Ecosystem Management and Sustainability

Ecosystem Management and Sustainability analyzes myriad human-initiated processes and tools developed to foster sustainable natural resource use, preservation, and restoration. It also examines how humans interact with plant, marine, and animal life in both natural and human-altered environments. Experts explain the complex ecosystem relationships that result from invasive species, roads, fencing, and even our homes—by addressing topics such as fire and groundwater management, disturbance, and ecosystem resilience. Because most people in the twenty-first century live in urban environments, the volume pays special attention to the ecology of cities, with detailed coverage on topics ranging from urban agriculture to landscape architecture. The volume focuses on how ecosystems across the world can be restored, maintained, and used productively and sustainably.
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Berkshire Encyclopedia of Sustainability

- Volume 1: The Spirit of Sustainability
- Volume 2: The Business of Sustainability
- Volume 3: The Law and Politics of Sustainability
- Volume 4: Natural Resources and Sustainability
- Volume 5: Ecosystem Management and Sustainability
- Volume 6: Measurements, Indicators, and Research Methods for Sustainability
- Volume 7: China, India, and East and Southeast Asia: Assessing Sustainability
- Volume 8: The Americas and Oceania: Assessing Sustainability
- Volume 9: Afro-Eurasia: Assessing Sustainability
- Volume 10: The Future of Sustainability
Ecosystem management for sustainability in the broadest, ecological sense is a concept that expresses a simple outcome—the complete preservation of nonrenewable, natural resources from one generation to the next. Fundamental to this quest for sustainable ecosystems, as we know them today, is the preservation of biological diversity. Ecologically, the preservation of natural ecosystems and their component species is based on theoretical and empirical studies evaluating ecosystem stability, including resistance to change, and recovery and resilience, both of which are important components of sustainability. As many of the contributions to this volume demonstrate, the loss (or removal) of one seemingly inconsequential species has led to devastating ecosystem impacts and the collapse of whole populations and communities of organisms in many different habitats found worldwide. In the twenty-first century, scientists have now identified and measured overwhelming evidence for anthropogenic (human-caused) perturbations of our Earth’s ecosystem, the biosphere. Carbon dioxide and other greenhouse gases have led to an increase in global temperatures with a rate of change unmatched in geological timeframes. Moreover, there are frightening scenarios for future large-scale changes in the Earth’s landscape that could have devastating impacts on humankind. Sea level rise and increases in extreme episodic events are two impacts predicted for the future (IPCC 2007). Yet little is currently known about the specific changes in ecosystem properties and services that will unfold in this portentous future.

Maintaining Species Diversity
Understanding how to maintain species diversity—a component of which involves the capability for species to recover following both natural and anthropogenic disturbance of minor to cataclysmic portions—is the foundation of ecosystem sustainability. (These ecosystem properties are active areas of research today, and many of them are presented as separate topics in this volume.) But even this simple definition of sustainability requires clarification because virtually every natural resource, including biological species, could be considered renewable, or replaceable, over a long enough timeframe. According to the fossil record, ecosystems today have survived major, mass extinctions through the evolution of new species: in fact, well over 90 percent of all species formerly on this Earth are now extinct. Seemingly irreversible extinctions of a biological species may become reversible in some sense if one considers today’s advances in technology for cloning and amplifying genes. Moreover, those within some circles of societies around the world argue that problems related to the sustainability of our biosphere will be taken care of, ultimately, by the actions of a supreme deity, or by as-yet-unknown technological advances. Following that premise, environmental problems such as global change or high species extinction rates might seem less pressing to individuals, especially when overwhelming economic problems require their immediate attention.

If an individual, community, or organization is not extinquishing nonrenewable resources (including biological species) from the biosphere, or if our management practices are not building toward a future of unforeseen, negative events, a sustainable status (again, by definition), has been achieved. Yet sustainability in other arenas of society can be interpreted quite differently. For example, sustainable economics or commercial development can take on a new dimension when considered in the context of sustaining an acceptable standard of living for a particular society. Consuming only renewable resources, and only at a rate that will not diminish the current pool of a renewable resource, is one obvious solution to this problem. But because many resources used in common practice
by humankind are far from being sustainable, this presents a monumental challenge for most societies across the globe, especially for those economies and standards of living that are deeply dependent on resources harvested excessively from the environment. Unfortunately, societies today with the greatest capabilities for accessing natural resources are also the ones with the highest standard of living and greatest usage. How will sustainability components such as biodiversity be accomplished on a global scale, including all societies regardless of their standard of living? This fundamental question facing the global community today can only be answered as an orchestrated effort directing a multidisciplinary, integrated approach that will span virtually all fields of study, including the “hard” sciences, the humanities, business, and law. Within this network, research methods and accompanying, specific measurements need to be identified, and then standardized, as the most accurate indicators of the quantitative degree of sustainability. Examples of this could include hidden costs of pollution, depreciation, ultimate depletion of natural resources, overproduction, negative alterations in esthetic value, or effects on health care expenses, just to name a few. Any quantitative index employed to quantify sustainability success will thus have to include a wide array of variables that are intertwined in a complex network of feedback and feed-forward interactions (see Volume 6, Measurements, Indicators, and Research Methods for Sustainability). In fact, these interactions may be the most difficult to understand. All of these multidisciplinary fields of study must be called upon to provide for the future, successful management of ecosystems necessary to prevent potentially serious consequences for our own species.

Effectively, with the current existence of global change issues such as elevated carbon dioxide in the atmosphere and warming temperatures, there are no habitats on Earth where anthropogenic disturbance is absent; that is, purely natural ecosystems (i.e., pristine areas, using the word in its purest sense) no longer exist. Many so-called official wilderness areas are in essence legislative constructs, built or restored habitats in which government mandates must approve attempts to restrict species or restate species. We now have to understand and manage already impacted ecosystems that were being managed previously from a simpler preservation approach. That is not to say that protection of critical areas will not be important for addressing the serious anthropogenic impacts already underway. For example, new management strategies for sustainability should now involve ideas for lowering global atmospheric carbon dioxide content and greenhouse gases that generate global warming. Encouraging plantation development for biofuels or as sinks for carbon dioxide absorption from the atmosphere are examples of ecosystem management techniques that are perceived to benefit the global community. But is this temptation to manage ecosystems to rectify anthropogenic impacts a sound strategy considering the complexities and difficult challenges of ecosystem management? Instead, is it not wiser to understand more comprehensively the potential harm of these impacts and act to eliminate pollution-point sources altogether? The same can be asked about the excessive harvesting that has led to serious declines and extinctions of species, plus community/ecosystem collapse. Just like common promulgations from the health sciences field, prevention is much less taxing and expensive compared to recovery and restoration. Species reintroduction, especially of those species high in the food web, is an active area of research and a current example of efforts at ecosystem sustainability. This approach is still very much in the experimental phase and, unfortunately, has evolved in response to major disturbances caused primarily by human alteration of the landscape and resulting pollution. Ecosystem Management and Sustainability contains articles covering all of these topics in greater detail.

Predictions and Forecasts

In the ecological sciences, in general, the capability for prediction (forecasting) has been a goal for over a century. Yet progress has been slow at best. For ecosystem management to become a legitimate “hard” science, predictability and forecasting future changes in ecosystems is a prerequisite. Science areas such as chemistry, physics, and math have played a vital role in predicting all sorts of events that are driven by physical/chemical forces. These predictions involve very rapid reactions that occur over a fraction of a second to much more lengthy astronomical time scales that predict the motion of planets and solar systems over millennia and well beyond.

The impact of perturbations due to anthropogenic forcings such as climate change have elicited a sense of urgency among ecosystem managers at all spatial scales, and the field actively pursues this capability to predict and forecast. Legislators want to know how much and
how fast ecosystems will be influenced, either negatively or positively, as the result of global change. Ecological forecasting is problematic, however, because so many variables are involved at differing degrees of influence, and myriad feedforward and feedback interactions could as well play prominent roles. Even most physical scientists admit that biological systems from the cell level to the landscape incorporate an almost overwhelming matrix of variables. This complexity impedes the capability for predicting the future and, it follows, efficacious management strategies for sustainability. Today’s technological advances, however, especially computer capabilities for data storage and rapid processing, should make the most complex systems understandable and predictable in the future. Acquiring this capability is the ultimate goal for ecosystem managers.

The encyclopedic approach taken with this volume and the other nine volumes of the Berkshire Encyclopedia of Sustainability provides a venue for communication between experts in the field and nonexperts, which in turn allows the latter to better appreciate the serious challenges ahead for humankind, especially if plans for the future lack a sustainability objective. After all, if only experts are convinced of the value of sustainable ecosystems, an electorate will rarely vote into office politicians who are knowledgeable about and supportive of ecosystem management principles. In this volume, numerous experts in their respective fields examine basic principles of ecology and ecosystem management. Aside from these fundamental ideas, their articles cover important topics that address specific issues of ecosystem management related to pollution impacts, agriculture, hunting and fishing, forestry, water, indigenous people, the esthetic value of natural resources, shale gas extraction, tree planting, and rain gardens. Because climate change is projected to bring greater episodic extremes, such as floods, minimum and maximum temperatures, droughts, and so forth, one article is dedicated entirely to this topic.

Politics and Ecosystem Management

The volume also addresses the growing field of environmental law because much of our future may involve litigation and court challenges to federal and state attempts at regulating anthropogenic pollution and the protection of endangered species. (In his bestselling book Storms of My Grandchildren, James E. Hansen [2009] offers an interesting look at a case history describing the interaction of science and politics, in particular dealing with the problem of greenhouse gases in the atmosphere and global warming.) Many experts in ecosystem management believe that the political arena is a crucial target for the ultimate success of ecosystem management in the United States and elsewhere around the world, and that economic interests instead of the science of sustainability often drive political decisions about ecosystem management. For example, politicians in the United States are elected and re-elected based upon their membership in and financial support from two major parties, both of which raise huge sums to back their candidates, especially those at the highest levels of government. In many cases the major contributors to these parties are (or represent) industrial corporations that are responsible for the pollution driving global change. This realization and the social paradox generated are related to the critical regulatory role of governments discussed in several of the articles in this volume. For instance, in “Administrative Law,” Bruce Pardy of Queen’s University, Canada, explains how in Western legal systems with a constitutional separation of powers, government officials carrying out and enforcing ecosystem management directives must operate within the bounds of a statutory mandate (i.e., authorizing legislation). This process ensures the protection of individual rights and will reassure ordinary citizens who see ecosystem management as coercive, just another case of government telling them what they can and cannot do, that checks and balances are in place.

Politicians are the ones who must legislate the rules driving strategic ecosystem management, a crucial ingredient for installing sustainability ideas into a society that can perpetuate the process only by electing the regulatory officials. But, as many articles in this volume attest, individual action, as well as interaction between public and private sectors, are powerful catalysts to bring about change.

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Suggested Reading

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Adaptive Resource Management

Adaptive resource management is a management strategy appropriate for use when there is uncertainty in how the natural resource system would respond to a management action. The goal of adaptive resource management is to iteratively improve management actions by carefully developing, monitoring, and assessing each management action. Adaptive resource management is especially important for maintaining sustainable resources as environmental conditions alter because of climate change or other threats.

The term adaptive resource management describes a structured method for managing natural resources in systems or situations with uncertainty in how the system/situation would respond to a particular management action. In theory, adaptive resource management is an iterative process in which managers learn more about the resource and enhance the effectiveness of their management strategy by designing management actions as experiments. Adaptive management, which is often learning by doing, can enhance understanding about the functioning of the managed resource and thereby improve the effectiveness of future management actions.

Adaptive resource management can be an important tool in achieving sustainable ecosystems. In the face of climate change, habitat modification, and other emerging threats, an adaptive resource management strategy allows management to alter their actions as environmental conditions change.

The Process

Adaptive resource management should not simply be management by trial and error; instead, it should involve an experimental design so that the effects of management actions can be understood and improved through a methodical, cyclic process. (See figure 1 on the next page.) The process breaks down into three phases: planning, implementation, and evaluation.

Planning

The planning phase starts by clearly defining a goal. Managers establish specific, measureable, and attainable objectives to help them reach their goal. Once they identify objectives, the planning phase continues as they develop management alternatives hypothesized to meet these objectives. These management alternatives are individual, testable hypotheses. In the formal adaptive management process, managers develop alternatives that involve modeling so that they can understand the potential effects of each management action on the resource and identify an optimal strategy (e.g., most cost effective, most effective at achieving the goal). For each alternative, managers must explicitly predict the effects.

Implementation

To apply an adaptive management strategy, managers implement alternatives in an experimental framework. A carefully developed monitoring program to assess the impact of each alternative management action on the resource is central to this phase.

Evaluation

Managers synthesize and assess results of the monitoring program in the evaluation phase. Based on the information obtained, managers can return to the planning phase and identify new goals, objectives, and alternate management strategies or make other adjustments. As managers
modify their methods, they repeat the adaptive management cycle. This process provides new knowledge about the functioning of the resource and improves future management actions.

Alternative Definitions

Despite the formally accepted definition and process of adaptive resource management described here, managers regularly use the term to describe a wide variety of resource management approaches. They often use the term in the broadest sense to define any resource management program in which actions change over time. These management programs often lack one or more of the key components required for a true adaptive resource management strategy. The loose use of this term to describe many different management programs has confused the meaning of what constitutes a true adaptive resource management program. Many resource managers say they are adaptively managing their resources, when in reality they are not.

Implementation of Adaptive Resource Management

Since the late 1970s, when policy makers first formally proposed the idea of adaptive resource management, it has garnered worldwide attention and support. Managers have attempted adaptive resource management in coastal, terrestrial, and marine systems. They have used it as a strategy to manage ecosystems, the harvest of species, water resources, and forests across a spectrum of scales and in geographically diverse places.

Figure 1. Adaptive Resource Management Diagram

Source: author.

This diagram identifies the major components of the adaptive resource management process and illustrates their conceptual sequence.
Adaptive resource management has often proved complicated and difficult to implement in the real world. Managers have successfully carried out few large-scale projects. Difficulties with implementation include flaws in experimental design or data analysis, high costs of monitoring, time lags between management actions and their impacts, difficulties in modeling outcomes, institutional barriers caused by stakeholders’ conflicting management philosophies, or the difficulty of adapting policy in large bureaucratic organizations with many levels of decision making.

Glen Canyon Dam

In the mid-twentieth century, the US government built Glen Canyon Dam and other dams along the course of the Colorado River in the southwestern United States, significantly altering the river’s hydrology. They built these dams to control flooding, store water, and generate electricity. Before the US government built the dams, massive seasonal floods shifted large amounts of sediment, creating sandbars and protected backwaters and altering the temperature of the water. The altered post-dam hydrologic regime lost or endangered many native species.

In the 1990s the US government (Department of the Interior) created the Glen Canyon Adaptive Management Program and Working Group with a goal of restoring the Grand Canyon’s riverine ecosystem and protecting native fishes. Managers of the Grand Canyon and other interested parties were uncertain about the best methods to restore ecosystem function. The adaptive management strategy focused on restoring sandbars and backwaters through massive water releases that mimicked pre-dam floods. The management alternatives included the timing, duration, and volume of water released. As of 2011, three iterations (1996, 2004, and 2008) of this management strategy had been completed. After each release, managers monitored the program to determine whether the alternative they selected created sandbars, created nearshore habitat for the endangered fishes, and benefited other resources. Managers have evaluated the results and adapted each subsequent release to improve the outcomes. They have also used the results to advance development of sediment models and aquatic-ecosystem models for the river.

The Future

Resource managers are often required to make decisions based on uncertain or incomplete data. Adaptive resource management provides a framework managers can use to make more informed decisions as they collect additional data to reduce uncertainty. Adaptive resource management will continue to play an important role in the management of natural resources in the twenty-first century as climate change and human impacts alter environmental conditions. Key to its continued use will be whether policy makers and managers develop novel approaches and tools to support components of adaptive management that currently impede its full implementation.

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See also Administrative Law; Best Management Practices (BMP); Complexity Theory; Disturbance; Ecological Forecasting; Extreme Episodic Events; Plant-Animal Interactions; Safe Minimum Standard (SMS)

Further Reading

Administrative law is a branch of law that governs the actions of the executive branch of government, including administrative agencies and government officials. The principles of administrative law apply to ecosystem management conducted by state agencies, including the actions and decisions of officials, scientists, policy advisors, and others involved in its practice.

From the perspective of ordinary citizens, ecosystem management is a prescriptive phenomenon. It consists of government telling them what to do. As such, it is a coercive process, dependent not upon agreement or consent of individuals or communities, but upon the authority of the agency giving orders. A basic principle of administrative law, as it exists in Western legal systems, is that the executive branch of government is empowered to do only what statutes grant it the mandate to do. The corollary of this principle is that any executive official who takes action without a statutory mandate acts without jurisdiction. While science and politics play a large role in ecosystem management, the law defines what the practitioners of ecosystem management can do. Therefore, officials who carry out and enforce ecosystem management directives must find their mandate in authorizing legislation.

This administrative law principle is a derivative of the constitutional separation of powers between the three branches of government: legislature, executive, and judiciary. Separating powers between these branches is a basic feature of the rule of law in Western legal systems. It protects citizens by giving each branch a role in controlling the actions of the others, thereby preventing concentration of power and diminishing the potential for arbitrary measures. Traditionally, the legislature passes general rules and authorizes executive action to put them into effect; the executive carries out these directives, in accordance with the powers and mandate provided in the statute; and the judiciary applies the general rules to particular cases, including the power to review executive actions. Separation of powers is observed to different degrees in different countries. For example, in the United States, there is a generally strict division between Congress (the legislative branch), the office of the president (the executive branch), and the judiciary, while in parliamentary democracies like the United Kingdom and Canada, the separation of powers between legislative and executive branches takes a different form, although under both systems, as a general rule, executive power requires legislative authorization.

Courts are authorized to scrutinize executive action when aggrieved parties apply for judicial review. In judicial review, the court assesses whether a state agency has acted within its jurisdiction, as established by statute; whether it has exceeded its discretion, the bounds of which also depend to a significant degree upon the wording of the statute; the degree of deference that the agency should be afforded; whether its actions are consistent with applicable procedural standards, such as providing notice and a right to respond to those who will be affected by certain kinds of decisions; whether there was bias, conflict of interest, or predetermination in the decisions reached; and a number of other considerations the court may consider to determine whether the actions of the agency were lawful.

In principle, through the application of administrative law, courts play an important supervisory role over the practice of ecosystem management. But modern environmental statutes commonly provide broad discretion to environmental agencies. The language in such statutes tends to be permissive (“the Agency may . . .”) rather than mandatory (“the Agency shall . . .”), and gives officials
authority not simply to execute general rules but also to establish objectives and the means of achieving them in regulations or policy documents. Since the role of courts in judicial review is to compare executive actions with statutory mandates, the grant of broad discretion provides more room for ad hoc and therefore potentially arbitrary decision making, thus diminishing the theoretical protections provided by the principles of administrative law.

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See also Adaptive Resource Management (ARM); Best Management Practices (BMP); Comanagement; Safe Minimum Standards (SMS)

**FURTHER READING**


Intensified agricultural production involves the use of land for food or livestock production in a way that increases output. The rapid increase in human population has made it imperative to meet the demands for food and other natural products. Injudicious and uninformed use of agrochemicals and technology often leads to land, air, and water pollution and degradation. Environmentally sound farming practices enhance production while sustainably supporting future generations.

Agricultural intensification or intensified farming is the practice of using farm- and pastureland in a way to increase production by means of increasing the frequency of cropping, increasing the use of inputs (tillage, labor, fertilizer, pesticides, etc.), or grazing higher numbers of livestock per unit land area (Boserup 1965; WRI 2011). Intensification of agriculture is an unavoidable consequence of the growing global human population, which demands ever more food, energy, and other resources. Whereas intensive use of land-based resources is not, in and of itself, a bad thing, lack of careful planning and management, vigilant regulation, and judicious use of inputs (especially agrochemicals) could lead to a decline in productive capacity and a degradation of the land along with pollution of air, rivers, lakes, and groundwater (Matson et al.1997; WRI 2011). Because of the potentially negative consequences of agricultural intensification and its adoption, scholars, researchers, and development workers have shown considerable interest in this topic.

Historical Development

In the course of the evolution of agriculture, a number of factors have influenced the nature of practices and tools adopted by farmers. The technological know-how available to them has evolved from simple, handheld tools to large-scale mechanical implements. The development and availability of inputs like irrigation water, fertilizers, pesticides, and hybrid seeds have enabled production beyond natural climatic and soil limits. But an overriding factor causing farmers to intensify production is population pressure (Boserup 1965). Prior to the industrial and technological revolution of the nineteenth and early-twentieth centuries, farming communities worldwide practiced “low-tech” agriculture using simple tools made from locally available materials.

As the industrial era progressed into the twentieth century, advances in technologies, including the tractor and farm implements for tillage, sowing, harvesting, and processing crops, along with fertilizer, improved seed, and pest control technologies, led to increased farming intensity. Population growth as well as changes in demography and dietary patterns fueled this transformation (WRI 2011).

An increasing proportion of the human population gradually concentrated in urban areas, and the proportion of people engaged in agriculture diminished. Increasing mobility, the changing nature of employment, and a transformation in lifestyles all led to changes in dietary preferences and demand for agricultural products. A shrinking farm-labor force and simultaneous increase in world food demand during the mid-1900s meant that fewer people had to produce greater quantities of food from smaller areas of land. Mechanized farming and monoculture production systems thus began in Europe and North America.

During the late twentieth and early twenty-first centuries, it has become evident that agricultural intensification will undoubtedly continue well into the future. Intensified farming has now spread to the emerging economies of Asia and South America, as well as to the
less-developed countries of the world where population pressures are tremendous (Ali 2007; Kates 1994). In these developing nations, which are already densely populated, habitation, infrastructure, manufacturing, and agriculture compete with farming for land use. Agricultural production has to come from the existing farmland. This lack of room to expand will require increasing use-intensification of the land already under cultivation.

Environmental Impacts

Although intensified use of land resources need not necessarily lead to adverse environmental consequences and degradation, the risks of ecosystem imbalance and disruption of natural systems are high. Inadequate precautionary measures and improper management practices on intensively cultivated lands invite adverse impacts on the quality and productive capacity of the land, biodiversity and gene-pool resources, surface and ground water quality, and atmospheric composition. (See figure 1.)

Soil Properties

Alteration of soil properties is the most common and readily observable environmental effect intensified farming practices cause. Intensified agriculture increases manipulation of the soil (number of tillage operations and depth of tillage) and produces a higher number of crops per unit land area per cropping cycle (usually one year). This generally leads to a reduction in the soil organic matter content unless high rates of farmyard manure, plant litter, or compost compensate and substantial amounts of crop residues are retained. Soil organic matter is a key component of soils. It maintains soil structure, stability, and water-holding capacity, facilitates

Figure 1. Consequences of Sustainable and Unsustainable Approaches to Agricultural Intensification

Source: author.
A reduction in the agro-biodiversity will result in a reduction in the abundance of their natural enemies (Matson et al. 1997). Outbreaks of unwanted organisms (i.e., pests) and weeds, insects, and microbial communities will be common. A shift away from excessive application of chemical fertilizers and synthetic pesticides will be essential to avoid its adverse consequences. This loss has implications for the genetic varieties of crops and their ability to adapt to environmental changes, which normally operates through the natural selection process.

Pollution and Health Effects

Agricultural intensification affects the environment through pollution from the indiscriminate and excessive use of chemical fertilizers and pesticides. Because water and crop nutrients are commonly the main constraints to production, intensive farming systems inevitably require substantial inputs of fertilizers and supplemental irrigation. A significant portion of the applied inorganic fertilizers leach into the groundwater system, volatilize, or travel long distances by air (Matson et al. 1997). Fertilizers removed from the farm field contaminate ground- and surface waters, posing a threat to human or animal health.

Toxic synthetic chemicals (e.g., organo-chlorine, organo-phosphate, and carbamate compounds) applied to control crop pests such as weeds, insects, and diseases have serious implications for human health as well as ecosystem functioning. Many of these toxic compounds, some of which persist for decades in the environment, do not reach the intended target organisms but rather lead to the contamination of soil, water, and food products. The persistent compounds, like chlorinated hydrocarbons (DDT, for example) accumulate in the tissue of organisms like fish and birds. When other organisms higher up in the food chain, including humans, consume them, these compounds can cause health problems such as cancer and birth defects.

Soil Biota and Agricultural Biodiversity

Monoculture—planting a single crop type over large areas—and continuous cropping of one type of crop across multiple seasons reduce the diversity of plants in and around the farm fields. This reduction influences the diversity and abundance of other biota, such as soil fauna, weeds, insects, and microbial communities (Matson et al. 1997). Outbreaks of unwanted organisms (i.e., pests) and a reduction in the abundance of their natural enemies can result. A reduction in the agro-biodiversity will ultimately lead to the loss of crop gene-pool diversity. This loss has implications for the genetic varieties of crops and their ability to adapt to environmental changes, which normally operates through the natural selection process.

The Future

Production systems will likely intensify to meet the ever-growing demands for food, fiber, and basic natural raw materials. This trend is likely to persist at least through the twenty-first century, by the end of which the world population probably will have stabilized and begun to decline.

This change in human population is expected to be a combined consequence of depletion of available space and natural resources as our planet approaches and exceeds its carrying capacity (Arrow et al. 1995; Cohen 1995); as nation-states take population growth control measures; and as climate change brings increasingly frequent droughts, famines, floods, hurricanes, and severe storms (Forster 2007). Until such a time when population pressure and increasing demand for the basic necessities diminish, it is unlikely that intensification of production, whether it be food crops, medicinal plants, fiber, and construction materials, will decrease. Faced with this scenario, farming communities, scientists, and conservation workers must devise lasting and sustainable means of intensifying production systems while avoiding its adverse consequences.

Sustainable Agricultural Intensification

Our quest for sustaining intensified production with minimal adverse impacts on the environment, natural ecosystems, and human health will require that we learn from and work in harmony with natural systems. Sustaining intensive agriculture is likely to necessitate innovative and unconventional approaches to production. A shift away from excessive application of chemical fertilizers and synthetic pesticides with concomitant adoption of diversified cropping systems that optimize the use efficiency of water, light, and plant nutrients will be essential. Some examples of such systems include mixed, multiple, and relay cropping, agroforestry and other


permaculture systems, use of biofertilizers and biopesticides, hydroponics, and so on. The need for sustainable intensification of agriculture is even more acute in developing countries because of the continued increase in human population and lack of financial capacity of the farmers for capital investment in advanced technology. The careful selection and adoption of a combination of appropriate alternative practices, crops, and pest control measures will be instrumental for ensuring the sustainability of farming systems (Dahal, Sitaula, and Bajracharya 2008). Failing to achieve sustainable agricultural intensification will mean the further degradation of land, water, biodiversity, and other natural ecosystem services, which will in turn reduce the Earth’s capacity to support and nurture human society.

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See also: Agroecology; Biodiversity; Ecosystem Services; Fencing; Groundwater Management; Human Ecology; Irrigation; Population Dynamics; Soil Conservation; Urban Agriculture; Urban Vegetation

**FURTHER READING**

The environmental impacts of the Green Revolution, which emphasized increasing crop yields often at the expense of the environment, led ecologists to encourage agroecology, a sustainable approach that protects not only the environment but small shareholders. The movement has grown worldwide with positive effects particularly in developing countries, where it has not only ecological impacts but social and economic effects as well.

Agroecology is defined most simply as “the application of ecology in agriculture.” Whereas many of the principles of agroecology are as old as agriculture itself, interest in the topic has expanded in response to industrialized agriculture’s negative environmental and social impacts. The Green Revolution in agriculture that occurred from the 1940s to the 1970s promoted the adoption of new technologies for boosting crop yields through monoculture systems, improved crop varieties, chemical fertilizers, synthetic pesticides, and irrigation. Although this strategy increased production in an effort to feed a growing population, a number of unintended consequences emerged. New high-yielding varieties displaced traditional varieties that were well adapted to local conditions and served as a source for genetic diversity. The focus on specific grain crops such as corn, wheat, and rice reduced the nutritional quality of the human diet because these replaced fruits, vegetables, and traditional crops. Environmental impacts are the most publicized consequences: biodiversity and habitat loss in the landscape; pollution of water resources from pesticides, nutrients, and sediment; reduction in water quantity from irrigation; and others. The growing environmental movement of the 1960s raised awareness of these issues. Agroecology emerged as an approach that would support production of food and other materials while also protecting the environment and small shareholders.

Charles Francis, a US expert on sustainable agriculture, and his colleagues expanded the area of agroecology as the “integrative study of the ecology of the entire food system, encompassing ecological, economic and social dimensions” (Francis et al. 2003, 100). This broader focus on food systems establishes agroecology as an integrated and multidisciplinary field that encourages involvement from sociologists, economists, regional planners, policy makers, and public health experts. Interest in agroecology is likely to grow because society needs creative and sustainable solutions to balance the need for greater food security with the limited availability of natural resources to produce food.

Historical Context

The emergence of agroecology as a unique discipline dates to the early part of the twentieth century. Scientists explored early topics such as crop ecology (Klages 1928), crop-environment interactions (Papadakis 1970), and ecology in agriculture (Hanson 1939). Agroecology gained a strong foothold in the 1970s, as scientists publicized the impacts of industrialized agriculture. Miguel Altieri, a US agroecologist and a sustainable agriculture advocate, published Agroecology: The Science of Sustainable Agriculture in 1987. Steven Gliessman, another US expert in agroecology, published his textbook Agroecology: Ecological Processes in Sustainable Agriculture in 1998. Alexander Wezel (2009), a French agroecologist, and his colleagues chronicled the development of agroecology primarily as a scientific discipline in the 1970s and earlier, as a set of practices in the 1980s, and as a social
movement in the 1990s. Broader definitions that encourage the integration of multiple disciplines have emerged in the twenty-first century.

A number of topics related to agroecology are worth defining because they have contributed to the development of the field and will likely impact directions. Sustainable agriculture seeks to meet the needs of humans today without compromising the needs of future generations by integrating goals for environmental stewardship, economic viability, and social equity. Many of the practices recommended for sustainable agriculture align with those in agroecology, but the concept of sustainable agriculture is used less in the context of traditional agricultural systems, practices, and knowledge. Organic agriculture refers to a very specific set of farming standards that limit the use of pesticides, chemical fertilizers, and other synthetic inputs. The Agricultural Marketing Service of the US Department of Agriculture and international bodies regulate the marketing term. Products labeled “organic” often receive a premium price. Multifunctionality in agriculture developed out of the recognition that agricultural landscapes can provide multiple noncommodity outputs including ecological functions (e.g., biodiversity, water protection, and habitat) and cultural functions (e.g., recreation, visual quality, and education) that traditional markets do not capture. Policies to encourage multifunctionality often promote landscape features and practices that could support agroecology. Other holistic approaches such as permaculture and biodynamic agriculture overlap with agroecology, drawing in a diverse audience to include residential gardeners, collective farms, and concerned consumers. These approaches share knowledge on the ecology of agriculture in simple and small-scale formats.

Contemporary Approaches, Impacts, and Challenges

Contemporary agroecology includes a broad set of approaches and methods, most of which seek to mimic the characteristics of natural ecosystems. In general, the methods intend to reduce reliance on off-farm resources, avoid synthetic inputs, minimize toxic materials, conserve energy, and protect natural resources such as soil and water. Reduced tillage minimizes the regular disturbance of the system, resulting in lower energy requirements, reduced soil erosion, and conservation of soil moisture. Diversification of the agroecosystem through crop rotations and polycultures helps to reduce pests by disrupting their life cycles, to improve soil fertility when nitrogen-fixing legumes are included, and generally to increase resilience against local disturbances. Cover crops can also contribute to biodiversity and nutrient cycling while suppressing weeds. Crop species that are well adapted to the given environment can reduce the need for irrigation and other inputs. Perennial crops, including trees, minimize disturbance of the system while providing additional benefits such as carbon sequestration, soil stabilization, and microclimate control. Integrating livestock into the agroecosystem improves soil organic matter and enhances nutrient cycling because animals process plant material into readily available nutrient forms in their manure. From the social perspective, agroecology seeks to support the livelihoods of farmers, to protect the welfare of farm workers, and to strengthen the rural communities in which they live.

The impacts of agroecology on the environment are intended to be positive when compared with conventional systems. Critics of conventional industrial agriculture say it displaces natural habitats, depletes the soil of nutrients and organic matter, pollutes and depletes water resources, and contributes to greenhouse gas emissions (agriculture and food systems globally contribute one-third of emissions). Agroecology is one strategy for protecting natural resources through appropriate design and management of a production system.

The approach is not without challenges, however. The knowledge required to manage such a complex system is vast. Current institutional support to help build the knowledge base through research and extension activities is lacking. Farmers themselves, along with some key advocates, share much of the knowledge. Agroecology can also be more labor intensive, particularly in the startup phases. The availability of skilled laborers is limited in some regions. Finally, profitability can be a challenge in countries where government subsidies provide support for commodity crops and associated inputs, but not for
agroecology production. Farmers of diversified and small-scale production systems may find it difficult to compete under such conditions. They are often pushed into high-value markets with a more elite or wealthier customer base. The outputs (such as organic foods) are not equally available to all consumers, particularly the members of disadvantaged communities.

**International Distribution**

The roots of agroecology, as well as current applications, can be found across the globe. Much of the focus in the literature is on developing countries, but the growing interest in healthy food systems is driving a greater awareness in developed countries.

**United States and Canada**

Ecologists led the rise of agroecology in the 1970s in the United States, where the conflict with industrial agriculture was most intense and visible. Several key advocates have promoted agroecology for decades and continue to contribute to the field through writings, research, and curriculum development: Miguel Altieri at the University of California at Berkeley; Steven Gliessman at the University of California at Santa Cruz; Charles Francis at the University of Nebraska; and John Vandermeer at the University of Michigan. A number of land grant institutions provide some curricular offerings in agroecology or related topics through a dedicated major, a specialization within a major, or individual courses. Increasingly, other institutions throughout the United States and Canada, including liberal arts colleges and community colleges, offer programs in agroecology to meet the demand growing from greater public awareness of the role of agriculture in environmental and social issues.

Many of the publications on agroecology originate from US authors, yet developing countries (particularly those in Latin America) are the target for much of the research in the articles. One reason may be that experts rarely consider the applications of agroecology from the United States and Canada to be transformative. Few examples exist of whole farms planning to closely mimic the natural environment. Instead, US agriculture has focused on implementing individual, specific practices such as cover cropping, intercropping, and crop rotations, which are often promoted for commercial organic production systems. Some of the more aggressive applications include integrated crop-livestock systems, urban food production systems, and perennial polycultures for grain or biofuels. The growing public interest in local foods in the United States could have important implications for the future of agroecology as consumers request higher environmental standards from the nearby farmers providing their food.

**Latin America**

Latin America has played an important role in the development of the international agroecology scene in two ways. For one, the region includes a number of sites that have been the focus of research and the source of models of integrated systems relying on local knowledge. A second role is in the grassroots political movements driven by peasant farmers, such as the Campesino a Campesino (Farmer to Farmer) movement that Eric Holt-Giménez, an internationally known researcher in agroecology, documented. These movements call for agricultural reform through a return to sustainable agriculture practices, local knowledge protection, and food sovereignty for the poor.

In some ways, Latin America serves as a nexus of agroecology between the developing and developed world. These countries offer innovative solutions, host development projects, and lead reforms. In Havana, Cuba, the transition from reliance on imported foods and agricultural inputs to local, organic food production following the collapse of the Soviet Union may be one of the most inspiring examples of widespread transformation to agroecology production. The Tropical Agriculture Research and Higher Education Center, or CATIE (Centro Agronómico Tropical de Investigación y Enseñanza), in Turrialba, Costa Rica, is recognized globally for programs and expertise in agroecology, particularly for its emphasis on social issues such as poverty alleviation and rural development. A number of current agroecology experts have trained at the center.

**Europe**

In many European countries, agroecology is being applied at a broad scale in a comprehensive and integrated manner responsive to local and regional landscape characteristics. Policies to subsidize landscape features that improve ecological and cultural functions while still encouraging productivity have embraced the concept of multifunctionality of agriculture. Many of the programs emphasize landscape design (i.e., including noncrop habitats), as opposed to field-scale management practices. Several European researchers are contributing to the broader conversation about agroecology. Eduardo Sevilla-Guzman’s research group at Spain’s University of Cordoba focuses on the sociological perspective of agroecology, engaging small-scale farmers and supporting rural development through participatory approaches. Alexander Wezel at ISARA Lyon in France studies the history and applications of agroecology around the world. The Nordic Agroecology
University Network (AGROASIS) is a collaboration between several institutions to provide education in agroecology. Wageningen University in the Netherlands offers programs that bridge disciplines through applied research and education based on a theme of “healthy food and living environment.”

Asia

In Asia, the most interesting applications of agroecology occur in and around urban centers. High population densities require innovative approaches to growing food and reusing resources, in some cases integrating crops, livestock, and aquaculture. In urban areas, agroecology combines with landscape ecology, integrating and protecting production functions as part of the urban planning process. Peri-urban zones, which juxtapose urban and rural areas, contain agritourism enterprises that often emphasize sustainability. In rural areas, protecting the heritage of traditional practices that relied upon complex nutrient cycling and waste management is the primary focus, providing important lessons for agroecology today. Water and air quality are specific issues that could strengthen the importance of agroecology in Asia in the coming years, along with the threat that climate change poses to food security in densely populated areas.

Africa

For Africa, agroecology offers an opportunity to deal with human hunger in a more sustainable manner that could also support self-reliance and community empowerment. The lack of organized agricultural policy that could subsidize sustainable approaches has limited the widespread adoption of agroecology in Africa. Instead, food insecurity has led to exploitation of resources, causing a decline in soil fertility and crop yields. To date, most international efforts to assist African communities in dealing with the immediate threat of starvation have relied on Green Revolution technologies such as nonlocal varieties, irrigation, and synthetic pesticides and fertilizers. Recently, however, results of a countrywide study in Malawi provided evidence of the benefits of sustainable approaches, even in terms of food provision. Monitoring the impacts of a broad government program to provide farmers with improved maize seed and synthetic nitrogen fertilizer, the research team of US ecologist Sieglinde Snapp and her colleagues found that adding diversity to the system (in the form of rotations including legumes) increased yield consistency, grain quality, production profitability, fertilizer efficiency, and farmer preference compared with synthetically fertilized monoculture systems. This study challenges the notion that agroecology is inconsistent with the need for high production to feed the world (Snapp 2010).

Australia

Agroecology has existed in Australia for decades, mostly through alternative sustainable farming practices. The Australian ecologists Bill Mollison and David Holmgren formalized the permaculture approach (or “permanent agriculture”) in the 1970s as a way to promote ecological principles through productive perennial habitats that are diverse and resilient. Permaculture, which is often implemented as part of the homestead, can become a lifestyle for some advocates. Biodynamic farming, a related holistic and organic approach focusing on a closed system of nutrient cycling, is popular in Australia (see, e.g., Biodynamic Agriculture Australia, a nongovernmental organization that promotes the concepts). Australia was also an early leader in the organic agriculture movement, with the first organic society, the Australian Organic Farming and Gardening Society.

Controversy and Debates in Agroecology

The debates between proponents of industrial agriculture and advocates of agroecology have simmered throughout the decades. The primary point of contention has been the extent to which agroecology could actually meet the global food needs if implemented on a broader scale. Governments justified the Green Revolution based on the assumption that the growing population required high-yielding grain crops that could be stored for extensive periods. Critics have brought the sustainability of this strategy into question. With the growing awareness of the impacts of agriculture on climate change, water quality, and other environmental issues, the controversy has evolved to consider the best strategies for conserving biodiversity and protecting natural resources. The question then becomes whether it is better to intensify production in an effort to conserve land elsewhere (“land sparing”), or to reduce the negative impacts of agriculture locally (“wildlife-friendly farming”).

The controversy between industrial agriculture and agroecology extends beyond environmental issues to include societal issues. One debate relates to the importance of protecting local knowledge, conserving local genetic resources, and supporting farmer livelihoods in developing countries. Critics have accused agribusinesses of exploiting the diverse crop genetic resources to improve the performance of marketable crop varieties. Governments have encouraged indigenous farmers to purchase the
“improved” varieties. Farmers often abandon local and adapted crops, along with the local knowledge to manage such systems. Agroecologists, on the other hand, view the genetic resources and local knowledge as an essential part of the agroecosystem and a contributor to livelihoods of small farmers.

Another debate revolves around the issue of human health related to food consumption. Historically, proponents of traditional agricultural systems compared systems based on metrics of quantity (producing calories, or high grain yields) and not quality (nutritional value). Agroecology advocates argue that much of the grain that conventional systems produce is destined for livestock feed or highly processed foods (particularly in the United States with the focus on corn), so the efficiency is low in terms of human consumption and food quality. The recent concerns about the quality of the human diet have fueled this argument, as health experts call for diets with more fruits and vegetables to reduce obesity and related illnesses.

The controversies related to production, environmental impacts, and social implications become particularly heated when government subsidies and incentives are considered. In the United States, for example, agroecologists and other critics question policies originally designed to avoid food shortages and support farmers, for their unbalanced support of large, commodity-based production systems. Agroecology advocates argue governments should remove or redistribute subsidies and incentives to align with the public benefits provided by agriculture. Furthermore, a large portion of the agricultural subsidies go to wealthy individuals and corporations that are not based in rural communities where the land is located. Predicting the broader implications of an overhaul of these policies, however, is a great challenge.

Outlook

Governments are exploring agroecology as a realistic solution for balancing food production and environmental health, with increasing consideration of the noncommodity outputs provided by agroecosystems. The expansion of the definition of agroecology to encompass the entire food system will encourage multidisciplinary approaches that consider social and political issues. An important contemporary concern is the need to develop appropriate assessment and monitoring strategies in order to evaluate the impacts, beyond yield alone, of different types of farming systems. These assessments will consider “ecosystem services” provided by agroecosystems, as well as impacts on food security, human health, and farmer livelihoods. The results of these assessments could help guide agricultural policy.

The outlook for the discipline of agroecology is bright. United Nations Special Rapporteur Olivier De Schutter’s report to the United Nations on the “right to food” specifically identified agroecology as an appropriate strategy to improve food availability for vulnerable groups (De Schutter 2010). And since vulnerable groups have been identified in cities throughout the world, even in developed countries, urban agriculture is a particularly interesting application of agroecology to address food insecurity and obesity while promoting neighborhood revitalization.

With dense populations, the urban environment offers unique opportunities to provide healthy foods, reuse organic waste products, reduce transportation and processing, educate consumers about food and nutrition, and create jobs.

The future of agroecology will most certainly be guided by sustainability, as the environmental implications of agricultural activities must be balanced with the need to feed the world. Research related to climate change—both mitigation and adaption—will be critical in the coming years. How can agroecosystems be designed to sequester carbon? What cropping systems will be adapted to future conditions? To what extent will agriculture compete for fresh water resources? Answers to these questions will certainly require collaboration across disciplines. Although agroecology initially gained strength directly from the negative outcomes of industrialized agriculture, this discipline has the potential to dissolve some of the tensions between agriculturalists and environmentalists as we face an uncertain future dealing with food insecurity, climate change, and limited resources.

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See also Adaptive Resource Management (ARM); Agricultural Intensification; Best Management Practices (BMP); Biodiversity; Ecosystem Services; Global
Climate Change; Human Ecology; Irrigation; Nutrient and Biogeochemical Cycling; Permaculture; Soil Conservation; Urban Agriculture

**Further Reading**


**Best Management Practices (BMP)**

**Best management practices (BMP) are practical measures implemented to reduce the impacts of human activities on water resources. BMPs identify the best available pollution control technologies for nonpoint (diffuse) source pollution, taking into account practical, societal, and economic considerations. Use of appropriate BMPs can significantly improve runoff water quality from land uses like agriculture, forestry, and urbanization.**

Nonpoint source (NPS) pollution (diffuse sources resulting from storm runoff or snowmelt) often is the largest contributor to water quality degradation. Particularly in developed countries, once government regulates large industries, municipalities, and other direct dischargers of pollution, the remaining sources of water pollution are often runoff from land uses like urbanization, agriculture, and forestry. These nonpoint sources can be difficult to regulate under discharge-permitting systems because of the sporadic nature of storm events, the diffuse nature of pollution sources, the difficulty separating background sources from anthropogenic sources, and the diverse human activities in a watershed.

Sustainable water resources management requires reduction of all anthropogenic sources of pollution below thresholds that will harm aquatic ecosystems and human water uses. Water quality regulatory programs can significantly reduce water pollution by changing the management of lands within a watershed. Practical measures that nonpoint sources can implement economically to reduce NPS pollution below critical thresholds are called best management practices (BMP).

**Types**

Managers employ a wide range of BMPs based on land use and type of NPS pollution. BMPs can be grouped into the following general categories.

**Riparian Buffer Strips**

A riparian buffer is a zone of vegetation retained along the banks of a water body. The land immediately adjacent the water body is the most sensitive part of a watershed. This riparian area may extend from ten to several hundred meters in width. In this riparian buffer, canopy-forming vegetation (like trees) is retained, and soils remain covered with ground vegetation like grasses and forbs or by forest leaf litter.

Riparian buffer strips protect water bodies from adjacent and upstream land uses, enhance soil infiltration, filter overland flows, reduce bank erosion, and provide shade along the water course. Pollutants retained in the buffer can be transformed and bioremediated. Vegetation also traps eroded soil particles, reducing sediment losses (McBroom and Young 2009). Riparian buffers are one of the most important BMPs for significantly reducing NPS pollution (McBroom et al. 2008b). These areas can also produce other benefits, including recreation, silvipasture, production of perennial food crops like nuts, and selective timber harvesting. Riparian buffers also provide ecosystem services such as wildlife habitat and enhanced biodiversity.

**Structural BMPs on the Land**

People have built structural BMPs like terraces for many centuries. This type of BMP slows overland flows, increases water infiltration into soils, and reduces soil
erosion. Depending on soil erodibility (i.e., texture, cohesiveness, etc.), slope, climate, and land use, managers can design terraces for optimum water retention and minimal soil erosion. Native soil, organic matter (wood chips, straw, etc.), rocks, synthetic fabrics, or other materials may make up terraces. Managers can add mulches to bare soils or use silt fences and hay bales as effective temporary BMPs in road or building construction. Structural BMPs include filter beds, retention basins, detention ponds, rain gardens, and constructed wetlands. If these BMPs are located downslope from high intensity agriculture, urban areas, and construction, they can effectively reduce a variety of NPS pollutants (Clayton and Schueler 1996; Zhang, DeAngelis, and Zhuang 2011).

**Municipal and Residential BMPs**

Because urban stormwater runoff is a significant pollution source, managers have developed programs to minimize the effects of common daily activities on water resources. These BMPs include proper disposal of household hazardous wastes, landscaping and lawn care that minimize nutrient and pesticide runoff, pet waste management, and trash and debris management. Water conservation practices such as rainwater harvesting reduce both urban runoff and demands on municipal water supplies.

**Land Management Decisions**

A critical component of BMP implementation is appropriate land use. On steep slopes with highly erodible soils, for example, BMPs for row crop agriculture like terracing may be expensive and labor intensive; such areas may be more effectively managed as forestland or under some other permanent vegetative cover, as in the Yangzi (Chang) River basin in China (Zhang, DeAngelis, and Zhuang 2011). On compaction-prone soils, minimizing traffic results in greater long-term productivity and less runoff and erosion (McBroom et al. 2008a). Governments often inappropriately zone urban developments, resulting in floodplain incursion. Floods cause significant economic and social losses, in addition to chronic NPS pollution as urban areas encroach into riparian zones. The most economically viable and environmentally sustainable land use requires significant knowledge of the watershed characteristics and must have local community involvement.

**Effectiveness**

Researchers have tested specific BMPs, examining the costs of these practices and determining how to efficiently implement these strategies on the landscape. Forestry BMPs in particular can reduce NPS pollution by up to 99 percent (Ice 2004). These rates compare to pollution reductions that effluent treatment and point source control programs achieve. The challenge then becomes to get local communities and land users to implement BMPs.

**Implementation**

Educating landowners, managers, and regulatory agencies about BMP implementation and effectiveness is critical because it increases the perceived benefits of BMPs (Husak, Grado, and Bullard 2004). Governments and managers can then offer incentives and cost-share programs. They can reduce property taxes on riparian buffers, for example. In addition, environmental certification programs may require BMP implementation. Particular BMPs may become mandatory, and violations may be fined. Implementation rates for mature programs can reach over 90 percent. In the United States, states with voluntary forestry BMP programs generally have implementation rates comparable to those with regulatory programs (Ice, Schilling, and Vowell 2010). Inadequate funding and staffing of organizations and agencies responsible for education, incentive, and cost-share programs is a significant challenge to BMP implementation.

**The Future**

Managers and land users will continue to develop innovative BMPs, addressing local land uses and emerging water quality problems while protecting water resources and allowing economic growth. This protection requires greater coordination among land users. BMP programs are often divided among land-use categories subject to diverse regulatory jurisdictions. Because different land uses often occur within a watershed, managers need a coordinated, systematic approach. Road surfaces, ditches, and cut-and-fill slopes are significant NPS pollution sources, for example. Roads often fall under several jurisdictions and ownerships, making effective BMP implementation on a road network challenging.

All land uses that affect water quality must adopt BMPs. In the United States, for example, Texas exempts oil and gas development from construction BMPs. A gas well built in the middle of a stream resulted in over ten times more erosion than from a nearby clear-cut forest without any BMPs and almost twenty times more sediment than from a clear-cut forest using BMPs (Thomas and McBroom 2009). The area needs a more systematic, coordinated BMP program with significant local community support.
In *A Sand County Almanac*, the US scientist and environmentalist Aldo Leopold stated, “A thing is right when it tends to preserve the integrity, stability, and beauty of a biotic community. It is wrong when it tends otherwise” (Leopold 1949, 224). By this definition, BMPs are a sustainable means of controlling NPS pollution, preserving the integrity of aquatic ecosystems, and ensuring the availability of clean, contaminant-free water for future generations.

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See also Adaptive Resource Management (ARM); Administrative Law; Agroecology; Buffers; Comanagement; Ecological Forecasting; Ecological Restoration; Groundwater Management; Pollution, Nonpoint Source; Pollution, Point Source; Road Ecology; Shale Gas Extraction; Water Resource Management, Integrated (IWRM)

FURTHER READING


The Earth’s biodiversity comprises the entire range of living species, the genetic variation that occurs among individuals within a species, and, at a higher level, the biological communities in which species live. It also includes their ecosystem-level interactions with the physical and chemical environment. Natural and anthropogenic occurrences can alter these complex levels of biodiversity with cascading consequences.

The protection of biological diversity is central to conservation biology. Conservation biologists use the term *biological diversity*, or simply *biodiversity*, to mean the complete range of species and biological communities, as well as the genetic variation within species and all ecosystem processes. By this definition, biodiversity must be considered on three levels, all of which are necessary for the continued survival of life as we know it:

- **Species diversity.** All the species on Earth, including single-celled bacteria and protists as well as the species of the multicellular kingdoms (plants, fungi, and animals)
- **Genetic diversity.** The genetic variation within species, both among geographically separate populations and among individuals within single populations
- **Ecosystem diversity.** The different biological communities and their associations with the chemical and physical environment (the ecosystem)

**Species Diversity**

Species diversity includes the entire range of species found on Earth. Recognizing and classifying species is one of the major goals of conservation biology. A species is generally defined in one of two ways:

1. A group of individuals that is morphologically, physiologically, or biochemically distinct from other groups in some important characteristic; this is the morphological definition of species.
2. A group of individuals that can potentially breed among themselves in the wild and that do not breed with individuals of other groups; this is the biological definition of species.

Because the methods and assumptions used are different, these two approaches to distinguishing species sometimes do not give the same results. Increasingly, differences in DNA (deoxyribonucleic acid) sequences and other molecular markers distinguish species that look almost identical, such as bacteria. To further complicate matters, individuals of related but distinct species may occasionally mate and produce hybrids, intermediate forms that blur the distinction between species. Sometimes hybrids are better suited to their environment than either parent species, and they can go on to form new species. Hybridization is particularly common among plant species in disturbed habitats. Hybridization in both plants and animals frequently occurs when a few individuals of a rare species are surrounded by large numbers of a closely related species. For example, the endangered Ethiopian wolf (*Canis simensis*) frequently mates with domestic dogs, and declining British populations of the European wildcat (*Felis silvestris*) are being swamped with genetic material due to matings with domestic cats.

Much more work is needed to catalog and classify the world’s species. At best, taxonomists have described only one-third of the world’s species, and perhaps as little as 1 percent. The inability to clearly distinguish one species
from another, whether due to similarities of characteristics or to confusion over the correct scientific name, often slows down efforts at species protection. It is difficult to write precise, effective laws to protect a species if scientists and lawmakers are not certain what name should be used. At the same time, species are going extinct before they are even described. Tens of thousands of new species are being described each year, but even this rate is not fast enough. The key to solving this problem is to train more taxonomists, especially for work in the species-rich Tropics.

The Origin of New Species

The origination of new species—called speciation—is normally a slow process, taking place over hundreds, if not thousands, of generations. The evolution of new genera and families is an even slower process, lasting hundreds of thousands, or even millions, of years. Even though new species are arising all the time, the present rate of species extinction is probably more than one hundred times faster than the rate of speciation and may even be one thousand times faster. The situation is actually worse than this grim statistic suggests. First, the rate of speciation may actually be slowing down because so much of the Earth’s surface has been taken over for human use and no longer supports evolving biological communities. As habitats decline, fewer populations of each species exist, and thus there are fewer opportunities for evolution. Many of the existing protected areas and national parks may be too small to allow the process of speciation to occur. Second, many of the species threatened with extinction in the wild are the sole remaining representatives of their genus or family; examples include the gorilla (Gorilla gorilla), rapidly declining throughout its range in Africa, and the giant panda (Ailuropoda melanoleuca) in China. The extinction of taxonomically unique species representing ancient lineages is not balanced by the appearance of new species that are closely related to existing species.

Measuring Species Diversity

Conservation biologists often want to identify locations of high species diversity. In the broadest sense, species diversity is simply the number of different species in a place. Ecologists, however, have developed many other specialized, quantitative definitions of species diversity as a way to compare the overall diversity of different communities at varying geographical scales. Ecologists have used these quantitative measures to test the assumption that increasing levels of diversity lead to increasing community stability and biomass production. In controlled experiments in the greenhouse or gardens, or in grassland plant communities, increasing the number of species growing together generally leads to greater biomass production and resistance to drought. The significance of this result to the broader range of natural communities, such as forests and coral reefs, however, still needs to be convincingly demonstrated.

At its simplest level, diversity has been defined as the number of species found in a community, a measure often called species richness. Quantitative indexes of biodiversity have been developed primarily to denote species diversity at three different geographical scales. The number of species in a certain community or designated area is described as alpha diversity. Alpha diversity comes closest to the popular concept of species richness and can be used to compare the number of species in particular places or ecosystem types, such as lakes or forests. For example, a 100 hectare deciduous forest in New York or England has fewer tree species than a 100 hectare patch of the Amazon rain forest; that is, the alpha diversity of the rain forest is greater. More highly quantitative indexes such as the Shannon diversity index take the relative abundance of different species into account and assign the highest diversity to communities with large numbers of species that are equally abundant and the lowest scores to communities in which there are either few species, or a large number of species, one or a few of which are much more abundant than the others.

Gamma diversity applies to larger geographical scales. It refers to the number of species in a large region or on a continent. Gamma diversity allows us to compare large areas that encompass diverse landscapes or a wide geographical area. For example, Kenya, with one thousand species of forest birds, has a higher gamma diversity than Britain, which has only two hundred species.

Beta diversity links alpha and gamma diversity. It represents the rate of change of species composition along an environmental or geographical gradient. For example, if each lake in a region contained different fish species, or if the bird species on one mountain were entirely different from the birds on neighboring mounts, then beta diversity would be high. But if the species composition along the gradient does not change much (“the birds on this mountain are the same as the birds on the mountain we visited yesterday”), then beta diversity will be low. Beta diversity is sometimes calculated as the gamma diversity of a region divided by the average alpha diversity, though other measures also exist.

A greater diversity of species provides a larger range of potential human products, including everything from food and medicine to building materials and fuelwood. Species-rich ecosystems are also better able to provide...
ecosystem services that supply us with natural flood control, clean water, and pollution reduction.

Genetic Diversity

At each level of biological diversity—genetic, species, and community—conservation biologists study the mechanisms that alter or maintain diversity. Genetic diversity within a species is often affected by the reproductive behavior of individuals within populations. A population is a group of individuals that mate with one another and produce offspring; a species may include one or more separate populations. A population may consist of only a few individuals or millions of individuals, provided that the individuals actually produce offspring.

Individuals within a population usually are genetically different from one another. Genetic variation arises because individuals have slightly different forms of their genes (or loci), the units of the chromosomes that code for specific proteins. These different forms of a gene are known as alleles, and the differences originally arise through mutations—changes that occur in the DNA that constitutes an individual’s chromosomes. The various alleles of a gene may affect the development and physiology of an individual organism.

Genetic variation increases when offspring receive unique combinations of genes and chromosomes from their parents via the recombination of genes that occurs during sexual reproduction. Genes are exchanged between chromosomes, and new combinations are created when chromosomes from two parents combine to form a genetically unique offspring. Although mutations provide the basic material for genetic variation, the random rearrangement of alleles in different combinations that characterizes sexually reproducing species dramatically increases the potential for genetic variation.

The total array of genes and alleles in a population is the gene pool of the population, while the particular combination of alleles that any individual possesses is its genotype. The phenotype of an individual represents the morphological, physiological, anatomical, and biochemical characteristics of the individual that result from the expression of its genotype in a particular environment. Some characteristics of humans, such as the amount of body fat and tooth decay, are strikingly influenced by the environment, while other characteristics, such as eye color, blood type, and forms of certain enzymes, are determined predominantly by an individual’s genotype.

Sometimes individuals that differ genetically also differ in ways related to their survival or ability to reproduce—such as their ability to tolerate cold, resistance to disease, or the speed at which they can run away from danger. If individuals with certain alleles are better able to survive and produce offspring than individuals without these alleles, then gene frequencies in the population will change in subsequent generations. This phenomenon is called natural selection.

The amount of genetic variability in a population is determined by both the number of genes that have more than one allele (polymorphic genes) and the number of alleles for each of these genes. The existence of a polymorphic gene also means that some individuals in the population will be heterozygous for the gene; that is, they will receive a different allele of the gene from each parent. On the other hand, some individuals will be homozygous: they will receive the same allele from each parent. All these levels of genetic variation contribute to a population’s ability to adapt to a changing environment. Rare species often have less genetic variation than widespread species and, consequently, are more vulnerable to extinction when environmental conditions change.

Although most mating occurs within populations, individuals occasionally move from one population to another, resulting in the transfer of new alleles and genetic combinations between populations. This genetic transfer is referred to as gene flow. Natural gene flow between populations is sometimes interrupted by human activities, causing a reduction in the genetic variation in each population.

Genetic diversity allows species to survive in the face of a changing environment, giving them the greatest number of allele combinations and with it the traits necessary to survive and reproduce under new conditions. It also provides the basic material for the improvement of domestic species. Without genetic variation, improvements in agriculture would be more difficult.
Ecosystem Diversity

A biological community is defined as the species that occupy a particular locality and the interactions among those species. A biological community, together with its associated physical and chemical environment, is termed an ecosystem. Many characteristics of an ecosystem result from ongoing processes, including water cycles, nutrient cycles, and energy capture.

Within a biological community, species play different roles and differ in what they require to survive. For example, a given plant species might grow best in one type of soil under certain conditions of sunlight and moisture, be pollinated only by certain types of insects, and have its seeds dispersed by certain bird species. Similarly, animal species differ in their requirements, such as the types of food they eat and the types of resting places they prefer. Any of these requirements may become a limiting resource when it restricts population size of the species. For example, a bat species with specialized roosting requirements—forming colonies only in small grottoes on the ceilings of limestone caves—will be restricted by the number of caves with the proper conditions for roosting sites. If people damage the caves to collect limestone, then the bat population will probably decline; if the bats are able to adapt to human presence and roost under bridges, however, their population might increase.

Ecological Succession

As a result of its particular requirements, behaviors, or preferences, a given species often ends up appearing in a given site at a particular time during the process of ecological succession. Succession is the gradual process of change in species composition, community structure, soil chemistry, and microclimatic characteristics that occur following natural and human-caused disturbance in a biological community. For example, sun-loving butterflies and annual plants are most commonly found early in succession, in the months or few years immediately following a hurricane or after a logging operation has destroyed an old-growth forest. At this time, with the tree canopy disrupted, the ground is receiving high levels of sunlight, with high temperatures and low humidity during the day. Over the course of decades, the forest canopy is gradually reestablished. Different species, including shade-tolerant, moisture-requiring wildflowers, butterflies whose caterpillars feed on these plants, and birds that nest in holes in dead trees, thrive in these mid- and late-successional stages. Similar cases of species firmly associated with early, mid-, or late succession are found in other ecosystems, such as grasslands, wetlands, and the intertidal zones of oceans. Human management patterns often upset the natural pattern of succession; for instance, grasslands that have been overgrazed by cattle and forests from which all the large trees have been cut for timber no longer contain certain late-successional species.

Successional processes in modern landscapes might represent a combination of natural and human-caused disturbances. A grassland and forest community in the Rocky Mountains of Colorado, for instance, might be affected by natural fires, cycles of drought, and grazing by elk. Now succession in such a community is increasingly dominated by human-caused fires, cattle grazing, and road construction. Often, the largest number of species occurs in landscapes with intermediate levels of disturbance and a mixture of early, mid-, and late stages of succession.

Keystone Species and Guilds

Within biological communities, a particular species or groups of species with similar ecological features (guilds) may determine the ability of large numbers of other species to persist in the community. These keystone species affect the organization of the community to a far greater degree than one would predict, if considering only the number of individuals or the biomass of the keystone species. Protecting keystone species and guilds is a priority for conservation efforts, because loss of a keystone species or guild will lead to loss of numerous other species as well.

While we can sometimes identify such keystone species, it is also true that other species may be significant for ecosystem functioning in ways that are not immediately obvious. Top predators are often considered to be keystone species, because predators can markedly influence herbivore populations. The elimination of even a small number of individual predators, although they constitute only a minute amount of the community biomass, may result in dramatic changes in the vegetation and a great loss in biological diversity, sometimes called a trophic cascade. For example, in Cape Cod (Massachusetts) salt marshes, the common plant-eating marsh crab (Sesarma reticulatum) increased dramatically after populations of predators such as blue crab were reduced by overharvesting and water pollution. The subsequent increase in grazing pressure of the Sesarma has denuded 70 percent of the salt marsh cordgrass on Cape Cod, leading to soil erosion and a loss of protection for other species inhabiting the salt marsh (Bertness, Holdredge, and Altieri 2009).

As should be evident from the discussion thus far, the identification of keystone species has several important implications for conservation biology. First, the elimination of a keystone species or group from a community may precipitate the loss of other species. Losing keystones can create a series of linked extinction events, known as an extinction cascade, that results in a degraded ecosystem with much lower biological diversity at all trophic levels.
This may already be happening in tropical forests where overharvesting has drastically reduced the populations of birds and mammals that act as predators, seed dispersers, and herbivores. While such a forest appears to be green and healthy at first glance, it is really an “empty forest” in which ecological processes have been irreversibly altered such that the species composition of the forest will change over succeeding decades or centuries (Redford 1992).

Implications

Ecosystems provide the basic environmental services that human societies depend on. Overall, high biodiversity can help a species, population, or ecosystem persist, and it is therefore an important consideration for scientific researchers, planners, and managers. Understanding the dynamic relationships that exist can ensure that the actions we take are appropriate and sustainable.

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See also Biodiversity Hotspots; Biogeography; Biological Corridors; Boundary Ecotones; Buffers; Community Ecology; Ecosystem Services; Edge Effects; Food Webs; Habitat Fragmentation; Marine Protected Areas (MPAs); Population Dynamics; Refugia; Resilience; Species Reintroduction; Succession; Wilderness Areas

Further Reading


Many different conservation organizations have identified sets of “hotspots,” regions of the planet’s surface that they consider to be high priorities for conservation actions. Most are characterized by their high density of endemic species. By protecting these small regions, disproportionate amounts of the Earth’s biodiversity can be conserved. Hotspots have therefore been called conservation’s “silver bullet.”

The Earth is currently experiencing a mass extinction event, with species being lost at more than one hundred times the background extinction rate (i.e., long-term, non-anthropogenically influenced rates of species extinction). The majority of species extinctions are the result of human-caused habitat degradation, climate change, and invasive species (Mace et al. 2005). The creation of protected areas has proven to be an effective method of locally halting the process of degradation in many locations. Although more than 10 percent of the planet’s terrestrial area is currently designated as some form of protected area, habitats that are underrepresented require greater protection. Historically “protected areas” have been established where lands are not suitable for human usage (e.g., the Northeast Greenland National Park) and are therefore unlikely to be degraded in the near future; in contrast, many regions that are currently undergoing rapid habitat loss contain very few protected areas (e.g., tropical forests in Southeast Asia). Unfortunately, global biodiversity conservation has too few resources to protect all of the habitat at risk of degradation. Furthermore, protected areas cannot stop habitat loss entirely, and much of the degradation that would have occurred in new protected areas is simply displaced to unprotected habitat elsewhere. Finally, we do not have the financial resources to protect and manage all areas that require it.

Conservation organizations—both government and nongovernment—are responding in three ways to rapid biodiversity loss: (1) they are attempting to increase the resources available for conservation action by raising additional funds from donors or influencing the allocation of resources by governments; (2) they are working to slow the problem’s growth by reducing the opportunities for, and the drivers of, habitat degradation; and (3) they are ensuring that those resources that are available are being used as efficiently as possible. This final response—conservation resource prioritization—promises to increase the efficiency of conservation actions by targeting them where they are most needed. Its most familiar realization at a global scale is the “biodiversity hotspot.”

History of Biodiversity Hotspots

Hotspots are regions that contain a disproportionately large amount of biodiversity. If these hotspots are protected, it is argued, large amounts of the Earth’s biodiversity can be conserved for comparatively small investments. Their proponents argue that hotspots are therefore conservation’s “silver bullet”—a solution to both biodiversity’s overwhelming problems and conservation’s underwhelming resources. The original set of biodiversity hotspots was proposed by the British ecologist Norman Myers in 1988; it consisted of ten tropical forests. Covering only 0.2 percent of the Earth’s surface, they contained over thirty-four thousand endemic vascular plant species, or 13 percent of the world total. But these regions were also in immediate risk of losing their
many endemic species: less than 10 percent of the original vegetation remained intact in each area. Hotspots were thus identified as regions with high species richness and high levels of threat (as measured by past habitat loss). In 1990 Myers expanded the original set to a total of eighteen biodiversity hotspots, including four Mediterranean areas. In 2000, Myers and others classified an additional seven regions, and in 2004 the set of biodiversity hotspots was increased to thirty-four regions. (See figure 1.) The most compelling aspect of

**Figure 1. The 34 Biodiversity Hotspots**


The list of biodiversity hotspots as first proposed by the British ecologist Norman Myers in 1988 has increased over the years. The ten tropical forests Myers originally identified has since grown to thirty-four regions considered unique in their biological diversity.

Key:

1. The California Floristic Province
2. The Madrean Pine-Oak Woodlands
3. Mesoamerica
4. Tumbes-Chocó-Magdalena
5. The Caribbean Islands
6. The Tropical Andes
7. Chilean Winter Rainfall-Valdivian Forests
8. The Cerrado
9. The Atlantic Forest
10. The Mediterranean Basin
11. The Guinean Forests of West Africa
12. The Succulent Karoo
13. The Cape Floristic Region
14. Maputaland-Pondoland-Albany
15. Madagascar and the Indian Ocean Islands
16. The Coastal Forests of Eastern Africa
17. The Eastern Aframontane
18. The Horn of Africa
19. The Caucasus
20. The Irano-Anatolian
21. The Mountains of Central Asia
22. Eastern Himalaya
23. Western Ghats and Sri Lanka
24. The Mountains of Southwest China
25. Japan
26. The Philippines
27. Polynesia and Micronesia
28. Indo-Burma
29. Sundaland
30. Wallacea
31. East Melanesian Islands
32. Southwest Australia
33. New Caledonia
34. New Zealand
the biodiversity hotspots was their staggering species-to-area ratio. The most recent 2004 set of hotspots contained 150,000 endemic vascular plant species (50 percent of the world’s total) in just 2.3 percent of the Earth’s land surface. This high species-to-area ratio was assumed to be a proxy for a high species-protected-per-dollar ratio. As well as being species rich, biodiversity hotspots had experienced extensive habitat loss, with each region containing no more than 30 percent of their original vegetation.

In 1989, the biodiversity hotspots were adopted by the international nongovernmental organization (NGO) Conservation International as their institutional blueprint. After a number of revisions, the program took a central role in the organization’s conservation strategy. The set of biodiversity hotspots was relaunched in 1999 with an extensive global review, a scientific analysis, and an online publication that detailed each of the regions’ attributes (Mittermeier et al. 2005). By providing a clear and compelling focus to their global conservation decision making, the biodiversity hotspots greatly enhanced Conservation International’s fund-raising activities, and by 2003 it was estimated that the biodiversity hotspots program had attracted more than $750 million in funding for the NGO’s global conservation efforts (Brooks et al. 2006).

In retrospect, such a prioritization program was long overdue. The enormous scale of the threats to biodiversity made the challenges facing global conservation difficult to appreciate. Hotspots provided the NGOs with a plan for resource allocation that emphasized the ambitious scope of their actions, and yet it could be easily communicated to the public. Other NGOs and governments quickly appreciated that hotspots had allowed Conservation International to simultaneously address two of their key goals—prioritize their resource allocation while enhancing the total amount of those resources—and began to devise their own unique (and uniquely branded) hotspot programs.

Different Sets of Hotspots

The first decade of the twenty-first century consequently saw a number of other global conservation NGOs and national governments adopt the formula of the biodiversity hotspots, adapting and further developing the approach to deliver new methods that focused on their particular conservation issues of concern, carried their individual organizational branding, and corrected some of the oversights of the original program. Numerous lists of global priority regions have been described in the peer-reviewed scientific literature, as well as regional-scale hotspots typically defined within national boundaries (e.g., Australia’s National Biodiversity Hotspots). Conservationists now consider a broader definition of hotspots: a set of regions that have been identified as high priorities for conservation action and resources. The different hotspot programs can be broadly classified according to three features.

1. **Biodiversity definition.** Some hotspots focus on single taxonomic classes (e.g., plants); others combine multiple biodiversity attributes into a single aggregate value or identify areas that meet specific criteria (e.g., presence of species on the International Union for Conservation of Nature [IUCN] Red List of critically endangered species).

2. **Reactive or proactive.** “Reactive hotspots” identify target regions that have already lost large proportions of habitat. “Proactive hotspots” focus on areas that have endured the least human impact but which may be threatened in the future.

3. **Selectivity.** Each program implicitly chooses a different proportion of the land surface to define as high priority by defining criteria that are more or less common.

The new hotspot programs each chose different definitions of biodiversity. Most moved away from the original biodiversity hotspots’ focus on tropical forests toward more inclusive definitions of biodiversity. The biodiversity hotspots also broadened through the 1990s toward arid and Mediterranean ecoregions, and Conservation International also began to report the number of other taxa (e.g., mammals, freshwater fishes, amphibians) that are endemic to the priority regions. Other programs narrowed their focus onto other aspects of biodiversity, such as Birdlife International’s Endemic Bird Areas. Some sets of priority regions changed the focus from reactive hotspots (that had lost large amounts of habitat) to proactive hotspots, which remained almost entirely intact (e.g., the Wildlife Conservation Society’s Last of the Wild). For each new set of hotspots, new methods were developed for dividing the world’s surface into high- and low-priority regions. Few retained the threshold criteria of the biodiversity hotspots (i.e., more than 1,500 endemic plant species; more than 70 percent of the original vegetation lost), instead following more complicated systems that amalgamated the various regional attributes into a single score for each region. Some hotspot programs became less selective; for example, the set of “megadiverse countries” (countries that contain more than five thousand endemic vascular plant species) comprise more than one-third of the Earth’s surface (Mittermeier, Gil, and Mittermeier 1997). Figure 2 on the next page shows a classification of the different hotspot programs according to the primary criteria by which they were devised.
Figure 2. Global Programs for Conservation Priorities and Categorical Axes

Source: authors.

Each circle represents one global hotspots program (described in the Key). The location of the circle on the axes indicates the type of region the program considers a high priority. The x-axis measures the degree to which a program is focused on factors that are not replicated elsewhere (high irreplaceability) or are present in a broader range of regions (low irreplaceability). The y-axis indicates whether the program's focus is proactive or reactive. Circles have a larger radius if those hotspots consider only endemic species.

Key:

BH: Biodiversity Hotspots (Conservation International) prioritize areas that contain more than 1,500 endemic vascular plant species, where more than 70 percent of the original habitat has been cleared.

EBA: Endemic Bird Areas (Birdlife International) prioritize all areas where the distributions of at least 2 restricted-range bird species (species with ranges of less than 50,000 square kilometers) overlap.

G200: Global 200 (World Wide Fund for Nature) identify areas with high levels of species richness or endemism and/or significant evolutionary/ecological processes.

CE (V): Vulnerable Crisis Ecoregions (the Nature Conservancy and World Wide Fund for Nature) prioritize areas with moderate habitat degradation rates and ratios of degraded to protected habitat.

CE (C): Critically Endangered Crisis Ecoregions (the Nature Conservancy and World Wide Fund for Nature) prioritize areas with the highest habitat degradation rates and ratios of degraded to protected habitat.

MC: Megadiverse Countries (Conservation International), 17 countries each containing more than 5,000 vascular plant endemics.

FF: Frontier Forests (World Resource Institute) identify the most intact forests with high biodiversity values.

LW: Last of the Wild (Wildlife Conservation Society) select the 10 largest contiguous areas in each biome that are least affected by human impacts.

HBDA: High Biodiversity Wilderness Areas (Conservation International) prioritize areas of more than 10,000 square kilometers with human population density less than 5 people per square kilometer where less than 70 percent of the original vegetation remains intact.

Irreplaceability: If a global program prioritizes an area that is irreplaceable (Yes/No).

Vulnerability: If a program prioritizes area with high human impacts (reactive), avoid human impacts (proactive), or irrelevant to human activities (neutral).

Endemism: If a program focuses on endemic species (Yes/No).

Limitations of the Hotspots Idea

The proliferation of new programs highlighted several problems with the idea of hotspot programs.

First, conservation organizations could not agree on a common definition of biodiversity, and as a result, different programs identified different areas as high priority. Biodiversity is multifaceted, of course, and conservation organizations with distinct objectives routinely and appropriately pursue goals in different areas. But by 2006, 80 percent of the Earth’s surface was considered a conservation priority by at least one of the major global hotspot sets, voiding the argument that these sets could provide a “silver bullet” for global conservation’s funding mismatch.

Second, the methods applied by the different hotspot programs proposed diverging objectives. Some argued...
that focusing on the most damaged and threatened ecosystems would deliver the best conservation outcomes, since it was in these regions that species were facing imminent extinction. Others responded that conservation gains would be maximized by targeting intact landscapes, as it was only in these regions that viable populations of species and ecosystems could be protected in the long term. This conflict exposed unresolved questions in conservation resource allocation theory about the most effective way to combat ongoing degradation.

Finally, many critics pointed out crucial aspects of biodiversity conservation that were routinely omitted from the methods used to identify priority regions. Hotspots are almost exclusively defined by the distribution of species—either their richness, endemicity, or threat status—but global conservation is interested in objectives that are much broader and much harder to quantify. Genetic diversity and uniqueness, ecosystem services and function, and spatially equitable protection are each very difficult to include in sets of priority regions. As static maps, hotspots cannot respond to changing threats in a manner that theorists consider essential. They also generally ignore global variation in the costs of land acquisition and management, yet these factors will overwhelmingly determine whether a given set of hotspots represents cost-effective areas for investment. More recent analyses explicitly include cost-effectiveness when identifying hotspots, aiming to preserve those areas where large amounts of biodiversity can be conserved for relatively small costs.

The Future of Hotspots

As these criticisms arose, conservation organizations took steps to revise their methods for delineating hotspots and reduced their reliance on the priority sets. The focus of conservation organizations was beginning to shift away from a focus solely on the conservation of biodiversity, and toward activities with a dual focus of conserving both biodiversity and ecosystem goods and services (e.g., biodiversity and carbon storage and sequestration; or biodiversity and the provision of clean water). Actions that pursue multiple benefits can potentially access funding from a diversity of sources, including some that are inaccessible to projects that focus only on biodiversity conservation. These include multigovernment programs and emerging economic markets. For example, the United Nations’ Reducing Emissions from Deforestation and Forest Degradation in Developing Countries program (REDD+) is designed to reduce atmospheric greenhouse gases while delivering “co-benefits” of biodiversity conservation and poverty alleviation. Hotspots currently play a limited role in identifying priority areas for achieving multiple objectives.

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See also Biodiversity; Biogeography; Boundary Ecotones; Buffers; Charismatic Megaflora; Edge Effects; Keystone Species; Marine Protected Areas (MPAs); Microbial Ecosystem Processes; Refugia; Resilience; Wilderness Areas

The authors highly recommend the “A–Z Guide to Areas of Biodiversity Importance” (UNEP-WCMC 2010) as an introduction to various conservation programs around the world.

FURTHER READING


Biogeography analyzes the geographic distribution of taxa and their attributes in space and time. Biogeographic approaches broadly constitute two subdisciplines: ecological and evolutionary biogeography. Some of the issues addressed by evolutionary and ecological biogeography include species distribution, the geography of diversity, the geography of traits, endemity, biogeographic regionalization, biotic assembly and evolution on islands, biotic history, and conservation biogeography.

Biogeography is the study of the geographic distribution of taxa and their attributes in space and time. A taxon (plural taxa) is a unit or group of biological organisms that is considered distinct enough to be formally recognized and assigned to a category, such as a kingdom, phylum, class, order, family, genus, or species. Biogeography encompasses the recognition of distributional patterns, the biogeographic regionalization of the Earth, the identification of the processes that shaped distributional patterns, the prediction of global planetary changes, and the selection of areas for biodiversity conservation (Morrone 2009).

Biogeographic approaches broadly constitute two subdisciplines: ecological and evolutionary biogeography. Ecological biogeography analyzes patterns at the species or population level, accounting for distributions in terms of biotic (living organisms) and abiotic (nonliving physical and chemical elements) interactions that happen in short periods of time. Evolutionary biogeography—also known as historical biogeography—analyzes patterns of species and supraspecific taxa. It concerns processes occurring over long periods of time. This distinction, however, is rather artificial, because it implies splitting a continuum, where extremes are easily identifiable as “ecological” or “evolutionary.” It is more difficult to justify such division in the middle range. In fact, ecological factors may have wide geographic effects, while historical factors may be responsible for local distributional patterns. Biogeographers have criticized the lack of interaction between evolutionary and ecological biogeography since the 1990s. Several authors have discussed the possibility of integrating them into a unified discipline (Morrone 2009).

Some of the issues addressed by evolutionary and ecological biogeographers include species distribution, the geography of diversity, the geography of traits, endemity, biogeographic regionalization, biotic assembly and evolution on islands, biotic history, and conservation biogeography.

Distribution of Species

Each plant and animal species occupies a particular geographic range. Some species, known as cosmopolitan species, have extensive ranges over several continents. Others have more restricted geographic ranges; they are distributed in a small area on a single continent.

There are different ways to represent the geographic range of a species on a map:

- Dot maps represent each locality where a species has been recorded as a point on a map.
- Outline maps depict an irregular area where the species is presumably distributed.
- Contour maps illustrate some variation among individuals or populations.
- Individual tracks show localities where a species has been recorded joined by a line graph connecting them according to their geographic proximity.
Ecological niche models are predictive maps of species distributions based on known or inferred distributions and data layers summarizing the distribution of some environmental parameters (e.g., temperature, altitude, ocean depth, days of ice cover, etc.).

Areography—also known as chorology—is the quantitative analysis of distribution areas (Rapoport 1982). Areographers analyze the delimitation of distribution areas, the variation in size of distribution areas of different species, endemism versus cosmopolitism, the shape of different distribution areas, and population density within the distribution area of a species.

**Geography of Diversity**

Different measures describe the structure of communities and regional biotas. Species richness, one of the most commonly considered measures, is simply the number of species in an area. Species richness can be classified into four categories:

1. **Alpha diversity**: the number of species recorded for a local community
2. **Beta diversity**: the change or turnover in species composition over a relatively small distance between different communities
3. **Gamma diversity**: the total species number of a large area, from a combination of local communities to entire continents
4. **Delta diversity**: a broad-scale measure of the richness between large geographic areas, as biogeographic regions

Biogeographers have identified some patterns in species richness. The latitudinal gradient measures increasing species numbers from the poles to the equator. This pattern has proved to be true for several taxa and has been also detected for other measures of diversity. Since the 1990s, biogeographers have proposed different factors influencing the latitudinal gradient, such as evolution, immigration, extinction, and ecological interactions.

**Geography of Traits**

Macroecology deals with the study of relationships between organisms and their environment at large spatial scales to characterize and explain statistical patterns of different traits, such as species abundance and richness, latitudinal diversity patterns, the species-area curve, range size, and body size (Brown 1995; Gaston and Blackburn 2000). Macroecological analyses use a top-down approach to understand properties of the ecosystems as a whole. A typical macroecological question may analyze, for example, the relationship between the abundance and range size within a supraspecific taxon or ask why species that maintain large local population sizes tend to be widely distributed, whereas species that are less abundant tend to have restricted ranges.

**Endemicity**

Endemicity or endemism refers to a taxon restricted to a particular geographical area. It represents a basic feature of geographic distributions: species are rarely cosmopolitan. Most species and even supraspecific taxa are confined to restricted regions. Endemism occurs on a variety of spatial scales, from areas as large as continents to small areas such as islands or mountaintops. Organisms can be endemic on different taxonomic levels; usually the size of the area depends on the category of the taxon, with genera having larger areas than species, and families having larger areas than genera. This situation, however, is not comparable between different taxa: the distribution of a plant family may correspond to the distribution of an insect genus.

Endemic taxa may be classified as follows:

- **Autochtonous endemics**: taxa that evolved in the areas where they are currently found
- **Allochtonous endemics**: taxa that evolved in a different area from where they are found today
- **Taxonomic relicts**: sole survivors of a once diverse group
- **Biogeographic relicts**: narrowly endemic descendants of a once widespread taxon
- **Neoendemics**: taxa that have evolved relatively recently and may be restricted in their distribution because they have not had yet time to disperse farther
- **Paleoendemics**: taxa that have a long evolutionary history and usually are restricted by barriers to dispersal or by extensive extinction in the remaining areas where they were distributed in the past

Areas where the distributional areas of two or more taxa overlap are called areas of endemism or endemic areas. If biogeographers map the distributional ranges of relatively well-known taxa, the substantial overlap in their ranges determines an area of endemism. With only a few taxa, this is an easy task. Difficulties may arise with a high number of taxa to analyze. Methods have been designed to deal with this, however. An alternative approach for analyzing endemity, known as panbiogeography, plots distribution of different taxa on maps, connecting their separate localities together with lines called individual tracks (Craw, Grehan, and Heads 1999). When different individual tracks are superimposed, the resulting summary lines are considered generalized tracks. Generalized tracks indicate the preexistence of ancestral biotas, which subsequently become fragmented by tectonic and/or climatic changes (Morrone 2009).
Biogeographic Regionalization

Ecological biogeographers classify communities on the basis of structure of the vegetation, assuming that plant life-forms reflect the influence of climate and soil. Earth’s vegetation can be classified into the following main biomes (geographical areas classified according to the predominant vegetation—or lack thereof—and characterized by adaptations of organisms to particular environments; see figure 1):

- **Tundra**: treeless biome, found between the taiga and the polar ice cap or at high mountainous elevations, in areas characterized by harsh winters and short growing seasons
- **Taiga**: boreal or swamp forest, occurring in a broad band across North America and Eurasia, in cool and moist areas
- **Temperate deciduous forest**: tree-dominated assemblage with a continuous canopy that occurs in temperate latitudes where there is enough water during the summer growing season to support large trees
- **Subtropical evergreen forest**: tree-dominated assemblage with a continuous canopy that occurs in hot lowlands outside the equatorial zone, where rainfall is more seasonal
- **Temperate grassland**: assemblage in which grasses predominate, situated biogeographically and climatically between deserts and temperate deciduous forests, most extensive in the interior plains of the Northern Hemisphere.
- **Desert**: assemblage with very sparse plant cover in which most of the ground is bare, found around the world at low to intermediate elevation with dry climate.
- **Tropical deciduous forest**: tree-dominated assemblage with a continuous canopy that occurs in hot lowlands outside the equatorial zone, where rainfall is more seasonal.
- **Tropical savanna**: nearly continuous layer of xerophytic perennial grasses (grasses that do not need water) scattered among fire-resistant trees or shrubs that occurs at low to intermediate elevations along tropical latitudes.
- **Tropical rain forest**: tree-dominated assemblage with a continuous canopy, found at low elevations along tropical latitudes with abundant rainfall.

**Figure 1. Earth’s Main Biomes**

Source: Lomolino et al. (2010).

Ecological biogeographers divide the Earth’s vegetation zones into the following biomes: 1, tundra and ice; 2, taiga; 3, temperate deciduous forest and subtropical evergreen forest; 4, temperate grassland; 5, desert; 6, tropical deciduous forest and tropical savanna; 7, tropical rain forest. Evolutionary biogeographers divide the world up differently than ecological biogeographers do; see figure 2.
Evolutionary biogeographers take a completely different approach, basing their regionalizations on endemicty. The fact that areas of endemism are nested, with large areas including smaller ones, allows proposing a hierarchical biogeographical classification that parallels the taxonomic Linnaean hierarchy, employing the following subdivisions: realms (or kingdoms), regions, dominions, provinces, and districts. Based on a consensus of studies undertaken since the 1980s, evolutionary biogeographers have proposed the following system for the world (Morrone 2002; see figure 2):

- **Holarctic realm**: Europe, Asia north of the Himalayan mountains, northern Africa, North America, and Greenland. From a paleogeographic viewpoint, it corresponds to the paleocontinent of Laurasia. It is divided into the Nearctic and Palearctic regions.
- **Holotropical realm**: Tropical areas of the world, between latitudes 30° south and 30° north. From a paleogeographic viewpoint, it corresponds to the eastern portion of the Gondwana paleocontinent. It is divided into the Neotropical, Afrotropical, Oriental, and Australian Tropical regions.
- **Austral realm**: Southern temperate areas in South America, South Africa, Australasia, and Antarctica. From a paleogeographic viewpoint, it corresponds to the western portion of the Gondwana paleocontinent. It is divided into the Andean, Antarctic, Cape, Neoguinean, Australian Temperate, and Neozelandic regions.

### Biotic Assembly and Evolution on Islands

Biogeographers find islands particularly interesting. Their isolation and particular biotas make them natural laboratories for studying different biogeographic patterns and processes. The first general theory for

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**Figure 2. Earth’s Biogeographic Realms and Regions**


Evolutionary biogeographers, as opposed to ecological biogeographers, instead divide the world into sometimes overlapping biogeographical realms: 1–2, **Holarctic realm**: 1, Nearctic region; 2, Palearctic region; 3–6, **Holotropical realm**: 3, Neotropical region; 4, Afrotropical region; 5, Oriental region; 6, Australian tropical region; 7–12, **Austral realm**: 7, Andean region; 8, Cape region; 9, Neoguinean region; 10, Australian temperate region; 11, Neozelandic region; 12, Antarctic region.
explaining differences in species richness among islands was formulated by the US biologists Robert MacArthur and Edward O. Wilson in the 1960s. Their theory proposes that the number of species found on an island is determined by the equilibrium of immigration and extinction. The distance of an island from the source of colonists in the mainland affects the rate of immigration, while the size of the island affects the rate of extinction. Larger islands contain larger habitat areas and opportunities for different habitats and reduce the probability of extinction due to chance events (Whittaker and Fernández-Palacios 2007). The countervailing forces of immigration and extinction result in an equilibrium of species richness.

Biotic History

In order to reconstruct the evolutionary relationships of biotas, cladistic or vicariance biogeography assumes a correspondence between phylogenetic relationships and area relationships. A cladistic biogeographic analysis comprises three basic steps. First, biogeographers construct taxon-area cladograms, or evolutionary diagrams, from the cladograms (phylogenetic hypotheses) of different taxa by replacing their terminal taxa by the area(s) of endemism where they are found. Then they convert these taxon-area cladograms into resolved area cladograms, so that each terminal taxon is endemic to a single area and each area has a single taxon. Finally, they derive a general area cladogram that represents the most logical solution for all the resolved area cladograms analyzed and represents a hypothesis on the evolutionary history of the analyzed areas.

Outlook: Conservation Biogeography

Biodiversity is in global crisis. Natural habitats are disappearing at a very high rate, and the numbers of plant and animal species going extinct are alarming. For example, about two thousand species of Pacific Island birds (about 15 percent of the world total) have gone extinct since human colonization. One of the major goals of conservation is the maintenance of as much of the diversity of life as possible, in order to allow a sustainable use to future generations. We have to measure and compare priorities regarding areas to be protected, and we need to measure and compare local biodiversity, taking into account not only the number of species, but also the degree of difference among them. One criterion for measuring biodiversity is species richness, exemplified by the megadiversity countries, which are the countries that harbor the majority of Earth’s species, namely, Australia, Brazil, China, Colombia, Democratic Republic of the Congo, Ecuador, India, Indonesia, Madagascar, Malaysia, Mexico, Papua New Guinea, Peru, Philippines, South Africa, United States, and Venezuela. Many important components of biodiversity may be poorly represented in megadiverse countries, however, and some areas could harbor a large number of widespread species, with no great conservation concern.

Ecological biogeographers have developed diversity measures that combine species richness with information about abundance among species, and information on the vulnerability of those species, including the “hotspots” analysis. Some authors have suggested that endemicity may help determine priorities for biodiversity conservation, whereas others have argued that it is not an appropriate measure of diversity, and that it is an ineffective means for selecting areas for conservation. Some evolutionary biogeographers have suggested that panbiogeographic nodes (areas of overlap of different generalized tracks) should be considered as prioritary areas because of their biotic richness.

Whatever these diversity measures may mean, preservation of biodiversity is essential for the sustainability of Earth. The long-term maintenance of areas harboring biodiversity has profound environmental, economical, and social dimensions, which are a necessary precondition for human well-being.

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See also Biodiversity; Biodiversity Hotspots; Biological Corridors; Boundary Ecotones; Buffers; Charismatic Megafauna; Edge Effects; Forest Management; Habitat
Fragmentation; Human Ecology; Marine Protected Areas (MPAs); Population Dynamics; Species Reintroduction

**FURTHER READING**


With increasing fragmentation of our planet’s natural communities, biological corridors are critically important tools for helping us to maintain as much connectivity as possible among the fragments. Fostering connectivity is essential for conserving biodiversity and for maintaining human welfare and sustainability on planet Earth. In conservation planning projects, potential negative effects of corridors must be considered, as well as their economic, social, and political context.

All parts of the Earth are interconnected to some degree. This means that people cannot isolate themselves from the extraordinary assemblage of living creatures with whom they share the planet, and upon which they are dependent for their existence. The challenge is to sustain a healthy biosphere on Earth so that humanity has a chance to thrive into the foreseeable future. That is the message of sustainability.

The impact (footprint) of humanity on the biota and nonliving resources of the planet is accelerating, and most experts believe it already exceeds sustainable levels. One consequence of this predicament is that the distributions of living organisms are becoming increasingly fragmented. This fragmentation brings reductions in connectivity among the fragments, which is one of the important factors leading to species extinctions, threats of extinctions, and disruptions in the functioning of biological communities. The question is: are biological corridors useful in helping to achieve sustainability through improved connectivity?

What Are Corridors?

Biological corridors are any real estate, terrestrial, aquatic, or both, that increases the ability of organisms to move among habitat fragments in search of something they need for their livelihood. They may need a suitable place to live, some specific resource (for example, a hollow tree or preferred prey), a desirable mate, or a chance to avoid threats of competition, predation, parasitism, or disease.

Corridors come in many shapes and sizes and can have diverse objectives. The ultimate corridor is a large strip of habitat much like that of the patches that it connects. This is the best way to ensure that the entire community of organisms found in the fragments or patches can readily move between them. It is unrealistic, however, to expect that this kind of supercorridor can be conserved or reconstructed very often. Most often, corridors must have more limited objectives as well as being subject to existing land use constraints and economic considerations. Rather than providing for the movements of all species in the fragments to be joined, a corridor may be useful for certain species that are most harmed by the fragmentation or are of special conservation concern, such as rare and endangered species. A corridor might simply be a culvert under a highway that can be used by large and mobile species. Other corridors might be passable only seasonally or under unusual or extreme conditions such as flooding or prolonged drought. Corridors may have native vegetation, be dominated by invasive species, be planted with horticultural varieties, or even be an agricultural crop. An essential consideration is that corridors can exist over a wide range of spatial scales. They range from a few meters in length, perhaps connecting two patches of grass, to continental in extent, such as the entire length of the North American Rocky Mountains. The critical criterion for a corridor is that it improves connectivity.

Connectivity

Movements across landscapes have always been an issue for organisms. Places suitable for each kind of organism are not found everywhere. Moreover, there are serious
barriers separating continents, ocean basins, lakes and rivers, climatic zones, mountain tops, and so on. Organisms are always exploring, searching, and experimenting in their attempts to find what they need or to improve their chances for success.

Species, however, differ greatly in their abilities to move across space, and this movement is complicated by humans when we convert space to our own needs, add toxic substances to the environment, introduce alien species, and co-opt valued resources. Generally, humans make it more difficult for species to move successfully (disperse), so that we reduce their connectivity. In other cases, we enhance connectivity by transporting organisms inadvertently in our baggage or by intentionally bringing them to new places.

**Extinctions**

Why should reduced connectivity be a problem for organisms? Organisms are constantly searching for new, better, or safer places to live. They may also have to move in response to seasonal changes, that is, migrate, or to adjust to long-term climatic changes. Of special concern is that isolated populations carry higher risks of extinction. Survival is especially precarious if population numbers are low or experience low points regularly or occasionally. Random catastrophes can readily eliminate a small population, whereas a larger one will usually have some survivors. Small populations also risk genetic deterioration through inbreeding that can lead to reduced fitness because of loss of hybrid vigor, or suffer the loss of genes that are critical for population persistence (polymorphisms). Examples of such critical variants are males and females, perhaps, or one type of individual that is more fit for winter conditions and another that does better in the summer. Another genetic risk is the loss of the population’s ability to adapt to long-term changes.

Small populations of some species face another important threat, and that is so-called Allee effects (Lidicker 2010). These occur when some aspect of a species’ environment or life history makes it increasingly difficult to survive as numbers decline. Generally, these are found in species with complex social systems that require a certain minimum number in order for the social group to function properly. This might be joint protection from predators, for example. It might also take the form of a flowering plant that requires an insect pollinator. If the plant lives in only a small, isolated patch, the pollinators may never find the patch, and hence there will be no reproduction.

Still another threat faced by organisms living in small, isolated patches is cascading extinctions. Suppose one species becomes extinct in the patch. This may lead to another species becoming extinct, and this in turn to still another, and so on. The chain reaction is the result of the fact that, just like humans, all other living organisms depend for their existence on other species in their community. Extinction of a species that is the favorite or essential prey of another will lead to the predators’ extinction. Loss of a tree species that provides hollows needed by others for nesting will cause the nesting species to become extinct as well. Extinction of a predator may result in higher densities of its prey, which in turn might cause the extinction of several rare species that successfully competed with the prey species as long as it was not highly populous.

**When Are Corridors Not a Good Idea?**

Corridors are a useful, often essential, tool for conservation and land management. Moreover, this need will be increasingly important as humans continue to convert natural habitats to their own immediate use, and thus intensify the fragmentation of the world’s biota. Accelerating losses of biodiversity can be confidently predicted as can a parallel decline in the Earth’s carrying capacity for humans. Does this mean that corridors are always the solution to conservation problems? They are not.

There are many reasons why a particular corridor may fail to provide the benefits for which it was intended. It might even make a bad situation worse. For every conservation effort in which corridors are considered as connectivity tools, the pros and cons need to be carefully evaluated. Moreover, site-specific or context-dependent issues enter into the equation for each project. This is the reason no universal formulae apply to solving connectivity problems. Land managers, politicians, and biologists need to be aware of potential causes for failure so that they can anticipate problems, plan for their mitigation, monitor their results, and prepare to adaptively manage their projects accordingly.

There are several potential disadvantages of corridors (see Hilty, Lidicker, and Merenlender 2006, chapter 6, for an expanded discussion).

1. **Edge Effects**—Most corridors are relatively narrow, and hence, because their edge-to-area ratio is large, they are likely to be dominated by so-called edge effects. These effects emerge when two types of communities or habitats share a common boundary. Species from one community may invade the other, some species may avoid living in or even moving through edge zones, and species that live only in edges may appear and even become abundant. Plants and animals attempting to use a corridor may be thwarted by unfamiliar predators, parasites, or competitors, and so may not use what seems to
7. **Genetic Impacts**—Genetic connectedness is generally a positive influence in helping populations to avoid the negative effects of inbreeding and of maintaining sufficient genetic heterogeneity to allow adaptation to local or changing conditions. Where long-isolated populations are newly connected by a corridor, however, negative genetic effects can be anticipated (Rhymer and Simberloff 1996). Local adaptations can be disrupted, lowering the overall fitness of a population to its environment. More unusual are instances where higher taxonomic levels are impacted by new corridors. Subspecies and even species have disappeared, and native species have been replaced by exotic species or hybridized out of existence (Rhymer and Simberloff 1996).

### Economic and Social Factors

People may preserve, enhance, create, and study corridors from a strictly scientific perspective, but these activities are invariably done in an economic, social, and political context (Hilty, Lidicker, and Merenlender 2006). Costs include land acquisition and possibly construction, maintenance, monitoring, and lost opportunity costs. In addition, potential costs of negative impacts on adjacent areas may be unforeseen or underestimated.

The benefits of conservation objectives are rarely measured accurately in monetary terms. A new breed of economist is struggling with trying to improve the economic quantification of conservation activities (Costanza et al. 1997). The reality, however, is that conservation activities address quality of life issues, the sustainability of human life, and the critical role of Earth’s biota in making both these objectives possible. It may be difficult, but it is critically important to keep nonmonetary benefits in the balance sheets.

### Outlook

Although there are many potential negative outcomes of establishing corridors, they are frequently a good choice in conservation planning. Corridors will continue to be important, even critically necessary, in most situations, especially if potential pitfalls are carefully considered.

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See also Adaptive Resource Management (ARM); Boundary Ecotones; Edge Effects; Habitat Fragmentation; Invasive Species; Landscape Planning, Large-Scale; Light Pollution and Biological Systems; Outbreak Species; Plant-Animal Interactions; Refugia; Regime Shifts; Rewilding; Road Ecology; Wilderness Areas
FURTHER READING


Nowhere else can the tenets of ecology and ecosystem science be studied and tested better than within the boundary ecotones that separate communities across the landscape. They can also provide sensitive and early evaluation of changes in species and community distribution patterns resulting from such disturbances as agriculture and climate change. These important zones offer unique challenges that ecosystem managers are only beginning to understand.

Ecological boundary ecotones are most often defined as the transition areas between two distinct plant communities, including the overlap of their respective flora and fauna. These same areas may also incorporate the concept of edge effects, wherein ecotonal species may include more or less individuals of the same species found in the flanking communities, and even species unique to the ecotone (i.e., edge species). Common examples of ecotones include the boundaries between desert and shrubland, grassland and shrubland, shrubland and forest, forest and alpine or arctic tundra, wet and dry meadows and forest, and forest and steppe, to name only a few.

In addition, there are boundary ecotones separating both freshwater and saltwater communities from a large variety of terrestrial communities, such as beach/ocean, marsh/estuary, stream/riparian, and bog/forest. Many species of these ecotones must navigate from the terrestrial to the aquatic community, and vice versa, for reproductive purposes.

Traditionally, ecotones have been of particular interest to ecologists owing to the abundance of animal species that utilize these transition zones on a daily basis in order to complete their life cycles. For example, mammals and birds may utilize a forest meadow for foraging but the forest interior for nesting, bedding down, birthing, protection from predators, and so forth, at different times of the day and season. Predators may use one community for camouflage while stalking prey in the ecotone and adjacent community. Thus, the importance of ecotones on ecosystem processes of energy exchange and nutrient cycling depends directly on the amount of edge perimeter compared to the total area of the adjoining communities. For these reasons, land management objectives emphasizing game management and not necessarily ecosystem sustainability often attempt to regulate vegetation patterns (e.g., forest timber harvesting) by maximizing the ratio of edge to total area—a potential benefit to species dependent on the ecotone that also favors hunters of those same species.

Additionally, when species travel across an ecotone to access the advantages of both communities, they may also encounter trade-offs such as greater exposure to stress factors that are both abiotic (e.g., greater sun exposure, drier soils) and/or biotic (e.g., predators, competition).

Why Study Ecotones?

Ecotones can provide the most sensitive and earliest indication of potential shifts in the spatial distribution patterns of plant communities. It has been argued that the impacts of global change, such as elevated carbon dioxide in the atmosphere, atmospheric warming, and sea level rise, may be first recognized within the transitional boundaries between communities. For example, the degree of establishment of plant seedlings or vegetative sprouts from either community into the transitional ecotone can provide early evidence for the ultimate movement, expansion, or contraction of community...
boundaries. These data can also provide estimates of ecological encroachment, species extinction potential, and the possible replacement of one community by another with consequential losses in biodiversity across the landscape.

Boundary ecotones are also thought to be the most stressful situation for growth and survival of individual species of either community and, thus, provide an outdoor experiment evaluating adaptation, or why the individual species of a particular community grows and survives at a particular location. The physiological limitations to growth and the capability of producing viable offspring in the boundary ecotone can explain more mechanistically the spatial distribution pattern observed for a species or community. These limitations may be generated by abiotic conditions of the physiochemical environment, or by biotic interactions such as ecological competition and facilitation, or by both. The interaction between competition and facilitation in boundary ecotones is a relatively unstudied research topic (e.g., Baumeister and Callaway 2006).

An Ecotone Example

One of the more commonly studied ecotones since the 1900s is forest treelines on mountains, especially at the upper elevational limit—the alpine treeline. The ecophysiological and environmental causes of why trees occur at different elevational limits worldwide has intrigued forest scientists for decades, while more recently the movements of treelines in response to climate change (elevated carbon dioxide and atmospheric warming) is of interest for validating some predicted climate change effects on forest spatial patterns of the future.

These treelines often appear as sharp boundaries from a distance (e.g., view from an airplane), although their transitional nature becomes apparent on the ground. A blend of both communities, forest and alpine tundra, occurs gradually across the treeline ecotone, with fewer and fewer trees at greater distances from the leading edge of the intact forest (Smith et al. 2003, 2009). These trees commonly become more and more distorted in growth form and are separated by greater and greater distances from one another. Closer to the forest edge, trees may cluster into islands and are severely flagged (indicating wind direction). These tree islands diminish in size both vertically and horizontally with greater distance from the forest edge, becoming progressively more stunted and bush-like in appearance. Ultimately, single trees that are found at the upper elevational limit of the species resemble small shrubs with a strong distortion in form that reflects the prevailing wind direction during winter (similar to the flagging of taller trees lower in the ecotone).

Detecting early changes in this boundary ecotone is possible through the study of tree seedling occurrence and age class distribution as an indication of which particular year’s seedling germination and establishment took place. Ultimately, the establishment and growth of new seedlings and saplings, along with the distorted mature tree forms, will provide enough protection (facilitation) for trees to grow to maturity, forming a new subalpine forest with forest-stature trees. In other words, forest-like trees will not be possible until some degree of forest develops that can protect the developing trees.

Tree seedlings in this ecotone also compete with alpine species such as grasses, sedges, and herbaceous species for such resources as limited water; at the same time, they may be facilitated by the other species’ effects on tree seedling microclimate, for example, shading from intense sunlight that can cause high seedling mortality. Thus, the question of the interplay of competition versus facilitation seems not to be answered for this ecotone, or virtually all others. To date, most research on ecotonal interactions among the overlapping species from each bordering community has been limited. In fact, the majority of alpine treeline studies have involved mature trees at treeline, not establishing seedlings in the ecotone between the forest and alpine tundra, a fundamental determinant of changes in treeline altitude.

Ecotones and Sustainability Management

Boundary ecotones such as the alpine treeline deserve particular attention for ecosystem management and the future sustainability of natural ecosystems. These transitional areas between two distinct communities are where changes in spatial distribution patterns may be most detectable earliest. If anthropogenic disturbances, including global change factors, occur in ecotones, we may expect almost immediate effects on the spatial stability of the contiguous communities. For example, rising sea level will submerge great amounts of coastal shoreline and shift the intertidal community to higher elevations. Similarly, disturbances influencing propagation in the ecotone by either of the adjoining communities will have a potentially strong influence on their spatial stability. What ultimate effect this will have on existing spatial patterns and species composition in terrestrial/oceanic ecotones is unknown. What can be expected is the potential loss, and possible extinction, of species that are endemic to these ecotones and not found in either of the contiguous communities. Thus, some of the most highly adapted organisms may be lost to extinction if sea level rise is rapid enough to disallow adaptation and selection
of phenotypes and, ultimately, new genotypes and species.

The inherent nested and hierarchical complexity of ecotones reflects the kind of complexity observed across the ecological sciences. There are ecotones within ecotones, depending on spatial and temporal scale. As one reduces spatial scale, such as from the size of the entire ecotone separating the forest from the alpine to a single large boulder within the ecotone, other smaller ecotones associated with the boulder become apparent. These smaller ecotones result from varying abiotic factors associated with the microclimate generated by the boulder itself (e.g., sunlight exposure, temperature, soil water, wind) as well as biotic factors associated with the abiotic variation. Similar problems of scale arise when considering changes in this concept of ecotones-within-ecotones due to seasonal variation (e.g., summer versus winter), or even variation in abiotic and biotic conditions during a day.

Differences in the abundance and variety of smaller-scale ecotones found within larger ecotones could have a strong influence on such important ecological characteristics as species composition and frequency across an ecotone. Comprehensive measurements of scaling effects on ecotonal disturbance could be pivotal in estimating the impacts of ecotonal disturbance on species diversity and abundance and ecosystem sustainability, as well as for designing land management strategies under a future of global climate change. Few such studies have been attempted to date (see Gosz 1993 for a review), but some research campaigns have been designed to address the contribution of ecotones to overall ecosystem services and functioning (e.g., Naiman, Decamps, and Fournier 1990; McArthur, Ostler, and Wambolt 1999), and there have been reviews dealing with the conceptual foundations of ecotone ecology (Hufkins, Scheunders, and Ceulemans 2009). There have also been attempts at characterizing ecotones as “fuzzy sets” in an effort to describe more precisely the landscape distribution and abundance of land area classified as ecotonal (Morris and Kokhan 2007).

The influence of ecotones on ecosystem properties across the landscape has received considerable study. Yet much more research is necessary to achieve a comprehensive understanding. Sustaining ecosystems in the future will certainly demand attention and focus on the critical importance of ecotones, despite the common practice of altering the character and abundance of these zones by humans.

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See also Biodiversity; Biodiversity Hotspots; Biogeography; Biological Corridors; Buffers; Ecological Forecasting; Edge Effects; Global Climate Change; Habitat Fragmentation; Hunting; Indicator Species; Invasive Species; Keystone Species; Outbreak Species; Plant-Animal Interactions; Species Reintroduction; Wilderness Areas

FURTHER READING


Brownfield Redevelopment

Brownfields are areas that have been polluted through industrial or commercial use and abandoned. Redeveloping such areas to beneficial and sustainable uses involves the often difficult and expensive process of cleaning up contaminants before new development can proceed. Reclaiming brownfield lands is, however, a significant aspect of long-term sustainable land use.

Brownfield redevelopment is a three-step process by which developers purchase a contaminated property from its previous, often industrial, owner, remediate it, and finally redevelop it. Though the process itself sounds simple, it is often difficult to entice investors to purchase blighted land for which they may be liable for future environmental and health risks. Additionally, the high cost of remediating the various contaminants on the property, including polluted ground- and surface water, may discourage investment. The benefits, however, of reclaiming despoiled sites include improvements to the environment, the tax base, and the economic and social well-being of the area.

The majority of literature on brownfield redevelopment quotes the definition put forth by the United States Environmental Protection Agency (US EPA 1997) of brownfields as “abandoned, idled or underused industrial and commercial facilities where expansion or redevelopment is complicated by real or perceived contamination.” Notably, this definition is open-ended and suggests that a site need only be perceived as contaminated in order to be considered a brownfield. Similar definitions are provided by the National Round Table on the Environment and the Economy in Canada (NRTEE 1998), the United Kingdom Environment Agency (UKEA 2011), and by the US geographer Linda McCarthy in her 2002 article in Land Use Policy.

Social, Economic, Environmental, and Legal Issues

Brownfields are a widespread problem in North America, Europe, and the developing world. There are more than 500,000 brownfields in the United States; 360,000 brownfields in Germany; 33,000 hectares of brownfields in the United Kingdom; and over 30,000 such properties in Canada (NRTEE 2003). Brownfield redevelopment projects can impact the social, economic, environmental, and legal sectors. Economic impacts are important to the redevelopment of brownfields and can lead to social consequences; for example, the presence of brownfields within a neighborhood can often lower property values of adjacent properties and result in a lower tax base for the whole community. The income reduction may lead to a decrease in services, including policing, fire protection, hospital facilities, road maintenance, and garbage collection (Greenberg and Lewis 2000). This reduction in services results in additional industries leaving the affected area. For instance, the US professor of environmental management and urban development Christopher De Sousa (2003) notes that many of the brownfields in the city of Toronto, Canada, are a result of the exodus of industry in the 1970s from the blighted downtown core. Because brownfields can lead to a reduction in services and property values, the adjacent communities are often plagued by crime, unemployment, and poor schools. In some cases brownfields can pose threats to human health from groundwater contamination. These environmental justice challenges associated with brownfields too often occur in communities of color and poverty.
Financial risk due to the unknown cost of remediating a brownfield site is an important consideration for private redevelopment investors. De Sousa (2000) found that two key financial issues—liability concerns and high remediation costs—were among the most pressing obstacles to private brownfield redevelopment. He also discovered that uncertainty related to a site-specific risk assessment, a lack of government incentives, obtaining financing, and lack of knowledge and negative attitudes on the part of the public and stakeholders were pressing financial concerns for the developers. In spite of numerous economic risks taken by the developer, there are many economic benefits to be obtained by communities whose brownfields are redeveloped, including an increased tax base, higher property values, and the attraction of outside investment.

The environmental aspect of brownfield contamination is at the root of most remediation and redevelopment work. Subsurface contamination, a common brownfield issue, can infiltrate into groundwater, which in turn affects the surrounding ecosystem (Murray and Rogers 1999). The presence of these contaminants in the groundwater and subsurface water is also a human health hazard. Traditionally, environmental management has focused on brownfield remediation as the key to successful projects (Lawrence 2000).

Due to the widespread impacts of brownfields, most countries have a range of laws to deal with them. Globally, the US EPA has been a leader in developing appropriate standards and cataloging known brownfields. In Canada, Environment Canada and each of the provincial environmental ministries have followed the lead of the US EPA and often refer to the standards that the United States has developed. Both the EPA and Environment Canada agree that the responsibility of future environmental and health concerns belongs to the property owner who contaminated the land. The 2007 report of the Japanese Ministry of the Environment concerns out of concern that liability could extend to subsequent property owners. This onus of liability on property owners is intended to protect future purchasers. A side effect, however, is that it also encourages some owners, such as fuel companies, not to sell their contaminated sites out of fear of future lawsuits (US EPA 2010a). In Europe, brownfields remain an important issue, though the extent of the problem is uncertain in many nations (Grimski and Ferber 2001).

**Stages in Brownfield Redevelopment**

The Canadian systems design engineers Sean Bernath Walker and Keith Hipel and urban planner Terry Boutilier (2010) have outlined three main stages in the redevelopment of brownfield properties. The first step is the *acquisition* of the property, which usually involves negotiations among a developer, a local government that wishes to entice the developer to take on the risk of a contaminated property, and an owner who may be either the industrial polluter or another organization that owns the property. Subsequently, *remediation* work is carried out to remove or treat contaminants before *redevelopment* of the land for a new use, such as a shopping mall.

**Acquisition**

One of the greatest difficulties for land owners and local governments is the sale of brownfield property to a suitable developer. Often, local governments struggle to find developers who are fully capable of taking on the many expenses involved in the remediation and redevelopment stages and must offer economic incentives to attract investors. A common enticement used in the United States is tax incremental financing by which the local government agrees to provide a break in taxation by increasing the tax level in increments from that of an unoccupied property to a full commercial property over the course of a number of years. The goal is to provide taxation savings to compensate the developer for the cost of remediating the property (US EPA 2010b).

**Remediation**

There are currently a number of options available to developers and engineers to remediate contaminated lands. Before any of these methods are used, developers or their contractors must consider the following: what are the contaminants on the property; what type of soil is on this property; and how does the water table interact with these pollutants? After considering these important issues, the future land use, and the possibility of exposure to contaminants by land users, then the cost of redevelopment and any legal restrictions must be taken into account so that an appropriate remediation plan can be selected. In Canada and the United States, the decision must be between remediating the land to exacting physical standards laid out by bodies such as the EPA or the creation of a site-specific risk assessment, which must be approved by the EPA, or, in Canada, by the provincial environmental ministry, to allow a less stringent remediation while still not harming human and environmental receptors.

Remediation methods can generally be classified into two categories: physical remediation and biological remediation. Physical methods rely on phase changes or physical transport to remove the offending toxin(s). One process, for example, is excavation, which involves the
physical removal of contaminated soil. Generally, excavation is completed using heavy construction equipment, and the removed soil is placed in landfill sites. If the entirety of the contamination is not removed, synthetic blanketlike fabrics called geotextiles can be used to provide a protective barrier, which prevents the migration of particular contaminants. An alternative to removing the soil to a landfill site is to wash the soil; in this process, the soil is excavated, treated off-site, and then returned to the ground. Finally, soil vapor extraction (SVE) can be implemented to remove volatile chemicals from the soil. With SVE, a system of wells and pipes is installed in the soil, and air or carbonated water is pumped into the soil. Through vaporeous and liquid transport, the chemical leaves the aquifer and is collected.

If the contamination is biological, myriad biological tools are available to break down the contaminants. For example, microbial remediation occurs when microbes are used to degrade contaminants into less toxic forms. This technique can be very effective in the treatment of hydrocarbons, polynuclear aromatic hydrocarbons, pesticides, and polychlorinated biphenyls (PCBs). To remediate organic contaminants or toxic metals, phytoremediation is occasionally employed through the planting of chemical-absorbing plants in the contaminated soil, while fungal remediation can be used to degrade specific types of hydrocarbons (Dahn and Reyes 1992; Hollander, Kirkwood, and Gold 2010).

Redevelopment

The redevelopment of brownfield properties is the final, and perhaps most rewarding, stage of the three-step process. At this point, the benefits for the four separate but interacting sectors—social, economic, environmental, and legal—can be fully realized (De Sousa 2003; Greenberg and Lewis 2000; McCarthy 2002). Where a historic building is located on the property, the developer may need to work with the community’s historic preservationist society to obtain appropriate approvals. Famous redevelopments, such as that of the former Atlantic Steel mill in Atlanta, Georgia (Georgia Department of Community Affairs 2011) and the Kaufman shoe factory in Kitchener, Ontario, Canada (Bernath Walker, Boutilier, and Hipel 2010), are examples that show how developers have respected the industrial history of a property while renewing the property at the same time.

Brownfield Decision Making: A Systems Approach

Because brownfield redevelopment involves a range of interconnected physical and social systems elements coupled with deep uncertainty, risk, and conflict, decision making should be carried out within a systems thinking framework. In practice, governance and associated decision making should be integrative to handle interconnections, adaptive to deal with unexpected consequences caused by complex systems interactions and high uncertainty, and participatory to involve the key stakeholders linked to a given brownfield redevelopment problem and its resolution. The purposeful inclusion of all relevant stakeholders in the responsible governance of brownfields ensures that the principle of environmental justice will be followed. A rich range of formal decision models, developed in fields such as systems engineering and operational research, and associated decision support systems (DSSs) (see Hipel, Fang, and Kilgour 2008; Sage 1991) for applying these decision methods to practical problems, are available for addressing tough problems that arise in brownfield redevelopment (see, for example, Hipel et al. 2007; Hipel et al. 2008; Jamshidi 2009; Sage and Biemer 2007; Sage and Rouse 2009).

Brownfield development is complicated by two key characteristics: a high amount of uncertainty and different types of stakeholders. From the very definition of brownfields put forward by the EPA, there is uncertainty in the extent and existence of contamination at most brownfield sites (Greenberg, Lee, and Powers 1998; McCarthy 2002). Additionally, there is uncertainty with respect to remediation and legal costs, future liability and legal risk, as well as the amount of time necessary to undertake the projects. There is even uncertainty as to how many brownfields are present in many communities (Coffin 2003). The implementation of an appropriate decision support tool thus requires considerable research. This means not only researching case studies of similar
developments but also conducting an inventory of the brownfields and contamination in a given region. Such research and the adoption of a brownfield information system, such as a database to track brownfield properties in a given administrative area, are needed for the effective use of a decision support tool. To deal with uncertainty, planners have applied a number of different types of decision support tools to the consideration of brownfield redevelopment. In the United Kingdom, for example, the Environmental Information System for Planners (EISP) is a helpful online resource that allows planners to examine development control decisions by making information such as flooding risk and possible contamination available to non-specialists (Culshaw et al. 2006). Similarly, the EPA operates an online risk management resource for North American brownfield decision makers. Using the tactical information provided by such online resources, a number of strategic analytical tools can be applied to brownfield decision making. To encourage more investment in brownfield redevelopment, the Canadian water quality specialist George Wang and Professors Keith W. Hipel and D. Marc Kilgour (2011) proposed a fuzzy real options methodology to minimize investment risks in the presence of environmental and financial uncertainty. In real options modeling, option pricing models from the financial market are used to price real assets within which the concept of “fuzziness” can be utilized to take into account the accompanying high uncertainty and risk.

Negotiations among stakeholders take place throughout the acquisition, remediation, and redevelopment stages. Effective methods may include using optimization techniques to determine who among a property owner, purchaser, and government body should pay what toward the remediation of a property (Sunderpandian, Frank, and Chalasani 2005). Such strategic problems lend themselves to the application of systems methods for conflict resolution (Fang, Hipel, and Kilgour 1993) that can provide strategic insights and improved brownfield policy solutions (Bernath Walker, Boutilier, and Hipel 2010; Hipel and Bernath Walker 2011).

Brownfields in the Making

Given the costs and complexity related to brownfield redevelopment, there is significant reason to avoid creating new brownfields. In places like the Oil Sands of Alberta, Canada, the in situ and surface mining of bitumen as well as upgrading processes are causing widespread pollution of the soil, water, and air, thereby continuing to create massive brownfields that future generations will inherit (Kunzig 2009; Tar Sands Watch 2011). Throughout the developing world, there is concern over the installation of single-walled tanks for gasoline storage and the creation of the same types of brownfields that occurred in the developed world in the middle of the twentieth century (Taylor et al. 2009; US EPA 2010c). In nations that are developing quickly into global economic powers, such as India and China, there is concern that the growth is coming without environmental awareness and stewardship (Gardner 2007).

Looking Forward

The redevelopment of brownfields is a multidisciplinary practice that must balance social, economic, environmental, and legal elements. Not only do natural environments need to be restored, but so do properties, communities, and human health. Using a wide range of tools and indicators, policy makers must be proactive in preventing the creation of additional brownfield lands and in rehabilitating existing brownfields.

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See also Adaptive Resource Management (ARM); Community Ecology; Dam Removal; Disturbance; Ecological Restoration; Groundwater Management; Hydrology; Microbial Ecosystem Processes; Pollution, Point Source; Pollution, Nonpoint Source; Urban Agriculture; Urban Forestry; Urban Vegetation

Further Reading


A buffer is a protective barrier that can exist in natural or managed systems, such as urban areas or agriculture. In the latter, buffers often serve as a landscape design solution to human-made problems like soil erosion, surface water contamination, habitat fragmentation, and biodiversity loss; buffers have the potential to improve the health of human-dominated ecosystems. Obstacles to the use of buffers are primarily economic.

Buffers serve important functions in protecting our natural resources from the negative impacts of human activities. The term buffer refers to an entity that serves as a protective barrier, reducing or eliminating the flow of undesirable substances. In natural systems, vegetated buffer zones can protect rivers, wetlands, and other sensitive features from natural disturbances such as fires and floods. In the context of managed systems, buffers represent a landscape design solution that has the potential to reduce the impacts of a number of different anthropogenic problems such as soil erosion, surface water contamination, biodiversity loss, and habitat fragmentation.

Agriculture is one type of land use that can have detrimental impacts on our natural resources. Pesticides and fertilizers used in agricultural systems can be transported to rivers, lakes, and oceans, where they contaminate the water and harm aquatic organisms. Intensive soil disturbance from agricultural activities contributes to erosion and depletion of valuable topsoil. The expansion of agriculture across the landscape has resulted in a net loss of biodiversity, as natural habitats rich with a variety of species are replaced by homogeneous cropping systems. Urban areas, which are also managed systems, are not immune to these problems. In cities, lawns serve as sources of pollutants, building construction contributes to soil erosion, and new development threatens diverse ecosystems. Buffers, when designed and located appropriately, have considerable potential to address these problems, offering a solution that could improve the health of human-dominated ecosystems.

Background

Historically, features such as riparian forests, wetlands, and hedgerows were retained and planted in agricultural landscapes, serving as buffers around cultivated fields and pastures. With the intensification of agriculture over the past 150 years, many of these features were removed and replaced with crops. Without noncrop habitats to divide fields and protect water resources, soil erosion from wind and rainfall became an increasing problem. The 1934 Dust Bowl, a great dust storm that occurred across the farmlands of North America, increased public awareness of the problem. In the years following that event, new soil conservation practices were introduced, including vegetated buffer zones within fields, along waterways, and beside roadways. Not until the 1970s, however, was the term buffer used to describe such features. At that time, conservation scientists began to study the effectiveness of buffers in improving water quality.

Today, buffers exist in a wide range of settings, offering a variety of functions depending on their configuration in the landscape and the composition of plant materials. Riparian and wetland buffers consist of perennial vegetation, oftentimes forest communities, located directly along water courses. These habitats can play a critical role in protecting water quality in agricultural and urban landscapes. Wetlands can themselves serve as very effective buffers, treating contaminated water before it enters rivers, lakes, or oceans. Field margins or
hedgerows are located along the borders of crop fields in agricultural areas, and they serve an important function in reducing soil erosion and offering habitat for wildlife. Windbreaks or shelterbelts typically contain trees or shrubs to reduce wind erosion and protect crops, livestock, or homesteads from the harsh weather conditions. Grass filter strips are designed to intercept contaminants from storm-water runoff before it enters a water body. In urban settings, features such as vegetative swales, rain gardens, and constructed wetlands serve as buffers, if they are located between a source of storm-water runoff and a sensitive water body.

Benefits of Buffers

In the context of managed systems, buffers offer many benefits for the environment and society in general. They have the potential to reduce a number of negative impacts on our natural resources that result from current farming practices and urban development. Buffers reduce soil loss resulting from wind and water erosion, because the perennial plants with extensive root systems help to stabilize the soil and infiltrate water. Taller plantings, particularly trees, reduce the wind current that can carry uncovered topsoil. Buffers also protect water supplies by intercepting fertilizers, herbicides, heavy metals, and other contaminants from storm-water runoff from crop fields, residential lawns, or impervious surfaces. The mechanisms for treating storm water can include physical filtration of sediment-bound materials, chemical or biological transformation of materials in soils, and uptake of materials by vegetation. Performance in removing pollutants, therefore, varies substantially depending on the chemical structure and soil binding properties of the material. Research studies have shown buffers to be effective in capturing significant fractions of nitrogen, phosphorus, heavy metals, soil-bound herbicides, and organic materials.

While the primary function of buffers is typically to improve water quality by reducing erosion and intercepting pollutants, buffers offer many other environmental benefits. The perennial vegetation in buffers, often consisting of many different species, increases the biodiversity of flora and fauna, while also offering habitat for wildlife. With riparian vegetation, wildlife and aquatic organisms benefit from a more favorable microclimate (regulation of light and temperature), which provides greater access to food and water. Buffers can also serve as corridors to connect natural habitats and support the dispersal of organisms between fragmented patches.

In addition to the environmental benefits of buffers, they provide a range of other benefits for society. The water-quality benefits of buffers, for example, can improve the safety of drinking water and reduce the degradation of recreational waters. Buffers reduce flooding by infiltrating water and retaining flood water within wetlands. The result is a reduced hazard for people and less damage to built structures following flooding events. Trees and shrubs in buffers can also filter dust and unpleasant odors from the air, including around large livestock facilities. When they work as part of a greenway system, buffers can serve as corridors for wildlife as well as for people. They also offer important visual-quality benefits by greening the space, diversifying the landscape structure, and screening views of undesirable features. In addition to protecting recreational features, buffers can themselves provide recreational opportunities such as hunting, hiking, and bird-watching.

Barriers to Buffer Adoption

Despite the extensive benefits of buffers, many obstacles limit their widespread adoption. First is the cost of lost opportunities from competing land uses, many of which offer greater potential for profit. In rural areas, this is typically the yield from crops that would be grown on the area used for a buffer. Some farmers are tempted to use the buffer area for grazing livestock, but this can reduce the integrity of the buffer by compacting the soil, limiting the growth of vegetation, and allowing nutrient-rich manure to be deposited near sensitive sources (i.e., rivers). In urban areas, the values of competing land uses are even greater, with opportunities for residential or commercial development. This issue is particularly critical in the areas surrounding scenic lakes and rivers, where land values are often relatively high. Even in a floodplain, where development is not practical, nearby residents sometimes oppose the establishment of treed buffers because they obstruct the scenic view.

A second barrier to expanding the adoption of buffers is related to the direct costs of establishing and maintaining them. Some buffer types require expensive earth-moving equipment to grade the site to convey or retain water. For many buffers (particularly those located in riparian zones), native trees, shrubs, and herbaceous vegetation must be purchased to develop the appropriate plant community. Initial establishment may also involve labor and materials to install barriers to protect the young plants from grazing and browsing by wildlife. After the buffer is established, additional costs may be incurred by the maintenance of mowing, weed control, and sediment removal. The cost of establishing and maintaining a buffer may need to be covered by public funds, through subsidies to landowners or some other program.

This leads to the third barrier—the role of the government or nongovernmental agencies in allocating funds...
for buffers, which provide public benefits. In much of North America and Europe, programs have been established to subsidize buffers (along with other conservation practices), particularly in rural areas where agriculture is the dominant land use. Programs such as the Conservation Reserve Program (CRP) in the United States or agri-environmental programs in Europe are designed for this purpose. Agri-environmental programs reward farmers for environmentally friendly management practices such as establishing riparian zones or wetlands, enhancing hedgerows, and conserving areas with high biodiversity. Landowners, however, are often resistant to allowing the government to play a role in the land-use decisions, since the programs often involve multiyear contracts. In many other countries, nongovernmental agencies often play an important role in promoting buffers to protect natural resources by supplying landowners with planning tools, providing labor to help with establishment or maintenance, or even purchasing the land directly to convert to buffers. Even in that situation, landowners (particularly smallholders) may still be skeptical of strategies that appear to allow another organization to control the land-use decisions.

**Landscape Design Considerations**

The obstacles to expanding the use of buffers may be partially addressed through landscape design by focusing on opportunities to optimize buffer performance, while also considering the preferences of stakeholders. Important factors for buffer design include overall size, placement within the landscape, and selection of plant species. Water quality benefits, for example, will only be incurred if the system is designed to convey water through vegetation or retain water long enough to allow treatment. Riparian zones are often considered to be the best location for treating water before it enters a water course, but if the area of land has been constructed with tile drains, the water may bypass the buffer treatment system altogether. If the primary function of the buffer is to filter odor and dust from the air, the vegetation must be located downwind of the source area. In order to serve as corridors, the buffer should be designed to connect natural habitat areas. Overall, the buffer design will depend on the primary functions to be fulfilled.

Not only are the ecological functions important to consider, but cultural and social functions might also be supported by good buffer design. These functions might include recreation, visual quality, education, artistic expression, and historic preservation. Recreational opportunities can be supported by integrating trails or pathways through buffers, and by establishing vegetation that would draw in birds and other wildlife for bird-watching or hunting. Buffers designed to support the visual quality preferences of the landowners, nearby residents, and other stakeholders might be more widely adopted and protected for the long term. Where scenic views are important, for example, the buffers might be designed with openings consisting of short shrubs or herbaceous plants, allowing taller vegetation to frame the views. In agricultural landscapes, farmers and other landowners often prefer buffer designs that demonstrate an ethic of stewardship, with well-managed vegetation that reflects the organization of the cropping systems. Educational, artistic, and historic components might be integrated into the design, particularly in urban areas where it is important to have the support of many user groups to establish and maintain buffers. The cultural functions will depend to a great extent on the local landowners and residents, as well as the context of the site.

The regional context of buffers can play a very critical role in the success of the design, so consideration should be given to the primary environmental issues, competing land uses, and preferences of stakeholders. In tropical regions, for example, deforestation drives a very specific need to reduce erosion and protect water resources, particularly for vulnerable rural communities. There the focus is often on reestablishing stream corridor vegetation. In Africa, buffers have been used to reverse some problems created by land degradation from deforestation and improper use of herbicides, while at the same time trying to optimize systems so they do not compete with production of valuable food resources. In regions where much of the native vegetation was replaced with introduced species, such as parts of New Zealand, the interest in reestablishing native plants along streams and wetlands in urban and urban fringe areas has driven the design and establishment of buffers. Some regions have tied the establishment of buffers directly to locally important endangered species. Buffers have been promoted in the US Pacific Northwest to improve habitat and reduce pesticides that harm salmon, which are a highly valued species—both economically and ecologically. In Europe, hedgerows and other buffer types have been recognized and promoted for the positive impact they have on the aesthetics of the countryside landscape, which impacts agritourism in the region. Around the world, buffers have been used regularly to protect water resources, but their implementation is often more successful when tied to other specific goals that fit with the interests and preferences of the people living there.

**Outlook for the Future**

Several trends are likely to impact the adoption and design of buffers in managed systems in the future. First is the growing interest in developing landscapes to be sustainable
and multifunctional. Buffers should be considered as a standard layer of rural and urban landscape planning. Even the design of individual buffer features can be conceived in a way that supports many more functions and synergizes conservation goals. Tree and shrub species in buffers, for example, might be selected to offer edible fruits and nuts that can be harvested by the landowner or users of the site. Increasingly, the interest in integrating biofuel production in the more sensitive and less productive areas has been considered. This strategy could be appropriate if the biofuel crops are native perennial plants with very low likelihood of invasiveness, and they would not be replacing another important plant community.

A second trend with buffers is to improve their performance (typically related to water quality) using advanced technologies such as specialized filters containing materials that have a high capacity for absorbing pollutants. Recycled steel slag, for example, has been used to remove high levels of phosphorus through subsurface-flow constructed wetlands. As new technologies for water treatment become available, they are likely to be integrated into the design of buffers. Finally, future strategies to increase the adoption of buffers will probably encourage greater participation from a wide range of stakeholders, including coordinated efforts among multiple landowners. Participatory planning approaches have been shown to encourage commitment from stakeholders, increase satisfaction with results, build trust in the process, and create more realistic outcomes. These trends, taken together, could have a significant, positive impact on the contribution buffers make in protecting our natural resources.

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See also Agroecology; Biodiversity; Biological Corridors; Brownfield Redevelopment; Community Ecology; Ecological Restoration; Habitat Fragmentation; Hydrology; Irrigation; Landscape Architecture; Landscape Planning, Large-Scale; Rain Gardens; Stormwater Management; Tree Planting; Viewshed Protection

FURTHER READING


Carrying capacity has been used to assess the limits of a wide variety of things, environments, and systems to convey or sustain other things, organisms, or populations. Four major types of carrying capacity can be distinguished; all but one have proved empirically and theoretically flawed because the embedded assumptions of carrying capacity limit its usefulness to bounded, relatively small-scale systems with high degrees of human control.

The concept of carrying capacity predates and in many ways prefigures the concept of sustainability. It has been used in a wide variety of disciplines and applications, although it is now most strongly associated with issues of global human population. The idea that Earth has a finite ability to support humans, and that exceeding that limit will result in famine or other cataclysms, is at least three hundred years old (Cohen 1996). British political philosopher William Godwin’s estimate of 9 billion, published in 1820, may seem prescient today. The term carrying capacity was not coined until the middle of the nineteenth century, however, and it was not originally conceived in relation to population at all. Rather, it emerged in the context of international shipping and subsequently was applied in a series of other fields—including engineering, range and wildlife management, agriculture and anthropology, and finally biology—before neo-Malthusians took it up in the second half of the twentieth century. An understanding of this history sheds valuable light on the limits of carrying capacity as a tool for evaluating and managing humanity’s impacts on Earth.

Intuitively, carrying capacity is a simple relation or ratio: the quantity of some \( X \) that a given (amount of) \( Y \) can “carry.” The myriad uses of carrying capacity distilled into a single definition probably would be “the maximum or optimal amount of a substance or organism \( X \) that can or should be conveyed or supported by some encompassing thing or environment \( Y \).” But the extraordinary breadth of the concept so defined renders it extremely vague. As the repetitive use of the word or suggests, carrying capacity can be applied to almost any relationship, at almost any scale; it can be a maximum or an optimum, a normative or a positive concept, inductively or deductively derived. Better, then, to examine its historical origins and various uses, which can be organized into four principal types: (1) shipping and engineering, beginning in the 1840s; (2) livestock and game management, beginning in the 1870s; (3) population biology, beginning in the 1950s; and (4) debates about human population and “overpopulation,” also beginning in the 1950s. Carrying capacity continues to be used in all these senses, but in all except the first, it has been forcefully criticized and largely discredited among scholars, often after a lengthy period of enthusiastic use in both research and policy making. Its widespread popular use and continuing traction in public debates stand in sharp contrast to these critiques.

Shipping and Engineering

The earliest use of carrying capacity is the most literal, and it has been partially supplanted by other terms such as payload. It referred first to the amount of cargo that a ship could carry, measured in volume. This measurement served a specific purpose in the context of international trade in the 1840s, when steam propulsion was overtaking the older, wind-powered technology of sailing vessels. Previously, tariffs and duties had been imposed on cargo ships in terms of their “tonnage,” a measure of...
volume descended from casks of wine known as tuns. A ship’s hull was measured to compute its overall volume, crews’ quarters were deducted, and the resulting figure was used to assess levies on all of that ship’s voyages, regardless of the amount of cargo it carried on any particular trip.

Although somewhat imprecise, this method was a reasonably accurate way of calculating the volume of cargo a sailing ship could transport, because the hull was wholly available for cargo. With the rise of steamships, however, the tonnage system appeared faulty, at least to those whose interests lay in the newer technology—notably the British, whose steam-powered merchant marine fleet led the world. Steamships had to devote much of their “tonnage” to coal and fresh water (to generate steam), and to the huge boilers and engines that propelled them and gave them decisive advantages over sailing ships (e.g., speed, power, and independence from the vagaries of the wind). It seemed unfair to pay levies on this portion of a ship’s volume, as it could not be used to transport cargo. Carrying capacity was invented to capture this distinction and provide an alternative basis for tariffs and duties.

Around 1880, carrying capacity began to be used to measure other human constructions, including canals, railroads, pipelines, irrigation systems, hot air balloons, lightning rods, and electrical transmission lines (Sayre 2008). No longer limited to shipping, it served the practical need of engineers and public planners to know how much X a particular Y was designed to carry without exceeding its tolerances. As in the case of shipping, it was possible to determine such limits with reasonable precision and accuracy; they were static, fixed by the design and materials used; and they were ideal—that is, they referred not to the amount of X actually carried by Y at a given point in time, but the amount that could or should be carried. These features—numerical expression, stasis, and idealism—gave carrying capacity its analytical power and have persisted in subsequent uses of the term (Sayre 2008).

Livestock and Game Management

Carrying capacity was transferred to the measurement of living organisms and natural systems beginning in the 1870s: how much X a human or a pack animal could carry; the amount of pollen carried on the legs of bees; the moisture carried by prevailing winds; the floodwaters that a river channel could carry. These were not engineering questions, but they shared the literal sense of something “carrying” another thing from one place to another.

The second type of carrying capacity emerged from a more figurative notion that transposed the earlier subject and object. Livestock, previously a Y that carried an X, became instead an X “carried” in the sense of “supported or sustained by” a new Y: pastures or land. Scientists in Australia and New Zealand appear to have been the first to use carrying capacity in this way, as they struggled to determine how many sheep and cattle these British possessions could reliably produce on their recently settled frontiers. Carrying capacity helped administrators allocate rangelands to as many settlers as possible while simultaneously avoiding overstocking. The idea quickly caught on in the United States, which experienced calamitous episodes of rangeland degradation in the 1890s, especially on the unclaimed public domain and in areas prone to drought. Between 1905 and 1946, the government implemented a system of leases for the vast areas of land the Forest Service and the Bureau of Land Management held, in which carrying capacity served the key role of measuring the number of stock and the amount of time they could be grazed each year in fenced areas known as allotments. These measurements were averages calculated over periods of years, often extrapolated from study sites to much larger areas of similar climate, soils, and vegetation.

The US conservationist Aldo Leopold, who worked for the Forest Service’s Office of Grazing in 1914–1915, extended this use of carrying capacity from livestock to game animals. He formalized the concept in his famous 1933 textbook, Game Management, the founding work of the discipline now known as wildlife management. Leopold understood carrying capacity as an attribute of a piece of land (rather than a particular animal species) and as a function of multiple variables—including vegetation, weather, predation, competition, and disease—that together determined the size of a local wildlife population by affecting reproduction and survival. By identifying the limiting or deficient variable and manipulating it to improve the carrying capacity, the game manager could achieve conservation and optimize game populations for human uses such as hunting and fishing.

Leopold’s ideas influenced wildlife management in the United States and abroad for most of the twentieth century, resulting in many notable successes in sustaining popular species of game and fish, but also many outcomes that are now regretted by conservation biologists, such as the introduction of non-native species and the loss of biodiversity (Botkin 1990).

In both range and wildlife management, scholars in the second half of the twentieth century began to critique carrying capacity, due primarily to practical shortcomings and on-the-ground failures. International development projects aimed at replicating the US model of range leases, fences, and carrying capacities in Africa and other developing world areas routinely failed, in part because fixed carrying capacities, based on averages of rainfall or
forage production, overlooked the large year-to-year variability of many rangelands (Behnke, Scoones, and Kerven 1993). The same problem occurred in wildlife management: if actual habitat conditions varied from place to place and year to year, and wildlife populations both responded and contributed to these changes, then carrying capacity was merely an ephemeral or local descriptor rather than a predictive or prescriptive tool for management. In shifting from engineered to natural systems, carrying capacity lost its static and ideal qualities and therefore much of its coherence and usefulness.

Population Biology

The third type of carrying capacity emerged from laboratory experiments in which scientists observed population growth in carefully controlled environments. Provided with optimal conditions of temperature, food, and so forth, populations of flour beetles and fruit flies grew slowly at first, then accelerated, and then slowed in asymptotic fashion toward a stable upper limit at which births and deaths balanced each other. When graphed, the line had a sigmoid shape, like a stretched-out \( S \). These experiments took place in the 1920s, and the US biologist Raymond Pearl, who helped pioneer the research, also rediscovered the forgotten work of the nineteenth-century Belgian mathematician Pierre-François Verhulst, who had found a similar pattern in human population statistics and had quantified it as “the logistic curve” (Hutchinson 1978).

As population biology grew into a new scientific field, the logistic curve provided scientists with a way to redefine carrying capacity as a core concept that linked research, theory, and application. In his famous textbook, *Fundamentals of Ecology*, the US ecologist Eugene Odum (1953) called Pearl’s and Verhulst’s asymptote “carrying capacity” or, in mathematical language, \( K \). Because scientists observed \( K \) under ideal environmental conditions, they took it as the maximum possible population of an organism, independent of the environment. Odum thus reversed Leopold’s view that carrying capacity was an attribute of particular places or habitats, defining it instead as a fixed attribute of species themselves. In a fixed, ideal environment, one could observe fixed, ideal carrying capacities.

This new carrying capacity enabled major advances in applied and theoretical population biology for two reasons. First, it provided a benchmark or baseline against which to evaluate population dynamics in field settings. Odum noticed that the logistic curve also approximated patterns observed when a new species arrived (or was introduced) in previously unoccupied habitats: sheep in Tasmania, pheasants on Protection Island, Washington, and starlings in the United States. The pattern similarity helped validate the logistic curve empirically, while the difference between values of \( K \) in the lab and the field suggested that actual environments imposed restrictions on population growth, which Odum termed “environmental resistance.” Second, by expressing population growth as an equation, the logistic curve allowed scientists to develop mathematical models of organism-environment interactions for single or multiple species. They could test the models in lab experiments or compare them to field data, informing both management and research.

As in range and wildlife management, carrying capacity in population biology eventually proved faulty. Although the sigmoid curve could indeed be found in field settings, the actual value of \( K \) varied over time and space. Odum had conceded that “one should not use the sigmoid curve to predict the maximum size of future populations of man or organisms unless one is sure that the carrying capacity of the environment will remain largely unchanged during the interval” (1953, 125). This condition is rarely if ever met outside of the lab, however, except over very short time periods and in small or clearly bounded settings such as ponds or islands. It follows that models built on the logistic curve are unlikely to yield robust predictions of actual population dynamics. As the US ecologist Daniel Botkin noted, “[L]ogistic growth has never been observed in nature” (1990, 40), and fifty years of research has found little or no empirical support for the concept of carrying capacity (see also Hutchinson 1978).

Neo-Malthusianism

The fourth type of carrying capacity emerged concurrently with the third, and it drew on many of the same scientific developments. It applied the concept to human populations, however, and at much larger scales—countries, continents, and the world as a whole—with a view to influencing not scholars but policy makers and the public at large. Buttressed by the scientific authority of ecology, this final kind of carrying capacity helped to revive the arguments made famous in T. Robert Malthus’s *Essay on the Principle of Population* (1798).

Carrying capacity had been applied to human populations before. In addition to his scientific work, Raymond Pearl had been active in debates in the 1920s and 1930s about eugenics, birth control, and the specter of overpopulation—although he had not employed the term carrying capacity. And in the 1940s, the British colonial administration of Northern Rhodesia (now Zambia) used soils maps, agricultural data, and population statistics to calculate the carrying capacities of different portions of
the colony for various forms of native farming. It relocated some fifty thousand native Africans on the basis of the results (Allan 1949). The work went on to inform research by anthropologists studying native agricultural practices elsewhere in Africa and beyond.

Ecologists enlarged and popularized carrying capacity as a tool for promoting population control beginning with the US ecologist William Vogt’s popular 1948 book, Road to Survival. Vogt was an ornithologist who spent World War II doing rural reconnaissance for the US government in South and Central America. His book sounded a plea for conservation and development to improve the lives of poor people throughout the world, both for their own sake and as a means to support the United States in the global struggle against communism. He built his arguments around what he called a “bio-equation”: \[ C = B : E \]

in which \( C \) stood for carrying capacity, \( B \) for biotic potential, and \( E \) for environmental resistance (Sayre 2008). He applied the equation to the world’s continents and concluded that all but North America and Antarctica had already exceeded their carrying capacities, as evidenced by poverty, malnutrition, soil erosion, and other forms of environmental degradation. Humanity faced a stark choice: raise the carrying capacity by reducing environmental resistance through conservation and agricultural modernization or risk “the searing downpour of war’s death from the skies” (Vogt 1948, 16).

Vogt’s concept of carrying capacity contained the same flaws as its predecessors’ ideas had. As in Odum’s case, the idea of environmental resistance was tautological, because it purported to explain something that arose necessarily from an ideal, fixed concept of carrying capacity: namely, the disparity between that ideal and actual empirical cases. The very fact that humans could change their environment, and thereby raise (or lower) the carrying capacity, meant that Vogt’s bio-equation could really produce only ephemeral and local inductive conclusions, just as with wildlife. As the US geographer Nathan Sayre (2008, 131) concludes, “If carrying capacity is conceived as static, it is theoretically elegant but empirically vacuous; but if it is conceived as variable, it is theoretically incoherent or at best question-begging.” These weaknesses did not prevent Vogt’s arguments from recurring, in remarkable detail, in the works of subsequent neo-Malthusians such as Garrett Hardin (1968, 1986) and Paul and Anne Ehrlich (1990).

The concept of carrying capacity originated in contexts in which human control can be effectively wielded over discrete objects and bounded systems at small to medium scales such as a ship, a city, or a transportation system. In such settings, a quantified, static, and ideal measurement of limits was both desirable and achievable. As carrying capacity spread to other applications, however, these conditions were difficult or impossible to meet, except in laboratory experiments. Scholars in a wide range of social and environmental sciences concluded long ago that it is fundamentally flawed. The US anthropologist Stephen Brush (1975, 806) summarized the problem: “the principal empirical weakness of the concept of carrying capacity lies in the fact that the theory of homeostasis inherent to the concept is neither testable nor refutable.” Similarly, the famous Anglo-American zoologist G. Evelyn Hutchinson (1978, 21) offered this judgment in 1978: “When the possible value of \( K \) is constantly increasing, Verhulst’s equation loses its value.”

The limits of carrying capacity as a concept have direct relevance to debates about sustainability today. Given its flaws, the question that must be asked is why the concept of carrying capacity has persisted. This is due in part, no doubt, to the concept’s intuitive obviousness: everyone can understand the idea that a ship can carry only so much cargo, or that a pasture can support only so many livestock, and so forth. Also important is the authority various advocates gave carrying capacity along the way, before empirical evidence caught up with early enthusiasm. Finally, agencies of the state embraced and promoted most of the uses of carrying capacity as they sought to measure, regulate, tax, plan, allocate, or otherwise control people, commerce, land, wildlife, and natural resources of various kinds. But such control is elusive when sought over large, complex, and unbounded systems that are poorly understood and difficult or impossible to control. The history of the concept of carrying capacity teaches us that ideal, static, quantitative limits are extremely unlikely to exist in such cases; the same is probably true for sustainability.

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See also Community Ecology; Complexity Theory; Ecosystem Services; Fisheries Management; Global Climate Change; Extreme Episodic Events; Human Ecology; Natural Capital; Population Dynamics

FURTHER READING


A catchment is another term for a watershed or river basin. Catchment management recognizes that the land, water, and ecosystems in the catchment are interconnected and therefore tries to manage these elements as a whole instead of individually. This approach to conserving land and water resources is being used more and more, but an “implementation gap” presents challenges to more widespread adoption.

A river catchment (also known as a river basin or watershed) is the land area that drains into a single river’s system of tributaries. It is the basic unit for studying river systems. A catchment is an open system made up of fluxes (areas where elements flow through the system) and stores (areas where elements are stored in the system). Water and sediment are the main elements that flow through a catchment area, and they in turn pass through a series of stores, such as clouds, soil, vegetation, river channels, and sometimes aquifers. Catchments are complex mosaics where all components of the system are connected to one another, and management of the whole system must consider not only the natural processes that take place, but also how they interact with human social, economic, and political actions.

Integrated Management Considerations

Catchment management attempts to manage a river's whole system at the watershed level in an integrated and holistic way. It is a multi-objective approach to management that takes into account water quantity (i.e., floods and droughts), water quality, sediment transport, and aquatic and terrestrial biodiversity. This comprehensive strategy contrasts with one that addresses and manages each of these areas separately on a short-term, site-specific basis—an approach that leads to fragmented and often counterproductive treatment of resources. Holistic catchment management is a much more sustainable and effective way to deal with the many factors that need to be considered.

Water Quantity

Incidents of flooding have been increasing because of climate change and urban encroachment into floodplain areas. The report *Future Flooding* (UK Office of Science and Technology 2004) states that 4 million people and £200 billion worth of assets in the United Kingdom are at risk in a once-in-a-hundred-years flood event. In the United Kingdom there are several individual policies, which have gone through different phases, directly focused on flood management. In the 1930s fields were drained to aid agricultural intensification (the use of marginal land and cultivation throughout the whole year), and catchment boards were established to fund and coordinate these programs. During the 1970s and 1980s, flood management was addressed with engineering solutions, including channel modification, straightening, and embanking. These programs employed local fixes to address specific problems, and often simply forced flooding to occur in other locations. In more recent decades, the flood-control paradigm has shifted from structural defenses to sustainable flood management (Werritty 2006). Formal policies in the United Kingdom include Making Space for Water, which encourages reconnection of rivers to their floodplains, as well as Learning to Live with Rivers, the Floods Directive, and the River Basin Management Plans, the last of which is required by the European Union (EU) Water Framework Directive. All
of these policies concentrate on flood risk management at the catchment scale.

Water Quality

The main policy for managing water quality in Europe is the European Water Framework Directive, which sets a goal of achieving “good” ecological status (as defined by the Water Framework Proposal) by 2015. River pollution comes from either point sources, such as sewage-treatment works and industrial outfalls, or diffuse (also called nonpoint) sources, such as agricultural fields and urban stormwater runoff. Regulation of point sources is much easier, and over the past couple of decades these have been managed stringently with agreements (e.g., discharge consents from industry) and laws such as the Clean Water Act in the United States. Diffuse pollution, however, remains a widespread problem. Amendments to the Clean Water Act that were passed in 1987 addressed some nonpoint sources (e.g., municipal stormwater runoff), but agricultural runoff is still mostly exempt from regulation, and in general nonpoint source pollution control still lags way behind point-source control.

Sediment Issues

Sediment aggradation and degradation—situations in which the level of a streambed is altered through sediment deposition or sediment erosion, respectively—increase the risk of floods, cause bank erosion, and impact fish and invertebrate habitats negatively. These problems are particularly prevalent in areas downstream of dams, where there are sediment deficits due to sediment being stored in reservoirs. The reach downstream of the Hoover Dam in the United States has degraded by up to 7.5 meters, with up to 120 kilometers being affected (Owens 2005). Sustainable management of sediment transport would correct for sediment that is lost by helping to achieve a level that is natural for the river.

Biodiversity

Catchments, especially riparian areas (areas where land and water meet), provide a diverse habitat for multiple species, both aquatic and terrestrial. Therefore, the entire landscape needs to be managed to maintain this biodiversity. Recent decades have seen steps forward in riparian management. For instance, riparian buffer strips have been established in many watersheds to help slow high runoff, thereby helping to prevent floods and limit the transport of contaminants into the river system. Riparian buffers also provide good habitats for many species. Farmers and other landowners often demand compensation for the land lost to production, however, and so land costs can be a barrier to establishing these areas.

Expansion of Integrated Management

Scientists and policy makers have come to realize that changes at the local level can have much broader consequences, and that different elements of an ecosystem (for example, sediment delivery and water quality) are nested and linked. As a result, management that in the past would have been limited to certain aspects of rivers is now evolving to include landscapes and multiple land uses. An advantage of this kind of holistic management is that a plan intended to mitigate one problem can be designed to produce multiple benefits—as opposed to unintended negative consequences—for other catchment functions.

Holistic management programs were first used in the early part of the twentieth century. The Canadian province of Ontario established catchment-based conservation authorities as early as the 1940s, and currently there are thirty-eight of them, although their effectiveness has been impacted by budget cuts and the overlapping authority of other government organizations. The US government engaged in river-development activities (such as dam building, navigation improvements, and power production) at the catchment level in the 1930s and 1940s as well, but the broader impact of these activities on the river basin was not considered in a long-term, sustainable fashion.

New Zealand began to manage at the catchment scale during the International Hydrological Decade (1964–1975), establishing distinct hydrological regions and benchmark catchments. New Zealand’s program recognized the need to focus management on the source of the problem rather than trying to cope with the impact downstream. In recent decades catchment-scale management has declined as regional councils have taken over the responsibility for oversight. Meanwhile, Australia has adopted “total catchment management” (also known as integrated catchment management), which consists of multiple organizations working together to achieve their separate objectives. This arrangement has successfully addressed some of the problems of fragmented management, such as overlapping authority and duplication of work.

Widespread international support for an integrated approach to water management came in 1992 with the Dublin Principles and Agenda 21, the latter of which encouraged sustainable management of the whole environment and considered stakeholder participation to be a key aspect of successful programs. The World Water
Council and the Global Water Partnership also recommended at that time that governments adopt a more integrated approach to management of both water and land resources.

More recently, the main driver for the adoption of catchment management within the EU has been the Water Framework Directive, which was adopted in 2000. As discussed previously, it requires participating nations to achieve “good” ecological status by 2015. Good ecological status includes the biological, chemical, and hydromorphological (the shape, boundaries, and content of a water body) aspects of the river ecosystem. Its River Basin Management Plan, whose purpose is to identify all pressures on river catchments, will be updated every six years. For example, the 2009 draft Thames River Basin Management Plan identified the need to address the pressures of securing sustainable amounts of water, reducing the impact of the built environment, and addressing point source pollution.

Barriers to Integrated Management

There are several barriers to widespread adoption of integrated catchment management. First of all, there is a mismatch between environmental and political boundaries, with the latter being more important when it comes to catchment management (Witter, van Stokkom, and Hendriksen 2006). This makes the management of large catchments—which consist of multiple administrative units and often more than one country—particularly difficult.

Second, different organizations are often in charge of managing different parts of the catchment system (Lerner and Zheng 2011). In the United Kingdom, management at the catchment level has been the norm since about 1990. The Environment Agency is responsible for authorizing uses of water resources and managing flood risk; different departments within the agency oversee these areas, however, with a lack of coordination and integration among them. A second statutory nondepartmental public body, Natural England, is mainly responsible for the agricultural environment. It oversees agri-environmental stewardship programs, which are intended primarily to conserve biodiversity, enhance landscape character, promote access to the countryside, and protect natural resources by improving water quality and reducing surface runoff. Flood management is only a secondary objective of Natural England’s programs, even though reducing flood risk would benefit most land-management options. There are also additional groups that control other aspects of catchment management, including local governmental councils that are responsible for planning, national parks/trusts that have specific management responsibilities for particular parcels, and water companies that manage water supplies for human use. China has an administrative structure similar to the United Kingdom in that different catchment issues are covered under different laws and managed by different organizations—for example, surface water is managed by the Ministry of Water Resources, while groundwater is managed by the Chinese Geological Survey and the Ministry of Land and Resources.

The third problem is the “implementation gap”—the recognition that integrated catchment management is complex conceptually and therefore difficult to convert into practice (Watson 2004). Social and institutional constraints prevent the full application of current scientific knowledge, and although stakeholder participation is an important aspect of catchment management, it currently falls short. Most management projects are still top-down, piecemeal, and intended to fix specific problems in particular parts of the catchment. Consultation is viewed as important, but it’s time consuming and leads to conflict over balancing the priorities of different groups. Moreover, there is still confusion over exactly what “catchment management” means, and therefore gauging the success of projects is difficult.

Future Outlook for Integrated Management

Recent decades have seen an increase in the adoption of integrated catchment management, but it’s doubtful that it has been achieved fully in any major catchment in the world (Lerner and Zheng 2011). The concept is widely accepted and believed to be the best form of management, but implementation is easier said than done. There is hope for the future however. The EU Water Framework Directive is still in its first adoption cycle, but it seems to be a good legislative tool in achieving holistic catchment management because it puts the catchment unit at the forefront of management and takes a multifunctional approach to management. Technological improvements such as remote sensing, geographical information systems, and computer models are making catchment-scale studies and management easier. There also has been an increase in the number of interdisciplinary studies on catchment issues, which use the knowledge of researchers from different fields to examine these issues from different angles. All of these developments are helping to change the status quo, in which users of the catchment are isolated from those who manage it.

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See also Adaptive Resource Management (ARM); Agroecology; Best Management Practices (BMP); Coastal Management; Dam Removal; Ecological Restoration; Food Webs; Groundwater Management; Hydrology; Irrigation; Landscape Planning, Large-Scale; Pollution, Point Source; Rain Gardens; Stormwater Management; Water Resource Management, Integrated (IWRM)

**FURTHER READING**


Focusing public attention and conservation efforts on large, popular endangered animals—charismatic megafauna—has been seen as a way of obtaining funding for projects and of blazing trails for the conservation of less popular species and their habitats. Detractors argue, however, that this approach diverts resources away from more deserving causes and neglects the vast majority of species that are neither large nor popular.

The term ecosystem management implies a holistic approach to nature conservation—in other words, managing ecosystems as a whole to achieve the goal of sustainable living. One might assume that all life forms—vertebrate and invertebrate animals, plants, fungi, and microorganisms—feature in the management strategies of nature conservation agencies and in government decisions. But the realities of politics and community perceptions often narrow the way public policy issues are approached, and in the area of nature conservation the focus frequently contracts to a few large, well-known animal species (such as the koala shown above in a photograph by Daniel Lunney). What is it about the nature of ecosystem management that makes it subject to public preference for so-called charismatic animals, and does this mean that it is a popularity contest, pandering to public appeal, and what kind of strictures are ecosystem managers under? These are critical governance issues, and if not discussed, they become the elephant in the room.

The phrase charismatic megafauna refers to large, well-known animal species that gain a disproportionate share of the public's attention. The context for this term is that much of the world's wildlife is declining or becoming endangered, and increasing numbers of species are heading for extinction, as is evident from such sources as the United Nations Millennium Ecosystem Assessment's biodiversity report (2005) and the frequent reports on the status of the world's wildlife by the International Union for Conservation of Nature. The growing size of the human population is placing demands on the world's ecosystems, and wildlife is being hit hard. Media and conservation organizations often publicize this message through the examples of a small suite of well-known endangered animals, mostly large mammals—the so-called charismatic megafauna. The question arises: is focusing on charismatic megafauna helpful to wider sustainability efforts?

What Is Charismatic Megafauna?

The Australian Macquarie Dictionary defines charisma (a Greek word, meaning gift) as “the special personal qualities that give an individual authority over a large number of people; ability to influence or impress people.” The same dictionary defines fauna as “the animals of a given region or period, taken collectively.” Fauna commonly refers principally to vertebrates (animals with backbones), although its meaning clearly includes the vast array of insects, mollusks, and other animals without backbones. Because of this slant toward vertebrates, the term charismatic megvertebrate is also commonly used. Zoologists customarily classify vertebrates into five groups: mammals, birds, reptiles, amphibians, and fishes. It follows that megavertebrates are large animals from these groups, such as elephants, whales, eagles, crocodiles, bullfrogs, or sharks.
Big, Popular Mammals

An early reference to charismatic megavertebrates appears in “Saving ‘Charismatic’ Animals,” an article published in the 22 April 1985 issue of Newsweek magazine. It both defines the concept, with examples, and presents a case for its use by one of the world’s leading conservation biologists, Edward O. Wilson. Wilson’s specialty is the study of ants, and thus his support for the concept lends extra weight in those zoological circles that complain that popular vertebrates steal the limelight in the conservation debate.

The latest tactic acknowledges that public support cannot be mobilized to save the snake mite—or thousands of other homely beasts. Instead, within the past year many wildlife conservationists have forged a policy of preserving and promoting “charismatic megavertebrates,” the pandas, tigers, okapis and other glamorous rarities that rivet public sentiment. If this is a rude repudiation of the conservation purist’s all-or-nothing creed, it’s also a deftly plotted political practicality. “There is a sense of mission now, and of encouragement,” says eminent Harvard zoologist Edward O. Wilson. “Our most easily appreciated species can call attention to the plight of our entire ecosystem.” (“Saving ‘Charismatic’ Animals,” 10)

In this application of the strategy, the charismatic megafauna are employed for the conservation benefit of all other species and, in fact, entire ecosystems. Environmental problems, such as habitat loss and fragmentation, pollution, introduced species, and climate change impacts, affect all species. Charismatic animals allow us to publicize these generic sustainability issues and to explore, with popular support, ways of countering their effects, to the benefit of all wildlife that suffer the same problems and utilize the same geographical areas. Furthermore, protecting popular species and their habitats helps us conserve a habitat network across landscapes, to the benefit of all fauna. The conservation zoologist Norman Myers (1996) points out that once large vertebrate species are lost, the opportunities for new large vertebrate species to evolve will also be lost in a planet much modified by human activities.

The Newsweek article names three species, describing them as “glamorous rarities.” The panda is arguably the best-known glamorous rarity, a pin-up charismatic megafauna. In a 1998 editorial in the journal Oryx, Jacqui Morris observes that mammals make up a relatively small proportion of the world’s fauna, yet over the previous decade, half of the papers published in Oryx had mammals as their primary focus. Morris notes that, in the opening paper of that 1998 edition of Oryx, Jeffrey McNeely makes a plea to find new ways of conserving mammals beyond research, survey, and anti-poaching, saying that mammal conservationists need to tackle underlying issues such as habitat destruction, overexploitation, and introduced species. Morris thus points to the tension of highlighting individual species versus addressing root causes.

This theme became the subject of a book by Abigail Entwistle and Nigel Dunstone, Priorities for the Conservation of Mammalian Diversity—Has the Panda Had Its Day? (2000). The book notes that, since recent analyses have shown that about a quarter of all mammal species are threatened with extinction, the conservation movement is moving rapidly away from a traditional “protectionist” approach to nature toward a more integrated view of wildlife and landscape conservation. The British magazine The Economist ran an article on 7 January 2008 entitled “Branding Land: Conservation Marketers Choose Land over Beast,” which makes the point that although conservation organizations have long understood the fund-raising value of charismatic megafauna, the money raised to save these animals often cannot be spent on broader conservation goals. The article discusses an alternative response adopted by the Zoological Society of London, which is to focus on “evolutionarily distinct and globally endangered” (EDGE) species. These species, the article notes, are “rarely cuddly” and may “look quite weird,” but they are often the last representative of an entire animal group. EDGE species—which include the duck-billed platypus, the long-beaked echidna, the aardvark, and the dugong—may not be considered endangered, but it can be argued that because they are rare from an evolutionary standpoint, conservation action should be undertaken before they become endangered.

All Creatures Great and Small

While the public presses for iconic species to be protected, government resources are often narrowly focused on threatened species for research and conservation effort, to the neglect of the far greater number of species that are not listed as threatened. It is important for managers working to maintain biodiversity to have statistically workable numbers to determine the impacts of change, such as from logging, fire, and climate change. The animals that provide the most effective answers are the common species, and often the least charismatic, such as native bush rats. Few people are keen to hear about the ecology of rats, even though they provide insights for conservation of certain ecosystems, such as forests or riparian strips (riverbank
habitats), which rare animals never can. In contrast, an investigation on the remediation of rural lands for koala conservation was given detailed television coverage in Australia on 14 April 2011, when the recovery of the koala population was set back sharply by climate change (Lunney et al. 2012). The native rats would not have gained the coverage, and they do not attract the funding dollars.

The article in The Economist (“Branding Land” 2008) notes the value of so-called flagship areas—that is, entire regions identified as threatened and worthy of protection. For example, Conservation International, based in Arlington, Virginia, has designated a number of flagship areas, including the tropical Andes, the Brazilian Atlantic forests, and Africa’s Cape floristic region. The World Wide Fund for Nature, based in Gland, Switzerland, has also identified what it calls “global ecoregions.” Money raised as a result of designating flagship areas can be used for a variety of conservation projects, not just to protect certain well-known species.

What about conservation of the small animals? The vast majority of animals are invertebrates, and their contribution to ecosystem function overwhelms the contribution of the vertebrates. With a few exceptions, such as the octopus, invertebrates are not megafauna, but many are charismatic. Butterflies are one example; coral reefs are another. Despite a long list of engaging animals, the invertebrates languish in attracting public attention. So incensed was the Australian Museum mollusk expert Winston Ponder that he wrote a paper entitled “Bias and Biodiversity” (1992) and coedited a large book on the biology and conservation of invertebrates entitled The Other 99% (1999). Invertebrates are now commonly referred to as “the other 99 percent.”

Even within the vertebrates, the large charismatic species are an insignificant minority. The two mammal orders with the most species are the rodents and the bats, but finding charismatic rats or bats is at best a matter of individual taste. The beaver qualifies as a charismatic rodent, but most beavers look like the despised rats and mice. Flying foxes are large fruit-eating bats, with wingspans up to a meter when in flight. They are utterly captivating to some people, but they are also problematic to propose as charismatic to a broad public. They induce fear in some people because they are bats, and in their vast camps they are seen by some as pests, particularly so when they raid orchards. The worry that they potentially carry lethal diseases has further tarnished their image. Public ignorance and fear of bats reaches back two thousand years (Lunney and Moon 2011).

Implications

From this overview of charismatic megafauna, the conclusion can be drawn that the term is about capitalizing on the preexisting views of the public about their concept of “fauna,” which animals they like, and where people are willing to assist conservation programs aimed at conserving nature. Does a focus on a few high-profile species reinforce an already constrained agenda for those who work to conserve the diversity of species and the ecosystems of which they are vital elements? It does, but it also facilitates conservation gains. Given the multitude of pressures on ecosystems and their faunal inhabitants through the twentieth century, it is fair to say that all conservation gains are welcome. There has been a phenomenal growth in fauna conservation efforts in the last four decades, and the rate of loss of biodiversity has been slowed in many places. That is a great achievement considering the path that humanity had been on, and a new generation now has the challenge of reversing these losses, fortunately with most of the charismatic megavertebrates still here to enjoy.

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See also Biodiversity; Biodiversity Hotspots; Biogeography; Community Ecology; Complexity Theory; Ecological Restoration; Fencing; Food Webs; Habitat Fragmentation; Hunting; Marine Protected Areas (MPAs); Population Dynamics; Species Reintroduction

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Coastal Management

The conceptual basis for coastal management has evolved from sector-based approaches to conflict reduction to proactive consideration of trade-offs through spatial management and ultimately to coastal ecosystem governance, which advances human objectives with cognizance of limits imposed by natural systems. Enhancing sustainability may occur through reinterpretation of actions under the sector-based management system, through incremental changes to it or, alternatively, through fundamental changes that emphasize ecosystems and the services they provide to advance human welfare.

Coastal ecosystems include land and sea environments as well as the people who inhabit and use them. One way to define the geographic extent of this region is by elevation. Some describe the coastal zone (simply defined as the place where the land meets the sea) as the area ranging from the 200-meter land-elevation contour to its 200-meter-depth equivalent (Crossland et al. 2005). In many areas this zone would extend inland 70–100 kilometers and offshore to or just beyond the shelf break on the continental margin. Furthermore, for research purposes the extent of the zone would be expanded to encompass additional area to consider material flows (water, sediments, nutrients, contaminants) and social systems, which do not neatly fit within the elevation contours. In certain settings, coastal airsheds can be important too. Materials traveling by air settle to land and water areas of the coast, and attention to their sources becomes a part of research or management activity.

This land-sea geographic region has important biological properties (Day et al. 1989). Estuarine habitats show primary production levels that are among the highest on the planet. Estuarine food webs rest on phytoplankton, detritus, and submerged aquatic vegetation and produce very high abundances of benthic organisms per unit area. The fish community present varies in response to cycles of spawning, migration, and feeding in addition to year-to-year changes in absolute abundance. The diet of a single species of fish may include twenty food types, and as a result fish have important impacts on lower trophic levels through predatory pressure. These linked ecological systems are both sensitive to anthropogenic changes and important to human well-being through food and a variety of other services.

The land portion of the coastal zone is a highly desirable place to live and work. In fact, as of 2004, 53 percent of the US population lived in coastal counties, which account for only 17 percent of the land area of the country (Crossett et al. 2004). This means that coastal counties contain three hundred people per square mile, whereas the national average is ninety-eight individuals per square mile. In the New York coastal area, densities run as high as almost thirty-nine thousand people per square mile (Crossett et al. 2004). The coastal zone also includes the social system of its inhabitants, who bring values and expectations as well as an elaborate and growing system of laws, practices, and behaviors that structure how people relate to the natural systems they inhabit.

The political boundaries that are superimposed on this land-sea ecosystem vary by country, and the US system exemplifies the types of jurisdictions that can be established. Coastal counties are parts of states. Towns and municipalities reside within counties, and this nested system is subject to a variety of federal authorities. These jurisdictional boundaries are for the most part completely independent of the natural boundaries, such as watersheds. Nature thus is working with one set of boundaries...
while political jurisdictions operate with another rationale for area delimitation. Coastal management must address this incongruity if it is to be successful. In the United States, the federal Coastal Zone Management Act encourages states to designate a management zone that extends outward from the land-sea boundary 3 miles (or in some instances 3 leagues) offshore. Outside of state waters the federal government manages a 200-mile exclusive economic zone.

This combination of environmental processes—a high and growing density of people, increasing anthropogenic alterations of the coast, and multiple governmental jurisdictions—creates difficulties for the managers of these complex coastal systems and for the larger societies. Coastal management as it is practiced today consists of three distinct approaches (Burroughs 2011).

Sector-based management is managing in response to separate activities (e.g., dredging, waste disposal, sprawl, oil development) that have an impact on others (e.g., recreation, fishing, tourism, aesthetics). This approach attempts to retroactively address the effects of one use on another use.

A second approach, known as spatial management, is to plan by geographic area either on land or at sea. In this process, the planning entity identifies uses that will be compatible in the same area or in areas adjacent to one another. This method looks across uses and seeks to avoid conflict largely by planning the location of activities. By virtue of its proactive approach, spatial management can reduce conflict, delay, and uncertainty as coastal lands and waters are developed.

A third approach, coastal ecosystem governance, adopts proactive planning within the context of both human needs and ecosystem capacity. Coastal ecosystem governance promotes sustainable natural and social systems through diverse management techniques that shape human activity in concert with the limits of the natural system. This third approach provides the strongest foundation for sustaining coastal systems.

Sustaining Coasts through Ecosystem Management

Coastal societies face a continuing conundrum concerning conflicting goals for the land/sea region. The conflict is aptly captured in the 1972 US legislation for coastal management that states both the protection and development of the coast as goals. Fifteen years later, the Brundtland Commission raised additional challenges for coastal managers when it defined sustainable development as the ability of humanity “to ensure that it meets the needs of the present without compromising the ability of future generations to meet their own needs” (WCED 1987). The commission thereby established sustainability over long time periods as a measure on which to determine action. For example, actions in a coastal region are unsustainable if contaminated runoff is allowed to render coastal waters uninhabitable for valued organisms.

The dimensions of coastal management include what is to be sustained (nature, life support, and human community), and what is to be developed (people, economy, and society) as noted by the US National Research Council (NRC 1999). Furthermore, a time horizon and the linkage between sustaining and developing are implicit in whatever solutions are adopted. In this framework, the difficult trade-off for the coast is between nature and economy. The NRC wisely asks whether the link should be nature only, nature mostly, nature but, nature and, or nature or economy; articulating these alternatives highlights the difficult choices to be made. In this context the central challenge of coastal management is determining the values associated with common interests as well as creating and implementing coastal programs that implement the type of sustainability most support. Historical records of coastal environments show this to be difficult.

Lotze et al. (2006) demonstrate that anthropogenic transformation of coastal systems can be tracked over thousands of years by examining species richness and associated indicators. Almost without exception, in the twelve predominantly Atlantic systems examined, the researchers found that important species and habitats (sea grass, wetland) have declined substantially. Nature in these settings has not been sustained as human societies moved from hunter-gatherer, to agricultural, and on to market-colonial periods. Other researchers (Fischer et al. 2007) envision the task of management as returning the biophysical, social, and economic system to a state that is sustainable, which means that human societies and economies fit within the capacity of the Earth to support life through the foreseeable future.

Since governance systems for coastal regions seldom invoke sustainability directly and never provide specific instructions concerning which elements (nature, life support, human community, people, economy, or society) are to be given precedence, creating and implementing management systems remains difficult in the absence of explicit goals and authority.

Competing objectives for collective coastal action may be viewed in terms of ends and means (Burroughs 2011). As pressures on resources increase and societal values shift, the end or objective changes. Those values are quite different from the pre-1970s practices of filling
or draining marshes and dumping wastes in coastal waters in primary pursuit of development goals. In fact the value shifts ultimately resulted in current discussions of ecosystem governance as a means to satisfy new goals.

Managing for long-term sustainability ultimately relies on the ability to create, adopt, and provide effective ways of governing human behavior. Orchestrating these policy elements to achieve sustainable coastal systems, both human and natural, results in positive sustainability trajectories (Burroughs 2012).

Coastal Ecosystem Governance: The Basis

For many, ecosystem management has become a way to reconcile the value conflicts discussed above. Ecosystem principles—properly understood—support sustainability. Ecosystems include people. This move toward ecosystem-based management as a means of refining both the definition and implementation of sustainability for coastal environments is evident in the results of a recent national commission in the United States. The US Commission on Ocean Policy (2004) represented many interests and provided a national perspective. The commission declared the need to manage all components of the ecosystem—including people—collectively. This principle has widespread support but is only sporadically reflected in current governance or policy directives.

Nonetheless, the potential for ecosystem-based management is great, and many have detailed what it would entail (Burroughs 2011). Statements about ecosystem-based management establish a goal of healthy, productive, resilient ecosystems able to provide services people need and want. Institutions with the capacity to manage holistically using the principles of ecosystem health, sustainability, and precaution are required. Such institutions, if present, would be particularly helpful in resolving management of issues related to dead zones because administrators would be empowered to manage agricultural practices with full cognizance of the impacts on marine systems.

As summarized by Richard Burroughs (2011), individual authors emphasize the value of a common goal under which multiple sectors of coastal activity may be managed in an integrated fashion. Furthermore, natural boundaries with management regimes fitted to them (as opposed to political jurisdictions) are important for success. Collaborative planning built on public participation and equity should lead to flexible and adaptive program implementation. Ideally ecosystem-based management will account for cumulative impacts, incorporate precaution, and promote appropriate trade-offs among services. As more experience is gained with these concepts, they will inevitably be refined.

How much change is really required to implement ecosystem-based coastal management successfully? Burroughs (2011) considers three approaches. One approach would be to maintain the conventional management system, characterized by sectoral management, and simply try to improve its techniques to meet the new ecosystem goals. Unfortunately, the record suggests that this strategy would be difficult. In spite of many enhancements over the years, important measures of environmental health such as water quality are at best stable or declining. In fact, the increasing size of the low oxygen zone in the Gulf of Mexico makes clear that orchestration of fragmented government initiatives to control nutrient flow have not been adequate. There is also the matter of intent.

Some purported transformations are mere rhetoric, rather than a substantive change in policy. By simply declaring current practices a form of ecosystem management, officials can avoid making needed changes.

A second approach is to undertake incremental change. This strategy involves changing parts of the existing policy system in small ways and then evaluating the consequences before contemplating additional changes (Lindblom...
Shelf Lands Act instructed government officials to Management Act, and the revised Outer Continental National Environmental Policy Act, the Coastal Zone and the US legal regime reflected this change. In the 1950s, were altered in the 1970s (Juda 1993). Public offshore oil development, originally established in the ocean management in the past. An important test for an incremental solution is whether it has moved forward without overwhelming objections. By muddling through, only part of the goal can be achieved, which means that the process must be repeated endlessly. In an incremental change, officials could utilize ecosystem thinking within the confines of existing legal structures and routines to enhance results. Almost all examples of current ecosystem-based management fall under incremental change.

In fact, incremental change has reshaped coastal and ocean management in the past. The arrangements for offshore oil development, originally established in the 1950s, were altered in the 1970s (Juda 1993). Public attitudes about the values of marine waters had shifted, and the US legal regime reflected this change. The National Environmental Policy Act, the Coastal Zone Management Act, and the revised Outer Continental Shelf Lands Act instructed government officials to consider the environmental consequences of decisions. The growing importance of environmental impacts was confirmed through judicial decisions and, together with changes in the law, altered the collection and use of environmental data (Burroughs 1981). After the transformation the government was increasingly charged with anticipating the detrimental environmental impacts of offshore oil development and taking action to avoid them by suspending part or all of certain developments.

A third possibility is that change becomes revolutionary—that is, both rapid and fundamental. Fundamental change becomes particularly important when the characteristics of the problem demand bold solutions, or when a shift in goals requires action that is beyond the reach of incremental change (Birkland 2005; Cortner and Moote 1999). Responding to the Great Depression, landing a human on the moon, and protecting equal rights required fundamental changes in policy. Coasts and oceans are receiving similar transformative attention by many today. The chair of the Pew Oceans Commission stated that their group found the ocean in crisis and advocated for “a fundamental change in this nation’s posture toward its oceans,” noting that “reforms are essential” (Pew Oceans Commission 2003, i). Just as the early 1970s were a time of rapid and fundamental changes in environmental policy (see the discussion of the Clean Water Act in the section below), ecosystem-based management could inspire similar advances in the twenty-first century.

Coastal Ecosystem Governance: The Practice

What would a fundamental shift to ecosystem-based management look like? Issues to consider include the rationale for such a change, the policy elements involved, and an example of a proposed change of this magnitude.

Rapid and dramatic policy transformation is not unprecedented. For example, the Federal Water Pollution Control Act Amendments of 1972, commonly known as the Clean Water Act, represented a profound shift from reliance on voluntary pollution-control measures to a “command and control” approach. To protect recreation, fisheries, and the environment, the new law set up a permitting system to control discharge of effluent from pipes, with treatment requirements based on technological feasibility. The act specified discharge limits for individual contaminants and created a monitoring and enforcement system to ensure compliance.

When pursuing change of this magnitude, it is important to consider both the principles driving it and the means of implementation. First as Robert Costanza et al. (1998) have noted, stakeholders should be involved in formulating and implementing policies, and those policies should be ecologically sustainable and socially equitable. Second, institutional scales for decision making should match the ecological setting. Third, potentially damaging activities should be approached with caution, and there should be ample opportunity to adapt and improve policies. Finally, Costanza and his colleagues recognized that sustainable governance of oceans rests on full allocation of social and ecological costs and benefits. If markets are used as a means of determining best policies, they will have to be adjusted to reflect full costs.

Changing to ecosystem-based management to enhance sustainability requires a shift in how we conceive the process of reaching decisions. In constitutive decision making, basic allocations of authority and control are made (Lasswell 1971; Clark 2002). The ideal constitutive decision process for ecosystem-based management will give precedence to the common interests that support human and natural systems while producing adequate authority to ensure implementation. The
reliance on stakeholders in the planning process and proposals for adaptive management exemplify constitutive changes.

The central challenge of ecosystem-based management is to consider all factors that affect ecological systems and to manage activities so as to sustain the services provided (Levin and Lubchenco 2008). Ecosystem services are defined as the “conditions and processes through which natural ecosystems sustain and fulfill human life” (Daily 1997, 3). The Millennium Ecosystem Assessment divides ecosystem services into four categories (UNEP 2010, 8). The products or goods obtained from ecosystems are known as provisioning services. In coastal environments they include fisheries, mariculture, genetic diversity, medicines, transportation, minerals, and energy from wind, tides, and waves. Ecosystems also provide regulating services by controlling climate through carbon storage and land cover, water quality through decomposition, and natural hazards through protective features such as marshes and reefs. In addition, ecosystems also provide spiritual enrichment and aesthetic and recreational values, which are known as cultural services. Finally, the indirect and long-term benefits that are the basis for the production of all other ecosystem services are known as supporting services. Water or nutrient cycling and photosynthesis are examples.

To what extent and how might explicit consideration of ecosystem services advance coastal management? A change of that type would reorient the discussion of governance techniques to rely on principles of ecosystem-based management. Several investigators have laid out a conceptual basis for it (Slocombe 1998; Yaffee 1999; Layzer 2008; McLeod & Leslie 2009). The Millennium Ecosystem Assessment (UNEP 2006), among others, proposed that ecosystem services are an effective means to convert concepts into practices. Because governmental programs are not designed to protect and enhance ecosystem services in a direct manner, implementation of this approach will almost certainly require use of new tools. Pertinent tools include markets, tradable pollution permits, government ownership, government regulations, incentive payments, voluntary payments, and other means (Brauman et al. 2007; Ruhl, Kraft, and Lant 2007). Ecosystem services require thought and action based on natural systems, not political jurisdictions. Working at the regional scale will almost certainly lead to better understanding of cumulative impacts and to actions less likely to jeopardize the ecosystem services at stake.

Ecosystem service districts are one way to achieve this new goal (Heal et al. 2001; Lant, Ruhl, and Kraft 2008). The United States has many regional organizations that manage geographic areas for specific purposes. State law or local initiative can create a district to manage human behavior associated with watershed health and services, such as erosion, water supply, or floods. These serve as prototypes for ecosystem service districts, which can be linked in ways that allow their geographic jurisdictions to match ecosystem properties, thus ensuring effective oversight. An ecosystem service district is a governmental entity managing a geographic region and empowered to coordinate, zone, and tax. Hypothetically the ecosystem service district would select the least costly means of providing a service. For example, New York City faced a trade-off between managing its watershed to protect drinking-water quality and installing water treatment. By establishing a prototypical ecosystem service district surrounding the reservoirs, the city could and did benefit from the ability of ecosystem processes to purify water in lieu of technology investments to achieve the same results (Heal et al. 2001). Sorting out the advantages and disadvantages of different proposals becomes more complicated as more services and values are considered, but the objective remains to produce the maximum possible value.

Managing for the production of desired ecosystem services is not without liabilities. Managing ecosystems solely on the basis of the monetary advantages they provide could promote the view that “nature is only worth conserving when it is, or can be made, profitable”
(McCauley 2006, 28). Given this pitfall, proponents must design tools and facilitate policy-making decisions with great care to ensure that the broadest reach of ecosystem functions and biodiversity is protected. Otherwise the initiative may result in minimum gains to selected services and potential harm to others (Daily and Matson 2008). In spite of these issues, an ecosystem-services approach provides a new model for coastal and ocean regions, which may presage a fundamental shift in coastal governance. If effectively implemented, this new approach could provide significant gains in sustainability.

Summary

The coast, a region of land and sea, is home to rapidly increasing human populations. Higher standards of living for coastal citizens frequently result in greater environmental impacts, while at the same time many in the populace increasingly value healthy, natural waters. As a result, coastal management professionals have confronted sustainability issues for four decades or more. In the Coastal Zone Management Act of 1972 administrators are challenged to both protect and develop the coast through the goals statement in the legislation. The 1987 Brundtland Commission report added a time dimension to the challenge and, while seeking sustainability of social and natural systems, noted the importance of managing the present in a way that does not jeopardize the future. More recently, coastal management analysis and action have been informed by ecosystem principles, which provide a means to shape the discussion about protect-develop-sustain within the context of environmental limits and human needs. In coming years coastal management will continue to test the ability of society to create and implement new governance systems matched to the desire for a sustainable future for coasts.

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See also Best Management Practices (BMP); Catchment Management; Ecosystem Services; Edge Effects; Extreme Episodic Events; Food Webs; Global Climate Change; Large Marine Ecosystem (LME) Management and Assessment; Marine Protected Areas (MPAs); Ocean Resource Management

Further Reading

Comanagement of natural resources, by which governments work in partnership with local groups, has been developing since the nineteenth century around the world but did not gain pace until after World War II, initially in western Europe. Since the 1970s, comanagement has been part of a major global shift in natural resource policy making, but implementation challenges continue.

Comanagement is the process of sharing responsibility for the management of natural resources between a government agency (or other formal institution) and local stakeholders, such as rural communities, indigenous peoples, and nongovernmental organizations. Also referred to as community-based conservation, collaborative resource management, joint management, or conservation partnerships, comanagement has been applied to marine and freshwater fisheries, wildlife populations, and other natural resources, as well as to national parks, forest reserves, and many other conservation settings worldwide. Although not without its challenges, comanagement represents one of the most important innovations in natural resource conservation to emerge since the 1980s.

Rise of State-Centered Conservation

Although comanagement is often touted as a novel approach to conservation, the concept actually is not entirely new. Cases in which governments have passed responsibility for managing natural resources to local communities go back to at least the nineteenth century, such as at Lofoten, Norway, where a major cod fishery has been managed by participating fishers with oversight for more than a century.

When the modern conservation movement took shape in the late 1800s, however, most conservationists in North America and western Europe argued that the protection and management of natural resources were the responsibility of the state, employing the knowledge and wisdom of professional managers and scientists trained in forestry, biology, engineering, and other emerging scientific disciplines. As societies industrialized and transportation and economic links improved, the exploitation of forests, wildlife, and water sources advanced on an unprecedented scale, involving powerful national and international economic interests. Conservationists argued persuasively that a strong and progressive central government was the best way to counterbalance the juggernaut of exploitation and achieve conservation goals, be they the preservation of wild lands or the wise management of natural resources. The role of the state was duly codified in legislation authorizing government action to protect wildlife and endangered species, establish forest reserves and national parks, manage fisheries, and control environmental contamination.

Although the scale at which federal and colonial governments established parks and reserves during the late nineteenth and early twentieth centuries was impressive, these accomplishments came with a price. Many units were superimposed over territory that had been farmed, fished, hunted upon, gathered over, grazed, and otherwise called home by indigenous cultures and traditional communities. For instance, the territory that in 1872 became Yellowstone National Park (Wyoming, Montana, and Idaho) was used seasonally by Shoshone Native American bands. Likewise, Yosemite National Park in California, designated in 1890, was inhabited by Miwok peoples who gathered a wide variety of plant and animal resources from the Yosemite environment.

As conservationists strove to preserve nature, they often failed to recognize that many landscapes they perceived as pristine actually resulted from traditional
habitat modification activities such as regular burning, seasonal grazing, or shifting cultivation. Local residents who relied on plants and animals as subsistence resources often were forced to abandon or greatly modify their traditional harvests. Indeed, in many cases they were physically evicted from parks and reserves. In some instances, to protest such treatment, communities purposefully exploited or overharvested animal and plant resources they previously had managed with care. All too often, the centralized model of conservation that took shape in the nations of the Americas and in European colonial empires produced highly polarized relations between local residents and natural resource professionals, each with different perspectives on their interaction with the natural world.

**Changing Assumptions, Changing Policies**

The first major instance in which modern conservation began to chart an alternative course occurred in western Europe when nations such as Great Britain began to establish modern protected area networks in the years after World War II. Working in rural landscapes that had been settled and managed by agrarian populations for centuries, conservation planners realized from the onset that a US-style national parks system would not be feasible. The agencies that they ultimately established to design and manage national parks and reserves, such as Britain’s National Parks Commission (later renamed the Countryside Commission), did not acquire large extensions of land for protection. Rather, from the start they functioned more as regional advisory committees, working closely with local planning authorities inside designated park boundaries and attempting to provide incentives for land uses that would be compatible with landscape conservation goals.

By the 1960s and 1970s a broader consensus had begun to emerge that governments needed to give greater consideration to local people’s concerns when managing natural resources. The more progressive social climate of the time made it possible for disenfranchised indigenous communities, native peoples, and minority cultures to consolidate their fundamental human rights. In legal and political arenas, as well as in the court of public opinion, indigenous peoples began to press conservation and resource management agencies to give them a greater role in managing natural resources of traditional importance, including forest estates, fish stocks, wildlife populations, and public lands with religious, subsistence, or culturally relevant historical value. In the United States, for instance, federal court rulings such as *US v. Washington* (1974) reaffirmed the right of Native American tribes to manage, in conjunction with state agencies, fisheries resources originally granted to state agencies, fisheries resources originally granted to the tribes in treaties signed with the US government.

At about the same time, concern over the economic, social, and environmental costs of centralized, state-sponsored development models began to spur a shift to more democratic grassroots development approaches. This provided an opportunity for rural communities to claim a greater role in managing natural resources upon which they depended and would prove particularly important in developing countries. By the early 1980s conservation and development in Africa, Asia, and Latin America were being discussed as potentially complementary rather than opposing concepts, further emphasizing the importance of addressing local people’s concerns in formal conservation and management.

The claims of local communities were also buoyed by advancements in academic arenas, where a growing body of research in ecological anthropology, ethnobiology, and human geography fueled appreciation of the complexity and sophistication of local and indigenous resource management knowledge and traditions. Rather than mere opportunist and exploiters of natural resources, local people were revealed in many studies to be careful observers and effective stewards of land, fish, forests, and fauna. As early as the 1960s and 1970s, farsighted scientists were drawing on these new views of local capacities to argue for the adoption of comanagement approaches, as the biologists Raymond Dasmann, from the United States, and David Western, from Kenya, did while studying wildlife populations and nature preserves in eastern and southern Africa.

By the end of the 1970s it was clear that a major shift in natural resource policy was taking hold at both national and international levels. In the case of protected areas, a systematic attempt to promote comanagement began in 1979 with the establishment of the Man and the Biosphere Programme within the United Nations Educational, Scientific and Cultural Organization (UNESCO). The term *biosphere reserve* describes a conservation area where a core protected zone (such as a national park) is surrounded by buffer zones and transition areas allowing traditional land use and economic activities, such as ecotourism, that are compatible with conservation goals. The program has promoted the designation of biosphere reserves by national governments worldwide, with more than 560 designations in 110 nations by 2011.

By the 1980s even the venerable US National Park Service (NPS) was embracing more collaborative approaches to park management, through designations of new national parks, national heritage areas, and national trail and river corridors in which the federal government owned little land and management was shared with state
and private partners. At long-established parks, the NPS increasingly began to collaborate with volunteer groups and neighboring communities in order to address management issues, such as development pressures on adjacent lands, that could not be dealt with adequately through more traditional administrative channels alone.

Putting Comanagement into Practice

Amid the lofty rhetoric of collaboration, natural resource professionals have faced an ongoing challenge of translating the ideals of comanagement into a new kind of conservation on the ground. In practice, comanagement can encompass a range of approaches that are likely to vary in how well they achieve meaningful collaboration. At one end of the scale, under what is termed “passive collaboration,” natural resource managers may simply inform local resource users about management decisions or seek their input and opinions prior to implementing management prescriptions. At the other end of the scale, self-determinant collaboration may mean that local residents themselves make and implement management decisions, with government professionals serving in an advisory role to help analyze problems and resolve conflicts.

Comanagement efforts initially tended to proceed in a piecemeal, experimental fashion, often beginning at the initiative of dynamic individuals—perceptive local leaders as well as innovative resource management professionals—who advocated collaborative management of natural resources particularly important for local residents. Comanagement of state-owned forests in parts of India, for instance, was spurred initially by concerned villagers and later expanded with the support of professional foresters.

Nongovernmental organizations (NGOs) have also played a prominent role in sparking the implementation of comanagement, particularly in developing countries. In 1985 a leading conservation NGO, the World Wildlife Fund, launched a program called “Wildlands and Human Needs,” which sponsored model comanagement projects in Latin America, Asia, and Africa. Another major conservation NGO, the Nature Conservancy, shifted its emphasis in the early 1990s from establishing strict nature preserves to one of protecting larger “bioreserves,” where the organization helps bring together government actors, local residents, and private landowners to protect the biological wealth of native ecological landscapes.

Natural resource agencies in the United States, Australia, Brazil, Canada, India, and other nations have had mandates to implement comanagement since the 1970s and 1980s. Initially, however, most were reluctant to share decision-making authority that previously had been theirs alone and often resorted to court appeals and bureaucratic foot dragging in an attempt to delay implementation. Over time more progressive administrators began to recognize that comanagement offers the possibility of stretching limited agency budgets further and of helping satisfy public demands for reforms and change. By the 1990s natural resource agencies were moving much more rapidly on implementing comanagement agreements for fisheries, forest products, wildlife populations, and public lands. Nearshore fisheries have proven to be particularly suited to the adoption of comanagement arrangements, with the most recent global analysis identifying 130 comanaged fisheries in forty-four countries. One of the largest of these is a lucrative fishery for Chilean abalone that employs twenty thousand fishers in 700 comanaged areas along 2,500 miles of Chile’s coastline.

Even with clear incentives, many government agencies have had difficulty accepting local communities as equal partners in natural resource management, particularly when indigenous communities or minority cultures are involved. In New Zealand, Parliament passed the Conservation Act in 1987, authorizing the government to pursue collaborative arrangements with native Maori communities for managing wildlife populations and protected areas, but implementation has taken time to develop. Cultural differences play a role. For instance, certain Maori concepts of resource stewardship such as those invoked by the term kaitiakitanga—an intergenerational responsibility to safeguard, protect, and shelter—are not easily translated into the scientific language of ecology that natural resource professionals are accustomed to using. A lack of prior models for collaboration also slowed the transition to comanagement, along with inadequate funding of new institutions on the part of the government and objections by more preservation-oriented conservation activists.
Even for the countries and institutions that have gone the furthest in embracing comanagement approaches for natural resources, positive results have not been automatic. The integration of traditional management practices for flora and fauna populations is no guarantee that every group or community will be effective stewards in all situations, particularly when cultural belief systems and subsistence patterns are undergoing rapid flux. Inequities or corruption in governance systems can overwhelm even the best intentions at collaborative management, as can social or ethnic divisions within local communities.

In the case of the Dzanga-Shaanga forest region of the Central African Republic, conservation groups have struggled to achieve workable comanagement regimes for many years. The region is populated by a variety of ethnic groups, many of whom have emigrated from other parts of the country, and communal management traditions for forest resources are not well developed. While ecotourism opportunities that bring greater benefits to local residents are now present on a limited scale, it has been a challenge for conservationists to offer local residents incentives for conservation that are equivalent to the profits promised by organized wildlife poaching and logging companies active over the years in the region.

**Future Prospects**

The role of local communities in resource conservation and their capacity to enhance sustainable management approaches are now unquestioned, but examples of successful comanagement arrangements still remain elusive in many settings. Perhaps the single most important factor in continuing to refine and improve comanagement regimes is the commitment of federal and state governments to continue supporting partnerships and collaboration with funding and resources. Comanagement is not a panacea by itself, but it remains the most viable approach available to governments, communities, and organizations alike for meeting twenty-first century conservation challenges.

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**See also** Administrative Law; Best Management Practices (BMP); Biogeography; Ecosystem Services; Fire Management; Fisheries Management; Forest Management; Human Ecology; Indigenous Peoples and Traditional Knowledge; Large Marine Ecosystem (LME) Management and Assessment; Natural Capital; Ocean Resource Management; Wilderness Areas

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**FURTHER READING**


Community Ecology

In an ecological community, species interact in often complex and subtle ways, depending on intricately balanced food webs and energy cycles that maintain the ecosystem. The study of community ecology—examining the distribution, abundance, and diversity of organisms in an area—provides understanding that people can use to prudently manage their own interactions with other species.

An ecological community is generally characterized as a group of species of interest interacting in a particular geographic area. Community ecology is thus the study of species interactions within a particular area. One can watch any nature documentary and get some idea of the complexity of the natural world. The visceral drop of blood on the face of a top predator that has recently killed a grazing herbivore in the grassland savannas of Africa, the parasitic microbial infection that causes ants to climb to the top of plants so they become more visible and likely to be eaten by birds, or the mutualisms between plants and root fungi that allow plants to access soil nutrients via the extended fungal network in exchange for sugars—these dramatic images highlight the species interactions within and across environments that make up community ecology.

Communities can be simple, composed of two or three different species, or complex, composed of many hundreds or thousands of species that interact in dynamic food webs. (Terms in italics are defined in the accompanying sidebar.) Community ecology is largely focused on how species interactions influence the distribution, abundance, and diversity of organisms within a location across scales of space and time.

Understanding community ecology is important to the human population because ecosystems provide services, like a breathable atmosphere, food, water, material for living, and an environment in which to live—services that are mediated by species interactions. For example, soil fertility is related to the interaction of plants and soil communities. Plants contribute organic material in the form of dead leaves that fall to the ground in autumn or roots that grow and die and remain in the soil. This organic matter is fragmented into smaller pieces by soil arthropods and colonized by soil fungal and bacterial communities; the organic nitrogen from the leaf or root is converted to an inorganic form of nitrogen by the microbial communities through the process of mineralization; the inorganic form of nitrogen in the soil can then be reused by the plants. Thus the ecosystem process of nitrogen mineralization is related to aboveground and belowground interactions within the community of plants and a diverse group of soil organisms (Schweitzer et al. 2004).

Species Interactions

Species interactions occur in two major ways: (1) direct interactions, where the effects of one species are felt immediately by a second species, or (2) indirect interactions, where the effects of one species on another are determined by a third species. Species interactions may result in trophic interactions through patterns of herbivory and predation or may vary along gradients of mutualism to parasitism or competition to facilitation. For example, in an experimental setting, the evolutionary ecologist Joseph Bailey (2011) demonstrated strong trophic interactions among a common forest tree species, an herbivore, and avian predators. He found that leaf size in cottonwood trees was related to the abundance of a gall-producing insect herbivore—an example of herbivory and a case of
THE LANGUAGE OF COMMUNITY ECOLOGY

The following is a list of terms commonly used in the field of community ecology. Not all of them are used in this article.

**Alpha diversity**—The number of species found in a small local area.

**Beta diversity**—The differences in species among small local areas.

**Biodiversity**—Usually refers to the total number of species found in an area, but may also include genetic diversity within species in an area.

**Community**—A group of species found in a particular space and time. While some ecologists reserve the term for species that interact with one another (Whittaker 1975, Price 1984), others simply define a community as the collection of species found in an area of interest (Emlen 1977).

**Competition**—An interaction between organisms sharing similar resource requirements that negatively affects the growth, reproduction, or survival of one or both organisms. Competition may result from a direct interaction where one organism actively prevents another from accessing resources (interference competition) or from an indirect interaction whereby one organism depletes the resources used by the other organism (exploitative competition).

**Detritivory**—Consumption of nonliving material (plant or animal) to obtain resources.

**Facilitation**—An interaction between species that positively affects one organism (but not both). For example, many legumes have nitrogen-fixing bacteria (*Rhizobium*) living in their roots that allow them to colonize nutrient-poor habitats. Once legumes are established, they increase nutrient availability in the soil, which positively impacts the growth of new plants that do not host such terra-forming bacteria. The new plants, however, do not positively affect the legumes in turn.

**Ecosystem service**—A service provided by a natural ecosystem that benefits humans, such as the pollination of crops, water and air purification, and protection from floods by wetlands.

**Food webs**—Connections among species in a community that describe the way in which energy flows through a system. As a variety of resources and consumers interact with one another, energy moves through a system from primary producers (plants and algae), to herbivores that feed on the plants or algae, to mesopredators that feed on herbivores, to top predators that feed on other predators.

**Gamma diversity**—The total number species found in a large area containing several small local areas.

**Herbivory**—Consumption of living plants to obtain resources.

**Mutualism**—An interaction between species where both species benefit from the relationship. For example, in the legume-*Rhizobium* interaction, *Rhizobium* bacteria live in the legume’s roots and supply the plant with nitrogen; in return the legume supplies the *Rhizobium* with carbon resources.

**Parasitism**—Partial consumption of a living organism by another living organism. Unlike predators, parasites usually do not kill their host outright but live in intimate association with their host (often inside a single host), feeding in a way that keeps their host alive and provides them with a constant source of food.

**Predation**—Consumption of a living organism by another living organism in order to obtain resources. Carnivores consume other animals, while omnivores consume a mixture of prey (e.g., herbivores and carnivores, or plants and herbivores). Herbivory is a type of predation because herbivores consume living plants.

**Trophic interactions**—Interactions among species that acquire energy and resources in different ways. For example, grasses acquire energy from sunlight, whereas deer acquire energy by feeding on the grass. While grasses and deer obtain energy in different ways, they interact with one another when the deer feed on the grass.
a direct effect of more gall insects resulting in larger leaves. The abundance of this herbivore was positively correlated with the foraging behavior of avian predators—an example of predation and of trophic interactions, and a direct effect of increased bird foraging when gall abundance was high. Moreover, another study has indicated that foraging by avian predators on arthropods positively affected cottonwood tree growth (Bridgeland et al. 2010), resulting in an indirect effect for the trees. The direct effect of birds eating galls indirectly benefitted the trees by reducing herbivory. These interactions among a plant, herbivore, and predator are a classic example of trophic interactions where energy from the sun is captured by plants as carbon; this carbon is consumed by herbivores; herbivores are consumed by predators; and thus energy flows through an ecosystem.

In a classic example of parasitism and mutualism, the US ecologist Nancy Collins Johnson and colleagues J. H. Graham and F. A. Smith (1997) showed that there were beneficial effects of soil fungi on the reproduction of grasses when the grasses were grown in unmanaged soil and received normal light, suggesting that soil fungi help plants acquire nutrients to grow. When soils were fertilized, however, and the plants no longer needed help acquiring nutrients, the mutualistic benefits disappeared, and the soil fungi became parasitic and negatively affected plant fitness. This example shows how species interactions can shift from parasitism to mutualism, or vice versa, as interactions with their hosts or environment change. While we can characterize species interactions in many ways, community ecology is primarily concerned with how these interactions influence patterns of biodiversity on the landscape.

Biodiversity

Measurements of biodiversity within communities include alpha, beta, and gamma diversity. Alpha diversity is the number of different species within a local area; for example, how many species of birds are found in a local city park. Beta diversity is the difference in alpha diversities of local areas; for example, are the same species of birds found in each of several city parks, or are different bird species found in each different park (greater beta diversity being when more different species are found in the different parks)? Gamma diversity is the total number of species in a larger region; in this example, the total bird species in all of the parks within the city. Understanding patterns of biodiversity on the landscape provides important fundamental information that can reflect on the ability of an ecosystem to (1) respond to environmental perturbation (e.g., a hurricane), (2) resist invasion by pest species (e.g., the invasive kudzu vine, fire ants, brown rats), or (3) provide services such as soil fertility, pollination, and products. For example, when biodiversity is high there are more species to provide “backup” for different ecosystem services, similar to backup systems in a factory (Naeem 1998). When one species is no longer able to provide an ecosystem service, other species may take over that function.

Understanding the factors that lead to the amazing diversity of living creatures we find on our planet is a worthy goal in itself, but some researchers have found that such diversity can also affect the sustainability of ecosystem function and services. In 1994, the US ecologists David Tilman and John Dowling were conducting experiments in a grassland habitat where they planted plots with either a single species of plant (a monoculture) or multiple species of plants (a polyculture). After a severe drought, they noticed that plants growing in the polyculture plots were more resistant to the drought and also recovered more quickly. There are at least two explanations for the pattern Tilman and Dowling found. First, some species are so successful at obtaining resources, growing, and reproducing that they become the most abundant (or dominant) species in a habitat. The odds of finding a hardy, dominant species would increase in a polyculture plot, relative to a monoculture plot. Second, different species acquire resources in different ways, creating their own niche in the ecosystem. Niche differentiation could also explain the patterns found in Tilman and Dowling’s plots, as polyculture plots would have more species acquiring resources each in their own way, which would decrease competition among plants, relative to monoculture plots. Recent studies have also found that genetic differences among plants of the same species can affect plant growth and productivity, soil fertility, associated species interactions, and also increase resistance to severe weather conditions. One study found, for example, that eelgrass plants growing in
genetically dissimilar groups were more productive and resistant to spikes in water temperatures and recovered more quickly after temperatures declined (Reusch et al. 2005). Species diversity and even genetic diversity in a community can therefore have important consequences for many aspects of ecosystems and the services they provide, including their ability to respond to disturbances such as climatic change.

Application

Community ecology—studying how species interact—gives us insight into how humans interact with other humans and other species. From a practical standpoint, understanding how species interact can improve human activities such as agriculture and economics, or indicate the importance of natural habitats such as forests and wetlands. For example, a study of Costa Rican coffee farming demonstrated a clear economic incentive for coffee farmers to preserve natural forests in order to protect a fundamental component of community ecology—species interaction. Because native bees that pollinate the coffee plants could only persist and pollinate the plants within a small range of preserved forest patches, those plants grown far away from forest fragments had less value to farmers. Approximately 150 hectares of natural forest provided US$60,000 per year in economic benefits to the farmers (Ricketts et al. 2004).

An understanding of community ecology also tells us that conserving species interactions and natural habitats is likely to be more effective than focusing on one species. If the survival of an individual species depends on the presence of another species, then no amount of conservation effort directed at the focal species will be successful unless the other species is also preserved. Community ecology is of fundamental importance to our understanding of how species interact, the effects of species interactions for whole ecosystems (interactions of communities with their environment), and the consequences of these interactions for sustainability and management of ecosystem services on which humans rely (Millennium Ecosystem Assessment 2005).

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See also: Biodiversity; Biodiversity Hotspots; Biogeography; Buffers; Complexity Theory; Ecosystem Services; Edge Effects; Food Webs; Human Ecology; Indicator Species; Invasive Species; Keystone Species; Microbial Ecosystem Processes; Mutualism; Nutrient and Biogeochemical Cycling; Outbreak Species; Plant-Animal Interactions; Population Dynamics; Refugia; Regime Shifts

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Complexity theory deals with systems that exhibit complex behaviors, such as nonlinear responses, self-organization, sophisticated information processing, and learning. Complex systems include social-ecological systems (linked systems of people and nature) such as those associated with agriculture, fisheries, and forestry. These systems may demonstrate rapid and potentially irreversible shifts between states. Understanding and managing complex dynamics in social-ecological systems are of fundamental importance for our long-term future.

Complexity theory consists of a set of general principles and concepts that relate to the definition, analysis, and prediction of the structure and behaviors of complex systems (Simon 1962; Simon 1977; Gell-Mann 1992). It is particularly focused on phenomena that are difficult for science to explain and predict by classical methods. Examples of topics of interest in complexity theory include chaotic behavior, learning and adaptation, information processing, the structure and function of networks, self-organization, nonlinear relationships between cause and effect, and group decision-making processes (Mitchell 2009; Norberg and Cumming 2008; Holland 1992; von Neumann and Burks 1966).

Complex systems are defined, somewhat circularly, as systems that exhibit complex behaviors or dynamics. Complex behaviors include, but are not limited to, nonlinear relationships between cause and effect (a small input produces a large output, as in the butterfly effect, or vice versa); the action of feedback loops, which can regulate or amplify trends (such as sweating as a regulating response to overheating, or panic buying in response to fears of scarcity); the potential for alternate system states that are maintained by different regimes (sustained, long-term changes in system controls and functions); and the ability to process information and respond to it.

Although complex systems are often built from simple units, they are irreducibly complex—that is, they cannot be fully understood by reduction. For example, a single ant exhibits relatively simple, predictable behaviors, but an ant colony is capable of far more sophisticated decision making. According to the Canadian ecologist C. S. Holling’s “rule of hand” (Holling 2001), systems of people and nature that exhibit truly complex behaviors generally have a hierarchical arrangement (Allen and Starr 1982) with at least three levels and often include variables that act at three or more different rates.

Nearly all systems exist in a context or environment that structures and influences their dynamics and interactions. Complex systems are systems precisely because they show the properties of cohesion (the pieces hang together) and spatiotemporal continuity (Cumming and Collier 2005). The primary challenge of defining a complex system is determining boundary conditions. Even apparently well-bounded complex systems, such as individual people, can have fuzzy boundaries. For example, the human stomach is effectively an external chamber inhabited by vast numbers of free-living bacteria; the case for its inclusion as part of a human organism is strong, but not without question.

There are many different kinds of complex systems, ranging from avalanches of sand grains to the global climate system to communities of carbon-based life forms. In natural resource management, researchers are particularly interested in complex adaptive systems, which are capable of responding to environmental change and of permanently modifying their behaviors and/or internal
structure. Adaptation in lineages of organisms arises over evolutionary time from the action of environmental selection on phenotypic (body form) diversity. Adaptation in complex systems can also occur deliberately over shorter time periods. In human societies, for example, responses to climate change may include active adaptation (taking deliberate steps to reduce potential impacts of sea-level rise on low-lying towns) or passive adaptation (encouraging a diversity of solutions and adopting those that are most successful).

The first analyses of complexity in ecological contexts dealt with the behavior and organization of relatively small, discrete systems, such as ant societies, the human brain, swarms of bees or flocks of birds, and the immune system (Mitchell 2009). These analyses were heavily influenced by models of systems, such as Conway’s game of life (Gardner 1970), which promised to generate complex behaviors from sets of simple rules. The early focus on understanding the minimal elements of complexity provided useful theoretical insights and computational tools, as well as a starting point for ecologists with interests in complex systems, but this type of analysis has not yet been compellingly linked to many real-world problems in ecology or ecosystem management (although some elements of the original approach are starting to become more influential through advances in multiagent modeling).

In recent years, research on complexity in natural resource management has focused less on understanding complexity itself and more on understanding the properties of social-ecological systems, which are linked (complex, adaptive) systems of people and nature (Berkes, Colding, and Folke 2003). One of the most relevant concepts in this rapidly expanding body of knowledge is that of regime shifts, and particularly the idea that a system can quite suddenly cross a threshold beyond which it undergoes rapid, fundamental, and possibly irreversible change (Scheffer 2009). A social-ecological system can be conceptualized as a ball that moves along a three-dimensional surface (state space) that is in turn defined by the range of possible combinations of variables. Most complex systems exhibit some local stability, with the ball (system) being trapped in a localized set of conditions, the basin (attractor). For example, if the ball represents the climate system, exact rainfall patterns (the ball’s trajectory) differ each year, but there is a level of predictability (the ball remains in the cup) between years. If a force is applied to the system, it may move far enough to find another local attractor, depending on the size of its basin of attraction. In the case of the climate, warming due to the greenhouse effect may permanently alter precipitation patterns in both space and time. A shift to a new local attractor is termed a regime shift; the system crosses a threshold as it changes.

Examples of thresholds and regime shifts are increasingly being documented in a wide range of social-ecological systems. Some, such as the possibility that Amazonian rain forests may tip into a drier state (as did the Sahel) or the potential that major oceanic currents may be reversed, carry profound implications for global biophysical systems and biodiversity (Rial et al. 2004; Scheffer 2009). Complexity theory is also contributing to a range of related topics of high importance for the environment, such as our understanding of social and ecological networks (Bodin, Crona, and Ernstson 2006), the influence of spatial heterogeneity on system dynamics (Cumming 2011), and the rules that best facilitate sustainable management in common property systems (Ostrom 2007).

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See also Adaptive Resource Management (ARM); Biogeography; Boundary Ecotones; Disturbance; Ecological Forecasting; Edge Effects; Food Webs; Human Ecology; Large Marine Ecosystem (LME) Management and Assessment; Mutualism; Plant-Animal Interactions; Regime Shifts; Resilience

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Dams have played important roles in human civilization, providing a variety of services including stable water supply, flood control, and hydroelectricity. In turn, however, dams have altered flowing water ecosystems, and many dams that are no longer needed are now being removed to restore the natural flow and functioning of stream ecosystems. Globally, dams are being individually evaluated in terms of whether they are contributing to or detracting from a sustainable environment and economy.

Humans have been building dams for over five thousand years. Most of the world’s rivers have been modified through dams and other diversion measures for reasons including recreation, hydropower, irrigation, and water supply. As scientists have become aware of the often far-reaching consequences of altering waterways and landscapes, the removal of existing dams has become a consideration for the sustainability of water resources and aquatic life. (See table 1.)

Although the exact number of dams in the world is not known, according to the World Commission on Dams (WCD) there are about forty-five thousand dams exceeding 15 meters in height with a reservoir capacity greater than 3 million cubic meters (WCD 2000). The number of smaller dams (less than 15 meters in height) is not well known, and worldwide estimates are in the millions (Smith 1971; WCD 2000). For example, in the United States, the National Inventory on Dams managed by the US Army Corps of Engineers has approximately seventy-nine thousand dams on record (Gleick et al. 2009), though it is estimated there are over 2 million when dams less than 2 meters in height are included (Shuman 1995; Graf 1999). Many of these dams are on private property, and many are no longer used for the original intended purpose (e.g., to run a mill).

Historically, building dams has been a means to increase social, cultural, and economic development by harnessing the power of water and converting it to usable energy forms (e.g., to cut lumber or grind wheat), by providing a stable water supply for consumption and

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Table 1. Reasons for Building and Removing Dams

Source: Compiled from Nilsson et al. (2005); McCully (1996); Gleick et al. (2009); American Rivers (2011).

Many dams are multipurpose (e.g., they provide hydropower as well as recreation and water supply); they may also be removed for several reasons (e.g., to restore fish passage or to eliminate liability).
irrigation, and by controlling flooding. In turn, many of the impounded areas (the lakes or reservoirs created) became areas of recreation and focal points in communities. In the early to mid 1900s, dam building occurred in many nations, with some of the largest dams being built from the 1930s to the 1960s (McCully 1996). For example, Hoover Dam, on the border between the US states of Arizona and Nevada, was built in the 1930s to tame the floods of the Colorado River but also served to create a stable water supply (Lake Mead) and generate electricity. In the southern United States beginning in the 1930s, several dams were built on the Tennessee River system in order to control floods, promote navigation, provide national defense, and generate electricity by the Tennessee Valley Authority.

Despite the costs of building (tens of billions of US dollars), the numbers of large dams is expected to increase in the next few decades, as several countries and regions of the world are developing large-scale dam projects (Ross 2011). The most notable of these recent projects is the Three Gorges Dam on the Yangzi (Chang) River in China. Three Gorges is currently operational, and other dams as well as diversion projects are also planned on the Yangzi and its tributaries. Chile, India, Ethiopia, Southeast Asia, and several other countries and regions are developing projects that are designed to generate thousands of gigawatt hours (GWh, equal to 1 billion watt hours, or 3.6 terajoules) of electrical energy annually as well as to provide flood control, improve navigation, and provide a stable source of water. While these countries are experiencing a renaissance in dam building, other countries, such as the United States and Spain, have begun decommissioning and removing dams.

Over the three decades since 1990, more than 900 dams have been purposely removed from stream systems around the world for a variety of reasons, including the restoration of fish passage and other ecological issues, maintenance costs, liability, and safety issues. (See table 1.) Globally, dam removal is receiving increased attention as a management and restoration tool (Lejon, Malm Renöfält, and Nilsson 2009). As of 2010, 888 dams had been removed from river systems in the United States (American Rivers 2011), and in Europe, Spain has removed over 50 dams and has plans to remove over 100 more (Brufao 2008). France and Sweden have begun employing dam removal as well (Lejon, Malm Renöfält, and Nilsson 2009). It should be noted that at present there is no master list of global dam removals, so actual numbers of removals around the world are not known and are underestimated. At present the United States is the world leader in dams that have been purposely removed.

**Why Choose Dam Removal?**

Consideration of whether to remove a dam must include examination from a number of viewpoints, including the ecology and environment of the waterway and surrounding area, maintenance issues, liability and legal concerns, and socioeconomic drawbacks and benefits.

**Environmental Perspectives**

Environmentally, the changes that dams make to stream systems can be detrimental. Dams degrade rivers by altering the natural flows of water and sediment; by changing the quality of the water system (e.g., increasing nutrients such as nitrogen and phosphorus that encourage algae growth, and toxic contaminants such as polychlorinated biphenyls and heavy metals); by flooding and altering terrestrial systems; by changing the temperatures of the water; by limiting the movements of biotic communities; and by changing the biota from a lotic (flowing water) system to that of a lentic (lake) system.

The conversion from lotic to lentic system and the loss of hydrologic connectivity can have far-reaching effects, including increased production of greenhouse gases such as methane and nitrous oxide, the loss of biotic communities such as migratory fishes whose remnants fertilize the surrounding floodplains, and the loss of natural flooding regimes that replenish the floodplains with nutrients and sediment. For humans, these changes manifest as the loss of many ecosystem services, such as the availability of water on floodplains that are farmed, important sources of protein (e.g., migratory fishes), native flora and fauna, and aesthetics (e.g., geologic formations or bends in rivers).

The removal of a dam has the potential to reverse many of these alterations, though it must be understood that ecosystems are dynamic, and it is likely that over the life of a dam there have been changes in land use, habitat availability, and fishing pressures (Bushaw-Newton, Ashley, and Velinsky 2005; see figure 2). For example, in the case of anadromous fish (those that ascend rivers from the sea for breeding) such as salmon, alewife, sea lamprey, and American shad, a return to historical numbers may not be achievable due to the current size of the populations, but opening the passageways will at least create the potential for restoration. The removal of Edwards Dam on the Kennebec River in the US state of Maine in 1999 opened almost 28 kilometers of historic spawning grounds, and migratory fish species have been returning (Maine Department of Marine Resources 2011). The removal of the Saint-Etienne de Vigan dam on the Loire River in France in 1998 has resulted in restored salmon spawning in the river system upstream...
of the old dam (RiverNet 2008). In Spain over fifty dams have been removed for the purpose of improving salmon stocks and preventing floods (Brufão 2008).

**Maintenance and Functioning Perspectives**

Dams must be maintained to work properly, whether it is for flood control, recreation, hydropower, or a combination of functions. Potential maintenance concerns are often handled during mandated safety inspections. Additionally, for dams being used for hydroelectricity in the United States, concerns over the ability of the dam to function are handled through the Federal Energy Regulatory Commission (FERC) relicensing process. If the current owners or operators of a dam decide that they no longer have the ability to maintain the integrity of the dam system or it no longer will serve the desired functions, then removal of the dam should be considered as an option.

The long-term structural integrity of a dam is based in part on the materials used. Many of the dams built in the eighteenth and nineteenth centuries have a rock and timber core that may or may not have been covered later with concrete, while large modern dams use both earthen materials and concrete. A dam’s materials and construction must be able to withstand a variety of forces including those of water, sediment, and climate. Some reservoirs can fill with sediment in less than fifty years, at which point it no longer serves its original function (McCully 1996). Further, predicted changes in climate (precipitation and temperature) would alter storm intensities and overall precipitation patterns, resulting in changes in the magnitude of flows and sedimentation, and ultimately the severity of consequences (Emanuel 2005). Rivers impounded by dams lack the capacity to adjust to changes in discharge due to unusual circumstances (Palmer et al. 2008). For example, in 1975 more than 200,000 people died in the Henan Province of China when the Banqiao Dam and at least sixty others collapsed due to the combination of sedimentation and an intense typhoon (McCully 1996).

**Liability Issues**

In many regions, dam owners are held accountable for the safety of dams and are responsible for damages caused by their failure, even though they may not have built the dams. For example, in the US commonwealth of Pennsylvania, dam owners must do what is necessary to prevent injury to persons and damage to property. This includes the placement of signs alerting others to the presence of the dam as well as regular inspection, and maintenance and repair when necessary. Failure to comply can result in civil and criminal actions resulting in jail time and monetary penalties in the millions of US dollars. Rather than deal with the costs of repair, maintenance, potential for lawsuits, and insurance premiums, private owners of small dams in Pennsylvania, as well as other areas, have chosen removal. To date, Wisconsin and Pennsylvania have removed over two hundred dams and lead all other states in dam removal in the United States (Gleick et al. 2009).

**Legal and Policy Perspectives**

In addition to laws related to safety, other laws and policies have helped to drive dam removal. For example, in Wisconsin, removal of abandoned dams by the Wisconsin Department of Natural Resources is mandated by the state. In Sweden, the government’s outline of sixteen ecosystem goals that should be achieved by 2020 includes the restoration of 25 percent of the valuable and potentially valuable streams and rivers by 2010 (Lejon, Malm Renöfält, and Nilsson 2009).

**Socioeconomic Perspectives**

The decision to remove a dam must factor in the value of the goods and services provided by the environmental resource as well as the effect of removal on jobs, incomes, and surrounding communities in the long term (Whitelaw and Macmullan 2002). Cost-benefit analyses for small dams have demonstrated that in some cases the cost of repair was at least three times higher than the cost of removal (Born et al. 1998). As stated earlier, the liability costs have driven many dam owners to the removal decision. Other important cost factors include those specific to a dam’s function (e.g., profit margins for the production of electricity or loss of water supply if removed). Facets of the local or regional economy may be affected by the loss of the impoundment, including recreational and fishing opportunities, tourist revenue, and associated jobs. In 1997, Edwards Dam was denied renewal of its Federal Energy Regulatory Commission license for hydropower operation after the FERC determined that the amount of electricity produced did not justify the amount of environmental harm that resulted from the operation of the dam.

Fish passage restoration is a driving economic force for the consideration of dam removal. In the case of the Elwha and Glines Canyon dams on the Elwha River in the US state of Washington (removal of which began in September 2011), restoration of the ecosystem, including anadromous fish passage as well as the cultural and economic concerns of indigenous peoples, played key roles in the dam removal decisions (NPS 2011). (See figure 1.) Neither of these dams has fish ladders, nor are other means such as trucks or boats used to move fish around
Figure 1. Removal of a Dam on the Elwha River, Washington State, October 2011

Source: Photo courtesy of Tami Heilemann, US Department of the Interior.

Fish passage restoration is a driving economic force for the consideration of dam removal, which is typically a large operation. The dam on the Elwha River being removed in this photo was not built with fish ladders or other means of moving fish past the dam.

the dams (which can require considerable resources). In Sweden, the cost-benefit analyses for some dam removals have included the socioeconomic benefits to the indigenous Sami people, as well as issues of reindeer passage and fish passage (Lejon, Malm Renöfält, and Nilsson 2009). Additionally, cost-benefit analyses for dam removal, depending on the history of the watershed, may have to take into account the costs of remediation of contaminated sediments, the costs of restoring the riparian area, or other environmental or health concerns related to that stream system.

Challenges of Dam Removal

The removal of a dam from a flowing water system is a disturbance to that system. Complex relationships exist among the physical, chemical, and biological factors that are related to the presence of the dam (Hart et al. 2002; Bushaw-Newton et al. 2002; Bushaw-Newton, Ashley, and Velinsky 2005). Understanding how the removal of the dam will affect the stream system is a challenge for several reasons. First, the current body of knowledge on the effects of dams on stream systems relates primarily to large dams. The majority of removals, however, are occurring for dams that are considered small; more than 80 percent of dams removed in the United States in recent years were less than 20 meters high (American Rivers 2011). Without a comprehensive understanding of how small dams affect physical, chemical, and biological processes in an ecosystem, it is difficult to predict all the ecological outcomes of removing a dam. Dam and impoundment size likely play roles in determining the extent of alteration, but other factors such as climate, geology, geography, overall ecology, and human use also factor into the magnitude of change (Hart et al. 2002).

Second, fewer than fifty dam removals have been studied to examine the ecological effects on a stream system.
Figure 2. Possible Consequences of Dam Removal

(For a list of studies, see Hart et al. 2002.) Most of these studies are not comprehensive and only cover a few aspects resulting in strong knowledge of fish movement but little knowledge of chemical components (Hart et al. 2002). Unfortunately, none of these studies have continued for time periods necessary to assess the responses of all ecosystem components. While some responses are seen in the first days to months—such as the movement of biota, movement of sediment, and changes in the physical characteristics of the water—other processes such as stream formation and tree growth take years, decades, or longer for full maturity (Hart et al. 2002). Also, flowing water systems are dynamic and represent a variety of interactions of hydrology, geology, climate, ecology, and other factors. The interactions, responses, and rates of response for components of a system will not all be the same after dam removal. (See figure 2.) In some systems, the sediment behind the dam may all be fine-grained and move downstream very quickly, while in other systems, the trapped sediment may consist of gravel and take years or decades to move downstream. Further, the magnitude and frequency of precipitation events are factors in the rates of sediment movement. Understanding projected future climate change and the accompanying changes in the water cycle will be pivotal in improving our understanding of dam removal processes.

Third, removing a dam requires a strong understanding of the history of the watershed and its hydrology, geology, ecology, and biology. (For a comprehensive list of recommendations, see Bushaw-Newton et al. 2002.) Pre-removal assessments are time consuming, require expertise, and are expensive, but they are necessary to determine if dam removal is the right option and how and when the removal should proceed. Other factors should also be considered:

- Are there endangered or threatened species present?
- Do fish spawn in this system?
- Are there mussels and/or other sessile creatures in this system?
- Are invasive species present in the impoundment and/or downstream?

*The possible consequences of dam removal are numerous and interdependent. This diagram illustrates general categories of effects that should be assessed when considering or planning a removal. Note that wetlands associated with riparian areas would most likely continue to exist or form.*
• Are there contaminants (e.g., polychlorinated biphenyls [PCBs], polyaromatic hydrocarbons [PAHs], heavy metals) present in the sediment of the impoundment?

Does this system regularly flood?

When Fort Edward Dam on the Hudson River in the US state of New York was breached in 1973, the dam owner had not adequately tested for PCBs in the sediment, although it was known that upstream of the impoundment there was an industrial facility that used and discharged the known carcinogenic compounds (Shuman 1995). Today, due to the volume of sediment that has moved through the river system, parts of the Hudson River are hazardous waste sites requiring cleanup, and certain fish there are deemed not edible by the government (EPA 2011). Dam removals must be viewed within the context of a larger watershed management plan and long-term goals. If, for example, the sediment is highly contaminated, but the rest of the watershed downstream has far lower levels, then the dam removal plan must account for the ultimate fate of the sediment (e.g., dredging before removal) and determine if dam removal is the best option. If, however, the sediment in the watershed downstream is already highly contaminated, then the removal of the dam is not adding any contaminant into the system that is not already present and dam removal should still be viewed as an option. The latter was actually demonstrated in the removal of the Manatawny Creek Dam in Pennsylvania in 2000, which was part of the larger Schuylkill River watershed. Contaminant concentrations in sediments in downstream reaches were similar to those measured in sediments in the impoundment (Ashley et al. 2006).

Fourth, dam removal plans must take into consideration stakeholders’ concerns and interests. Finding common ground and understanding can be difficult. There are several common stakeholder concerns throughout the world that are continually addressed:

• fear of change and unknown consequences
• the cultural and personal significance of the dam (e.g., the Hoover Dam is a major tourist attraction, although the chances of a dam of this size being removed are extremely slim)
• loss of revenue and recreation opportunities (e.g., fishing, boating, hunting, swimming)
• decrease in property value (e.g., due to loss of lakefront property or easy access to a recreational opportunity)
• aesthetics of the former impoundment (e.g., will there only be muddy banks remaining?)
• is it necessary?

The majority of these concerns may be overcome by holding meetings with stakeholders that carefully explain the process and potential outcomes as well as the current state of knowledge. For example, the removal of dams has not been shown to cause property values to decrease (Provencher, Sarakinos, and Meyer 2008), and several researchers have demonstrated that shortly after removal (weeks to months) vegetation returns. Stakeholders represent many interests and are key components in the present and future state of a watershed, so understanding their concerns is critical in developing watershed management plans.

Finally, funding is often a challenge for a dam removal. Finances should cover not only the removal of the dam, but also meetings with stakeholders as well as any pre- and post-removal assessments to be conducted. The removal of very small dams (less than 2 meters high) can run into the tens of thousands of dollars or higher for the removal component alone. When assessments are factored into the finances, the cost can be in the hundreds of thousands. Therefore some dam owners may be inclined to forgo the assessments and a detailed removal plan and just blow up the dam, and this can have serious consequences if contaminated sediments or endangered species are involved.

Outlook

Dam removal is an emerging tool in river restoration that is gaining interest internationally. Although dams continue to be important to development and maintenance of land and water resources and are still being built, they also are now being removed, as they are recognized as being potentially detrimental and thus not to be added or kept unless their usefulness is judged to outweigh their environmental and socioeconomic impacts. In the United States, Spain, Sweden, France, and other countries, dams are being removed to improve the ecology of the water system; to eliminate maintenance, safety, and liability costs; and to restore cultural values. Dam removals represent the result of a complex interplay, including ecological, socioeconomic, and cultural factors, making the planning and execution of removals challenging. Understanding how these factors interact within a context of a sustainable watershed management plan, and how dam removal fits within the goals of that plan, are key to improving the overall science and decision-making process. As more removals occur, our knowledge of the types and rates of responses of ecological components will continue to improve. In turn the uncertainties and concerns associated with removal will decrease, leading to better decisions about the use of dam removal as a watershed management tool.

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See also Adaptive Resource Management (ARM); Disturbance; Ecological Restoration; Ecosystem Services; Eutrophication; Extreme Episodic Events; Fish Hatcheries; Hydrology; Irrigation; Water Resource Management, Integrated (IWRM)
FURTHER READING


Desertification can be defined as the transformation of arid or semiarid land into desert. Desertification has been blamed on everything from poor land management to naturally changing weather patterns. While billions of dollars have been spent since the 1960s to combat desertification, there remains no consensus on causes, remedies, or whether the issue even requires human intervention.

The UN Convention to Combat Desertification officially defines desertification as "land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities" (UNGA 1994, 4). In less formal language, desertification could be defined as a process by which a previously arid or semiarid but productive land becomes a desert.

Drylands (arid or semiarid regions) occupy about 40 percent of the Earth’s surface and are home to one-third of the world’s population. It is believed that, among arid regions of the world, those most affected by desertification are sub-Saharan Africa and the former Soviet republics of Central Asia, where the risk of land degradation is also very high.

According to the United Nations, desertification is displacing large populations of people and forcing them to leave their homes and lands in search of better livelihoods. It is estimated that 135 million people—the combined populations of France and Germany—are at risk of being displaced by desertification (UN 2007, 14). The problem appears to be most severe in sub-Saharan Africa, the Sahel, and the Horn of Africa. It is estimated that some 50–60 million people will eventually move from the desertified areas of sub-Saharan Africa toward North Africa and Europe by the year 2020 (UN 2007, 20). The UN further reports that between 1987 and 2007, nearly half of the total male population in Mali in western Africa had migrated at least once to neighboring African countries or to Europe. In Burkina Faso, desertification can be identified as the cause of 60 percent of the swelling of the main urban centers (UN 2007, 20).

What Is Desertification?

The term desertification originally goes back to the French colonial forester André Aubréville (1897–1982), who described the process of “savannization” taking place in western Africa, in the tropical forest zone, as desertification. His ideas became part of a pan–west African forest orthodoxy shared by British and French foresters, one constituent of this orthodoxy being the important role played by traditional farmers’ activities (such as shifting cultivation) in destroying forest ecosystems.

In contrast to this first approach, which included regions with annual precipitation of up to 1,500 millimeters (the tropical forest zone), most contemporary definitions of desertification focus on arid, semiarid, and subhumid areas. These are frequently desert fringes.

By the beginning of the 1980s more than one hundred definitions of the term already existed. Some are complementary, whereas others appear contradictory. The great diversity and mix of definitions have led to miscommunication among researchers and policy makers. As many scholars claim, this overabundance of definitions may indicate that the very nature and causes of the problem remain uncertain.

Origins and First Assessments

In the late 1960s and 1970s the Sahel, a transitional band between the Sahara desert to the north and the humid savannas to the south, underwent severe droughts and
famine. As a consequence of drought, approximately 250,000 people and millions of animals died in six African countries. This was the first environmental catastrophe to be televised. Public opinion in developed countries was one of shock, and, for the first time, desertification raised international concern. Another terrible drought devastated the same region from 1983 to 1985.

The response of the United Nations to the environmental and humanitarian disaster of the Sahel was the organization, in 1977 in Nairobi, of the first international Conference on Desertification. The conference drew attention to the phenomenon of desertification, brought together representatives of the many countries affected or at risk of being affected by the problem, and encouraged scientists to research the topic. After the conference, desertification became a major issue of investigation in the academic world and was the subject of hundreds of articles and books. The problem was also identified in many other arid and semiarid zones outside Africa, such as the European-Mediterranean region.

Since 1977 desertification in Africa has become big business, politically sensitive, and a major North–South aid issue that has brought considerable financial support from the global North to desertification-related programs (Thomas and Middleton 1994).

Desertification is considered one of the biggest environmental problems of the last few decades and is among the World Bank’s priorities of major global environmental issues (Thomas and Middleton 1994). Since the 1970s billions of dollars of aid have gone into the Sahel. Yet, perhaps as little as 2 percent of foreign aid entering the Sahel countries has been spent on ecological projects that could improve the environment in the long run. The total investment between 1978 and 1990 on antidesertification projects was around US$6 billion, with little tangible effects. During that period, money was spent on projects that could be better described as economic development: building feeder roads, improving water supply, establishing seed multiplication facilities, and controlling animal diseases (Thomas and Middleton 1994).

The conference in Nairobi, more a political meeting than a scientific one, gave way to much misinformation and to a popular image of sand dunes advancing on villages and towns. The threatening vision of sudden burial by sand stimulated urgent calls for action from international organizations and donor countries. Nevertheless, as this “advancing desert” narrative faded in the absence of convincing evidence, attention turned to human actions. Humans were blamed for the onset of desertification and the disaster it had provoked. It was believed (and to some extent it is still believed) that the main causes of desertification were anthropogenic: deforestation, overcultivation, overgrazing, increased fire frequency, salination of soil from intensive irrigation, plowing of marginal areas, and overdrafting of groundwater. In developing countries these practices are intensified by population growth, high population density, poverty, poor land management, and inappropriate use of new and traditional agricultural technologies (Thomas and Middleton 1994; Pearce 1992).

Some attempts have been made to demonstrate that population growth has no direct bearing on desertification. It is obvious, however, that in certain situations increasing population may put pressure on drylands in at least two different ways. Firstly, increased population implies an increased demand for food, which has to be addressed by increasing production (area under cultivation) or productivity (yields per area). In most cases, the additional food comes from increasing production, including an expansion into marginal and more fragile zones that are usually susceptible to degradation. Secondly, with the number of families increasing comes a parceling of land, with each beneficiary owning smaller plots that are frequently overcultivated with negative consequences.

Economic globalization also puts pressure on farmers and pastoralists (people who raise livestock) in drylands. In developing countries, unfavorable trade terms and decreasing commodity prices encourage or compel dryland farmers to produce more to increase returns on poorly priced primary products, thus degrading natural resources.

On the other hand, in some countries of western Africa where desertification is usually considered to be a serious threat, land degradation has been ascribed to governmental policies. Incentives given for cultivating cash crops, at the expense of subsistence agriculture, expanded farmland into the southern fringes of the Sahel, traditionally reserved as dry-season grazing areas. Similarly, replacing nomadic pastoral systems with intensive fodder-crop cultivation and year-round grazing in Central Asia has led to severe pasture degradation (Sneath 1998).

Myth or Reality

From the onset of the drought in the Sahel, intense discussions over the nature and causes of desertification have been raging. Some researchers and scholars have even begun to challenge the existence of desertification and have referred to the topic as a myth, an imaginary environmental problem, or an “institutional fact” that has been perpetuated by organizations to warrant certain actions.

It is clear now that most of the data on desertification derives from places and periods that were affected by either severe drought or long-term reduction in precipitation. As a consequence, there is no agreement about the causes of desertification, the extent to which the reported
Changes are natural or human induced, the number of countries affected or at risk, or the reversibility or irreversibility of the phenomenon. The issue is poorly understood; some scientists in the early 1990s wondered whether a new convention to halt desertification might be more effective than previous flawed efforts. The United Nations Conference on Environment and Development (held in Rio de Janeiro in 1992 and also known as the Earth Summit) culminated in the creation of the UN Convention to Combat Desertification (UNCCD), which entered into force in 1996. The objective of this convention is “to combat desertification and mitigate the effects of drought in countries experiencing serious drought and/or desertification, particularly in Africa, through effective action at all levels.” This should be achieved through “long-term integrated strategies that focus simultaneously, in affected areas, on improved productivity of land, and the rehabilitation, conservation and sustainable management of land and water resources, leading to improved living conditions, in particular at the community level” (UNGA 1994, article 2). But taking into account the limited success of other similar international environmental processes in sectors like climate change (Brand et al. 2009) or forests (Humphreys 2006), it is questionable whether the UNCCD will make a greater difference.

During the 1970s and 1980s, the UN Environment Programme (UNEP), a UN agency then claiming a global mandate to address desertification, supported the view of desertification as “spreading deserts.” The UNEP frequently stated that annually 21 million hectares of once-productive soil were reduced by desertification to a level of zero productivity, and at the beginning of the 1990s it estimated the amount of the world’s land area threatened by desertification at 25 percent, although for many years UNEP had claimed that it was 35 percent (Pearce 1992, 42). In fact, at the time that UNEP was disseminating these figures, the effects of desertification in Africa and elsewhere had not yet been documented according to scientific standards. Correspondingly, many researchers contested the validity of the data, wondering where they came from.

What has really improved the understanding of arid and semiarid ecosystems is the application of remote sensing technology, particularly high-resolution satellite images that can be compared over different periods of time. Within such temporal frames, the edge of the Sahara appears to both advance and retreat, and a net expansion of the desert cannot be detected. Previous assessments of desert spread were wrong in assuming that trends observed in a few isolated places of the Sahel were happening across the entire continent.

Research in 2001 showed how the ecosystems of arid and semiarid regions fluctuate, their biological productivity governed by variations in prevailing weather patterns, such as annual rainfall or El Niño Southern Oscillation (ENSO), the atmospheric component of the cyclical climate shift known as El Niño (Lambin et al. 2001).

These results have also led to a reconsideration of the role played by the people in the process of desertification. Remote sensing studies have found little evidence of desertification around villages or watering holes in Africa. On one hand, traditional nomadic pastoralism (usually blamed for degrading the environment), characterized by frequent migration and fluctuating herd sizes, is well adapted to unpredictable arid and semiarid ecosystems. On the other hand, it appears that, at least in some cases, the land in high-population areas might benefit from water and soil conservation efforts and vegetation management practices (Mortimore 1989). As late as the beginning of the 1990s, however, UNEP still perceived desertification as a principally human-induced process. But based on more-recent research, scientists now question that view. Their criticism caused the United Nations to reconsider desertification drivers, taking a more moderate view and recognizing the role of natural factors, particularly climatic ones.

Climate Change and Afforestation

The effect of global climate change on desertification is complex and not yet sufficiently understood. On the one hand, higher temperatures can have a negative impact through increased loss of water from soil and reduced rainfall in drylands. On the other hand, an increase in carbon in the atmosphere can boost plant growth for certain species. Although climate change may increase aridity and desertification risk in many areas, the consequent effects of biodiversity loss on desertification are difficult to predict.

Despite contradictory evidence, the traditional view about the causes of desertification persists. For example, it is still commonly believed that in Africa the environment is being destroyed by traditional land users as a consequence of unregulated land access. In Europe, Mediterranean countries allocate considerable amounts of money for planting trees to halt the supposed advance of the desert, even though a rural exodus and the depopulation of inner regions since the 1950s has provoked a natural large-scale expansion of *matorral* (shrubland) and forests. Large-scale afforestations have been carried out in the Alpes-Maritimes, central Sicily, Calabria, and southern Sardinia (Thirgood 1981). (Afforestation, in contrast to reforestation, is the practice of planting trees where they have not previously grown in recent history.) In Spain, for instance, some 3 million hectares were afforested between 1940 and the mid-1980s (Groome 1988). Scholars explain the persistence of the old
In many countries, land-rights systems have been traditionally transformed, limiting the access of rural communities to natural resources, in order to allow more state intervention in the countryside (Davis 2005; Larson and Ribot 2007). There are also international environmental agencies, development-aid organizations, and non-government organizations (NGOs) “seeking to establish their authorities and legitimacy as environmental advocates and stewards. The desertification narrative persists in part because it serves to mobilize support for these groups’ varied agendas” (Bassett and Bi Zuéli 2000, 91).

Potential Pitfalls

Controversy also surrounds possible measures to address desertification. One popular practice implemented to halt the spread of deserts is planting trees. This was supported, for instance, by UNEP’s Plan of Action to Combat Desertification after the 1977 Nairobi conference, but the initiative did not succeed, mainly due to poor funding and the lack of engagement with the local population. Other failures have been reported in different geographical regions. The tradition of Mediterranean countries carrying out plantation forestry has already been mentioned; such afforestations have included vast areas of bulldozed semiarid slopes replanted with pines. The result has been soil erosion and the emergence of dense stands of pine without a shrub layer (Grove and Rackham 2001). In spite of these experiences, the popularity of tree planting still leads politicians to fall back on afforestation as a demonstrative environmental policy measure. For instance, in April 2010 the leaders of the member states of the South Asian Association for Regional Cooperation (SAARC) committed themselves to plant 10 million trees from 2010 to 2015 (SAARC 2010).

Other measures include privatization of natural resources (promoted by the World Bank), state interventionism in rural areas, transformation of land-rights systems, and exclusion of local people from protected zones.

Taking into account that many aspects of the topic are still badly understood, care should be used when practically addressing the problem. Researchers note that particular consideration should be given to the effects that proposed measures might have on poor, local communities. Specifically, they warn against cookie-cutter planning models typically applied by international institutions, instead emphasizing “locational and culturally appropriate technical and economic options” and “the necessity of moving away from regulation and intrusive administration” (Bassett and Bi Zuéli 2000, 76).

Outlook

Despite repeated criticism, it is widely recognized that desertification represents an important threat to drylands. Given the relevance of this problem, it is surprising that there is neither consensus on a definition nor is there consensus on the right way to assess the desertification status of a region. Since the 1980s, contradictory definitions have produced both different assessment methodologies and divergent estimates.

The coexistence of conflicting definitions and divergent estimates affects social perception negatively, leading to skepticism and, finally, to a delay of possible solutions. Societies, as well as international conventions, institutions, and agencies, must recognize the real progress that desertification research has made, leave behind notions that no longer represent current knowledge, grasp the opportunity to better understand the extent and intensity of the problem, and realize that assessing desertification accurately is still an unresolved issue.

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See also Adaptive Resource Management (ARM); Agricultural Intensification; Comanagement; Disturbance; Global Climate Change; Indigenous Peoples and Traditional Knowledge; Irrigation; Reforestation; Tree Planting

FURTHER READINGS


Disturbances are relatively discrete events in time that substantially influence ecosystem composition, structure, and function. Natural disturbances (such as hurricanes, avalanches, fires, and floods) play important roles in shaping landscapes and the biota that evolved with them. With increasing human population growth and resource demand, the direct and indirect effects of human-caused disturbances are posing an increasingly complex challenge for balancing ecosystem and societal needs.

In common usage, the word disturbance refers to the breaking up of a settled order or a departure from normal conditions. Most ecologists do not support a strict order in nature or a “normal” set of conditions for any given ecosystem, however. From an ecological perspective, disturbances are events that markedly affect ecosystem composition, structure, and function. A variety of physical (fire, windstorm, floods) and biological (outbreaks of insects, parasites, or pathogens) agents cause ecological disturbances. They elicit ecosystem change through directly modifying the biophysical environment, which indirectly affects the composition of plant and animal species. Disturbances also affect biota directly, by selectively killing certain species (e.g., flooding may kill all but the most flood-tolerant tree species in a floodplain forest) and initiating changes in the mutualistic and competitive interactions among species. When the focus is on an established community, disturbances clearly are events that disrupt community development. Given a broader perspective, however, disturbances from volcanism and tsunamis to local avalanches, floods, and fires are essential processes shaping landscapes and the biota that evolved with them.

The US plant ecologists Peter S. White and Steward T. A. Pickett define disturbance as “any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment” (White and Pickett 1985, 7). This broad definition encompasses a variety of actions that modify landforms (landslides and volcanoes), soil (floods, erosion, debris flow), biota (fire, insect outbreaks), or any of the processes that link these ecosystem components. Most disturbances produce patchy, heterogeneous effects that depend in part on the state of the ecosystem at the time of the event (e.g., windstorms may blow down large old trees but leave nearby younger stands relatively intact) and on factors acting during the disturbance (e.g., forest fire patterns driven by changes in wind speed or direction). One of the most important effects of disturbances is causing an ecosystem to depart from a trajectory of relatively predictable changes in composition, structure, and function, and either resetting or accelerating the sequence, or deflecting the ecosystem toward an alternate development pathway.

Human-caused disturbances are superimposed on the suite of natural disturbances in any landscape. Humans may dam rivers, clear forests, dump toxic chemicals, and transport plant and animal species to new environments. In some cases, human-caused disturbances are an integral part of the ecosystems as we know them. Although a controversial topic, burning by Native Americans is widely believed to have been a key feature of many ecosystems throughout the Americas (Denevan 1992). Other human-caused disturbances, such as extensively clearing tropical rain forests for agricultural use, have drastic effects that may be irreversible.

Importance of Scale

The events viewed as disturbances vary according to the spatial and temporal scale of consideration. For example,
the windthrow of a single forest tree (i.e., the tree’s uprooting and overthrowing caused by wind), and the associated changes in soil, microclimate, and available light may be important disturbances that alter, accelerate, or reset forest succession in the forest gap. When ecologists focus on broad forest landscapes, however, they see the variation in forest composition and structure associated with such treefall gaps as part of the heterogeneity characteristic of an otherwise undisturbed forest, and only those windstorms that affect numerous trees over extensive areas are viewed as disturbances (Everham and Brokaw 1996). Similarly, at short timescales, scientists may view the annual flooding and debris flows along the margin of a glacier as disturbances that create new landforms and influence local patterns of species colonization and elimination (Matthews 1992). If ecologists focus at a scale of tens of thousands of years, however, they may consider the terrain and biota relatively stable for centuries at a time, but they may see glaciation as a disturbance that reshapes the landscape and promotes the gradual development of new assemblages of plant and animal species.

Biological Legacies

Most natural disturbances leave living and dead organisms at various densities and patterns across the landscape. These biological legacies of the predisturbance ecosystem strongly influence the direction and rate of development in the recovering ecosystem (Franklin et al. 2000). For example, following the 1980 eruption of Mount St. Helens, the blast zone was not a barren moonscape to be recolonized only by dispersal from along its margin. Instead, numerous plants and animals survived in various types of refugia (e.g., tree seedlings that survived beneath snowbanks and seeds, spores, roots, and hibernating animals that survived below ground) that provided multiple sources for recolonization (Dale, Swanson, and Crisafulli, 2005). Biological legacies of the predisturbance ecosystem persist in three general forms: (1) live individuals or seeds, spores, fungal hyphae, and other structures (e.g., rhizomes) capable of regenerating into new organisms; (2) nonliving structures, such as dead wood in forests or coral in marine systems, that moderate microclimate and provide an energy source and critical habitat functions (e.g., protection from predators); and (3) biological modifications of the physical environment, including soil aggregates, rooting channels, ant mounds, and pit-and-mound microtopography (i.e., hollows with new soil coverage that can foster the growth of buried seeds or seedlings), a condition resulting from uprooted trees. Human-caused disturbances, such as clear-cutting, tend to remove more of these legacies and do so in a more uniform pattern than natural disturbances. As a result, patterns of recovery following human-caused disturbances typically are simplified compared to those following natural disturbance.

Adaptations

Plant and animal species exhibit numerous adaptations and life-history strategies for (1) avoidance of damage by disturbances, (2) recovery following disturbance, (3) colonization after disturbances, and even (4) promotion or facilitation of disturbances. Thick bark, a tree’s self-pruning of its lower branches, and rapidly decomposing foliage, for example, help protect trees from fire damage. Many plant species can sprout from the root crown, rhizomes, or other structures, which enables rapid recovery following damage by fire, grazing, or numerous other factors. Light, wind-borne seeds, water-dispersed seeds in areas prone to flooding, and long-term seed storage in dormant seed banks enable rapid colonization following the removal of plant cover. Several species found in fire-prone environments (e.g., jack pine and lodgepole pine in North America and species of the genus Banksia in Australia) store seeds in closed (serotinous) cones that open to release the seeds only when exposed to high temperatures associated with burning. Some researchers hypothesize that in fire-prone environments, species whose regeneration depends on the reduction of competition or exposure of seedbeds by fire tend to exhibit characteristics that promote the spread of fire, including flammable foliage (conifers and chaparral shrubs, for instance, generally are more flammable than deciduous trees) and the retention of dead leaves and branches (Gagnon et al. 2010).

Animal species also exhibit numerous strategies for persistence in disturbance-prone environments. For example, salmon are well adapted to the highly dynamic
stream systems in the Pacific Northwest of the United States. Relatively infrequent fires in the surrounding forests followed by floods and debris flows episodically deliver, transport, and rearrange large volumes of sediment and logs in these streams (Swanson et al. 1998). Such processes destroy salmon habitat in the short term, but they sustain many of the habitat features that salmon require in the long term. Adaptations that help salmon persist under the dynamic habitat mosaics characteristic of these stream systems include high fecundity rates, adult straying, and juveniles’ high mobility (Reeves et al. 1995).

Regimes

Although individual disturbance events may gain much attention, especially those affecting large areas, ecologists need to understand multiple, successive events within a given region to understand the ecological role a disturbance agent plays in that region. Scientists commonly describe the series of multiple, overlapping disturbances as a disturbance regime and define it by the following characteristics: (1) disturbance type or the agent of disturbance; (2) seasonality; (3) frequency (the number of disturbances at a given point per unit time); (4) extent; (5) magnitude, described either as intensity (a measure of physical force, such as the energy released per unit time in a fire or wind speed in a hurricane) or severity (a measure of the effects of the disturbance on organisms or the ecosystem); (6) internal patchiness, described in terms spatial variation in disturbance magnitude; (7) synergism, or the effect of disturbances on subsequent disturbances by the same or different agents; and (8) predictability or variability in the above characteristics (White and Pickett 1985; White and Jentsch 2001).

The classification of a series of recurring disturbances and their effects into generic types of disturbance regimes repeated in multiple regions provides a means for comparing the ecological roles of disturbances among regions. A regime of relatively infrequent and severe forest fires, for example, is common to most boreal and subalpine forests around the world. Under such a fire regime, each fire produces abrupt changes in the relative abundance of certain plant and animal species and in nutrient cycling and hydrology that are likely to persist for several decades (Romme et al. 2011). By contrast, in other regions where fire-free intervals rarely last longer than ten to twenty years and flame lengths usually are too short to kill more than individual or small groups of trees, each fire is likely to have comparatively little effect on plant and animal communities or ecosystem dynamics. In the case of such a frequent disturbance, however, activities that suspend or exclude these disturbances (e.g., active suppression of fire or river damming) may constitute the disturbance that leads to new and unexpected consequences (Allen et al. 2002; Gergel, Dixon, and Turner 2002).

Human-Caused Disturbances

To some degree, human-caused disturbances affect almost every ecosystem around the world. People intentionally clear forests for agricultural use, accidentally spill toxic chemicals, and unintentionally alter global carbon and nitrogen cycles as a consequence of agricultural methods and burning fossil fuels (Vitousek et al. 1997). Human-caused disturbances (also known as anthropogenic disturbances) are not new to ecosystems, but as human population and resource demand grows, the extent and magnitude of many of these disturbances increase, as they increasingly affect areas that previously had little human impact.

Early anthropogenic disturbances include widespread burning during the initial human colonization of Australia (c. 45,000–50,000 years ago), which may have triggered a series of megafaunal extinctions and vegetation change (Miller et al. 2005). Similarly, the initial Polynesian settlement of the South Island of New Zealand in the thirteenth century brought the widespread use of fire to forests that had little history of fire (e.g., fires previously had
occurred about every 1,000 years). Because most vegetation was poorly adapted to fire, a relatively small human population was able to rapidly deforest much of the island (McWethy et al. 2009).

Many recent anthropogenic disturbances have had severe effects. Oil spills such as the Exxon Valdez incident in Prince William Sound, Alaska, and others in the Gulf of Mexico even before the recent British Petroleum spill in 2010, have profoundly affected marine wildlife in the short term, and they may lead to unsuspected long-term impacts on trophic chains and other species interactions (Peterson et al. 2003). In addition to the direct consequences of anthropogenic disturbances, humans indirectly alter natural disturbance regimes. Few regions around the world are unaffected by grazing by introduced livestock species. The late nineteenth- and early twentieth-century initiation of livestock grazing in the American Southwest profoundly impacted the fire regime. By decreasing the continuity of grasses that previously carried surface fires, grazing contributed to abundant establishment of pine seedlings, which eventually grew to form a taller fuel capable of carrying fire through the crowns of the widely spaced older trees (Cooper 1960).

Management

Ecosystem management in the context of disturbances requires the combination of insight and knowledge from multiple disciplines ranging from ecology (how will the ecosystem respond to treatments?) to sociology (what is the acceptable degree of disturbance for an inhabited area?). Disturbance management approaches are highly variable and depend on the objective they pursue. In some cases, the objective could be to maintain the disturbance regime within the natural bounds of variability (Keane et al. 2009). In others, the objective may be complete suppression of disturbance to protect objects of value, such as homes. Managers may employ multiple strategies to reach a common objective, but determining the most effective strategy requires detailed knowledge of the local landscape.

Disturbance management is a contentious subject. Wildfire management in the western United States is one of the most controversial natural resource management issues today, as it transitions from a policy aimed at complete suppression of all fires, toward greater recognition of fire’s valuable ecological role in some landscapes, and addressing the negative consequences of nearly one century of fire suppression in these landscapes (Keiter 2006). Most forest regions of the western United States have experienced a marked increase in the occurrence of large fires starting in the late twentieth century and continuing into the twenty-first century (Westerling et al. 2006). Multiple factors have contributed to this increase, each with different implications for management. In some cases, the suppression of formerly frequent fire has enabled an increase in tree density, which leads to greater fuel amount and connectivity, and thus, greater potential for extensive, severe fires. In these forests, thinning small trees followed by the reintroduction of low-severity fires may be a useful strategy for reducing future severe fires. Other forests historically burned infrequently and only under the most severe (dry and windy) weather conditions. High densities of small trees in these forests are not a consequence of the suppression of formerly frequent fires. Thinning these trees in an effort to reduce fire risk may have little effect under severe weather conditions, when these forests are most likely to burn. It is critical for planners to avoid broadly applying generalized management prescriptions (Baker, Veblen, and Sherriff 2007).

The Future

Managing disturbances is a complex endeavor that requires a thorough understanding of multiple aspects of ecosystems, ecology, and societal needs (Turner 2010). Despite the increasing experience and knowledge that scientists are gaining, they need more research to successfully manage the complex disturbance scenarios, such as disturbance interactions and novel disturbances, which are occurring on Earth today.

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See also Adaptive Resource Management (ARM); Complexity Theory; Dam Removal; Desertification; Ecological Restoration; Extreme Episodic Events; Fencing; Fire Management; Indicator Species; Outbreak Species; Refugia; Regime Shifts; Resilience; Shifting Baselines Syndrome; Succession

FURTHER READING


Ecological forecasting is the integration of physical, biological, and sometimes social models to predict how organisms and ecosystems will respond to environmental change. Forecasts vary spatially from a local area to the entire globe and temporally from a single event to centuries. Applications include predicting distribution shifts of species and ecosystems in response to climate change and predicting the spread of diseases and introduced species.

Weather forecasts allow us to appropriately prepare for encounters with the environment on a daily basis. Forecasts are essential to deciding whether to leave the house with an umbrella or a sun hat. Human activities responsible for greenhouse gas emissions and land conversion are fundamentally altering the physical environment on a global scale. Predicting how organisms and ecosystems will respond to these changes is imperative for policy and planning. Analogous to weather forecasts, ecological forecasting seeks to predict how organisms and ecosystems will respond to chemical, biological, and physical changes in the environment.

A primary application of ecological forecasting has been predicting how species’ abundance and distribution will shift in response to environmental change. Other important applications include predicting the spread of diseases and introduced species, and ecosystem responses to altered nutrient concentrations and land use change. Physical applications include predicting ecosystem nutrient cycling and hydrological dynamics. Ecological forecasting initiatives ultimately aim to predict the implications of environmental change on biodiversity and ecosystem services.

Ecological forecasting may be applied at spatial scales ranging from a local area to the whole world, and at temporal scales ranging from months to decades or even centuries. Ecological forecasting may also be applied to single events such as predicting nutrient leaching from a forest following a rain storm. It seeks to ask specific questions such as “how will precipitation over the next season influence the grape harvest?” or general questions such as “what will be the impact of climate change on the distribution of biodiversity?”

Methods

Most ecological forecasting models involve a budget balancing inputs and outputs. At the ecosystem scale, ecosystem models account for nutrient fluxes between nutrient reservoirs, and hydrologic models account for the movement of water between pools. At the scale of individual organisms, energy losses and gains from interactions with the environment can be accounted, often as a heat budget. Most predictions of organismal responses to climate change or new environments, however, have involved using statistical correlations to relate the presence or absence of individuals to environmental conditions such as temperature and precipitation. These models rely on defining an environmental niche or climate envelope. They assume that a species will maintain a constant niche and will thus track their environmental limits through space as climate changes. While this method can readily produce forecasts with only geographic coordinates of species’ localities and gridded environmental data, the method has several limiting assumptions (Buckley et al. 2010).

One limitation concerns the likelihood that large portions of the globe will experience novel climate conditions by 2100 due to climate change (Williams and Jackson 2007). These novel climates pose two challenges to environmental niche models. First, the models assume that the relationship between climate gradients such as
temperature and precipitation will remain fixed. Second, the validity of the models in extrapolation is uncertain. Additionally, the models generally omit all biological details. These omissions include geographic variation in species’ characteristics including thermal tolerance and well as species interactions. The models are based on environmental conditions averaged over long time periods (e.g., annually), while environmental conditions can influence organismal physiology over the scale of minutes.

Models that incorporate additional biological details are rapidly emerging as a complement to correlative species distribution models that attempt to overcome these limitations. These mechanistic models attempt to describe the processes that constrain a species’ abundance and distribution. The models generally require detailed information on physiology, morphology, and the environment. Biophysical models account energy losses and gains from an organism’s interaction with the environment. This energy budget enables predicting the organism’s body temperature. The predicted body temperatures, termed *operative environmental temperatures*, can be compared to the organism’s thermal limits to forecast thermal and water stress and ultimately mortality. Some mechanistic models use biophysics as a basis for directly predicting abundance and distribution. These models translate body temperatures into energetics and performance and ultimately demography. Plant growth models are employed to estimate photosynthesis rates as a basis for demography. The implementation of these models is currently sharply limited by the availability of data on organismal traits and detailed environmental data.

### Data Resources

Ecological forecasting is an emerging approach that focuses on assembling detailed organismal and environmental data and using this data in detailed models to predict organismal responses to environmental change. The collection and assembly of plant trait data are accelerating. For example, a network of vegetation scientists comprise the TRY Initiative (n.d.), which aims to assemble a global database of plant traits. Databases for animals have lagged somewhat behind. Information on species’ abundances and distributions is increasingly available with the expansion of online bioinformatics tools. Long-standing surveys of consistently located grid cells or transects continue to provide important abundance and distribution data. The most extensive survey programs are in Europe, particularly in the United Kingdom, where the British Trust for Ornithology monitors birds and the Biological Records Center monitors a wide variety of freshwater and terrestrial organisms. In North America, long-running seasonal surveys of birds (e.g., Breeding Bird Survey, Christmas Bird Count) and butterflies (e.g., Fourth of July Butterfly Count) provide important abundance and distribution data.

The biophysical models used in ecological forecasting require environmental information including surface and air temperature, radiation, wind speed, reflectively of the surface, and measures of water availability. Compilations of weather station data both regionally and globally provide this information. This weather station data has been interpolated to produce gridded environmental data. Another important source of information on spatial variability is remotely sensed data from aircrafts and satellites. Within the United States government both NOAA (National Oceanic and Atmospheric Administration) and NASA (National Aeronautic and Space Administration) sponsor initiatives to make remotely sensed data available for ecological forecasting. Additionally, NOAA has proposed a national climate service to expand its weather forecasting to longer time scales. NASA’s Ecostart program focuses on “monitoring, modeling, and forecasting ecosystem change.” In Australia, CSIRO (Commonwealth Scientific and Industrial Research Organization) and its climate adaptation flagship are focused on ecological forecasting. Similar initiatives are distributed globally.

Emerging initiatives are implementing biological and climate data collection programs specifically focused on ecological forecasting. The National Ecological Observatory Network (NEON) is a proposed observatory network across the United States that aims to “enable forecasting of ecological change at continental scales over multiple decades” (NEON n.d.). NEON has partitioned the United States into twenty domains based on similarities in habitat and climate. Representative sites in each domain will be the focus of airborne observations, and experiments and will be linked in a sensor network. Raw data on climate and atmosphere, soils and hydrology, and a variety of organisms will be synthesized into data products that can serve as the base for ecological forecasting.

Validating and testing ecological forecasting methods is essential to increasing confidence in their predictive capacity. The models may be validated by predicting current abundances or distributions based only on organismal traits and environmental conditions. This is known as *hindcasting*. A robust model test is to parameterize the model based on past conditions and test the model by predicting more recent distributions and abundances. Predicting responses to past environmental changes is known as *hindcasting*.

### Applications

Ecological forecasting is a relatively new endeavor and has had only a limited application to ecosystem management.
as of 2011, but its implementation is rapidly accelerating as data resources accumulate and climate change accelerates. While much legislation aimed at curbing greenhouse gas emissions focuses on constraining the temperature increase at a future time period, say a 3°C temperature increase by 2100, little is known about how these temperature increases will impact organisms. Will a threatened plant species be able to withstand a 3°C warming but face extinction if temperatures rise by 5°C? Ecological forecasting is essential to considering the potential ramifications of environmental policies and informing future management needs. Understanding the potential impacts to biodiversity and ecosystems service of continuing to emit greenhouse gases at current rates is central to motivating policies to curb emissions.

Rising temperatures are already pushing organisms toward their thermal limits, and atmospheric concentrations of greenhouse gases commit us to further temperature increases. It is thus essential to identify and mitigate species and ecosystems that are likely to be particularly impacted. One notion is that ecological forecasting should be used by biologists and resource managers to implement ecological triage. Concentrating on areas where species are likely to be pushed just over their stress limit may provide the most effective use of limited resources and personnel effort. Consider a series of reserves threatened by a gradient of habitat degradation in addition to climate change. It may be futile to expend resources on the most degraded reserve where the multiple stressors may commit species to extinction regardless. Conversely, in the least degraded reserve, the ecosystem may be sufficiently robust to withstand increased temperature stress. Resources may be best allocated to a reserve with intermediate habitat integrity. Restoration efforts there may increase resiliency to thermal stress and enable persistence.

Ecological forecasting is providing potentially crucial information about which geographic regions are likely to be most sensitive to climate change. Traditionally, researchers have thought that the ecological impacts of climate change will be concentrated in polar regions, where temperature increases are expected to have the largest magnitude. But when one uses organismal biology to translate temperature changes into thermal stress, it appears likely that tropical organisms may incur the most severe ecological impacts. This is because the lesser annual temperature fluctuations in the tropics drive species to specialize their physiology to a narrow range of environmental temperatures (Tewksbury, Huey, and Deutsch 2008).

Assisted migration is the act of deliberately moving potentially imperiled species to areas where they can thrive in an appropriate environmental niche when they cannot move there on their own. It is increasingly considered as a last-ditch conservation effort particularly in response to climate change. Ecological forecasting predictions would be central in identifying the appropriate areas to which species should be moved. Indeed, in an initial test of the viability of assisted migration, environmental niche modeling was used to identify where a species of butterfly should be relocated. Ten years since the assisted migration, butterfly populations are thriving in their new locations (Willis et al. 2009).

Ecological forecasting has been applied to inform agricultural management. For example, the NASA Terrestrial Observation and Prediction System (TOPS) program has been involved in producing forecasts to aid planning by the California wine industry. Warmer satellite-observed sea surface temperatures (SST) have been found to enhance wine quality via reduced humidity and frost frequency and a lengthened growing season (Nemani et al. 2001). SST has thus been used to predict vintage quality. Measures of vegetation growth such as leaf area index have been used to parameterize ecosystem models, which predict optimal irrigation practices for maintaining vines at preferred water stress levels (Nemani et al. 2001).

Outlook

Despite the imperative for consistent and reliable ecological forecasts, progress toward this goal has been slow. One impediment is the need to coordinate expertise across disciplines. Integrating surface and remotely sensed environmental data sets requires expertise and substantial computing capacities for processing and storage. The environmental data must be processed in a manner that is relevant to organisms and ecosystems, and biologists must assemble and contribute relevant organismal data. New modeling approaches are required to integrate environmental and organismal data. The potential for rapid
progress in ecological forecasting is offered by emerging initiatives bringing together computer scientists, remote-sensing specialists, climate modelers, hydrologists, and organismal and quantitative biologists. Improvements in computing and sensor technologies and in climate change projects should facilitate this progress. Understanding the impacts of environmental change on species, hydrology, agriculture, and ecosystems will be essential to the maintenance of ecosystem services and biodiversity.

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See also Adaptive Resource Management (ARM); Agroecology; Best Management Practices (BMP); Biogeography; Boundary Ecotones; Complexity Theory; Ecosystem Services; Edge Effects; Global Climate Change; Habitat Fragmentation; Indicator Species; Keystone Species; Plant-Animal Interactions; Population Dynamics; Species Reintroduction

FURTHER READING


Ecological restoration refers to the process of restoring damaged ecosystems. This is accomplished through an accurate assessment of the ecosystem’s processes, identifying and reversing the environmental stressors or causes of dysfunction, conducting an inventory of native plants and animals, and reintroducing absent species so the ecosystem regains ecological balance and stability.

No ecosystem on Earth has escaped at least some level of disturbance resulting from human activity, ranging from complete conversion to artificial structures, such as cities, to minor contamination by toxic chemicals. The result is a diminished resiliency, the ability of damaged ecosystems to maintain themselves against continued insult. Ecological restoration is human intervention intended to restore or improve impaired or dysfunctional ecological processes.

Healthy ecosystems perform a myriad of complex processes carried out by teams of component species. When one or more populations of species is lost or reduced, the processes with which those species are involved are diminished.

Ecosystems can be viewed from three perspectives: composition (biodiversity); structure (primarily the size and age of the vegetation present); and processes (all the physical and chemical interactions between and among the component species). It is the composition—the species—that creates the structure and carries out the processes. While the aim of ecological restoration is restoration of the processes, the approach is nearly always through adjustments in structure, or more commonly, species.

Because of overlapping niches, many species can perform similar processes. Rich diversity usually means there will be more overlapping niches, such that ecological processes suffer relatively little when one, or a few, species is missing. For this reason a full complement of native diversity is not essential for ecosystems to function. As diversity is lost, however, the ecosystem processes with which those species are involved will begin to diminish. Restoring missing species, therefore, is a primary approach to ecosystem restoration.

Structure

Whereas every species in an ecosystem plays some role in one or more ecological processes—some far more important than others—only a handful of species play important roles in structure. Because most species in an ecosystem are not easily seen, and, indeed, most may not even be known, it is easy to forget that the vast majority of species are cryptic if not microscopic. Most of the myriad insects and many of the smaller vertebrates, for example, are seldom seen. It is the fungi, bacteria, and archaea (a specialized group of single-celled microorganisms), however, that comprise most of the diversity in an ecosystem, and many of these species have yet to be discovered. The structure of an ecosystem is primarily provided by plants, with a few important exceptions, such as coral in a reef. Consider a forest. There are big trees, smaller trees, shrubs and vines, epiphytes (plants, such as bromeliads, or those commonly called air plants that attach themselves to other plants), and perhaps ferns and other herbaceous species on the forest floor. In undisturbed forests, there will be standing dead trees in various stages of decay, fallen trees and limbs, and other coarse and fine litter over the soil. Even the soil is structured, from the recently fallen litter through various stages of decay and incorporation into the mineral layers. Not only does this structure provide the habitat for all the other species that are part of the community, but the species, in turn, interact with the plants to create and
develop the structure. This is especially true of the soil, where the decomposition process and incorporation of organic material depends largely on organisms associated with the plants. It is sometimes possible to adjust structure and restore habitat and species that, in turn, correct dysfunctional ecological processes. For example the restoration of savannas or prairies overgrown by invasive woody plants is best achieved with combinations of cutting and fire to reduce the woody vegetation.

At a basic level, ecological processes involve energy transformations and shifts in chemical structures. Nutrient cycling is a good example. This process, fundamental to healthy ecosystem function, involves use of potential energy primarily by plants, fungi, bacteria, and archaea to absorb dissolved compounds and convert them to needed vitamins, enzymes, or tissues. Animals largely obtain needed nutrients from other organisms that make up their diet. Following death or excretion, the complex molecules are oxidized by decomposing organisms, releasing energy that is used to support their metabolic requirements. Eventually the nutrients are released into the soil or water in water-soluble forms that can again be absorbed. In a typical nutrient cycle, hundreds of different species from two or more kingdoms will be involved.

There are some ecological processes in which a species or small group of species plays a unique role. Examples include the nitrogen cycle, in which very specialized bacteria are responsible for critical chemical transformations, or pollination of some flowers that rely on specific pollinators. These relationships can give rise to cascading effects whereby loss of one species will result in the decline or loss of others that in turn lead to the decline of still others. The term *keystone* is applied to a species whose influence in an ecosystem is disproportionately large compared to its numbers or biomass. In restoration, it is especially important to recognize when a keystone species is missing.

**Assessment and Reference Areas**

Successful ecological restoration must begin with identification of missing species and dysfunctional processes. There are two components of this initial investigation. First, one must assess the nature of the original ecosystem. While this will be immediately obvious if the disturbance is relatively minor, it may require quite a bit of investigation if the alteration is severe. For example, it will not be obvious what natural ecosystem was present where one now finds only a cultivated field or a pasture. That will require investigating the soil, topography, historic documents, or interviewing older neighbors who may remember the area before it was badly disturbed. Second, with a good idea of the nature of the original ecosystem, including dominant species, one needs to locate nearby reference areas that are matched as closely as possible to soil, topography, and hydrology. Protected natural areas are ideal references, but there may also be remnant communities tucked away along railroads, field corners, or “back 40s”—remote, uncultivated areas on a ranch or farm—where many of the species have survived. The aim is to determine what species were part of the original community and how the ecosystem functioned. Was it maintained by periodic fires? Did it flood in the spring? Have any important species become extinct or been extirpated from the area? This initial investigative phase may take a few hours or several months, but it is essential to do it well, before beginning restoration.

In ecological restoration, one must strive to work with nature. In most instances, succession will lead an ecosystem toward what occupied the site before it was disturbed. Sometimes, all that is needed is to mitigate the stressors that resulted in the disturbance. Stressors are those perturbations or altered conditions that caused the ecosystem to lose its integrity in the first place. For example, fragmentation of landscapes, coupled with fire-control efforts, were largely responsible for the invasion of woody vegetation into prairies and savannas in the Midwest. Initiating a prescribed fire regime often is all that is needed to restore these ecosystems.

Working with nature means understanding where succession would go if given the opportunity, and assisting the process. It is at least theoretically possible to convert a damaged ecosystem to something quite different from what was there, but to do so requires more effort, and the end result would likely be unstable and require considerable maintenance. For these reasons the preliminary investigation to determine the nature of the original
ecosystem and the stressors that altered it is critical to successful restoration.

Mitigating Stressors

Mitigating stressors is the first step in restoration once the ecosystem has been defined. The factors that result in deterioration of ecosystems can be natural or caused by humans. Most species, and communities as a whole, are adapted to deal with natural phenomena, unless they are extreme or unique. Ordinarily, ecosystems will restore themselves when disturbed by events such as droughts, fire, floods, and the like. While these can create stress in natural communities, they usually are not considered stressors. Human disturbances, on the other hand, often are beyond the evolutionary experience of at least some of the species. Clearing and cultivating land or establishing pastures with periodic mowing, introducing exotic agronomic species, and using fertilizers and herbicides can lead to elimination of a high percentage of the species that constituted the original ecosystem. Even more extreme disturbances, such as draining wetlands, damming streams, removing topsoil, or paving can result in nearly complete removal of, or at least a change in, the species present. These stressors, at least, are usually obvious.

Often stressors are more subtle. One of the most common, for example, is alteration of hydrology. Use of drainage ditches, tiles, or partial damming by highways or railroads, even when culverts are installed, can lead to deterioration of ecosystems. Change in fire regime, usually as a result of fragmentation and fire control, is another stressor that can be overlooked. Invasive species are another growing problem in many ecosystems. Unless stressors can be accurately identified and mitigated, restoration efforts may be futile, or the restored ecosystems will require high maintenance.

Realistic assessment is needed at this juncture before investing more time and money. If the primary stressor is off-site, restoration may be impossible or impractical. Air pollution, altered flood regimes caused by upstream dams or diversions, and climate change are examples of off-site stressors that, if determined to be the primary problem, may make restoration impossible. Local off-site stressors sometimes can be addressed. For example, erosion coming from careless use or development of neighboring property might be addressed by installing artificial ponds or wetlands to intercept the excess water flowing onto the property. Likewise, pesticides or excessive nutrient loading from neighboring feedlots or fields can sometimes be captured in artificial wetlands, allowing the rest of the site to be restored. Better yet, recruiting owners of surrounding properties to join the restoration process and approach it on a broader ecosystem scale is preferable.

Fragments of ecosystems are always more difficult to maintain and are often too small to adequately address primary stressors.

Restoration Practices

The kinds of restoration practices, and the sequence of applying them, should be dictated by the stressors. When the investigation and planning is completed, initial restoration efforts should be directed at mitigating the stressors. Often that is the only restoration required unless key species cannot return without assistance.

A list of the dominant species that populated the original ecosystem is one of the products of the initial investigation. Because plants comprise the majority of structure in most ecosystems, the focus is usually on reintroducing the dominant plant species, filling in lesser species where possible, and allowing other organisms to find their way back. Site preparation is usually required and largely consists of removing or reducing exotic species that would interfere with successful establishment of desired native vegetation.

It is important that sources for species being reintroduced be as local as possible. Ideally, seeds or other propagules (i.e., cuttings, spores) should be gathered from nearby refugia (an area of undisturbed or unaltered habitat) and from similar communities. This provides some insurance that the genotypes being introduced are adapted to local conditions.

Patience

When restoring severely altered ecosystems, it commonly takes many years to see satisfying results. An ecosystem can be destroyed in hours, but it requires years, even decades or centuries, if ever, to regain the full complement of species required to restore the ecological processes that lead to stability. Species that must find their own way back, depending on fragmentation of the landscape and distance to refugia, may not show up for decades. Most badly disturbed ecosystems can never be fully restored and will require a higher level of maintenance than ecosystems with more complete integrity. Nevertheless, with diligence, effort, and patience, most ecosystems can be restored to the point where they can continue to develop on their own with periodic maintenance, such as the use of prescribed fire and removal of invasive species.

Monitoring

An often-neglected aspect of ecosystem restoration is monitoring. A well-developed restoration plan may have benchmarks for different degrees of recovery. Monitoring
is required to make sure the practices being used are moving the ecosystem toward these recovery goals. If not, reevaluation is required to make sure goals are reasonable and the practices proper. Perhaps an important stressor was missed, or maybe a critical species was overlooked. Even after the ecosystem is largely restored, monitoring is required to guide maintenance work. For example, the easiest time to address invasive species is during their first growing season, or certainly within the following year. Monitoring need not be onerous, but it should be thorough and regular.

Group Effort

Ecological restoration lends itself well to family or group activities. Restoring a wetland or prairie in the backyard can be a lifetime family project. Scout troops, garden clubs, or even communities can tackle a vacant plot of abused land and turn it into a vibrant natural community, complete with butterflies, birds, and amphibians that find their way back. Investigating the possibilities leads to discovery of community history, lessons in biology, and engagement with nature that is worthy of any school class. Some argue that the most important thing achieved in ecological restoration is reconnecting people to nature.

The field of restoration ecology, which became recognized as a profession toward the close of the last century, continues to rapidly expand. In addition to scientific journals and reference books, professionals regularly meet to exchange information under the auspices of the Society for Ecological Restoration International. Thousands of scientists now engage in research related to restoration ecology with projects in virtually every kind of ecosystem on Earth. Employment opportunities are largely with companies that provide restoration services and public agencies that manage parks, national forests, and grasslands. The expansion of knowledge about restoration ecology parallels a growing desire to restore the natural ecosystems that have been damaged.

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See also Biodiversity; Brownfield Redevelopment; Community Ecology; Complexity Theory; Dam Removal; Disturbance; Forest Management; Habitat Fragmentation; Human Ecology; Hydrology; Indicator Species; Invasive Species; Keystone Species; Outbreak Species; Plant-Animal Interactions; Refugia; Resilience; Species Reintroduction; Urban Agriculture

FURTHER READING


The benefits people obtain from nature are known as ecosystem services. The concept has come to play important roles in both research and policy since the 1970s. Ecosystem services are being integrated more explicitly into multilateral environmental agreements, national accounting frameworks, corporate strategy, and public policy. Accounting for many different ecosystem services at a sufficiently large scale to promote sustainability, however, remains a future challenge.

People depend on nature for their livelihoods, health, and welfare. The benefits that people get from nature include clean water for drinking, food and recreation from fishing, and wood for building houses and furniture. At the same time, people affect nature in ways that limit its ability to provide these benefits. For instance, forests help to regulate climate by capturing and storing carbon, but each year landholders reduce forests’ ability to provide this service by clearing thousands of hectares of forests in the tropics. Clearing this land releases up to 20 percent of all human emissions of carbon dioxide—a greenhouse gas that contributes to climate change. This is just one example of the ways we are transforming nature and altering the benefits—or ecosystem services—that nature provides to people.

What Are Ecosystem Services?

The most common definition of ecosystem services comes from the United Nations Millennium Ecosystem Assessment (MA): “the benefits people obtain from ecosystems” (MA 2005). Ecosystem services are also referred to as environmental goods and services and nature’s benefits. The services flow from the functions and processes of ecosystems, including the species that make them up (Daily 1997). The MA (2005, 57) identifies four categories of ecosystem services:

1. **provisioning services** that deliver goods such as food, water, timber, and fiber
2. **regulating services** that stabilize climate, moderate risk of flooding and disease, and protect or enhance water quality
3. **cultural services** that offer recreational, aesthetic, educational, and spiritual experiences
4. **supporting services** that underpin the other services, such as photosynthesis and nutrient cycling

Alternative definitions and classifications have been proposed for specific contexts, such as landscape management, environmental accounting, and policy development (Boyd and Bhanzaf 2006; De Groot, Wilson, and Roelof 2002; Fisher, Turner, and Morling 2009; Wallace 2007). In 2010, the Economics of Ecosystems and Biodiversity (TEEB), an international initiative led by the United Nations Environment Programme (UNEP), proposed a definition that differentiates between the services provided by ecosystems and the benefits that humans receive from them: “the direct and indirect contributions of ecosystems to human well-being” (Kumar 2010, 19). The TEEB classification for ecosystem services redefines **supporting services** as ecosystem processes and includes a new category of **habitat services**, which provide nurseries for hunted or fished species, and preserve future options by protecting genetic diversity.

History

The understanding that people rely on nature for their well-being dates to antiquity. Some of the earliest known texts on this topic describe the loss of ecosystem services
and the impact of that loss on society. Chief among these is a description in *Critias*, one of the famous dialogues of the Greek philosopher Plato:

What now remains of the formerly rich land is like the skeleton of a sick man with all the fat and soft earth having wasted away and only the bare framework remaining. . . . Once the land was enriched by yearly rains, which were not lost, as they are now, by flowing from the bare land into the sea. The soil was deep, it absorbed and kept the water . . . and the water that soaked into the hills fed springs and running streams everywhere. Now the abandoned shrines at spots where formerly there were springs attest that our description of the land is true. (Daily 1997, 5–6)

Many scholars trace modern concern about ecosystem services to George Perkins Marsh, a nineteenth-century lawyer, politician, and scholar. Marsh’s 1864 book *Man and Nature* describes a range of services and the consequences of their loss. In the first half of the twentieth century, prominent environmental writers, including Henry Fairfield Osborn Jr., William Vogt, and Aldo Leopold, wrote about the value of ecosystems and wildlife for human welfare. In addition to nature’s value for people, Leopold also espoused a *land ethic* that places a value on the existence of nature itself, without regard to the ways humans use it.

Environmental health became an important issue during the 1960s and 1970s, sparking the first ecological economics research. In 1968, Stanford University ecologist Paul Ehrlich published *The Population Bomb*, which describes human disruption of ecosystems, the costs for society, and possible solutions. In 1970, the Study of Critical Environmental Problems, a group of scientists meeting together at Williams College in Massachusetts, presented the term *environmental services* for the first time, with examples such as fisheries, climate regulation, and flood control. Since then, *ecosystem services* has become the most common term in the scientific literature for benefits that derive from nature.

By the 1980s, research and debate centered on two questions: how much ecosystem function and services depend on biodiversity, and how to measure and value ecosystem services. In 1997, two groups of ecologists and economists synthesized scientific information about ecosystem services and their value (Costanza et al. 1997; Daily 1997). The Millennium Ecosystem Assessment (MA) that began in 2001 involved 1,360 researchers in a four-year global study that evaluated the state of ecosystems and the services they provide. The MA reported that, out of twenty-four ecosystem services tracked over the previous fifty years, fifteen services had seriously declined, four had shown some improvement, and five were generally stable but under threat in some parts of the world (MA 2005). The assessment also revealed that some provisioning services, such as food, had improved at the expense of regulating, supporting, and cultural services. A number of smaller assessments of ecosystem services were undertaken as part of, and following, the MA. One study found that of the ecosystem services delivered by eight broad aquatic and terrestrial habitat types in the United Kingdom and their constituent biodiversity, about 30 percent were declining and others were reduced or degraded (UK National Ecosystem Assessment 2011). A separate international study of the Economics of Ecosystems and Biodiversity (TEEB) assessed the economic benefits of ecosystems and biodiversity and the costs of ecosystem degradation and biodiversity loss (Kumar 2010). As of 2011, several countries, including Brazil and India, were conducting national-level TEEB studies.

**Emerging Policies and Programs**

Spurred in part by international assessments like the MA, various organizations instituted new policies and programs for ecosystem services. In 2005, the United Nations (UN) began to consider establishing a new UN authority that would review ecosystem service research and disseminate conclusions relevant to policy making. As a result, the Intergovernmental Platform for Biodiversity and Ecosystem Services (IPBES) was established in 2010. Also in 2010, the United Nations Framework Convention on Climate Change (UNFCCC) adopted an international framework for reducing greenhouse gas emissions from deforestation and forest degradation (REDD), which provides a way for industrialized countries to offset their emissions by purchasing credits from developing countries that store additional carbon in their forests. The World Bank has developed a partnership for wealth accounting and the valuation of ecosystem services (WAVES) to encourage and enable countries...
to incorporate nature’s value into national accounting frameworks and indicators, such as gross domestic product. In addition, the Convention on Biological Diversity and the Ramsar Convention on Wetlands have explicitly incorporated ecosystem services and ecosystem-based approaches in their principles.

At national and state levels, too, ecosystem service policies and markets are in place. In the United States, the Clean Water Act (1974) protects wetlands and bodies of water to avoid loss of hydrological, cultural, and habitat services. The act provides for mitigating the loss of wetlands and their functions. One provision of the law is that developers who build on wetlands are required to restore or protect an equivalent or greater amount of wetland area to offset the loss of a wetland’s fisheries, recreational opportunities, water purification, and erosion control services, among others. In Australia, too, several state laws require mitigation of damage to ecosystems for the habitat services they provide. Brazil’s Forest Law (1965) requires landowners in the Amazon to maintain 80 percent of their landholdings under forest to preserve the benefits from intact forests.

Civil Society

Various civil society programs have also emerged since the Millennium Ecosystem Assessment, including widely cited work at the World Resources Institute (WRI), Forest Trends, and the Natural Capital Project. In 2008 the WRI released the guide Ecosystem Services: A Guide for Decision Makers, which provides practical guidance on policies that sustain natural capital (Ranganathan et al. 2008). The WRI has also developed the Corporate Ecosystem Services Review (ESR) to help companies identify the ecosystem services they affect and depend on (Hanson et al. 2008). Forest Trends aims to expand the value of nature to society through the creation of markets for ecosystem services. Among other initiatives, Forest Trends has developed a clearinghouse for ecosystem service projects, an ecosystem service project incubator, a program for marine ecosystem services, and a voluntary biodiversity offset framework. The Natural Capital Project is an academic–civil society partnership that develops science and tools to measure, map, and value ecosystem services (see Kareiva et al. 2011); apply these tools with government, business, and civil society partners around the world; and spread the science, tools, and lessons from those efforts worldwide.

Tools

New diagnostic tools for monitoring, measuring, and valuing ecosystem services are constantly being developed. By the end of 2011, more than a dozen ecosystem service assessment tools were in use. Some tools focus on the provision, use, value, and trade-offs of multiple ecosystem services under different resource management scenarios. Two of these are ARtificial Intelligence for Ecosystem Services (ARIES) and Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST). Both ARIES and InVEST use maps to assess the spatial distribution of ecosystem services, as does the Natural Assets Information System (NAIS). Other tools, such as the Wildlife Habitat Benefits Estimation Toolkit and WRI’s coral reef valuation package, estimate the value or amount of ecosystem services without spatial representation. Additional tools assess the benefits from a single ecosystem service (for example, carbon sequestration and storage calculators) or consider changes in ecosystem services in a particular context (for example, benchmarking tools for policies and practices in the mining industry).

Payments for Ecosystem Services

One of the areas of greatest growth since about 2000 is payments for ecosystem services (PES), in which users compensate the providers of services for maintaining or enhancing them (Gómez-Baggethun et al. 2010; Wunder, Engel, and Pagiola 2008). In the developing world, two types of PES mechanisms are prominent: payments for watershed services and payments for climate regulation through the REDD framework. Payments for watershed services, also known as water funds, are a way for downstream water users to pay upstream landholders for the delivery of water services, such as purification and erosion control. One of the first PES programs in Costa Rica is described in a later section, Ecosystem Services in Practice.

Valuation Methods

Economic valuation involves assigning a monetary value to nature’s benefits. Existing market prices often do not reflect ecosystem service values, and special valuation methods based on similar or hypothetical market situations are required. A 2004 white paper published by the World Bank clarifies the aims and uses of economic valuation, outlining four principle objectives: assessing the value of the total flow of benefits from ecosystems, determining the net benefit of an intervention that alters ecosystem conditions, determining how the costs and benefits of ecosystem conservation are distributed, and identifying beneficiaries to ascertain potential funding sources for conservation (Pagiola, von Ritter, and Bishop 2004).

The analytical approach for economic valuation of ecosystem services must be shaped to meet the specific objective. The framework often used to value these
that market mechanisms might have unintended adverse environmental and social consequences, because consumers sometimes act in unexpected ways when they are given external incentives. In addition, these researchers say that the distribution of financial or other benefits is likely to magnify existing economic disparities.

**Ecosystem Services in Practice**

Ecosystem services are increasingly being considered in decision making as policy makers take into account the value of services and how actions affect those values; establish innovative market-based mechanisms that ensure service values are reflected in market transactions; implement policy, organizational, and institutional reform; and develop tools that help people to do all of this quickly and easily. Following are several illustrative examples.

**Costa Rica**

By 1986, forest cover as a share of total land area in Costa Rica had fallen to 32 percent from 63 percent in 1960. This dramatic loss of forest resources prompted the Costa Rican government to pursue a new set of forest conservation and restoration policies, including market-based mechanisms. The passage of Forest Law 7575 in 1996 laid the groundwork for a PES program to pay landowners to maintain and restore forest resources. The program is focused on four ecosystem services generated by forests: watershed protection, biodiversity, landscape beauty, and climate regulation. Program payments to landowners vary by activity. The most common activity is forest protection; landowners agree to forgo use of their forests, transferring their use-rights to the government in exchange for a fixed payment per hectare disbursed over five years. Landowners are also paid by the hectare for reforestation activities or are paid for each tree planted in an agroforestry system, in which crops are interplanted with trees (Fondo Nacional de Financiamiento Forestal n.d.).

**China**

Following devastating droughts and floods in 1997 and 1998, China instituted a series of conservation programs to reduce damage from extreme weather. One of these...
programs, the Sloping Land Conversion Program (SLCP), also known as the Grain to Green Program, is exceptional in its longevity and geographic expanse. Established in 1999, the SLCP restores erosion control and flood mitigation services in twenty-five provinces through grain and cash subsidies to farmers who convert agricultural fields on steep slopes to forests and grasslands. Initial studies have found that the SLCP has increased key ecosystem services while also having a positive effect on household income (Li et al. 2011). Following the SLCP, China has developed Ecosystem Conservation Function Areas (EFCAs), which are newly established zones identified for conservation because of their high levels of biodiversity and ecosystem services, including sediment retention and carbon storage and sequestration. When development is complete, these EFCAs are projected to cover 25 percent of China’s landmass. Land use master plans at the provincial and county level will steer development activities away from these areas, mandating low to no infrastructure development in these zones.

**Belize**

In 1998 and again in 2011, the WRI released a global *Reefs at Risk* report that mapped and analyzed threats to the world’s coral reefs in order to visualize where reefs could be lost. In a complementary series of *Coastal Capital* reports, the WRI produced economic valuations of the reefs and mangroves in the Caribbean to raise awareness of the benefits that people get from these ecosystems and build support for policies that promote their sustainable management. In the case of Belize, the study prompted the government to impose a number of fishing restrictions to protect its coastal resources, including size limits for Nassau groupers caught, a ban on spearfishing within marine protected areas, and a mandate that all fish fillets must be brought to landing sites with a skin patch to enable species identification. Moreover, after the container ship *Westerhaven* ran aground on a reef in January 2009, the Belizian government worked with civil society partners to calculate compensatory ecosystem-related damages that were used in a subsequent court case.

**Wild Bee Pollination**

In 2011, the agribusiness Syngenta assessed the value of wild bee pollination to blueberry farms in Michigan and the added value created by providing foraging habitat for native bees. The purpose of the study was to show that conserving bee populations gave a positive return on investment. The Syngenta valuation was a pilot test for the World Business Council for Sustainable Development’s (WBCSD) *Guide to Corporate Ecosystem Valuation*, which is designed to improve companies’ understanding of the benefits and value of ecosystem services. The guide explores tools and methods for ecosystem valuation to manage risks and opportunities related to ecosystem services. The Syngenta study determined that Michigan blueberry farmers received $12 million annually from wild bee pollination of their crops. The company has since launched Operation Pollinator to support conservation programs that growers can integrate into their farms (WBCSD and IUCN 2011).

**Challenges and Future Directions**

The ongoing challenge with ecosystem services is to build on the many new tools and approaches. Many decisions made by individuals, communities, corporations, and governments still do not reflect the value of nature’s benefits to people. There are critical gaps in both the scientific basis of ecosystem services and policy and finance mechanisms.

The relationships between ecosystem services and biodiversity, human well-being, and poverty remain unclear. Many studies do not address multiple ecosystem services and their interactions or the consequences that changes in ecosystem services in one place have on distant places (Seppelt et al. 2011). In addition, few systematic studies reveal the effects of different policy instruments on ecosystem services and the people that provide and benefit from them. New global research programs, however, are taking up these challenges.

Ecosystem service programs and policies are often piecemeal and poorly coordinated. In many cases, they are based on unproven assumptions or sparse information (Carpenter et al. 2009). In addition, disproportionately few programs and policies are focused on dryland, grassland, subterranean, or marine ecosystems. In a 2009 study, the Bridgespan Group observed that 73 percent of the ecosystem service projects they looked at focused on forests and wetlands (Searle and Cox 2009). Moreover, many existing policies and programs address one or two ecosystem services, rather than multiple services. Last, relatively few nations have adopted ecosystem service policies, although the number is growing. An important future challenge is to take multiple ecosystem services into account at a large enough scale to ensure that the environment provides the many benefits society needs to prosper.

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FURTHER READING


Edge Effects

Changes in ecology near the interface of two habitat patches are often referred to as “edge effects.” Edges were once considered positive features in natural landscapes, but the realization that they could be zones of high mortality made them a focus of major conservation concern. Modern attempts to incorporate edge effects into resource management plans focus on understanding species-specific responses in complex and dynamic landscapes.

Habitat edges are landscape features formed at the interface of two different patches of habitat. Such edges support distinct ecological conditions and are ubiquitous in both natural and human-altered landscapes. Edges and their effects on biota were first studied in 1907 by the US botanist Frederic Clements, who noted the gradual change in plant communities within the transition zone from one ecotype to another. Clements called these transition zones ecotones.

Evolution of Perspectives about Edges

Throughout much of the twentieth century, studies of edge characteristics focused on natural edges, especially where habitats were naturally patchy and interspersed. (See figure 1 on page 119.) During that era, botanists like Clements focused on the gradual changes between different habitat zones, while animal ecologists focused on the different groups of animals that were associated with these edges. In the 1930s, the US naturalist Aldo Leopold noted that many species, including game species, often congregated near edges, and for many years, edge creation was a popular tool to enhance habitat for wildlife. Consequently, through the early 1970s, the term edge effect generally referred to the increase in abundance and diversity of species near habitat edges, and overall, edges were considered to be positive features in the environment.

The view of edges and their role in the landscape shifted dramatically in the late 1970s as scientists realized that edges created by forest fragmentation, rather than by natural processes, might actually reduce, not increase, habitat quality. The landmark work of the conservation ecologist J. Edward Gates demonstrated that some bird species were experiencing higher rates of predation and parasitism near forest edges. Because birds had been known since the 1930s to have higher abundance and diversity along forest edges, this led to the idea that edges might be acting as an “ecological trap,” attracting individuals by virtue of increased resources and shelter, but then exposing them to higher risks of mortality from predators that also frequented edges. The US ecologist David Wilcove then suggested that such ecological traps, in concert with increasing creation of edges through habitat fragmentation, could be a contributing factor in the decline of North American songbirds. That research, along with other studies suggesting that habitat edges could exacerbate the already extreme consequences of habitat loss, caused a paradigm shift in how ecologists think about edges. Indeed, by the mid-1980s, the term edge effect came to be associated with increased mortality near habitat edges, and edges were more commonly seen as negative features in the environment.

This paradigm shift led to an explosion of research documenting the impacts of habitat edges on the ecology of many species in many different landscapes. Studies documenting the ecological effects of edges attended to diverse issues, including edge-induced changes in predator densities, parasitism rates, species abundances, and community composition. At the same time, a series of articles emerged laying a foundation for understanding
Fragmentation experiments were also initiated, including the groundbreaking Biological Dynamics of Forest Fragments Project (BDFFP) in the Brazilian Amazon. This large-scale ecological experiment involved the purposeful creation of replicated forest fragments of different sizes in a region of the Amazon already slated for development. The BDFFP fragments were created in 1979 and have been monitored ever since, with many studies focusing on the ecological changes taking place near patch edges. By the late 1990s, numerous scientific articles drew on data from the BDFFP and other large-scale experimental studies of fragmentation in forests and other habitats to quantify many different types of edge effects. These studies spanned the globe, involving diverse landscapes, edge types (e.g., grassland edges, desert edges, sea grass edges), and species. In the 1990s and early 2000s, synthetic review papers (those that summarize findings from hundreds of research papers), the ecological mechanisms by which edge effects occur in patchy landscapes. Researchers identified four fundamental mechanisms that could shift distributions of individual species and ultimately lead to a new community structure at the edge: (1) ecological flows of both organisms and nonliving materials between adjacent patches; (2) increased access to resources divided between adjacent patches (or “cross-boundary subsidies”)—for example, nesting sites in forests and forage sites in meadows; (3) resource “mapping,” whereby a shift in the distribution of one species causes an accompanying shift in the distributions of species that use it as a resource; and (4) altered species interactions, such as decreased pollination or increased predation. All of these mechanisms can interact with each other, highlighting the complex forces that can influence species near habitat edges.

In addition to hundreds of studies designed to document and quantify edge effects, several large-scale habitat fragmentation experiments were also initiated, including the groundbreaking Biological Dynamics of Forest Fragments Project (BDFFP) in the Brazilian Amazon. This large-scale ecological experiment involved the purposeful creation of replicated forest fragments of different sizes in a region of the Amazon already slated for development. The BDFFP fragments were created in 1979 and have been monitored ever since, with many studies focusing on the ecological changes taking place near patch edges. By the late 1990s, numerous scientific articles drew on data from the BDFFP and other large-scale experimental studies of fragmentation in forests and other habitats to quantify many different types of edge effects. These studies spanned the globe, involving diverse landscapes, edge types (e.g., grassland edges, desert edges, sea grass edges), and species. In the 1990s and early 2000s, synthetic review papers (those that summarize findings from hundreds of research papers),
highlighted the high degree of variability in edge responses. The result was a further change in perspective, from “edge effects are bad” to “edge effects are highly variable and idiosyncratic.”

**Factors Influencing Edge Effects**

That edge effects are variable and hard to make sense of is perhaps not surprising given the diverse combinations of species and conditions created when different types of habitat abut. Yet the variability has turned out to be less intractable than originally imagined, and many of the main sources of variation have been identified. One of the major realizations was that the type of so-called matrix habitat in which a patch is embedded (e.g., grassland, agriculture, mining, urban development) would surely alter the type of edge effects observed. This concept, known as *patch context*, has proved critical as a means of accounting for much of the observed variability within the edge literature. In fact, if the quality of the adjacent habitat is considered, then some basic predictions can be made about the types of edge responses expected based on the resource needs of most species. (See figure 2.) For instance, species can be expected to avoid edges with adjacent habitat that contains inferior resources, but they show no edge response if resources are found equally in both habitats. On the other hand, in cases where species’ resources are divided between habitats such that the edge offers “cross-boundary subsidies,” then a positive edge response can be expected. This simple conceptual model has been successful in predicting the direction of observed edge responses for birds, mammals, butterflies, and plants.

Several other factors also influence the nature and magnitude of edge effects, including the orientation of the edge (especially north versus south) and its structural contrast. Finally, different species simply respond differently to edges, and those differences may be due to species’ traits, such as whether they are specialists or generalists.

**Figure 2. Predicted Edge Responses**

![Predicted Edge Responses Diagram](image)


Species’ edge responses are likely to be positive, negative, or neutral depending on the relative quality of the two adjacent habitats and how resources are distributed between the habitats. Expected edge responses are transitional (a), neutral (b), or positive (c and d). Lower habitat quality is indicated by a white box; habitats of higher or equal quality are shaded. Either the same resources are available in both habitats (supplementary), or different resources are divided between habitats (complementary).
their position in the food chain, their vulnerability to predation, and perhaps other factors. Among the generalizations that have emerged from the scientific literature are that predators and habitat generalists are more likely to prefer edges, whereas habitat specialists are more likely to avoid edges. These generalizations, while never being perfectly accurate, have proven to be good rules of thumb when on-the-ground studies are lacking. It is critical to remember, however, that many species will exhibit no response to edges. Furthermore, even within species, edge responses may vary depending on the type of edge encountered.

Implications for Conservation Efforts

Given that edge effects are seen as complex and arising from multiple, conflicting forces, the edge effect concept has limited utility as a tool for conservation efforts. Nevertheless, resource managers have long tried to consider edge effects when doing landscape planning. One of the first attempts to incorporate edge dynamics into conservation planning was the development of the core area model by William Laurance, a research scientist working on the BDFFP project. (See figure 3, part a.) This approach required conservation planners to estimate the depth of edge effects (i.e., how far the changes in density or species diversity penetrated into a habitat patch) and then identify an area of edge influence along the boundaries of each patch. These zones were then effectively clipped from the patches so that only the “core area” of the patch, which was argued to be insulated from edge effects, would be considered when making management decisions. Unfortunately, this approach has many limitations. First, many landscapes are so extensively fragmented that the only type of habitat left is effectively all edge. Second, as edge habitats come to dominate a landscape, it is risky to ignore them because edges can be both zones of high mortality and also the preferred habitat of many species. An expansion of the core area model, called the effective area model, allowed density estimates to be extrapolated throughout an entire patch based on the response of each species to each specific edge type in the landscape. This model, however, is more difficult to apply because managers usually lack information on edge responses that are specific to particular combinations of species and edge types. Another approach to managing for the potentially negative impacts of edges was the idea of multiuse buffer zones. (See figure 3b.) This idea, championed by the US conservation biologists Reed Noss and Michael Soule, imagined core areas of protected habitat buffered by zones featuring similar habitat structure (thus decreasing edge contrast) in which human uses were more broadly permitted. This approach has been implemented worldwide in many conservation plans that incorporate core areas surrounded by buffers, often with habitat corridors to connect isolated patches.

Although edge effects remain a difficult dynamic to grapple with in landscapes featuring multiple edge types and complex ecological communities, scientists have made great strides in determining how to leverage information about edges to make better management decisions. For example, efforts to identify edges from remotely sensed imagery can help quantify the dynamic nature of patch structure and track structural changes in edges over time. This is especially vital as climate change can cause abrupt changes to landscape structure, as has been shown by rapid migration of treelines in some systems. Likewise, efforts to understand how edges influence animal movement (though infrequently undertaken because of complexity and cost) have proved essential in efforts to quantify landscape isolation. Nevertheless, many challenges remain. For example, there has been little progress in incorporating the complex structure of edges into landscape-scale models of species’ distributions and community dynamics. Many studies continue to focus on unrealistic binary landscapes, even though most landscapes are a complex mosaic of patches that create multiple types of edges. Moreover, most models seeking to quantify edge effects assume straight edges, thereby ignoring the influence of complex edge geometry. Efforts continue to allow scientists to understand why certain species show strong responses to edges while other species seem to ignore them (and why edge responses can vary substantially even within species). Finally, much more work needs to be done to
Figure 3. Incorporating Edges into Conservation Planning

Source: Leslie Ries.

For any patch that might be targeted for conservation, managers could employ the core area model (a) where they determine how deep the zone of edge influence is (black area) and consider only the interior "core" zone, or they could design the landscape so the targeted patch is surrounded by one or more buffers (b) that have less intensive uses and so may exert weaker edge effects.

understand the complex interactions that occur among species near edges. A promising new research direction is integrating edge dynamics with spatial food web theory. (Food web theory seeks to describe feeding relationships among species.) These continuing efforts to understand how edges impact the abundance and distribution of organisms will enable more effective conservation in landscapes that are increasingly fragmented.

See also Biodiversity; Biodiversity Hotspots; Biogeography; Biological Corridors; Boundary Ecotones; Buffers; Community Ecology; Complexity Theory; Ecosystem Services; Food Webs; Habitat Fragmentation; Plant-Animal Interactions; Population Dynamics; Refugia; Regime Shifts

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Further Reading


The term eutrophication has come to describe the enrichment of waters with nutrients (mainly nitrogen and phosphorus) from human activities and their associated impacts. The process results in excessive algal growth, loss of species diversity, creation of dead zones, and consequent effects through the entire food web. Eutrophication can be reversed, but restoration of a eutrophied ecosystem is difficult when nutrient removal is focused only on a single nutrient.

The term eutrophication has had several formal definitions through its history. In the early twentieth century ecologists applied the concept of eutrophication mostly to the natural aging of lakes. By the mid-twentieth century eutrophication came to mean the enrichment of waters with nutrients from human activities and their associated impacts—in other words, nutrient pollution.

The complex responses of an ecosystem to eutrophication depend on the nutrient load (the amount of nutrients), the forms of the nutrients, the type of ecosystem, and the extent to which the system is stressed by other factors. Systems such as rivers, well-flushed estuaries, and wave-swept marine coasts are less susceptible to the effects of nutrient pollution than are lakes, coastal lagoons, and other systems that do not have natural flushing. The more retentive the system, the more opportunities occur for nutrients that enter the system to be recycled, prolonging their effects. Multiple stressors, including habitat change, overfishing, and climate variability and change also synergistically interact with nutrient loading, affecting the ways nutrients are taken up in the ecosystem, how food webs function, and ultimately the extent to which a system may respond to nutrient loads or reductions.

Effects on Ecosystems

One of the first ecosystem responses to nutrient enrichment is an increase in algae growth and accumulation. In turn, the turbidity (cloudiness) of the water increases and thus the amount of light reaching the bottom decreases, a condition that results in stress to, or the death of, submerged aquatic vegetation (SAV). When these algae and SAV die, bacteria and fungi decompose them, consuming oxygen in the process, and eventually reducing or depleting the oxygen in the water if the levels of organic matter in the system are high enough. These areas are considered hypoxic if oxygen is significantly reduced or anoxic if the oxygen is depleted completely; these areas are called dead zones. The US scientist Robert Diaz and his Swedish colleague Rutger Rosenberg (2008) have identified more than five hundred coastal regions globally, totaling more than 24 million hectares, affected by dead zones. Dead zones have large ecological and economic costs. The dead zone of the Chesapeake Bay in the eastern United States, for example, is responsible for the loss of tens of thousands of tonnes of fish annually, while the dead zone in the Gulf of Mexico is responsible for a loss of fish about three times that of the Chesapeake due to its much larger size and the larger amounts of nutrients exported to the Gulf from the Mississippi River.

Dead zones are one of the most recognized and severe effects of eutrophication, but there are many others. Species of algae well suited for nutrient uptake and growth in low-nutrient conditions may be replaced by those better at competing in high-nutrient conditions. This often results in a system shifting from a broad diversity of algae species to fewer and more harmful species. Harmful algae are those that produce toxins that directly kill fish or shellfish, affect human consumers, or disrupt the normal trophic (nutritional) pathways supporting food webs. Among
the algae considered harmful are toxic cyanobacteria and many dinoflagellates that form so-called red tides, named for the often reddish or pinkish algal bloom.

The toxins produced by harmful algae are potent and have a diverse array of effects. About 60,000 people are exposed to algal toxins annually in the United States, with effects ranging from mild to severe. One of the most common toxic syndromes from red tides is that of paralytic shellfish poisoning, which can cause respiratory distress or paralysis in people who eat mussels or other shellfish that have consumed the toxic dinoflagellates. In the natural environment, toxins from dinoflagellates affect the food web in many ways, from reproductive failure in whales to embryonic deformities in oysters. Some freshwater cyanobacteria produce toxins that have been associated with cattle, dog, and bird deaths as well as human illnesses and death. As reviewed by Geoffrey Codd (2005), a scientist from the United Kingdom, cyanobacteria mostly affect people through skin and respiratory irritation following recreational exposure to affected water, but can also have far more serious effects, including deaths, when people are exposed to these toxins directly through ingestion. These toxins have been shown to promote liver tumors in rats and may contribute to increased rates of liver cancer in people who live near and consume water from nutrient rich lakes (Grosse et al. 2006). In the natural environment, toxins from other species of dinoflagellates affect the food web in many ways, from reproductive failure in whales to embryonic deformities in oysters.

Eutrophication has also been associated with increased infectious diseases in the aquatic food web (Johnson et al. 2010). Nutrient pollution is also associated with elevated levels of bacteria or pathogens and this has been the cause of beach closures and swimming advisories (along coasts and in fresh- or saltwater) throughout many parts of the world, especially in the United States. In some parts of the world, evidence suggests a link between eutrophication and cholera, and other research points to links with other diseases, as described by the US scientist Pieter Johnson and colleagues (2010). Deformities in amphibians, such as missing, misshapen, and extra limbs, have been associated with parasites that become more numerous with eutrophication.

Macroalgae, or seaweeds, may also develop in response to eutrophication. These plants can form visible scum or mats in the water, either floating or on the bottom. A massive bloom of such macroalgae affected the sailing events at the 2008 Olympic Games in Qingdao, China, for example. More than 1 million tonnes of algae had to be cleaned up before the competition. Such overgrowth is also thought to be contributing to coral reef decline. Brian Lapointe (1997), a US biologist, has documented such changes in nutrient-rich, coral-rich areas in Florida and Jamaica. Various investigators, for example J. D. Voss and L. L. Richardson (2006), have experimentally demonstrated disease outbreaks in some coral reef systems to be associated with nutrient enrichment.

Eutrophication also results in changes in the structure of an affected ecosystem's food web. Increases in jellyfish populations have been observed in many dead zones—jellyfish have been showing explosive growth in the Mediterranean, the Gulf of Mexico, the Black and Caspian seas, off the northeastern coast of the United States, and in Asian coastal waters. Jellyfish may do well in these areas because they can better tolerate low oxygen conditions than can most fish. Shifts in food webs away from crustacean zooplankton to jellyfish negatively affect fish populations that normally feed on zooplankton. The range of effects of eutrophication on all aspects of the aquatic ecosystem is broad, affecting humans and wildlife, ecology, and economies.

Causes

Although eutrophication is occurring globally, nutrient export is far from evenly distributed either regionally or around the globe. Human sewage — and increases in sewage because of expanding human population in many parts of the world—is a major component of nutrient discharge to waterways, with the amount and forms of nutrients in the effluent depending on the sophistication of the treatment. Eutrophication is also attributed to the large increase in the use of agricultural chemical fertilizers that began in the 1950s and is projected to continue to escalate (Smil 2001). Half of all nitrogen fertilizers used in the world have been applied since 1950, and human population now accounts for half of the nitrogen exported from rivers. China illustrates this trend well. In the 1970s, China used less than 5 million tonnes of nitrogen fertilizer annually, but since 2010 it has used more than 20 million tonnes per year, which has led to significant increased nitrogen pollution of its coastal waters (Glibert 2006).

Significant nitrogen and phosphorus waste also arises from animal husbandry, especially intensive factory farms. Unlike human sewage, wastes from animal operations are not treated in treatment plants; instead, farmers often either hold those wastes in lagoons or spray them on nearby lands where they can run off into adjacent waters or percolate into groundwater. The release of ammonia from animal wastes on farms is another source of nitrogen that makes its way into the atmosphere and then can be deposited on land some distance from its source. Intensive animal operations also include aquaculture farming, a practice that creates nutrient waste in water directly from feed and indirectly as a result of the
farmed fish’s metabolism changing the nutrient concentrations and forms of nutrients available in the water.

Additional nutrient pollution comes from human energy production and consumption. The combustion of fossil fuels in power plants and in individual vehicles results in nitrogen oxide (NOx) production, which is discharged into the atmosphere where it contributes to smog and acid rain. In many estuarine and coastal waters, the atmosphere may contribute up to 40 percent of the total inorganic and organic nitrogen inputs and recent estimates suggest that 30 percent of oceanic external nitrogen supply may be coming from the atmosphere (Duze et al. 2008).

Phosphorus is the other major source of nutrient pollution, and historically, phosphate detergents have been a large contributor. Phosphorus makes its way into waters from both household and industrial use. Use of phosphorus in detergents, however, has been declining in most parts of the developed world.

The total loads of nitrogen and phosphorus are important, but so too are their different forms. Different types of algae thrive on different forms of the same nutrient. For example, some types of harmful algae grow faster or produce more toxins when they grow on urea than when they grow on nitrate, although both are forms of nitrogen.

Adding to the complexity is the fact that nitrogen and phosphorus have different effects in fresh and marine waters. Phosphorus is generally considered the limiting nutrient for freshwater, while nitrogen often limits primary production in estuaries and coastal waters. (The limiting nutrient is the one that is in least supply relative to the needs of the organisms.) Some natural resource managers believe that controlling the limiting nutrient will control unwanted algae growth and thereby control eutrophication. In general, control of phosphorus is less expensive than control of nitrogen, and there have been numerous “successes” in ecosystem restoration following phosphorus load reduction. (See the section “Prospects for Recovery,” below.) This would seem to support the theory that limiting the availability of one nutrient will limit the amount of algae growth, and that an excess of any other nutrients will have no ecosystem effect because they will not be able to be assimilated. This simplistic view has been challenged, however, for two reasons: excess nutrients are often transported downstream, where they contribute to eutrophication spatially displaced from the original site of discharge; and disproportionate nutrient availability may have ecosystem effects on the larger food web.

When one nutrient is controlled relative to another, just as when one nutrient is added disproportionately to another, unintended ecological changes can occur, such as increased growth of non-native (“invasive”) species that are opportunistic under such conditions. Different types of organisms have different needs for nitrogen and phosphorus, and thus it is not surprising that the prevalence of different species changes when the proportions of these nutrients change in the environment. While individual species and processes may respond to single nutrients, the relative proportion of nitrogen and phosphorus collectively alters metabolism, species composition, and food webs. Nutrients collectively thus exert a strong regulatory control on ecosystem structure as a whole. The nutrient balance, or stoichiometry, has many consequences for which species are successful under different nutrient condition, as the research by US freshwater ecologists Robert W. Sterner and James J. Elser (2002) on stoichiometry in aquatic systems has shown. Studies by the US scientist Patricia Glibert and colleagues (2011) have shown that altered nutrient ratios have affected the food web structure of a number of estuaries in the United States and Europe, from plankton to bivalves to fish.

Prospects for Recovery

Some ecosystems have shown significant recovery following nutrient removal. One example is Lake Washington, in Washington State, which was highly polluted in the 1960s. The limnologist W. T. Edmondson (1970) led efforts to divert sewage from the lake, and once controls were enacted, algal biomass, including filamentous cyanobacterial scum, declined, and the food web began to recover to earlier conditions. The reduction of phosphorus loading to Lake Erie, in the 1970s, is also hailed as a success, because a dramatic reduction in nuisance algal blooms followed almost immediately from the reduction in phosphorus. Another example of recovery from nutrient pollution occurred in the 1990s in the Black Sea. After the collapse of the Soviet Union, subsidies for fertilizer use in Russia and Ukraine were reduced, and as a result, there was less nutrient runoff, and dead zones and algal blooms in the Black Sea declined in intensity.

Not all events called successes are true successes, however. Eutrophication is a process, and so too is recovery following nutrient reduction. Although dramatic changes in algal biomass and food webs may occur following reduction in phosphorus loading, the ecosystems do not necessarily return to their pre-eutrophied condition. When one nutrient is controlled without controlling the overall nutrient balance, the nutrient in excess is exported downstream or offshore where it can contribute to blooms displaced in time and space from the source of the pollution. To fully address the challenges of eutrophication, ecologists and ecosystem managers will need to reduce both nitrogen and phosphorus loads. Dilution does not solve the problem—it just displaces it.

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See also Dam Removal; Fisheries Management; Food Webs; Human Ecology; Invasive Species; Microbial Ecosystem Processes; Nitrogen Saturation; Nutrient and Biogeochemical Cycling; Pollution, Nonpoint Source; Pollution, Point Source; Resilience

**FURTHER READING**


Extreme Episodic Events

Weather, volcanism, evolution of new diseases, and human actions are among the factors that impose changes, of varying degrees of extremity, on organisms and the environment. Such extreme events also affect humans directly and indirectly. It is challenging to recognize, study, and predict extreme events. The human-induced rise of atmospheric carbon dioxide is probably the most important extreme event in human history.

Major challenges exist in recognizing extreme events and their causal chains. Some, but not all, extreme events are recognizable, and then to varying degrees. Events of long duration, particularly the positive extremes, are not readily apparent while they are taking place, such as the long intervals of equable climate in the Medieval Warm Period (950–1250 CE) or at the peak of the Roman Empire. Some extreme events, such as earthquakes and hurricanes, originate without essential involvement of humans or other biological systems (biota). Others are principally biological in origin, such as the evolution of new diseases, or insect outbreaks that defoliate or kill trees. Whatever the ultimate causation may be, all organisms are susceptible to negative extremes that exceed their ability to acclimate at low cost in metabolic effort or via minimal losses in Darwinian fitness (ultimate reproductive output). Overdesign (by natural selection) to cope with the rarest extremes is too costly, for organisms just as for engineering projects.

Biological Extreme Events

Biological extreme events (BEEs) of numerous types have long been known, but most have received only anecdotal study. There are no rigorous theoretical frameworks to categorize them, quantify them, and, consequently, to study them, and to devise means of avoidance or amelioration (or, in some cases, to comprehend their inevitability), although some frameworks are in development (Gutschick and BassiriRad 2003). BEEs may be analyzed for their distribution over time and space, and cause and effect.

Biological extremes occur naturally over wide ranges of space and time, and most are distributed over a continuous range of extremity (exceedance). Some events involve humans directly in their origination or progress. Human involvement may be proximate, or direct, as in the introduction of exotic species, such as the rabbits that have devastated Australian vegetation. In other cases, human involvement may be more remote but perhaps no less causal, as when the use of fossil fuels and deforestation add greenhouse gases to the air. The consequent climate change appears to be affecting precipitation regimes globally, and the increase of carbon dioxide (CO₂) itself appears poised to have a major impact on coral reefs.

Extreme events may affect humans directly, influencing health, agricultural productivity, water supplies, and other services. Some events affect nonhuman biological systems directly and humans more indirectly. The indirect impacts may nonetheless be significant, as, for example, when key ecosystem services, including flood control by intact vegetation, or pest and disease control by birds, bats, and other insects, are altered.

Some BEEs may appear to occur at single points in time, such as the dislodgement of tidal-zone organisms by rare waves. The US biologists Steven D. Gaines and Mark W. Denny offered a mechanistic understanding of such events and some rigorous formulation of the statistics of the environmental fluctuations that drive them. Other BEEs unfold over extended intervals and may involve many environmental variables. One example is the extensive death of coniferous trees in western North America from 2000 to 2003, driven by the combination...
of warm winter and the surprisingly rare combination of summer heat and drought.

The limitations of knowledge of how organisms respond to environmental extremes are compounded by the limited knowledge of how the environmental extremes occur over time—that is, their statistical distribution. The uncertainties are likely to increase with ongoing changes in land use and in the emission of greenhouse gases and a variety of pollutants. Some research groups are performing experiments on small scales, aimed at modeling current and future extremes. The experiments range can be very expensive, such as those in which CO₂ is added to open air in a limited area (free-air CO₂ enrichment, or FACE). Researchers also attempt to capture extreme events in progress, as well as to simulate them, though with imperfect knowledge of what combinations of factors constitute extreme events.

Abiotic Extremes

While most knowledge of BEEs is case-specific or anecdotal, there is a vast body of knowledge on abiotic extremes, both as observations and as models such as climate models (general circulation models, or GCMs). Patterns of increases in extremes of rainfall and of attendant floods and droughts are clear over large regions of the globe. The implications for agriculture alone are sobering. Satellite- and ground-based observations have extensively documented shifts in seasonality in temperature and precipitation, extending to entire hemispheres. The effects of these shifts on timing (phenology) of plant and animal activity, including growing offsets between flowering plants and pollinators as well as greater activity of plant-eating insects and of insects that spread human diseases, have also been documented. Nonetheless, interpretation of the observed effects remains limited. In addition, data indicate that severe storms on land and hurricanes may be increasing, although the observations are insufficient in number. Moreover, it is not possible to attribute any changes in extremes to human actions such as increasing CO₂ content of the air. It is feasible only to estimate the risk ratio—that is, to compare the probability that a given event would occur with human effects to the probability of its occurrence without human effects. Such assessments rely on the accuracy of models such as GCMs; their accuracy is improving but is far from sufficient, particularly for small regions and long time intervals.

Rising CO₂ Levels

The most significant BEE in human history is likely to be the rise in atmospheric CO₂. This event is global in extent and long-lasting (CO₂ has a mixture of residence times in air, but the shortest is about 150 years), affecting temperature and precipitation, and altering the performance of plants of all species (and all their dependent species, including humans) in various ways. Moreover, we may expect that very few plant species, wild or cropped, possess in their populations the genetic variants that respond positively or adaptively to the changes in climate and in CO₂ directly. As a result, some may disappear or radically change their performance. Levels of CO₂ as high as those seen in the beginning of the twenty-first century have not occurred for some 20 million years, over which time variant genes (alleles) that are not immediately useful can be lost by a process called genetic drift.

The modern rise in CO₂ levels is not the first time that the Earth has seen such a phenomenon. An increase in CO₂ also took place at the beginning of the Proterozoic eon, 2.2 billion years ago. Cyanobacteria were the first organisms to develop the form of photosynthesis that generates free oxygen. The free oxygen oxidized methane that dominated in the air, and the CO₂ that was produced by oxidation was a much weaker greenhouse gas. With the sun only 70 percent as bright as it is today, the cooling effect led to glaciation over most of the Earth. Only continuing volcanic output of CO₂ eventually restored a strong greenhouse effect after about 50 million years.

Organisms, communities, and ecosystems have persisted through other major changes in environmental conditions in the distant past, albeit with massive population reductions. Atmospheric CO₂ levels have periodically spiked during most of Earth’s history. There is evidence that, during the spikes in the Cretaceous period (145 million–65 million years ago), coral animals became unable to build solid reefs but persisted as free-swimming individual plankton (Medina et al. 2006). The effects on reef-reliant fishes are not known, while they are projected to recur by the end of this century. Surface temperatures have likewise spiked higher, as in the Paleocene-Eocene Thermal Maximum of 56 million years ago (5°C–6°C higher), or in the Cretaceous period; the latter era has been dubbed the Saurian Sauna, for the conditions that the dinosaurs experienced. The human population may have been severely reduced by the eruption of the Toba volcano in Indonesia seventy thousand years ago. Humans as a species survived this extreme event and other events and are likely to survive future cataclysms, but the potential loss of the vast majority of the population would certainly rank as worse than all wars, plagues, and natural disasters.

Outlook

The human-imposed changes in environmental conditions in modern times demonstrate unique characteristics that make it challenging to predict the outcome based on historical records. One is their unprecedented rate. Until now, the most rapid CO₂ changes have been near-doublings
over about five thousand years during ice age transitions. In the present cycle, CO$_2$ is projected to double in less than one hundred years. Second, the multiplicity of changes—elevated CO$_2$, widespread releases of “reactive nitrogen” (nitrate pollution in water from fertilizer use, nitrogen oxides in air from combustion), ozone depletion in the stratosphere, ozone increases at ground level, land-use changes—has no equal. Organisms, including humans, have never had to acclimate individually or adapt genetically in populations to so many changes so fast. Finally, humans have reworked the surface of the Earth, primarily through agriculture but also as a result of species extinctions from habitat change and hunting. Humans now command about 70 percent of the net productivity of land that is supported by photosynthesis and plant growth. For all these reasons, the effects of the current extreme events are very difficult to assess but appear to constitute a mass extinction event that we can avert, with sustained great effort.

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See also Agricultural Intensification; Carrying Capacity; Complexity Theory; Disturbance; Ecological Forecasting; Ecosystem Services; Fire Management; Global Climate Change; Human Ecology; Keystone Species; Nitrogen Saturation; Ocean Acidification—Management; Resilience; Shifting Baselines Syndrome

FURTHER READING


Fencing fragments landscapes and reduces the ability of animal species to shift ranges in times of nutritional need or stress. Migrant large mammals are particularly affected. Controlling animal diseases and human conflict with wildlife are two of the primary reasons for the erection of fences. Removal or realignment of long fences requires the development of alternative conservation strategies such as transfrontier conservation and biofences.

Fencing and other forms of human-constructed separation barriers—such as road, walls, pipelines, and artificial water bodies—fragment landscapes, create habitat patchiness, and form isolated wildlife enclaves. Barriers protect humans and their infrastructure from natural risk and disturbance. Like medieval castles, fencing and other deliberately erected barriers stop humans or animals and diseases from getting into or out of some predefined space. Since the dawn of the agricultural age humans have protected their lives and livestock from the real and perceived threats that emanate from nature, especially predation (on crops or domestic animals and occasionally people) and cross-species disease transmission. Within the safety of more permanent and larger settlements, farmers could grow grain and raise domesticated herds. As this process expanded, the conflict with nature widened.

Animal control fences that prevent migrant or nomadic animal movement across landscapes may endanger the last of the spectacular mass migrations of large wild mammals. Fences and other barriers create hard, impassable edges, expose animals to dangerous areas where they may be hunted, and interfere with the ecological functioning of a wildlife area. David Wilcove (2008), a US ecologist and evolutionary biologist with an abiding interest in animal migration, has illustrated how humans have designed fences for the sole purpose of stopping animal movement or have incidentally constructed barriers such as railway lines. François Lamarque, a French program officer with the International Foundation for the Conservation of Wildlife, and his colleagues have reviewed various forms of human-wildlife conflict and possible mitigation methods that include fencing (Lamarque et al. 2009).

Barriers also increase the spatial distance between related populations of animals and plants. Both plants and animal species show significant variations on the other sides of divides, leading to changes in the ecosystem and extinction of some localized species.

Benefits and Costs

Increasing human-caused habitat loss has led governments to create protected areas for animals worldwide. Removing fences to allow cross-border movement of wildlife has not made all park fences redundant, however. In eastern and southern Africa, governments are fully or partially sealing off some protected areas for a number of reasons. When a large human population adjoins the park, diseases can flow in both directions. Fences prevent poaching and allow parks and reserves (private and state owned) to legally import and safely enclose dangerous animals that tourists wish to see.

Local people who suffer livestock and crop losses due to wild animals often push for increased fencing. Human-wildlife conflict may become so contentious that fencing offers a respite to continuously fractious relations between wildlife parks and people. Matt Hayward, an Australian ecologist who has worked extensively in Africa, and his colleague Graeme Kerley, a South African expert in African wildlife ecology, have published a comprehensive list of the costs and benefits of fencing at all scales of
impacts (Hayward and Kerley 2009). Hayward has further emphasized the importance of fencing for conservation terms: “The IUCN [International Union for Conservation of Nature] lists ten key threatening processes in its Red List of Threatened Species, and eight of these can be managed via the use of conservation fencing” (Ferguson and Hanks 2010, 168). Fencing can protect wildlife corridors that span the inhospitable terrain between two or more protected areas.

Sometimes fencing can be a victim of its own success. Kruger National Park in South Africa was entirely fenced in the 1970s. Ian Whyte, a South African elephant specialist, and Salomon Joubert, an ecologist and former director of Kruger Park, have found noticeable opposing conservation effects of the fence. Blue wildebeest numbers declined 87 percent because fencing severed an external migration route. Fencing increased the number of elephants, however, because they lacked dispersal opportunities, leading to overabundance and population compression. Both these authors, on balance, believe the benefits of the fence—its role as a disease barrier and protection from human encroachment—outweigh its costs (Whyte and Joubert 2010, 137–143).

In most of the world humans do not fence themselves in to keep nature at bay. Instead, they increasingly use fencing and other barriers to keep wild animals in protected areas and out of an ever-expanding human domain. This paradigm shift, which has increased within the last few hundred years, creates havoc, particularly with the remaining populations of large migratory wild mammals.

Human-created barriers change evolutionary processes, such as changing the trajectory of adaptations or the resistance to diseases, by blocking the flow of genes and encouraging species differentiation, often over very short time spans. The subdivision of landscapes by means of fencing leads to smaller isolated animal or plant populations more vulnerable to other forms of habitat fragmentation, leading to localized extinction (Hayward and Kerley 2009). A case in point is the precipitous decline of the springbok (Antidorcas marsupialis) from population sizes in the order of millions to a complete collapse of the trek in the Karoo drylands of South Africa. Chris Roche (2008), a South African conservationist, attributes this collapse partly to the expanding role of wire fences in South Africa during the early part of the twentieth century.

Animal control fences can change ecosystems. They reroute large migrating populations, which trample and denude vegetation at fences. Fences benefit large predators by acting as a net within which the predators can trap and catch prey. A fence that excludes large segments of land from the actions of wild browsing and grazing animals can lead to changing fire regimes and upset the tree-to-grass ratio of a savannah. This fencing assists in creating a series of unequal environmental gradients, or changes in the conditions of the environment. Rangeland ecologists have shown that fencing that creates smaller parcels of land can decrease its ecological carrying capacity (Boone and Hobbs 2004).

A further worry for conservationists is that as global warming increases, human-made barriers will prevent certain species from shifting their ranges to more suitable areas. Large animals with long life cycles and small population sizes may be especially vulnerable (Milner-Gulland, Fryxell, and Sinclair 2011, 16). Evidence from the discipline of island biogeography shows that ever-decreasing parcels of land bounded by fencing would not offer a conducive platform for large mammal species to evolve or evolve in a different trajectory. A large antelope that spends its life in a small, intensively managed fenced game farm, for example, lacks the same evolutionary potential of an animal of the same species living in a vast unbounded savannah. Some species may benefit from habitat loss and subdivision, however. They may avoid viruses and bacteria that evolve rapidly to fill new opportunities provided by humans taking over these previously natural spaces.

Fencing and ancient walls built primarily to defend civilizations from invasions can profoundly affect wild species divided as an indirect consequence. Researchers have studied a number of plant species over the large expanse of time since the Great Wall of China was built. Their work shows that “significant genetic differentiation was found between the subpopulations on both sides of the Great Wall” (Su et al. 2003, 212).

Such walls do not belong only to antiquity. The British geographer Reece Jones (2009) notes that the so-called Great Wall of India, which runs for 4,096 kilometers along the border of India and Bangladesh, is a recent anti-human fence. It was built to stem the projected future millions of climate-change refugees from moving from Bangladesh to India. The high-tech Mexico-US border fence and the separation wall between Palestine and Israel not only separate peoples and nations, but also, by default, divide their shared biodiversity. Fences are symbols of marginalization for some local people, who may feel dispossessed from their land by their presence. A recent major review edited by two UK conservationists provides examples of the social, economic, and ecological complexities of separating wildlife and human activities, especially in Africa (Ferguson and Hanks 2010). The long disease-control fence that surrounds the western side of Kruger National Park in South Africa exemplifies “human-fence conflict.” The fence has long been a source of conflict between park authorities and the adjacent communities. The creation of the park in the 1960s evicted some of these communities from their land. The fence
itself was built in the 1950s. Many rural people now connect these two events (Ferguson and Hanks 2010, 55).

Barriers to Conservation Efforts

“Migration is a complex adaptation arising as a result of interaction between individuals, their genes, and the environment . . . evolving in response to spatiotemporal variation in resources or threats” (Cresswell, Satterthwaite, and Sword 2011, 8). Fences subdivide landscapes into ever smaller parcels of land, directly blocking large mammal migration routes. A continuous process of fragmentation, aided and abetted by human-made barriers, creates islands of natural landscapes (surrounded by a matrix of disturbed land) and isolated populations of plants and animal species. Without recourse to wildlife corridors that foster the vital genetic interchange between populations of the same species, these populations become vulnerable to extinction processes associated with small sizes of area and animal numbers (Akçakaya, Mills, and Doncaster 2007).

In human terms, fencing serves as a vital physical and metaphorical marker of private ownership, a means to protect state assets, and as an agricultural tool. Fencing will become more globally prevalent as the human population increases and nation-states compete for increasingly scarce resources.

Neither subsistence farmers nor the more recent commercial farming sectors favor mixing domestic and wild mammals. Diseases jump the species barrier, devastating animal health and economic yields and leading to extreme poverty in rural communities, especially in the developing world. Farmers are locked in a war of attrition between their stock and natural predators and with crop-raiding wild herbivores. Local people living around Kenya’s Shimba Hills National Park demanded a protective fence that would ring the entire park to prevent elephants escaping and damaging their livelihoods. After the park had been completely enclosed, however, local people illegally removed sections of fence to allow themselves access to the park’s natural resources (Knickerbocker and Waithaka 2005).

Long and straight veterinary fences traverse many thousands of kilometers of southern Africa. Managers gave little thought to migration routes when they erected and aligned these fences in the 1950s and 1960s and have added an increasing number since then. The fences are built to prevent livestock diseases, especially foot-and-mouth disease. These zones protect cattle destined for local and export markets. The fences therefore are a crucial bulwark safeguarding the national and private herds of cattle and other livestock.

The fences do not come without a high environmental price. Fences that bisect wildlife migration routes have led, in conjunction with dry periods, to a precipitous decline of migratory large herbivores. Opponents of veterinary fencing estimate that Botswana’s blue wildebeest (Connochaetes taurinus) population has fallen from approximately 250,000 animals to just 4,500 within two decades. They attribute this decline to fencing (Albertson 2010, 86).

Martyn Murray, a Scottish conservation planner, writes in his book _The Storm Leopard_ of “wildebeest dying in thousands along the fence, and in the wastelands around Lake Xau” in Botswana as they tried to reach water and grazing resources (Murray 2010, 113).

Scientists currently know very little about how migrations collapse, in a demographic sense, due to the imposition of fencing. The initial structure certainly may cause mass immediate starvation and dehydration and sometimes death by direct entanglement. The remainder of the population may decline more insidiously because they are more vulnerable to disease and predation or are trapped by fire. If the fences deny them critical nutritional resources, their fertility may decrease. Compounded by a series of steplike population plunges whenever drought returns, this decrease may well herald the final cascade effect as populations get smaller and more vulnerable to random extinction events.

Migrating large herbivore populations in many African settings are doubly tasked. The core protected area often does not fully cover the entire migration route. Fences often circumscribe the park boundaries or the surrounding rangeland that encompass both high livestock and human densities. The migrant population must move “nervously” through these areas. Ecologists predict that plans to bisect the spectacular Serengeti National Park with a road protected by a fence will cause the catastrophic collapse of one of the last great large herbivore migrations.
Global treaties designed to protect the migration routes of large herbivores have a long history of ineffectiveness (Cioc 2009). According to some conservationists (Shuter et al. 2011, 202–203) the Bonn Convention (entered into force 1983) currently offers the best hope for rallying nations to the cause of protecting terrestrial migrations that have internal or transboundary routes.

Ecologists’ efforts to persuade governments or private landowners that these migrations can be economically beneficial for tourism or sustainable utilization will increase the protective momentum. Kenya has directly paid its citizens to remove fencing or banned fences that cut across migration routes. Some of the recently settled Masai peoples previously had used barbed wire to stake their claims to group ranches. Masai resource ecologists such as David Ole Nkedianye championed the plans to remove the existing fences and to keep these remaining Kenyan wildlife corridors open (Nkedianye 2010, 263–266).

Removal, realignment, or circumvention of fencing to aid the free flow of wildlife is a complex and time-consuming conservation practice. Fencing left to deteriorate still causes problems for many large mammals and should therefore always be removed completely. Once governments have removed fencing or other human-made barriers, previously stunted migrations may resume. Researchers using telemetry concluded that Mongolian gazelle (*Procapra gutturosa*) migrations blocked by the presence of a railway “could cross the railroad within a week if there was no barrier” (Ito et al. 2005, 947). Other researchers have used telemetry to pinpoint vital narrow migration corridors used by Saiga antelope (*Saiga tatarica mongolica*) (Berger, Young, and Berger 2008).

Mike Chase, a Botswanan elephant ecologist, and Curtice Griffin, a US environmental conservationist, have spent many hundreds of hours using satellite telemetry to map where elephants and fences interact and cause congestion in southern Africa. These advances pinpoint critical wildlife corridors or funnels that must be kept open in order for the migrations routes to survive or even be re-created (Chase and Griffin 2009).

British and Botswanan researchers studied the population increase and range expansion of the plains zebra (*Equus burchelli*) after the removal of a major fence in Botswana. It took about four years for the population to retrace its former migration route (Bartlam-Brooks, Bunyongo, and Harris 2011). Botswana has emerged as a key testing ground for new ways to deliver increases in agricultural output without the collateral damage to wildlife populations they experienced in the past. Along with the superkraal (cattle stockade) concept, a policy that seeks to control wildlife diseases that affect livestock is emerging. This policy relies less on fencing. Commodity-based trading seeks to sanitize and verify the disease-free status of beef at the abattoir, or slaughterhouse, stage, rather than to rely on rangewide surveillance. This policy may allow for a more mixed (wildlife and livestock) economy to emerge (Ferguson and Hanks 2010, 62–65).

Similarly inventive is the reasoning behind transfrontier conservation in Africa, as described by John Hanks, a UK expatriate with forty-five years conservation experience on the continent. This policy may eventually lead governments to remove major fences, thereby allowing “transfrontier wildlife corridors [that] will have a vitally important role to play in regional conservation activities by presenting or consolidating opportunities for various species to move freely across international borders” (Ferguson and Hanks 2010, 25).

Fencing is a core management tool in two innovations in the development of African protected areas. Game conservancies combine individual private farms by removing the internal fences and relying instead on a single perimeter fence. The increasing development of privatized or leased protected areas often requires newly installed fences to protect the large private financial investment in rehabilitating defunct game reserves.

Where fencing must remain (for animal disease control or military purposes, for example), conservationists seek new methods to curb the excessive damage to wildlife these structures cause by direct entanglement and long-term population declines due to the cessation of
migrations. “Virtual fencing” techniques, such as olfactory barriers or biofences, more selectively prevent certain species from crossing out of a proscribed area. J. Weldon “Tico” McNutt, a US animal behaviorist, and his colleagues have tested the use of chili fences and bombs and slow-burning dung bricks. Local African people can easily deter elephant crop raiding with these methods. At the same time these methods will not restrain the movement of elephant herds as would traditional fencing methods.

Farmers and ranchers now use more sophisticated fencing to subdivide the ever-increasing human-dominated landscape. In an intensive agricultural setting, “live” fencing—refuges made from hedges and thorny shrubs to separate fields and plots of land—benefit biodiversity. In developing countries ranchers use natural materials such as thorn branches to kraal (stockade) animals, especially at night, to protect them from human theft and natural predators. Dedicated grazing rangeland, forming fenced-in superkraals, helps deal with the increases in livestock numbers. Such developments in Botswana could help protect the national cattle herd from wildlife-borne diseases and still provide larger parcels of range land between these “cattle islands” for the exclusive use of wildlife.

Managers can make even traditional fencing more selective. In certain areas of the Kruger National Park in South Africa, elephant-restraining fences built with shoulder height cables stop adult elephants from getting through to damage mature trees. This fence allows the passage of movement for smaller game and avoids the hazard of electric fencing, which increases the mortality of small animals like tortoises. Governments and planners have put much effort and money into finding the ultimate cost-effective, elephant-proof fence.

The Future

Governments can preserve wildlife corridors using ingenious means. Elephants, moose, deer, gazelles, and badgers are among the many species that have benefited from such means of circumventing major barriers. Only when governments fully appreciate and value the importance of species and ecosystems will the barriers come down to save or revive some migration events.

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See also Adaptive Resource Management (ARM); Agroecology; Biogeography; Biological Corridors; Charismatic Megafauna; Complexity Theory; Ecological Restoration; Edge Effects; Food Webs; Habitat Fragmentation; Human Ecology; Plant-Animal Interactions; Population Dynamics; Road Ecology; Soil Conservation

Further Reading


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Fire management involves making decisions and taking action to prevent, maintain, control, or use fire in a particular region or landscape, and it depends on knowledge of fire frequency, severity, size, and seasonality. The tools of fire management include suppression of fires, application and management of prescribed fire, and the management of fuels. Knowledge and experience gained during the twentieth century guide current fire management goals and practice.

Fire management is defined as the range of “decisions and actions available to prevent, maintain, control or use fire in a given landscape” (Myers 2006). In the broadest sense of the phrase, humans have been managing fire for hundreds of thousands of years. There is strong evidence for controlled use of fire by Homo erectus beginning 400,000 years ago and by our own species, Homo sapiens, for more than 100,000 years. Indeed, anthropologists view the controlled use of fire as a major milestone in human evolution. It provided warmth, heat for cooking and protection from predators and enemies. Humans also purposefully set fires in ecosystems to facilitate travel and improve habitat for the plants and animals they depended on. (It should be noted that the human use of controlled fire to manage and improve vegetation differs from slash-and-burn [swidden] agriculture, in which entire forests and woodlands are cut and burned to create fields for planting and pasture for livestock.)

The tools of modern fire management include suppression of fires, application and management of prescribed fires, and the management of flammable materials or fuels. Fire suppression involves a variety of tactics including simple and direct extinction of flames with shovels and water pumps, the setting of backfires to eliminate fuels in advance of a wildfire, and aerial applications of water and fire retardants by planes and helicopters. Prescribed fires are either intentionally or naturally ignited fires that are allowed to burn under predetermined conditions of weather, fuel moisture, and fire intensity. Fuel management includes the removal of flammable materials from fire lines cut by fire fighters and fuel breaks installed on landscapes to limit the spread of wildfires. Fuel management also includes the modification of fuels within an ecosystem, such as the removal of shrubs, saplings, and woody debris from forest understories (growth at the lowest height under the forest canopy).

The fire regime is a foundational concept in fire management and recognizes that, under natural conditions, fires behave in different ways in different ecosystems. Land management agencies in the United States often define fire regimes in terms of fire frequency or return time (i.e., the average time between fires at a given place), and fire severity, the impact of fire on the vegetation (NIFTT 2010). High frequency–low severity fire regimes are typical for grasslands, savannas, and some forests such as the giant sequoia-mixed conifer forests of the Sierra Nevada. In these systems fires may burn as frequently as every one to ten years, consuming surface fuels but killing very few plants. At the other extreme are low frequency–high severity fire regimes such as those occurring in high-elevation conifer forests. In these ecosystems fires recur much less frequently and kill most of the canopy trees. In between are so-called mixed severity fire regimes that are typified by high variation in fire return times and impacts on vegetation. For example, fire return times in pine- and fir-dominated forests in the western United States vary from surface fires burning every five to ten years to local crown-killing fires every hundred to two hundred years. In addition to frequency and severity, fire managers...
understand that variations in fire size and seasonality are also important components of fire regimes that influence post-fire vegetation responses (Keeley et al. 2009).

Fire regimes and fire behavior are influenced by a variety of factors. Sources and patterns of ignition can limit the frequency of fires. Lightning, the most important natural ignition source, is quite frequent in the southeastern United States, with more than four strikes per square kilometer per year (> 4 strikes/km²/yr), but far less frequent in the far west, with less than a quarter of a strike per square kilometer per year (< 0.25 strikes/km²/yr). Ignitions by humans are most common near roads and urban centers.

Weather conditions greatly influence both ignition and fire spread. Fuel moisture is directly influenced by temperature and relative humidity. During the heat of afternoon, temperature is typically high and relative humidity is low; fuels are dryer and are more likely to ignite and burn more severely. Overnight and in the early morning hours, low temperatures and high humidity have the opposite effect. Winds influence the drying of fuels as well as fire spread.

Fire frequency and behavior is also influenced by the amount, dead/live ratio, size structure, chemistry, and spatial arrangement of fuels. Amount of fuel is influenced by the relationship between net primary production and decomposition. Where decomposition is slow, fuel dead/live ratio is higher and this increases ignition probability and fire spread. Size structure refers to the distribution of fuels among fine material (leaves and twigs) and larger material (branches to large logs). Fine fuels dry out much more quickly than larger fuels; thus a higher proportion of smaller fuel lends itself to more frequent fires. Fuel chemistry includes the content of inorganic salts and volatile organic compounds that influence volatility. Spatial arrangement of fuels is especially important with respect to fire spread. Where fuels are evenly distributed vertically, fire can easily spread from the forest floor through so-called ladder fuels up to the fine fuels in the forest canopy. Where ladder fuels are absent, such as in savannas, fires tend to be confined to the surface. The horizontal distribution of fuels affects the spread of fires across landscapes. Large, contiguous expanses of dense forest or shrubland thus tend to support very large fires; fires may be limited to smaller patches on heterogeneous or fragmented landscapes.

History of Fire Management

Although the following examples illustrate fire management research and practices in the United States during the twentieth and early twenty-first centuries, the field itself is practiced worldwide with the same general principles, and is especially important in dry climates such as Australia’s.

At the beginning of the twentieth century, US land managers engaged in a vigorous debate on fire management, although the term “fire management” did not actually appear in the discourse. The debate was about “light burning”—whether fires should be intentionally set and allowed to burn in the understories of forests in order to maintain open conditions and diminish risk of severe wildfires. The debate about light burning was to be suppressed (Pyne 1982). Complete suppression of fires regardless of ignition source (lightning or human-set) became official national policy soon after. Despite accumulating evidence and voices to the contrary (e.g., Chapman 1932 and Garren 1943), there was consensus among many, if not most scientists and managers during this time that (1) wildfires were a mostly human-caused problem, (2) wildfires could be eliminated from ecosystems, and (3) suppressing fire had no adverse consequences.

Over the next quarter century, a welter of evidence from a variety of ecosystems challenged this consensus (e.g., Biswell 1961; Cooper 1960; and Weaver 1968) so that, by the 1960s, there was widespread agreement
among most ecologists and some managers on four points that were completely contrary to the previous conventional wisdom. (1) Fires are not random or chance occurrences; they ignite and burn in particular ways as a consequence of the confluence of climate, weather, ignition sources, and the growth of fuels. (2) Fire regimes—the typical frequency and behavior of fire—and ecosystems have co-evolved. The flora and fauna of many ecosystems are not simply resilient to fire, they depend on it. Fires come and go at least in part, in response to the growth patterns of the fuels they burn. (3) Exclusion of fire from these co-evolved ecosystems does have significant consequences, often affecting the establishment and growth of shade- and fire-intolerant species, and closure of the forest understory. (4) The absence of fire can actually increase the amount and flammability of fuels, increasing both the risk and severity of future fires.

In 1967, the Forest Service relaxed its “10 a.m. policy,” which stipulated that fires were to be under control by 10 a.m. on the day following their outbreak, for early- and late-season fires. The Park Service began implementing prescribed fire programs soon after, using deliberately ignited fires in the Everglades of Florida and in giant sequoia groves of California, and allowing lightning fires to burn within preset guidelines in the high-elevation forests of the Sierra Nevada and, beginning in 1972, Yellowstone National Park. (The lodgepole pine common in Yellowstone actually depends on fire to reproduce, both to open its cones and release its seeds, and to provide a mineral surface for seed germination.) In 1977, the Forest Service further modified its policy to allow local fire managers to consider alternatives to full suppression, including the use of prescribed fire.

The 1988 fires in Yellowstone National Park were a watershed event for fire management. To a greater or lesser extent, they burned nearly half of the park’s forested landscape, closing the park to visitors and threatening nearby communities. These fires were the first of many events that would keep fire management in the limelight of public attention and politicize its management. Subsequent major fires in Yosemite; the devastating 1991 fires in and around Oakland, California (close to San Francisco); and the 1994 fire season with its thirty-four fatalities were all cause for scrutiny of the concept of fire management.

Since 2000, numerous very large fires, so-called megafires, have occurred in ecosystems as diverse as coniferous forests in Arizona, Colorado, and Oregon, chaparral in southern California, shrub and grasslands in Texas, and pine flatwoods in Florida. In the case of western coniferous forests, the extent and severity of these fires is at least in part due to the accumulation of fuels as a consequence of historical fire suppression (Keeley 2009). Abnormally hot, dry weather, however, has been a contributing factor in all of these situations (Brown et al. 2004). Development and increased human access to these ecosystems have also increased the probability of fire starts (Hansen et al. 2005).

More importantly, development and the blurring of boundaries between urban and wilderness landscapes have increased the consequences of wildfires for human life and property (Dombeck et al. 2003).

Fire Management Goals

Wildfire prevention and control are often the central goals of fire management. This has been particularly true over the past decade because of concerns regarding the accumulation of excessive fuels in many North American forests (Schmidt et al. 2002). The central focus is on reduction of fuel continuity between the forest floor and canopy and across the landscape. Removal of understory trees and shrubs has been a useful and sometimes controversial technique applied to many western forests (Wallin et al. 2004; Peterson et al. 2005). Where fuel accumulations are light, thinning may be accomplished with prescribed fire. Where fire suppression has resulted in heavy fuel accumulation, however, thinning must be done mechanically and often at considerable expense. Such fuel treatments appear to have little influence on fire spread in the most extreme weather conditions. Fuel breaks are an important
class of fuel manipulations that can limit fire spread under moderate weather conditions. But they may be more important in providing access for fire fighters and staging areas for ignition of prescribed burns.

Fire management, including the use of mechanical thinning and prescribed fire, is also widely used to restore ecosystem structure and composition to presettlement conditions (e.g., Covington and Moore 1994). Here the goal is often to recreate conditions under which natural fire regimes can be maintained. Such restoration assumes an understanding of presettlement conditions, which is often a matter of controversy (Christensen 2006). Recognizing that presettlement ecosystems were themselves constantly changing, fire managers often determine restoration targets based on historic range of variation, the range of fire behaviors, and ecological conditions that may exist over a particular historical period (Keane et al. 2009).

Fire management is also used to achieve silvicultural goals (those related to the development and care of forests). Pre- and postharvest prescribed fire is widely used in pine plantations in the southeastern United States to limit growth of competing hardwoods. It is also used in hardwood stands to create conditions favorable for germination and growth of shade intolerant trees such as oaks (Brose et al. 2001).

**Fire Management Guidelines**

Over the past two decades, fire scientists and fire managers have learned a great deal regarding the behavior and role of fire in specific ecosystems and across landscapes. Norman L. Christensen Jr. (2009), a Duke University professor whose area of expertise includes research on ecosystem responses across fire regimes in North America, summarized the most important of these lessons as follows.

*Know what it is you are trying to accomplish and why.* While fire is essential in many ecosystems, it is not the endpoint of management. Rather we manage fire—we suppress it, restore it, and prescribe it—in order to conserve key things such as fuel conditions, natural and historic objects, wildlife, and key processes such as energy flows and element cycles. Our goals must be formulated in terms of these measures of forest sustainability.

*Set realistic goals.* Fire managers set fires, extinguish fires, and in various ways manage fuels across a range of fire regimes. The fact that certain things are easy to do at one end of that range too often leads to hubris regarding what can be accomplished elsewhere. Prescribed fire is virtually an oxymoron in many fuels.

*Manage the cycle—the entire process of change—not just the fire.* Fire is just one moment, albeit a transformational moment, in a process of change. The nature of a fire, any fire, is determined only in part by conditions—weather, fuel moisture and so forth—unique to that moment. Much fire behavior is a consequence of a century or more of ecosystem change preceding it. Furthermore, its behavior will influence the patterns of change that proceed from it over the decades and centuries that follow.

*Manage less for desired future conditions and more for desired future change.* Change is constant, and efforts to restore a particular condition with no thought about the change that will follow are likely to produce unhappy consequences.

*Variation and complexity matter—conserve them!* Perhaps the greatest ecological lesson of the 1988 Yellowstone fires was their variability and the equally remarkable diversity of recovery patterns and biological communities they produced. The diversity of many ecosystems is a consequence not just of disturbance, but of variations in disturbance and the processes of change they produce. For this reason, managers should avoid homogeneity in their practices.

*Beware of arbitrary boundaries.* This is, of course, a basic tenet of ecosystem management. The megafires of the past decade demonstrate that the spatial extent of fire and of the many processes that are affected by fire has little relationship to jurisdiction or ownership boundaries or the boundaries we use to define social and cultural categories such as urban and wildland.

*The world is changing—expect surprise and manage to accommodate it.* Climate change, invasive species and changing patterns of land use are creating conditions that have no historical precedents. Fire management that mimics the historic range of variation may therefore produce unexpected and undesirable change.

*We are mostly managing people.* Fire management is not an academic matter; it has great consequences for human life and property. If nothing else has been learned on this matter in the past twenty years, it is that attempts to manage fire and fuels at landscape scales and across jurisdictional boundaries must have the engagement of all communities and stakeholders.

*You only think you know what you’re doing—be humble, manage adaptively.* There is much we do not understand about fire behavior and fire effects; we have no choice but to learn on the job—adaptive management is critical. This requires monitoring that is directly relevant to goals and objectives and research that addresses the most pressing uncertainties. The world is changing, but uncertainty is an unacceptable excuse for inaction. Indeed, in a world of change, there is no such thing as inaction.

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See also Adaptive Resource Management (ARM); Administrative Law; Best Management Practices (BMP); Boundary Ecotones; Community Ecology; Complexity Theory; Disturbance; Ecological Forecasting; Extreme Episodic Events; Forest Management; Habitat Fragmentation; Human Ecology; Indigenous Peoples and Traditional Knowledge; Invasive Species; Reforestation; Regime Shifts; Succession

FURTHER READING

Fish hatcheries have two purposes, first, to breed fish for release into the wild to augment natural populations, and second, to introduce new species to river habitats. Releasing more hatchery fish has led to several problems, and the existence of fish hatcheries allows people to ignore core problems such as overfishing and loss of habitat.

Breeding in a fish hatchery involves stripping female fish of their eggs, which are then fertilized and incubated until they hatch. In a way, hatcheries are similar to agricultural farms, in that production is increased to meet demands. The fish produced in hatcheries generally are released into either rivers or reservoirs. Broodstock (parents) are best obtained from the location of planned release or an ecologically similar stream, which increases the likelihood of survival. Fish hatcheries are usually a cost-effective way of enhancing populations, with it costing about $40,000 to produce four million fry (Aprahamian et al. 2003).

The two main reasons for setting up fish hatcheries are to conserve, restore, or enhance depleted natural populations and to introduce new species into habitats where they don’t naturally occur. Fish populations have been severely depleted in some areas, often because of habitat loss or degradation or obstructions in rivers (by dams). The River Tyne in northeastern England, for example, lost 11.2 kilometers of fish habitat because of the development of Kielder Reservoir (Aprahamian et al. 2003). Restocking the river upstream of the reservoir using fish reared in a hatchery has somewhat lessened the dam’s interference with natural fish reproduction. The most successful results, however, combine restocking with habitat restoration to increase the carrying capacity (the number of fish that can be supported by resources) of the river. Hatcheries aimed at boosting depleted populations are generally seen as beneficial even though these programs have had limited success. Fish populations can also be introduced into locations where they have previously not been, generally for recreational or commercial fisheries. This is a controversial activity in terms of ecosystem sustainability, but can bring short-term economic benefits. Introducing new species to a habitat in which they do not naturally live is ultimately unsustainable, however, and may damage the existing ecosystem, especially if populations exceed the natural carrying capacity of the river.

Fish hatcheries produce large quantities of fish. For example, 250 million Atlantic salmon smolts are produced every year, of which three quarters are bred in Norway, with smaller numbers produced in Scotland, Iceland, Ireland, and the Faroe Islands (Bergheim et al. 2009). Fisheries production is increasing rapidly, from between 100,000 and 300,000 fish in 1985 to between 500,000 and 2 million in 2000 (Bergheim and Brinker 2003). Production in Scotland, for example, doubled in the ten years leading up to 2006. One of the main reasons this increase in production has been possible is the method used to incubate the eggs. Previously, single-pass-through systems or partial reuse systems were used. Hatcheries have replaced these methods with recirculating systems, which allow for lower water use and improved water quality (Fivelstad et al. 2003). Unfortunately, hatchery-bred fry have a high mortality rate when they are released into the wild. One study showed a 48 percent mortality rate of salmonid fry after only two days of release (Henderson and Letcher 2003). Other studies have shown that the time and location of release, as well as allowing for acclimatization, is critical to survival (Aprahamian et al. 2003). Spawning normally is induced earlier in fish hatcheries than would naturally
occur, and as a result hatchery fry are ready for release before wild fish hatch. This timing difference leads to a mismatch between reared fish and the flow regime (characteristics of the river), leading to both mortality and competition with natural fish when they hatch later. Clearly, releasing fish from hatcheries is not always a sustainable approach to augmenting fish populations.

## Controversies

Fish hatcheries are not intrinsically good or bad, and their success or impact can be determined only in the context of the aims of the hatchery (Waples 1999). Some ecologists criticize hatcheries because they are seen as a substitute for dealing with the underlying reasons that fish populations are decreasing, such as habitat degradation, overharvesting, and obstructions (Levin et al., 2001). Furthermore, introducing reared fish into the native population can have negative impacts. Many researchers believe that fish hatcheries result in reduced genetic diversity within the natural population, as well as other negative genetic changes. Artificial selection for certain traits in reared fish (Campton 1995), interbreeding (McMeel and Ferguson 1997), and unnatural competition result in more homogeneous populations. One strategy hatcheries pursue is to release fish early in their life cycle. Such early releases, however, lead to competition with natural fish and reduce the benefits that hatcheries provide in terms of absolute number of fish that survive. Furthermore, releasing reared fish into the wild can introduce diseases or parasites into the natural population. One example of this is the spread of Gyrodactylus salaris into populations of Atlantic salmon in Norway (Waples 1999). One US study shows that the introduction of hatchery fish reduced the resistance to disease of native coho salmon in the Fishhawk Creek tributary of the Nehalem River in Oregon (Wade 1986). Furthermore, the State of Oregon released 1 million steelhead trout between 1966 and 1975, but had no adult returns. The problem was traced to the Ceratomyxa shasta parasite, present in the Willamette River, to which the reared fish were not resistant. One possible solution to this kind of problem is to use local broodstock that is resistant to endemic parasites and diseases (Maynard, Flagg, and Mahnken 1995).

Success in augmenting fish stocks has been mixed. One attempt to restore depleted populations of striped bass, in the Albemarle Sound / Roanoke River (1981–1996) demonstrated that restocking has only a small impact unless the original populations are extremely low (Patrick et al. 2006). Nevertheless, the active management program is highly valued by the public. An attempt to restore chinook salmon populations in the Snake River in the Columbia River basin demonstrated one of the negative effects of introducing hatchery fish, when the population exceeded the carrying capacity of the river and many fish starved (Levin, Zabel, and Williams 2001; Hilborn and Eggers 2000).

## Future Prospects

Fish hatcheries have proved to be an effective way to manage fish populations. Clearly defining the aims of specific hatcheries will enable researchers to evaluate their performance meaningfully. The success of a restocking program should be measured in terms of adult returns rather than the number of fish released. More research is required, however, to assess the long-term impact of introducing reared fish into natural populations, especially in terms of their impact on genetic diversity. Hatcheries will need to be more flexible and adaptive to the latest research, because they are currently quite resistant to change. Fisheries biologist Robin Waples (1999) calls for a common understanding of the realities of hatcheries, dispelling the myths surrounding them, which will enable researchers to compare their successes objectively. Finally, restocking using hatcheries should be only part of the solution to depleted fish populations, with other measures, such as more stringent regulation of fishing and habitat restoration, enacted to preserve existing stocks. Fish hatcheries should be viewed more as artificial tributaries integrated into the larger ecosystem than as farms (Lichatowich 2003). Taking this view would lead to more sustainable ecosystem restoration and fish populations.

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See also Agroecology; Biodiversity; Catchment Management; Dam Removal; Eutrophication; Fisheries Management; Food Webs; Hydrology; Invasive Species; Plant-Animal Interactions

## Further Reading


Wade, Mark. (1986). The relative effects of Ceratomyxa shasta on crosses of resistant and susceptible stocks of summer steelhead. Corvallis, OR: Oregon Department of Fish and Wildlife, Research and Development Section.


Since the mid-1950s, economists and fisheries scientists have highlighted the problems of managing sustainable fisheries—such as the open access nature of the industry and harmful fisheries subsidies—and have also stressed the need to re-establish the natural protection afforded fish in marine-protected areas and to address the problem of illegal, unreported, and unregulated fishing. Accordingly, fisheries managers need to overcome shortsightedness that focuses on the current generation without accounting for the interests of future generations.

It is widely acknowledged that the world’s fish stocks and fisheries are in crisis, even though some may be recovering. Summaries of recent reports published in industry journals and papers have discussed trends in catches or biomass of fishes at the species, regional, and global levels, and they provide useful documentation to assess the level of the crises.

Catch Profiles: Individual Species

Mapping of catch profiles can depict the trend for many commercial fish stocks that have been exploited over time (see figures 1a and 1b on page 147): the catch usually begins at a very low level; gradually the catch increases as demand for the particular fish increases and the fishing technology improves (thereby reducing the cost of fishing); the increase continues until the stock can no longer support the catch; then increases in catches stop, and are followed by declines of catches toward zero.

The Canadian ecosystem modeler Villy Christensen and colleagues (2003) report a dramatic decline of fish biomass in the ecosystems of the North Atlantic between 1950 and 1999. They estimated the biomass of high trophic level fishes (i.e., those high on the food chain) in the North Atlantic based on twenty-three spatialized ecosystem models, each constructed to represent a given year or a short period from 1880 to 1999. Their results indicate that the biomass of high trophic level fishes declined by two-thirds during the last fifty years of the twentieth century, and by a factor of nine over the full century. (See figures 1a and 1b.)

Global Fish Catch Data

At the global level, the German and Canadian scientists Rainier Froese and Daniel Pauly (2003) analyzed the Food and Agriculture Organization of the United Nations (FAO) global fisheries catch-data set from 1951 to 1998. Using FAO’s five definitions of the status of a fishery (namely: undeveloped, developing, fully developed, overfished, and collapsed), the authors categorized 932 species of fish in the FAO’s global catch statistics according to the status that applied to the species for each year from 1951 to 1998.

Their analyses showed a steady erosion of oceanic fishery resources worldwide during that time period. Compared to 1951, when approximately 70 percent of global fish catches came from undeveloped fisheries and almost none came from collapsed fisheries, the reverse was the case in 1998, when almost none of the total global catch came from the undeveloped fisheries and over 10 percent came from collapsed ones.

Even though there is ongoing debate in the fisheries literature regarding the usefulness of catch data in telling us the state of a fish stock (see, for example, Branch et al. 2011) academic research shows that whether one looks at the trends in fisheries catches at the species, regional, or global level, fisheries have been declining, and therefore it must be concluded they are not being
Figure 1a. Biomass Distributions for High Trophic Level Fishes in the North Atlantic in 1900

Source: Christensen et al. (2003).

Figure 1b. Biomass Distributions for High Trophic Level Fishes in the North Atlantic in 1999

Source: Christensen, et al. (2003).

Figures 1a and 1b show the decrease of high trophic level fish stock (i.e., fish high on the food chain) in the North Atlantic during the twentieth century. The loss of fish stock around Newfoundland is particularly striking.
managed sustainably. These results are not mere academic findings—they are widely accepted by world leaders, as demonstrated by the various commitments undertaken at a number of global forums, including the Johannesburg World Summit on Sustainable Development (2002), and the Convention on Biological Diversity in Nagoya, Japan (2010). At these and others summits, governments have committed to protecting biodiversity and improving ecosystem management, including the commitment to restore fisheries to their maximum sustainable yields by 2015, and to establish a representative network of marine protected areas by 2012 (Johannesburg Summit).

Economic Data on Sustainability

Key reasons advanced by fisheries scholars, especially economists, for the inability to manage fisheries sustainably include the open access nature of fishery resources; government subsidies to the fishing sector; technological progress and increasing trade in the face of ineffective management; problems related to illegal, unreported, and unregulated (IUU) fishing; and what can be called “shortsightedness” in the valuation of fishery benefits.

Open Access Nature of Fishery Resources

Open access areas are those in which fishers may use the marine ecosystem, either totally uncontrolled or where no well-defined access rights—whether individual, communal, or state—exist and are enforced. Experts have convincingly shown that under open access the tendency is to overcapitalize fisheries, resulting in overexploitation of the resource. This has led to various measures, especially at the international level, aimed at turning open access fishery resources into well-defined and functioning common or private access rights to resources. A case in point is the coming into force of the UN Convention on the Law of the Sea in 1982, which formally established access rights to coastal nations within their 200-mile (322-kilometer) exclusive economic zones (EEZs). The Law of the Sea therefore turned what used to be global commons into the private property of coastal nations. But clearly, the law does not solve the problem of domestic open access or the problem of open access in the high seas (Norse et al. in press), or the problem that comes about due to the transboundary (Munro 1979) or shared (Sumaila 1997) nature of fishery resources. Hence, in many cases fisheries are still effectively open access.

Government Subsidies

Some subsidies to fisheries are recognized worldwide as serious threats to sustainable fisheries management, as they intensify overcapitalization and overfishing (FAO 1998; OECD 2000; Porter 2002; Munro and Sumaila 2002). In the late 1990s Mateo Milazzo (1998), writing for the World Bank, estimated the total amount of subsidies paid by governments around the world to their fishing sectors to be between $15 and $20 billion per year. An estimate from 2010 puts the amount of subsidies at approximately $27 billion per year (Sumaila et al. 2010). This estimate is more than 30 percent of the industry’s annual revenue of $81 billion a year, implying that fishers can keep on fishing even when the cost of fishing is up to 30 percent higher than the amount they bring in.

Increasing Trade/Ineffective Management

The FAO reports increasing global trade in fish and fish products since about the year 2000. Fishing gear and vessel technology have achieved the capacity to impact radically the Earth’s marine ecosystems. Fleets have become powerful enough to overexploit essentially all stocks in the world.

In fact, technological progress in fishing gear, which has contributed to overfishing, for instance, has virtually removed the “natural protection” afforded fish in earlier times when fishing was more labor- and time-intensive. Also, improvement in fish product preservation and transportation technology has significantly increased the scope of international trade in fish products, thereby removing market barriers to fishing. Fishers who were able to do business in a domestic market now have available a global market with significantly higher demand for fish.

In general, trade economists agree that if fisheries are managed effectively to meet a stated objective, then increasing trade in fish and fish products will not result in unsustainable practices (Neumayer 2000). There are three issues to note here. First, the flow of trade in fish and fish products comes increasingly from the global South (in general, the Earth’s Southern Hemisphere, where the majority of countries are developing or underdeveloped) to the global North (in general the Northern Hemisphere, where most of the world’s more developed countries are located); that is, from fisheries that lack the resources required to manage their operations effectively, with the consequence that increasing trade results in unsustainable fisheries management (Alder and Sumaila 2004). Second, even in the resource-rich and developed North, effective management is not very common due partly to the existence of domestic open access in many fisheries. Third, usually the discussion on trade and fishery sustainability is framed under the assumption that effective management is not directly related to increasing trade. It is likely, however, that increasing trade will impact on the ability to manage fishery resources.
sustainably. Potentially, increasing trade may weaken or strengthen the ability of countries to manage their fisheries sustainably; the actual direction will depend on the specific country and fishery.

Illegal, Unreported, Unregulated (IUU) Fishing

Illegal, unregulated, unreported (IUU) fishing occurs in many places—not only in the high seas but also within exclusive economic zones (EEZs) that are not properly regulated (Sumaila et al. 2006). IUU fishing leads to a failure to achieve both management goals and sustainability of fisheries (Pitcher et al. 2002). When stock assessments are performed on fisheries, reported catch and effort data is used. The underreporting of illegal catches, however, results in the absence of a significant part of the annual catch and therefore makes the stock assessments nearly useless (Pauly et al. 2002, FAO 2001). The good news is that the issue of IUU fishing has begun to receive the attention of scholars, fisheries managers, and governmental, intergovernmental and nongovernmental organizations. For instance, the FAO has begun the implementation of an International Plan of Action (IPOA), which encourages all states and regional fisheries organizations to introduce effective and transparent actions to prevent, deter, and eliminate IUU fishing and related activities (FAO 2003). And, the Organization of Economic Cooperation and Development (OECD) sponsored the OECD Workshop on IUU Fishing Activities in April 2004 as part of their effort to help find solutions to the problem.

Shortsightedness in Valuation

Shortsightedness stems from the general human perception that what is closest to us appears to be large and weighty, while size and weight decreases with our distance from things. This human tendency to be shortsighted is facilitated by the economic concept of discounting—that is, the approach by which values to be received in the future are reduced to their present value equivalent using a discount rate (Koopmans 1960). Discount rate assumptions, as used to reduce a stream of net benefits into net present value (NPV), can have a big impact on the apparent best policy or project (Nijkamp and Rouwendal 1988; Burton 1993; Fearnside 2002). In particular, high discount rates favor myopic fisheries policies that result in global overfishing (Clark 1973; Koopmans 1974; Karp and Tsu 2011).

If, for example, there are five twenty-year generations of fishers and fish consumers within the time horizon being analyzed, nearly 75 percent of the NPV accrues during the period of the first generation, and close to zero percent of the NPV accrues during the last generation’s period.

Is this outcome a problem? Is it fair to future generations? Assuming that future generations would also like to enjoy fish protein and other benefits derived from having viable and healthy fish populations in the ocean, many people may be tempted to say that the outcome is not fair to future generations.

To explain why so many came to that conclusion, Rashid Sumaila of the University of British Columbia proposed a concept called the discounting clock (Sumaila 2004). The term applies to the time period used to discount the flows of benefits. In the equation $t = 0–T$, “t” represents the time period and “0–T” the range of time; “T” is the terminal period, which can be infinity. If, for example, the discounting clock with a time period that runs from zero to one hundred years ($t = 0–100$), conventional practice calls for the use of one discounting clock, which starts at the beginning of the first generation’s period and stops at the end of the fifth generation’s.

Instead of using only one discounting clock for the current generation, it is possible to consider five clocks, one for each of the five generations. Each clock then starts and ends at the beginning and end of a given generation’s time. Discounting the flow of constant benefits in this way, the percentage of the modified NPV that accrues during each generation’s time is the same and equal to 20 percent. So, in economic and social terms conventional discounting of a constant flow of benefits is not a problem. It appears to be a problem only because by using the current generation’s discounting clock, we count the benefits that would accrue to future generations as if they were those to be enjoyed by the current generation.

The above analysis assumes a constant flow of annual benefits. In the real world of fishing, however, constant
flows of benefits do not happen (recall the earlier discussion in this section that introduced the topic of discounting clocks). This is because it is simply not economically rational to assume so when one uses only the current generation’s discounting clock. Future benefits are given much smaller weights in NPV calculations, therefore it does not pay to wait to catch the fish in the future. This leads to the tendency to “frontload” fisheries benefits, resulting in overfishing in the current period (Clark 1973, Heal 1997, Sumaila 2004), to the detriment of future catches.

This tendency to frontload, which results from shortsighted valuation as captured by the concept of discounting, poses the biggest problem to sustainable fisheries. Discounting encourages the current generation to frontload fisheries benefits, thereby threatening the ability of future generations to meet their fish protein needs.

The Future

To ensure sustainable fisheries, we need to deal with the problems economists and other fisheries scientists have highlighted since the mid-twentieth century—namely, the open access nature of fisheries and harmful fisheries subsidies—to re-establish the natural protection afforded to fish (by creating marine protected areas, for example), and to address the problem of IUU fishing.

More importantly, we need to break the shortsightedness tendency. We need to value benefits in a manner that explicitly takes into account the interest of future generations by employing the discounting clocks or time perspective of all generations, rather than follow the current practice of using only the discounting clock the present generation. Many might agree that this shortsightedness is a very powerful human tendency that will be difficult to break, but which must be broken if the world really wants to succeed at managing fisheries and other environmental resources sustainably.

Identifying a way to conceptually and intellectually deal with the shortsightedness problem—that is, to integrate in our valuation methodologies the ways by which benefits to future generations can be counted as benefits that will accrue to them, and not to the current generation—has opened a line of research that has led to a host of contributions (e.g., Ainsworth and Sumaila 2005; Ekeland et al. 2010; Karp 2011). Using these and other findings as the model, future research efforts could provide solutions to what is arguably the most difficult problem facing humanity when it comes to managing natural resources and the environment in general, and fishery resources in particular.

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See also Best Management Practices (BMP); Carrying Capacity; Coastal Management; Complexity Theory; Eutrophication; Fish Hatcheries; Food Webs; Human Ecology; Indicator Species; Keystone Species; Large Marine Ecosystem (LME) Management and Assessment; Marine Protected Areas (MPAs); Ocean Resource Management; Population Dynamics; Shifting Baselines Syndrome

Further Reading


Food webs are a way to understand feeding interdependencies among species within an ecosystem. The degree to which such interdependencies remain intact determines the sustainability of a system. Humans are closely linked with natural systems and have therefore shaped food webs throughout the world. The conservation of food webs is essential to ensure sustainable resource use and to preserve ecosystem health.

A food web is a way to think about how plants and animals in an ecosystem are connected. Species are organized into food webs according to the resources they consume. Primary producers (plants) draw on inorganic nutrients from the soil and carbon dioxide from the atmosphere, primary consumers (herbivores) consume plants, secondary consumers (carnivores) consume herbivores, and decomposers (for example, bacteria) consume and recycle all other uneaten, dead organic matter.

A food web is depicted as a network of nodes interconnected by directional links. (See figure 1.) Nodes represent variables, such as species or nutrients, and directional links represent the feeding dependencies between consumers (i.e., the eaters) and resource species (i.e., the eaten). Energy is cycled through the system as it is repeatedly consumed and converted into edible tissue at each level of the chain and finally returned to the soil through decomposition. Most of the energy captured in food ends up lost as heat and indigestible waste during metabolic processes, and consumers convert an average of only 10 percent of chemical energy into tissue. This ecological inefficiency means that most of the biomass, or amount of living matter, in food webs tends to be tied up in plants and progressively less in herbivores and carnivores, creating a pyramid-shaped structure of food webs where plants are abundant and carnivores are rare. Abiotic factors, such as climate and nutrient availability, as well as biotic factors such as predation and competition, caused by living organisms, affect species’ populations. As their environments change, species may adapt, for example, by changing their diet. The structure of a food web is quite flexible, as species turn to different food resources in response to environmental change.

Drivers of Ecosystem Stability

Prior to the mid-twentieth century, abiotic (nonbiological) factors were considered the dominant driver of ecological processes and species interactions. Weather and nutrient availability were considered to be the primary limitation on plant productivity, which in turn limited animal productivity. This so-called bottom-up control perspective was challenged in 1960, when zoology professors Nelson Hairston, Frederick Smith, and Lawrence Slobdkin proposed the green world hypothesis, which suggests that carnivores play a very important role in structuring ecosystems by consuming herbivores and thus limiting grazing on plants. So-called top-down control, in which predators limit herbivores and indirectly benefit plants, transformed the way that ecologists understand biological processes governing ecosystem functioning. Under top-down control, a change in a top predator species’ abundance has a ripple effect on animals and plants lower down in the food web. This is known as a trophic cascade. Top-down, rather than bottom-up, control is now recognized as the stronger driver of food web processes in many systems, highlighting the important role that top carnivores play in maintaining food webs. The widespread extermination of large carnivores, such as wolves, bears, and cougars in North America, and the resulting shifts in vegetation due to overgrazing by herbivores (Beschta and Ripple 2009) is but one example of...
the large-scale effects on ecosystem structure arising from the dismantling of food webs.

The sustainability of an ecosystem is determined by both the species composition of its food web and the degree to which species are connected via feeding dependencies. While the sheer diversity of species alone was originally considered to be a fundamental driver of stability, it is now clear that the robustness of a food web is more related to its connectance, or the number and strength of feeding interactions between species, than to species diversity. In other words, the loss of any single species threatens the sustainability of an ecosystem because it reduces connectance between members. Not all species affect stability equally, however. Food webs are composed of many weak and a few strong interactors. Intuitively, the strong interactors should play the greatest role in determining food web structure and ecosystem functioning and accordingly the loss of a strongly interacting species can drastically alter ecosystem function and threaten sustainability. For example, keystone predators are species that play significant roles in maintaining species diversity, and their removal from food webs can substantially shift interactions between other species, leading to changes in species composition, even to the point of ecosystem collapse. These species tend to be large, rare vertebrate consumers that are also most

![Figure 1. An Example of a Food Web](image-url)
vulnerable to human threats from hunting and habitat degradation (Duffy 2003; Estes et al. 2011). Human-driven extinctions have dramatically skewed the structure of food webs by removing the higher-level species from them. These losses have caused complex rippling effects on ecosystem function, and as a result conservationists have made efforts to protect and restore these species; the reintroduction of wolves to Yellowstone National Park in the United States is one example. Counterintuitively, however, it is becoming apparent that ecosystems may be most sustainable when many weakly interacting species counterbalance the effects of a few strongly interacting species (McCann, Hastings, and Huxel 1998). In combination, many weakly interacting species can serve as important threads that hold ecosystems together. Efforts to restore and protect only the rare, strongly interacting species thus may be insufficient to protect the long-term sustainability of ecosystems.

**Human Effects**

People often recognize the importance of top predators in ecosystem sustainability only after the predator population is severely reduced or completely lost. For example, in the Scotian Shelf ecosystem off Nova Scotia, Canada, commercial overexploitation caused cod and other bottom-dwelling fish predator populations to crash in the late 1980s (Strong and Frank 2010). The decline in top fish predators generated a series of trophic cascade effects, including an increase in the abundance of prey species such as the northern snow crab and northern shrimp and a decrease in zooplankton due to the larger number of fishes. Top predator levels were also affected; large shark populations such as hammerhead sharks declined up to 99 percent in the western Atlantic due to loss of a significant food species. Many marine systems have recovered from overfishing disturbances within twenty years (Jones and Schmitz 2009), but despite a ban on cod fishing since 1993, the Nova Scotia ecosystem shows no sign of return to its original state. Unsustainable overfishing may have changed the interdependencies among species to the point that food web structure and dynamics shifted permanently, with unforeseen impacts on ecosystem function and species survival.

Besides direct impacts such as hunting and overfishing, humans have affected food web sustainability indirectly through urbanization. A prime example of this is so-called *mesopredator release* in southern California, where land fragmentation has caused a decline of the apex (top level) carnivore—the coyote—and the rise of smaller carnivore species (mesopredators). Apex predators can suppress mesopredators by killing them for food and by instilling fear, which changes mesopredator behavior, habitat use, distribution, and abundance. As a result of coyote decline, increasing numbers of mesopredators such as skunks, raccoons, foxes, opossums, and domestic cats killed more native birds, driving some species to extinction and significantly reducing bird diversity (Crooks and Soule 1999). Even without directly removing top predators, human influences on the environment can change the nature and strength of feeding dependencies between species, generating unsustainable predation pressure and indirectly causing local extinctions.

**Sustaining Human and Natural Systems**

Conserving top predators is essential for maintaining ecosystem function and biodiversity, yet it can be logistically challenging and ethically controversial. Large carnivores range across broad, heterogeneous landscapes and seascapes that are often difficult to protect alongside expanding human development. Furthermore, top carnivores attack humans and livestock and generate substantial losses to the livelihoods of people living adjacent to protected areas (Woodroffe, Thirgood, and Rabinowitz 2005). They are viewed as pests in many parts of the world and are exterminated by the millions to protect game animals and domestic livestock. Understandably, farmers and villagers often resist conservation efforts involving the reintroduction of large carnivores. The continuing controversy regarding protection of wolves from renewed hunting now that they have rebounded in number in the greater Yellowstone Park area and adjacent states is a prime example of the social and political complications involved in conserving top predators in human-dominated landscapes (Musiani and Paquet 2004; Taylor 2011).

The preservation of food webs is crucial not only for ethical reasons but also for sustaining natural economies and preventing widespread poverty and food insecurity. The interconnectedness of large food webs can lead to unexpected declines in resources and shifts in livelihoods across the world. The collapse of the cod fishing industry in Nova Scotia, for example, led European Union fleets to shift to the oceans of western Africa, where they competed with local fisheries. The resulting decline in fish harvests by local fisheries motivated people in Ghana to turn increasingly to an unsustainable bushmeat (jungle animal) trade for income and food, which in turn caused sharp declines in forty-one tropical wildlife species (Brashares et al. 2004). Globalization has tightened the interdependency between international biological conservation, food security, and economic development to the extent that protection of food web linkages is now mandatory for the survival of human and natural systems alike.
Future Applications

As humans continue to influence natural processes, the food web is used increasingly as a tool for understanding how global ecosystems change. In addition to overexploitation and habitat fragmentation, climate change is now acknowledged as a major factor in trophic interactions. Climate shifts often create situations in which animals’ life cycles are out of phase with their accessibility to food. For example, empirical evidence indicates that faster spring warming in the High Arctic of Canada has led to a mismatch in timing between the arrival of migratory caribou herds and the growth of their plant foods, which results in greater calf mortality (Post and Forchhammer 2007). In a European forest ecosystem, where spring has been arriving earlier over the past twenty years, caterpillars and predatory songbirds have adapted to the twenty-day advance, but the top predators—sparrow hawks—have not adjusted to their prey’s life cycle development. Consequently, the hawks are unable to provide enough food for their young during breeding season. Fewer offspring survive, and the number of hawks overall is declining (Both et al. 2009). Analyzing food webs reveals that human influences on a single species can cause rippling effects throughout the food web network that change the ways individual animals behave and how populations change.

As the ecological and conservation sciences broaden from a species-specific to ecosystem-based understanding of natural processes, food webs will continue to aid in conceptualizing connections between species. Global interconnections between human economies and natural biodiversity indicate that people are intricately linked with environmental systems. To maintain ecosystem function and biodiversity, the integrity of food web structure, particularly the presence of top predators that drive top-down control of system processes, must be preserved.

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See also Agroecology; Biodiversity; Community Ecology; Complexity Theory; Ecosystem Services; Global Climate Change; Habitat Fragmentation; Hunting; Keystone Species; Microbial Ecosystem Processes; Nutrient and Biogeochemical Cycling; Plant-Animal Interactions; Population Dynamics; Regime Shifts; Resilience; Wilderness Areas

FURTHER READING


As our understanding of the ecosystem services provided by forests (such as water treatment and carbon sequestration) continues to grow, so does public interest in protecting these systems. Sustainable forest management balances the dynamic social and environmental requirements placed on forests against the needs of future generations. Maintaining the health of forest ecosystems for future generations requires consideration of forest biodiversity, soils, and hydrology.

In the late twentieth century, following widespread degradation and fragmentation of forest ecosystems in developed countries and decades of tropical deforestation, the international community observed the need to promote and maintain the biological diversity of forests. In 1992 the United Nations Conference on Environment and Development held in Rio de Janeiro (also called the Rio Summit or the Earth Summit) emphasized the need for sustainable forest management (SFM) as one of the means to address these concerns. Following this, the Helsinki Process and Montreal Process were initiated to develop a framework of criteria and indicators to evaluate forest management. These developments have attempted to shift forest management practices from those focused on sustainable yield toward a more holistic forest management approach. Historic forest management approaches were primarily focused on yield at the jeopardy of other ecosystem components, such as soil fertility and biodiversity. In contrast, sustainable forest management relies on meeting the needs of the present generation without compromising the ability of future generations to meet their own needs. Conceptually, SFM is rather well defined, but its implementation and data to support this management approach are in their infancy.

Why Practice Sustainable Forest Management?

Forest ecosystems directly and indirectly serve a variety of important functions. Ecologically, forests provide habitat and cover for wildlife and pollinators and thereby represent areas that are rich in biodiversity. Forests produce most of the oxygen needed to sustain life on Earth. They also sequester carbon that would otherwise exist as carbon dioxide in the atmosphere. They stabilize soils and slopes against erosive forces, regulate climate, mediate hydrology, and maintain water quality. In addition to a steady stream of wood products, forests also provide a wide array of nontimber products, such as fruits, oils, and medicines, forming an essential component of the world food supply and contributing over $90 billion worth of nontimber products each year (Pimental et al. 1997).

Furthermore, forests provide a wide range of aesthetic, recreational, and other socioeconomic benefits and values. Sustainable forest management approaches seek a balance between providing timber yields and ensuring that forest ecosystems can provide these other services.

Historically, humans have directly relied on forests for the timber, fuel, and wildlife contained within them. The scarcity of wood fuel in Europe prompted the use of coal at the onset of the Industrial Revolution in the late eighteenth century and spurred the so-called carbon economy. The rapid global population growth underway in the twenty-first century will likely put continued pressure on forest ecosystems in the form of increased fragmentation and increased demand for the resources therein. In response to this increased demand for forest resources, forest managers will be faced with the daunting task of producing a greater yield of wood products from a finite land area. Even though per capita wood
consumption is declining, and is projected to do so well into the twenty-first century, the world’s population has nearly doubled since the 1950s, resulting in an 80 percent increase in the demand for wood fuel (Hammett and Youngs 2002). The growing demand suggests that the future of the forestry industry must be dynamic and innovative while weighing the economic, social, and ecological costs and benefits. This is particularly difficult due to the fact that successional forest dynamics occur over long time periods (decades to centuries), which can be marked with stochastic disturbances (fires, pests, wind damage). Additionally, our understanding of forest ecosystem dynamics and impacts of disturbance events on these systems remains incomplete.

In spite of uncertainty, however, a number of trends are apparent. First, the amount of forest land will continue to be reduced and more fragmented as populations increase and suburban areas increase. Forest land must therefore become more productive, whether this is through increased biomass accumulation and yield or through the development of better wood processing technology, or both. Since the 1960s, we have seen the more efficient use of a wider range of forest-based resources that are used in engineered wood products like oriented strand board and particleboards, which rely on low-quality wood from trees that mature relatively quickly. Additionally, timber production is expected to increase with advances in fire control. The United States has already witnessed a 50 percent increase in forest output between 1961 and 2000 due to advances in fire control and better silvicultural practices (Hammett and Youngs 2002).

Society’s growing wood demands will likely be met through the increased exploitation of forest ecosystems from abroad. Unfortunately, these will most likely involve previously unlogged areas, thus perpetuating the growing problems of fragmentation, extirpation of native species, and land degradation. Land managers and the timber industry must work to develop sustainable logging techniques and practice adaptive management to incorporate new information as it becomes available (Johnson 1999). It is important to keep in mind that forestry is only practiced with societal approval, so forest managers must appeal to people’s interests, providing not only goods but services.

**History of US Forest Management**

Forests and the natural resources therein represent an important economic and environmental asset upon which humans have relied for millennia. Prior to the colonization of North America, Native Americans lived in close association with forests, utilizing them for fuel, berries, wildlife, and other materials. Following colonization, these forests were primarily exploited for their timber. The wood materials and associated goods produced by the lumber industry drove the development of the infrastructure and economy of the maturing United States during the eighteenth and nineteenth centuries. Further reductions in forest cover were caused by the clearing of forests for agricultural land. From 1630 to 1930, poor land management practices depleted the inventory of sawtimber (live trees suitable for sawing into lumber) by over 70 percent from 26,535 to 7,708 million cubic meters (Birdsey, Pregitzer, and Lucier 1930). It was not until the creation of the US Division of Forestry in 1881 (which became the US Forest Service in 1905) that an agency was enacted to develop a coherent management plan encompassing the nation’s forests. Largely shaped by the destructive practices of early timber industrialists, forestry in the United States adopted sustained timber yield principles first developed in Europe. While these principles have been effective at sustaining yield, evidence suggests that they need to be revised if sustaining the forests’ integrity is a management objective.

**Sustainable Standards and Forest Certification**

In the 1990s international policy makers recognized the need to conserve forest ecosystems and developed a system of standards to promote environmentally focused forest practices. They turned to forest certification as a voluntary market-based approach to promote the multidimensional (economic, ecological, social, cultural, and spiritual) value of forests, reasoning that consumers concerned with forest degradation and deforestation will prefer to purchase timber products from certifiably well-managed forests.

Given the ecological and economic objectives of SFM, policy makers at the Helsinki Process and Montreal Process developed lists of criteria and indicators (C&I) for assessing the success of forest management. Specifically, the Montreal Process, which focused on countries in temperate and boreal forest zones, defined the following criteria: (1) conservation of biodiversity, (2) maintenance of productive capacity of forest ecosystems, (3) maintenance of forest ecosystem health and vitality, (4) conservation and maintenance of soil and water resources, (5) maintenance of forest contribution to global carbon cycles, (6) maintenance and enhancement of long-term multiple socioeconomic benefits to meet the needs of society, and (7) existence of a legal, policy, and institutional framework that facilitates
Sustainable forest management (Fujimori 2001, 242). Associated with these criteria are sixty-seven specific indicators that are helpful for defining SFM components. While the C&I are clearly defined, they contain no targets, timetables, or performance requirements (Ramatsteiner and Simula 2003). The Helsinki Process, which involved European countries, is largely comparable to the Montreal Process except that it lacks a legal, policy, and infrastructural framework. Over 150 countries have participated in one or more of these processes (FAO 2001).

In forest certification, an independent third party assesses the quality of forest management relative to available standards and offers written assurance that standards are being met. Due to the market-based nature of sustainable forest management, the interests of the certification organizations have largely driven forest certification standards. Because of this, such standards, which can be regionally specific, are variable between organizations, reflecting the different stakeholder viewpoints.

Sustainable forest management standards seek to ensure that certified forests will be managed to address timber and non-timber forest values, maintain forest productivity and biodiversity, and protect soil and hydrology, in addition to providing cultural, aesthetic, and recreational value. Over the last decade a number of major forest certification programs have been established to address these specific management objectives. The major ones—Canadian Standards Association (CSA), Forest Stewardship Council (FSC), Programme for the Endorsement of Forest Certification (PEFC), and Sustainable Forest Initiative (SFI)—have similar structural characteristics, including the use of third-party auditors, a chain of custody, public reporting, stakeholder consultation, and on-product labels. While many of the criteria to compare forest legislation and certification standards are similar, discrepancies and inconsistencies may provide an uneven playing field in international markets (Wood 2000).

In total, these forest certification programs have certified greater than 239 million hectares comprising over 9 percent of the global forest area as of July 2011 (PEFC 2011). Barring disruptions in the global economy, which could reduce the willingness of landowners to participate, the continued steady growth of forest certification seems likely. While the forest certification process has been successful at recruiting larger landowners interested in certifying their lands, it has left out smaller landowners not capable of covering certification costs, and the overall ecological and social benefits remain an area of concern. As of 2002 the vast majority of certified lands were in Europe and North America, leaving the tropical areas for which this program was developed largely underrepresented (Atyi and Simula 2002). The existing complex certification systems may be displaced by simplified approaches that focus on single issues or products. Forest management standards that can address these issues may prove more resilient.

Sustainable Management Practices

Sustainable forest management requires that forests be managed not only for wood production but also for their biodiversity, hydrology, recreational, and other values. Management of these components must be evaluated across the ecological landscape, which can be viewed as a shifting mosaic of ecosystem characteristics. At the regional level this requires maintaining ecosystem processes such as nutrient cycling, while at the landscape level this management will focus on ecosystem integrity or the range of historic biological diversity.

Traditional silvicultural techniques focused on productivity may be insufficient to meet the multiple objectives of SFM, and new management approaches may need to be developed. Public pressure has been successful at encouraging many forest managers to pursue alternatives to traditional clear-cutting methods, which practice the complete removal of valuable lumber. To meet the objectives of SFM, forest managers may employ...
lower-impact harvesting techniques such as variable retention, single tree selection, or shelterwood methods. Variable retention methods were adopted from the idea that natural disturbances regularly alter forest landscapes, creating a mosaic of forest patches at different successional stages. The variable retention approach allows the selection of key structural elements of forest ecosystems to be retained for at least one rotational period to achieve particular management objectives, with tree selection and the duration of retention depending on those objectives. Trees can either be retained in high-density groups throughout an area, creating patches of core wildlife habitat, or uniformly over the landscape to aid regeneration and seed dispersal. The variable retention method is largely comparable to the single tree selection approach in which trees are selected for removal if their canopy is damaged (via insects, disease, or wind) or their crown is not well developed, inhibiting high photosynthetic rates and thereby growth.

Shelterwood harvest is another silvicultural technique used to meet SFM objectives. This technique is mainly used in situations where wildlife, soil erosion, and/or specific distribution of species regeneration are of concern. In this approach, tree harvesting occurs in two or more series where larger trees are conserved as a seed source for saplings and to provide cover from harsh climate phenomena. This method is able to target species for regeneration by retaining a seed source for new seedlings and can be used to redistribute the relative dominance between present species. The retention of larger trees throughout the shelterwood harvest process prevents erosion, and when coupled with multiple harvests, this technique will perform well on steeper topography. Regardless of harvest approach, it is required in SFM projects that the rate of harvest should not exceed the rate of regeneration; therefore proper accounting must be conducted to ensure this balance.

Unlike traditional silvicultural techniques that rely on stand height and spatial regeneration methods, SFM has identified the following criteria as being important: stand structure, regeneration method, tending methods, rotation period, objectives of the management, the potential natural vegetation and present forest type, social circumstances, and the desired forest type or stand development stage required to fulfill the desired forest function (Fujimori 2001). Broadly applicable SFM techniques are difficult to prescribe due to the complexity of forest ecosystems and associated external factors, such as climate, invasive pests, and disease. Additionally, the diversity of forest types in different regions of the world will require unique SFM approaches. The fact that SFM programs operate in political and socioeconomic settings also complicates the proscription of broad management approaches.

Going Forward

International consensus has identified sustainable forest management as a means to conserve forest ecosystems and maintain a yield of forest products for generations. While sustainable forest management is conceptually straightforward, its implementation is considerably more challenging. Specifically, our understanding of forest ecology is limited, and compromises must be made to permit the extraction of timber. Additionally, sustainable forest management is difficult to make operational because definitions of sustainability are dependent on the time frame and spatial range that are considered. A possible precautionary approach that the international community might take to forest management would be to set aside a network of protected forest preserves to serve as a repository for biodiversity (Groves 2003, 216–259); such areas would be useful from an experimental perspective to assess the success of sustainable forest management approaches. If sustainable forest management practices are to be successful at conserving forest ecosystems, new management approaches and technologies to rapidly assess forests will need to be developed. Additionally, market-based systems that value ecosystem services derived from sustainably managed forests (i.e., biodiversity, water filtration, pollutant reduction, and reduced erosion) need to be developed before the value of forests can truly be realized.

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See also Biodiversity; Biological Corridors; Buffers; Ecosystem Services; Fire Management; Habitat Fragmentation; Hunting; Hydrology; Indigenous Peoples and Traditional Resource Management; Microbial Ecosystem Processes; Natural Capital; Population Dynamics; Reforestation; Resilience; Soil Conservation; Tree Planting; Wilderness Areas

Further Reading


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The factors and processes involved in climate change are many and complex, ranging from fluctuations in the Earth's orbit to changes in biota. The Earth's climate has always been in flux, but indications are that human impacts from industrialization, land-use change, and growing population are speeding a warming of the planet that could have substantial effects on ecosystems and the services they provide.

Global climate change refers to alteration in climate (temperature, precipitation, and wind patterns) over a significant area lasting for an extended period. The complex set of processes involved in climate change includes impacts from land use (ice coverage and vegetation shifts, deforestation, development, urbanization, infrastructure deployment), natural and human-induced forcing factors, and feedback processes within the climate and Earth systems. It has long been recognized that the Earth's climate is in constant flux and that human activity can induce change, but the apparent complexity and underlying drivers of climate change have only come to light during the past century aided by technological advances and accumulated evidence. Population and economic growth are the major anthropogenic (human-generated) drivers of change in natural resources, land use, and their climate feedback processes. This discussion addresses forcing factors for global climate change and associated feedback mechanisms.

Climate

Climate is primarily regulated by the amount of energy absorbed and dissipated by the Earth's surface. Incoming shortwave radiation emitted by the sun passes through the atmosphere and strikes the Earth's surface. There it is either absorbed or is reflected as longwave radiation, depending on the albedo (the reflective property of the surface, including cloud cover) at that location. Some of the reflected radiation is trapped by greenhouse gases in the atmosphere (e.g., carbon dioxide $[CO_2]$, methane $[CH_4]$, nitrogen oxides $[NOx]$, and water vapor), resulting in what is known as the greenhouse effect. This effect is largely responsible for Earth's average surface temperature of approximately $15^\circ C$, to which we have grown accustomed; removal of greenhouse gases would reduce the average surface temperature to about $-18^\circ C$.

The amount of energy Earth receives from the sun varies with latitude. The sun's rays hit the equator directly, causing tropical regions to receive a large amount of energy. At higher latitudes, the same incoming solar radiation is distributed over a larger surface area of the Earth, creating the temperate and polar zones. The uneven distribution of heat across land and oceans fuels atmospheric circulation (Hadley circulation), thereby creating climate and precipitation patterns across the planet. This translates into weather patterns that develop in the lower atmosphere and are driven by incoming heat from the sun, the Earth's rotation, and heat stored in oceans and the atmosphere. The storage capacity of heat in the atmosphere is a function of the relative amount of the incoming radiation that can be absorbed by the different greenhouse gases in the atmosphere. Large water masses (oceans) have a high capacity to store heat and therefore cool and warm very slowly.

The temperature differences between land, oceans, and air ultimately drive climate and explain temperature and precipitation patterns along the Earth's surface in a predictable fashion. As a consequence of this predictability, biota adapts to geographical locations of the Earth in highly recognizable forms such as tropical, temperate,
Recent Change

Satellite, weather balloon, and ground observations all agree that there has been a steady warming trend in the Earth’s surface temperatures and that it has been more apparent over the course of the nineteenth and twentieth centuries. Based on meteorological data, the twentieth century can be divided into three sections: early twentieth-century warming, a mid-century cooling episode, and late twentieth-century warming (Anderson, Goudie, and Parker 2007).

Descriptions of early twentieth-century warming documented the changing time periods marked by the occurrence and intensity of first and last snowfalls or ice covers: for example, they noted how the snow period declined from 150 days to 113 days in London, or when the period of time during which ice cover in the Arctic Sea prevented navigation shortened from 12–13 weeks to 3–4 weeks per year, or when polar ice thickness declined 20–40 percent depending on location. During the early twentieth century sea temperature records reveal about a 1°C–2°C increase until the 1960s in northern latitudes. These increases in temperature were corroborated by independent biological observations, including the northward expansion of cod, halibut, or haddock in Greenland, displacement of fish by warm-adapted species in the southern limits (though overfishing has contributed somewhat to these effects), and the northward range shifts of plant species and birds, including the invasion of tundra by trees between 1920 and 1940. This warming period also impacted agricultural and silvicultural practices, as the number of growing days increased, and cultivation of rye, barley, or oats expanded into high latitudes in Scandinavia (expansion not caused by breeding) (Anderson, Goudie, and Parker 2007).

The mid-century cooling period occurred between about 1945 and 1970 on land and 1955 and 1975 in the oceans. Unlike the early twentieth-century period, when 85 percent of the Earth’s surface experienced warming, during the mid-century cooling period 80 percent of the total Earth surface area experienced cooling. During this period glaciers stopped retreating, snowbanks were formed in the Canadian Arctic, snowfall increased in Europe, Baltic Sea ice increased, and the plant-growing season was documented to be shortened in parts of northern Europe.

The late twentieth-century warming period is characterized by a rapid increase in temperatures over continents at mid latitudes (40–70 degrees north). Temperatures rose 0.6°C in about two decades, the fastest and largest increase in temperature known over the last thousand years. This warming trend has primarily affected night-time temperatures as increased cloud cover has contributed to reduced diurnal temperature oscillations. During this current period similar climatic and biological trends that characterized the early twentieth-century warming period have been observed. Glacier retreat and melting of the permafrost (at about 4–5 kilometers a year) have been particularly well documented. Also, the onset of spring for both plant and animal life is occurring five to eleven days earlier than indicated in the historical record.

In addition to increases in temperature during the twentieth century, global precipitation has increased by about 2 percent in response to the higher evaporation rates of ocean waters. The magnitude of rainfall events has increased in many areas of the Northern Hemisphere and Australia. The increase in precipitation at northern latitudes is contrasted with decreased precipitation and increased aridity at low latitudes, particularly in northern Africa and Asia, indicating that climate shifts will not be uniform. Much of the variability observed in precipitation patterns is also related to the El Niño Southern Oscillation (ENSO), the complex of warm ocean current and associated atmosphere that influences continental climate in many regions of the world.

Multiple lines of evidence indicate changes in climate over the last 150 years. Debate continues, however, on what is causing the temperature and precipitation changes since the late nineteenth century. Changes in atmospheric chemistry due to human activities that can lead to both warming (greenhouse gases) and cooling (aerosols) seem to explain a large part of the surface temperature oscillations at short-term scales.

Natural Drivers

The Earth’s climate has continuously changed during the planet’s history. In the past, climate was largely impacted by natural physical, chemical, and biological processes and the feedback between these. Tectonics, which creates land and moves continents on the Earth’s surface, clearly influences climate, but because continental movement is very slow, tectonics alter climate over tens of millions of years. Over the last 2–3 million years climate has changed more rapidly, with spans of tens of thousands of years forming cold (glacial) periods and warm (interglacial) periods. These climate changes can largely be explained by planetary forcing agents that affect the amount of
incoming energy from the sun hitting the Earth’s surface. The theory of orbital forcing developed by the Scottish scientist James Croll in the 1860s and advanced by the Serbian civil engineer and mathematician Milutin Milankovitch in the 1920s describes how the eccentricity, axial tilt, and precession of the Earth’s orbit in relation to the sun drive glacial–interglacial variations (Imbrie and Imbrie 1979). Slight variations in these parameters directly impact the amount of solar radiation reaching the Earth and subsequently impact the seasonality and location of solar energy.

The Earth and all other planets in our solar system orbit the sun in an elliptical manner. The eccentricity of the orbit, or the departure of the ellipse from circularity, is determined by the interactions between the gravitational fields of Jupiter and Saturn. The ellipticity of Earth’s orbit varies from 0 to 5 percent on a cycle of roughly 100,000 years. Variations in eccentricity account for how far the Earth is from the sun and have contributed to historic glacial regimes. The angle of Earth’s axial tilt in relation to its plane of orbit around the sun is responsible for seasonal variation in daylight and temperature. Currently the axial tilt of the Earth is close to 23.5 degrees and decreasing; Earth’s tilt naturally varies from approximately 21.4 degrees to 24.5 degrees on a roughly 41,000-year cycle. Additionally, the Earth’s precession governs how the Earth wobbles as it spins on its axis and operates on a periodicity of about 23,000 years, further modulating seasonality. Evidence from deep-sea sediments and ice cores suggest considerable climate variability is associated with orbital forcing (Imbrie et al. 1992).

Regarding shorter time scales, it has been hypothesized that shifts in the quality (via changes in ultraviolet [UV] range) and quantity (via sunspots) of solar radiation at the Earth’s surface can also result in changes in climate (Lean 2010). Research suggests that the number of sunspots varies on a roughly eleven-year cycle and can alter solar output by approximately 0.01 percent. During periods of high sunspot activity, the Earth receives increased radiation compared to periods with low activity. It is thought that since 1750 increased solar irradiance has been responsible for a positive radiative forcing of 0.06 to 0.30 watts per square meter (W/m²) (IPCC 2007). This is sufficient to contribute to moderate increases in temperature in the upper atmosphere but cannot account for most of the observed increases in surface temperatures.

Volcanic eruptions may also play an important role in the Earth’s climate through two primary pathways: first through the emissions of CO₂ and other greenhouse gases into the atmosphere and second by emissions of aerosols (suspensions of fine particles in gas) such as ash and sulfur gases. Water vapor and CO₂ are the primary greenhouse gases emitted and represent between 50–90 percent and 1–40 percent of annual volcanic emissions, respectively. The water vapor dissipates from the atmosphere rapidly, resulting in a negligible effect on climate, while the magnitude of CO₂ from volcanic origins is less than 1 percent of annual CO₂ emissions (Gerlach 1991). Additionally, ash and sulfur gases are projected into the stratosphere and can contribute to global cooling. These aerosols reflect incoming radiation back to space, leading to cooling of ground surface temperature. Volcanic ash is usually removed rapidly (within one month after the eruption) from the atmosphere by sedimentation (Pinto, Turco, and Toon 1989). Sulfur gases from volcanic activity represent about 36 percent of the annual tropospheric sulfur emissions (Graf, Feichter, and Langmann 1997) and are largely responsible for the climatic effects associated with eruptions because of their longer residence times in the atmosphere and their role of scattering solar radiation back to space.

Additionally, natural fluctuations in Earth’s albedo resulting from shifts in land or cloud cover can impact climate patterns by altering the amount of solar radiation that is reflected or is absorbed by the Earth’s surface. For instance, increased snow cover can increase reflectance and thereby alter the Earth’s albedo, resulting in further cooling. In contrast, increased vegetation can result in what is called vegetative forcing, which lowers the land surface albedo and results in increased absorption of heat, thereby raising surface temperatures.

As previously mentioned, the Earth’s atmospheric and oceanic conditions are closely coupled, and thus alterations in patterns of oceanic circulation can have considerable impact on global climate. The combined effects of heating/cooling and salinity drive the oceanic currents to circulate water throughout the Earth’s oceans. This is known as thermohaline circulation and is responsible, for instance, for warming the North Atlantic regions by as much as 5°C. Evidence suggests that thermohaline circulation has been disrupted a number of times in the past, resulting in considerable alterations of regional temperatures. For example, evidence suggests the Younger Dryas, a millennium-long cold period about twelve thousand years ago, at the beginning of the Holocene, may have been triggered by the release of freshwater into the North Atlantic, altering ocean circulation (Broecker 1997). A shift in the ocean’s thermohaline circulation could occur with increased precipitation at higher latitudes, which would reduce salinity and thereby disrupt circulation (Stocker and Schmitten 1997).

 Anthropogenic Drivers

Humans, like most organisms, modify their environmental surroundings. As such, it is logical that the magnitude of modification by humans has grown in conjunction
with population. The rapid growth of the human population has been fueled by the consumption of natural resources. Extraction of these natural resources is both energy and land intensive. Recently an increasing body of scientific literature suggests that there is compelling evidence that human activities are modifying forcing factors that influence climate (IPCC 2007). These human impacts are due in large part to the increased emission of greenhouse gases through the burning of fossil fuels (such as coal and petroleum), industrial activity, land-use change, and deforestation practices, all of which became prevalent during the industrial development of the past 250 years.

Human activities in large part bear responsibility for the increases in atmospheric greenhouse gases, which alter the Earth’s energy budget through a process known as radiative forcing. Increased concentrations of these gases in the atmosphere contribute to global warming by absorbing energy reflected from Earth and re-emitting this energy, resulting in a net increase of energy. Humans have increased atmospheric CO$_2$ concentrations through fossil fuel combustion (estimated at 7.2 gigatons of carbon [GtC] per year from 2000 to 2005) and to a lesser extent by land clearing (estimated at 1.6 GtC per year during the 1990s) (IPCC 2007). These emissions have increased global CO$_2$ concentrations from preindustrial-era levels of about 280 parts per million (ppm) to about 389 ppm in 2011, far exceeding the range (180 to 300 ppm) from the last 420,000 years as determined from ice cores (Petit et al. 1999; IPCC 2007). Evidence suggests the increased atmospheric CO$_2$ has contributed to the global temperature increase from 1850/1899 to 2005 of an average 0.76°C (range of 0.57°C–0.95°C) (IPCC 2007). These patterns are particularly concerning due to the fact that CO$_2$ has a one-hundred- to two-hundred-year residence in the atmosphere, resulting in potentially long-lasting consequences. Three lines of evidence show that CO$_2$ increases are anthropogenic (Prentice et al. 2001). First, atmospheric oxygen is declining in line with CO$_2$ combustion. Second, the isotopic signature of fossil fuel (lack of carbon 14 $^{14}$C and depleted carbon 13 $^{13}$C) is detected in atmospheric measurements. Finally the increase in CO$_2$ is more rapid in the Northern Hemisphere where the majority of fossil fuels are combusted.

Humans have also contributed to the increase in a variety of other trace gases (principally methane, nitrous oxide, and halocarbons) that may have radiative forcing effects that are comparable to and higher than that of CO$_2$. Humans are now responsible for nearly 70 percent of annual methane atmospheric accumulation as a result of agricultural practices (i.e., livestock farming and rice cultivation), fossil fuel combustion, and decomposition associated with landfills. This has led to an increase in global methane concentrations from about 320–715 parts per billion (ppb) during the preindustrial period to 1,774 ppb in 2005 (IPCC 2007). Although methane has a relatively short residence time in the atmosphere (about twelve years) compared to CO$_2$, it exhibits 3.7 times more global warming potential per mole (Lashof and Ahuja 1990). Nitrous oxides have also increased from preindustrial levels of about 270 ppb to 319 ppb in 2005, primarily due to the burning of fuels at high temperatures such as in factories and cars (IPCC 2007). Finally, concentrations of halocarbons have increased significantly due to their use in synthetic organic compounds. The combined impacts of these trace gases have been estimated between 0.88 and 1.08 W/m$^2$, which constitutes nearly 60 percent of the radiative forcing of CO$_2$ (IPCC 2007).

Unlike CO$_2$ and other greenhouse gases that warm the atmosphere via positive radiative forcing, aerosols cool the atmosphere by reflecting incoming radiation (as in the case of large volcanic eruptions). Aerosols can contain a broad collection of particles with different chemical properties causing each to interact uniquely with the atmosphere. They can attract water and serve as cloud condensation nuclei, resulting in more diffuse clouds, which reflect more solar radiation. Sulfur dioxide produced from fossil fuel combustion and the burning of vegetation is the primary atmospheric aerosol. Aerosols are not long-lived in the atmosphere and are generally localized to the region of production. Although anthropogenic aerosol emissions have declined in North America and Europe due to more stringent regulations, emissions have increased in Asia as urbanization has increased.

Human population growth has relied on the widespread transformation of the Earth’s surface to provide necessary resources, and current consensus suggests that humans have transformed or degraded somewhere between 39 and 50 percent of the Earth’s surface (Vitousek et al. 1986 and 1997). Land surface change through deforestation, reforestation, and urbanization alters the albedo of the Earth’s surface, impacting the amount of energy absorbed. Estimates indicate that the impacts of land transformation on Earth’s albedo accounts for a loss of 0.4 W/m$^2$ (IPCC 2007), and therefore affecting the energy balance of the Earth’s surface.

**Feedback Mechanisms**

To further complicate the variable climate system on Earth, global conditions are also modified by natural and human-induced feedback mechanisms, which operate on complex spatial and temporal scales. Climate feedback can be either positive or negative: positive feedback processes magnify an effect, causing increased warming or
cooling, while negative feedback processes dampen change in climate.

An important feedback mechanism impacting climate is the flux of CO₂ into and from oceans; when global temperatures become warmer, CO₂ may be released from oceans. Increasing CO₂ concentrations may amplify warming by enhancing the greenhouse effect. When temperatures become cooler, CO₂ enters the ocean and contributes to additional cooling. During the last 650,000 years, CO₂ levels have tended to track glacial cycles: during warm interglacial periods, CO₂ levels have been high, and during cool glacial periods, CO₂ levels have been low.

Another important positive feedback process is the natural emission of CO₂ from soils (soil respiration), specifically in boreal ecosystems. As boreal ecosystems experience warming, they will release the large stocks of carbon that are currently immobilized (sequestered) in soils by frost, leading to further increases of CO₂ in the atmosphere. Furthermore, soil respiration has been shown to be correlated with temperature and moisture, such that increases in soil temperature in conjunction with moisture may result in increased natural CO₂ emissions rates from ecosystems (Wildung, Garland, and Buschbom 1975).

Alterations to the Earth’s surface can also result in complex feedback effects on climate. Ice-albedo feedback refers to the lower albedo that snow and ice have compared to ground and vegetation, resulting in increased reflectance of energy into space. Periods of low temperatures allow snow cover to last for a longer duration, causing increased reflectance and the cooling of Earth’s climate, which in turn can result in further expansion of ice cover (positive feedback). This process can also work in reverse, whereby reduced ice coverage creates feedback in which the Earth’s surface warms, resulting in glacial recession. Current scientific consensus indicates that glaciers and ice caps have been losing mass, particularly since the early 1960s (Kaser et al. 2006).

Water vapor feedback in conjunction with other processes can amplify climate change. Despite its short residence time in the atmosphere, water vapor is a potent greenhouse gas. It has a heat-amplifying effect and tends to increase in conjunction with temperature, thus creating positive feedback. But water vapor also forms clouds that block incoming radiation, resulting in the cloud negative feedback effect (Ramanathan et al. 1989). A negative feedback effect by clouds is thought to be partly responsible for the observed moderate increases in surface temperature, which are less than expected given the accumulation of greenhouse gases in the atmosphere.

A study investigating feedback between climate and boreal forest vegetation cover during the Holocene epoch highlights the significance of orbital forcing, vegetation shifts, and feedback between these two parameters (Foley et al. 1994). The study suggests that orbital forcing (i.e., shifts in Earth’s eccentricity, tilt, and precession), while capable of increasing global temperatures by 2°C during the mid Holocene, was not solely responsible for the observed warmer temperatures during the period. Instead, orbital forcing paired with its positive feedback effect on the northward expansion of boreal forests in high latitudes was likely to have contributed to the warmer temperatures observed. While not all feedback mechanisms are understood, research indicates that they are important in determining Earth’s climate.

**Sustainability, Biodiversity, and Resource Management**

It is likely that human impacts on the global climate will continue, as global population and per capita energy consumption continue to rise. These impacts may be beyond the capacity for many ecosystems to adapt naturally, leading to losses of biodiversity and impacting ecosystem function and services. Consequently, climate change will be a major driver of decision making in natural resource management.

Global climate change can impact ecosystems and associated organisms in many ways. For instance, plants and animals may shift their ranges dramatically, moving as much as 6.1 kilometers per year toward the poles (Parmesan and Yohe 2003). The ability of individual species to respond to climate change will, however, likely be impeded by human-induced land habitat fragmentation, which breaks down ecosystem connectivity and produces isolated islands of habitats (Honay et al. 2002). An additional change may be in timing, specifically in the advancement of spring events in temperate ecosystems, which is occurring 2.3 days earlier per decade (Parmesan and Yohe 2003). In coastal areas, ecosystems and urban systems will need to adapt to increases in sea level. These changes may not occur uniformly across the globe, however, but may occur faster in coastal areas or near the poles where temperature changes are more rapid.

As species shift in range, timing, and composition, ecosystem function will also change. The potential degradation of ecosystem function will threaten the ecosystem services provided to human society by nature, resulting in increased economic costs for these services. The impacts of global change on humanity will be contingent on society’s ability to adapt to this change. Such an adaptation may necessitate coordinated ecosystem management across geopolitical boundaries to minimize the global impacts of climate change—a significant challenge considering the uncertainties regarding the rate and magnitude of change that has created difficulty in garnering international support to curb climate change.
A proposed adaptation technique is active ecosystem management. For instance, humans can mediate natural CO₂ storage by speeding up the rate of sequestration and reducing the release of already stored carbon. This may be accomplished through reforestation efforts or by increasing the growth rate of forests and practicing no-till agriculture. Additionally, deepwater or geologic sequestration of CO₂ may provide an alternative means to reduce atmospheric greenhouse gas concentrations.

Making informed ecosystem management decisions regarding global climate change comes with many challenges. Uncertainty surrounds the rates and magnitudes of change, due in part to the availability of data sources and the fact that many observations cannot be tied to a particular sampling station (Berliner 2003). Furthermore, considerable uncertainty arises from the fact that scientists have not yet solidified their understanding of many of the driving forces behind climate change models. A further area of uncertainty arises from not knowing how humans will continue to impact the Earth. Future use of fossil fuels, land-use change, and population increases are all variables, and they are largely dependent on societal decisions.

Recent climate change has the capability to dramatically alter the state and functioning of natural ecosystems. Much more must be learned quickly about the functioning of and the change in these systems in order to mitigate damage. By improving our understanding of natural ecosystems and their links to climate, we can be better prepared to properly manage our resources for future generations.

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See also Biodiversity; Biodiversity Hotspots; Carrying Capacity; Coastal Management; Complexity Theory; Ecological Forecasting; Food Webs; Human Ecology; Nitrogen Saturation; Regime Shifts; Resilience; Safe Minimum Standard (SMS)

**FURTHER READING**


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Groundwater is essential to life. It plays an important role in water and food safety for many peoples in rural settlements and cities worldwide, as well as in irrigated agricultural land. Groundwater is also a part of the hydrologic cycle and is intimately linked to surface waters and aquatic ecosystems. Although there are not comprehensive and reliable statistics regarding populations’ dependence on groundwater (i.e., water beneath the Earth’s surface, which often seeps from saturated soil between rocks to become the source for springs and wells), there are examples that show its importance. For example, over 2 billion people worldwide (with at least 1.1 billion in Asia and 175 million in Latin America) are estimated to be directly dependent on underground sources for their potable water supply, including those living in 12 out of the 23 megacities in the world (i.e., cities with a population of 10 million or more) (Howard 2006).

Groundwater resources are extensively and intensely exploited in many regions, particularly in cities and irrigated agricultural areas. Such misuse, however, is generally carried out in an unplanned or uncontrolled way, causing problems of overexploitation. Another serious problem that puts this vitally important resource at risk is contamination resulting from human activities that degrade the generally good natural quality of groundwater.

Surface- and Groundwater: One Resource

Aquifers (groundwater-bearing units) and rivers, inherently different types of water resources, can nevertheless supplement each other. If they are well managed and treated as one resource, their joint capacity can be used to the fullest. For example, aquifers generally work more effectively as sources of stored water, because aquifer recharge rates—the rate at which water is returned to an aquifer—are generally very slow. As a result, aquifers generally yield far less water, especially on a sustainable basis, than rivers, even though a river’s surface waters are replenished every time it rains. By integrating groundwater and surface water management, users can, for example, depend on aquifer use during dry seasons, while during wet seasons count on surface water for a water supply as well as a source for recharging the aquifer.

If well managed and protected, however, aquifers have an advantage over rivers and surface reservoirs due to their lower vulnerability to pollution (although, once polluted, aquifers tend to be harder to clean up than surface water). Passage of water through the soil layer and underlying rock strata to the water table can attenuate some but not all types of pollution (Foster and Hirata 1988). Deep groundwater typically has a much slower response to polluting activities on the surface, often many years. For some shallow or fissured groundwater sources, however, the response can be much faster, sometimes as little as a few days.
Quantity Protection

Overexploitation of a groundwater resource occurs when pumping water from the aquifer causes: a) exhaustion of the aquifer (generally associated with the withdrawal rates higher than the recharge rate for long periods of time or when closer wells provoke unsustainable hydraulic interferences); b) a prohibitive increase of groundwater pumping costs because wells need to be deeper, pumps need to be replaced, and more energy is required; c) impacts on sensitive ecosystems, such as when the contributions of groundwater (i.e., the baseflow) to rivers, lakes, wetlands, and associated flora and fauna is reduced; d) contamination of the aquifer, including by saltwater intrusion into coastal aquifers; e) social inequity, generally associated with the exploitation of the groundwater resource through large wells, reducing the water availability to small users; or f) land instability, subsidence, and geotechnical problems.

Sustainable use of aquifers requires that demand for the groundwater match the aquifer’s capacity, taking into account its role in the environment. A smart way of managing water, furthermore, should consider the characteristics of the various sources of water (water matrix) and make the most of each resource, minimizing the social, environmental, and economical expenses. When it comes to groundwater, then, any management program should initially identify the resource, its production capacity, the reliance on it, and the possibility of being replaced by other sources of water.

The assessment of the current and future demand for water compared to the capacity of the aquifer will allow managers to identify areas where overexploitation is occurring or is likely to occur. The evaluation of areas with potential for overexploitation and the results of those evaluations when integrated with public policies, help to prioritize areas where corrective or preventive actions should be undertaken.

The prevention and correction of problems should be based in actions taken by the state (i.e., local and national government). Instruments for controlling drilling and installation of wells and pumping rates, based on licenses and authorizations, should be implemented in priority areas. Since some members of the civil society (a term somewhat broadly defined by the World Bank to include nongovernmental and not-for-profit organizations with a presence in public life) are private users of groundwater, good management of groundwater use should necessarily involve their participation, in the form of social communication, shared information, and consultation. The water user could therefore indicate problems and inadequate use of the resource. This matter is particularly sensitive in developing countries, where traditionally there is not a relationship of trust between the civil society and the state, and the obedience to laws and state control are limited.

The monitoring of groundwater exploitation (and its evolution) can serve as an essential tool for controlling parties to evaluate the effectiveness of the instruments applied to achieve groundwater management best practices.

Quality Protection

Two complementary tools are often used in the protection of groundwater against contamination. These tools do not assess or consider problems that are neither related to natural water quality from hydrochemical anomalies of the aquifers nor those directly related to overexploitation.

The first tool, which aims to protect groundwater sources, is based on mapping the vulnerability of aquifers and evaluating the existing or future anthropogenic contaminant load. This type of mapping identifies activities that will threaten groundwater quality while also assessing the vulnerability of the aquifer in an area. The vulnerability of the aquifer is defined by a combination of factors such as permeability of the material covering the aquifer, distance from surface to the top of the aquifer (i.e., the thickness of the material covering the aquifer), and others.

The vulnerability map will therefore chart the vulnerability factors and the contaminant load. This combination will enable management experts and planners to identify areas or activities with greater risk of aquifer contamination. Specifically, highly vulnerable areas with high potential contaminant loads generate areas with greater risk of degradation (Foster and Hirata 1988).

The other tool is defining Source Protection Areas or SPAs (called wellhead protection areas in the United States), where perimeters of protection against the installation of potentially contaminating activities are delineated around wells or springs where water is used for public supply, and which are increasingly restrictive with the proximity to the water supply source. In this case, the protection does not apply to the whole aquifer, but a portion of it, associated with the supply to a specific population.

The SPA delineates a surface area that contributes to recharging a certain well or specific water source. Because many of these areas are extensive, it is common practice to apply subdivisions to the delineated area based on the distance to the well or source, or the travel time of persistent or mobile contaminants to groundwater. Therefore, the SPA is commonly divided in distances as short as 10 meters to as long as 2 kilometers or in times as short as fifty days to as long as ten years in the upgradient direction from the well or source.
There are various techniques for delineating an SPA. Some techniques are simply based on the definition of an arbitrary fixed radius (based on the experience of a hydrogeologist) and some are based on complex numerical models of flow and contaminant transport.

These two tools enable the identification of areas where the aquifer presents greater or lower susceptibility of being degraded by any anthropogenic event and can serve as land-use planning tools for a specific region. Contamination control aims at avoiding the installation of potentially harmful or toxic activities in specific areas that contribute to the flow of water to the well, or in areas of high vulnerability.

In land-use planning, when there is incompatibility between the activity and the zoning, based on vulnerability or SPA of public water supply sources, an assessment of the activity should be carried out. If the activity is incompatible with protecting the groundwater resource, various measures can be required: a) relocating the activity to a more adequate area; b) reducing the risks of generating contaminant loads to the aquifer, by implementing effluent treatment systems, adequately storing products, and disposing wastes; c) being ready for treating the contaminated water of the aquifer; and d) substituting the water source.

**Institutional Arrangements**

A “virtuous” (as described in a World Bank briefing about groundwater management, and as compared to the “vicious circle” created by supply-driven groundwater development), must be created for the correct management of groundwater resources, especially in developing countries (Tuinhof et al. 2006). As a result, it is necessary to recognize that managing groundwater involves managing people (water and land users) as it involves managing water (aquifer resources). In other words, managers must recognize that the socioeconomic factors (demand-side management) is as important as the hydrogeological factors (supply-side management) and integrate both into a groundwater management regime (Tuinhof et al. 2006).

Therefore, an adequate aquifer management program should take into account adequate protection measures as well as institutional realities. When considering institutions, some factors should be addressed, such as the construction of a regulatory framework, where the definition of water rights should be clear and separated from property rights; the participation of stakeholders; and the use of economic instruments.

A formal structure created by the government is necessary, and most managers and planners believe it should control and provide technical and financial assistance to agencies that issue and enforce environmental permits. It should consider as well the groundwater resource for installation of new enterprises, and assess those measures already in place. Similarly, a government institution should also oversee well drilling, new well installation, and water withdrawal permits. If two or more government entities divide these responsibilities, there must be clear and good communication between them, as well as with other institutions responsible for land use and economic planning.

In many instances, the fact that a regulating agency defined a management policy that is hydrogeologically and economically sound does not mean that the policy will be accepted and implemented. Stakeholder participation will thus be crucial to overcome resistance to the introduction of logical groundwater management policies. Similarly, the regulatory interventions (such as water rights or permits) and economic tools (such as abstraction tariff and tradable water rights) become more effective if they are not only encoded in water law, but also implemented with a high level of user participation (Tuinhof et al. 2006).

Finally, the public must be informed not only about the negative impacts on the sustainability of the resource caused by overexploitation and contamination, but also that the lack of such management policies can result in real, immediate, financial losses to the community because pumping and water treatment become more expensive.

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*See also* Agricultural Intensification; Catchment Management; Coastal Management; Desertification; Ecosystem Services; Hydrology; Irrigation; Natural Capital; Rain Gardens; Resilience; Shale Gas Extraction; Soil Conservation; Water Resource Management, Integrated (IWRM)

**Further Reading**


Habitat fragmentation is the division of continuous habitats into smaller, more isolated remnants. Fragmentation can directly impact habitat-dependent species by reducing available habitat area, increasing isolation of subpopulations, and intensifying the negative effects of surrounding land use. Fragmentation can also indirectly affect biodiversity and human welfare by exacerbating other threats. Predicting and managing fragmentation effects presents a formidable challenge to sustainable land-use management.

Habitat fragmentation is the process by which habitat loss leads to the subdivision of a continuous area of habitat into multiple smaller remnants isolated from one another by areas of dissimilar land use. Fragmentation typically occurs as a direct result of habitat clearance for human land uses, and it is therefore tightly coupled with the processes of land-use and land-cover change. Together these processes are widely recognized as the principal threats globally to biodiversity, and the study of habitat fragmentation in particular has been among the most active fields of ecological research in recent decades. Moreover, the rate and spatial extent of fragmentation will continue to increase in the foreseeable future as escalating human resource use continues to drive conversion of natural habitats to production land uses.

The direct effects of habitat fragmentation on biodiversity and ecosystem processes have been comparatively well studied. But fragmentation can also have indirect effects, where the impacts of other threats (such as land-use intensification, species invasion, climate change, or overharvesting) are exacerbated or mitigated by the amount or spatial arrangement of remaining habitat. Indirect consequences have been largely overlooked, but they can make the overall effects of habitat fragmentation variable and difficult to predict.

Understanding the circumstances under which fragmentation effects are realized and predicting the magnitude of direct and indirect results will be crucial for managing habitat fragmentation in a way that ensures sustainable production while simultaneously minimizing biodiversity loss and maximizing human welfare.

Global Scale of Habitat Fragmentation

Globally, all major terrestrial biomes have been affected by habitat fragmentation. Forest ecosystems have been particularly affected, with more than 40 percent of the estimated 6 billion hectares of the world’s forests already lost to agriculture and deforestation, with continuing losses of approximately 13 million hectares per year (FAO 2010). Moreover, the absolute area cleared is small compared to the area of remaining forest that is negatively affected by close proximity to anthropogenic land uses. Only one-fifth of the world’s forests remain in contiguous unfragmented tracts (Bryant, Nielsen, and Tanglely 1997), with much of this restricted to the vast areas of coniferous forest across the boreal zone.

Since the middle of the twentieth century, deforestation and fragmentation have occurred predominantly in tropical countries. For example, tropical dry forests have become highly fragmented, with no extensive areas remaining in Africa and nearly 80 percent classified as highly fragmented in Southeast Asia and Australia. Tropical rain forests have also suffered extensive recent deforestation, particularly throughout Southeast Asia, where forest cover is less than 30 percent of its former...
extent, and clearance continues in landscapes that are already highly fragmented. The high rate of forest loss and fragmentation in developing tropical countries has implications for human societies, because human welfare is tightly linked to local natural resources. Analyses show that deforestation might lead to limited short-term economic gain, but in the long term loss of forest fragments threatens the sustainability of basic ecosystem services (Rodrigues et al. 2009).

Mechanisms Driving Fragmentation Effects

Researchers have traditionally conceptualized fragmented landscapes as a set of patches of remnant habitat embedded in a matrix of human-modified habitat. Fragmentation decreases patch size, increases patch isolation, alters patch shape, and increases the influence of the matrix on biotic and abiotic processes at habitat edges, all of which can have strong effects on ecosystem dynamics in fragmented landscapes.

Decreases in patch size lead to smaller populations that are more vulnerable to extinction from chance events, such as fluctuations in environmental conditions. Any rare or irregularly distributed resources may also be lost as patch size declines, with the possible subsequent extinction of species dependent on those resources. For example, decreases in patch size may reduce the number of large, cavity-bearing trees to a level that can no longer support populations of cavity-nesting birds or mammals. Increased isolation reduces dispersal between patches, preventing individuals in one patch from rescuing a declining population in another patch or recolonizing an empty patch. This may be exacerbated by decreasing patch size, because smaller patches are also smaller targets for colonization. Together, decreased patch size and increased isolation alter the balance between extinction and colonization rates, which can result in a lower proportion of patches occupied or eventual extinction from the landscape.

Subdivision of habitat into isolated patches creates edge zones where patch and matrix meet. These areas are subject to edge effects, in which environmental conditions and species occurring at the patch edge differ from those in the patch interior due to the influence of matrix conditions. For example, the researchers Andrew Young and Neil Mitchell (1994) examined how microclimate and vegetation were affected by proximity to edges in forest patches in New Zealand. They found that temperature, light levels, and vapor pressure deficit (a measure of water content in the air) within 50 meters of a forest edge differed from conditions in the forest interior. This edge zone also had a higher density of mature trees, supported more species, and had different species dominating the community. Edge effects are often studied as a one-sided phenomenon, extending from the patch edge to its interior, but in reality edge effects also extend into the matrix, where proximity to the habitat patch influences the composition of matrix-dwelling communities as well. Edge effects within patches become increasingly pervasive in more fragmented landscapes because the proportion of habitat exposed to edges increases as patch size declines.

Habitat patches are often irregular or convoluted in shape. Convoluted shapes increase the length of a patch’s perimeter, resulting in a higher proportion of habitat exposed to edge effects and more habitat affected by multiple edges, which increases the frequency with which animals encounter edges. This potentially increases movement into and out of patches, and leads to greater fluctuations in population size within patches. Mathematical models have suggested such fluctuations in abundance will increase the probability of population extinction. Finally, patches with convoluted shapes may have discontinuous areas of “interior” habitat separated from one another by edge-affected habitat, creating isolated subpopulations of those species for which edge habitat is unsuitable.

In some cases, studies of habitat fragmentation have produced inconsistent findings across different biomes and different taxa; one explanation is that most studies only focus on patch-level processes (measuring attributes of individual patches such as area, isolation, shape complexity, or amount of edge-affected habitat) without considering that the patch-level effects of fragmentation depend sensitively on the landscape context in which patches occur. Factors such as the total amount of habitat remaining in the landscape, variation in habitat quality of the human-modified matrix, time since fragmentation, and composition of the regional species pool all influence patch-level processes. For example, the degree of contrast between patch and matrix will alter the intensity of edge effects, and the extent to which patch isolation translates into reduced colonization rates.

Indirect Effects of Habitat Fragmentation

The impacts of fragmentation can also operate indirectly by amplifying or mitigating the impacts of other processes, which is termed an interaction between the processes. Research into this aspect of fragmentation is recent and limited, but such interaction effects may be
common. Threats from climate change, invasive species, pollution, erosion, flooding, overharvesting, and other factors are likely to interact with habitat fragmentation. This has implications for managing the impacts of fragmentation, not only on ecosystems but also on human societies, because many of these threats directly impact human welfare.

In one example of this type of interaction, the Brazilian biologist Carlos Peres found that the impacts of subsistence hunting on forest-dwelling birds and mammals in the Brazilian Amazon were amplified in fragmented habitats, because fragmentation both increased the hunters’ access to forest remnants (i.e., absolute rates of hunting increased) and at the same time reduced the ability of animals in the surrounding forest to recolonize habitat fragments and replenish populations that were subject to hunting (Peres 2001). The net effect of this interaction on the sustainability of harvested populations was larger than would be predicted based on the combined impacts of the two threats acting independently. Many species that persist in continuous forest are expected to disappear from fragmented regions, with implications for both the biodiversity of the region and the human communities that rely on hunting for food.

Moreover, a series of studies in the northeastern United States has shown that fragmentation may indirectly affect human health by increasing Lyme disease (Borrelia burgdorferi) transmission. Small woodland patches in heavily fragmented landscapes have substantially higher densities of infected tick vectors (Ixodes scapularis) than larger patches, due to shifts in the diversity and composition of mammal communities. Species-poor mammal communities in small patches tend to be dominated by the white-footed mouse (Peromyscus leucopus), which is a good reservoir host for the Lyme disease spirochete. Fragmentation may also increase transmission by creating more woodland edges in the landscape, which provide suitable habitat for tick reservoirs (including P. leucopus) and increase human access to woodland where transmission can occur. In some cases, these factors have been shown to cause increased human risk of Lyme disease infection in more fragmented habitats (Jackson, Hilborn, and Thomas 2006), although this has not been the case in all regions (Killilea et al. 2008). These findings suggest that careful land-use planning, reduction in habitat fragmentation, and restoration of mammal biodiversity could reduce Lyme disease transmission to humans.

The potential for indirect interactions is of particular relevance for predicting and managing the impacts of fragmentation. These interactions can lead to synergies (where the magnitude of the overall effect is greater than the independent effects of either process), positive feedbacks, and threshold effects, which mean that under certain circumstances fragmentation has much larger and less predictable impacts than otherwise expected. For example, the US scientists William Laurance and G. Bruce Williamson (2001) found that forest fragmentation in the Brazilian Amazon creates forest edges that are drier and more predisposed to fire than the forest interior. Subsequent fires cause further forest loss and fragmentation, reducing regional rainfall by decreasing transpiration (the release of water vapor by plants) and increasing drought and the incidence of fire, which in turn drives further forest loss and fragmentation. These synergies and positive feedback effects between fire, regional climate, and fragmentation processes suggest there may be a threshold level of forest loss and fragmentation beyond which rain-forest cover can no longer be sustained in some regions of the Amazon. In this context, habitat fragmentation can result in considerable, large-scale effects that could not be predicted without explicit consideration of interactions between fragmentation, fire, and regional climate.

Management Challenges

A detailed ecological understanding of the direct patch-level effects of habitat fragmentation has been the cornerstone of nature reserve design and planning since the 1970s. Future conservation management strategies, however, face significant challenges in effectively integrating emerging perspectives on how landscape-level factors (such as quality of the matrix and the total amount of habitat remaining in the landscape) alter biotic responses to fragmentation at the patch-level, and on the importance of indirect interactions between fragmentation and other threats.

One of the few examples where there has been active development of management intervention strategies to mitigate indirect effects is in the Australian wheat belt, a vast area of open Eucalyptus woodland where clearance of trees from the landscape indirectly threatens agricultural sustainability by increasing soil salinity. This area has been extensively cleared (up to 93 percent in some regions) for grazing and crop production. Conversion from deep-rooted tree cover to short-rooted grasses and crops has caused groundwater levels to rise dramatically in the soil, leading to a steady increase in surface deposition of dissolved salts that occur naturally at high concentrations in groundwater in these regions. This process of “dryland salinization” gradually reduces the health of any remaining trees, leading to further loss of tree cover and acceleration of the salinization process. These positive feedbacks between tree cover, groundwater levels, and salinization suggest that there may be a threshold level of forest loss and fragmentation
beyond which natural vegetation dynamics are permanently altered.

In an important series of studies by the Australian ecologist Sue McIntyre and colleagues, management guidelines have been developed that address the direct and indirect effects of habitat fragmentation in the wheat belt. Estimates of the total tree cover actually required to mitigate or prevent dryland salinization have been derived from landscape-scale revegetation programs, which suggest that at least 30 percent of drainage catchments must be tree covered to ameliorate soil degradation. Furthermore, guidelines recognize the role of landscape context in mediating fragmentation effects—the recommended minimum habitat patch size is increased from 5 hectares to 10 hectares in areas dominated by intensive land uses, and the maximum cover of intensive production land uses is set at 30 percent of total landscape cover. These guidelines merge concepts of the maintenance of biodiversity with concepts of the sustainability of future crop production in the face of increasing dryland salinization, leading to better integration of conservation and production goals in the landscape.

Ultimately, understanding when habitat loss and fragmentation effects are likely to occur, and when they are likely to exacerbate other threats to humans and biotic communities, will be vital to managing their combined effects. Only through mitigation of the direct and indirect effects of fragmentation can we minimize biodiversity loss while maximizing sustainable food production and human welfare.

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See also Agricultural Intensification; Biodiversity; Biodiversity Hot Spots; Biological Corridors; Boundary Ecotones; Disturbance; Edge Effects; Fencing; Fire Management; Food Webs; Forest Management; Light Pollution and Biological Systems; Mutualism; Population Dynamics; Reforestation; Refugia; Regime Shifts; Rewilding; Road Ecology; Succession; Tree Planting

FURTHER READING


Bryant, Dirk; Nielsen, Daniel; & Tangle, Laura. (1997). *The last frontier forests: Ecosystems and economies on the edge; What is the status of the world’s remaining large, natural forest ecosystems?* Washington, DC: World Resources Institute, Forest Frontiers Initiative.


Home ecology addresses topics that extend beyond human health to evaluate the impact our actions have on the condition and health of the planet. Manufacturing and transportation systems exist to supply things we use at home and at work, and major consumer purchases often “reside” where we do. Domestic choices, even small ones, must be magnified in terms of the total global population, and the lifestyle choices made by people in affluent countries influence the aspirations of people in developing countries. For instance, preferences common in the United States—such as super-sized restaurant portions, the “bigger is better” notion when it comes to houses and cars, and the fascination with leisure activities that depend on expensive or ecosystem-unfriendly equipment—have come to define an “ideal” lifestyle far removed, even in our globalized world, from what many remote populations could imagine. For countries like China, with its rise in economic power, its large population, and its constant exposure to the West, such “ideal” lifestyles will be impossible to sustain. As one of China’s leading environmentalists, Liang Congjie (1932–2010), declared, “if Chinese wanted to live like Americans, we would need the resources of four worlds to do so” (Kynge 2004).

History and Debates

Many people consider the first celebration of Earth Day, in the United States on 22 April 1970, as the birth of the modern environmental movement. In the late 1960s, most of the US population seemed oblivious to environmental concerns—automobiles were powered by V8 engines guzzling leaded gas, and industry’s smokestacks spewed polluted sludge with little fear of legal ramification. And yet public
Consciousness about the environment was slowly rising in the wake of Rachel Carson’s 1962 bestseller *Silent Spring*. Earth Day founder Gaylord Nelson, then a US senator from Wisconsin, believed that channeling the energies of student antiwar protests into a “national teach-in on the environment” could put environmental protection on the national political agenda (Earth Day Network n.d.).

It was a gamble, Gaylord later admitted, but he and Earth Day co-chair Pete McCloskey, a conservative Republican congressman at the time, recruited Denis Hayes to organize a team and promote Earth Day across the country. Some 20 million individuals in the United States came together to rally in streets, parks, college campuses, and public forums against actions ranging from dumping raw sewage to the use of pesticides. That single day aligned political, public, and individual efforts that would result in the formation of the US Environmental Protection Agency and the passing of the Clean Air, Clean Water, and Endangered Species Acts. Earth Day went global in 1990, bringing out 200 million people in 141 countries; the event sparked worldwide recycling programs and the formation of the 1992 United Nations Earth Summit in Rio de Janeiro.

Earth Day 1990 brought a surge in individual awareness of environmental degradation. People who once thought of environmental writers as “cranks” soon began to seek out an ever-more accessible range of green books and articles. (E. F. Schumacher, the author of *Small Is Beautiful*, was happy to be called one, identifying a “crank” as a small, efficient tool that makes revolutions.) Now, nearly twenty-five years later, the range of blogs and online information sharing about home ecology issues is breathtaking—there is even a blog about how to make cloth diapers. Not all environmentalists, however, shared Schumacher’s belief in the power of individual people. And some were positively dismissive of it.

**Individual Action**

The idea that the simple things each of us do can help save the planet came to prominence as Earth Day 1990 approached. Still, a number of environmentalists today remain dubious, arguing that one person’s “progress” on one avenue toward sustainable living fosters complacency about other avenues, and that ecofriendliness needs to be assessed in a truly global context.

A 2011 study conducted by the Center for International Climate Change and Environmental Research—Oslo (CICERO) reveals that green families and other members of environmental groups are no more ecofriendly than their nongreen counterparts. For instance, people who recycle, or bicycle to work, or buy green products to reduce their carbon footprint, have something else in common: they travel more during holidays and vacations, which nullifies their attempts at home to reduce their impact on the planet. Kristin Linnerud, a senior research fellow for CICERO, aptly theorizes, “when we sacrifice something, we think we deserve a reward” (CICERO 2011).

**Green Consumerism: Personal vs. Political**

Green consumerism, the premise that individual consumers can change industry simply by asking for and buying environment-friendly products, pushes the debate about individual action into the political realm. Some of the same analysts who doubt the efficacy of individual effort use green consumerism as the rationale to shift blame for environmental problems (and the lack of solutions) toward the individual and away from corporate polluters and government policy. John Elkington, the author of the bestselling *The Green Consumer*, called attention to the irony when he hailed green consumerism as a method of personal empowerment. “People’s ordinary spending is the most powerful agent of change they possess,” he declared.

Two problems emerge from those trains of thought. On the one hand, by treating the personal and the political as adversaries, it’s easy to ignore the type of synergistic energy that has infused all Earth Days (and any resulting reform in public policy and governmental regulations, nationwide or internationally) since 1970. Personal, ecofriendly changes made at home can serve as a catalyst to influence public attitude at the grassroots level. Telling friends and neighbors that you’ve switched to biodegradable soap might prompt them to do the
same. Gardeners, who often participate in a subcommunity of knowledge- and technique-sharing in their urban neighborhoods or rural towns, manage their local ecosystem constantly, producing pesticide-free foods or habitats for fauna such as hummingbirds and butterflies, species that will pollinate plants on a larger scale. Through everyday actions like using fabric grocery bags, switching to eco-bulbs, and installing thermostats with time-triggered temperature settings, people exert control over their environment. In this way individual action can cause a groundswell of public support and demand for the development of green technologies, green industrial practices, and government policies and regulations to set safety standards for hazardous products and processes.

On the other hand, green consumerism does not address the quantity of goods we consume, or assess consumerism as a basis for shaping a culture or civilization. Elkington's eventual bestseller hit bookstands in Britain in the autumn of 1988, was rewritten with co-authors for a US market and published there in 1990, and has since been revised in “sequels” that target special audiences such as kids and supermarket shoppers. The original edition, however, seemed to catch UK environmental groups by surprise, many of which (although not the UK Green Party) helped organize a “Green Consumer Week” to promote the book. A deeper irony had emerged with the idea of consumerism as personal empowerment.

Think of it: we all queue up at the cash register to vote with our environmentally sound purchases. But implying that people are capable of no more than putting a different brand of dishwashing detergent in their shopping cart is damaging and demeaning. While boycotting a coffee shop that refuses to abandon Styrofoam cups is indeed a positive step, the idea of green consumerism seems positioned to privilege the effects of a consumer mentality within a culture.

Fortress Thinking

A worrying mindset emerges as some individuals, who obsess over the hazards of modern life, concentrate on creating an insulated indoor environment while cheerfully allowing the world outside to become a wasteland. Such thinking harks back to the survivalist mentality at the height of the Cold War: “I can’t do anything about the bomb, but I can build a shelter in my basement.” The triumph of consumerism may arguably connect to the nuclear threat, and in general to uncertainty and pessimism about the future. The following quote from Isaiah 22:13 and 1 Corinthians 15:23, evokes the fleeting presence of humans on Earth and in the universe: “Let us eat and drink, for tomorrow we die.”

Practicing Home Ecology

What we consume, what we conserve, and what we throw away—that is to say, the “give and take” aspect of our relationship to the Earth—most often occurs in our personal abodes. The following sections explore ways to make our personal spaces (and the choices we make about how to manage them), more ecofriendly. We need food, adequate energy, water, and clean air to sustain our lives; we must be mindful of how we can reduce and dispose of waste, a byproduct of our not always essential “needs.”

Food

Some people consider “eating organic” a matter of personal health rather than a necessary lifestyle change that affects the health of the ecosystem as a whole. But with public awareness of environmental sustainability, more and more chain grocery stores in the United States responded by expanding their organic offerings—from fruits and vegetables to grass-fed beef and hormone-free poultry to canned beans, soups, and artisanal cheeses. Fear of widespread food contamination (notoriously in infant formula) and agricultural pollution in China has drawn significant attention to organic food products available there. But many other issues complicate the debate. For example, the luxury pricing of “organic” and “local” foods exacerbates its inherently “expensive” inaccessibility to
many, and then there’s the question of whether organic agriculture could possibly feed the world’s growing population.

Most of the produce found in large grocery stores comes from commercial farming operations that practice what is known as industrial agricultural. The cheapness of such mass-produced food often comes with a deferred price for the environment: farm production depends on agricultural subsidies, the depletion of aquifers and soil erosion, and massive input of chemicals, pesticides, and fertilizers. Long-distance shipping has a dual impact on global warming: the transport itself and the need for refrigeration. Consumers can rarely find apples, peppers, or cucumbers that haven’t been coated in a waxy substance to keep them “fresh.” Varieties are limited and arguably less nutritious after being stockpiled for months.

For individuals who can’t or don’t want to garden, local growers, including neighbors tending their plots, provide plentiful sources of local produce. Following the seasons for each type of produce, instead of buying products forced to grow out of season, or ones that have to be shipped from farther away, can reduce environmental impacts. There are also methods of preserving food that don’t involve chemical processes. Home canning, pickling, and freezing are great ways to use abundant harvests that couldn’t all be eaten in one season. Making kimchi, sauerkraut, jams, and jellies are just a few of the ways fruits and vegetables can be processed naturally for long-term storage.

Energy

When it comes to energy consumption, residential users consume 14 percent of power worldwide as of 2011; commercial use accounts for 6 percent; transportation for 27 percent, and industry for 52 percent (EIA 2011). The average household in the United States used 927 kilowatt hours per month in 2010 (EIA 2010), the largest percentage worldwide, not surprisingly, and nearly twice that of Japan, another highly industrialized country. It is difficult to calculate a meaningful average for a country like India, with its wide range of class levels. While approximately 80 percent of households in China use 140 kilowatt hours per month or less (China Daily 2010), the accumulated effect of its huge population bumps China up to the second-largest energy consumer in the world. Due to increasing populations in China and elsewhere, the demand for energy is expected to rise. With issues of global warming attributed to increased levels of CO₂ in the atmosphere, we must balance the ecological impacts of obtaining coal and other fossil fuels (such as transporting oil by pipeline or using hydraulic fracturing to bring shale or sand oil to the surface) with the finite supply of these resources. Ecosystem managers (including industries, governments, organizations, and regulatory committees) need to find renewable sources of clean energy to meet this projected demand.

Receiving energy from clean, renewable sources is only part of the battle. Energy and environmental government agencies around the world have been implementing programs and laws to put more energy-efficient appliances on the market, to provide more sources of green energy, and to offer tips to consumers about energy conservation. But government agencies can only go so far. To make a difference individuals must take steps such as replacing worn out appliances (especially refrigerators) with energy-efficient models and cutting energy costs by buying more fuel-efficient cars. Even the smallest changes in personal habits—unplugging chargers when a cell phone battery is full, for instance—can reduce the amount of energy consumed. Insulating walls and roofs will make a difference, as will replacing windows with upgraded units or simply covering them with a sheet of cellophane. In many homes in Japan and India, on-demand water heaters are used to conserve the energy wasted by keeping water hot all day. Outside of the house, alternatives to driving such as walking, biking, and public transportation can have a diminishing effect on the amount of fossil fuels consumed and CO₂ produced.

The price of everything we buy includes a percentage for the energy needed to produce and transport it. Taking this cost into count—including the impact of energy generation—is a change we should look forward to.

Water

When industry dumps chemicals and toxic waste into water sources like rivers, lakes, and oceans, and fertilizers and pesticides from industrial agriculture flow into rivers as runoff or sink into the groundwater, contamination results. When assessing the scope and complexity of these problems, the impact that a single household might make seems insignificant. But changes in personal habits regarding water use, and the advance of water-saving technology, can go a long way toward water conservation, as much of a concern in ecosystem management as pollution and contamination.

Most homes bring in water through a single-pipe system, and that single source of (usually) treated water suffices for drinking, bathing, washing, and other uses. This means that the water coming out of the tap may be questionable for drinking, and that water used to flush toilets is cleaner than it needs to be—therefore, the effort of treating the water is largely wasted. While the idea of a
dual-water system (with one set of pipes devoted to industrial-type usage and another set dedicated to carry water for drinking and hygienic use from underground springs) seems like an ideal fix, it may not be the most practical or cost-effective solution for a household not in need of upgrading pipes. (As of 2011 this dual-pipe system still has not been put widely into effect.) Some people buy bottled water, or filter their tap water, but even these practices aren’t cure-all solutions. Plastic-bottled water is processed, packaged, and transported, and the plastic can leak carcinogenic polymers into the water. (The 2009 documentary Tapped, produced by Atlas Films, is an eye-opening account of the bottled-water industry.) Home filtering systems are also becoming more sophisticated, but regular maintenance is crucial to their effectiveness and safety, since filters need replacing once a month to trap impurities and avoid bacterial growth.

Putting water to multiple uses can go a long way toward conservation. According to the Environmental Protection Agency (EPA), each person in the United States uses an average of 100 gallons of water per day. That amount can be reduced by 30 percent through the use of water-efficient fixtures and appliances, such as low-flow showerheads and toilets, and small changes in personal habits (EPA 2011). For example, cooking- and dishwater (with biodegradable soap) can be used to water gardens. Newer washing machines have a suds-saver mode that stores the wash water from a load in a holding tank with the option to use it for the next load, and so on, until it is too dirty. Taking showers instead of baths can reduce water use on an individual level. In countries like Japan, bath water is used multiple times, usually by each member of the family, after washing with a small bucket of water outside the bath to keep the bathing water clean for other users. If a low-flow toilet is out of the household budget range, a small container filled with rocks or marbles can displace water in the tank, reducing the amount of water used per flush.

By using water more efficiently, and switching to biodegradable soaps and detergents, individual households can greatly reduce their impacts on local water sources. When fewer pollutants flow down the drain, less nitrogen and phosphorous enters the local water supply. With less nitrogen and phosphorous, the effects of eutrophication (rampant growth of algae that feed on nitrogen and phosphorous, which thus reduces oxygen in bodies of water) can be slowed and potentially reversed.

Air

People spend more time than ever before indoors, but the air inside our houses and offices is often two to five times more polluted than the air outside. Building materials, consumer products, and personal activities all create invisible pollution known as volatile organic compounds (VOCs). VOCs are present in particleboard, soft plastic, plastic foam, caulking, paint and varnish, office machinery, cleaning products, personal cleansers, and even some foods. Emissions from most of these products decline rapidly after a few days or weeks, but some, such as those from new carpet and particleboard, last longer. Over-exposure can result in dizziness, headaches and nausea, respiratory problems such as asthma, and weakened immune systems.

Aerosol sprays, ostensibly used to make homes cleaner, contribute to household air pollution, and many may be doing more harm than good. Air fresheners advertised to eliminate odors just cover up one smell with another more potent (and supposedly pleasant) smell, or deaden the olfactory nerves. Aerosol particles disperse widely throughout the air and are easily inhaled, allowing the body to absorb dangerous chemicals. This can be especially dangerous in draft-proof houses that are sealed tight and prevent the movement of air to carry away the aerosol mist. While it’s a good idea to seal windows to save on heating bills in the winter, that same seal can make the ecosystem of the home a closed system, locking the air pollutants inside.

Offices can be even worse. The phenomenon known as “sick office syndrome” can be attributed to many factors in the workplace. An office building can be a closed system, with no natural ventilation. Windows that don’t open can’t provide fresh air. Air conditioners allow the buildup of bacteria and viruses, as does centralized heating with toxic gases, which then all get released into the air and dispersed across the office. Even the chemicals used to clean the office are suspended in the air where they can be inhaled and absorbed. Simple changes like improved airflow, switching to less-harsh chemicals when possible, and keeping plants in the office, can drastically improve air quality, and reduce occurrences of headache, fatigue, and depression.

Waste

Just as we carefully consider what comes into the ecosystem that is our home, we also must consider what goes out, and where it then goes. In most urban and suburban neighborhoods, garbage collectors drive their trucks up to the curb, operate huge metal “jaws” that clamp onto cans or receptacles, turn the cans upside down, and dump the contents into the holding bin. The truck driver takes the garbage to the landfill, and that’s the end of it as far as many of us are concerned.

But landfill sites throughout the developed world are filling to capacity. Many have already closed, and
recent zoning laws or regulations can make it harder to open new ones. Trash has to be transported farther and farther for burial, adding to costs and further polluting the air. Landfills pose other problems. Bacteria decompose food, yard, and other household waste in the anaerobic atmosphere of a landfill to produce dangerous landfill methane that is emitted into the atmosphere (although methane-capturing techniques exist to turn the gas into fuel). And leachates (the toxic “soup” created when household or industrial chemicals thrown into landfills mix with the chemicals produced by decomposed garbage), seep into groundwater supplies.

Incineration, a potentially useful source of heat and energy, has special hazards. Some materials when burned emit dangerous gases. In a small country like Japan, where there is no available landfill space and no feasible alternatives, trash must be carefully separated into bins, often by individuals who bring the trash to sorting centers to isolate products that can be safely incinerated from those that must be carefully disposed of or recycled.

When it comes to landfills and incinerating stations, the NIMBY principle (not in my backyard) applies. But every place is someone’s backyard. Past incidents of unethical dumping have included American trash being buried in Cornish tin mines, West German industrial waste deposited in northern Cyprus—and developed countries’ garbage being shipped to poor African countries. Most people would never sanction depositing garbage on a neighbor’s lawn or wood-line property boundary, so it should follow that our country’s waste is our problem, while the waste of other nations is theirs. Responsible waste management is essential domestic housekeeping for everyone.

Product packaging, from food wrappers and cellophane to the hard plastic that cocoons even the simplest of gadgets, contributes immensely to waste—as stuff we throw away, of course, and as a wasteful use of the resources it takes to produce the packaging. Why do oranges and bananas, for example, with their own natural packaging, need to be bagged in plastic and then bagged again with the rest of the groceries? Short of petitioning stores and companies to change packaging practices, there are several ways we can eliminate excess and unnecessary packaging at the household level. Choosing products that come in recyclable, reusable, or biodegradable wrappers can make a huge impact on what gets taken to the curb every week. Another way to cut waste at the store is to buy in bulk, and buy refillable containers when possible. Bringing cloth totes to stores will reduce the number of plastic bags taken home. In 2011, some US grocery chains took responsibility for recycling plastic bags brought to the store by their customers (King Soopers 2011). In China shoppers use up to 3 billion plastic bags daily and dispose of more than 3 million tons of them annually, mostly in unofficial dumping sites, landfills, and open fields along expressways, where the ultra-thin plastic bags are called “white pollution.” On 1 June 2011, the central government implemented a nationwide ban that prohibits shops, supermarkets, and sales outlets from giving customers free plastic bags; it also bans the production, sale, and use of plastic bags under 0.025 millimeters thick (Liu 2011).

So what can we do with all those biodegradable packages and banana peels? A simple Amazon search will yield over a thousand results for books all about composting, including how to build composting containers and composting for apartment dwellers. Composting food leftovers creates nutrient-rich soil for gardens or flowerbeds, instead of a plastic-encased pile of rot in a landfill.

With the proliferation of computers, there has been a surge in e-waste (electronic waste). According to Global Futures Foundation (Earth911 2011) “electronic waste accounts for 70 percent of the overall toxic waste currently found in landfills.” Old televisions contain cathode rays that contain lead. Computers and other electronics
with lithium or nickel cadmium batteries have to be disposed of carefully. Many different electronic devices also contain aluminum and mercury (Earth911 2011). If simply thrown in a landfill, these hazardous substances contribute to the formation of toxic leachates. Some electronics stores will dispose of and recycle electronics, and more and more communities have toxic and e-waste drop off points.

Community Ecology

Ecology, as a science, can be most basically defined as the relationship of organisms to their environments; in that sense, ecology is the study of communities and their interactions with the world. Community as an aspect of home ecology thus merits further study. As long as 100,000 years ago, for instance, Homo sapiens settlements fostered the sharing of the technologies like fire and tool making; irrigation methods in agriculture developed as hunter-gatherers abandoned nomadic lifestyles. Fast-forwarding to the twentieth-first century, much sharing of information is done through online forums and networks. As social scientists have found in studies of other forms of social behavior related to health, the lifestyle choices we make are greatly influenced by choices the people around us make. Domestic and personal choices that are better for the environment, and for human health, are more likely to “stick” when our neighbors, friends, and loved ones are willing to make similar choices (and when there is social pressure not to do harmful things).

Our homes, however, put more pressure on the planet than dwellings of early communities because of the isolation—or call it privacy—that has become the norm in urban societies. In the latter half of the twentieth century, for the first time in human history, at least some people—those in the urban, developed world—could truly get along without cordial relations with their neighbors. The existence of supermarkets (many of which now deliver), big box stores, banks, and the whole panoply of modern institutions make it possible for one to earn money and secure essential services from perfect strangers. And yet we use more space, and have more material goods, because we are less often in a position to share “things” with people we live with.

Managing the human ecosystems of home and workplace more sustainably includes accounting for goods and services and the structures of our buildings, as well as taking stock of the way we live and how we live together. Home ecology should be factored into town and regional planning, with a goal of creating healthy habitats for humans without harming the wider environment, not by building fortresses against an outdoors that is perceived as being dangerously polluted.

While some may dismiss the idea of an individual’s capacity to have a positive impact on the larger problems associated with environmentalism and sustainability, it is important to realize the echoing changes that well-intentioned individual action can make. Parents pass on lifestyle habits to their children, who will in turn adapt those habits to future goals and needs, and instill them in the next generation. (The system can work in reverse, and thus come full circle.) An individual may influence a community, effecting political change from the bottom up, either passively or actively. But the most important thing to keep in mind is that every aspect of life has an environmental footprint, from Internet use to vacation travel, and although such uses may be unavoidable, falling into complacency about them is the biggest threat against change.

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See also Agroecology; Human Ecology; Landscape Architecture; Light Pollution and Biological Systems; Permaculture; Urban Agriculture; Urban Forestry; Urban Vegetation; Waste Management; Water Resource Management, Integrated (IWRM)

Further Reading


Human ecology is an interdisciplinary approach to understanding human-environmental systems. The field seeks to combine understanding of the biophysical realities of human existence (such as dependence on natural resources) with the social and psychological dimensions of human health and well-being.

Human ecology focuses on understanding humans and their environments as parts of a whole. Although the field predates current debates about sustainability, it shares concern for the limited capacity of the Earth to meet the demands that humans are placing on it. Human ecology is also concerned with ethical questions about how fairly environmental resources are shared among people and other living things, and identifying the rights and wrongs of existing situations and proposed alternatives. Where the human-environment system is changing in ways that cause problems for people, human ecology focuses on what is ultimately driving that change, and the consequences. By seeking the ultimate, rather than immediate, causes of change, human ecology typically locates the sources of many problems in aspects of the dominant culture, its attendant values, and resulting human behavior. The challenge then is to identify interventions that will result in improved environmental outcomes and that are fair and acceptable to the people who are affected. Human ecologists are then typically committed to act as change agents by seeking interventions that improve the health and well-being of people in a humane and sustainable manner.

Human Ecology and Sustainability

Human ecology is able to make a valuable contribution to understanding and improving situations that are labeled as sustainability problems because it provides approaches that rely on multifaceted analyses. Sustainability problems typically involve a degree of scientific uncertainty, in that it is not always possible to gauge accurately the status of the key environmental variables that have to be managed. For example, it is hard to assess the carbon stored in agricultural landscapes under different management regimes, yet this could be a key element in promoting sustainable farming practices. In addition, sustainability problems often have no clear boundaries and cut across institutional and jurisdictional boundaries, including different departmental responsibilities or state and national borders, such as when atmospheric pollution from a smokestack in one state causes acidification in the forests of its neighbor. In many cases the ultimate cause of a problem is at a great distance in time or space from its effect. For example, a consumer’s choice of coffee bought from a local supermarket can affect, for better or worse, the environmental health of a coffee plantation landscape in a distant country. Many problems are the unforeseen and unintended consequences of human activity that made perfect sense at the time. Irrigating a property to increase food yields makes sense, as does building freeways to relieve traffic congestion. Yet an unintended consequence of irrigation might be to mobilize naturally occurring salts in the soil and eventually render it useless for farming; similarly, freeways can make private vehicle travel choices more attractive and result in even greater congestion as more people adopt this method of transport. Problems can even defy definition. Building a hydropower station on a river might be the solution to a problem of renewable energy generation but the cause of a problem for river ecology and fish populations. This complexity is compounded as individuals and groups promote often-conflicting options for intervention to improve a
situation, and various judgments about the desirability of proposed solutions.

With sustainability problems, pathways to improvement will often involve changes to human attitudes and behaviors as much as they will involve new or changing technology. Certainly technology can improve the efficiency with which an environmental resource is extracted and managed and reduce the unit cost of the good produced, but improving efficient supply cannot always keep up with total increasing demand. Modern jet aircraft can be more efficient as measured by the energy cost of each passenger transported per mile, but with soaring passenger numbers the total energy cost of the sector will increase. Expecting changes in people’s behavior requires ethical considerations of justice and fairness in terms of how people are engaged in the development of proposed solutions and, when costs are involved, how the burden of those costs is distributed. This ethical concern may also extend to species other than humans. With its concern for the ethical dimension, human ecology has a normative dimension that other sciences often lack or downplay.

Human ecology’s concern for the sociocultural dimensions of sustainability problems is coupled with the recognition that ecosystems have a finite capacity to service the demands humans place on them. If human demands on environmental resources exceed the rate at which those resources are naturally replenished, then the resource will inevitably be exhausted. At best, for very large stocks, this point of exhaustion may be far off into the future, which might allow postponing the inevitable requirement for human behavioral change. A similar principle applies to pollution: problems arise when rates of accumulation exceed the natural capacity of the ecosystem to absorb pollutants. The unequivocal evidence is that human use of key resources is now rapidly approaching their limits, and many people already do not have access to sufficient resources to maintain minimal standards of health and well-being.

The complex nature of sustainability problems means that they are best tackled using integrative, holistic approaches that combine traditional disciplinary knowledge with other insights into the human condition, people’s beliefs and values, and their aspirations and motivations. Human ecology draws these insights from the social sciences, as well as the humanities; arts and design; and lay, community, and nonacademic knowledge bases. The field seeks to provide a conceptual framework for research and learning that combines knowledge about what is and what needs to be done, with understanding about what motivates and enables individuals and societies to act on the basis of that knowledge. Human ecologists are agents of change, seeking to help societies achieve humane and sustainable futures.

**Ecology and Human Ecology**

Human ecology has its origins in ecology more generally. The term **ecology** was originally coined by the German zoologist Ernst Haeckel in 1866 to mean, loosely, the “science of the habitat” (Lawrence 2001, 675). At this basic level, human ecology can be thought of as the study of the environmental conditions in which human beings developed, and the relationship of humans to the ecosystems that support them and which they affect. The same principles apply to the ecology of any species, but in the case of humans, the number of humans on the planet, their presence in almost all terrestrial ecosystems, and their impact on the planet is largely the product of the evolution of human capacity for culture. If success is measured by sheer numbers and ability to colonize nearly every environment on the planet, then culture was an evolutionary advantage.

The need to take culture and its effects seriously makes the study of the ecology of humans different from the ecology of other animals. Other species exhibit behavioral adaptation to their surroundings, but for humans sociocultural adaptation is the prime mechanism for responding to environmental change. Humans can learn and adapt their behavior based on information provided by other humans, in stories passed from generation to generation and enshrined in enduring social institutions. Humans have a highly developed ability to imagine consequences of future action, although they do not necessarily act to avoid those consequences. Artistic creativity enables the celebration of traditional ways of living and the imagination of alternative futures. Imagination and inventiveness also allow humans to develop tools and technologies that extend their capacity to access resources from the environment and to rapidly change the efficiency with which they can convert resources into a service. These characteristics of cultural adaptation, social and individual learning, institutional arrangements, art and creativity, imagination, and technology, while not necessarily unique to humans, are developed to highly complex levels in them.

**Development of Human Ecology**

As human ecology developed out of ecology and the natural sciences, it followed a number of different trajectories. Current academic programs using the term **human ecology** often exhibit different characteristic concerns depending on which of these pathways they followed.

Ellen Swallow Richards, born in Massachusetts in 1842, is one pioneer of the discipline. Richards was the first woman graduate of the Massachusetts Institute of Technology (MIT), graduating with a bachelor of science degree in chemistry in 1873. She also obtained a master’s
degree, but the award of a PhD was beyond what MIT at that time was prepared to confer on a woman. Her disciplinary expertise was in industrial chemistry, but she had broader interests in the social movements of the day, including women’s issues and progressive social change. Generally, in 1892 she used Haeckel’s term *oekology* to mean the science of the conditions of the health and well-being of everyday human life, elaborated as human ecology in 1907. But the biological scientists resisted extending the concept of ecology to include social dimensions, and the term *home economics* was adopted in its place. Human ecology programs that derive from this lineage are still common in the United States today, typified by programs in education and childhood studies, nursing, family and community well-being, and local applied policy issues.

Today the study of human ecology may be defined by the interests, methods, and intellectual domain of particular disciplines. One strand of human ecology as a field of study arose within the sociology department at the University of Chicago in the 1920s. In this school of thought, ecological terms from the biosciences are applied to social change processes, using concepts such as competition, succession, web of life, and mutual interdependence. Versions of human ecology also developed within other social science disciplines, such as geography, anthropology, and ethnology, but beyond contesting ownership of the name, these versions of human ecology have little in common.

Growing environmental awareness in the 1960s gave rise to a different approach to human ecology. This multidisciplinary human ecology recognized the limited contribution that single disciplines could, by themselves, make to understand the complexity of human-induced ecological problems. Typically arising in applied professions such as urban design and regional planning, these programs brought together a range of disciplines to apply their unique insights jointly to a given problem. Pragmatically focused, these approaches collected insights from the contributing disciplines, without defining a unifying framework. The frameworks, assumptions, and methodologies are not transformed through their interactions with other disciplinary experts. Nevertheless, this multidisciplinary approach continues to be practiced by those who place a premium on maintaining forums in which different disciplines can share their insights.

**Interdisciplinary Human Ecology**

Human ecologists have sought to define a genuinely interdisciplinary field of study, in which the understanding achieved is more than the sum of the contributing parts. The challenge is to retain the power of disciplinary thinking while avoiding the partiality of such work. Interdisciplinary human ecology tackles complex sustainability problems with approaches that have rigor but are not rigid. It may use a systems-based framework, which enables consideration of the influence and constraints of both social and biophysical drivers of change. The system in question, its boundaries, component parts, and its dynamics are defined by participating stakeholders. Such an approach enables the integration of the diverse values, desires, and needs of the human actors with the capacity of the environment.

As a college dedicated to human ecology, the College of the Atlantic in Bar Harbor, Maine, exemplifies the graduate attributes typical of a student of human ecology. It bases its curriculum on the following characteristics:

- **Be creative:** use the imaginative and inventive powers of the human mind to tackle sustainability problems with original and adaptable approaches. This commitment to creativity has to include a willingness to take risks and sometimes fail, so long as failure is acknowledged and learned from.
- **Think critically:** reflect critically on the partiality of information, including the unavoidable prejudicial elements that arise from human habits, biases, and assumptions. This ability to think critically includes reflection on one’s own limitations and preconceptions.
- **Engage with community:** involve individuals, communities, and institutions in the design and implementation of solutions to their problems. This includes a willingness to learn from their knowledge and traditions. It also requires participants to connect theory and practice.
- **Communicate:** communication is understood to be processes of learning and not merely the transmission of knowledge from one party to the other. Communication can include artistic and motivational elements.
- **Integrate elements:** think comprehensively about situations as wholes. The characteristic behavior of the whole emerges from, but cannot be reduced to, the interactions of the parts. Sustainability is a description of the behavior of a system of interest to an individual or group as it changes or remains constant over time.
- **Practice interdisciplinarity:** recognize the strength and depth of disciplinary thinking and the crucial contribution it can make to solving problems, while being aware of its partiality. Combining different contributions within suitable conceptual frameworks is crucial to creating the new knowledge needed to tackle sustainability problems.

As an evolving approach to defining and investigating its subject matter, human ecology may appear to have a complex and occasionally contradictory history. This same diversity, however, prepares human ecology to
contribute new approaches to complex and previous intractable problems.

**Human Ecology in Action**

Many university-based human ecology programs around the world are sponsoring research and publishing that continue to define the field and also lead to public sustainability initiatives. In addition, organizations such as the Society for Human Ecology, the Commonwealth Human Ecology Council, and the German Society for Human Ecology bring practitioners together to advance the field.

As early as 1972, following a meeting organized by the Commonwealth Human Ecology Council (CHEC) in Hong Kong, a group of researchers, primarily based in the Human Ecology program of the Australian National University (ANU), undertook a major study of the city of Hong Kong and its population. Published under the auspices of the United Nations Educational, Scientific and Cultural Organization (UNESCO) as *The Ecology of a City and Its People* (Boyden et al. 1981), this landmark study applied a whole-of-system approach to the material and energy flows of Hong Kong and the sociocultural drivers of those flows. The study was a seminal work in research into urban metabolism and is an exemplar of research incorporating both the quantifiable and qualitative dimensions of sustainability. In the twenty-first century, ANU continues to be a leader in the field of human ecology through its teaching and research program, as well as through open discussion at the Human Ecology Forum.

The aforementioned College of the Atlantic (COA) is entirely dedicated to the study of human ecology. COA places great emphasis on developing an applied approach through working collaborations with local and international communities. Over the years a number of initiatives originating from student projects have, through community engagement, been adopted in practice, such as the Maine legislature’s adoption of beverage recycling as a consequence of work conducted by COA students. The college offers ongoing demonstration projects of applied human ecology in areas of collaborative decision making, environmental design, conservation ecology, ecological education, green business, and watershed-based regional planning. COA has also received numerous awards for institutional resource management, campus buildings, and leadership in its commitment to carbon neutrality.

Human ecology research focuses on a variety of development issues. Researchers from the Department of Human Ecology at the University of Tokyo have been involved in interdisciplinary research into relationships between human and environmental health in rapidly developing Asian countries. Studying rural areas, research teams have tracked changes to a broad range of indicators of human well-being, such as income, labor arrangements, food and nutrition, exposure to various chemical compounds, and health. Policy interventions designed to influence national development can be initially successful, but then after a delay, negative consequences appear. An example is rising food production: this may result in improved nutrition, but a consequent rising chemical burden in the local environment eventually leads to potential risks for community health. The relationship between chemical levels and community health is nonlinear, however, with different individuals and communities affected to different degrees for a range of reasons detectable only at the local scale. Interdisciplinary human ecological research can suggest ways that policy interventions can be targeted to community involvement to achieve a balance of technological and social initiatives that can produce more consistently positive outcomes.

**Future Directions**

Research in human ecology has been hampered by the field’s nontraditional nature, including perceptions that it is not a proper science. It has been challenging to find funding bodies willing and able to support interdisciplinary research or journals willing to publish the results. Because it is an awkward fit within the traditional departmental structures of the university system, recognition of merit and pathways for career development have been limited. Although these barriers continue to exist, there is growing recognition of the need for approaches like human ecology to tackle sustainability challenges. The Ecological Society of America’s (ESA) Planetary Stewardship Initiative (Power and Chapin 2009) is devoted to fostering interdisciplinary collaboration and
defining the scientific needs to encourage sustainable social and ecological change. ESA’s Human Ecology Section was established to discuss and apply the ideas and methods of human ecology and allied disciplines. The Proceedings of the National Academy of Science (PNAS) also include a sustainability science section committed to publishing research dealing with the interactions between natural and social systems and their impacts on sustainability. Leading academic institutions, including Stanford University in California, are also establishing groups to investigate sustainability issues with a human-ecological perspective. With these and other arenas opening up, human ecology will be well placed to make its contribution to sustainability challenges.

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See also Agricultural Intensification; Biogeography; Ecosystem Services; Fisheries Management; Home Ecology; Indigenous Peoples and Traditional Knowledge; Permaculture; Shifting Baselines Syndrome; Urban Agriculture

**Further Reading**


Hunting is the practice of killing an animal—called game—for consumptive, recreational, or commercial purposes. Hunting as a management tool for ecosystem sustainability remains a highly divisive and controversial issue. In places where hunting is carefully regulated, hunting can be considered an effective method for keeping ecosystems in a healthy balance. In regions where hunting is difficult to regulate, hunting can be considered a destructive practice that reduces biodiversity and increases vulnerability to ecosystem collapse. This article briefly provides information on history and current participation in hunting, and then explores situations where hunting fosters or hinders ecosystem sustainability.

Background

Hunting commonly refers to the pursuit of a mobile species, usually a mammal or bird, by humans with the intent of killing for food, recreation, cultural reasons, or trade. In many societies, these species are referred to as game, and the act of killing them is referred to as harvest or take. Trophy hunting is a type of recreational hunting primarily focused on harvesting rare, exotic, or exceptionally large individuals of a species for the head, hide, horns, or antlers, with consumption of the meat being secondary or not important. This may be a legal or illegal practice depending on local laws. Legal hunting is practiced following a set of regulations that controls when, where, and how a person can pursue and kill game, and specifies how many of a species a hunter may kill during a specific period of time (e.g., per day or per year). Illegal hunting, or poaching, occurs when people are hunting in a manner that violates agreed-upon regulations of the governing institutions. Fishing, trapping, and gathering of wild plants may have similar intentions as hunting, but these practices typically use different weapons and tools, involve different species, and are considered separate activities.

Hunting can be considered either an effective method for keeping ecosystems in a healthy balance or a destructive practice that reduces biodiversity and increases vulnerability to ecosystem collapse. Because hunting involves complex and changing interactions among humans, wildlife, and the environment, consideration of circumstances is required. This article briefly provides information on history and current participation in hunting, and then explores situations where hunting fosters or hinders ecosystem sustainability.
in many developed parts of the world (e.g., North America and Europe), followed by restoration of many game populations to bountiful levels (Geist, Mahoney, and Organ 2001; Brainer and Kaltenborn 2010). Interestingly, the movement to save wildlife populations beginning in the early 1900s also was led by hunters such as US president Theodore Roosevelt (Wilson 2010).

Hunting continues to be a popular practice around the world today. There are an estimated 12.5 million active hunters in the United States (USFWS 2006) and 7 million European hunters (FACE 2011). In the forests of Latin America, Southeast Asia, and Africa, nearly 150 million people support their livelihoods through the hunting of game (Department of International Development 2002).

**Ecosystem Sustainability**

From an ecosystem perspective, modern hunting can be simplified (with a few exceptions) into two general categories: (1) hunting game for recreation and/or personal consumption under a strict regulatory system with sufficient enforcement, and (2) hunting game for sale of meat on the market (i.e., market hunting) or for personal consumption under weak regulatory systems with insufficient enforcement. The former occurs in wealthier, developed nations (e.g., United States, Canada, Europe), and is considered a sustainable practice from an ecosystem perspective. The latter occurs in poorer, developing nations (e.g., west Africa, Latin America), and hunting of wild meat (often referred to as “bushmeat” hunting in tropical areas) is considered to be on an unsustainable trajectory (Milner-Gulland, Bennett, and SCB 2003; Robinson and Bennett 2004).

**An Unsustainable Practice**

As human densities have increased in developing regions, so has pressure on local game resources, creating a conflicting relationship between healthy ecosystems and hunting. In tropical forests, increased commercialization and land conversion (e.g., logging) in remote areas have created roads and better access for people to exploit game (Robinson and Bennett 2000; Franzen 2006; Brinkman et al. 2009). Data suggests that numerous species of primates and ungulates are facing local extinctions in remote parts of Vietnam, Africa, and Latin America, largely because of hunting (Milner-Gulland, Bennett, and SCB 2003). Excessive hunting often reduces populations of large-bodied and slow reproducing species while rapidly reproducing species adaptable to human disturbance may increase (Peres 2000). From an ecosystem perspective, this may result in a shift in ecological communities (interactions of plants and animals living in the same place) toward less biodiversity.

Bushmeat hunting in tropical forests threatens biodiversity, but it also sustains the world’s poorest people. Ecologically sustainable solutions have been elusive for political, economic, and ethical reasons (Robinson and Bennett 2000; Ostrom et al. 1999). Researchers contend that issues related to poverty (e.g., income, education, human health) must be addressed before negative patterns associated with hunting can be mitigated. For instance, long-term conservation of biodiversity is of lesser concern when people are facing short-term circumstances such as famine.

**An Effective Tool**

In North America, Europe, and some sub-Saharan African countries, hunting has been considered one of the most effective management and conservation tools for maintaining game populations at healthy levels, sustaining healthy ecosystems, and alleviating human development and poverty problems. Reliance on hunting as a management tool has increased in North America as top predators have been reduced in numbers or removed from ecosystems, and the game species that they preyed on subsequently have increased. Today, game managers are more likely to contend with issues related to overabundance rather than conservation (McShea, Underwood, and Rappole 1997). For example, populations of white-tailed deer (*Odocoileus virginianus*) (Rooney 2001; Côté et al. 2004), geese (*Branta canadensis*, *Chen caerulescens*) (Ankney 1996), elk (*Cervus elaphus*) (Ripple and Beschta 2004), and many other popular game species are at population levels harmful to the ecosystem they inhabit (Garrott, White, and Vanderbilt-White 1993). Overabundant game populations contribute to reduced plant and animal diversity and threaten human life and livelihood. For example, wildlife damage (e.g., agricultural damage, vehicle collisions, disease transmission) is estimated at nearly $22 billion in the United States, with overabundant game species being the main problem (Conover 2002). Although alternative strategies (e.g., fertility control, fencing) for managing overabundant game exist, hunting remains the most socially, economically, and ecologically responsible method (Carpenter 2000; VerCauteren, Dolbeer, and Gese 2005). Unlike most alternative approaches, hunting helps to keep game wild, preventing habituation of game to human activities, and potentially reduces wildlife-human conflict.

Hunting also may be used to reduce or eliminate the presence of an undesirable species (e.g., invasive or exotic species). Some introduced domestic swine (*Sus scrofa*) and Eurasian boars have turned feral and are now considered one of the most threatening invasive species in North America. Numbering between 4 and 6 million, feral pigs are destroying native plants, damaging human
infrastructure, causing erosion, competing with native wildlife, and spreading disease to livestock and humans (Campbell and Long 2009; Tegt et al. 2011). Recreational hunting may be the best strategy for reducing the feral pig populations while potentially creating income (e.g., selling of meat, hunter expenditures) to offset management costs.

Economic benefits of hunting cannot be ignored (Williams 2010). Hunters have organized to form several nonprofit conservation groups (e.g., Ducks Unlimited, Pheasants Forever, Rocky Mountain Elk Foundation, and Boone and Crockett Club) that fund the protection of critical habitat for game species. Passed in the United States 1937, the Pittman-Robertson Act (P-R Act) levied a manufacturer’s tax on firearms and ammunition to provide revenue for wildlife agencies. Funding from the P-R Act in combination with hunting license fees has resulted in a significant source of annual funding (over $1 billion in the United States in 2009) for wildlife restoration, research, management, and education (Williams 2010). In Europe and North America, fees that private landowners accrue from allowing hunters to access game on their property serve as an incentive for landowners to preserve wildlife habitat. In sub-Saharan African countries, paid trophy hunts generate approximately $200 million a year (Lindsey 2008).

Although more difficult to quantify, advocates note that hunting helps to keep an increasingly urbanized human population connected to nature (Swan 1995). Interaction with nature improves physical and emotional health (Louv 2005), increases appreciation of the services provided by ecosystems, enhances awareness of potential threats (e.g., overpopulation, global warming), and cultivates stewardship of intact ecosystems. Despite these benefits, many people view hunting negatively.

A Contentious Issue

Even in areas where hunting is considered ecologically sustainable, more people have become critical of the practice and are debating its justification and continuance. Hunting proponents will argue the social, economic, and ecological benefits discussed above. People against all forms of hunting argue that it is unethical and unneeded in modern societies because of the availability of other foods (Baker 1985; Singer 1985). Ethical hunting varies by culture, but it commonly refers to pursuing and killing an animal using a protocol that is respectful to the animal, minimizes suffering, and obeys local laws and customs. Those opposed to using hunting as an ecological tool argue that reintroducing top predators (e.g., wolves, cougars [Puma concolor]) into ecosystems where they are absent would be the best way to find ecological balance. But this argument is less applicable when game populations are overabundant in areas heavily populated by humans, and reintroduction of large predators puts people or livestock in harms way. The pro- versus anti-hunting debate is a complex and sensitive issue that requires careful and respectful communication among all stakeholders.

Future of Hunting

Hunters per capita have declined in recent decades. In the United States, the percentage of hunters has declined from 10 percent in 1980 to 5 percent in 2006 (USFWS 2006). A summary of research findings suggests several explanations for declines, including urbanization of societies, increased dependence on cash economy and commercial foods, less access to areas with hunting, loss of knowledge about hunting, lack of time and money to hunt, a less physically active society, and competition with a growing number of nonhunting outdoor recreationists (Enck, Decker, and Brown 2000; Manfredo et al. 2009).

With the global economy promoting privatization and commercialization of recreational access, hunters with lower incomes may be excluded due to unaffordable fees. In North America, hunters generally view wildlife as a public resource and are strongly opposed to this trend. Some forms of paid hunting (i.e., trophy hunting), however, are thought to play a crucial role in rehabilitation of threatened wildlife areas and in generating an economic incentive (e.g., tourism) to promote conservation. For example, the growing trophy hunting industry in Africa requires a low and sustainable harvest to ensure future opportunities (Lindsay 2008). The future structure and function of hunting, undoubtedly, will vary in form and potential by location and customs.

When human population growth, climate change, habitat conversion, human values, and other influential factors are considered, the efficiency of hunting as a management tool for ecosystem sustainability becomes more uncertain. The unique services provided by hunting will need to be accounted for during times of rapid change to fully evaluate its future social and ecological utility. Individual hunters and the institutions that regulate hunting will have to be adaptive and flexible to unforeseen changes. Integrative strategies (approaches that consider social and ecological components across small and large geographic and time scales) to managing and monitoring hunting practices are thought to have the greatest chance for fostering sustainable game populations and ecosystems.

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See also Administrative Law; Biodiversity; Community Ecology; Complexity Theory; Fencing; Fish Hatcheries; Fisheries Management; Food Webs; Human Ecology; Indigenous Peoples and Traditional Knowledge; Invasive Species; Outbreak Species; Regime Shifts; Species Reintroduction; Wilderness Areas

FURTHER READING


Hydrology is the science that examines the properties and behavior of water on Earth, including its interaction with and reaction to the environment. Increasingly, as water resources respond to the pressures of population growth and climate change, hydrological investigations are used to inform decision makers on how much water is available, its quality, and the limits to its sustainable use.

Hydrology is the study of water: its physical and chemical properties; its circulation; its distribution across the Earth; and its interactions with the environment. It is a very broad field within the earth sciences, and because of this breadth some aspects of the study of water are deemed the province of other areas of scientific inquiry. For example, the study of marine waters is included within oceanography, while the study of permanent ice (glaciology) and atmospheric water (hydrometeorology and climatology) are considered to be discrete interdisciplinary domains of earth and environmental sciences.

The earliest evidence for the development of hydrology dates from ancient Mesopotamia, Greece, and Egypt. For example, from about the fifteenth century BCE, agricultural taxes were based on measurements of Nile flooding events, evidence for an understanding of the critical importance of water for survival and economic prosperity. In ancient Greece, Thales of Miletos (624–584 BCE) studied the regular flooding of the Nile River; Plato (428–348 BCE) developed basic concepts of the water cycle and recognized the impacts of human activities on water quality. By the time of the Roman Empire, engineering hydrology had developed to the point at which aqueduct systems, up to hundreds of kilometers in total length, distributed water from springs, lakes, and rivers to large urban populations.

With the exception of some refinement of hydrological concepts by great thinkers such as Leonardo da Vinci and Galileo, it was not until the eighteenth and nineteenth centuries that significant developments in hydrology occurred. During this latter period of time, the “founders of British hydrology,” Edmond Halley and John Dalton, developed an understanding of evaporation processes and the atmospheric phases of the hydrological cycle, while British engineers such as Robert Manning developed formulae for estimating the velocity of flow in streams and channels. Elsewhere in Europe, during the design and construction of water supply systems for Dijon, France, and Brussels, Belgium, Henri Darcy developed a formula for the flow of water through a porous medium (sand), now referred to as Darcy’s Law, and, with his colleague Henri Bazin, developed equations for quantifying flow in open channels.

This focus in the development of hydrology on meeting the design needs for urban water infrastructure led to a strong emphasis on engineering, a disciplinary approach that has also been effective in the development of irrigation and water supply dams around the world. Toward the latter part of the twentieth century, hydrology expanded beyond this engineering focus to include ecological and environmental considerations, particularly in terms of water quality. Now, hydrologists are increasingly not only defining current freshwater systems, but also predicting the distribution, changes in, and quality of surface and ground waters in response to natural and anthropogenic change.

The Properties of Water

Water is a unique substance. Water molecules consist of two hydrogen atoms and one oxygen atom (H₂O). The bonding between these atoms is very strong, which
means that it takes a lot of energy to break water molecules down into their separate components of hydrogen and oxygen. In addition, the way that the two hydrogen atoms (which have a positive charge) and the oxygen atom (which has a negative charge) are arranged means that water molecules have a positive “end” and a negative “end”—that is, they are dipolar. The polarity generates an attractive force between the positively charged end of one water molecule to the negatively charged end of another. This force, referred to as hydrogen bonding, is almost unique to water.

Water’s molecular structure is responsible for many of its distinct properties. At temperatures less than 0°C, hydrogen bonds lock the molecules into a tight lattice of ice. When temperatures rise above 0°C, however, a small proportion of the hydrogen bonds break, partly collapsing the rigid ice lattice and reducing the density of the substance. This is the process of melting, and the hydrogen bond lattice explains why ice is less dense than liquid water, causing ice to float.

Ice flotation has important implications for aquatic life. If water froze from the bottom up, aquatic plants and animals would be increasingly forced upward as the water body froze and ultimately would die. But because water freezes from the top down, aquatic life can survive below the icy surface in the underlying waters. Colder water with a temperature around 4°C sinks, and in still bodies of water such as lakes and reservoirs, this sinking can create layers of colder water underlying warm water at the surface. These temperature and density differences mean that colder water does not mix with the warmer water above, and it is consequently separated from the atmosphere—the source of dissolved oxygen in the water column. As a result, the deeper, cold water becomes anoxic or depleted in oxygen. This has consequences for biochemical processes and for the distribution of living organisms. For example, sulfur-reducing bacteria proliferate in the deeper, cold anoxic waters, while fish, invertebrates, phytoplankton, and zooplankton populate oxygenated waters closer to the surface.

As the temperature of water increases, important molecular responses occur. More hydrogen bonds break, separating the molecules until all are broken above the boiling point (100°C) and the liquid water changes to steam. Significantly, water is the only substance that occurs in all three states—solid, liquid, and gas—within the normal surface temperature range on Earth.

When water is heated, much of the heat energy serves to break the hydrogen bonds rather than increasing the temperature. Consequently, in contrast to many other substances, water can absorb a large amount of heat before its temperature increases significantly. Because of this high specific heat or thermal capacity of water, a large body of water, such as the ocean, will not heat up as much as nearby land. Consequently, oceans, lakes and reservoirs moderate the magnitude and rates of temperature changes at daily, seasonal, and annual scales, which, among other things, drives the Earth’s ocean and global climatic circulations. At a smaller scale, this property means that warm-blooded animals can regulate their temperatures.

Latent heat is the energy that is released or absorbed by a system when it is changing from solid to liquid, liquid to gas, or vice versa. When water changes form, there is no change of temperature because all the energy is being used to break the hydrogen bonds rather than in actual warming. Energy is used in the transition from water to steam but is released when steam condenses to water. A significant consequence of latent heat absorption in evaporation and its release on condensation is that heat is transported from the place of evaporation to the place of condensation. On a global scale, such latent heat transfer results in a transfer of heat energy toward the poles from the tropical oceans, where water evaporates in large volumes, and from land and water surfaces to the atmosphere. This process prevents extreme temperature variations and moderates Earth’s climate. This moderated climate is one of the major reasons that life exists on Earth.

Hydrogen bonding and polarity of the water molecule also influence the physical properties of cohesion (sticks to itself), adhesion (sticks to other materials), and surface tension (behaves elastically). Cohesion causes water to “bead” on waxy surfaces. Surface tension produces the spherical shape of raindrops and dew. Surface tension also allows some invertebrates to move across the water’s surface without sinking.

Water’s adherence to other materials is the process of wetting. When water is poured into glass tubing, it sticks to the sides, and the attraction between the positively charged hydrogen ends of the water molecules and the negatively charged oxygen electrons in the glass is strong enough to cause the water to move upward along the glass surface. Such capillary action is how water flows upward through rock or soil against the force of gravity and how plants raise water from their roots.

The polarity and hydrogen bonding in water molecules mean that many substances are soluble in water to some degree, giving water its nickname of “the universal solvent.” Salts, such as NaCl, dissolve readily because the oppositely charged ions of sodium (Na+) and chlorine (Cl−) attach to the oppositely charged ends of the water molecule. When each ion is surrounded by a cloud of water molecules, the ions are prevented from recombining. The ability of water to dissolve most substances has enormous implications for weathering of rock minerals, dissolution of pollutants and their transport in streams, the high salinity of ocean water, and the transport of solutes within living organisms.
The Hydrological Cycle

During the era of lunar exploration astronauts sent back to Earth images of our “blue planet”—a planetary body dominated by water. From these images, a popular belief has emerged that Earth has a plentiful supply of water to sustain life. Water does cover approximately 70 percent of the planet’s surface, but only a tiny fraction (around 2.5 percent) of the total resource occurs as freshwater, which is mostly locked up in ice caps, glaciers, and groundwater. Available freshwater stocks in the world’s rivers, freshwater lakes, swamps, and water vapor in the atmosphere add up to a mere 0.77 percent of the total global water reserves. The stocks and flows of water at a global scale are shown in table 1.

The hydrological cycle (see figure 1) is a simplified conceptual model that describes the continual circulation of water through the atmosphere, the biosphere (the zone of life on the planet), and the lithosphere (the outer layers of the Earth’s crust and mantle). Solar energy and gravity drive this circulation. Water within the ocean, for example, evaporates as it is heated by the sun and becomes condensation and cloud formation. The water vapor then falls to the surface as precipitation.

Table 1. Earth’s Water Resources

<table>
<thead>
<tr>
<th>Water Source</th>
<th>Percent of Freshwater</th>
<th>Percent of Total Global Water Reserves</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lakes and swamps</td>
<td>0.29</td>
<td>0.008</td>
</tr>
<tr>
<td>Rivers</td>
<td>0.006</td>
<td>0.0002</td>
</tr>
<tr>
<td>Glaciers, permanent ice cover, and permafrost/ground ice</td>
<td>69.38</td>
<td>1.73</td>
</tr>
<tr>
<td>Groundwater</td>
<td>30</td>
<td>0.76</td>
</tr>
<tr>
<td>Atmospheric water</td>
<td>0.04</td>
<td>0.001</td>
</tr>
<tr>
<td><strong>Total global freshwater</strong></td>
<td><strong>100</strong>*</td>
<td><strong>2.5</strong></td>
</tr>
</tbody>
</table>

*Includes soil moisture and biological water, and arctic islands.

Nearly 70 percent of the Earth’s freshwater is frozen; most of the rest (30 percent) occurs as groundwater.

Figure 1. The Hydrological Cycle

In the hydrological cycle, water is exchanged between the atmosphere and the Earth and takes numerous pathways across the surface and deep into the ground due to the action of the sun’s energy and gravity. Water is always in motion and always in storage.
desalinated atmospheric water vapor, which, upon subsequent cooling, condenses to form clouds. Some clouds produce rain that falls to the surface. This water can then take various pathways, either being evaporated back into the atmosphere, intercepted by vegetation, infiltrating into the soil and recharging groundwater, or running off the surface into streams, lakes, and reservoirs and ultimately flowing back to the sea, where the cycle begins again. The hydrological cycle describes these processes of evaporation, precipitation, interception, infiltration, surface runoff, throughflow (flow just beneath the ground surface) and groundwater flows. Rates of flow and storage times for each of these processes vary: for example, storage of water can range from days (as streamflow) to millennia and millions of years (groundwater and glacial ice).

The volume of water involved in the hydrological cycle for any given time and place can be described and quantified by the water balance equation, which describes the balance between the inputs and outputs of water:

\[ I = O + \Delta S \]

where \( I \) refers to the total inflows of precipitation, runoff, and groundwater, \( O \) equals the total outflows of evapotranspiration, surface runoff, and groundwater, and \( \Delta S \) is the change of water storage in the soil, vegetation, and other forms of retention. Hydrologists can apply the water balance equation by comparing values of precipitation and evapotranspiration, and estimating water surplus and deficit, soil moisture status, and water runoff for a given place and/or time. Almost all hydrological studies are based around this equation and the theory that underpins it. Hydrological modeling, catchment assessment, and processes of water resource management and decision making all apply the water balance.

The Importance of Water

The properties and abundance of water have immense significance for life on Earth, not simply by producing suitable environmental conditions but also because water plays a vital role in the biochemical processes that regulate, nourish, and maintain living organisms. These processes include the following:

- photosynthesis and respiration
- enzyme catalysis
- metabolism
- transport of solutes (for example nutrients, amino acids, glucose in blood; waste products in perspiration, exhaled air, and urine; nutrients in plants)
- transport of heat and thermoregulation.

Human life is even further dependent on water, because of the highly organized social structures that have evolved. Humans rely on water for food production, sanitation, transportation, manufacturing, processing, and energy production. It is thus not surprising, given the central focus that water has to human survival, that water also underpins spirituality, religious ritual, and symbolism in many past and present cultures. Management of water resources is essential for maintaining health and well-being, the integrity of environmental systems, and productivity. Unsustainable exploitation of the freshwater resource not only places these values at risk at local and regional scales but also has broader, global implications.

Supply, Development, and Management of Water Resources

Freshwater resources are not evenly distributed around the globe: some regions have abundant water while others experience aridity. For example, a single river system, the Amazon in South America, discharges 20 percent of the total global runoff. By contrast, the entire continent of Australia generates only 1 percent. Furthermore, a significant amount of rainfall and associated runoff may occur only at certain times of the year (for example, the tropical monsoons), so that some regions may not have enough water at certain times to meet the needs of users. Added to these timing and distribution issues is the challenge presented by the growth in world populations. At the turn of the twentieth century, there was a global population of 1.6 billion people. This increased to 3 billion at the beginning of the Green Revolution and the expansion of irrigated agriculture in the 1960s, and early in the twenty-first century it approached 7 billion. Because the world’s freshwater resource remains roughly constant, population increase means that the availability of freshwater per person has significantly diminished over the last century, with consequences both for humans and for broader environmental conditions. Of particular concern is the fact that water use has been increasing at more than twice the rate of population growth, because of increasing agricultural use of water (which sat at 87 percent of the available freshwater resources early in the twenty-first century) and urbanization (Bates, Kundzewicz, Wu, and Palutikof 2008).

To address increasing water requirements by the agricultural, industrial, and energy sectors, as well as the drinking water needs of a rapidly growing population, the hydrology of whole catchments and regions can be modified by the construction of dams, weirs, canals, and stream diversions. These engineering structures have profound downstream impacts on the volume of water flowing through river systems, the flow regime (for example, seasonal patterns of flow, or periods of low or no flow), and water quality. Large-scale constructions such as the Three Gorges Dams in China and the Sardar
Sarovar project on the Namada River in India exemplify the complex hydrological, ecological, and socioeconomic impacts produced when a drainage system is modified for human use.

Despite these attempts to create and maintain reliable water supply, water scarcity is an emerging threat in many regions of the world. Water scarcity means that there are insufficient water resources to meet long-term average requirements in a particular area or region. Currently, some 1.2 billion people live in areas where water is physically scarce, and a quarter of the world’s population lives in developing countries where inadequate infrastructure for water delivery and/or treatment imposes water shortages (WHO 2009). Continuing efforts to meet water requirements have resulted increasingly in the unsustainable management and use of water, which occurs when the rate of demand exceeds supply.

A further challenge, associated not only with population growth but also with intensifying urbanization, is the risk to water quality from poor sanitation. Approximately 2.6 billion people live without basic household sanitation, using only rudimentary septic systems, latrines, or open fields and beaches. This lack of sanitation risks human health through waterborne and fecal-oral transmission of bacterial, viral, and protozoan diseases, including, but not limited to, cholera, typhoid, shigella, polio, hepatitis, giardia, and cryptosporidium.

It is not only areas of poor sanitation that are at risk from poor water quality. Waterways downstream from urban sewage treatment plants can be affected by the various solutes that are excreted by humans: endocrine disruptors from plastics, the contraceptive pill, and steroids; pharmaceuticals such as beta-blockers, antiepileptic, antidepressant, and antihypertensive medications; and salts and heavy metals. Humans are at the top of the food chain, and so the ingestion of complex compounds and bioaccumulation can have environmental consequences downstream from our own discharge points. Furthermore, increasing urban populations generate waste from manufacturing, industry, and domestic sources, while in rural areas there is broad use of herbicides, pesticides, and fertilizers. All of these substances, particularly those that dissolve or are adsorbed to sediments, can pollute surface and ground waters at the source and far down catchment. Indeed, the adequacy of water supply for human use in many parts of the world has more to do with water quality than quantity, generally as a result of anthropogenic pollution of water supplies. These factors represent significant and developing pressures on human health and mortality, ecosystem function and health, and sustainability.

The Future

The challenges that face decision makers regarding water in the twenty-first century are numerous and multifaceted. Water is essential for ecosystem function and health, and water supply must meet the needs of a rapidly growing population with its associated water-dependent agricultural and industrial users. There is a lower limit to water availability, below which stress and scarcity occur, with consequences that may not necessarily be reversible in the short term. Risks to water availability as well as to water quality from anthropogenic activities can occur at local, regional, or basin scales and can cross geopolitical boundaries.

These challenges must be managed within a dynamic global setting because of climate change and evolving biophysical, socioeconomic, and geopolitical settings. Management also needs to be based on the principles of sustainability, which aim at optimizing and integrating the triple bottom line of social, economic, and environmental outcomes. The imperative for managing water has been captured by Thomas Odhiambo, past president of the African Academy of Sciences, who said: “The art and practice of equitable distribution of and access to fresh water for all people in the twenty-first century, as a fundamental human right and international obligation, is the mother of all ethical questions of all transboundary natural resources of a finite nature.”

Guaranteeing water access globally requires efforts to conserve water through improved efficiencies. These efficiencies may take the form of water restrictions in urban settings, improvements in irrigation technology, sustainable caps on surface and groundwater use through allocation processes, priority allocation of water for high value crops with low water use, and the development of crop species with low water requirements. In addition, equity of water access and availability needs to be supported through governance and institutional arrangements.
The science of hydrology underpins all these efforts to manage the Earth’s water resources for future generations. Key hydrological knowledge includes the properties of water; its availability as freshwater; the stocks and flows of water that occur at a range of scales across time and space; the connectivity between a stream at its source and its mouth at the bottom of a catchment and between surface and ground waters; the processes which degrade water quality; and the implications of anthropogenic activities on both water quantity and quality.

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See also: Catchment Management; Community Ecology; Dam Removal; Groundwater Management; Human Ecology; Irrigation; Large Marine Ecosystem (LME) Management and Assessment; Marine Protected Areas (MPAs); Ocean Acidification—Management; Rain Gardens; Viewshed Protection; Waste Management; Water Resource Management, Integrated (IWRM)

FURTHER READING


I
The effects of the numerous threats to the sustainability of our ecosystems are often hard to detect, sometimes manifesting themselves when it is too late to take remedial action. Monitoring using species, or groups of species, selected from certain taxa can provide an early warning of changes in ecosystems and can also be used to measure the degree of success with attempts to restore such areas.

Ecosystems around the world face a number of threats, ranging from the local scale, as with pollution or introduced species, through to habitat fragmentation across entire landscapes, and ending in climate change, potentially affecting the whole world. These threats must be monitored so that an understanding can be derived about the impact they are having on the biota and the ecosystem functions and processes that they are associated with. Although direct observation has its value here, such as inspecting photographs taken over several years of a heavily used park, or interviewing locals who have long-term memories of a particular area, changes are often too subtle to detect by such means. It is here that the use of indicator species can provide a sensitive metric for detecting change; but which species?

The use of indicator species, generally referred to as bioindicators, is nothing new. (See sidebar on page 201 for more on this subject.) In times of antiquity, villagers used their observations of plants and animals to understand the seasons and the state of soils or waterways. It was only during the early part of the last century, however, that botanists formally adopted species of plants to indicate various soil types or the existence of various plant communities. Also around that time, European scientists developed a system for using aquatic organisms, such as midge larvae and tubifex worms, to measure the impact of sewage and other wastes, which at the time were freely allowed to enter our rivers. Since then, rigorous attempts have been made to find ideal bioindicators, with books, countless journal articles, and even journals dedicated to the subject (e.g., Ecological Indicators, Environmental Bioindicators, and Journal of Environmental Indicators).

The following table lists a few of the types of organisms that have been used as bioindicators, along with examples of their application.

### Table 1. Bioindicators and Their Applications

<table>
<thead>
<tr>
<th>Indicator Taxon</th>
<th>Examples of Use</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diatoms</td>
<td>River health; water quality in wetlands</td>
</tr>
<tr>
<td>Lichens</td>
<td>Air pollution around industrial areas</td>
</tr>
<tr>
<td>Fungi</td>
<td>Air or water pollution; potential indicator of climate change</td>
</tr>
<tr>
<td>Higher plants</td>
<td>Conservation value of land; presence of metals in soil</td>
</tr>
<tr>
<td>Spiders</td>
<td>Progress with mine-site rehabilitation</td>
</tr>
<tr>
<td>Insects</td>
<td>Mine-site rehabilitation; impact of fire; environmental quality</td>
</tr>
<tr>
<td>Fish</td>
<td>River health; impact of invasive fish; pollution monitoring</td>
</tr>
<tr>
<td>Terrestrial vertebrates</td>
<td>Conservation value of land</td>
</tr>
</tbody>
</table>

Source: Author (2011).

Examples of indicator taxa that have been used for assessing various types of environmental modification.
### USING BIOINDICATORS AS ENVIRONMENTAL MONITORS

There is a centuries-long history of using bioindicators to monitor conditions in the environment; a classic example is that of miners taking canaries down into the mines to monitor the quality of the air. If the canary died, it was an indication that dangerous amounts of methane or carbon dioxide were present. In recent history scientists have begun genetically engineering bioindicators for specific purposes. For example, in 1998, the plant *Arabidopsis* was engineered to react to nuclear pollution left by the Chernobyl meltdown (Patent Lens 2011): the radiation breaks DNA bonds, causing mutations to grow. If radiation levels of a particular area are high, patches of mutated plants provide a visible indicator (Kovalchuk et al. 1998).

Another example of using plants to measure harmful substances in soil involves a weed. Scientists have found one specimen to be particularly helpful in locating land mines. The company Aresa, based in Copenhagen, genetically engineered a species of weed to react to traces of TNT in the soil. When the weed’s leaves turn reddish-purple, it is a visible indication of a land mine’s location (Patent Lens 2011).

There are many advantages to genetically engineering plants as bioindicators. For example, instead of spending time searching for an organism that responds to a stimulus in a desired area, local plants can more readily be selected and genetically engineered to respond to the desired environmental factor. In some cases, plants may replace animals as bioindicators. Today, using the “canary in the coal mine” is considered too cruel a practice to be allowed. Plants provide “an ethically acceptable alternative to animal systems” (Kovalchuk et al. 1998).

There is, however, always controversy around the idea of modifying any organism genetically. Whether modified plants are being used to detect radiation or yield larger crops, there are concerns that altering a plant’s genes can cause unexpected side effects in the ecosystem.

**The Editors**

Sources:


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### Characteristics of an Indicator Species

An indicator species is an organism whose presence, absence, abundance, or condition portrays some significant feature about the environment in which it occurs. Some species continue to exist across a wide range of environmental conditions and therefore may not be good candidates; others may be extremely sensitive to changing environmental conditions, which enhances their potential for bioindication.

The South African scientist Melodie McGeoch (1998) has pointed out that there are various categories of bioindicators. In cases where they are used to measure the health, state, or condition of the environment, they are termed *environmental* or *ecological* indicators. The former is usually reserved for cases involving direct human disturbance, such as mining, while the latter tends to relate to conservation of more natural ecosystems, such as looking at the impact of global warming on native forest ecosystems. In addition, certain groups may be used as surrogates for the diversity or assemblage composition of other taxa; these are referred to as *biodiversity indicators*. They are particularly useful in diverse ecosystems where it is impossible to sample all of the species that exist there.

Indicator species have applicability for biomonitoring at various levels. Some can be used at the individual organism level, as is the case with fish used to monitor the effects of various concentrations of pollutants in water. Individual fish may be placed in aquariums containing progressively increasing pollutant levels, and their response observed, or the occurrence of biochemical abnormalities can be measured, as reviewed by the US environmental toxicologist Michael H. Fulton and...
US fisheries researcher Peter B. Key (2001) in the context of the exposure of fish and invertebrates to organophosphorus pesticides such as malathion or chlorpyrifos. This is of crucial importance as organophosphates interfere with nerve synapse transmission in these organisms, as well as in humans. When biochemical changes are investigated, this is referred to as the study of biomarkers as opposed to bioindicators. Secondly, an entire species in a selected habitat can be used to indicate its condition or state. For instance, attempts have been made to identify European plant species that are confined to ancient woodlands and which therefore might have value in identifying areas of high conservation importance or of tracking progress with woodland restoration. This work has been extensively reviewed by Martin Henry and colleagues (1999). Thirdly, an entire assemblage of organisms may be used to indicate something about the condition or state of the environment. A stress on the environment often results in a range of changes, such as drying of the leaf litter layer, thinning of the canopy, or loss of plant species when a forest is subjected to drought or trampling. Here, a suite of species may be measured, leading to a complex data set that provides a range of information about the environment when analyzed. A good example of this is the GLOBE-net program, which has been proposed by Finnish scientist Jari Niemelä (2000) and others. This uses carabid ground beetle assemblages to assess human-caused landscape changes and assists subsequent management practices. GLOBE-net incorporates a standardized sampling protocol at points selected to represent a gradient of decreasing disturbance from city centers through to the surrounding countryside. Carabids were selected as the indicator group because they are sufficiently varied, both taxonomically and ecologically; they are abundant; and they are sensitive to changes in the environment.

Although not a complete list, indicator species tend to come from taxonomic groups that contain organisms having the following characteristics:

- abundant organisms that are easy to find and gather data on
- organisms that are easily sampled rather than those that require extremely specialized skills and collecting procedures
- taxa that are easy to identify or where local knowledge is available to name them
- groups with high species richness, leading to samples with high information content
- groups containing species with specialized habitats or feeding characteristics and hence responsive to changing environmental conditions
- species that tend to bioaccumulate chemicals, which are often representative of higher trophic levels

It should also be added that the target organism(s) preferably should not be subject to great seasonal variation, as this could present logistic difficulties for monitoring. This explains why macrofungi are not commonly used; their fruiting stage tends to be restricted to short time periods, which are not necessarily predictable.

### Taxa Used as Bioindicators

The choice of organisms as bioindicators in freshwater systems is well established and includes freshwater worms; mollusks; various crustaceans such as shrimps and crayfish; and insect larvae including caddis flies, stone flies, dragonflies, mayflies, and midges, among others. Until recently, when considering terrestrial ecosystems, biologists generally used plants as the primary source of bioindication; if they did consider animals, they tended to focus on the charismatic megafauna, birds, mammals, reptiles, and amphibians. The use of invertebrates has escalated more recently, with examples being found in most regions of the world. Today, we see spiders, mites, collembolans (springtails), hemipterans (true bugs), beetles, ants, and many other groups all being advanced as excellent indicators of environmental features or of biodiversity. According to the Australian scientists Alan Andersen and Jonathan Majer (2004), ants are by far the most commonly used group, and their value as bioindicators has recently been reviewed by these authors. Their value stems from the fact that they are ubiquitous, highly abundant, diverse, of great functional importance, sensitive to environmental change, and are easily sampled.

According to Alan Andersen (1999), the choice of taxon used as a bioindicator has tended to be influenced by personal interest in particular groups, availability of taxonomists, or simply which group has been promoted by practitioners most successfully for its potential as a bioindicator. There is now interest in finding out which taxa are the most effective, and which taxa are the most practical and inexpensive to handle. Several studies have endeavored to answer these questions by drawing up lists of criteria for an ideal bioindicator and then considering the attributes of various taxa against this list; examples of this approach include papers by the British researchers Jeremy Holloway and Nigel Stork (1991) and their US counterparts Jodi Hilty and Adina Merenlender (2000). There have also been attempts to compare the effectiveness of plants, vertebrates, and selected invertebrates in terms of their efficacy as bioindicators.

Although most of these groups have their merits for bioindication, few provide information on all facets of
the changes that an environment may be experiencing. For this reason, many people, including the British entomologist Peter Hammond (1992) believe that a “shopping basket” of taxa should be used, with the selected taxa providing complementary information about the changes that are taking place. For instance, if monitoring progress with mine-site restoration, a survey might include springtails to provide information on the process of decomposition, hemipteran sucking bugs as indicators of understory diversity and health, flies and wasps as indicators of pollination, and so on. Inclusion of one or more focal vertebrate species could also provide information on the conservation value of the area under restoration.

As our world experiences increasing pressure as a result of rising human activity and population size, the sustainability of our ecosystems—natural, forestry, and agricultural—is under threat. If we are to understand these threats and resulting changes, we need reliable, inexpensive, and easily applied tools to monitor these areas. Bioindicators provide such an opportunity and could regularly be applied by land managers and conservation agencies in order to make assessments and take remedial action if necessary.

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See also Biodiversity; Biogeography; Boundary Ecotones; Charismatic Megafauna; Edge Effects; Habitat Fragmentation; Invasive Species; Keystone Species; Outbreak Species; Plant-Animal Interactions; Refugia; Regime Shifts; Resilience

FURTHER READING


Indigenous and traditional knowledge systems are important to all of humanity as the wellspring from which all knowledge originates. The complexity of social, economic, political, and environmental sustainability challenges has prompted scholars, political leaders, and theologians on every continent to search for sources of knowledge that will provide the best solutions to problems that affect everyone and everything on the planet. Indigenous peoples themselves have joined the effort to address problems of sustainability by offering to share their knowledge in exchange for protections.

Knowledge systems originate in human cultures as societies develop their relationships to other peoples, the Earth, and the cosmos. What defines the terms indigenous knowledge and traditional knowledge? How do they differ, and how are they the same? Are some knowledge systems more important or valuable than others? Is there only one “science” or are there many sciences? Is indigenous knowledge or traditional knowledge applicable to illuminating the pressing challenges threatening humankind, such as poverty, food security, climate change, war and peace, illness and disease? Indigenous peoples offer their own perspectives on the content and form of knowledge, and the worldwide academic community has joined in an effort with governments, businesses, nonprofit organizations, and international organizations to explain, define, and comment on indigenous knowledge and traditional knowledge.

Social, economic, and political globalization beginning in the late 1960s thrust metropolitan and indigenous societies into closer proximity, resulting in greater demand for effective communication among them. As indigenous peoples actively engage in efforts to solve complex problems created by human action or natural phenomena, academics, national decision makers, nonprofit organizations, and business planners recognize the significance and relevance of indigenous knowledge and traditional knowledge to the development of new strategies for meeting the challenges of the twenty-first century. Bridging the significant knowledge gap between metropolitan and indigenous societies became an acknowledged priority when the United Nations Commission on Human Rights appointed José Martínez Cobo as a special rapporteur in 1973 to conduct a thirteen-year “Study of the Problem of Discrimination Against Indigenous Populations” (UNCHR 1986) and especially after the UN General Assembly adopted the UN Declaration on the Rights of Indigenous Peoples (UNDRIP) in 2007. International organizations, nongovernmental organizations, governments, and indigenous peoples themselves now seek to record and document indigenous knowledge systems to contribute to sustainability solutions.

Indigenous and Traditional Knowledge

The knowledge and intellectual traditions of indigenous peoples embody significant information on a diverse range of topics, including architecture, irrigation, health
and nutrition, child rearing, botanical sciences, forest management, and astronomy. Interpreting and understanding indigenous knowledge systems has become a top priority in the search for answers to the human survival questions of sustainability and the effective responses to adverse affects of climate change. The effort to understand indigenous and traditional knowledge is complicated because such language is often veiled in ancient languages and cultural practices, because of the complex diversity among indigenous peoples, and because their indigenous societies are sometimes located in remote and inaccessible places.

Indigenous peoples exist on every continent except Antarctica. Researchers disagree on the number of indigenous societies in the world, but most often the number has been placed between six thousand and seven thousand different peoples. What qualities identify these peoples as different from one another may be language, history, territorial location and climatic environment, heritage, social, economic, and political practices, and culture. A group’s degree of isolation from or interaction with other groups influences whether knowledge systems are unique to a particular people or part of a broader collection of peoples.

The expression indigenous knowledge is often equated with the expression traditional knowledge, and indeed they are often used interchangeably. Word usage is important, since using indigenous knowledge (IK), ethnoecology, local knowledge, indigenous technical knowledge (ITK), folk knowledge, traditional knowledge (TK), indigenous science, traditional environmental knowledge (TEK), or simply people’s science can signal how an individual or group is approaching a topic or presenting underlying assumptions (Ellen and Harris 1996). David Turnbull, quoting from several researchers in his article “Working on Incommensurable Knowledge Traditions,” gives specific meaning to local knowledge by suggesting that it results from observations of the “local environment or at a particular site and held by a specific group of people”; he goes on to explain that traditional knowledge is a “cumulative body of knowledge and beliefs, evolving by adaptive process and handed down through generations by cultural transmission” (Turnbull 2009, quoting Fikret Berkes and Carl Folke). Whichever form one chooses, indigenous knowledge identifies a specific body of knowledge associated with a specific people and locality involving an understanding or possession of information, facts, ideas, truths, or principles. Examples of indigenous knowledge include architectural and building principles and ideas for constructing the Egyptian (c. 2500 BCE), Mayan (c. 1000 BCE), and Mississippian (c. 800 CE) pyramids; the ancient city of Anasazi (1200 BCE); the city at Machu Picchu (c. 1400 CE); and the mountaintop city of Cusco (circa 1100 CE). Indigenous knowledge has informed systematically engineered aqueducts in the Tibetan Kingdom (c. 100 BCE) and in modern Sri Lanka. Throughout the world indigenous peoples not only engage in engineering that produces vast transportation systems on land and water (rivers, lakes, and oceans), but on health and healing systems (such as ayurveda, 1500 BCE) and cosmologies and mathematical or numbering systems (Swaziland numbering c. 35,000 BCE, Northern Europe numbers c. 3000 BCE, Egyptian mathematics c. 2000–1800 BCE, Mayan mathematics c. 2000 BCE, Chinese mathematics c. 300 BCE, or Persian mathematics c. 700 CE). Calendrical systems, social organization, economic systems, manufactured textiles, wood and stone construction, the smelting of metals for tools and ornamentation, and organized systematic food and natural resource management systems have all been developed by indigenous knowledge. Some of this knowledge informs contemporary knowledge systems, while much remains to be reclaimed.

Traditional knowledge often refers to a more generalized expression of knowledge associating a people or peoples with time-honored ideas and practices associated with an individual or family. Such knowledge may include spiritual incantations or healing practices; fishing, hunting, and other food producing methods; styles and methods for manufacturing baskets or other containers; and art forms such as drawing, carving, singing, playing a musical instrument, dancing, and sculpting. While there are distinctions to be made between indigenous and traditional knowledge, there is sufficient overlap between the meanings of these and related terms to allow for their interchangeability.

**Defining Indigenous Knowledge**

Although scholars (both indigenous and nonindigenous), organizational doctrines, and institutions contribute to a substantial body of literature that offers various definitions of indigenous knowledge, no common understanding or widely accepted definition has materialized. Depending on the intended use for the definition (academic, political, policy oriented, or demographic), authors have refrained from becoming too specific, making an effort instead to embrace the many different knowledge systems practiced by indigenous peoples.

Erica-Irene Daes, the chairwoman-rapporteur for most of the existence of the United Nations Working Group on Indigenous Populations (in effect from 1982 to 2006) offered what is both a scholarly and working policy definition of indigenous knowledge: “[the] heritage of an indigenous people is not merely a collection of objects, stories and ceremonies, but a complete knowledge system with its own concepts of epistemology, philosophy, and
scientific and logical validity” (Daes 1994, para. 8). This definition is intended to apply generally to all different indigenous knowledge systems and is therefore broadly useful for policy, but it is of limited benefit when addressing a specific knowledge system of a specific indigenous people or peoples. One may begin to explore an indigenous knowledge system with this definition, but not actually comprehend or understand the specific body of knowledge.

The UN Environment Programme (UNEP) combines the broader approach to defining indigenous knowledge with recognition of the variety of knowledge systems that exist in different indigenous communities. UNEP states its definition as follows:

Indigenous Knowledge (IK) can be broadly defined as the knowledge that an indigenous (local) community accumulates over generations of living in a particular environment. This definition encompasses all forms of knowledge—technologies, know-how skills, practices and beliefs—that enable the community to achieve stable livelihoods in their environment. A number of terms are used interchangeably to refer to the concept of IK, including Traditional Knowledge (TK), Indigenous Technical Knowledge (ITK), Local Knowledge (LK) and Indigenous Knowledge System (IKS).

IK is unique to every culture and society and it is embedded in community practices, institutions, relationships and rituals. IK is considered a part of the local knowledge in the sense that it is rooted in a particular community and situated within broader cultural traditions. It is a set of experiences generated by people living in those communities. (UNEP 2011)

The International Bank for Reconstruction and Development, more commonly known as the World Bank, notes the controversies surrounding different definitions for indigenous knowledge, but it tends to favor this view:

Indigenous knowledge is developed and adapted continuously to gradually changing environments and [is] passed down from generation to generation and closely interwoven with people’s cultural values. Indigenous knowledge is also the social capital of the poor, their main asset to invest in the struggle for survival, to produce food, to provide for shelter or to achieve control of their own lives. (World Bank n.d.)

The World Bank’s approach is functional and specifically directed at the application of indigenous knowledge to problems and solutions for development. These definitions attempt to give a broad interpretation of indigenous knowledge as complete systems, whereas some scholars tend rather to narrow indigenous knowledge, confining its meaning to local and environmental topics. Louise Grenier, a Canadian researcher, defines indigenous knowledge as “the unique, traditional, local knowledge existing within and developed around the specific conditions of women and men Indigenous to a particular geographic area” (Grenier 1998, 1).

As a result of their work in Bolivia and Wales, the sociologist Alberto Arce and researcher Eleanor Fisher suggest that a “utilitarian representation of knowledge” by individuals observing indigenous knowledge is only a vague interpretation of the everyday application of knowledge or “local knowledge,” and this approach misses the political and social challenges of a people (Arce and Fisher 2003, 80). By this view, using an observational “lens” prevents a full understanding of the knowledge placed within its social, economic, and political environment. How that knowledge is applied to the actual struggles of a people is lost. To effectively achieve the full application of indigenous knowledge it is essential to recognize the social and political context and bridge cultural boundaries. Arce and Fisher urge negotiated exchanges of knowledge and the application of agreed knowledge.

Scholars have taken on the task of defining indigenous knowledge responding to the challenge of an International Council for Science (ICSU) working group study, which takes the position that indigenous knowledge cannot be assembled. The ICSU asserts that such knowledge “differs from scientific knowledge in that it is local, place based, diverse, and hence incomparable and incapable of being validated by common standards,” but that it is a science that informs western science (Fenstad et al. 2002; Aikenhead and Ogawa 2007). Taking the idea that indigenous knowledge is diverse and turning that to a virtue, Turnbull cites the definition of George J. Sefa Dei, an Ontario-based scholar who has done much research in Nigeria and Ghana:

A body of knowledge associated with the long-term occupancy of a certain place. This knowledge refers to traditional norms and social values, as well as to mental constructs that guide, organize and regulate the people’s ways of living and making sense of their world. It is the sum of experience and knowledge of a given social group and forms the basis of decision making in the face of challenges both familiar and unfamiliar. (Dei, Hall, and Rosenberg 2000)

Clash of Cultures

Diana Taylor, a scholar focused on performance art in the Americas (especially Latin America), contends that culture has two parts. The first of these she attributes to the thinking of social scientists such as the nineteenth-century German sociologist Max Weber and
Conveying Knowledge across Cultural Boundaries

The knowledge systems of indigenous societies have long been set aside as if they are separate from what is commonly identified as Western knowledge. Closer examination of indigenous knowledge by Western scholars and scientists is revealing the importance of valuing diverse knowledge systems and thereby expanding the global knowledge base in order to meet the challenges of sustainability, climate change, food security, health, climate refugees and famine, and political stability. Reaching across cultural boundaries to share knowledge, insights, and solutions to a myriad of problems has become more urgent as the trend of globalization accelerates human interaction.

Diversity of Knowledge

Indigenous peoples' knowledge systems vary from locality to locality and from region to region, reflecting the cultural distinctiveness for each people resulting from their dynamic and evolving relationships with the land and the cosmos. There is not just one form of indigenous knowledge; there are many. While the sources, structures, and methods for acquiring knowledge differ, the themes of change and relationships occur repeatedly.

Rudolph C. Ryser, a member of the Native American Cowlitz tribe and founder of the Center for World Indigenous Studies (CWIS), notes that there are numerous ways of knowing that express the knowledge of different peoples. He writes that there are five different, but related, modes of thought [that] have led to knowing, achieving the ultimate expression of consciousness: apprehending the living universe (Ryser 1998, 19). While there are clearly many culturally animated systems of knowledge, Ryser credits the Greek, Chinese, Romans, Nubians, Indo-Arians, and Mayans for developing civilization-wide bodies of knowledge. The Greek system of thought-based knowledge developed through observation and cycles in which events repeated over time. The Chinese and Nubians contributed a system based in fatalism, where knowledge is expressed in terms of inevitability and certainty. The Roman system of thought was amplified by the Roman Catholic Church through the ages in the form of providentialism, where knowledge is based on the belief that God's will is evident in all things and that the will of God predetermines outcomes. Progressivism, a mode of thinking rooted in the ideas of the seventeenth-century French philosopher René Descartes, bases knowledge on reason, empirical evidence, and constant change. The view developed from Descartes's time posits that knowledge advances toward the good (progress) while inevitably relegating what is considered backward or primitive to the dustbin of history. Ryser offers a fifth mode of thought that generates new knowledge as well—typical of knowledge systems in the Americas before colonization. Likening the system to a spiral, he contends that indigenous peoples responsible for building pyramids, great cities, a mathematical system, calendars, agricultural systems, and social order in the Americas rely on constantly changing conditions where evidence of an event at one point may no longer serve as evidence in the future. These examples of knowledge systems reflect the diversity of human experience over time and at different locations in the world.
All of these knowledge systems contribute to “Western sciences” (or what Ryser calls progressivism) as defined by the European seventeenth-century Age of Enlightenment, in which “humanism produced a version of human nature by tethering to human-ness the requirement of rationality” (Watson and Huntington 2008, 258). Indigenous knowledge should be understood to be equal to Western sciences, and the knowledge of indigenous peoples, such as that having to do with hunting wildlife for food, for example, should be compared with the knowledge of wildlife biologists and ecologists. Indigenous knowledge systems express concepts and ideas in virtually all domains of Western science and over time have directly and indirectly informed Western science as a whole.

Native Knowledge and Modern Challenges

Faced with significant changes in the environment resulting from human activity and natural changes that were recognized beginning in the 1960s, economies around the world struggled with water shortages, desertification, soil erosion, forest degradation, social dislocation, and water pollution. International bodies notably led by the UN Environment Programme (UNEP) began searching for solutions. Indigenous peoples’ rights was introduced into the UN global agenda in the 1970s, and by the 1980s the possibility that indigenous peoples’ knowledge could benefit the world’s economies began to be considered in new international treaties and agreements—especially those dealing with the environment, natural resources, and climate.

Five years after the UN Environment Programme convened the Ad Hoc Working Group of Experts on Biological Diversity (in November 1988), the Convention on Biological Diversity (CBD) became official law in December 1993 with the support of 192 UN member states. This agreement is particularly noteworthy due to the inclusion of a specific article asserting that parties to the agreement will respect, preserve, and maintain indigenous knowledge; the agreement emphasized sharing the benefits of that knowledge as it applied to conservation and sustainability. The particular language used in this convention set in motion efforts to include similar language in subsequent treaties and agreements. In particular, Article 8(j) of the CBD (2011) states:

Each contracting Party shall, as far as possible and as appropriate: Subject to national legislation, respect, preserve and maintain knowledge, innovations and practices of indigenous and local communities embodying traditional lifestyles relevant for the conservation and sustainable use of biological diversity and promote their wider application with the approval and involvement of the holders of such knowledge, innovations and practices and encourage the equitable sharing of the benefits arising from the utilization of such knowledge innovations and practices.

In March 1994, UN member states approved the UN Framework Convention on Climate Change (UNFCCC), establishing a major commitment to documenting, understanding, and applying traditional knowledge to reduce the adverse affects of climate change and develop adaptation strategies. In 1996, the UNFCCC began negotiation of a new climate change treaty to replace the Kyoto Protocol that had been originally developed to implement the 1994 convention. Indigenous knowledge is an increasingly important part of the global debate over best approaches to sustainability.

Traditional knowledge became the focus of another international agreement, the UN Convention to Combat Desertification (UNCCD) in 1996, focusing on countries that face significant drought or desertification. The central locations for drawing on indigenous knowledge in this arena are Africa, the Middle East, and the Mediterranean. A very specific study conducted for the UNCCD centering on traditional knowledge was completed in 1999. The objectives of this study included explaining the main attributes of traditional knowledge, developing an inventory of traditional knowledge in the Mediterranean and identifying successful approaches, and assessing the uses of traditional techniques.

This agreement in its many forms is used to predict and aid in early escape from the consequences of tsunamis, predict and cope with droughts, and traverse the open oceans between islands in the Pacific, the Caribbean Sea, the Atlantic Ocean, and the Indian Ocean.

Samoan indigenous knowledge about the medicinal benefits of the bark of the mamala tree has been of great
interest to researchers at the University of California, Berkeley, who have eagerly sought access to the knowledge and to the trees for the purpose of extracting prostratin, a drug thought to be beneficial for treating the disease HIV (Shetty 2004).

Indigenous knowledge about the “sweet plant” cultivated and used for centuries by the Guarani people of Paraguay demonstrated the beneficial uses of _Stevia rebaudiana Bertoni_ (commonly known as stevia), as a sweetener for bitter teas. The plant’s natural sweetness is considered useful for sweetening beverages and baked foods while not promoting tooth decay, hypertension, and unbalanced flora in the intestines, as common sugar does.

Richard “Umeek” Atleo, hereditary chief of the Nuu-chah-nulth people located on Canada’s Vancouver Island, presents their perspective on indigenous knowledge as “an integrated and orderly whole, [which] thereby recognizes the intrinsic relationship between the physical and spiritual realms” (Atleo 2004). Atleo based this explanation on listening to, remembering, and interpreting origin stories. He regards the Nuu-chah-nulth knowledge system as conceived through the method of _osuminich_, which joins the physical and the spiritual realms to explain phenomena in life, as being similar to the vision quest undertaken as a rite of passage in many Native American groups. Since _osuminich_ is both a personal and secret method, the possibility of joining it with the Western scientific method is problematic. Atleo believes, however, that the Nuu-chah-nulth method of knowledge creation is not inconsistent with the empirical method and that the two methods applied together can bridge the cultural gap and permit the expansion of human knowledge for meeting human challenges.

Placing the Nuu-chah-nulth body of knowledge alongside the knowledge of other peoples to produce a synthesis that is beneficial to both exemplifies the expected outcome offered by those seeking to support indigenous peoples. The conventional approach of transferring knowledge used by development agencies such as the UN Development Programme (UNDP) presumes that one body of knowledge offers a superior solution to problems and challenges faced by “less developed peoples.” This approach has rapidly fallen out of fashion owing to increasing levels of resistance waged by peoples on whom “development” is promoted. A more productive approach in relations between development-oriented agencies is one of collaboration and negotiation where all parties presume a position of equality and sharing.

In the Ovamboland and Kavango region of Botswana and Namibia, the collaborative approach is being employed to promote economic and environmental sustainability through the domestication of local fruit trees. Indigenous knowledge about the best selection of trees and growing conditions in dryland areas is critical to the successful crop propagation (UNESCO 1994–2003).

Indigenous knowledge contributes to the reformation of institutions in India’s Ajmer District, Rajasthan, in the more than one hundred villages of Silora Block, through the Barefoot College. This organization was founded in 1972 to apply indigenous knowledge and skills to solve problems in the villages and the region in a manner different from the educational system introduced by the British. The result is that the community develops its own expertise, reducing the people’s dependency on outside help, which is often seen as useless by villagers.

Examples of applied indigenous knowledge in connection with human sustainability across the full spectrum of human endeavors may be found in indigenous communities, villages, towns, and cities throughout the world. When collaboratively negotiated, indigenous knowledge systems become effective contributors to the global knowledge base for meeting the challenges faced by humankind.

**A Twenty-First Century Shift**

Indigenous knowledge, traditional knowledge, and local knowledge are varied ways of labeling the knowledge systems developed and used by more than six thousand different groups of indigenous peoples throughout the world. These systems of knowledge are part of the global body of knowledge, but with the expansion of European, Asian, and African peoples throughout the world in the sixteenth through the twentieth century, the knowledge systems of indigenous peoples was subordinated to colonizing powers. The challenges of sustainability in the twenty-first century created a shift in attitude toward recognizing indigenous knowledge as not only equal to other forms of knowledge, but also essential to be understood and incorporated into the global body of knowledge for the benefit of all humankind.

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*See also* Biogeography; Comanagement; Human Ecology; Hunting; Permaculture

**Further Reading**


Long-distance movement of organisms to areas far from their native range by humans has produced the phenomenon of biological invasions that affects almost every ecosystem on Earth, homogenizing biotas and often disrupting ecosystem structure and functioning. The management of invasive species—biosecurity—requires multiple interventions; preventing the introduction of potentially invasive species is often the most cost-effective method.

Biological invasion is a phenomenon precipitated by the introduction of species by humans (intentionally or accidentally) to regions where they have never occurred before and which they would not have reached without human assistance. Whether the species are successful in the new region depends on their ability to survive, establish, reproduce, disperse, spread, and interact with resident species in recipient communities. Invasion ecology is the study of all aspects of the human-mediated introductions of organisms, and explores their capacity to survive, naturalize, and invade in the target region; it also considers the costs and benefits of their presence and abundance, with reference to human value systems.

Conceptual advances in the field have been hampered by the uncritical use of terms and concepts, which has complicated communication between researchers themselves, and between researchers, the public, and policy makers. In recent years, however, reasonable agreement has been achieved regarding a lexicon based on the stages through which a species passes during the introduction-naturalization-invasion continuum, and the barriers it must negotiate (Richardson et al. 2000; Richardson 2011; Blackburn, Lockwood, and Cassey 2009); see table 1 on page 212.

Concerns about Non-native Species

Some researchers advocate that the concept of biological invasions should more broadly embrace native species’ range expansions, since the fundamental processes are the same, and since both involve the movement of individuals from a donor community into a recipient community (Davis 2009). Indeed, native species may undergo marked range changes in response to human actions, sometimes resulting in substantially increased abundance and geographical ranges. Such range changes share some important features with invasive alien species, and some are considered undesirable and require management intervention. Some native species can become weedy; examples include the native grass *Elymus athericus*, which has recently spread in salt marshes throughout Europe, and many conifers that are weedy in their native ranges. Dynamics such as these, however, can almost always be traced to human-mediated changes to environmental conditions; in the examples above, this includes increased atmospheric nitrogen deposition for *Elymus*, and fire suppression or altered grazing pressure for conifers. In this article we deal only with biogeographical invasions of species that are non-native to regions because (1) there are often marked differences between alien and native species in behavior, traits, and impacts, and (2) on a global scale, problems caused by expanding native species are trivial compared to those caused by alien species.

The Introduction-Naturalization-Invasion Continuum

Although biological invasions were noted by Charles Darwin and other naturalists of his era, the foundation of invasion ecology as a distinct subdiscipline of ecology
is attributed to Charles Elton, who published his milestone book *Ecology of Invasions by Animals and Plants* in 1958 (Richardson 2011). An international program under the auspices of the Special Committee on Problems with the Environment (SCOPE) in the 1980s stimulated intensive research and defined three basic issues that still underpin most work in invasion ecology: (1) which species invade, (2) which habitats are invaded, and (3) what is the impact of invasive species and how can we manage invasions?

The concept of an introduction-naturalization-invasion continuum was suggested to describe the status of alien species in a given region. It invokes a series of environmental and biotic barriers that a given species needs to negotiate in order to become “alien,” “casual,” “naturalized,” or “invasive” (See table 1).

**Introduction Pathways**

Every invasion starts with a species being introduced from its native geographical location to another region. To become alien it must overcome a geographical barrier;

| Table 1. Overview of Terminology Used in Biological Invasions |
|---------------------------------|-------------------------------------------------|
| **Alien Species** (exotic, introduced, nonindigenous) | Those whose presence in a region is attributable to human actions that enabled them to overcome fundamental biogeographical barriers. Some alien species (a small proportion) form self-replacing populations in the new region, and a subset of these has the capacity to spread over substantial distances from introduction sites. Depending on their status within the naturalization-invasion continuum, alien species may be objectively classified as casual, naturalized, or invasive. |
| **Casual Species** | Those alien species that do not form self-replacing populations in the invaded region and whose persistence depends on repeated introductions of propagules. |
| **Introduction** | Movement of a species due to intentional or accidental human activity from an area where it is native to a region outside that range. |
| **Invasive Species** | Alien species that sustain self-replacing populations over several life cycles; produce reproductive offspring, often in very large numbers at considerable distances from the parent and/or site of introduction; and have the potential to spread over long distances (Richardson et al. 2000; Occhipinti-Ambrogi and Galil 2004; Pyšek et al. 2004). Invasive species are a subset of naturalized species; not all naturalized species become invasive. This definition explicitly excludes any connotation of impact, and is based exclusively on ecological and biogeographical criteria. |
| **Native Species** (indigenous species) | Those that have evolved in a given area or that arrived there by natural means (via range expansion), without the intentional or accidental intervention of humans from an area where they are native. |
| **Naturalized/Established Species** | Those alien species that sustain self-replacing populations for several life cycles or a given period of time (ten years is advocated for plants) without direct intervention by people, or despite human intervention (Richardson et al. 2000; Pyšek et al. 2004). The former term is mostly used with reference to terrestrial plants invasions, the latter with that to animal invasions. |
| **Weeds/Pests** | Cultural terms often applied to plants/animals (not necessarily alien) that live in places where they are not wanted and that have detectable economic or environmental impact, or both. |
| **Transformers** | Invasive species that change the character, condition, form, or nature of ecosystems (Richardson et al. 2000). |

*Source: Adapted from Richardson (2011).*
This happens by means of human-mediated introduction pathways. A universal framework applicable to a wide range of taxonomic groups in terrestrial and aquatic ecosystems recognizes six principal pathway classes (Hulme et al. 2008):

1. **Release**: alien organisms introduced as a commodity and deliberately released (e.g., biocontrol agents, game animals, plants for erosion control)
2. **Escape**: alien organisms introduced as a commodity but escaping unintentionally (e.g., feral crops and livestock, pets, garden plants, live baits)
3. **Contaminant**: unintentional introduction with a specific commodity (e.g., parasites and pests of traded plants and animals)
4. **Stowaway**: unintentional introduction with transport vector
5. **Corridor**: artificial corridors among marine basins
6. **Unaided pathway**: unintentional introduction through natural dispersal of aliens through political borders

**Invasion Hotspots**

Nowadays species are being introduced to new regions at accelerating rates; in Europe, for example, the rate at which alien species from many taxonomic groups have established on the continent has been consistently increasing, and the total number of alien plants, fungi, invertebrates, and vertebrates established in Europe has reached at least eleven thousand (DAISIE 2009). Very few regions and ecosystems remain untouched by invasions. For several reasons, a few regions have become invasion hotspots; these include Australia and New Zealand, western parts of North America, South Africa, and many oceanic islands. Some of these regions are also globally important hotspots of native species biodiversity. For example, in the Hawaiian Islands, the number of naturalized plant species (roughly eight to nine hundred) equals that of native, mostly endemic species. Among the most seriously affected ecosystems are temperate grasslands, especially in North America where large areas of prairies in the Intermountain West and California have been transformed to annual grasslands. Also, it is estimated that about 1 million square kilometers of humid and dry tropical and subtropical forests in North, Central, and South America have been transformed to pastures and invaded by African grasses. South African fynbos, tropical wetlands, and aquatic systems are examples of other severely invaded ecosystems. The Mediterranean Sea, the Ponto-Caspian Sea, and the Great Lakes are examples of regions with devastating aquatic invasions.

Many hypotheses have been proposed to explain why some species introduced to a new location invade and others do not. Some of these concepts address invasiveness of species and their populations, while others focus on the capacity of recipient communities, habitats, ecosystems, or regions to accept new species (i.e., invasibility). But species invasiveness and invasibility are two sides of the same coin, and both need to be considered.

**Concepts Relating to Species Invasiveness**

The extent of the invasion of an alien species generally increases with the time since its introduction to a region; this period is termed **residence time**. The earlier a species was introduced, the more time it has had to fill its potential range. Since alien floras and faunas include species with different residence times, models analyzing determinants of invasion need to filter out the effect of residence time to avoid biasing results in favor of species with long residence times (Richardson and Pyšek 2006).

A lag phase is the time between when an alien species arrives in a new area and the onset of the phase of its exponential increase. Among the reasons for this delay are the initial shortage of suitable (invasive) sites, the absence or shortage of essential mutualists and/or mating partners, and inadequate genetic diversity that hampers invasion processes. Lag times can last for decades or centuries, but some species spread immediately after introduction without any obvious lag time. Associated with lag times is the concept of invasion debt. This concept posits that even if introductions cease and/or other drivers of invasion are relaxed (e.g., propagule pressure is reduced), new invasions will continue to emerge, and already-invasive species will continue to spread and cause potentially greater impacts because large numbers of alien species are already present, many of them in a lag phase. Support for this concept is provided by the fact that the number of
alien species recorded in European countries at present is better explained by how powerful their economies were about a century ago than by more recent economic indicators (Essl et al. 2011).

Invasive species rarely move across the landscape as a continuous front; rather spatial patterns are determined by long-distance dispersal events that are infrequent, often via nonstandard means, but which are of overriding importance. Consequently, invasive populations typically spread via satellite populations that later coalesce. For plants, average rates of long-distance dispersal are at least two orders of magnitude greater than estimates of local dispersal. A daisy, _Wedelia trilobata_, spread from a single focal area to cover 2,500 kilometers of the Queensland coastline in fifteen years, averaging some 167 kilometers of spread per year (Pyšek and Hulme 2005). Eurasian cheatgrass, _Bromus tectorum_, one of the major transformers of North American prairies into annual grasslands, spread over more than 200,000 square kilometers between 1890 and 1930, supported by railway construction. Such rates of spread cannot be explained without invoking long-distance dispersal.

Several hypotheses address the effect of genetic factors in mediating invasions. To invade a new region, an introduced plant species must possess either sufficiently high levels of physiological tolerance and plasticity or undergo genetic differentiation, or both, to achieve required levels of fitness. Many invasive species have greater _phenotypic plasticity_ than co-occurring native species. _Post-introduction evolution_ can, however, be rapid enough to be relevant over the time scales at which invasions occur. Invasive species may evolve by genetic drift and inbreeding in founder populations, by intra- and interspecific hybridization in the introduced range to create novel genotypes, and by drastic changes in selection regimes imposed by novel environments that may cause adaptive evolutionary change. Hybridization has been shown to be an important mechanism of evolution of invasive species, and many widespread and successful invaders are recently formed allopolyploid hybrids. Some hybrid plant taxa or genotypes show increased invasiveness and vigor (e.g., taxa of _Carpobrotus_ in California, or _Fallopia_ in central Europe). Available evidence suggests that some invaders are “born” (released from fitness constraints), while some are “made” (they evolve invasiveness after introduction), and that the relative importance of ecological and evolutionary forces is unique to each plant invasion event (Ellstrand and Schierenbeck 2000).

The _enemy release hypothesis_ proposes that alien species have a better chance of establishing and becoming dominant when released from the negative effects of natural enemies that, in their native range, lead to high mortality rates and reduced productivity (Keane and Crawley 2002). Based on the same principle is the _evolution of increased competitive ability (EICA) hypothesis_, which predicts that plants introduced into an environment that lacks their usual herbivores will experience selection favoring individuals that allocate less energy to defense and more to growth and reproduction. The _resource–enemy release hypothesis_ suggests that fast-growing plant species adapted to high resource availability have less constitutive defenses against enemies and therefore benefit from enemy release more than species from resource-poor environments; the two mechanisms can act in concert to favor invasion (Blumenthal et al. 2009).

Some biological traits known to be associated with invasiveness in plants are those related to size, vigorous spatial growth, high fecundity, efficient dispersal, small genome size, and some physiological features such as high relative growth rate or high specific leaf area. For example, differences in invasiveness among pine species (_Pinus_) can be explained using only three traits that together form a syndrome that favors invasiveness (i.e., seed mass, length of juvenile period, and the interval between years of above-average seed production). If dispersal by vertebrates and
characteristics of fruits are included, invasions of woody species can be reasonably predicted by using this simple suite of traits (Rejmánek and Richardson 1996).

The theory of seed plant invasiveness highlights a low nuclear amount of DNA as a result of selection for the short generation time, membership of alien genera, and large geographic range as factors contributing to the invasiveness of seed plants. Large geographical range is a good predictor of invasion success, probably partly because widespread species are more likely to be known, appreciated, collected, and dispersed by humans, but also because they are more likely to be adapted to a wider range of conditions (Rejmánek 1996). Furthermore, recent studies have shown that the role of species’ biological traits is context dependent, interacting with factors like features of the receiving environment, propagule pressure, residence time, and climate, and that the importance of these traits increases at more advanced stages of the invasion process.

**Concepts Relating to Community Invasibility**

Associated with both species invasiveness and community invasibility is the concept of propagule pressure that encompasses variation in the quantity, quality, composition, and rate of supply of alien organisms to recipient communities or regions (Simberloff 2009). Propagule pressure fundamentally influences the probability of invasions both in space (by widespread release or abundant plantings) and/or time (by long history of cultivation or capture); the more propagules are introduced, the more likely it is that species will proceed along the introduction-naturalization-invasion continuum.

Variation in the extent to which a community, ecosystem, or region is invaded could be simply due to differences in the number of aliens that have arrived in the community. Consequently, it is imperative to ask not only whether a community has more alien species than another, but whether it is intrinsically more susceptible to invasions. We must distinguish between two measures. First, invasibility is the inherent vulnerability of a community to invasion and is ideally measured as the survival rate of alien species introduced to the system, thus accounting for losses due to competition with resident biota, effects of enemies, chance events, and other factors (Lonsdale 1999). Second, invasibility differs from the level of invasion, which integrates the effects of invasibility, propagule pressure, and climate, and is defined as the actual number or proportion of alien species present in a community, habitat, or region. Therefore, relatively resistant communities can be heavily invaded if exposed to high propagule pressure, and even relatively vulnerable communities will experience low-level invasions if propagule pressure is low (Chytry et al. 2008).

Global-scale studies have revealed robust geographical patterns, showing for example that islands are more invasible than mainlands, temperate agricultural or urban sites are among the most invasive biomes, the New World is more invasible than the Old World, and that tropical areas are generally less invaded than extratropical regions (Richardson and Pyšek 2006).

Among the concepts put forward to explain invasibility is the diversity-invasibility hypothesis, which holds that more biologically diverse communities are less susceptible to invasion than species-poor communities. Empirical tests of the effects of species richness on invasibility have produced ambiguous results. The hypothesis is usually tested by exploring the relationship between the numbers of native and alien species, which appears negative at very small spatial scales (reflecting competition among species, thus supporting biotic resistance) but positive at larger scales (as more alien species tend to occur in areas with high richness of native species).

Invasional meltdown refers to a phenomenon whereby alien species facilitate one another’s establishment, spread, and impacts (Simberloff and Von Holle 1999). Potentially facilitative effects include positive interactions of invading plants with soil biota, with documented switches from negative plant-soil community feedback in native ranges to positive plant-soil community feedback in the invaded ranges (Callaway et al. 2004). Similarly, the “grass-fire cycle,” in which invasive alien grasses change the distribution and abundance of fine fuels, results in more frequent fires, or even in introducing regular fires to non-fire-prone ecosystems. This profound alteration of ecosystem functioning favors further invasion of fire-tolerant alien species and has had radical effects on biodiversity in many semiarid systems. An example of direct facilitation is alien frugivorous birds in the Hawaiian Islands that promote the spread of the alien tree *Morella faya* by eating its fruits and dispersing its seeds; the tree itself is a nitrogen-fixer that invades nutrient-poor lava flows, making them more suitable for invasion by other plants. The latter interaction is an example of indirect facilitation when an alien species modifies environmental conditions or disturbance regimes in a manner that promotes the establishment of subsequent invaders.

The fluctuating resources theory of invasibility predicts that pronounced fluctuations in resource availability enhance invasibility of a community if they coincide with the availability of propagules required to initiate an invasion (Davis, Grime, and Thompson 2000). This is because invading species must have access to available resources (e.g., light, nutrients, water for plants, food, shelter, space, mates for animals) and because a species will be
more successful in invading a community if it does not encounter intense competition for these resources from resident species. An increase in resource availability can happen if the rate at which resources are supplied from external sources is faster than the rate at which the resident biota can use them, or if the resident biota’s use of resources declines.

Impacts on Biodiversity and Ecosystem Functioning

Adding a new species to an area often changes the structure or functioning of the ecosystem. Such effects, generally termed “impacts,” may manifest at the level of populations, communities, or ecosystems. Impact is the description or quantification of how an alien species affects the physical, chemical, and biological environment. It may be conceptualized as the product of the range size of the invader, its average abundance per unit area across that range, and the effect per individual or per biomass unit of the invader (Parker et al. 1999). Another approach, used by the Millennium Ecosystem Assessment, considers impacts relative to specific types of ecosystem services: supporting (i.e., major ecosystem resources and energy cycles), provisioning (i.e., production of goods), regulating (i.e., maintenance of ecosystem processes), and cultural services (i.e., nonmaterial benefits) (Vilà et al. 2010). Impacts of invasive species are sometimes rapid and dramatic, especially where they result in the transformation of ecosystems. Examples are invasive grasses radically changing fire regimes, or invasive insects that transform ecosystem functioning by altering carbon, nutrient, and hydrologic cycles. Invasive plants are known to change vegetation structure at large spatial scales, such as that of native rain forests over more than 200,000 hectares of the Hawaiian Islands, as a result of replacing native species at different canopy levels (Asner et al. 2008).

Other effects may be subtle, indirect, and slow, yet they may have radical consequences for ecosystem functioning over longer time scales. Invasive species, for example, via the introduction of alien pollinators, seed dispersers, herbivores, predators, or plants, may cause profound disruptions to plant reproductive mutualisms, and there is increasing evidence of severe impacts due to invasive species infiltrating such networks. Himalayan balsam (Impatiens glandulifera) reduces pollination and reproductive success by co-opting pollinators from co-flowering native plants. Another example of an invasive species indirectly reducing survival of a native species is the crayfish plague in central Europe, where the alien American crayfish Orconectes astacus is a main vector of the crayfish plague pathogen, Aphanomyces astaci, which causes massive mortalities of a native crayfish species but not of the alien crayfish.

The impact of biological invasions on species richness and diversity translates to biotic homogenization, a term used for reduced distinctiveness of biological communities. Over the past few centuries, globalization resulting from human activities has altered the composition of regional biotas through two fundamental processes: extinctions of native and introductions of alien plant species. In Europe for example, since invasions exceeded extinctions, both processes acting in concert have made European regional floras less unique (Winter et al. 2009).

In many parts of the world, impacts have clear economic implications for humans, for example as a result of reduced stream flow from watersheds in South African fynbos following alien tree invasions; increased drought and soil salinity following Tamarix species invasions in the southwestern United States; or through disruption to fishing and navigation after the invasion of aquatic plants such as Eichhornia crassipes. Impacts of alien plants are assessed using biological, ecological, and economic currencies. In South African fynbos, the estimated cost of clearing alien plants from catchments, although substantial, is very small—approximately 5 percent of the estimated loss in the value of services provided by these ecosystems, water in particular. Cost-benefit analysis of Tamarix invasions in riparian areas in the southwestern United States showed that, considered over fifty-five years, eradication is economically justifiable. Recent estimates of economic costs caused by biological invasions in Europe amounts to €12.7 billion annually. In addition, many invasive species’ impacts invoke various dimensions of human value systems: they cause or transmit human diseases or ailments, host parasites of pets and livestock, cause injuries and allergies, accumulate toxins that are transferred to human food, represent hazards to health by contamination of soil and water, impede recreational activities and tourism, exert aesthetic impact, and deteriorate the quality of environment (Pyšek and Richardson 2010).

Management of Biological Invasions

International, regional, and local strategies to manage invasions need to realize that most alien plant species are innocuous and many are highly beneficial. Objective means must be devised for focusing limited resources on those species that are known to, or could, cause substantial problems. Key management options are prevention, early detection and eradication, containment, and various forms of mitigation. Mapping these onto the introduction-naturalization-invasion continuum defines several broad
zones; these, and efforts toward preventing introductions of potential invasive species, define the domain of biosecurity, that is, the management of risks posed by organisms to the economy, environment, and human health through the key management options. Finally, various forms of anthropogenic change, synergisms, and nonlinearities are affecting invasions in complex ways. These factors, combined with rapid changes associated with climate change, must be borne in mind when assessing management options. In many parts of the world, the harmful effects of invasive alien species are widely recognized, and multiscale (local-regional-national-international) programs are underway to reduce their current and potential future impacts (Pyšek and Richardson 2010).

Risk assessment is the crucial first step in the risk-management process, related to prevention; it is undertaken to evaluate the likelihood of the entry, establishment, and spread of an alien species in a given region, and the extent and severity of ecological, social, and economic impacts. Preventing the introduction of species with a high risk of becoming invasive is the most cost-effective management strategy. Most attention has been focused on organism-based protocols, and screening procedures with good accuracy rates (greater than 80 percent in many cases) are now available for diverse regions and taxa (Pyšek and Richardson 2010). For example, it has been shown in Australia that the use of a weed-risk assessment scheme provides net economic benefits by allowing authorities to screen out costly invasive species. Even after accounting for lost revenue from the small percentage of valuable non-weeds that may be incorrectly rejected, they showed that screening could save the country US$1.67 billion over fifty years (Keller, Lodge, and Finnoff 2007).

In many instances, the best way of reducing introductions is through pathway management. For example trade with ornamental plants and shipping are the primary pathways for introductions of plants and aquatic organisms, respectively, and elucidation of the vectors that are implicated allows for specific management interventions (Pyšek and Richardson 2010). An important issue relates to responsibilities for invasions resulting from particular pathways. Some suggest that for organisms introduced by the release (see above for definitions) pathway, responsibility should be with the applicant; for escape, with the importer; for a contaminant, with the exporter; for stow-away, with the carrier; for dispersal corridors, with the developer; and in case of unaided pathways, the polluter-pays principle should be applied (Hulme et al. 2008). The first two pathways are subject to national regulations, whereas the others require international policies. This is one area where effective management of biological invasions demands complex multisector and multinational collaboration, and success in such ventures holds the key to reducing the influx of alien species.

Since the multiple pathways of introduction and the huge volume of traded commodities make the interception of all potentially invasive alien species unrealistic, early detection / rapid response initiatives are another crucial part of integrated programs for dealing with invasive species. Many new high-tech diagnostic tools have been developed, including, for example, gene probes for plankton trawls, or DNA barcoding and acoustic sensors to detect Asian long-horned beetles. But the issue of early detection highlights the crucial role of taxonomy in invasion biology. In many regions, alien species come from all over the world; identifying these species is a major challenge, and misidentification can have serious consequences.

Biological control has become the foundation of sustainable control efforts for many invasive species, especially plants, in many regions, but there is renewed interest in eradication (the extirpation of an entire population of an alien species within a designated management unit). Mammals are relatively easy to eradicate, and many successful eradications have been reported, mainly from islands, for cats, foxes, goats, rats, and other mammal species. Among the most widely cited projects were those on the seaweed Caulerpa taxifolia, eradicated from a lagoon in California in 2006, and the marine mussel Mytilopsis sallei, eradicated from a harbor in northern Australia. There are also reports of successful eradications of invasive alien plants, such as Cenchrus echinatus, eradicated from a Hawaiian island, and the herb Bassia scoparia from Australia. Costs of eradication projects increase dramatically as the size of the infestation increases, however, making eradication of plant species occupying more than one
thousand hectares very unlikely, given the resources typically committed to such operations.

Changing Management Approaches

Invasion ecology is rapidly becoming interlinked and interweaved with other disciplines such as conservation biology, restoration ecology, global change biology, and reintroduction ecology. This unification is only beginning, and there are considerable challenges. New frameworks are required for integrating insights from disparate disciplines—for example, to integrate ecological perspectives with socioeconomic considerations. Biosecurity policies and strategies are still being implemented without adequate conceptualization and verification of keystone assumptions. Every aspect of such policies needs to be researched with a view to improving their scientific underpinnings. There is a crucial need for research at the interface between invasion ecology and policy generation.

Better metrics are needed for quantification of impacts to allow for the objective prioritization of species for action and to facilitate the transfer of information between regions. It is not feasible to study the impacts of all invasive species; one way to go would be to select species representative of taxonomic groups and environments. If these were studied in enough detail they could serve as models for particular types of impact.

Multiple facets of global change pose significant challenges for ecologists and conservation biologists, and new approaches are needed for managing biodiversity. Every effort should be made to keep representative areas, such as protected areas, free of alien species, but in the increasingly human-dominated matrix, more pragmatic approaches will be needed. For example, management may in many cases be more effectively directed toward building and maintaining ecosystems capable of delivering key ecosystem services than attempting to steer degraded ecosystems back to some historic “pristine,” alien-free condition. Novel ecosystems are those comprising species that occur in combinations and relative abundances that have not occurred previously at a given location or biome (Hobbs et al. 2006). For example, many species are currently expanding their ranges in response to climate change. Recent invasion of the palm *Trachycarpus fortunei* into seminatural forests in southern Switzerland is driven by changes in winter temperature and growing season length, which are likely to continue in the future under a warming climate (Walther et al. 2007).

Such ecosystems result from either the degradation or invasion of natural ecosystems or the abandonment of intensively managed systems (Hobbs et al. 2006).

Possibilities for managing some invaded systems most effectively as “novel ecosystems” need careful consideration (Pyšek and Richardson 2010).

Further Reading


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See also: Biodiversity; Biodiversity Hotspots; Biological Corridors; Community Ecology; Ecological Forecasting; Food Webs; Indicator Species; Keystone Species; Plant-Animal Interactions; Population Dynamics; Refugia; Regime Shifts; Succession
Irrigation is an ancient technology that gave rise to organized civilizations. The basic technology stores rainwater and river water for agricultural and other purposes. The irrigation cycle’s four interconnected stages are water storage, water quality control, application, and drainage control. Environmental issues will require governments, organizations, and the public to address water accessibility, water quality, and higher efficiency and efficacy in the global irrigation cycle.

Irrigation has long been recognized as one of the earliest components of organized civilizations. To understand—and strive toward—sustainable irrigation practices, we must consider several factors when assessing the ecological impact of irrigation. These factors include the sociopolitical impacts where irrigation is practiced on a significant scale; an understanding of the irrigation cycle or the prevailing water management practices, as well as the geomorphic context within which irrigation is practiced; and those elements of technology that interface with the irrigation cycle.

Sociopolitical Impacts

Scholars working within historical fields related to ancient societies that practiced irrigation (such as those in Mesopotamia, Egypt, and China) have debated the nature of the emerging states that were created by intensive, irrigated agriculture. They also have addressed in part the ecological ramifications inherent in the relationship between societies practicing irrigation and the states that use it. In the 1950s, for example, the German Marxist Karl Wittfogel elaborated a highly provocative model of hydraulic societies, arguing that the despotic nature of Chinese, Egyptian, and Mesopotamian states was a result of the move toward extensive irrigation systems. In other words, societies that created large-scale systems of water diversion became increasingly bureaucratic and despotic. While Wittfogel’s model was impressive in scope, it lacked historical corroboration and suffered from its creator’s political motivations (Weber 1988, 79; Wittfogel 1981, xxi).

Wittfogel views the hydraulic civilization as a radical break from previous forms of social organizations. The economy and polity depended on a specific division of labor, intensified cultivation, and cooperation on a large scale. These new societies were also imperialistic. “The hydraulic agriculturists,” he notes, “outgrew and outfought the majority of all neighboring peoples wherever local conditions and international circumstances one-sidedly favored an agromanagerial economy and statecraft.” When not engaged in battle, the state mobilized the workforce to repair and construct irrigation and flood-control systems. Wittfogel notes that these labor requirements sometimes cut across class lines, and they were not restricted exclusively to slaves or peasants. “In ancient Mexico,” he observes, “both commoner and upper-class adolescents were instructed in the techniques of digging and damming.” Because of this unparalleled level of labor mobilization, hydraulic societies not only built impressive waterworks, but also constructed roads to improve the infrastructure, temples for worship, and tombs for burial of the despot (Wittfogel 1981, 19–22).

While Wittfogelian patterns of rule and intrastate relations may have prevailed in some societies practicing irrigation, several scholars have produced evidence revealing less dominant models of regional relations in ancient civilizations practicing intensive irrigation. The professor and archaeologist Pavel Dolukhanov argues that social categorizations and the development of arid societies were not as simplistic as previous descriptions.
suggested. Dolukhanov contends that, instead of appearing as an outgrowth of the Mesopotamian state, small canals were organized into larger systems during the middle of the third millennium BCE. Despite his rejection of the causal link between the emergence of the state as a result of building massive works, Dolukhanov recognizes that a significant relationship did exist between the emergence of bureaucratic institutions and large-scale intensive farming through the manipulation of the environment. “Although it is generally accepted now that the ‘hydraulic society’ alone could not result in the emergence of world empires,” he notes, “it is widely acknowledged that irrigation carried with it a substantial increase in the organizing capacities of human groups” (Dolukhanov 1994, 293–294).

A flexible development of irrigation practices revealed the true nature of institution building in Egypt. The archaeologist Karl Butzer emphasizes the dynamic nature of political systems along the Upper and Lower Nile. Waterfalls—and the concomitant break in the flow of the terrain—separated the Upper and Lower Niles, which made complete subjugation to the pharaoh difficult. Such a system facilitated local control over irrigation resources, instead of encouraging more complex, integrated regional systems, despite their association with the pharaoh’s regime (Butzer 1976). Butzer corroborates this with demographic evidence from ancient Egyptian civilization. During the New Kingdom period (c. mid-second millennium BCE), human communities were scattered throughout the kingdom, giving rise to numerous socioeconomic and ecological challenges (Butzer 1976, 50 and 80). The Egyptian scholar Michael Rice (1997), however, recognizes the unifying power of the Nile and the Egyptian state, but notes that the unification of the state often encountered resistance from local institutions. He also places the emergence of irrigation before the consolidation of the state. Furthermore, Rice (1997, 14) believes that the prospects of power and wealth were far more of an impetus for unification than any factors related to irrigation.

As in Egypt, the initial development of irrigation practices in China emerged as part of a noncentralized process. In his 1988 book The Food of China, Eugene Anderson observes: “The consensus among contemporary scholars is that in the Old World control over irrigation was usually decentralized, that the state was well established before the rise of large-scale irrigation systems, and that irrigation agriculture had little to do with the development of highly centralized government” (1988, 26). Anderson contends that there were no large irrigation works in the Shang dynasty (1766–1045 BCE), which preceded the Han dynasty by eight hundred years. While the Shang represented the earliest Chinese civilization, it was during the Han dynasty (206 BCE–220 CE) that major public works ushered in large systems of water control.

Nevertheless, a relationship existed between increased intensive agricultural production and the rise of the state from the earliest Chinese civilizations, including the Shang (Anderson 1988, 26 and 46).

Similarly, the historian Cho-yun Hsu has noted that before the Han dynasty, wells and ponds near fields served as the sources for irrigation instead of large-scale irrigation systems. During the Han period, local governing bodies engaged in relationships with state authorities when large-scale projects were undertaken. In addition, control over irrigation works was transferred from the central to the commandery (or provincial) administration. In effect, this revealed that “water controls had become so numerous and commonplace that initiative for their construction had to lie with local officials.” The private sector dabbled in hydraulic works as well (Hsu 1980, 5).

Legal structures within political economies also played a critical role in determining the sustainable potential for irrigation development. In the Americas, during the era of Spanish colonization and imperialism (fifteenth through nineteenth centuries CE), water rights were based on the Iberian concept of community rights over individual water rights. As a result, Spanish settlements in areas such as the present-day US–Mexico border region attempted to adapt an inclusive water policy to the varied constituents of a community located within an arid environment. While it is not surprising that well-heeled hacendados were often given privileges over the less fortunate constituents of these communities, scholars have expressed surprise at the number of instances in which legal disputes favored natives over Spaniards or Mexicans of pure European descent. As the Mexican historian Michael C. Meyer has observed in his landmark study, Water in the Hispanic Southwest: A Social and Legal History, 1550–1850:

[There] are enough examples of Indians, mestizos, and poor Spaniards coming out of the courts with more water than they entered to conclude that the voluminous legislation designed to protect the interests of the disadvantaged, both before and after Mexican independence, was not completely in vain. Compromise and concern for the common good were not merely lofty goals rejected cavalierly in the courts of the Hispanic Southwest. They were not simply guises making possible the cohabitation of the judge with his conscience. They were fundamental principles brought to bear even in the most complex of water adjudications and even when the status of one of the litigants would have suggested that his opponent stood no chance in the impending case. (Meyer 1996, 166)

The California water historian Norris Hundley Jr. sees the biggest differences between the American and the Spanish legal treatment of water rights in two places. First, the Spanish system in California, based on the Plan of Pitic (1783), guaranteed no one a specific quantity
of water, a step that provided flexibility in times of drought (Hundley 2001, 39–41). Second, looking back over the centuries, Hundley perceives a sea change in societal values, namely the onset of unfettered individualism, during the transition from Spanish and Mexican legal systems to American legal values in the late 1840s. He writes, “Viewed from the vantage point of the twenty-first century, Hispanic principles contrast sharply with the individualism and monopolistic impulses of those who flocked to California following the American conquest in 1846. Admittedly, Spain and Mexico’s imprint on the waterscape differed significantly from that of aboriginal Californians, but it paled in comparison with what was to come” (Hundley 2001, 64).

Just as the transition from Spanish, and then Mexican, to US control over water systems in the American Southwest occasioned a significant evolution in the ways in which water was apportioned, the early twentieth-first century has also given rise to new approaches to water distribution. According to the legal concept of prior appropriation, which largely dictates water distribution in the western United States, the first individual or entity to use water from a river develops a legal right for its use. While these rights have often been considered sacrosanct, legal challenges have sometimes prevailed over this system in favor of environmental concerns. From an economic standpoint, discussions regarding the reapportionment of water resources have also led to calls for the monetization of river resources. In actual fact, many farmers and water districts have sold their water rights to communities in need of water, thus facilitating the transfer of resources without recourse to litigation. Improved farming and irrigation techniques hold out promise to close the gap between growing urban needs and the continued importance of farming.  

Irrigation Cycle Management and Sustainability

While irrigation served as a catalyst for the emergence and evolution of ancient and modern civilizations, the management of the holistic irrigation cycle largely determined the sustainability of those civilizations. The irrigation cycle consists of four stages, intimately linked through the practice of agricultural development. These stages are water storage, water quality control, application, and drainage control. Though each functions separately, their interconnected natures require a delicate balance that can sustain or doom individual fields, as well as entire civilizations, if not properly attended to. In this section, we will consider the impact of water storage and water quality control on the sustainability of irrigation.  

While the earliest systems of irrigation used basins or reservoirs to retain rainwater or water from adjacent rivers and to capture rainfall, modern irrigation systems became increasingly dependent on large-scale dams to store water for agricultural (and other) purposes. Climatic conditions, however, often intervened to compromise the quality and the quantity of the water available behind large dams or reservoirs. As the US environmentalist Patrick McCully (1996) has noted, “In the region of 170 cubic kilometres of water evaporates from the world’s reservoirs every year, more than 7 percent of the total amount of freshwater consumed by all human activities.” Chief among the high-temperature, arid region dams responsible for the depletion of water resources for irrigation through evaporation is the iconic Hoover Dam, located in the western United States near Las Vegas, Nevada. Scholars estimate that as much as one-third of the water stored behind the dam evaporates before release and use for irrigation (McCully 1996, 40).

The retention of water behind large dams both affects the amount of water available to irrigators for agricultural use and impinges on the quality of the water accessible for irrigators. In the twentieth-century American West, for example, the series of dams and reservoirs placed on the Colorado River leads to the evaporation of impounded water and increases the salinity and the chemical content of the water for users downstream. (The chemical composition originates from urban use or pesticides and often reflects high levels of elements present in pesticides, such as selenium.) The paradox of these large systems is that as salinity increases in the available water downstream, irrigators need greater amounts of water to push the salt content of their water allotments through the drainage cycle on their fields. Furthermore, improper management of the
entire irrigation cycle creates collateral costs for irrigators. McCully (1996, 40) notes that high salinity not only harms plants, but also damages irrigation equipment. In addition, the number of times that water is used within a watershed basin compromises the initial quality of the water available for irrigators downstream. In the Colorado River basin, this became most apparent in the 1960s and 1970s as water deliveries in both the United States and Mexico, near the river’s delta, were blighted by highly saline water that traveled downstream (Ward 2003). Thus, the entire management of a river basin’s water usage and the chemical composition of residual returns to the river directly affect the quality of water available for users downstream. Finally, the availability and quality of water in turn affects the prevailing crop mix, also raising questions about efficient water usage. In arid regions, or areas of significant urban growth, for example, the demand of perennial plants, such as those cultivated in orchards, can place a serious strain on the amount of water available for urban areas, or vice versa. Consequently, the prevailing usages of water can also affect availability and water quality.

Perhaps the single, most difficult element of the irrigation cycle to manage is the drainage process. This is because of the geomorphologic orientation, or geologic structure, of fertile tracts of land, and, most important, because of its relationship to how water is stored and water quality is maintained. In ancient civilizations such as those in Mesopotamia, the scale of irrigation operations did not necessarily doom Mesopotamian civilization; instead the Mesopotamians’ inability to properly manage drainage operations in those locations doomed it. Irrigation officials not only miscalculated the impact that siltation would have on their canal systems, but also the impact of the salinization of fertile farmlands due to poor drainage. As the water expert Sandra Postel (1999, 26) notes, “By the seventh century AD, salt buildup had reached damaging levels in parts of the plain. Records show that 15,000 slaves were forced to work in the southern region peeling off the upper sterile layers of soil to get to the more fertile layers below.” In contrast, the Egyptian irrigation system that evolved ancienly in the Nile River basin provides one of the best examples of the relationship between natural flooding, water storage, and inherent drainage capabilities. Because of a better-regulated natural drainage system, farming in the Nile River basin far outlasted its counterpart civilizations in Mesopotamia and the Indus River valley in Pakistan and India. According to Postel (1999, 35), the overall stability of Egypt’s socioeconomic institutions, and the lack of salinization, contributed to the rise of the most sustainable irrigation-based civilization in human history. Much of this stability can be attributed to the balance sought between natural cycles of the river, local systems of water management, and an advantageous drainage system that forestalled the accumulation of salinity. Ultimately, management of the holistic irrigation cycle plays the most vital role in determining the sustainability of irrigation practices.

Technology and Application

Throughout human history, simple and advanced technologies have played a crucial role in irrigation and irrigation cycle management. Ideally, naturally occurring conditions where drainage and water acquisition reach equilibrium, negate the need for technological intervention in the irrigation cycle, yet such conditions are scarcely found in nature. In ancient times, a variety of technological devices were used to lift water into fields. In Mesopotamia, for example, small weirs made of brush and earth diverted water from streams into irrigation canals, while in ancient Egypt and Sumeria, waterwheels with buckets lifted water from rivers or canals for agricultural use (McCully 1996, 13–14). Aside from the evolution of dams, which indirectly aid irrigation and are meant to enhance water storage, technologies remained elementary until the twentieth century.

Advances in water purification and in delivery of irrigation water to plants offered promising alternatives in the post–World War II era. Perhaps the least sustainable of these alternatives for improving water availability and quality were offered by desalination technologies. The removal of salt from ocean water—or aquatic alchemy—has long been the pursuit of scientists. The key element that would make such a process sustainable is the identification of low-cost, low-risk forms of energy to power the process. During the Cold War, scientists in the United States, Israel, and other nations, under the umbrella of the International Atomic Energy Agency, promoted the concept of nuclear-powered desalination plants. Plans to place a plant in seismically active regions of Southern California and northern Mexico were dismissed. Conventionally powered desalination plants, however, were subsequently built near Yuma, Arizona, and in Israel during the 1970s and 1980s, as well as in other locations around the world.

In the twenty-first century, modifications and advances in irrigation application technology have stressed two features: efficiency and accessibility. In large measure, these advances recognize the emerging world’s severe problems of access to water for agricultural production and for urban growth. From the high-tech perspective, improved irrigation methods often include more than simply the tools for delivering water to plants. Improved water management techniques, including the precise tracking of weather conditions (such as, for
example, the California Irrigation Management Information Service, decreases the amount of water needed to irrigate a field traditionally flooded one or more times a year (Postel 1999, 180–181). These techniques also improve the quality and efficacy of irrigation, adding fewer salts and pollutants to local water sources (Postel 1999, 167). USDA advances in software development, such as with the Natural Resource Conservation Service Scheduler, provide farmers with up-to-date information on field conditions, weather, and moisture evaporation rates, which can be used in turn to optimize water usage (Postel 1999, 181–182). Drip irrigation technologies, like improved managerial techniques, require less water to irrigate crops but also tend to improve crop yields. Finally, the evolution of affordable, low-tech solutions has brought sustainability within reach of the emerging world. For example, pumps, such as the treadle pumps used in Bangladesh, have transformed the ability of small-scale farmers in the emerging world to farm in a productive and sustainable fashion (Postel 1999, 171–179 and 205–209).

Outlook in the Twenty-First Century

If present levels of governmental and public attention to environmental issues, such as climate change, are any indication, we can expect even greater attention paid to the issues of water accessibility, water quality, and higher efficiency and efficacy in the irrigation cycle throughout the world. While water pricing programs have been used with some effectiveness throughout the world, greater attention to a balance between access to water for irrigation, apportionment according to optimal pricing (which is accomplished either by selling the water at an agreed-upon price on a year-to-year basis or selling the actual rights to the water), and innovations targeting higher water quality and efficient application will dominate discussions between governmental planners, nongovernmental organizations, and the global citizenry.

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See also Agricultural Intensification; Agroecology; Community Ecology; Groundwater Management; Human Ecology; Hydrology; Water Resource Management, Integrated (IWRM)

FURTHER READING

Keystone species are those species whose importance to an ecosystem’s structure, composition, and function is disproportionately large relative to their abundance. These species can be of any life form, but they have in common an effect on their environment that is always greater than what can be expected based on their biomass. Well-studied examples include sea stars, beavers, bears, corals, elephants, and hummingbirds.

A species whose importance to community and ecosystem structure, composition, and function is disproportionately large relative to its abundance is referred to as a keystone species. As the name implies, keystone species play key roles in ecosystems. They are distinguishable from dominant species, which also have large roles in ecosystems but solely by virtue of being abundant. Keystone species, even when rare, can drastically modify or create habitats and influence the interaction between species in a community. An example of this can be beavers that create dams on rivers and streams, notably changing the previous habitat. Because keystone species are so important within communities, the removal of one often results in significant loss of biodiversity. The concept of the keystone species, originally proposed by the US zoologist and University of Washington professor Robert T. Paine, was a transformative notion in biology.

Keystone species can be any type of organism, including plants, animals, bacteria, or fungi. Ways to detect them vary, but an effective strategy to determine what is and what is not a keystone species is through removal experiments, in which a researcher excludes the suspected keystone species from some parts of a habitat and compares areas with and without the species. This is how Paine conducted his groundbreaking 1966 experiment, in which he excluded the sea star (*Pisaster ochraceous*) from a stretch of shoreline in Makah Bay, Washington, in the United States. (The sea stars in the photograph above by Marjolijn Kaiser are in Oregon.) His comparison showed that the relatively uncommon sea star had a huge influence on the tidal pool community. When the sea star was excluded from pools, the ecosystem lost almost half its resident diversity. Similar experiments involving other predators, such as bass, wolves, and jaguars, or herbivores, such as deer and elephants, have shown similar effects.

One factor that can help define a keystone species is functional redundancy. In other words, if a species were to disappear from its community, are there other species that can fill its role? Some communities have more species redundancy than others, and therefore fewer keystone species (i.e., fewer species with fundamental roles in the ecosystems that cannot be replaced by other species). In a given community, the extinction of a keystone species will produce drastic changes. Therefore, to maintain ecosystem functioning and services (like water purification and carbon sequestration), it may be critical to identify and protect those species.

Types of Keystone Species

There are many types of keystone species, and some of them have been thoroughly studied. Predators are typically defined as keystone species, because it takes only a few to regulate populations of other species in lower trophic levels. Many species that create or modify habitats, called ecosystem engineers, are also keystone species.
African Elephants as a Keystone Species

The World Wildlife Fund (WWF) is one organization that works to protect the habitat of numerous species; one of its programs is the African Elephant Program, which aims to conserve forest and savanna elephant populations through projects and policies. The following is an excerpt from their website:

African elephants once numbered in the millions across Africa, but by the mid-1980s their populations had been devastated by poaching. The status of the species now varies greatly across the continent. Some populations remain in danger due to poaching for meat and ivory, habitat loss, and conflict with humans.

Elephants are important because their future is tied to much of Africa’s rich biodiversity. Scientists consider African elephants to be keystone species as they help to maintain suitable habitats for many other species in savanna and forest ecosystems.

Elephants directly influence forest composition and density, and can alter the broader landscape. In tropical forests, elephants create clearings and gaps in the canopy that encourage tree regeneration. In the savannas, they can reduce bush cover to create an environment favorable to a mix of browsing and grazing animals.

Many plant species also have evolved seeds that are dependent on passing through an elephant’s digestive tract before they can germinate; it is calculated that at least a third of tree species in west African forests rely on elephants in this way for distribution of their future generations.


This is the case of the Canadian beaver and some species of African termites (in the genus Odontotermes), which build mounds that contain high levels of nutrients and thus can be colonized by many plant species. The presence of these nutritionally rich termite mounds can change an entire landscape. Large herbivores may also modify the habitat and the community through their feeding activity. One example of this is the African elephant (Loxodonta africana) in the savannas of southern Africa (discussed below). Also, many invasive species, which are exotic species that produce significant changes in a native ecosystem, can be keystone species in the invaded ecosystem. The main types of keystone species and the effects if they become locally extinct are described below.

Predator

Like wolves and sea stars, some predator species play unique roles in their ecosystem by regulating the populations of their prey. Their extirpation can affect the abundance and presence of other predators and lead to the elimination of both prey and competitors. The effects can cascade down to lower trophic levels; for example, the elimination of wolves leads to great increases in populations of deer, which in turn leads to destruction of certain plant species favored by deer.

Prey

When a prey species is removed from an ecosystem, this leaves fewer prey available to feed the predators. If the remaining prey species are more sensitive to the increased predation pressure, they might become rare or extinct within the ecosystem. Further loss of prey species could eventually lead to collapse of the predator population.

Plant

Many herbivores, pollinators, and seed dispersers specialize in and depend on specific plant species for food or shelter. The extinction of that plant could lead to a population crash of these other dependent animal species.
Link

Some species, such as bees and hummingbirds, play key roles in the maintenance of plant populations by providing pollination services that maintain gene flow and secure plant fecundity. Therefore, absence of these pollinators can affect all species that depend on them directly or indirectly.

Ecosystem Engineer

Species that create or modify habitats, such as beavers (*Castor canadensis*), can strongly affect ecosystem nutrient cycling. Shifts in available nutrients can directly and indirectly affect animal or plant species that use the same habitat.

Examples of Keystone Species

Even prior to Paine’s seminal work and terminology, biologists had studied and defined many species as unique and necessary components of a given ecosystem, despite their rarity or low numbers. Many species have been widely studied in their role as keystone species.

Sea Stars

This is the quintessential example of a key- stone species since Paine’s experiment in 1966. Sea stars are a key predator of mussels. The absence of sea stars can drastically impact ecosystems, including changes in diversity and abundance of other species in the habitat, affecting different trophic levels. For example, in the absence of sea stars, diversity was reduced from fifteen species to only eight.

Bears

Brown bears (*Ursus arctos*) as a predator constitute a keystone species by regulating the population of their prey species, but they also have a keystone species role regarding the cycling of nutrients, primarily nitrogen, by incorporating nutrients from rivers into riparian ecosystems. These bears capture Pacific salmon when the fish are spawning in upstream rivers. The bears feed and deposit salmon carcasses further inland, where they decompose and fertilize the riparian areas with nutrients that otherwise may not be incorporated into the local terrestrial ecosystem. Brown bears thus act as nutrient vectors that affect an entire ecosystem.

Beavers

Beavers (*Castor canadensis*) are the classic example of an ecosystem engineer because they create dams in rivers. These dams significantly alter the flux of nutrients and therefore the growth and abundance of local plants and animals. Their tremendous effect can be observed in Tierra del Fuego, an area of South America (in Chile and Argentina) where they have been introduced. Beaver are not native to South America, and no other native species has the ability to create dams in rivers, so beavers are altering the local ecosystems, replacing the slow-growing *Nothofagus* trees for meadows. This change in the structure of the ecosystem provides evidence on the fundamental role that this species has in its native and exotic range.

Corals

The compact ivory bush coral (*Oculina arbuscula*) is considered a keystone species because it creates new habitat. This coral species is endemic to the inshore and nearshore bottomland habitats of North and South Carolina in the United States. It is the only coral species found in this region. It forms complex branching colonies that provide shelter to over three hundred species of invertebrates that are known to live and complete much of their life cycle around the coral’s branches.

African Elephants

In the savannas of Africa, elephants (*Loxodonta africana*) are destructive herbivores that consume large quantities...
of woody plants and often uproot, break, and destroy the trees and shrubs on which they feed. The decreased cover and density of woody vegetation favors the growth and production of grasses, rapidly changing an area from woods to savanna. Many other herbivores that feed on the grasses benefit from the activities of elephants.

Hummingbirds

Hummingbirds are functionally important in many ecosystems by providing pollination services to many plant species. They exemplify a link keystone species. These highly specialized birds pollinate plants that have adapted to be pollinated only by these bird species. They serve as mobile links between plant populations in different landscapes, facilitating pollen movement (and therefore gene flow) often over considerable distances. Pollination triggers seed production and therefore plant population survival. An example of hummingbirds’ fundamental role can be found in southern South America in forests of Patagonia (in Argentina and Chile). The hummingbird species Sephanoides sephanoides pollinates nearly 20 percent of the local woody flora. These plant species could go extinct or become very rare if this bird disappeared, since no other species is adapted to pollinate them.

How Keystone Species Affect Ecosystems

Many ecosystem effects are attributed to keystone species. For example, in Paine’s original work, he reported that diversity of the tidal pool community decreased dramatically when the keystone species (the sea star) was removed. The predator preferred to forage on the most abundant mussel (Mytilus californianus), and when the predator was removed, the population of this mussel exploded in numbers that prevented many other species from existing in the tidal pools. So this keystone predator increases community diversity by foraging on the most abundant species, which benefits less abundant prey species. Other studies on predator species have found similar results.

Many dominant species depend on mutualism for their survival. Therefore, such mutualistic species can play a fundamental role in ecosystem function, and their removal can change drastically its dynamics, such as in food webs containing hummingbirds. Similarly, keystone species that are ecosystem engineers can, by creating or altering habitats, directly affect other species that need these areas to obtain food or shelter.

Problems with Other Definitions

The term keystone species has many definitions; and, in some areas of science, the term is more casually used to mean any species that has a very large impact on the studied ecosystem, no matter its abundance or biomass. This casual use of the term has led to attacks on the concept because it may be too vague and therefore meaningless. The phrase has even been freely and loosely borrowed outside biology; for example, it has migrated into business and economics, where “keystone” is used to describe organizations that enhance the business ecosystem by incorporating technological innovations, simplifying the connection between network participants, and/or providing a stable environment. Their importance to business ecosystems is such that their removal would lead to the collapse of the entire ecosystem.

The concept of keystones species helps determine priority species for conservation and habitats in need of protection. Identifying keystone species, however, is not simple, owing to the complexity of nature and its temporal and spatial variability. A species can be keystone under certain circumstances (e.g., a dry year) but redundant in others (e.g., wet or normal years). This complicates the use and detection of keystone species.

Furthermore, there may be inherent problems with basing conservation on keystone species. The concept implies that some species are more important than others in maintaining a given ecosystem, which suggests that more resources should be devoted to protecting them rather than other, more redundant species. This can be a problem given the complexity of natural systems and the well-known fact that interaction strengths change with space and time. Also, it is important to consider that keystone species may just be a human construct based on our limited observational and experimental capacities (i.e., the difference between redundant and keystone species may not exist in nature). Therefore, conservation plans made around them may not be ideal.

The Future

Keystone species are a central concept in biology. The term is widely used in theoretical, applied, and conservation biology and also serves as a heuristic tool in ecology to explain food webs and ecosystem functioning. Keystone species are recognized as a concept to help understand ecosystem diversity and functioning. A bibliographic analysis using the academic search tool Web of Science revealed that the term is not only widely used, but also that the number of publications using it is trending upward. This indicates that the study of keystone species remains an active part of the biological sciences and likely will continue, especially in areas related to invasive species, ecosystem engineers, and biodiversity conservation.

Although it may be problematic and controversial to base conservation plans on keystone species,
understanding what they are, how to detect them, and their full influence on ecosystems may be key for understanding, conserving, and protecting nature. This is especially important for sustainability of ecosystem processes and services that can be fundamental for human well-being and the functioning of ecosystems at the global scale.

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See also Biodiversity; Biodiversity Hotspots; Charismatic Megafauna, Complexity Theory; Edge Effects; Food Webs; Hunting; Indicator Species; Outbreak Species; Plant-Animal Interactions; Refugia; Regime Shifts; Species Reintroduction; Wilderness Areas

FURTHER READING
Landscape architecture involves a wide variety of outdoor design project types in both the natural and built environments, and as such includes making an ecological inventory and analysis to identify opportunities and constraints for land conservation and development. The US-based Sustainable Sites Initiative (SITES) exemplifies recent innovations being adapted in North America, northern and central Europe, Australia, South Korea, China, and Japan; these innovations are based on attention to the ways ecosystems function.

Landscape architecture is the art and science of arranging land so as to adapt it most conveniently, economically, functionally, and aesthetically to any of the varied desires of people. A landscape is the synthesis of all the natural and cultural features—fields, hills, forests, farms, deserts, water, and buildings or other structures—that distinguish one part of the surface of the Earth from another part. Landscape architecture, which involves the planning, design, and management of natural and built environments (Hooper 2007), positively contributes to ecosystem management approaches by incorporating such practices as effective drainage systems, self-sustaining vegetation and wildlife-supporting habitat, site planning to make the best use of solar energy, and resource recycling.

A Landscape Typology

Landscape architects are involved in a wide range of project types outdoors. William Tishler (1989), Stephen Carr et al. (1992), and Mark Francis (2001) have identified those project types, most of which are included in the following list. Frederick Steiner, a professor of landscape architecture and planning at University of Texas at Austin, expanded and/or consolidated the list, providing brief examples of some of the ways in which landscape architects contribute to sustainable ecosystem management:

- **Brownfield redevelopment**: finding sustainable designs for the reuse of industrial or commercial sites, many of which may be contaminated with chemicals and toxins.
- **Botanical gardens**: increasing the use of indigenous plants and sustainable irrigation practices.
- **Campuses**: adapting landscape planning to support sustainable practices in infrastructure development (e.g., using environmentally friendly materials and siting buildings to enhance the use of solar energy to reduce the carbon footprint); community building (providing an environment that contributes to human wellness and well-being); and learning (encouraging curriculum development in the field, which will foster public awareness and open opportunities for a new generation of landscape architects).
- **Cemeteries**: adapting principles (both aesthetic and functional) to enhance the spiritual connection of humans to the natural world and reduce the impact of human activity in it.
- **City, suburban, and rural town planning, as well as regional development**: supporting and contributing to the development of sustainable infrastructure (roadways, subways, and railways; municipal buildings; power and telephone lines; and the configuration of housing and commercial developments); and assessing sustainable land use (commercial, industrial, agricultural, and residential) as the focus or inspiration for landscape design.
- **Community open spaces (urban and rural), gardens (private and public), and plazas**: providing aesthetically pleasing and sustainability-focused environments to enhance people’s quality of life.
• **Green roofs:** adapting previously or newly built roofs as gardens by implementing lightweight soils (enhanced with minerals or nonorganic fillers), climate-appropriate plants (including grasses and shrubs), root barriers, drainage layers, and waterproof membranes to protect the roof—thereby providing insulation, absorbing rainwater rather than creating runoffs, and benefiting birds and other wildlife.

• **Green walls:** using vegetation on walls to mitigate climate and to provide food and habitat for birds and reptiles.

• **Greenways (coined as a combination of the terms greenbelt and parkway):** adapting former railroads, highways, or other transportation routes into a multipurpose “linear park” with vegetation (e.g., the High Line in New York City, the Gold Coast Oceanway in Australia, the EuroVelo cycles routes, and the Trans Canada Trail).

• **Historic landscapes:** using sustainable methods to conserve flora and fauna while preserving the aesthetic sense of the original architecture and design.

• **Housing environments:** planning or reassessing residential areas to increase sustainable practices, such as facilitating the use of solar power, providing open green spaces, and allowing access to public transportation.

• **Institutional and corporate landscapes:** finding innovative ways to redesign or renovate the spaces surrounding the architecture and infrastructure of preexisting (often dehumanizing and unsustainable) sites, and to utilize or adapt principles from other ecosystem friendly approaches when designing new sites.

• **National forests and parks, state parks, and other recreation areas:** working with federal, state, or municipal governments to make the best sustainable use of existing rules, regulations, and laws; and supporting newer and greener practices, including water, soil, and wildlife management.

• **Olympic, World’s Fair, Expo, and other special-use venues:** upholding a long tradition of the landscape architect’s role in developing special venues while focusing on newer and more sustainable building methods, energy use, and access to infrastructure (e.g., transportation).

• **Restored and/or reclaimed natural landscapes:** employing environmentally friendly methods for public access, such as boardwalks; removing invasive flora species, such as purple loosestrife and kudzu; and adapting principles of sustainable water management, such as rain gardens to control runoff.

• **Urban parks and playgrounds:** collaborating with practitioners in the field of urban forestry, which recognizes the benefits of using green spaces to combat air pollution, support biodiversity, and contribute to human emotional and physical well-being.

• **Waterfronts (and/or waterways):** using green practices to slow or prevent erosion and protect water quality, preserve and enhance the historic character and meaning of a waterfront landscape, and provide public access to waterfront areas.

• **Zoos:** providing environments that come as close as possible to the natural habitat of a species, and using sustainable practices (e.g., water management), to foster self-sustaining vegetation.

### Two Landscape Architect Pioneers

The American landscape architect Frederick Law Olmsted Sr. (1822–1903), who designed Central Park in New York City in 1858, worked hard in collaboration with his colleague Calvert Vaux, his sons, John C. Olmsted and Frederick Law Olmsted Jr., and others (including Charles Eliot), to establish landscape architecture as a profession. The senior Olmsted was also involved in the design of private gardens (The Biltmore in Asheville, North Carolina), college campuses (Stanford University), metropolitan park and parkway systems (Boston, Louisville, and Buffalo), new community design (Riverside, Illinois), a World’s Fair (the 1893 World’s Columbian Exposition in Chicago), institutional landscapes (the New York Hospital for the Insane and the McLean Hospital grounds), and state parks (Niagara Falls State Park). Frederick Law Olmsted Jr. continued these activities with his brother and also played a leadership role in establishing the national parks system in the United States. Before the Olmsteds, traditional commissions for landscape gardeners in North America and Europe were for private clients. Olmsted and his followers changed this and, in effect, took the English picturesque tradition public.

Jens Jensen (1860–1951), a landscape architect born in Demark (but who practiced primarily in Chicago; Door County, Wisconsin; Dubuque, Iowa; and Springfield, Illinois), brought to his practice a personal belief in the renewing and civilizing powers of nature. As the leader of the Prairie Style of landscape architecture, he inspired a movement to conserve threatened scenic natural areas; he has been described as more devoted to the landscape of the Midwest than many born there (Henderson 1985). His acquaintances and supporters included the sociologist and reformer Jane Addams; the architect Frank Lloyd Wright; Harriet Monroe, an editor, poet, and patron of the arts; the botanist Henry Cowles; and Illinois governor Frank Lowden. In the twenty-first century the Jens Jensen Legacy Project, sponsored by the Chicago Department of Cultural Affairs and the Chicago Park District, seeks to provide educational opportunities for children and adults, to bolster current efforts to restore and preserve Jensen’s projects, and to
raise awareness of Jensen’s Prairie Style to a new generation of landscape designers.

Landscape architecture, inspired in part by the early work of Olmsted and Jensen—and redefined for the late twentieth and twenty-first century by such notable figures as the co-designer of the National 9/11 Memorial, Peter Walker, whose five-decade-long career has stressed the dynamics among environmental, social, and economic aspects of a site—is a widely practiced environmental design profession in North America, northern and central Europe, Japan, Korea, and Australia. In China, landscape architecture is rapidly growing from ancient traditions in garden design and site planning. As the world becomes more urban, there are increasing concerns about making cities more livable and protecting both natural and cultural areas. Landscape architecture has a well-established competency in urban design and ecological planning and, as a result, its importance is becoming more broadly recognized.

**Traditional and Contemporary Practice**

A landscape architecture project begins with a commission or assignment with clear goals, including the scope of the project, proposed uses and users, and the site boundaries and context. The landscape architect may also be involved in site selection. Once the commission begins and the site is selected, the landscape architect then conducts an inventory and analysis of the site. The Scottish-American landscape architect Ian McHarg, author of *Design with Nature* (1969), advocated the use of ecology as a primary guide to structure, including inventories and analyses. An ecological framework enables the landscape architect to understand how physical and biological systems are structured and how they function (Rottle and Yocom 2011). An ecological inventory includes the climate, geology, physiography, ground and surface water hydrology, soils, plants, animals, settlement history, and current land use of the site. The inventory is compiled through maps, diagrams, and written descriptions. It may also involve transects (cross-sectional studies of the site that reveal relationships), among vegetation, drainage, and soils, for example. Inventories are used to conduct suitability analyses that display opportunities and constraints for the proposed uses.

Such site analyses enable the landscape architect to develop various design options. Frequently, these options are used in a formal environmental impact assessment. Many projects are also subject to review by citizens, public agencies, or the commissioning clients. Landscape architects often employ “before and after” drawings and physical models to illustrate their designs’ consequences. The process may conclude in a plan and/or a final design, both of which may require governmental approval for legal or regulatory reasons. A plan then may be implemented through public policy and/or through private actions. A design typically requires detailed construction documents that specify the dimensions and arrangement of its various elements before the project is built.

Traditionally, the maps and drawings were done by hand. In contemporary practice, computer aided design (CAD), computer renderings, geographic information system (GIS) technology, and geodesign techniques are utilized. CAD software systems are used during the design process as well as in design documentation. GIS technologies are essentially computer mapping programs capable of capturing, storing, analyzing, and displaying geographically referenced information.

**Innovations**

Increasingly, landscape architects are required to demonstrate and measure the outcomes of their designs. The concept of “ecosystem services,” the goods and services of direct or indirect benefit to humans that are provided by natural processes involving the interaction of living elements (such as vegetation and soil organisms) and nonliving elements (such as bedrock, water, and air) has proven to be especially helpful in this regard. Examples of ecosystem services include global and local climate regulation, air and water cleansing, water supply and regulation, erosion and sedimentation control, hazard mitigation, pollination, habitat functions, waste decompositions and treatment, human health and well-being benefits, food and renewable non-food products, and cultural benefits.

The Sustainable Sites Initiative (SITES) is an example of the formal application of the ecosystem services concept (Steiner 2011) with the goal of ecological stewardship. Beginning in 2006, SITES was developed through a partnership among the Lady Bird Johnson Wildflower Center of the University of Texas at Austin, the American Society of Landscape Architects, and the US Botanic Garden.

In the SITES system, ecosystem services are linked to specific actions that are considered as prerequisites and credits for SITES certification. The prerequisites and credits affect decisions concerning site prerequisites and credits. In the SITES system, ecosystem services are linked to specific actions that are considered as prerequisites and credits for SITES certification. The prerequisites and credits affect decisions concerning site selection, pre-design assessment and planning, site design, construction, and operations and maintenance to try to minimize aspects of a project that might potentially cause permanent ecological harm, such as the pollution of waterways or destruction of species. Meanwhile, SITES attempts to enhance or maximize any generative or productive project aspects that might produce cultural benefits or enhance the natural environment (such as increased tree
cover and recharging of aquifers that supply water). The SITES system establishes uniform, consistent standards, but standards that can adjust to the regional variations of climate, soils, and plants.

Of the sixty-six SITES prerequisites and credits, roughly 60 percent tie quantitative measures of performance to credit achievement, while the other 40 percent are primarily prescriptive in nature; all attempt to tie the attainment of credits with the production of ecosystem services (Windhager et al. 2010).

The credits vary significantly in terms of the performance. Of the thirty-nine credits that set quantitative levels of performance, the bulk remain prescriptive in method, with only seven (21 percent) allowing for open-ended attainment of those performance levels. One example of a high-performance-based credit, “Manage stormwater on site” (credit 3.5), provides a method for comparing regionally adjusted, model runoff-curve numbers for pre- and postdevelopment conditions, and it sets different point values based on preservation or reduction of runoff volumes. This type of credit leaves the landscape architect to determine the ways to achieve performance levels. The landscape architect may choose, for example, to incorporate conventional stormwater approaches (such as detention ponds) or low-impact design approaches (such as rain gardens, rainwater harvesting, or green roofs), so long as the methods used may be shown through modeling to meet the performance goal.

The SITES credits move beyond conservation to the restoration of resources. “Preserve or restore appropriate plant biomass on site” (credit 4.6), for example, intends to ensure regionally appropriate levels of vegetation biomass (referred to as biomass density index or BDI) on site that are sufficient to support ecosystem services. For “greenfield” areas—those never previously developed—postdevelopment vegetation-density levels must at least equal historic predevelopment conditions. For greyfield or brownfield sites that have lost significant levels of vegetation through earlier development, the credit provides a greater array of points based on improvement in the amount of vegetation incorporated into the new site design. Postconstruction vegetation amounts are estimates based on cover type after 110 years of growth and compared to appropriate region-specific vegetation levels based on climate and dominant habitat types. Since landscape architects determine how these biomass-density levels are attained, approaches may include everything from preserving existing areas of high-quality vegetation to creating dense, highly formal gardens, to incorporating green walls and roofs, to some mixture of several approaches.

Outlook

The future of landscape architecture is bright. Ecosystem services provide a restorative lens for landscape architecture practice. The concept of ecosystem services promises to help advance the profession by making its contributions to human and nonhuman health and well-being more explicit. Parks, for instance, have long been viewed as beneficial: as green refuges in the city and as places of recreation. Given a greater common understanding of landscape architecture’s aims, parks and other creations of landscape architects can now also be valued for their benefits, such as climate mitigation, improvements in air and water quality, habitat provision, and pollination.

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See also Adaptive Resource Management (ARM); Brownfield Redevelopment; Comanagement; Ecosystem Services; Landscape Planning, Large-Scale; Natural Capital; Nutrient and Biogeochemical Cycling; Permaculture; Rain Gardens; Soil Conservation; Stormwater Management; Urban Agriculture; Urban Forestry; Urban Vegetation; Viewshed Protection

FURTHER READING

Windhager, Steven; Steiner, Frederick; Simmons, Mark T.; & Heymann, David. (2010). Toward ecosystem services as a basis for design. Landscape Journal, 29(2), 107–123.

FURTHER READING
Landscape planning operates at a range of scales, from metropolitan to open countryside. It seeks to promote socioeconomic and ecological sustainability in various ways, especially by maintaining and improving important land-use functions, such as ecosystems, drainage, and local climate regulation. Key activities include conserving protected areas, assessing the visual impact of planned development, and joining up the green infrastructure of urban areas.

Most people are familiar with the work of landscape architects in relation to the design and regeneration of urban spaces. Landscape planning and design also take place on a large scale. There are no clear boundaries to the idea of “large scale”: at the lower end, it could relate to the open-space network of a new suburb (Williams, Joynt, and Hopkins 2010), and at the upper end, to the entire landscape resource of a small country (Kabat et al. 2005). The nature of “planning” in relation to landscape is similarly diverse, but broadly it refers to the ways that public authorities can conserve or promote important properties of land and coast. A core concern of landscape planning centers on the ways in which landscapes differ around the world and therefore help to make places special and recognizable. Unfortunately, many places are presently becoming more and more similar, or homogenized, so landscape planners now aim to protect and reinforce the special characteristics that make places distinctive.

Landscape planning promotes sustainable development (Benson and Roe 2007) and typically pursues a “triple bottom line” in relation to economic, social, and environmental sustainability. Environmental sustainability refers to the continued or improved capacity of physical and ecological functions associated with a landscape, and the ability to continually deliver a range of associated services (Termorshuizen and Opdam 2009). Social and economic sustainability relate, on one hand, to the delivery of services to people, such as helping to attract inward investment (i.e., money coming from outside), and on the other hand, to a self-reinforcing cycle in which local people invest in the landscape because it underpins the production of goods and services. Sustainability requires landscapes to be grounded in history and heritage, retaining and recovering the memories and physical traces of past landscapes, not least because they can furnish wisdom for future landscape care. It also needs to look forward to large-scale interventions, such as the managed removal of coastal defenses, rewilding, and reforestation.

Landscape Planning Concepts

A landscape is not the same as an ecosystem, though they may be closely related—a landscape typically has a distinctive visual identity and is more cultural in the sense that people have either helped to create it or at least share a widely recognized image of it. One popular definition of landscape is that it is “an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors” (Council of Europe 2000). A similar perspective (Phillips 2002) sees large-scale landscape as nature plus people, past plus present, and physical attributes (scenery, nature, historic heritage) plus associative values (social and cultural).

Large-scale landscape planning can be traced back to the designation of national parks from the nineteenth century onward, and it has subsequently diversified into a wider range of practices (Selman 2010). Initially, landscape planning was closely associated with protection, namely, safeguarding special qualities of scenically important rural areas. As the twentieth century
proceeded, other aspects rose in significance. First, as concerns grew about the impact of industrial and urban development, greater importance became attached to assessing and mitigating landscape and visual impacts, and to reclaiming the landscapes of deindustrialized areas (Fairbrother 1970). Second, the worrying declines in biodiversity around the world led scientists to pursue landscape-scale approaches to nature protection. Wildlife cannot thrive when it is reduced to a few preserves, separated by intensively exploited agricultural and urban land; hence, landscape planning often relates to ecological strategies, in an attempt to provide more continuously connected habitat networks across wide areas (Hopkins 2009). Third, while early landscape policy emphasized scenic and visual aspects, more recent studies have taken a multifunctional perspective, attaching greater value to landscape as a framework for integrating environmental, social, and economic systems (Lovell and Johnston 2009). These trends have led to an increasing concern for ordinary landscapes, including urbanized areas.

In view of the varied nature of landscape planning, it is helpful to note the distinction that has been drawn by the Council of Europe (2000) between landscape protection (actions to maintain significant or characteristic features) and landscape planning (strong forward-looking action to enhance, restore, or create). In essence, landscape planning has two different expressions: a protectionist approach, where highly valued cultural landscapes require safeguarding and traditional management to maintain their inherited qualities; and a proactive approach, where poor-quality landscapes benefit from improvement or new landscapes are created.

### Scales of Landscape Planning

Rural landscape planning typically operates at a scale of tens to hundreds of square kilometers. Here, the emphasis has predominantly been on acclaimed landscapes, but there is a growing concern to recognize the qualities of all areas. The most notable landscapes at this scale are national parks, which were created in response to a concern to defend visually iconic areas in the face of urban-related growth and agricultural intensification. Such parks usually comprise relatively unaltered and strictly controlled areas, and fall within the International Union for Conservation of Nature’s (IUCN’s) Protected Area Category II, “national park,” where there is an emphasis on ecosystem protection and recreation. In some densely populated countries, though, a more cultural interpretation is adopted, and they correspond to IUCN’s Category V, “protected landscape/seascape” (Phillips 2002). Thus, for example, the national parks of the United States are strictly protected against hunting, mining, and other consumptive activities, so that they might remain unimpaired for future generations to enjoy, and are directly managed by the US National Park Service. In contrast, the United Kingdom’s national parks are settled and mainly privately owned, where locally based National Park Authorities regulate development, influence farmers, and manage recreation. National parks can vary from the wilderness of Glacier Bay in Alaska (over 13,000 square kilometers) to England’s Peak District (about one-tenth the size), with its cement factories, thirty-eight thousand residents, and up to half a million visitors a week. In France, “regional parks” combine scenic safeguards with green tourism and sustainable economic development—for example, by marketing traditional local products such as specialty cheeses and fruit juices through a coveted “marque” that affirms quality and provenance, thereby stimulating the survival of vernacular farming landscapes.

A further scale of landscape planning has been associated with industrial activity. Many development proposals are subject to an assessment of their landscape and visual impact. Landscape impact assessment typically refers to the degree to which a proposed land-use change is compatible with the surrounding landscape character. It considers whether the area has the innate capacity to absorb the proposed activity and whether its distinctive qualities might be significantly altered. Visual impact assessment is concerned more with the direct physical effect of the proposal, often analyzed in terms of the “zone of visual intrusion,” or the area within which the development is clearly visible (Landscape Institute, IEMA, and Wilson 2002). The challenge for landscape planners is to make a judgment about the acceptability of the proposals and to ascertain ways of mitigating the predicted impact(s). For example, potentially destructive proposals such as new highways can be designed so they fit in with the contours of the landscape and are screened by ecologically diverse vegetation, while strip mining can be ameliorated by careful screening and imaginative restoration programs. At the other end of the industrial life cycle, landscape planning can enhance and restore derelict areas. Given the advances in reclamation technology and the amount of deindustrialization in recent decades, this scale of landscape planning may also be extensive (Ling, Handley, and Rodwell 2007). Thus, the Parc de la Deule, in northern France, comprises a coordinated strategy to reclaim coal mining areas, protect water resources, defragment farms, and improve public open space provision within a 30-kilometer swath (de Vogüé 2007). New rural resource proposals can also have a large-scale impact.

A third scale of landscape planning is associated with metropolitan areas, including the urban fringe. One of the main problems facing planners at this scale is the way in which the landscape has been fragmented by development and intensive farming. A major objective is
therefore to try and reconnect the fragments as a “green infrastructure” (Benedict and McMahon 2006). Rather than seeking to resist change, planners may actively couple new development to landscape creation. Within the urban network, there has been a long tradition of open space provision, occasionally with a conscious realization of landscape scale, such as the US landscape architect Frederick Law Olmsted’s creation of an “emerald necklace” in Boston over a century ago (Zaitzevsky 1982). The planning of green infrastructure often entails identifying natural signatures of relict ecological and hydrological systems within the urban structure.

Three ideas are central to the sustainability of green infrastructure. First, it must be highly accessible, so that it enables people to reduce their carbon footprints if they want to visit good-quality landscape. Second, it must possess spatial continuity, so that it supports landscape functions that depend on connectivity, such as biodiversity processes and floodplain regulation. Third, it must be multifunctional, so that there is simultaneous interaction between a range of co-located ecosystem services. These include wildlife protection, opportunities for health and fitness, promotion of psychological and emotional well-being, water circulation, mitigation of and adaptation to climate change, local food production, enhanced property values, and community engagement.

**Methods of Landscape Planning**

Landscape planners work with three main, often overlapping, methods: protection of special areas, “toolkits” to assess particular qualities, and strategic spatial planning models.

We have previously noted the approach toward protection in national parks. This method may be more generally understood as “designation,” where areas are designated on official maps (Selman 2009). While areas of national importance are firmly protected, other areas of regional or local scenic importance may also attract some degree of conservation. Planning mechanisms in designated areas usually comprise a mix of land acquisition by public or nongovernmental organizations, strict planning controls, site management for traffic and recreation, provision of visitor information, and grant aid for land management.

One popular tool, which we have already noted in relation to industrial and rural resource development, is “landscape and visual impact assessment.” Another widely used technique in older, cultural landscapes is “landscape character assessment,” which helps us to understand what makes places distinctive and special. It can be used by planners to control or encourage types of development relative to the capacity and characteristics of an area. Characterization methods have built on a longer tradition of trying to evaluate the relative values of landscape over wide areas (Bishop and Phillips 2004).

A landmark in the development of strategic landscape planning was Ian McHarg’s *Design with Nature*, which demonstrated how maps of different landscape attributes could be overlaid to identify where new development would integrate most satisfactorily with the intrinsic capacity of places (McHarg 1969). This approach has been superseded by the data storage and analytical capabilities of modern computers, and by modern theories of dynamic ecosystems, but McHarg’s fundamental rationale remains intact. In practice, most approaches to landscape planning now involve a representation and analysis of landscape services through geographic information systems, an assessment of areas where intervention can promote synergies between landscape services, an analysis of priority areas based on criteria such as human need, and implementation methods such as regulatory controls and financial incentives.

Landscape planning can be very complex and technical, yet landscapes matter enormously to ordinary citizens. Therefore, landscape planners increasingly make use of methods to involve other organizations and the general public. Methods typically engage people in mapping the character of their areas, for example by providing expert facilitators to help volunteers undertake desk studies and field surveys of their local landscapes (James and Gittins 2007), or use public participation techniques to inform the design and restoration of future landscapes (Collier and Scott 2010).

**Outlook**

The most striking trend in landscape planning is a shift in emphasis from protecting nationally important scenery
toward promoting landscape quality in all areas. This does not mean that protected areas have become less important—quite the reverse; landscape is now understood as possessing numerous, often unseen, functions in addition to its visual qualities. As green infrastructure becomes more important, so the traditionally rural emphasis of landscape planning declines, and the town-country divide becomes less relevant. Also, there is a growing awareness of economic, social, cultural, political, natural, and technological “drivers of change,” which cannot be stopped but merely influenced and channelled (Schneeberger et al. 2007). Policy makers now accept and even welcome a degree of change so long as it meets sustainability criteria and benefits local communities and economies. The degree to which we choose to embrace or control change drivers remains controversial—for example, how much alteration do we permit in iconic cultural landscapes, or how far can we allow nature to take over on rewilded land or realigned coasts?

Perhaps the major lesson for future landscape planners will be the degree of attention shift from the developed to the developing world. While the importance of multifunctional landscapes will continue to increase in the developed world, the greater challenge will be to safeguard the exceptional natural and cultural assets of rapidly developing countries and to embed green infrastructures within their burgeoning cities.

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See also Coastal Management; Community Ecology; Landscape Architecture; Large Marine Ecosystem (LME) Management and Assessment; Light Pollution and Biological Systems; Marine Protected Areas (MPAs); Rewilding; Road Ecology; Stormwater Management; Viewshed Protection

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Large Marine Ecosystem (LME) Management and Assessment

Large marine ecosystems (LMEs) are ecologically defined and highly productive coastal areas of at least 200,000 square kilometers located along Earth’s continental margins. The goods and services of LMEs that contribute an estimated $12.6 trillion annually to the global economy are in a downward spiral from overfishing, pollution, nutrient over-enrichment, and habitat degradation. An international effort is underway to recover and sustain LME resources.

Large marine ecosystems (LMEs) are defined as productive regions of ocean space at least 200,000 square kilometers (km²) in size that encompass coastal areas from river basins and estuaries seaward to the break or slope of the continental shelf, or to the seaward extent of a well-defined current system along coasts that lack continental shelves. (See figure 1 on page 241.) Earth’s sixty-four LMEs are defined by ecological criteria including (1) bathymetry (i.e., contours of bottom depth), (2) hydrography (i.e., seawater characteristics including salinity, density, and temperature), (3) productivity, and (4) trophically linked populations (i.e., predator-prey transfers of carbon energy through the food chain from plankton to marine mammals and other top predators).

The future is uncertain for the world’s large marine ecosystems. They annually produce 80 percent of usable marine biomass (e.g., fish, seaweed); however, LMEs are becoming increasingly stressed by both natural and anthropogenically induced changes, including climate change, overfishing, and pollution. The potential for consequent negative effects on the sustainable development of LMEs has aroused major international concern. A global effort is underway to restore and sustain LME goods and services; it is focused on the recovery and sustainability of depleted fish stocks, reduction and control of coastal pollution, nutrient over-enrichment and acidification, restoration of degraded habitats, conservation of biodiversity, and mitigation of and adaptation to the effects of climate change.

Levels of surface chlorophyll at the base of marine food chains and the source of primary production in the oceans are measured as grams of carbon per square meter. These rates are persistently higher around the margins of the ocean basins, within the boundaries of the LMEs, than in the open ocean. The high rates of primary production support the high biomass levels of fish within the boundaries of the LMEs.

Through the mid-1980s and 1990s the scientific basis for moving toward ecosystem-based assessment and management of marine resources to reverse the downward spiral in LME goods and services was put forward at annual meetings of the American Association for the Advancement of Science, the International Council for the Exploration of the Sea (ICES), and at international LME conferences. This movement represents a paradigm shift from single-species assessments to multiple-species assessments, and from small scale up to the LME scale for measuring changing ecosystem states on an annual basis, with a focus not only on ecosystem goods but also on ecosystem services. (See table 1 on page 242.)

An ecosystem-based approach has been introduced by the US National Oceanic and Atmospheric Administration (NOAA) in partnership with the UN system and international financial institutions (e.g., World Bank, Global Environment Facility) to improve assessment and management of marine resources in LMEs. A five-module LME indicator approach has proven useful for measuring changing states of LMEs and introducing ecosystem-based adaptive management strategies for (1) improving and sustaining LME productivity and (2) fish and fisheries,
Figure 1. Location of the Earth’s Large Marine Ecosystems

1. East Bering Sea
2. Gulf of Alaska
3. California Current
4. Gulf of California
5. Gulf of Mexico
6. Southeast US Continental Shelf
7. Northeast US Continental Shelf
8. Scotian Shelf
9. Newfoundland-Labrador Shelf
10. Insular Pacific-Hawaiian
11. Pacific Central-American Coastal
12. Caribbean Sea
13. Humboldt Current
14. Patagonian Shelf
15. South Brazil Shelf
16. East Brazil Shelf
17. North Brazil Shelf
18. West Greenland Shelf
19. East Greenland Shelf
20. Barents Sea
21. Norwegian Shelf
22. North Sea
23. Baltic Sea
24. Celtic-Biscay Shelf
25. Iberian Coastal
26. Mediterranean Sea
27. Canary Current
28. Guinea Current
29. Benguela Current
30. Agulhas Current
31. Somali Coastal Current
32. Arabian Sea
33. Red Sea
34. Bay of Bengal
35. Gulf of Thailand
36. South China Sea
37. Sulu-Celebes Sea
38. Indonesian Sea
39. North Australian Shelf
40. Northeast Australian Shelf-Great Barrier Reef
41. East-Central Australian Shelf
42. Southeast Australian Shelf
43. Southwest Australian Shelf
44. West-Central Australian Shelf
45. Northwest Australian Shelf
46. New Zealand Shelf
47. East China Sea
48. Yellow Sea
49. Kuroshio Current
50. Sea of Japan
51. Oyashio Current
52. Okhotsk Sea
53. West Bering Sea
54. Chukchi Sea
55. Beaufort Sea
56. East Siberian Sea
57. Laptev Sea
58. Kara Sea
59. Iceland Shelf
60. Faroe Plateau
61. Antarctic
62. Black Sea
63. Hudson Bay
64. Arctic Ocean


Large marine ecosystems are areas of the ocean characterized by distinct bathymetry, hydrography, productivity, and trophic interactions. Global efforts are focused on the recovery and sustainability of marine goods and services within Earth’s sixty-four LMEs.
Valuation and Governance of LMEs

Coastal waters encompassing LMEs annually contribute US $12.6 trillion to the global economy. The socioeconomic module emphasizes the practical application of scientific findings to optimize and sustain socioeconomic benefits to civil society.

With $3.1 billion in financial support from the Global Environment Facility (GEF)—an international financial organization located within the World Bank and focused on global environmental issues—and the World Bank, partnerships have been forged among five UN agencies (United Nations Development Programme [UNDP], United Nations Environment Programme [UNEP], United Nations Industrial Development Organization [UNIDO], Food and Agriculture Organization [FAO], and the International Oceanographic Commission–UN Educational, Scientific and Cultural Organization [IOC-UNESCO]), the US-NOAA, Norway, Iceland, Germany, and two nongovernmental organizations (International Union for Conservation of Nature [IUCN], World Wildlife Fund [WWF]) to assist 110 countries in Africa, Asia, Latin America, and eastern Europe in carrying forward LME projects. The projects are introducing ecosystem-based assessment and management practices that consider multisectoral interests (e.g., fisheries, transportation, energy production, wind farms, recreation) to recover and sustain depleted fish stocks; restore damaged habitats (e.g. sea grasses, corals, mangroves); reduce and control pollution, nutrient over-enrichment, and acidification; and mitigate and adapt to climate change.

The LME governance module is developed by each LME project to meet high-priority objectives for resource recovery and sustainability. The management framework engages national, regional, and local jurisdictions. Through GEF-supported LME projects, countries are moving toward joint governance arrangements to address the priority transboundary issues identified in the LMEs they share affecting LME fisheries, oil and gas production, transportation, tourism, and offshore energy production. The processes used to make determinations relating to governance include the joint preparation by participating countries of transboundary diagnostic analyses (TDAs) to prioritize issues, and strategic action programs (SAPs) to focus on issues to be resolved to optimize socioeconomic benefits derived from healthy LMEs. The SAPs serve as international agreements guiding the implementation of actions identified and prioritized in the TDAs for advancing toward recovery and sustainability of LME goods and services.

Adaptive Management of LMEs

An overarching adaptive management strategy in the form of a commission and/or serial management actions at different scales within the LME address multiple-user issues, including habitat restoration, fisheries recovery, and other goods and services issues at several scales. Experiences and lessons learned have proven effective at the level of the numerous ministries responsible for the various sectors (e.g., fisheries, transportation, environment, energy, and tourism). The ministerial approvals are obtained at the national levels with full knowledge that the ministries are entering into a five-year agreement, with the option for an additional five years, to address transnational and transboundary issues that have been prioritized through the GEF-supported TDA and SAP processes, thereby integrating local, national, and transboundary interests of the LME project for up to ten years. For example, agreements have been reached for moving forward with ecosystem-based adaptive management actions by the three southwest African countries—Angola, Namibia, and South Africa—sharing the goods and services of the Benguela Current LME (located in the southern Atlantic along the southwest coast of Africa).

Table 1. A Paradigm Shift to Ecosystem-Based Management

<table>
<thead>
<tr>
<th>From</th>
<th>To</th>
</tr>
</thead>
<tbody>
<tr>
<td>Individual species</td>
<td>Ecosystems</td>
</tr>
<tr>
<td>Small spatial scale</td>
<td>Multiple scales</td>
</tr>
<tr>
<td>Short-term perspective</td>
<td>Long-term perspective</td>
</tr>
<tr>
<td>Humans as independent of ecosystems</td>
<td>Humans as integral part of ecosystems</td>
</tr>
<tr>
<td>Management divorced from research</td>
<td>Adaptive management</td>
</tr>
<tr>
<td>Managing commodities</td>
<td>Sustaining production potential for goods and services</td>
</tr>
</tbody>
</table>

Source: Lubchenco (1994).
Africa) under the framework of the Benguela Current LME Commission. Other examples can be found developing among the sixteen countries located along the landward margins of the Guinea Current LME from Guinea Bissau in the north to Angola in the south, in the interim Guinea Current LME Commission. And in Asia, the People’s Republic of China has joined with the Republic of Korea in developing a joint commission for the sustainable development of the goods and services of the Yellow Sea LME.

Comparative LME Assessments

LME projects have had positive effects in transforming governance regimes from single sector foci to multisectoral ecosystem-based adaptive management practices that improve people’s awareness of important-at-risk ecosystem goods and services and support actions aimed at sustainable development of LMEs. The bottom-up TDA and SAP processes allow for national inputs to the LME project from coastal communities. The activities span the extent of country interest (established in the national strategic plans), transboundary resources, and the entire LME. Objectives of LME projects are consistent with the 2002 Johannesburg targets and Plan of Implementation of the World Summit on Sustainable Development to achieve substantial reductions in land-based sources of pollution, introduce an ecosystems approach to marine resource assessment and management, designate a network of marine protected areas, and restore and sustain depleted fish stocks.

The GEF LME project footprint on the global scale encompasses actions to recover and sustain marine goods and services affecting the livelihoods of hundreds of millions of people engaged in marine fisheries, aquaculture, tourism, shipping, energy production, and other marine industry activities in Africa, Asia, Latin America, and eastern Europe.

The application of the five-module LME approach to the assessment and management of marine resources is growing. From a scientific perspective, the importance of comparative assessments among the world’s LMEs is to advance an understanding of the sources and effects of human- and climate-induced changes on sustaining ecosystem goods and services. Major studies at the LME scale have appeared in the journals Science and Nature during the past decade; see references to these studies in the Further Reading section. In one of the published reports (Worm et al. 2006), the authors concluded that the “overfishing” trend in the world’s LMEs could result in the loss of viable wild-caught world marine fisheries by 2048. In a subsequent paper (Worm et al. 2009), the same authors joined with others who questioned the original results and concluded that, in fact, with proper application of science-based management controls, marine populations of fisheries stocks could be recovered from a depleted state and, under scientifically determined annual catch levels, be fished sustainably. Other LME global scale modeling reports published in Nature have focused on the use and the apparent misuse of the “mean trophic level of fish catch” index among the world’s LMEs (Branch et al. 2010) and the positive contribution of catch shares for sustaining marine fisheries in LMEs (Costello, Gaines, and Lynham 2008). Recent comparative LME studies have been published in other journals, including a study on the average annual carrying capacity of the world’s LMEs for supporting fisheries biomass (Christensen et al. 2009) and a study on the effects of climate warming on fisheries biomass yields (Sherman et al. 2009). Other studies project a doubling of nitrogen levels from river basin drainage by 2050, increasing the frequency and extent of dead zones within LME waters unless mitigation actions are implemented to control nutrient over-enrichment (Seitzinger, Sherman, and Lee 2008).

The results of studies at the LME scale are important to the implementation of ecosystem-based assessment and management practices in support of an emerging upward spiral in the recovery of depleted fish stocks and improvements in water quality in LMEs adjacent to the United States and in the Yellow Sea LME bordering the People’s Republic of China and the Republic of Korea. Other positive movements toward LME recovery and sustainability are underway in projects in Africa, Asia, Latin America, and eastern Europe. The key persons engaged in the LME projects are listed along with LME project objectives in a report compiled by NOAA entitled Scope and Objectives of Global Environment Facility Supported Large Marine Ecosystems

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Projects (Sherman, Adams, and Aquarone 2010). In November 2010 a volume describing the global LME recovery effort entitled Sustainable Development of the World’s Large Marine Ecosystems During Climate Change (Sherman and Adams 2010) was released at the 2010 Göteborg Award for Sustainable Development ceremony.

Among the more promising actions leading to the recovery and sustainability of the LMEs is the agreement reached by the People’s Republic of China and the Republic of Korea to improve water quality and reduce the fishing effort in the Yellow Sea LME by 33 percent by 2050. Other countries are supporting and achieving sustainable fishery practices, including Iceland within the Iceland Shelf LME and Norway in the Norwegian Sea LME. The United States is also recovering and sustaining fisheries in the eleven LMEs adjacent to the US coasts.

The global movement supporting the recovery and sustainability of the goods and services of LMEs is growing, particularly among the economically developing countries of the world with the aid of $3.6 billion in financial support from the GEF, the World Bank, and an increasing number of international donor nations (Sherman and Adams 2010).

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See also Coastal Management; Comanagement; Ecological Restoration; Ecosystem Services; Fisheries Management; Global Climate Change; Marine Protected Areas (MPAs); Natural Capital; Ocean Acidification—Management; Ocean Resource Management; Pollution, Point Source; Population Dynamics; Shifting Baselines Syndrome

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Artificial light, particularly outdoor lighting at night, disrupts the mechanisms plants and animals have evolved to cope with and use day–night and seasonal cycles. It affects behaviors such as preparation for winter, predator evasion, nighttime navigation, and migration, threatening the health and survival of life forms, from simple aquatic creatures to complex animals and humans. Fortunately, solutions are simple, inexpensive, and easy to implement.

For a long time light was considered benign, but evidence from biological research is accumulating that light at the wrong time and place can alter and damage the behavior of organisms and ecosystems. Artificial light at night changes the natural habitat, physiology, and biochemistry of organisms in ways that may lead to serious disruption of their behavior and their ability to cope in the environment.

The timing and intensity of light are important factors. During their evolution, species adapted to the changing day–night cycles of the seasons, and to the light of the moon, which varies in brightness over a month. Superimposing artificial light affects the health of animals, the balance between predators and prey, the seasonal behavior of plants, and even the fertility and mental health of humans. The most serious pollution effects of artificial light occur during the dark phase of the moon, or when artificial light intensity is greater than the intensity of the natural environment. Light pollution is a significant and dangerous consequence of modern civilization, and there is growing evidence that it needs to be controlled.

Light Pollution

The night is not dark. There are several sources of natural light at night, even in the most remote regions of Earth, to which life has adapted and on which life depends. The brightest constant source is natural sky glow emitted by the air (sometimes called airglow) as it releases energy absorbed from sunlight during the day. Another source is the stars, whose combined light illuminates the ground to 1/1,000 the level of a full moon. Finally, for a week each month the full moon overwhelms all light sources in the night sky. Wildlife has evolved and adapted to these sources of nighttime light. Some insects and aquatic life even produce low-level light using chemoluminescence as a predatory strategy.

Until the twentieth century, only stars, airglow, and the moon illuminated the Earth at night. The impact of human settlements, lit by isolated fires, was limited. During the past century, however, artificial light has become widespread. Illuminated cities, villages, farms, and roadways are now evident across the landscape.

Whether light is a pollutant depends on the characteristics of the light, including its brightness, color, extent, and duration. When any of these attributes are sufficient to alter the behavior and biology of organisms, the light is a pollutant.

Three additional points should be considered. First, light pollution, like air and water pollution, is not limited by political and administrative boundaries. Although populations are concentrated in urban areas, their combined illumination creates an expansive glow that extends beyond city boundaries. Gas flaring on oil rigs and searchlights on nighttime fishing fleets are also significant sources of sky glow in rural and maritime environments. Second, when the light from individual fixtures is directly visible, they can become significant sources of glare, which disturbs nocturnal organisms and reduces their ability to see into dimmer areas. Finally, the burning of fuel to generate electricity for artificial light releases nearly a quarter of a billion tonnes of carbon dioxide annually into the atmosphere (Mills 2002).
The remainder of this article will expand on these points by outlining the research on the impacts of artificial lighting on biological systems, both ecological and human.

Scotobiology

Organisms have not only developed to fit with the daily light–dark cycle, but this cycle is so engrained that most life depends on it. Altering this cycle affects the health of animals, the seasonal cycle for plants, and even reproduction.

Before effective and realistic controls on artificial lighting can be developed, its impact on biological systems must be known. We must determine at what level artificial light affects organisms. New measurement techniques permit the study of the behavior of organisms under faint natural light and determine lighting thresholds above which their behavior is altered. These illumination thresholds are typically about that of the natural level of the full moon.

As far back as the nineteenth century, researchers noted that light affects the behavior of some species. Some of this knowledge was used to enhance the growth of animals and plants. The adverse effects of an artificial photoperiod were rarely reported. Scotobiology is a new approach to studying the ways in which unpolluted darkness is essential to the normal behavior of organisms and ecosystems. This approach focuses on the negative effects of night lighting on the physiology, biochemistry, and behavior of plants and animals, including humans. Distinct from photobiology, which is the study of the effects of light on organisms, scotobiology focuses on the benefits of darkness and helps establish safe levels of brightness, duration, and color of nighttime illumination. It is now clear that periods of unpolluted darkness are essential for the normal functioning and development of many organisms and the levels to which we should limit light pollution. These thresholds are used to determine at what level artificial lighting becomes a pollutant of concern.

Influences of Light Pollution on Biological Systems

The difference between biological and behavioral evolution is critical to understanding the impact of artificial lighting on species. Behavioral changes can occur within a lifetime, but biological adaptation for the simplest organisms takes generations, and biological evolution for complex life forms requires hundreds to thousands of years. Furthermore, various species exhibit behavioral and biochemical needs for darkness in different ways.

Life forms may tolerate exposure in excess of certain thresholds but with increasing impact on their health and behavior. For example, foraging animals may curtail their activity during the full moon but then recover during the following dark nights. Such is not the case with artificial light, which remains the same night after night and, even when distant, may illuminate the ground over a large region.

Life forms undergo cyclic variations that are scheduled over twenty-four-hour periods. This schedule is maintained by the circadian rhythm. As complex as the circadian rhythm may be, it is generally accepted that the contrast between day and night plays an important role in synchronizing the biochemistry of organisms to their diurnal activity. Natural ambient light may also have either a strong or weak influence, depending on other factors such as temperature, food supply, and the nature of predatory risk. The regular cycle of natural light and dark periods allows organisms to synchronize their activity to longer seasonal changes.

Reviewing a few aspects of these influences on a range of life forms will help reveal the extent of this dependence on naturally dark nights.

Aquatic Life

Small marine animals such as zooplankton have successfully evolved to take advantage of the daylight cycle and tides. As a food supply for larger fish, they have developed strategies to improve their chances of survival. Surface predatory fish need light to see the small zooplankton. The light threshold for predation is about that of the full moon. Therefore the zooplankton remain at depth in daylight and approach the surface to feed only at night. This general pattern is evident in their vertical daily (diel) migration. Although temperature, food availability, and the degree of predation may modify this behavior, the vertical movement is based on ambient light.
Marine animals are very sensitive to night lighting levels. Shore lighting and sky glow from urban areas can exceed the brightness of the full moon. (See figure 1.) In these situations, the vertical movement of zooplankton in the water column can be suppressed, weakening the food chain. Zooplankton are sensitive to short-wavelength blue light because it penetrates the water more deeply than longer-wavelength red light. Therefore, white shoreline lighting will disrupt the shoreline environment. Light fixtures should be set back from the shore, and the use of short-wavelength light should be discouraged.

The impact of artificial lighting can have paradoxical effects. We usually think that illumination will help animals see and survive, but artificial lighting usually has the opposite effect when it confounds animal instinct. The eggs of sea turtles are buried in beach sand. When they hatch and dig to the surface they become easy prey for predatory birds. Their best chance of survival is to quickly run toward the surf.

The sea turtles instinctively get their bearings from the luminance of the waves breaking on the shore, but inland artificial lighting can attract them away from the water. This subjects them to greater predation or being hit by cars along coastal roadways (Salmon 2006).

Insects

Insects are critical to the food chain. At night their relatively simple navigation strategy is to keep their bearings with respect to the stars. They can even compensate for the diurnal motion of the stars across the sky. Nearby stationary artificial lighting disrupts this strategy by attracting them into a spiraling approach toward the light. (See figure 2 on page 248.) This distracts the insects from feeding, mating, and migrating. Furthermore, it concentrates them in specific locations, which greatly increases their exposure to predation.

Flying insects bother people sitting outside after sunset. Historically people used amber lighting, called bug lights, to reduce the attraction of flies. Modern lighting products use white light, which actually attracts the bugs. In fact, researchers use white light to attract and capture insect specimens.
night, which gets progressively shorter with summer and longer as winter approaches.

In temperate latitudes, autumn is marked by an increase in the length of the night. Plants interpret levels of nightly light pollution above the intensity of the full moon as the continuation of short summer nights, which may prevent them from preparing for winter, thus reducing or eliminating their chance of survival. All-night lighting may kill off some species while encouraging more tolerant species to invade the affected area.

Avian Life

Birds make use of the moon and stars to navigate across the sky, particularly during migration. Their ability to use these directional cues is affected by light pollution. As with insects, birds use stars to navigate during migration. This ability is compromised by glare when they confuse bright lights with the moon. It has been estimated that hundreds of millions of birds are attracted to city lighting during their biannual migrations and die from colliding with glass-clad buildings after being disoriented by direct and reflected lights (FLAP 2010).

For example, the city of Toronto, Canada, is along bird migration routes. Ten thousand birds were killed annually by exhaustion and collisions with buildings. With the volunteers’ efforts of the Fatal Light Awareness Program (FLAP) and their discussions with city officials, building managers have been required to turn off unnecessary lighting during the migration period to reduce bird mortality since 1997.

Influence of Light Pollution on Human Health

Some organisms survive environmental conditions by migrating to better climates, but humans can use their intellect to adapt to such problems in situ. There may be a limit to this strategy, however, particularly as it relates to light pollution. For humans, the most important effect of artificial lighting is how it affects our health.

Our circadian rhythm schedules the ebb and flow of hormones and is synchronized to our activities by the day–night cycle. Our hormones break down quickly and are then filtered from our blood. Therefore, if they are not used at the appropriate time, their effectiveness is reduced. Reducing the contrast between daytime and nighttime illumination causes the circadian rhythm to drift out of phase with this activity. This can result in a lack of mental awareness that in turn can exacerbate hazardous situations. It can also have effects that are only apparent in the long term. For example, women working night shifts have a 60 percent higher risk of developing breast cancer than those who worked day shifts (Davis, Mirick, and Stevens...
Light pollution impacts the health of seniors in several ways. The already degraded circadian rhythm of senior citizens can be further deteriorated by the confused light-dark cues that result from light pollution. The mechanism may be due to reduced melatonin levels coupled with the weakened circadian rhythm in older people. In addition, while our eyesight naturally degrades with time, artificial lighting, especially glare and white light, reduces visual acuity by scattering light from imperfections in the eye (Turnera, Van Someren, and Mainster 2010).

Studies have also shown increases in obesity and diabetes with nocturnal lighting. In response to the mounting evidence for the affects of light pollution on human health, the American Medical Association (2009) has passed a resolution identifying light pollution as a health risk.

The influence of lighting on the circadian rhythm does not depend only on the light detectors in our eyes. Our non-visual photoreceptor retinal-ganglion cells respond to illumination levels as low as 0.2 lux, about the brightness of the full moon, and have a peak color sensitivity to blue light. Urban lighting is designed to provide from ten to one thousand times this illumination level. And with the increased use of metal halide and light-emitting diodes, whose white emissions include high levels of blue light, our cities are becoming more inhospitable to human life after dark.

**Management of Light Pollution**

Astronomers first identified outdoor lighting as a problem by noting that the degradation of the night sky forced them away from urban areas to do their work. In the 1980s they took their concern to the public and governments through local initiatives and the creation of the International Dark-Sky Association. Unfortunately, because only 0.1 percent of the population is composed of astronomers and stargazers, most municipal officials and lighting professionals set aside these concerns and continued to increase illumination levels.

By the end of the twentieth century, public concern over environmental degradation had developed a stronger voice, prompting the development of more environmentally sound policies. Environmentalists came to recognize the damaging effects of artificial light, among other stresses on the environment. It ceased to be just an astronomy problem and has become an environmental and human health issue.

Governments have been slow to react to artificial light pollution, in part because of the influence of the outdoor lighting industry—a $26-billion-per-year business—and in part because outdoor lighting is considered necessary for the function and safety of human outdoor activity. The latter reasoning may be more psychological than fact based: although people feel safer in a brightly illuminated area, critical studies of crime statistics show that light at night has no significant effect on actual crime rates (Clark 2002).

Fortunately the solutions to overlighting, and the alternatives, are comparatively simple and inexpensive. The rising costs and environmental impact of energy production and distribution have begun to influence the use of outdoor lighting in many large cities. For example, Flagstaff, Arizona, developed a lighting bylaw in the 1980s to help protect the skies above the Kitt Peak Observatory, located forty miles west of the city; Ottawa has been illuminating its streets to half the level of most cities; Calgary has relamped many of their streets with fully shielded light fixtures and has also significantly reduced the illumination levels. The Czech Republic is the first country to adopt a national lighting policy to limit light pollution.

The amount of energy used for outdoor lighting is difficult to determine. Although there are good records on energy sources, electricity-use data rarely discriminates between indoor and outdoor uses. This is a global problem since, historically, distinguishing the energy use of outdoor lighting has not been considered necessary. In the case of roadway lighting, cities keep separate records, but this represents only a fraction of the energy used for all outdoor lighting. Private and commercial uses include yard and door lighting, advertising, and lighting for security and parking lots, but the energy for each end use is not recorded at this time.

It is usually thought that the brighter the light, the better one can see. Since humans are daytime creatures, artificial light at night improves our ability to see, but numerous studies have shown that too much bright light is less effective. Even a century ago, “Good lighting was defined as that which produces good seeing” (Nutting 1917). In some cases reduced brightness improves visibility. A bright light can scatter in the air, intervening glass and in our eyes to reduce the contrast of less brightly illuminated subjects, thereby reducing their visibility. Bright lights overwhelm our night-vision capacity, so that objects in shadow become more difficult to perceive. A person moving in semidarkness and blinded by glare may fail to see hazards that are dimly lit. A mugger lurking in the shadows, for example, may be invisible if the veiling luminance from an overly bright street light compromises the potential victim’s night vision. The use of outdoor lighting to illuminate buildings after the customers are gone and buildings are closed, and the common use of brightly illuminated advertisements, needs to be reexamined.

Light pollution now causes enough environmental and economic damage that laws may be required to control the levels of permissible pollution in the same way that air and water pollution have been controlled. This brings the
whole question of light pollution into the political and legal realm. Sustained and rational public pressure, including the wide reporting of light pollution issues, may be necessary to effect change. Fortunately, compared to other environmental stresses, light pollution may be the easiest to reduce, with the added benefit of reducing energy use and the carbon footprint of the communities involved.

Challenges and Solutions

For over a century we have been unwittingly performing a strategic experiment on a global scale by steadily increasing artificial outdoor lighting. Illumination levels are now so bright that the effects of artificial light can be readily identified and studied. At the same time, humans have become sensitive to environmental pollution of all kinds, economic factors involved in wasteful use of resources, and ecosystem sustainability. Reducing artificial outdoor lighting can help reduce energy costs, adverse health effects, and dependence on limited resources, while preventing environmental degradation from power generation and distribution. For example, city managers coping with the escalating costs of lighting are turning to fixtures that shine light downward and concentrate it on specific targets, which not only saves money because fewer and less powerful lights can be used, but minimizes glare and reduces light pollution. Such reductions of electricity consumption can contribute greatly to environmental sustainability, by improving the health and survivability of plants, animals, insects, and even humans.

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See also Best Management Practices (BMP); Buffers; Community Ecology; Edge Effects; Landscape Architecture; Landscape Planning, Large-Scale; Pollution, Nonpoint Source; Pollution, Point Source; Rewilding; Wilderness Areas

FURTHER READING


Marine protected areas (MPAs) are a promising tool for protecting marine resources, but they must be carefully designed to ensure success. MPAs tend to be most successful when they are planned with the input of diverse participants and implemented using sound scientific guidelines. Effective MPAs produce significant biological benefits and can contribute to an ecosystem-based approach to management.

Since commitments made by the international community at the 2002 World Summit on Sustainable Development and the Convention on Biological Diversity, the establishment of marine parks and preserves, most frequently termed marine protected areas (MPAs), has increased. Signatory countries are working toward conserving at least 10 percent of each of the world’s 232 marine ecoregions and establishing an ecologically representative and effectively managed MPA network by 2012. By 2010, more than 5,800 MPAs were documented worldwide, covering 4.72 million square kilometers and accounting for 1.2 percent of the ocean (Toropova et al. 2010).

Some marine reserves can be highly effective, consistently producing increases in organism density, biomass, size, and diversity (Lester et al. 2009), yet their estimated coverage is only 0.1 percent of the ocean (Wood et al. 2008). MPAs vary in their levels of protection, ranging from no-take areas that prohibit all fishing, called marine reserves, to areas that allow almost every form of extractive use. MPAs that allow some removal of organisms can still have positive effects on target and nontarget species (Beukers-Stewart et al. 2005). Research shows, however, that partially protected areas produce smaller changes in numbers and sizes of organisms and fewer conservation benefits than fully protected no-take marine reserves (Lester and Halpern 2008).

The broad range of levels of protection that MPAs provide for marine living organisms stems from the reality that many different factors and opinions influence the final choice of MPA rules. How extensive protected areas should be and what activities should be regulated depend on the balance between human uses and conservation goals (Jones 2009). Marine systems are subject to a multitude of uses, both consumptive and nonconsumptive. These uses include commercial, traditional, and recreational fishing; diving; recreational boating; swimming and surfing; aquaculture; renewable energy generated by wind and waves; mining; and oil exploration. Human uses can vary widely in their impact on marine organisms and habitats, yet each must be considered as MPAs are placed into the larger context of marine management.

To achieve conservation goals, MPAs should ideally be connected effectively. For example, even though reserves in the Mediterranean are in relatively close proximity to each other, data show that they are not effectively connected by movements of adult organisms or export of their offspring. As a result, they may be unable to meet the management goal of fully protecting populations of marine life (Grorud-Colvert et al. 2011). Thus the total number of MPAs worldwide may be misleading when assessing their benefits and contribution to conservation targets.

What Makes an MPA Successful?

It can be challenging to evaluate whether MPAs are truly successful at meeting their specific management goals. For example, some designated MPAs are not implemented or enforced—many are merely proposed on paper, and others may have lost their governmental support and protection over time. Verification of MPA status...
relies heavily on the knowledge of local managers, government officials, or resource users and stakeholders.

Evaluation of MPA planning and implementation around the globe reveals that there are certain key factors that lead to management success. Among these, clear objectives or goals for marine management are critical to begin a constructive dialogue about the establishment of an MPA. For example, the process to plan a network of marine reserves and other MPAs along the coast of California began with the enactment of the Marine Life Protection Act, which clearly states the objectives of the MPA network. These include goals to protect the diversity, abundance, and integrity of marine ecosystems and marine life populations; to provide educational and recreational opportunities; and to plan and monitor the MPA network using sound scientific guidelines.

The participation of diverse stakeholder groups that use and enjoy marine resources is also critical to the success of an MPA planning process. In places such as the Philippines, Australia’s Great Barrier Reef, the United Kingdom, and California and Oregon, certain MPA planning initiatives did not consult with stakeholders, leading to early setbacks; those conflicts, however, have led to the better integration of consumptive and nonconsumptive resource users. In fact, in many places the request for an MPA comes from the stakeholders themselves. For example, after witnessing a decline in the black murex snail and rock scallop they harvested, fishermen in Puerto Penasco, Mexico, defined and established a temporary no-take marine reserve to provide a refuge as well as a source for export of young snails and scallops to fished areas outside (Cudney-Bueno et al. 2009).

MPAs are more likely to be successful if their design is informed by the best available scientific information. Although a wealth of research exists about the effects of marine reserves and other MPAs, planning processes may not explicitly mandate the use of scientific guidelines. In addition, MPA planning staff may not know about the scientific resources available to evaluate proposed MPAs. As a result, educational resources to communicate marine reserve science to nonscientists are in development, and more scientists are contributing to MPA dialogues (Grorud-Colvert et al. 2010).

After an MPA is established, compliance, enforcement, and long-term support are critical to ensure continuing success. Involving stakeholders in the planning process encourages their compliance with the MPA rules since these individuals feel ownership of and investment in the MPA (Pollnac et al. 2010). For example, not only do the local fishers patrol and enforce their MPAs around the Philippines’s southeastern Cebu Island, but the bantay dagat (local volunteer enforcers) have banded together from nearby municipalities to protect their MPAs against large commercial fishing vessels poaching fish in their area (Eisma-Osorio et al. 2009). The enforcement of MPAs is a pressing issue around the world, including areas such as the Mediterranean, where a study showed that only three of fifteen Italian marine reserves were adequately enforced. Total fish density was greater in the Italian marine reserves with high enforcement (Guidetti et al. 2008).

Long-term support, both governmental and financial, is critical for managing, enforcing, and maintaining MPAs. Although the marine reserve in Puerto Penasco, Mexico, was initially successful, leading to larger and more abundant target species and empowering fishermen who saw the results of their conservation actions, the protection was not sustainable without formal government recognition.

Fishermen from another community began to harvest rock scallops from the Puerto Penasco reserve and resisted attempts by local fishermen to deter their poaching. Seeing their carefully conserved resource fall into the hands of outsiders, Puerto Penasco divers soon joined in the harvest, saturating the local market (Cudney-Bueno et al. 2009).

Without official designation of MPAs and the political backing for enforcement, an MPA is likely to fail. Because financial resources are key for supporting the enforcement and biological monitoring of an MPA, creative solutions are often required for financing. For instance, the consortium of fishers in the Philippines support their local regional council, which addresses issues of poaching, by contributing a small portion of each municipality’s internal revenue (Eisma-Osorio et al. 2009).

Adaptive Management

Even with clearly set goals, stakeholder buy-in, and effective enforcement, MPAs can vary in how effectively they achieve their goals. To evaluate the success of an MPA,
the assessment should be designed to evaluate the biological and socioeconomic responses to MPA-specific goals while accounting for the size and age of the MPA, fishing levels outside the MPA, differences in habitat types, and level of compliance and enforcement (Claudet and Guidetti 2010). A management rating system in the Philippines suggests that only about 20 percent of the 1,100 MPAs across the country are achieving their management goals (White, Alino, and Meneses 2006). Similarly, an analysis of marine reserves in the western Indian Ocean, Philippines, and Caribbean showed that success, measured in terms of greater fish biomass inside the marine reserves, was most strongly affected by compliance with reserve rules and human population density (Pollnac et al. 2010).

For MPAs or networks of MPAs that are not meeting their intended goals, it may become necessary to reevaluate the existing system and adapt the management plan accordingly. This is the concept of adaptive management, which promotes flexible decision making that can be adjusted in the face of uncertainties as outcomes from management actions and other events become better understood (Williams, Szaro, and Shapiro 2009). For example, after Australia’s Great Barrier Reef Marine Park Authority (GBRMPA) reviewed the park’s protected areas and management goals in the early 1990s, it identified a shortfall in the total area protected and the representation of habitats in the MPAs. The GBRMPA subsequently began an extensive process to consult with scientists, stakeholders, and the general public. The planning process employed both scientific and traditional knowledge to protect at least 20 percent of each bioregion in the park and to increase full protection in marine reserves from 4.2 percent to over 33 percent of the park’s total area (Fernandes et al. 2005). Clearly the need to repeatedly monitor and evaluate an MPA, and to proceed with adaptive management as necessary, is critical for ensuring that conservation and human use goals are met.

**MPAs and Broader Marine Management**

As successful MPAs meet their management goals, marine communities tend to become more robust, with greater numbers and sizes of organisms and more intact habitats than in fished areas outside. These healthier systems may be more resilient, allowing them to recover more quickly when they are affected by hurricanes, coral bleaching events, or other disturbances. It remains difficult, however, to demonstrate whether or not MPAs are more resilient than their unprotected counterparts. Moreover, while there is great potential for successful MPAs to provide significant and long-lasting ecological and social benefits, it is clear that MPAs alone cannot cure all that ails the ocean. Many stressors to marine systems, such as pollution and the effects of climate change, are simply too large to address effectively with MPAs.

As a result, MPAs are likely to be most useful when they are used as part of a larger, multisector management plan. For example, MPAs can be an integral component of ecosystem-based management (EBM). The goal of marine EBM is to sustain the long-term capacity of marine ecosystems to provide services that humans want and need (such as seafood production and protection from coastal storms) by balancing human health and ecosystem well-being and addressing the trade-offs that are required to account for multiple, often competing, goals (Halpern, Lester, and McLeod 2010). MPAs are a particularly effective component of EBM in areas where overfishing is the dominant human stressor, a situation found in many regions of the world including the Black Sea and the South China Sea. Healthy and resilient systems inside MPAs may provide a range of ecosystem services, including export of adults and juveniles to support productive fisheries outside the MPA or protection from erosion and storm damage provided by intact mangrove forests.

As marine resource management begins to move toward an EBM approach in many places around the world, MPAs can serve as a starting point, providing an existing framework from which to build up to larger scales and integrate other forms of management. For example, municipalities in southeastern Cebu Island in the Philippines have jurisdiction over their own waters to fifteen kilometers and have adopted local MPAs, which they enforce. By forming a coalition and convening a coastal resource management council, the community now considers MPAs collectively, outside fishing is closely regulated, tourism activities are controlled, and a social network of managers communicates across the region (Eisma-Osorio et al. 2009).

While MPAs are not a panacea, they are a powerful tool for protecting and conserving marine resources. When implemented scientifically and collaboratively, and when they are enforced and financed over the long term, MPAs can be a crucial part of an ecosystem-based approach to management.

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*See also* Adaptive Resource Management (ARM); Administrative Law; Best Management Practices (BMP); Coastal Management; Community Ecology; Fish Hatcheries; Fisheries Management; Hunting;
Large Marine Ecosystem (LME) Management and Assessment; Ocean Acidification—Management; Ocean Resource Management

FURTHER READING


Microbial communities can regulate many processes in ecosystems, including decomposition, nutrient cycling, and the degradation of toxic chemicals. These processes all have direct effects on soil and water quality. Microbial processes can be used to promote sustainability, but they also can be sensitive to changes in ecosystems that are caused by human activities. Understanding microbial communities is therefore crucial in managing our world for a sustainable future.

Soil

Soil and water contain communities of microbes (microscopic organisms like bacteria and fungi) that make up the majority of the Earth’s living matter and account for most of its diversity. These communities are responsible for many important environmental processes, including decomposition of plant and animal material, cycling of nutrients such as nitrogen and phosphorus, and the breakdown of toxic chemicals. Microbial processes contribute significantly to both soil quality (as measured by the structure of the soil and the amount of organic matter and carbon present, including nitrogen, phosphorus, and potassium) and water quality (as measured by the amount of nutrients, pollutants, or excess sediments present). Consequently, microbial communities play an important role in how human activities such as growing food, cleaning up oil spills, and addressing climate change will impact natural ecosystems. Understanding microbial processes and using them effectively can greatly improve our ability to manage the Earth’s ecosystems in a sustainable fashion.

Microorganism Habitats

Human management of ecosystems directly affects the habitats of microbes for better or for worse. Microorganisms are amazing in that they are adapted to live and function in any habitat on Earth where water, the basis of life, is in liquid form. This includes frozen habitats, where microbes produce sugars to decrease the temperature at which ice can be formed, and deep-sea thermal vents, where extreme pressure allows water to stay in liquid form at temperatures up to 407°C. An important characteristic of microbial habitats is that they contain both oxic (containing oxygen) and anoxic (containing either low or nearly no oxygen) zones, and a distinct set of microbial processes (termed aerobic and anaerobic, respectively) occurs in each. Soil and freshwater habitats are described below, as predominant systems of interest to sustainability and as examples of microbial habitats.

Soil

Soil is a living, complex system that provides physical space for microbes to live. The degree to which a soil has structure and how its particles clump together determine how well it is able to withstand erosion. The proportion of mineral particles ranging from very small clay particles (less than 0.002 mm) to large sand particles (0.05–2.0 mm), and organic matter in soil determines what the structure is like. Organic matter is a general term for decomposed plants, animals, or microbes; these materials are mainly carbon based, so the carbon can be measured and used to describe how much organic matter a particular soil has. The roots of living plants and filamentous microbes (fungi and filamentous bacteria) also contribute to soil structure by binding soil particles together and strengthening it (bacteria do this by producing complex sugars that they use to “stick” themselves to soil particles). Microbes also contribute significantly to soil formation by carrying out decomposition.

Certain other characteristics of soil (including pH, moisture, and the amount of oxygen present) will affect which microbial processes are most active. These characteristics are determined by climate, position in the
landscape (for example, hilltops versus low-lying areas), and local plant communities.

Soil quality (or soil health) is a term often used to describe how degraded an ecosystem is or whether management practices are sustainable. The amount of organic matter in soil, the strength of its structure, and how well it sticks together and retains nutrients determine soil quality. Building a healthy soil creates a positive feedback loop: a soil with lots of organic matter that holds nutrients supports the growth of microbial and plant communities, which create more organic matter and conserve more nutrients and minerals. On the other hand, it’s very easy to degrade soil structure (and therefore soil health) through human activities such as agricultural tillage, mining, urban development, and industrial activity.

Water

Microorganisms also are adapted to live in freshwater aquatic and wetland habitats. In lakes, photosynthetic organisms (those that convert carbon dioxide into organic compounds like sugars) are housed mainly in upper layers of the water where light can penetrate. Organisms that depend on organic compounds falling from above live in the deeper layers of the water and in bottom sediments, which can be a nutrient-rich habitat for microbial growth. Life in most lakes is adapted to relatively low nutrient levels, and fertilizer runoff (especially nitrogen and phosphorus) from urban areas or agricultural activities can endanger many forms of life there.

Water in rivers moves at different rates, and most microorganisms either grow attached to rocks or to subsurface sediments. In contrast, wetlands are terrestrial habitats with transient or permanent standing water and plant life that emerges from the water’s surface. In wetland habitats, microbes near plant roots in the uppermost layers of water or sediment can consume or transform nutrients rapidly, but these processes also deplete oxygen quickly, so anaerobic processes dominate lower layers. Wetlands are useful in filtering out chemicals or pollutants from urban areas because of this rapid nutrient uptake by microbes and plants, and because wetlands are usually situated in basins or low-lying areas—but at the same time they are also fragile habitats that can be degraded easily by nutrient or sediment runoff.

Microorganism Processes in Ecosystems

Just as microorganisms have adapted to life in diverse habitats, different organisms have adapted so that they can use nearly any material or chemical—from leaf litter (dead plant material) to oil to pesticides—for energy and growth. Microorganisms transform or degrade compounds in the environment for the same reasons that we eat: for energy and to obtain building blocks for growth. Microbes get energy from transforming different compounds based on oxidation-reduction reactions (described in the next section). Large organic compounds are broken down into inorganic building blocks (such as nitrate, ammonia, or phosphate) through mineralization.

Oxidation-Reduction Reactions

Oxidation is a process by which the outer electrons from a molecule are stripped away, changing its chemical state. Those electrons are passed to a donor electron acceptor molecule, which becomes reduced. Oxidation-reduction reactions yield energy for the microorganism but can also drastically impact whether nutrients or elements stay in the soil or leave, potentially polluting the surrounding environment.

Respiration

Respiration is the use or chemical transformation of any compound for energy production. Aerobic respiration (the use of oxygen as an electron acceptor, resulting in the production of carbon dioxide) is a very efficient process. Where oxygen is available, therefore, microorganisms capable of aerobic respiration will grow quickly and dominate the microbial community. Other chemical transformations used to produce energy, such as nitrification (the conversion of ammonia to nitrate through ammonia oxidation) or methanogenesis (the production of methane gas from carbon dioxide reduction or through acetate fermentation), are much less efficient. Organisms relying on these processes for the energy they need grow more slowly but process large amounts of the compounds used.

 Decomposition

Decomposition is the process by which large materials such as dead plants, animals, manure, or microbes are broken down into smaller subunits. Decomposition is nature’s way of recycling the nutrients found in living matter back into the soil. Decomposition also can function to store carbon as organic matter or, where the rate of aerobic respiration is high, to deplete supplies of organic matter. Microorganisms create enzymes because large organic molecules, such as cellulose, are too big to be absorbed through the microbial cellular membrane, and many types of microorganisms must produce a variety of enzymes to decompose different organic materials. Some cellular components—such as proteins, cellulose, and starches—are degraded quickly and therefore do not have a long turnover time in the soil before they are degraded and reabsorbed into newly growing microbial cells. Other components such as lignin are not degraded quickly and
Nitrogen Cycling

Microorganisms are responsible for many transformations of nitrogen that are important to soil and water quality. Nitrogen fixation, mineralization, immobilization, nitrification, and denitrification are all transformations of nitrogen that are important to sustainability, restoration, and soil quality. Nitrogen fixation is performed either by free-living bacteria or by bacteria living symbiotically with legumes (a family that includes peas, beans, and clovers). The legumes are able to acquire nitrogen because of their bacterial partner and can improve soil quality or help restore disturbed or low-nutrient soils (such as those on former mining sites). Nitrogen mineralization is the process where organic compounds are broken down, releasing inorganic forms that can be used by plants and microbes. Nitrogen immobilization is the process by which any soil organism takes up nitrogen so that it’s no longer available for other organisms. High rates of immobilization can be positive in that less excess nitrate nitrogen (nitrogen combined in a nitrate ion, as opposed to nitrogen in the form of ammonia, nitrites, etc.) leaks into lakes, rivers, drinking water, or wetlands—but it also can lead to problems with nitrogen limitations for agricultural crops. Nitrification is a process linked to pollution that converts nitrogen from a form conserved in the soil (ammonia) to a form that leaches readily from the soil (nitrate) through the energy yielding processes of ammonia and then nitrite oxidation. Often, ammonia fertilizer applied to crops is quickly converted to nitrate. Energy is lost so that crops do not receive the fertilizer they need to grow, and groundwater, lakes, and rivers are polluted by excess nitrogen. Conversely, denitrification is a form of anaerobic respiration in which nitrate is reduced into nitrogen gas for energy production. This process also produces nitrous oxide or other nitrogen gases that are considered potent greenhouse gases, but it also can act to remove excess nitrate that otherwise would pollute wetlands and groundwater.

Cycling of Other Elements

Microorganisms act as a middleman in the transformation of many other elements besides carbon and nitrogen. In fact, they can transform nearly any molecule on Earth, and the results can be good or bad for the environment. Acid mine drainage is an example of how the conversion of elements can lead to pollution: when mining or other human activities disrupt mineral rock, microbes transform sulfur (through aerobic respiration) into forms that are easily released, and water draining from the mine becomes acidic from too much sulfur. Microbes can release environmental toxins such as arsenic and selenium from mineral rock in a similar sulfur. Alternatively, microbes contribute to the nutrient levels of soil through mineralization, which transforms sulfur, phosphorus, iron, and other elements from forms that are bound to soil, or breaks down organic molecules into simpler mineral forms that can be taken up by other plants or microbes.

Degradation and Uptake of Toxic Chemicals

Microbes can use toxic chemicals to grow or get energy (through oxidation-reduction reactions) in the same way they use natural elements. These toxins include petrochemicals (such as oil or gasoline), pesticides, metals, and by-products of industrial activity. Sometimes microbial degradation or uptake of these chemicals can remove the contaminant, as with the slow but steady anaerobic degradation of polychlorinated biphenyls (PCBs) that contaminate many soils and aquatic sediments. Microbes can also transform or degrade a toxin only partially, however, resulting in something that is even more toxic or dangerous.

Microorganism Communities and Sustainability

Sustainable management or development has been defined as “the use of land and water to sustain production indefinitely without environmental deterioration” (Lincoln, Boxshall, and Clark 1998). We are managing our environment, for better or for worse, when we decide how to use it. Management can mean policies that are active and promote sustainability (as with agricultural land-management practices and actions that restore degraded landscapes), or it can mean indirect “management” that leads to environmental degradation through global change.

Agricultural Management

Land management practices in agriculture affect microbe communities in a variety of ways. Conventional methods of tilling soil break up the top 20 centimeters of the soil surface and, in the process, severely damage the filamentous cell networks of fungi. Tillage also can lead to organic carbon losses in soil, since it encourages bacterial aerobic respiration by introducing oxygen. These carbon losses can weaken soil structure, leading to erosion and poor-quality soil that doesn’t retain nutrients well, resulting in runoff, pollution, and the need to add more chemical fertilizers. Spreading fields with lime changes the pH of soil, affecting which microorganisms are present and
how they grow. Using nitrogen fertilizers decreases fungi relative to bacteria, thus altering the normal ratio of processes carried out by each. Nitrogen fertilizer (which is often in the form of ammonium) is then more likely to pollute groundwater, lakes, rivers, and wetlands, as now-abundant bacteria convert it quickly to nitrate. The effects of conventional agricultural cultivation may persist well after it has ended, altering fungal communities, microbial processes, the amount of soil organic carbon, and pH for as much as seventy years.

But agricultural management can become more sustainable. Reducing or eliminating tillage can increase fungal communities, resulting in better soil structure and fungal decomposition processes. Using cover crops (crops that are grown primarily to manage soil quality, weeds, diseases, pests, and water drainage) can prevent soil erosion and add nutrients back to the soil. Operations that grow food can be paired with those that raise animals for milk and food, so that animal waste can be used to add nutrients and organic matter back into the soil. These natural nutrient additions decompose more slowly and thus support long-term improvements in soil quality, unlike mineral fertilizers or repeated tilling. Organic amendments are like an investment bank account that matures over thirty years, while chemical fertilizers are more like a checking account that is drained regularly and needs to be replenished often.

Bioremediation

Using microorganisms to treat waste or pollution is called bioremediation, and understanding how microbial processes work is crucial to its success. For instance, certain organisms can be encouraged to grow by providing more or less oxygen, by adding nutrients, or by simply adding the desired organism to a contaminated soil. Sometimes (as with PCBs) just leaving contaminated areas alone and giving organisms time to go about their cleanup work is necessary. Other situations call for balancing different kinds of microbe communities, as when sulfate-oxidizing and sulfate-reducing organisms are paired to remove heavy-metal contamination from soils (a much more effective and cost-efficient process than has been used traditionally). Bioremediation processes don’t work in all conditions, but they have huge potential for cleaning up polluted soils.

Human activities can change landscapes greatly even when toxic chemicals aren’t involved, and understanding microbial processes can help to rebuild landscapes damaged by mining and other urban activities that lead to a loss of vegetation or excess sediments in wetlands, lakes, or rivers. The ideal path toward better sustainable management, of course, is to be aware of how human activities affect natural ecosystems in the first place and to try to limit the damage that they do.

Impact of Global Changes

Climate change and other global changes could drastically alter which plants can grow in which regions, meaning that agricultural practices will have to be altered and decisions about sustainable land-management policies will become a lot more complex. In addition, microbial communities are probably adapted to local weather in different ways, making it difficult to come up with a one-size-fits-all response to global changes in climate. Climate change also could cause microorganisms to add to the greenhouse gas problem (greenhouse gases are those that absorb radiation, trapping heat in our atmosphere), since their respiration converts stored soil carbon into carbon dioxide, a major greenhouse gas. In places where the climate becomes warmer but stays wet or gets wetter, denitrification and methanogenesis will create more nitrous oxide and methane, respectively, and these are also greenhouse gases. While nitrous oxide and methane are not as abundant as carbon dioxide in the atmosphere, they are much more potent in capturing heat and therefore are important for future changes in our climate.

Another major global change factor is nitrogen deposits from industrial activities. These by-products may act like fertilizers and increase microbial growth and respiration, acidify soil, or pollute rivers and lakes.
The Future

Understanding the many processes mediated by microorganisms can help us use them to make sustainable management decisions. In agriculture, low or no tillage systems with organic inputs to match the nutrients taken away through plant harvesting can improve decomposition processes and microbial contribution to soil structure and stability. Adding nitrogen-fixing microbes or growing legumes (plants with a nitrogen-fixing bacterial partner) to disturbed soil can help to restore it. Microbes can also be used to degrade or remove toxic chemicals that result from industrial waste, mining, and other human activities. Global changes may make microbial communities less able to carry out their usual processes depending on how they adapt. In the future these diverse communities should play a central role in how we think about sustainability and management.

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See also Agricultural Intensification; Agroecology; Biodiversity; Brownfield Redevelopment; Ecosystem Services; Eutrophication; Global Climate Change; Mutualism; Nitrogen Saturation; Nutrient and Biogeochemical Cycling; Plant-Animal Interactions; Soil Conservation; Waste Management

FURTHER READING


Mutualism is an interaction between species that benefits both. Such interactions are critical to reproduction and survival and thus to the continued provision of ecosystem services to human populations. Global environmental change threatens mutualisms by altering the timing of natural history events, shifting species’ ranges, reducing and fragmenting habitat, and promoting invasion by non-native organisms. More research is needed to accurately predict ecological response to these changes.

Interactions between species influence ecological processes within populations, communities, and ecosystems. Virtually all species on Earth are involved in multiple interspecific interactions at any one time. The best-studied species interactions are competition and predation, relationships that are harmful to one or both of the involved species. Mutualisms, in contrast, are interactions between two species that benefit both. Mutualisms occur in habitats throughout the world (Bronstein 2009) and are crucial to the reproduction and survival of many organisms, as well as to nutrient cycles in ecosystems. Moreover, the services that mutualists provide are increasingly leading environmentalists to consider mutualisms a priority in conservation and restoration.

Ecosystems provide goods and services that support human populations (e.g., food, clean water, energy, protection, and cultural enrichment), without which world economies would “grind to a halt” (Costanza et al. 1997). Some researchers define ecosystem functions as processes that facilitate biogeographical flow of energy among ecosystems, while ecosystem services are those processes that are beneficial to humans (e.g., Traill et al. 2010). Mutualism plays an important role in ecosystem services via pollination, seed dispersal, nutrient cycling, and biological control. It is likely that every organism participates directly or indirectly in at least one mutualistic interaction (Bronstein 2009). Since mutualism is involved in virtually every ecosystem service, understanding it and preserving it are of high priority.

Pollination is a classic example of mutualism. Animals pollinate over 87 percent of all flowering plants worldwide (Ollerton, Winfree, and Tarrant 2011). Pollination is essential for agriculture: animal-pollinated crops account for about 35 percent of global food production (Klein et al. 2007). Many plants also depend on animals, including birds and mammals, for seed dispersal. These animals feed on fruits, in the process moving seeds away from parent plants to other habitats, and hence are critical to the persistence of natural and managed vegetation. In another common mutualism, certain plants and insects (such as aphids) use ants for “biological warfare” against their enemies: ants are attracted by food rewards and then aggressively defend their partner against attack. Such interactions can be essential for allowing species to persist, which in turn benefits humans when these are species of economic or aesthetic interest. Conversely, these mutualisms are to our detriment when they involve species (e.g., agricultural pests) that we would prefer to extirpate. Finally, mutualisms between plants and certain microbes promote healthy nutrient and biogeochemical cycling. Mycorrhizae fungi are intimately (symbiotically) associated with plant roots from which plants gain essential nutrients. Approximately 80 percent of terrestrial plants participate in this type of mutualism (van der Heijden 1998). Mycorrhizal fungal diversity is important for the maintenance of plant biodiversity, structure, nutrient capture, and productivity. Similarly, rhizobia are symbiotic nitrogen-fixing bacteria that associate with legumes (including soybeans and peas). These
bacteria are responsible for most of the nitrogen that is fixed biologically, as well as for more than a quarter of crop production; thus, they are essential for ecosystem functioning and independence from nitrogen fertilizers. Indeed, mutualistic associations involving microbes are pervasive in nature and are critical to the maintenance of biogeochemical processes, including the nitrogen cycle.

Mutualism is crucial in sustaining coral reefs. Most coral species are dependent on mutualistic associations with specialized symbiotic algae. Environmental stresses, including rising temperatures, cause the expulsion of these algae in a phenomenon known as coral bleaching. While corals may completely recover after mild events, more extreme events can cause 100 percent mortality. Coral reefs provide the structure for highly productive, diverse coastal marine ecosystems that benefit humans by acting as nurseries and habitat for commercial fisheries, as well as recreational habitat. Their destruction through bleaching is expected to become more of a problem for the majority of the world’s coral reefs as Earth’s climate continues to warm (Donner et al. 2005).

The 2005 Millennium Ecosystem Assessment calculated that human activities have decreased ecosystem services by over 60 percent. Global environmental change negatively affects mutualism in several ways. It alters phenologies (timing of natural history events such as flowering), shifts species’ distribution ranges, and reduces or fragments habitat—changes that make it likely that organisms will not be able to find each other and thus to establish the mutualistic associations they require. Furthermore, global change often favors the invasion of species that monopolize or kill mutualists, to their native partners’ detriment. Consequences are particularly well documented in seed dispersal and pollination systems. Bee diversity in Britain and the Netherlands has declined since 1980, with a corresponding decline in the plant species reliant on those pollinators (Biesmeijer et al. 2006). Human land use and disturbance are significant threats to wild, native pollinators (Winfree et al. 2009), and thus to plant communities worldwide. Biologists have reported a few cases of phenological mismatches among plants and their pollinators as a consequence of climate warming (Hegland et al. 2009), although there are also cases known in which partners shift their own phenology to the same degree, leaving their associations intact.

Biologists have recently urged that species interactions need to be taken into account if we wish to accurately predict ecological response to global change (Kiers et al. 2010). Little is yet known about the effects of global change on biotic interactions, especially mutualism. Plant-pollinator systems have received more attention than any other mutualistic interaction; close attention to the size of intact habitat and ecosystem-specific thresholds has been shown to be essential for maintenance of wild pollinators and pollinator-dependent plants in agro-ecosystems (Keitt 2009). Continued accumulation and analysis of data for other biotic interactions will lead to understanding of what is needed to protect and sustain other ecosystems as well.

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See also Agroecology; Biodiversity; Charismatic Megafauna; Community Ecology; Disturbance; Global Climate Change; Food Webs; Habitat Fragmentation; Indicator Species; Keystone Species; Invasive Species; Microbial Ecosystem Processes; Nutrient and Biogeochemical Cycling; Outbreak Species; Plant-Animal Interactions; Resilience

FURTHER READING


Natural capital is an economic construct that describes the natural world, its ecosystems, and their value to society. How people value the natural world determines how businesses and societies both conserve and deplete it. Economists who think about natural capital as an irreplaceable resource and those who believe that it is like any other input into an economy have very different ideas about how society should treat the natural world.

Economists use the concept of natural capital to explain the contribution that the natural world's resources make to human economies. Different schools of economic thought have a number of different ways to approach the topic, and these approaches have different consequences for sustainable development.

Conceptual History

In the eighteenth and nineteenth centuries, economists identified the factors of production (that is, the resources that go into producing goods and services), as capital, labor, and land. Capital was defined as an input that is not consumed in the manufacture of a product (Smith 1776) or, alternatively, as something human made that contributed to production (Böhm-Bawerk 1891), for example, machinery. Land, which included all natural resources, was treated as distinct from capital because it was a gift of nature and because humans could not affect its supply. In the twentieth century, economists redefined capital as any asset that produces a stream of income over time (Fisher 1906). By this definition, land is lumped together with other capital, and the factors of production are reduced to two: capital and labor. After the redefinition, natural resources were increasingly ignored as a factor of production to the point where an eminent economist could suggest that “the world can, in effect, get along without natural resources” (Solow 1974). In the 1970s, however, growing evidence of the limitations on natural resources and environmental problems made worse by accelerating economic growth led many economists to call for explicitly recognizing natural capital—defined as a stock that yields a flow of natural services and tangible natural resources—as a distinct and essential factor of production.

The first explicit reference to natural capital appears in Small Is Beautiful (Schumacher 1973). In this book, the British economist E. F. Schumacher argued that irreplaceable natural capital stocks make up the larger part of all capital, and that modern economists erroneously treat their depletion as income. Schumacher identified two types of natural capital. The first was fossil fuels, which were rapidly being exhausted. The second was the ability of natural systems to regenerate themselves, along with other services provided by ecosystems, as a distinct and essential factor of production. Furthermore, both Daly and Georgescu-Roegen carefully distinguished between the two types of natural capital discussed by Schumacher. Fossil fuels, along with all other raw materials from nature (both renewable and nonrenewable), are identified as stock-flow resources, which are consumed and therefore depleted in the act of production. Humans can decide how quickly to deplete such resources. In contrast, the ability of ecosystems to reproduce themselves, along with other services provided by ecosystems, is a
fund-service resource. A fund is not consumed in the act of producing a service. For example, when a forest helps regulate water flow, processes waste, provides shelter for other species, or produces the seeds required for renewal, it is not consumed in the process. Nature’s fund services are generated from a particular configuration of its stock-flow components. Just as a car factory is a particular configuration of metal, glass, and concrete, a forest is a particular configuration of plants, animals, water, and soil. Funds provide services at a given rate over time. The term natural capital generally refers both to stock flows and to fund services.

The concept of natural capital caught on fairly quickly, particularly in the field of ecological economics, whose theoretical foundations stressed the dependence of the economic system on the planet’s finite supply of natural resources and the invaluable services they generate. The economists David Pearce (1988) and Herman Daly (1990) argued that sustainable development required a constant natural capital stock. Daly listed specific rules for maintaining a constant stock: extraction of renewable resources could not exceed regeneration rates, extraction of nonrenewable resources could not exceed the rate at which renewable substitutes were developed, and waste emissions could not exceed the ecosystem’s absorption capacity. The concept of natural capital suited ecological economic theory so well that the proceedings of the second international Ecological Economics Conference were published as the book Investing in Natural Capital (Jansson et al. 1994). The first section of this book focuses on maintaining and investing in natural capital, the second focuses on methods and research topics, and the third on policy implications and applications. These three topics parallel and anticipate the way researchers later developed the ideas around natural capital.

Much of the early research on natural capital focused on the economic value of ecosystem services. In May 1997, the journal Nature published a paper that integrated these studies into a single global assessment of natural capital. That paper, “The Value of the World’s Ecosystem Services and Natural Capital” (Costanza et al. 1997), has become one of the most cited in the environmental sciences.

At the same time, various nations attempted to integrate natural capital into their national accounts (Ahmad, El Serafy, and Lutz 1989), leading the United Nations to propose a System of Environmental and Economic Accounts (SEEA) (United Nations 1993), eventually implemented in 2003. The researchers William Rees (from Canada) and Mathis Wackernagel (from Switzerland) introduced the concept of the ecological footprint as a biophysical measure of humanity’s demands on natural capital (Rees and Wackernagel 1994). A growing number of countries, regions, and businesses around the world adopted the ecological footprint as a measure of sustainability.

The book Natural Capitalism (Hawken, Lovins, and Lovins 1999) and the nonprofit organization The Natural Step both played a critical role in popularizing the concept of natural capital outside of academia, particularly in the business world, by laying roadmaps for a society that obeys Herman Daly’s specific rules for sustaining natural capital. The triple bottom line is one increasingly popular approach to business that accounts for natural capital, human capital, and financial capital (Elkington 1997). National and international policies designed to protect and restore natural capital including cap and trade regulations for pollutants and fisheries and payments for ecosystem services are now multibillion-dollar global markets (Farley et al. 2010).

Strong and Weak Sustainability

Natural capital, however, remains poorly defined and subject to controversy. One ongoing debate is whether or not natural capital is, in fact, irreplaceable. If it is true that some natural capital is essential and has no substitutes, then that capital must be preserved and so cannot be lumped together with other forms of capital. This belief, generally shared by ecological economists and known as strong sustainability, led to the emergence of the concept of natural capital in the early 1970s. Other economists argue that human-made capital can substitute for natural capital and that as long as the value of both types of capital together is not declining, sustainability is achieved. According to this model, clear-cutting the Amazon could be viewed as sustainable as long as future generations were left with an equal value of roads and buildings. This approach is called weak sustainability. David Pearce was instrumental in developing the concept of weak sustainability (Pearce and Atkinson 1993), although he had initially stressed the irreplaceable nature of natural capital. Many scholars now use the phrase critical natural capital for natural capital that is essential to human welfare and has no substitutes (Ekins et al. 2003).

A related debate concerns the labels natural capital and ecosystem services, which, some argue, imply the treatment of nature as a commodity, or market good, and hence weak sustainability. Many economists do believe that natural capital should be treated as just another commodity and incorporated into the market model. Others, however, interpret natural capital as a metaphor that calls attention to the productive capacity of ecosystems and the need to invest in their protection and restoration. If natural capital is defined as an asset that produces a flow of income over time, then the term implies that we must
live only off the flow of income without depleting the capital stock. According to this way of thinking, the metaphor does not imply that natural capital can be bought and sold like any other asset.

Valuing ecosystem services goes a step further than the natural capital metaphor to suggest that such services have a monetary exchange value and are neither essential nor nonsubstitutable. Advocates of weak sustainability typically believe that markets will lead to the optimal provision of ecosystem services if the services are correctly priced. Even many advocates of strong sustainability argue that valuation calls attention to the importance of natural capital and that failure to value ecosystem services assigns an implicit value of zero to them. They point out that food is also essential and nonsubstitutable, yet it is nonetheless valued in monetary terms.

Critics of monetary valuation argue that it is based on preferences weighted by purchasing power, which gives no voice to the poor and prioritizes Western over indigenous values. Furthermore, preferences are often based on incomplete or inaccurate knowledge, because people rarely understand precisely how ecosystems generate services or how human activities will affect them. The dominant critique is that valuation implies weak sustainability. In fact, many conventional economists focused on dollar values have explicitly stated that global climate change is relatively unimportant because it primarily affects agriculture, which constitutes a negligible share of gross domestic product, or GDP (Schelling 2007). Most ecological economists argue that society should impose the specific quantitative rules for sustainable use of natural capital that Herman Daly suggested, and then let prices adjust to these ecological constraints (Farley 2008). They also suggest that because natural capital is part of a shared inheritance, scientific and democratic principles, rather than the market, should be used to value it.

Another controversial application of the concept of natural capital is payments for ecosystem services (PES). Those who favor the integration of natural capital into the market tend to favor PES systems in which private sector beneficiaries of ecosystem services pay landowners for land uses that provide specific services. Such payments are often for a single service and do not take into account other services provided by the system. Critics of PES typically argue that ecosystem services are public goods and cannot be forced into the market model. Protecting and restoring natural capital does, however, impose real costs on society, which somebody must pay. Many ecological economists believe that investments in natural capital should be a cooperative endeavor, in which the wealthiest nations and regions shoulder the financial cost of restoration and conservation wherever it is needed (Farley and Costanza 2010).

**Outlook for the Future**

The importance of the natural capital concept continues to grow, as measured by a steadily increasing use of the term in the scientific literature. Human society must recognize that we, like all other species, depend on the flow of goods and services provided by nature. In the past, humans have treated natural capital as if there were enough to meet all human and ecological needs for all time, with no trade-offs involved and hence no need to ration access. The market system is very effective at allocating natural capital toward market products, but it fails to take account of natural capital’s growing scarcity. As a result, societies are now depleting natural capital faster than it can regenerate and returning waste to the environment faster than it can be absorbed. Depletion of this natural capital will diminish not only nature’s capacity to regenerate itself but also the raw materials needed for all economic production and the flow of ecosystem services essential to human wellbeing. Future generations are left dependent on dwindling resources. The concept of strong sustainability makes it obvious that people must learn to live on the interest from natural capital, the annual flow of benefits, without depleting the stock.

Natural capital is inherently different from other forms of capital—produced capital and natural capital are ultimately complements, not substitutes for one another. It is not enough to simply put a price on natural capital and force it into competitive market boundaries. Instead, economic institutions must adapt to the fact that natural capital is irreplaceable and generates a flow of public good services best protected by cooperative efforts. Society faces a new allocation challenge: how much natural capital should be converted to economic production, and how much should be conserved for the provision of ecosystem...
services? Both uses of natural capital are essential and have no substitutes. The concept of natural capital, correctly applied, can help society make these choices.

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See also Agricultural Intensification; Agroecology; Ecosystem Services; Human Ecology; Irrigation; Landscape Planning, Large-Scale; Marine Protected Areas (MPAs); Ocean Resource Management; Permaculture; Reforestation; Soil Conservation; Viewshed Protection; Water Resource Management, Integrated (IWRM); Wilderness Areas

FURTHER READING

Farley, Joshua; Aquino, André; Daniels, Amy; Moulaert, Azur; Lee, Dan; & Krause, Abby. (2010). Global mechanisms for sustaining and enhancing PES schemes. Ecological Economics, 69(11), 2075–2084.
Nitrogen saturation occurs when the supply of reactive nitrogen (oxidized, reduced, or organic nitrogen) to an ecosystem exceeds biotic and abiotic demand. Reactive nitrogen has historically been a scarce nutrient, but recent industrial and agricultural activities have created an oversupply of nitrogen in many ecosystems. This shift from nitrogen scarcity to overabundance has a wide range of ecological consequences for terrestrial, aquatic, and marine ecosystems.

Nitrogen (N) saturation occurs when the supply of reactive nitrogen exceeds the ability of an ecosystem to absorb it through biotic and abiotic processes. Reactive nitrogen includes reduced forms such as NH\textsubscript{4}\textsuperscript{+} (ammonium), oxidized forms such as NO\textsubscript{3}\textsuperscript{-} (nitrate), and organic forms such as amino acids. Historically, plant growth in most terrestrial ecosystems has been limited by the availability of these forms of nitrogen (Galloway et al. 2004). Although nitrogen is plentiful in the atmosphere, nearly all of it is in an unreactive form (N\textsubscript{2}) that most plants and other organisms cannot use for growth (Galloway et al. 2004). Over time, human activities such as fertilizer production, cultivation of crops that can use N\textsubscript{2}, and fossil-fuel combustion have dramatically increased the global supply of reactive nitrogen (Galloway et al. 2004). Much of it is emitted into air and water, increasing the supply of reactive nitrogen to surrounding ecosystems to the point that it has become excessive (Vitousek et al. 1997).

For many ecosystems, this switch from nitrogen limitation to nitrogen saturation has ecological consequences.

From Atmosphere to Ocean

Nitrogen saturation was originally defined in terrestrial ecosystems (Aber et al. 1989), but the concept has also been applied to watersheds (Stoddard 1994) and aquatic ecosystems (Mulholland et al. 2008). Reactive nitrogen enters the atmosphere as a result of natural processes such as lightning and biotic fixation, but natural sources are now far outweighed by emissions from human activities, particularly in densely populated and heavily industrialized areas such as Europe, North America, and east Asia (Galloway et al. 2004). The primary sources of reactive nitrogen emitted into the atmosphere are fossil-fuel combustion from transportation and electricity generation (29 percent) and agriculture (52 percent). Fertilizer use (9 percent) and animal husbandry (22 percent) comprise the dominant agricultural sources (Galloway et al. 2004). Most reactive nitrogen stays in the atmosphere only for hours or days before it is deposited onto ecosystems downwind from emissions sources through precipitation, dust and fine particles, or chemical reactions that occur on surfaces, such as foliage (Holland et al. 2005).

Initially, reactive nitrogen deposited onto nitrogen-limited terrestrial ecosystems increases plant growth and ecosystem carbon storage, and the nitrogen is retained within the ecosystem (Vitousek et al. 1997). As nitrogen accumulates, however, and biotic demand for nitrogen is met, soil nitrogen is increasingly converted to nitrate (NO\textsubscript{3}), which leaches easily from soils and into the groundwater or aquatic ecosystems. Nitrate leaching is a key indicator of nitrogen saturation and implies a transition away from a nitrogen-limited system (Aber et al. 1989). Nitrate leaching causes soil acidification; the loss of plant nutrients, such as calcium and magnesium, from soils; and increased exposure of plants to soil aluminum, which can be toxic. Because of these effects, nitrogen saturation may reduce plant growth and increase plant mortality in acidic, nutrient-poor soils (Aber et al. 1989).

In terms of ecosystem carbon storage, there is evidence that negative effects on plant growth may be counteracted by reduced microbial decomposition of organic matter (Janssens et al. 2010). Nevertheless, excess soil nitrogen can reduce...
plant species diversity (Emmett 2007), lower the abundance of symbiotic mycorrhizal fungi associated with plant roots (Treseder 2004), and increase emissions of nitrous oxide ($N_2O$)—a powerful greenhouse gas (Aber et al. 1989).

Freshwater ecosystems are conduits for nitrogen to leach from terrestrial ecosystems into the oceans. Denitrification by microorganisms converts some of this nitrate to $N_2$ that is released back into the atmosphere. This process is also vulnerable to nitrogen saturation, because it becomes less efficient as nitrogen concentrations increase (Mulholland et al. 2008). In addition, nitrate leaching can damage aquatic ecosystems by contributing to both surface water acidification and eutrophication (Rabalais 2002).

Acidification of freshwater ecosystems reduces the growth and survival of fish and other biota. Although nitrate leaching is a minor contributor to surface water acidification in most of North America, it is a large contributor in Europe (Stoddard 1994). Eutrophication occurs when nutrient availability within an ecosystem greatly increases; in freshwater ecosystems, eutrophication can lead to noxious algal blooms, loss of aquatic vegetation, and oxygen deficiency (Rabalais 2002). Nitrogen inputs alone do not always result in freshwater eutrophication, because productivity in these systems is more strongly limited by phosphorus availability. Nitrogen inputs can, however, cause eutrophication in ecosystems that are phosphorus-rich from natural sources or pollution (Rabalais 2002).

In contrast, nitrogen can have major negative consequences when added to coastal ecosystems and estuaries (Vitousek et al. 1997). Unlike the phosphorus-limited open ocean, near-shore ecosystems are limited in productivity by nitrogen availability, because historical rates of nitrogen input have been small, the abundance of organisms that can use $N_2$ for growth is low, and rates of nitrogen removal through denitrification are high. Consequently, eutrophication caused by nitrogen can lead to loss of biotic diversity, harmful algae blooms, oxygen deficiency, and extensive die-offs of economically important fish and shellfish (Vitousek et al. 1997).

Each of these ecological systems is vulnerable to excess nitrogen availability, but susceptibility varies locally according to a wide range of abiotic and biotic factors that control the supply of and demand for nitrogen (Aber et al. 1989). Demand for nitrogen is reduced by factors that restrict biological growth, such as low temperatures, drought, the availability of other nutrients (chiefly phosphorus), the age of the dominant organisms, competition, and disease. Variation among species in their ability to take up and use available nitrogen also influences nitrogen demand. Past disturbances—both natural ones such as fires, and human ones such as timber harvesting and land conversion for agriculture—can change background nitrogen availability and make ecosystems more or less susceptible to chronic nitrogen additions (Aber et al. 1989).

In coming decades, atmospheric deposition of reactive nitrogen in North America and western Europe is predicted to remain relatively constant as a result of pollution controls, while it is predicted to increase in every other populated region of the world (Galloway et al. 2004). Reactive nitrogen is also predicted to become more abundant globally in streams and coastal ecosystems (Galloway et al. 2004). Although nitrogen inputs may not increase in North America and western Europe, the legacy of previous nitrogen inputs means that nitrogen saturation is likely to become more widespread. Lowering the production of reactive nitrogen could be achieved over the long term through reducing fossil-fuel combustion and increasing use of other energy sources, changing animal husbandry practices to improve management of animal waste, and reducing rates of fertilizer application in agriculture. If nitrogen additions can be limited to the point where they no longer exceed biotic demand, ecosystems will gradually return to being nitrogen-limited.

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See also Agricultural Intensification; Agroecology; Ecosystem Services; Eutrophication; Groundwater Management; Nutrient and Biogeochemical Cycling; Ocean Acidification—Management; Pollution, Nonpoint Source; Pollution, Point Source; Soil Conservation

FURTHER READING


Understanding nutrient and biogeochemical cycling (the movement of elements through ecosystem components) can provide relatively unambiguous and simple criteria for evaluating one aspect of sustainability. Based on a “mass balance” approach (which looks at the balance of ecosystem inputs and outputs), descriptions of elemental cycles over human time frames can determine whether management practices will result in losses in nutrient stocks (the total ecosystem inventory of nutrients in various forms) that will ultimately degrade ecosystem function.

The nutrient aspects of sustainability in land management (e.g., forestry, agriculture) perhaps provide the most unambiguous criteria for developing management plans consistent with sustainability principles. The contrasts between “strong” and “weak” sustainability—the proponents of the former saying that the Earth’s natural capital (the stock of natural ecosystems that yields a flow of valuable ecosystem goods or services) is irreplaceable, and proponents of the latter saying that economic growth can be accommodated by substituting technology and other resources for depleted natural capital—are less controversial in this context as nutrients are generally not substitutable. For example, if nitrogen is in short supply, extra phosphorus will not compensate for that limitation. The law of the minimum posited by the German chemist Justus von Liebig (1803–1873) provides the framework for this concept of nonsubstitution: growth will continue until the nutrient in the shortest supply becomes limiting; resupply of that nutrient will renew growth. Thus, understanding the basic dynamics of nutrients on human time scales provides a solid foundation for planning and management of ecosystems that would be consistent with principles of sustainability. More specifically, management of resource extraction to insure that nutrient stocks are not depleted and remain consistently available for continued net primary production (total plant growth via photosynthesis minus respiration of the plants) would be a key criteria of sustainable management. Because net primary productivity fuels the rest of the ecosystem (including primary, secondary, tertiary, etc., consumers and organisms that consume dead material and decompose organic matter), maintaining current levels of productivity sustains the entire ecosystem over time. Using a nutrient criterion for sustainability thus complements other sustainable management issues such as erosion, soil compaction, pollutant contamination, and loss of biodiversity.

Thinking like an Ecosystem

Prior to the advent of the ecosystem concept, concerns about forestry and agriculture focused more narrowly on issues such as species management and soil fertility. Foresters worried that after a tree harvest, their favored species would not regenerate to produce another crop of desirable trees to cut down the next time around. Agriculturalists worried that soil fertility would decline, and they would either have to find new land to till or purchase expensive fertilizers to apply to the land. The soil test was one solution to this concern. If the concentration of nutrients in the soil sample declined, then the remedy was to add the limiting nutrient (Liebig’s law again).

The foundation for understanding forests and agricultural fields in a different way—through knowledge of nutrient and biogeochemical cycling—began with the adoption of the ecosystem concept as a way of looking at forests, fields, and other areas. Articulated in 1935 by the British ecologist Arthur Tansley and experimentally employed to quantify ecosystem function in the 1960s (e.g., see Bormann and Likens 1967), the ecosystem concept provides the framework for describing nutrient stocks...
both conceptually and quantitatively. This concept focused on a bounded place as an interacting system of living and physical components (biotic and abiotic) that could be characterized by stocks and flows of matter and energy. For example, while the agriculturalist may have been happy when the concentration of a limiting nutrient was maintained at an optimal level, the actual stock of nutrients was declining as the depth of the soil was diminishing. This is analogous to evaluating the health of your bank account by the willingness of the bank teller to let you withdraw one hundred dollars a week. Sustainability requires you to keep track of the account balance.

By contrast, in using an ecosystem approach, places like forests and crop fields were studied by carefully establishing ecosystem boundaries and components, and then characterizing major inputs and outputs of substances like nutrients. In retrospect, this seemingly simple research perspective of defining a bounded ecosystem and considering inputs and outputs to be like deposits and withdrawals from the bank provided important new insights into the functioning of both natural and human-managed ecosystems. For example, early work by Harold Hemond (1980) on nutrient cycling in Thoreau’s Bog in Concord, Massachusetts, determined that it was ombrotrophic (rain fed), which accounted for the unusual plants, animals, and nutrient-retention survival strategies.

In addition, ecosystem analysis provided an integrative perspective on how living organisms and their abiotic components worked together. An ecosystem is a complex assemblage of hundreds and perhaps thousands of species of plants, animals, and microbes, along with the physical complexity of soil horizons, parent material, and ever-changing atmospheric conditions. The ecosystem approach simplified this complexity by focusing on a relatively small set of ecosystem components. This new understanding would eventually challenge existing management practices as described in sections below.

New Insights

New insights gained from studying nutrient cycling in ecosystems formed the basis for questioning the sustainability of human practices on the landscape. Not long after the development of techniques for ecosystem analysis in the late 1960s, F. Herbert Bormann and collaborators applied the ecosystem framework to the practice of clear-cutting. In one of many experiments they found manyfold increases in nutrient losses following cutting of trees. This finding generated a great deal of controversy and subsequent research, as well as new policies governing forest management. Clear-cutting could destabilize the biotic control of the cycling and retention of those nutrients, causing a loss of nutrients not only in the removal of the harvested wood but also in the leaching of nutrients into stream water. Another concern reported by the National Academy of Sciences in 1980 involved the conversion of moist tropical forest areas. In the previous decade, the ecologist Nellie Stark (1971a; 1971b) determined that the stock of nutrients in living biomass in Amazonian forests was very large in comparison with the stock of nutrients in the heavily weathered soils. This implied that removal of living biomass from the ecosystem might seriously jeopardize the ability of the forest to regenerate itself, given its reduced stock of nutrients. This was of particular concern with the conversion of primary forest, with their huge pools of nutrients in biomass, to secondary forest that then needed to re-accumulate nutrients from yet unquantified sources of inputs. In many areas, tropical deforestation may be unsustainable from a nutrient perspective.

Studies of nutrient dynamics in the context of management emphasized the need for research into the basic ecosystem dynamics of a whole suite of elements, including both nutrients and environmental contaminants. This new approach to understanding ecosystems has been subsequently applied to understanding acid rain, nitrogen saturation (over-fertilization), mercury pollution, and the biggest hurdle of the twenty-first century, global climate change.

Sustainability: Nutrient and Biogeochemical Cycling

As suggested above, nutrient cycling criteria may provide some of the most rigorous and unambiguous criteria for the evaluation of sustainability. Considering specific nutrients one at a time, we can use techniques of ecosystem analysis to achieve our best understanding of ecosystem function for that element. The joint consideration of many nutrient elements can be approached in the context of the previously mentioned Liebig’s law of the minimum. The measurement of the stock of this limiting factor can inform managers whether current practices are sustainable over the long term.

In the almost infinite diversity of natural and human-created ecosystems, the nutrient factors limiting primary productivity may vary considerably. One simplification is to consider two categories of nutrients: macronutrients (nutrients needed by life in large quantities, e.g., nitrogen, phosphorus, potassium, calcium, magnesium, sulfur) and micronutrients (nutrients needed in only minute quantities, e.g., boron, copper, iron, chlorine, manganese, molybdenum, zinc). Choosing a few to be illustrative of the application of nutrient and biogeochemical cycling to sustainability can provide the foundation for developing appropriate sustainability criteria.

In addition, as mentioned above, contaminants cycling is also a consideration for thinking about sustainable management. For example, natural wetland ecosystems
are increasingly valued for their pollutant “filtering” capacity, and constructed wetlands are often designed specifically for these functions. Understanding cycling in these contexts is essential for managing contaminants over the long term in relation to sustainability principles.

The terms nutrient cycling and biogeochemical cycling need some further discussion as they are both used in the title of this section. As used here, the distinctions in the terms lie in how they may be applied to nutrients versus contaminants. Biogeochemical cycling is the broader of the terms as it applies to any element moving through the biosphere and lithosphere, whether it is a nutrient of life or not. Thus the biogeochemical cycling of a contaminant like mercury (Hg) is not appropriately considered nutrient cycling as there is currently no known need of mercury in animal or plant physiology. Nutrient cycles on the other hand have biological, geological, and chemical components to their cycles, and thus are all properly biogeochemical cycles. Therefore, when discussing nutrient elements, context and semantic preference will govern the use of either term.

Cycling and Time

Most advocates of sustainability work in the context of human time scales. Even the long view of some native peoples (some who are said to plan in the span of “seven generations”) is framed within a time scale relevant to social processes and not geologic processes. Geologic processes figure prominently in the cycles of most nutrient elements however. Thus understanding biogeochemical cycles over large spans of time is critical to developing an understanding of sustainability and cycling.

For example, at human time scales, calcium cycling is not a cycle at all. Calcium weathers out of rocks, goes into solution, gets carried by ground-, subsurface, and surface water to the ocean, and then precipitates or settles on the seafloor through biotic and chemical processes. While this march to the ocean can be interrupted by ecological processes and recycled to the terrestrial system to then restart the journey to the sea, the vast majority of calcium atoms take a one-way trip to the sea. This deple- tion of calcium, like the mining of fossil fuels, degrades the pool of calcium with each successive millennium, weathering towering alps to gentle green mountains to pleasant hills and valleys. Also known as a sedimentary cycle to geologists, this calcium story is very much a cycle as the calcium-rich sediments accumulate to great depths over hundreds of millions of years, form rocks, and may eventually get uplifted by tectonic forces and reemerge as towering mountains that will weather the forces of physical erosion and acid dissolution, and thus start the process all over again. Thus sustainability principles applied in the context of nutrient and biogeochemical “cycles” must consider both the maintenance of elemental stocks and their dynamics (e.g., weathering).

The variability of nutrient dynamics in many developing ecosystems also requires consideration in the context of sustainability. For example, if we harvest trees on a continuous cycle in a way that does not deplete the pool of nutrients, thereby sustaining subsequent productivity for the next forest cycle, we might all agree that at least from a nutrient perspective, the practice is sustainable. But even in the relatively rapid span of human lives, natural ecological processes change the status of nutrient pools. For example, Bernard Bormann (the son of F. Herbert Bormann) and collaborators (1998) documented that primary succession of forest over sand can rapidly build biomass and available nutrient pools. Thus the “normal” dynamics of nutrient stocks, including available soil nutrients, would be increasing. Would sustainability criteria focus on increasing nutrient stocks or just maintaining a constant level? In another example, natural fires deplete ecosystem nitrogen (through conversion to nitrogen oxides that are lost to the atmosphere), lowering total nitrogen stocks below what would be present without fire. Right after fire, however, mineralized nutrients boost recovery through short-term fertilization. Thus managing fire frequency sustains different levels of productivity. What is the norm that we would want to manage sustainably? In both examples above, management may sustain stocks of nutrients in the ecosystem at static levels, where natural ecosystem development may not sustain constant stocks. Thus there are specific cases where principles of sustainability may require a more dynamic interpretation of nutrient “cycling.”

Sedimentary Cycles

In a geologic time frame, cycles have been categorized as sedimentary cycles and gaseous cycles, with some cycles having important elements of both. Each has some important distinctions in the context of sustainability. The calcium cycle, introduced above, is a good example of a sedimentary cycle. Elements categorized as having a sedimentary cycle have a very long cycling time associated with weathering from rock substrates, transport in surface waters, sedimentation in ocean waters, rock formation, uplift, and back to weathering.

For a terrestrial system, the implication of this long-term biogeochemical cycle is that nutrients need not be conserved within the ecosystem. Outputs will exceed inputs as the storage pool of nutrients in rocks is slowly depleted. Limestone weathering provides a simplified model of the sedimentary cycle starting with the following carbonate weathering equation:

\[ \text{CaCO}_3 + \text{H}^+ + \text{HCO}_3^- \rightarrow \text{Ca}^{2+} + 2\text{HCO}_3^- \]

Calcium (in ionic form: Ca\(^{2+}\)) participates in the internal or intrasystem cycle within the ecosystem, where calcium drives growth of plants and feeds the nutritional needs of the many animals, fungi, and microbes that make
up the ecosystem. But the net export puts these positively charged calcium cations into the surface waters of the world to then travel to the ocean. During this journey to the ocean, calcium continues to feed aquatic plant growth and all the organisms that depend on this productivity. Reaching the ocean, calcium is an important nutrient for many organisms, but to feed the long-term biogeochemical cycle, there are key organisms, such as diatoms, that use calcium as structural components. Upon their death, these calcium-rich structures settle to the bottom of the sea and begin the rock-formation process. Millions of years later, with an uplift event, these calcium atoms return to feed terrestrial organisms through the weathering process.

Thus, sustaining ecosystem productivity requires not the conservation of calcium, but the sustainable stewarding of the element on its way to the sea. Where humans seek to remove products of biology such as trees or crops from the ecosystem, the normal weathering rate might be a rough measure of how much can be removed without diminishing the stock of available calcium. This recognition of the role of weathering in nutrient cycling was first introduced by Nellie Stark (1978) with the proposal of a “biological life of soils.” Working in both young, recently glaciated forest soils and heavily weathered soils in the tropics, Stark described the natural evolution of sedimentary cycles and proposed that removal of nutrients through (a) crop or tree harvest and (b) land management practices that could increase or decrease weathering rates were important components of balancing human use and natural processes.

The components of this balance involve weathering rates, other natural (e.g., nutrients carried in precipitation) and anthropogenic inputs (e.g., particulate pollution), harvest removals, and aqueous outputs (e.g., calcium leaving the ecosystem in surface and ground water). This balance should be evaluated over the cycle of harvests (e.g., rotation rate of tree harvests) to evaluate sustainability. In between harvests, the balance also includes the accumulating storage of calcium in living and dead biomass. But if we assume no net change in biomass over the course of a harvest rotation, then the most important change in storage will be the available pool of calcium. A simplified mass balance equation can assist in conceptualizing the role of management on nutrient balance:

\[
\text{weathering} + \text{inputs} + \text{depletion of nutrient stock} = \text{harvest} + \text{outputs}
\]

(Inputs include natural phenomenon like precipitation and anthropogenic sources such as particulate pollution; outputs include losses to stream water and groundwater.)

Clearly, excessive harvests or human impacts that increase outputs of particulate or dissolved nutrients will deplete the stock of available calcium. If calcium scarcity, resulting from a decline in the available stock, reduces primary productivity, then the management practices could be considered to be unsustainable.

An interesting dimension of this mass balance model of ecosystem dynamics is that the stock of available calcium is not tied to harvest levels. As long as harvests and aqueous losses do not exceed weathering and other inputs, the ecosystem remains in nutrient balance. Thus this balance could be achieved under conditions of either low nutrient availability or high nutrient availability. Clearly there are some management advantages to designing the production system that maintained high calcium availability with consequent higher productivity. This would result in more of the nutrient outputs leaving as product and less as aqueous losses in stream water.

Gaseous Cycles

Gaseous global cycles do not have an important sedimentary phase. Thus rock weathering is not an important source of these elements. Again, each cycle has its own peculiarities, but nitrogen (N) is a good nutrient to illustrate a gaseous nutrient cycle. Nitrogen is also a limiting nutrient element in many cases and thus serves to illustrate the relationship between nutrient cycling and sustainability.

In a simplified model of the nitrogen cycle, the large reservoir of nitrogen is the atmosphere, in contrast to calcium, which has a huge reservoir in rocks. Like the calcium in rocks, nitrogen, as N\(_2\) in the atmosphere, is unavailable to life in that form. The entry of nitrogen into life cycles requires capture and conversion of nitrogen into forms available to living organisms (nitrogen fixation). This fixation can happen through either abiotic processes (e.g., lightning) or biotic processes (biological fixation, e.g., through specially adapted symbiotic organisms like *Rhizobium* and *Frankia*). While common members of the community in some ecosystems, nitrogen fixers are lacking in many ecosystems. The other major source of nitrogen in terrestrial ecosystems is precipitation. Nitrate (NO\(_3^-\)) and ammonium (NH\(_4^+\)) are found in precipitation. Natural levels of these compounds are generally low, but industrial pollution has increased the level of these and related compounds to many times the natural levels. The net input rate of nitrogen to most terrestrial ecosystems is still rather low however. Once accumulated in terrestrial ecosystems through fixation or precipitation, nitrogen must be conserved through limiting losses, or the stock of nitrogen in the ecosystem will decline. Thus most natural ecosystems have evolved mechanisms to limit nitrogen losses. Again using a mass balance approach, the following simplification approximates the nitrogen balance in ecosystems:

\[
\text{precipitation} + \text{fixation} + \text{depletion of nutrient stock} = \text{harvest} + \text{losses to ground-} \text{and surface water} + \text{atmospheric losses}
\]

In the absence of nitrogen fertilization, human management of nitrogen must then be designed to balance
output of nitrogen through harvest removals and land management with inputs of nitrogen. With the primary input of nitrogen being bulk precipitation for many ecosystems, this rate of input generally translates to long rotation times (e.g., in the case of forestry). For annual agricultural crops where the annual removal of nitrogen in crops is high, this generally means that some kind of inorganic or organic fertilizer is required to replenish the stock of nitrogen in the soil. Depletion of nutrients through poor land management can also be an important management issue as discussed earlier. If erosion and leaching losses of nitrogen are high, sustainable practices require that harvest outputs be low in order to allow for a balanced nutrient cycle.

Contaminant Biogeochemical Cycles

The biogeochemistry of contaminants in ecosystems is of great concern to both environmental scientists and regulatory agencies. Many long-term, persistent contaminants have chronic effects on human health and also may lead to mortality through cancers and other deadly diseases. It is difficult to generalize about contaminant biogeochemistry because the elemental dynamics are so variable (e.g., mercury versus lead versus synthetic organics like DDT). For example, mercury (Hg) is volatilized (turns to gas and is then released into the atmosphere) in coal combustion, enters terrestrial and aquatic ecosystems through precipitation, and then re-enters the atmosphere through volatilization of gaseous mercury. While the accurate quantification of mercury “cycling” (as discussed earlier, most of many elements may actually not complete a cycle) is still under development (see Grigal 2002), one of the concerns is that human alteration of natural mercury cycling is resulting in both higher fluxes of mercury through terrestrial ecosystems and greater accumulation of mercury in aquatic ecosystems. Also, because mercury bioaccumulates (i.e., is concentrated to very high levels as it moves through the food chain) in aquatic ecosystems, it has critical impacts on aquatic organisms and the humans who eat fish from the sea and many lakes.

Tracing contaminant movement through all parts of the ecosystem and quantifying how and where it accumulates is a key part to understanding contaminant biogeochemistry and consequently managing contaminant dynamics to the greatest extent that we can. Unfortunately, because of the large number of industrial contaminants created by human activity, this work is very challenging and will require increased and continuing funding to achieve a level of understanding that can accurately inform sustainable management and policy development.

Managing Biogeochemical Cycles

While the framework of balancing inputs and outputs to maintain available pools of nutrients is simple in concept and helpful in evaluating sustainability, uncertainty in measuring all components of the cycle results in some large ranges in possible outcomes, especially when extrapolated over longer time frames. Long-term ecological research is just beginning to approach the time spans under consideration in many management scenarios (e.g., forest rotations). Nevertheless, current levels of accuracy in measuring important ecosystem variables—such as nutrients in bulk precipitation, weathering rates, denitrification, nitrogen fixation, outputs of nutrients in stream water, and stocks of macronutrients in soils—can provide important insights into key ecosystem cycles that can inform sustainable management. In particular, advances in research on the cycling of nitrogen, phosphorus, potassium, calcium, magnesium, and sulfur, and contaminants like lead and mercury can aid in framing decisions about sustainable practices and policies.

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See also: Eutrophication; Food Webs; Groundwater Management; Microbial Ecosystem Processes; Mutualism; Natural Capital; Nitrogen Saturation; Pollution, Nonpoint Source; Pollution, Point Source; Safe Minimum Standard (SMS); Soil Conservation

Further Reading


Ocean Acidification—Management

The increasing carbon dioxide (CO₂) concentration in the Earth’s atmosphere is absorbed by the oceans and leads to changes in ocean carbon chemistry. This process of ocean acidification results in a range of biological and socioeconomic impacts. Accelerating acidification under current rates of CO₂ emissions is expected to compromise the function of global marine ecosystems during this century. Management actions are urgently needed to help counteract these impacts.

The concentration of carbon dioxide (CO₂) in the Earth’s atmosphere has increased dramatically since the Industrial Revolution (from around 280 parts per million [ppm] in preindustrial times to 392 ppm in 2011), primarily due to human activities such as the burning of fossil fuels and land-use activities (IPCC 2007). This buildup of CO₂ is recognized as one of the primary causes of global climate change. Over the last few decades, only half of the CO₂ released by human activities has remained in the atmosphere; 25 percent has been absorbed into the oceans (Sabine et al. 2004). This absorptive capacity of the oceans has helped to buffer the impacts of global warming associated with increased atmospheric CO₂ emissions, but it has come at a cost in the form of ocean acidification.

When atmospheric CO₂ dissolves into seawater, carbonic acid is formed, and hydrogen ions are released. As a result, the pH of the ocean surface waters decreases, making it more acidic. On the 14-point pH scale, lower numbers (0–6.9) designate acidic water, while higher numbers (7.1–14) designate basic water; a pH of 7.0 is neutral. Oceans are naturally slightly basic (on average, pH > 8.1), and acidification via CO₂ uptake is expected to drive pH as low as 7.6 by the end of this century. This pH change will affect the ocean carbon chemistry system, and the biological and ecological processes that depend on it in numerous ways.

When hydrogen ions are released in seawater, they combine with carbonate ions (forming bicarbonate), thereby lowering the carbonate ion concentration. Carbonate ions are the building blocks (calcifiers) for the shells and skeletons of many marine organisms, such as corals, crustaceans (e.g., lobsters and crabs), and mollusks (e.g., clams and oysters). Lowering the pH thus reduces the saturation state of calcium carbonate, which makes it harder for organisms to form the calcium carbonate needed for their shells. Calcification, or the process of “shell building,” depends on the availability (saturation) of carbonate ions in seawater. The calcification rates of marine calcifiers are generally sensitive to a decline in carbonate ion concentration. More specifically, changes in the carbonate ion concentration in the oceans can affect the saturation state—and hence biological availability—of several forms of calcium carbonate that these species depend on for shell building, including calcite, aragonite, or high-magnesian calcite (Feely, Doney, and Cooley 2009).

Currently, calcifying marine organisms in most areas of the ocean surface can build skeletons and shells because the water is saturated with calcium carbonate (Pelejero, Calvo, and Hoegh-Guldberg 2010). The pH of the ocean surface waters has already decreased, however, by about 0.1 units since the beginning of the Industrial Revolution (Feely et al. 2004), reducing the saturation of aragonite or calcite that these organisms need. Because the pH scale is logarithmic, a 1-unit decrease in pH is equivalent to a ten-fold increase in acidity. Ocean pH is projected to drop an additional 0.4 pH units by 2100 under a high CO₂ emission scenario (IPCC 2007), with carbonate saturation levels potentially falling below those required to sustain coral reef accretion (Royal Society 2005; Hoegh-Guldberg et al. 2007; Silverman et al. 2009). Such changes in the carbon chemistry of the open ocean probably have not occurred for more than 20 million years (Feely et al. 2004).
Changes in ocean acidity vary globally. High-latitude surface waters (waters near and right below the North and South Poles) have a naturally low concentration of carbonate ions because atmospheric CO$_2$ is more soluble in colder seas. As a result, these waters experience a higher degree of ocean acidification than warmer ocean waters and are therefore likely to become undersaturated with respect to aragonite before tropical and subtropical waters (Feely et al. 2004). Models suggest that oceanic waters will be undersaturated with respect to aragonite by 2020 in the Arctic Ocean and by 2050 in the Southern Ocean surrounding Antarctica (Orr et al. 2005; Steinacher et al. 2009).

Tropical coral reefs are also vulnerable to ocean acidification. Some globally important coral reef regions, such as the Great Barrier Reef, the Coral Sea, and the Caribbean Sea, are projected to attain dangerously low states of aragonite saturation more rapidly than other regions such as the Central Pacific Ocean (Hoegh-Guldberg et al. 2007). Despite global ocean acidification patterns, a number of local-scale ecological processes affect the rate and geographic scale of ocean acidification. Large variations in pH and aragonite saturation states have been documented on some coral reefs. On Heron Island Reef in the Great Barrier Reef, Australia, for example, variations in pH and aragonite saturation state over the course of one day were greater than the predicted changes that ocean acidification will cause for the oceans globally by the middle of the century (Anthony et al. 2008; 2011). These results suggest that although general patterns in aragonite saturation are evident for open oceans, they will vary significantly both spatially and temporally as a result of reef-scale processes.

Climate change and ocean acidification challenge marine conservation managers and research scientists because they force them to try to manage global threats at local scales. Unlike other global stressors, such as increasing sea surface temperature that leads to bleaching (visible whitening) and widespread mortality of coral reefs, ocean acidification is largely an invisible, insidious environmental problem. Because of the relatively recent awareness of the threat of ocean acidification, little guidance for managing its impacts exists. Further, the majority of research is focused on addressing the responses of marine organisms to changes in ocean chemistry or on projecting global-scale changes in ocean chemistry; little emphasis has been placed on developing management or policy recommendations to address these impacts (but see McLeod et al. 2008).

**Impacts of Ocean Acidification**

A number of groundbreaking studies conducted in the late 1990s predicted that coral reefs would show dramatic responses to changes in ocean chemistry during the twenty-first century (Gattuso et al. 1998; Kleypas et al. 1999; Marubini and Atkinson 1999). Since then, the effects of ocean acidification on marine organisms and ecosystems have become increasingly evident through experimental and observational research. Ocean acidification has demonstrable impacts on a number of biological and ecological processes in many marine groups—including phytoplankton, corals, other invertebrates, and fishes—around the globe (Kroeker et al. 2010). Most studies have focused on impacts of ocean acidification on calcification (shell building) and dissolution (shell dissolving or disruptions in formation), but impacts on other processes such as early life-history stages are reported with increasing frequency (Dupont and Thorndyke 2009; Albright and Langdon 2011). For example, ocean acidification can lead to excessive CO$_2$ levels in the blood (CO$_2$ toxicity) of fish and cephalopods and significantly reduced growth and fecundity in some invertebrates (Orr et al. 2005). For species with long generation times, slower growth and lower fecundity can lead to population declines. Recent works indicate that ocean acidification may lead to sensory and neurological dysfunction in marine fish larvae—that is, they can’t “smell” the reef and thus fail to distinguish predators from parents (Munday et al. 2010).

Importantly, ocean acidification doesn’t affect all marine organisms equally. Some hard corals have linear responses while others show accelerating responses to reductions in carbonate ion concentration (Reynaud et al. 2003; Jury, Whitehead and Szmant 2009; Rodolfo-Metalpa et al. 2010). While accelerating responses may result in a catastrophic tipping point for some species, it is not yet possible to define such critical points for individual species or broader ecosystem changes. The ability to define tipping points is limited because most studies on ocean acidification impacts are based on experimental work over short time scales and for a single species. Little is known about how populations and ecosystems will respond to ocean acidification, the combined effects from other stressors (e.g., pollution, overfishing, increasing ocean temperatures), and the ability of organisms to adapt. The variety of responses of marine organisms is partially the result of the wide variety of processes that ocean acidification affects, such as dissolution and calcification rates, growth rates, development, and survival (Kroeker et al. 2010). This variation in responses makes predicting the impacts of ocean acidification on species and marine ecosystems complex. Despite these complexities, however, recent studies indicate that, overall, ocean acidification will harm calcifying marine organisms (Hendriks, Duarte, and Alvarez 2010; Kroeker et al. 2010).

Whereas recent studies have explored the biological impacts of ocean acidification on marine organisms, less
emphasis has been placed on assessing the socioeconomic impacts. Because ocean acidification affects marine organisms’ abilities to form shells, it may decrease the abundance of commercially important shellfish species such as clams, oysters, and sea urchins, affecting the human communities that depend upon these resources for food and/or livelihoods (Cooley, Kite-Powell, and Doney 2009). Ocean acidification may thus affect human communities through a loss of goods and services provided by ecosystems such as coral reefs—for example, tourism revenues, fisheries, coastal protection, and cultural values. Such goods and services are valued in billions of dollars (Burke et al. 2011). The Great Barrier Reef, for example, contributes more than $5 billion annually to the Australian economy (Access Economics 2005).

Additional research is needed to assess the deeper socioeconomic impacts of ocean acidification in countries whose communities directly depend upon natural marine resources for survival (Cooley and Doney 2009). Such research could provide motivations for action that extend beyond ocean acidification and climate change, because economic analyses and models of ocean acidification’s impacts on fisheries and tourism are necessary to understand the true comprehensive costs of action or inaction to reduce global CO₂ emissions (Fulton et al. 2011).

Potential Management Options

The most critical action needed to address ocean acidification is to stabilize atmospheric CO₂ concentrations. Reviews suggest that policies that allow the global average atmospheric concentration of CO₂, currently approaching 393 ppm to reach or surpass 500 ppm of CO₂ are likely to be extremely risky for corals reefs (e.g., Hoegh-Guldberg et al. 2007). Thus, ocean acidification provides another impetus for comprehensive and effective global policies to address CO₂ emissions.

Unfortunately, reducing global emissions is beyond the scope of marine conservation managers. A more immediate need is to identify and implement local actions that support marine ecosystem (e.g., coral reef) health in the face of global threats such as ocean acidification. A study by the National Research Council (2010, 85), Ocean Acidification: A National Strategy to Meet the Challenges of a Changing Ocean, reviewed the current state of the knowledge of ocean acidification and stated that “there is not yet enough information on the biological, ecological, or socioeconomic effects of ocean acidification to adequately guide management efforts.” This conclusion is based on the fact that most research has focused on the impacts of ocean acidification on few species over short time scales. As a result, there are major research gaps regarding ocean acidification’s larger importance, including how ocean acidification will affect many ecologically or economically important species and communities, how it will affect a variety of physiological and biogeochemical processes, and what the potential will be for organisms to adapt to projected changes in ocean chemistry (Boyd et al. 2008).

The science needed to address these gaps will probably take decades to fully develop. Waiting to take management action until the science is complete, however, may put critical marine ecosystems at risk. Local actions can and should be taken now to protect marine ecosystems. Such actions include reducing the other stressors that affect most marine ecosystems, such as declining water quality, coastal pollution, and overfishing of important species and functional groups, such as herbivores (Hughes et al. 2003). Reducing land-based sources of pollution, such as nutrient runoff from agriculture and sediment runoff from coastal development, is particularly important for managing the impacts of ocean acidification, because nutrients like phosphorus and nitrogen and land-based carbon inputs can lower pH and aragonite saturation states in coastal and oceanic waters (Andersson, Mackenzie, and Lerman 2006). To achieve these goals, marine management efforts must be integrated with land-use and coastal-zone planning and practices to help reduce pollutant inputs. More generally, the reduction of stressors on marine ecosystems supports ecosystem health and will better allow marine organisms to channel resources to growth, calcification, and reproduction rather than to repairing damage and recovering from disease (McLeod et al. 2008).

Current ocean acidification research is exploring differences in the sensitivity of marine species and habitats to changes in ocean chemistry. If scientists can identify species or habitats that are less vulnerable to the impacts of ocean acidification, these could become priorities for inclusion in marine protected areas (MPAs). Less vulnerable areas may include coral reefs in carbonate rich areas, such as places where there are raised reefs and limestone islands, extensive reef flats, patch reef/coral head complexes, and carbonate sediment deposits. Other candidate areas for increased protection through MPAs may be high-diversity reef complexes that are well flushed by oceanic water, because these influxes of fresh oceanic water bring higher total alkalinity and saturation states that support reef and shell building. Well-flushed areas may be more vulnerable in the future, however, if ocean acidification causes significant decreases in the pH of oceanic waters. Therefore, a strategy of spreading the risk by selecting examples of coral reef areas in a variety of ocean chemistry and oceanographic regimes is a useful MPA design
approach. By protecting multiple examples of such reef areas, MPA managers are helping to ensure that these ecosystems are more likely to survive climate and other human threats.

Coral reefs located in areas with high variability in seawater temperature are thought to be less vulnerable to thermal stress and associated bleaching and mortality caused by increases in sea surface temperature (McClanahan et al. 2007). If reefs in areas with high natural variability in ocean chemistry are also less vulnerable to ocean acidification, then managers could prioritize reefs in these areas for inclusion in MPAs. In the absence of such information, MPA managers may choose to locate MPAs in a range of ocean chemistry regimes (including areas with high and low variability). Additionally, selecting reefs in a variety of pH and aragonite saturation regimes increases the chances that managers will identify and protect corals that are acclimatized to a variety of pH conditions and spreads the risk of any coral species’ survival being compromised by ocean acidification.

Actions that address threats such as increasing sea level, increasing sea surface temperature, and ocean acidification must be incorporated into MPA management plans to reduce the impacts of increasing atmospheric concentrations of CO₂ on marine species and ecosystems. It is also important to develop and test the efficacy of innovative interventions that reduce the effects of ocean acidification on high-priority areas and species. Such interventions include CO₂ capture and storage strategies. The geographic scale, time frame, and economic and environmental costs and benefits of these interventions must be explored further before they can be implemented.

Research Needs and Next Steps

A number of priority research needs should be addressed to support the management of ocean acidification. While models can project changes in ocean chemistry at global and regional scales, these models do not take coastal processes into account. Coastal areas already experience extreme variability in water chemistry because of natural and human inputs, such as acidic discharge of river water (Salisbury et al. 2008), atmospheric deposition of nitrogen and sulfur (Doney et al. 2007), and eutrophication resulting from land-use changes and agriculture, a process where a body of water (in this case the ocean) receives excess nutrients that stimulate algal growth (Borges and Gypens 2010). The processes that affect coastal carbonate chemistry are complex and not well resolved, and improved understanding is needed to manage the responses of marine organisms, ecosystems, and industries in coastal areas (National Research Council 2010).

The mitigation of local causes of ocean acidification using existing environmental laws has been proposed recently (Kelly et al. 2011). Local and state governments have the authority and capacity to address many stressors that can exacerbate ocean acidification conditions in coastal waters. For example, enforcement of the US Clean Water Act can help ensure that precipitation runoff and associated pollutants, which can increase acidification, are limited. Controlling coastal erosion can help reduce nutrient and sediment loading of water; such coastal inputs also may be enriched with fertilizers that can increase acidification, providing another reason for reducing them. Changes in land-use patterns, such as changes in deforestation practices, can reduce direct and indirect CO₂ emissions, runoff, and other threats. Enforcing federal emission limits for pollutants such as nitrogen oxide and sulfur oxide (e.g., from coal-fired power plants) can help reduce ocean acidification impacts in coastal waters (Kelly et al. 2011). Despite the clear benefits of minimizing additional stressors on coastal ecosystems through enforcement of existing environmental laws, such actions are inadequate to reverse the impacts of ocean acidification at a global scale.

To help conservation managers identify and protect marine species and communities that are most likely to survive changes in ocean chemistry, additional research is needed to investigate the response of organisms, populations, and communities to ocean acidification. Further, research exploring the capacity of marine organisms to acclimatize or adapt to changes in ocean chemistry will be important in understanding their susceptibility to future changes. The ability to identify species that are less vulnerable to changes in ocean chemistry is also useful to support aquaculture programs, because such species may be used for selective breeding. Field studies that document changes in ecosystem structure and function over
natural pH gradients are also useful for highlighting thresholds that trigger widespread ecological and biological changes. The ability to identify indicators of regime shifts (e.g., from coral to algal dominance) helps marine managers take actions to avoid such shifts or cope with them (Anthony et al. 2011).

No impact occurs in isolation, so studies need to address the interactive effects of multiple stressors. Marine ecosystems currently must deal with changes in ocean pH in addition to increasing sea surface temperatures, changes in sea level, and other human impacts such as pollution, coastal development, and overfishing. It may be challenging for researchers to attribute ecosystem changes to a specific stressor given the suite of challenges facing marine ecosystems. Managers, however, need the ability to understand how species, communities, and ecosystems respond to ocean acidification in concert with additional stressors in order to predict changes and develop appropriate management responses.

Decision makers need socioeconomic research on the impacts of ocean acidification, the projected timing of impacts, and ways to increase adaptability and resilience of socioeconomic systems. To prioritize research and mitigation and adaptation activities, assessments of the cost of ocean acidification on marine ecosystems and resources are essential. More broadly, the public needs educational and informational materials that communicate the implications of ocean acidification on marine ecosystems and dependent communities, and that emphasize actions that can be taken to reduce these impacts.

The ecological, biological, and socioeconomic impacts of ocean acidification pose significant challenges to marine species and the human communities who depend upon them for food and livelihoods. Anthropogenic CO₂ emissions must be dramatically reduced to mitigate these impacts. Our best hope for tackling the challenge of ocean acidification is the implementation of a suite of four key management actions: (1) curb and stabilize atmospheric CO₂ concentrations; (2) protect marine species and areas likely to be less vulnerable to ocean acidification; (3) explore and apply CO₂ capture and storage methods where feasible; and (4) use existing environmental laws to control stressors that exacerbate ocean acidification conditions in coastal waters.

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See also Catchment Management; Coastal Management; Ecological Restoration; Eutrophication; Fisheries Management; Food Webs; Global Climate Change; Indicator Species; Large Marine Ecosystem (LME) Management and Assessment; Marine Protected Areas (MPAs); Pollution, Nonpoint Source; Pollution, Point Source; Regime Shifts; Resilience

Further Reading


Rodolfo-Metalpa, Riccardo; Martin, Sophie; Ferrier-Pagès, Christine; & Gattuso, Jean-Pierre. (2010). Response of the temperate coral *Clatharea caspita* to mid- and long-term exposure to pCO₂ and temperature levels projected for the 2100 AD. *Biogiosciences, 7*, 289–300.


The ocean, provider of food, oxygen, even medicine for millennia, is no longer the unchangeable body it was once thought to be. It is heavily affected by human activities, from overfishing to waste runoff to mineral extraction, destroying habitats and irrevocably changing ecosystems. National governments and international organizations must strengthen and integrate their efforts toward sustainability to retain the viability of marine ecosystems and resources.

The ocean—covering 72 percent of Earth’s surface—provides vital, life-sustaining services to the global population. The world’s oceans generate half the oxygen on Earth, are the primary regulator of global climate, and provide economic and environmental services to billions of people. The oceans also act as an important sink, having absorbed over 80 percent of the excess heat and approximately one-third of all anthropogenic carbon dioxide (CO₂) since the onset of the Industrial Revolution. Marine biodiversity and ecosystem resources and services provide basic life necessities, including food, fresh water, wood, fiber, genetic resources, medicines, and cultural products. Coastal areas are heavily utilized—half the world’s population lives within 100 kilometers of the sea, and three-quarters of all large cities are located on the coast (UNEP and UN-HABITAT 2005).

Throughout history, people have had a long-standing connection with, and dependence on, the sea. Humanity’s relationship with the ocean has changed in recent decades, increasing the demands and impacts placed upon the ocean. Today, the greater ocean no longer seems so vast and unreachable. It has also become apparent that the marine environment is not unaffected by our actions, even distant actions, on land and at sea, but that our compounding impacts have far-reaching consequences for often fragile marine ecosystems. The ocean is not the resilient body we once thought it to be but is heavily impacted by human actions throughout all regions of the world, as well as at great depths.

Human Activities Affecting the Ocean

Humans interact with the ocean in numerous ways. The most intense uses of ocean resources and space are often associated with more traditional activities, such as fishing, shipping, offshore oil and gas extraction, laying of cables, tourism and recreation, and coastal development. More recently, human activities have expanded to include mineral extraction, the extraction of marine genetic resources, marine renewable energy, the construction of artificial reefs, land reclamation, coastal defense, and dredging and dumping. The United Nations Conference on the Human Environment recognized in 1972 the overexploitation of living marine resources, the physical alteration of habitats, and marine pollution. These major threats to the ocean persist today.

Global Distribution and Impacts

Fishing is perhaps the most long-standing and common use of ocean resources. It is also the activity that most heavily exploits ocean resources. The United Nations Food and Agriculture Organization (FAO) estimates that fish provide over 3 billion people with 15 percent of their total animal protein intake, and some type of fishing activity occurs in every region of the ocean. Global decline in fish stocks began in the early 1990s. Today, with total catch exceeding 100 million tons per year (including discards, bycatch, and illegal, unregulated, and unreported fishing), the FAO estimates that 85 percent of stocks are either fully exploited, overexploited,
depleted, or recovering from depletion, giving cause for concern about the long-term sustainability of these key resources (FAO 2010).

Destructive fishing practices also damage marine habitats. For example, bottom trawling, which involves dragging large, heavy nets along the sea floor, effectively decimates benthic habitats, that is, habitats of organisms living at the bottom of the ocean. In the deep ocean, huge reservoirs of marine biodiversity exist, particularly surrounding seamounts, hydrothermal vents, methane seeps, and deepwater corals. Organisms that thrive in these environments evolved under extreme environmental parameters and offer genetic material highly valuable for scientific discovery and commercialization. This material has numerous applications in pharmaceutical, biotechnological, and cosmetic fields. Many of these ecosystems are vulnerable to destructive fishing practices and potential overexploitation from commercial extraction.

The physical alteration and destruction of habitats is arguably the most important threat to coastal resources and environments. Social and economic development in these areas has led to coastal habitat destruction resulting from increasing pressures from population, urbanization, industrialization, marine transportation, and tourism. Such destruction often comes at significant environmental and economic (in terms of losses of ecosystem services) costs. For example, coral reefs, which provide habitat for over 1 million aquatic species, including thousands of fish species, and provide natural barrier protection from increased storm surges and wave activity, are estimated by the Economics of Ecosystems and Biodiversity project to be valued between US$130,000 and US$1.2 million per hectare per year (Diversitas 2009). Yet an estimated 58 percent of coral reefs worldwide are threatened, with habitat destruction one of the key contributors, though climate change and ocean acidification threaten far more. Mangroves are also highly valuable coastal ecosystems, providing protection against storms, nursery groups for offshore fisheries, and wood and non-wood forest products, with a total monetary value estimated at US$10,000 per hectare per year (Costanza et al. 1997), not accounting for additional services provided by these habitats (e.g., carbon sequestration). In the past century, over one-half of all mangrove forests have been lost, largely as a result of physical alteration. Wetland and sea-grass communities are also at risk and continue to decline worldwide, drastically reducing their ability to provide similar ecosystem services.

Marine pollution also results from human interactions with ocean resources and space. Approximately 90 percent of world trade is carried by ship. Shipping damages the marine environment through, among other activities, oil spills and accidental discharges, chemical accidents at sea, waste disposal and water pollution, sound pollution, and the discharge of ballast water—ships’ ballast water transports approximately three thousand species of plants and animals each day, which can lead to an uncontrollable growth of invasive species in some marine ecosystems.

Notably, however, the most significant threat of marine pollution comes not from activities at sea but on land. Land-based activities contribute some 80 percent of all pollution entering the oceans. Sewage continues to be the largest source of contamination by volume, but wastewater and agricultural nutrient runoff are also large polluters. Together, excessive nutrients from sewage outfalls and agricultural runoff have contributed to a rise in the number of dead zones (hypoxic or anoxic areas) in the marine environment, from 149 in 2003 to over 200 in 2006, resulting in the collapse of some ecosystems (Nellemann, Hain, and Alder 2008). Plastics and other debris that make their way to the ocean accumulate and further affect marine resources and ecosystems. While it is difficult to calculate the distribution of waste in surface waters, water columns, and on the seafloor, recent studies and observations confirm that debris is transported by ocean currents and tends to accumulate in a limited number of convergence zones, or gyres, in what has been termed “garbage patches.” For example, in the North Atlantic and Caribbean convergence zone, over 200,000 pieces of plastics per square kilometer have been found (UNEP 2011). Additional patches have been confirmed in an area midway between Hawaii and California, around the North Pacific Subtropical High, as well as off the coast of Japan in a small recirculation gyre. Another area of high concern is the North Pacific Subtropical Convergence Zone, where a high degree of marine debris concentrates (and where a high diversity of marine life also exists). One model simulation of global marine debris distribution after ten years shows plastics converging in the five gyres, namely the Indian Ocean, North and South Pacific, and North and South Atlantic (IPRC 2008). The debris has lethal and sublethal effects on biodiversity, entanglement, chemical contamination, and the alteration of community structures. In a recent study of planktivorous fish from the North Pacific gyre, an average 2.1 plastic items were found per fish (Boerger et al. 2010). The global community needs to study, better understand, and address the potential impacts of accumulation and releases from plastic particles, including persistent, bioaccumulating, and toxic substances. Enhanced understanding is needed as well on the long-term impacts and effective strategies for addressing other types of hazardous substances leeching into the marine environment, such as mercury, lead, polycyclic aromatic hydrocarbons (PAHs), and polychlorinated biphenyls.
Management Challenges

The overarching framework for global ocean management is the 1982 United Nations Convention on the Law of the Sea (known as UNCLOS). Specific management provisions and challenges vary across sectors and geographic areas and through time. Managing human interactions with respect to ocean resources means balancing environmental and developmental needs. In parts of the world where persistent poverty and inequality loom large, strategies must focus on the longer-term benefits of sustainable management practices.

Inherent inequities exist in the global trade of ocean products, and there is no sound framework for benefit sharing. Sixty-four percent of the global ocean is beyond national jurisdictions, and there is no clear international framework governing the exploitation of high-seas marine resources and the protection of the marine environment. The global financial recession has complicated the picture by rearranging national priorities and capacity; sustainable practices compete with priorities in agriculture, infrastructure, energy, health, and education.

Global climate change increases the vulnerability of ecosystems and coastal populations, especially the world’s poor. For the millions of people and local economies highly dependent on marine resources, climate variations increase poverty and food insecurity and lead to loss of livelihood and living space.

International, regional, and national governance issues pose major barriers to the sustainable ocean and coastal agenda. In many countries ocean resource management agencies are chronically underfunded and understaffed. Even in countries with strengthened institutions, addressing management challenges under national jurisdiction in the 200 nautical-mile exclusive economic zone requires expertise, equipment, and vessels for monitoring, control, and surveillance. A high level of technology, capacity, and coordination, supported by broader legal and institutional frameworks at the international and national levels, are needed to address multiple uses and expectations in ever-more-crowded oceans and coasts.

Management of Sustainable Ocean Governance

To overcome obstacles toward sustainable ocean governance and address increasing resource and user conflicts in ocean areas, national governments and international authorities recognize the need to adopt approaches for integrated coastal and ocean management and ecosystem-based management, approaches that shift focus from managing specific, single-sector marine uses to managing multiple uses on an ecosystem basis. The 2002 World Summit on Sustainable Development (WSSD) called for the application of the ecosystem approach by 2010 and the promotion of integrated coastal and ocean management at the national level. These paradigms realize the interrelations of marine ecosystem services and seek to implement holistic, sustainable management and governance.

The WSSD also agreed to achieve a significant reduction by 2010 in the current rate of biodiversity loss at the global, regional, and national levels as a contribution to poverty alleviation, as well as the establishment of marine protected areas, consistent with international law and based on scientific information and including representative networks, by 2012. It further called for the development of diverse approaches and tools, with a focus on the ecosystem approach and the elimination of destructive fishing practices. The 2006 Eighth Conference of the Parties to the Convention on Biological Diversity provided further clarity to marine biodiversity targets, calling for the effective conservation of at least 10 percent of each of the world’s marine and coastal ecological regions and for the protection of particularly vulnerable marine habitats, such as tropical and cold-water coral reefs, seamounts, hydrothermal vents, mangroves, sea grasses, spawning grounds, and other vulnerable marine areas.

Today, however, only about 1 percent of the world’s oceans have been afforded any protection, and renewable marine resources continue to be depleted. Strengthened efforts toward improving international coordination and national capacity are needed to move toward a more integrated approach. Marine protected areas are a useful tool in this approach, though they are but one of the measures needed for sustainable governance. For example, marine spatial planning optimizes the use of marine space to benefit economic development and the marine environment by balancing sectoral interests and the sustainable use of ocean resources, and is emerging as an important decision-making tool. Ecosystem valuation is also critical, providing analyses of the full economic valuation of, for example, coral reef ecosystems. Such valuation estimates both market and nonmarket goods and services to promote better understanding of the economic and societal consequences of environmental change.

Governance gaps and institutional deficiencies related to ocean resource management must be addressed in the coming years. International cooperation, compliance, and enforcement mechanisms must be enhanced. The United Nations Conference on Sustainable Development, held in Rio de Janeiro, Brazil, in June 2012 (Rio+20, being held twenty years after the first Rio Conference) represents a unique opportunity for the international community to
secure renewed political commitment for the sustainable management of ocean resources, assess progress to date, and review remaining gaps in implementation.

Though no common approach effectively addresses all ocean resource management issues, decision makers must pursue strategies fully supported by common tools and techniques, independent science, monitoring and assessment, sustainable finance mechanisms, and methods of evaluation. The three pillars of sustainable development—economic development, social development, and environmental protection—cannot be achieved without the sustainable management of ocean resources. Coordinated and proactive measures must be pursued to ensure the viability of marine ecosystems and resources, thus securing the continued life-support functions humanity receives from the global oceans.

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See also Best Management Practices (BMP); Catchment Management; Coastal Management; Fisheries Management; Large Marine Ecosystem (LME) Management and Assessment; Marine Protected Areas (MPAs); Ocean Acidification—Management; Pollution, Nonpoint Source; Pollution, Point Source

FURTHER READING

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Outbreak species are those species that have the potential to undergo rapid population growth and attain levels that are impossible for them to sustain over long periods. They range from vertebrates (e.g., rodents) to protozoans (e.g., malaria parasites), bacteria (e.g., cholera-causing bacterium), and viruses (e.g., influenza viruses). Because the study of outbreaks of protozoans, bacteria, and viruses is mostly within the realm of epidemiology and medicine, the focus of this article is on outbreaks of animal species only.

Although a small fraction of known species undergoes outbreak events, these have received a disproportionate amount of research attention. Outbreak episodes have alarmed and intrigued humans from early times (e.g., locust outbreaks are mentioned in both the Bible and the Qur’an) because of their unpredictability and great environmental impacts. Outbreaks continue to be a major source of concern in various ecosystems on Earth, in part because of the striking changes recently documented in the behavior of numerous outbreak species associated with global change.

Conditions that Promote Outbreaks

Outbreak events are generated by a wide variety of processes but generally occur when a population experiences increased reproductive rates and reduced mortality as a result of improvements in food or habitat, decreased predation, and favorable climate changes. Nevertheless, despite the great amount of research on outbreak species, definitive answers that explain outbreaks remain elusive. In most cases, a combination of factors is responsible.

Bark beetle outbreaks, such as those of the mountain pine beetle *Dendroctonus ponderosae* and the spruce beetle *Dendroctonus rufipennis*, are strongly influenced by climatic factors. Warmer summers and milder winters favor greater survival and accelerate life cycle development of these species, allowing multiple generations in a season and hence rapid population buildup (Bentz et al. 2010). In addition, drought increases the susceptibility of host trees, enabling beetles to more easily overcome tree defenses. Finally, natural factors (e.g., extensive fire) or land-use practices can create large areas of mature trees that are more susceptible to bark beetle infestation. Favorable climatic conditions are believed to be essential for driving major bark beetle outbreaks, but the pattern and severity of a regional outbreak are also determined by the availability of suitable trees (Raffa et al. 2008; Bentz et al. 2009).

Outbreaks of several rodent species are strongly influenced by increased food availability. Rodent outbreaks in parts of India, Myanmar, and Bangladesh (Singleton et al. 2010), as well as in southern Chile and Argentina (Sage et al. 2007), occurred shortly after synchronous bamboo blooming (called “masting”), which generates sudden availability of massive amounts of seeds in these ecosystems, enhancing rodent reproduction and survival rates (Singleton et al. 2010).

Outbreaks of many agricultural insect pests are promoted by the reduction of natural enemies from insecticide application or habitat destruction. In tropical Asia, the brown plant hopper, *Nilaparvata lugens,*
became a serious outbreak species in rice fields by the 1970s, after intensive application of chemical insecticides aimed at reducing crop damage by another pest, the stem borer. The use of insecticides in rice fields killed most natural enemies of the brown plant hopper, in turn causing large population eruptions and massive crop damage by this species in many parts of Asia (Settle et al. 1996).

**Consequences for Ecosystems and Humans**

Outbreak species cause great impacts on the environment and consequently on humans. Outbreak events can modify ecosystems in multiple ways, from short-term changes in productivity to broad-scale alterations in vegetation cover that may last for centuries. For instance, in western North America, the outbreak of mountain pine beetle *Dendroctonus ponderosae* that started in the mid-1990s has affected a cumulative forest area of 13 million hectares—an area about the size of Greece (Raffa et al. 2008). Through tree mortality, bark beetle outbreaks cause changes in forest structure, species composition, carbon cycles, and hydrology (e.g., snow melts earlier due to more insolation) (Veblen et al. 1991; Pugh and Small 2011). Furthermore, predicted increases in tree mortality caused by future outbreaks are expected to release massive amounts of carbon into the atmosphere, which may create positive feedbacks that enhance climate warming (Kurz et al. 2008).

Outbreaks affecting agroecosystems constitute a clear example of how they can impact humans. Agricultural insect outbreak species significantly reduce crop yields and threaten food security as well as human health and well-being in many regions of the world. The desert locust, *Schistocerca gregaria*, can experience rapid population growth, then aggregate and form swarms that migrate over large distances in search of food. This species affects agricultural lands in about fifty of the poorest countries in the world, especially Africa, the Middle East, and southwest Asia (Roffey and Magor 2003). Even a small fraction of an average size swarm (approximately 1,000 kilograms of locusts) can easily consume in a day an amount of food equal to that consumed by 2,500 people. It is not surprising, then, that during extreme outbreak events, desert locusts can affect the livelihood of nearly 10 percent of the world’s population (FAO 2009).

Species outbreaks can also promote drastic social and political changes. For instance, in 1959 in the Mizoram Hills of India, a bamboo masting event was followed by a large rodent outbreak, which devastated crops and stored foods and led to a widespread famine. This famine contributed to social unrest among the Mizo people, triggering a long civil war that ended in 1986 with the creation of the Mizoram State (Nag 1999; Singleton et al. 2010). Rodent outbreaks can also directly impact human health by transmitting diseases. For example, in southern Chile and Argentina, outbreaks of the long-tailed pygmy rice rat, *Oligoryzomys longicaudatus*, a hantavirus reservoir, generally coincide with increases in hantavirus pulmonary syndrome cases in humans (Toro et al. 1998). This disease has one of the highest mortality rates (30–50 percent) known for acute viral infections (Custer et al. 2003).

Although the majority of species outbreaks are perceived to have negative effects on the environment and human welfare, they can also be essential for ecosystem function and enhance biodiversity. For example, tree mortality caused by bark beetles creates habitat and food resources for several species of invertebrates and wildlife (Raffa et al. 2008; Bentz et al. 2009). Consequently, some outbreak species can be viewed as threats to human livelihoods but also necessary for ecosystem function. Thus, it is important to broadly consider the environmental consequences when attempting to manage outbreak species.

**Global Change and Outbreak Species**

Global changes in climate and human land use are causing variable and complex alterations in the frequency, severity, and extent of outbreaks around the world. Despite the positive effect of current warming trends on populations of many outbreak species, such as mountain pine beetle and spruce beetle, climate
warming is not necessarily favorable for all outbreak species. For example, for the past 1,200 years in the European Alps, larch budworm, Zeiraphera diniana, periodically defoliated European larch, Larix decidua, as their populations underwent regular eight- to nine-year cycles in abundance over that millennia period. No larch budworm outbreaks have occurred since the early 1980s, however, coinciding with the marked warming trends in the late twentieth century (Esper et al. 2007). Recent studies suggest that the collapse of these periodic outbreaks results from a combination of a mismatch in the distribution range of the tree host and the optimal range of the larch budworm (driven by climate), and the asynchrony between foliage flush and egg hatching, which may cause the larvae to starve (Johnson et al. 2010).

Changes in human land use can also cause significant changes in outbreak species’ dynamics. For instance, to recover from Cyclone Nargis, which destroyed crops in Myanmar in 2008, farmers planted rice whenever and wherever they could, resulting in asynchronous and aseasonal planting. Consequently, there was a constant food supply for rodents in the region. In 2010 a large outbreak of the main pest species, the lesser bandicoot rat, Bandicota bengalensis, caused additional crop losses that further threatened food security (Normile 2010). It is believed that the stable food supply promoted an extended breeding season of this rodent, resulting in the outbreak (Singleton et al. 2010).

Changes in land use can also decrease or completely eliminate an outbreak species. The Rocky Mountain locust, Melanoplus spretus, once a common outbreak species across much of western North America, devastated large agricultural areas in the Great Plains until the late 1800s, when it is believed to have gone extinct (Lockwood 2004). The most likely cause of extinction is related to the settlement and agricultural activities in the Rocky Mountain river valleys, which were the permanent habitat of locusts. Changes in the riparian soils, such as flooding for irrigation and soil compaction due to cattle raising, disturbed locust oviposition and development populations (Lockwood and DeBrey 1990).

Managing Outbreak Species

Humans have tried to suppress and manage species outbreaks in diverse ways ranging from direct control of populations to modifications of the environment in hopes of reducing the reproduction and survival rates of the outbreak species. Despite the numerous efforts and resources invested in management, controlling outbreaks remains extremely difficult, with inconsistent results.

Management strategies vary according to the condition of the outbreak population. For instance, when bark beetles attack a few high-value trees in a stand, insecticide application may be a practical method to reduce beetle damage. If an outbreak reaches an entire stand of trees, however, insecticide application will most likely be an ineffective control measure. Thinning and other silvicultural practices aimed at reducing the number of susceptible trees in a stand may be effective for reducing damage when beetle populations are not in an epidemic stage, but these are inadequate for controlling a broader-scale outbreak. Once a bark beetle outbreak has started and affected tens of thousands of hectares of forest, the outbreak becomes virtually unstoppable by human management actions and only collapses due to natural causes, such as weather or food depletion (Raffa et al. 2008).

In this last case, making no attempt to control the outbreak is a management strategy that can save a significant amount of resources.

Indeed, sometimes the most efficient management strategy is to take no action at all and let the outbreak develop and decline by itself. Conversely, for some outbreaks, where human livelihoods and health are at stake, direct control measures on outbreak populations are the only viable option.

Chemical pesticides traditionally have been used to control a wide variety of outbreak species when these are at epidemic levels. A
common method used to control locust outbreaks is to apply synthetic insecticides by land vehicles and aircraft directly on the swarms. These control methods continue to be used; although the chemicals are less damaging to the environment than those formerly used, they are not completely harmless to other species. Successful applications of a biopesticide composed of fungal spores of *Metarhizium anisopliae* in Africa and Australia in the 1990s and early 2000s suggest that this could be an effective and environmentally safe control for locust outbreaks (Lomer et al. 2001; Hunter 2004). Nevertheless, the effectiveness of these methods is debated (Enserink 2004). Despite advances in pesticide knowledge, controlling outbreaks of locusts is still extremely difficult because population eruptions are unpredictable and can occur over large regions, requiring the action of several countries simultaneously. Efforts by the Food and Agriculture Organization (FAO) of the United Nations and other agencies focus on monitoring populations and ecological conditions in source habitats to allow mitigation activities before an outbreak actually occurs.

**Future Directions**

Outbreak events are natural phenomena that play a key role in the dynamics of many ecosystems. Human activity, however, is contributing to the frequency, severity, and extent of many outbreaks around the world. More research is needed in order to understand the fine-scale mechanisms (e.g., species interactions) in conjunction with the broad-scale patterns (e.g., climate and topography) that drive the population dynamics of outbreak species. Interdisciplinary approaches combining diverse disciplines and techniques, such as physiology to understand why locusts swarm (e.g., Anstey et al. 2009) and satellite imagery analysis to monitor changes in locust habitat that may promote outbreaks (e.g., FAO 2009), will likely provide the most useful answers.

Human population growth and expansion of agricultural lands is intensifying the impacts of outbreak species on human subsistence. In some cases, direct measures to control the magnitude of outbreaks can lessen the consequences of outbreaks on the environment and on human welfare. Conversely, direct measures can further increase population levels of outbreak species or cause severe detrimental effects on the environment. Therefore, a thorough understanding of the ecology of outbreak species is critical for guiding mitigation and human adaptations in response to predicted and actual outbreak events, especially in the context of global change.

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See also Agricultural Intensification; Biodiversity; Complexity Theory; Disturbance; Extreme Episodic Events; Food Webs; Global Climate Change; Indicator Species; Keystone Species; Microbial Ecosystem Processes; Population Dynamics; Regime Shifts; Succession

**FURTHER READING**


A permanent agriculture underpins a permanent culture. While permaculture is often thought of as a gardening system, it also encompasses the design of urban areas and ecovillages. Advocates of permaculture argue that most food, water, and shelter requirements can be met from local sources, from systems powered mainly by renewable energy. With limited resources globally, permaculture is a significant tool for the design of sustainable human habitats.

Permaculture is an integration of many skills and disciplines, brought together to design ways of living sustainably in the twenty-first century. The essence of permaculture is ancient in origin—taking inspiration from those civilizations of the world that have survived for thousands of years. The modern permaculture movement emerged as a response to the oil crises of the mid-1970s and continued to respond to various later environmental crises. Permaculture has broadened from its origins in designing sustainable landscapes to be recognized as a tool to help create sustainable societies. It has been used to design ecovillages, community gardens, and city farms. A global network of permaculture activists demonstrates practical solutions through design of sustainable systems, developing a human-centered approach, in which community, economics, legal structures, and the built environment are included.

What Is Permaculture?

Permaculture’s advocates start with the axiom that the Earth’s resources are finite but that infinite external energy is available from the sun. From this follows that the best way to convert sunlight into something people can use is by growing plants. A given area of perennial plants will return food, fiber, and timber. The same area of solar panels could generate more energy, but unlike solar panels, the plants are low cost and can be grown by most people.

Perennial plants are especially useful because they don’t need as much labor as annual crops do and because they use less fossil fuel to grow than monoculture crops such as wheat, rice, maize, and potatoes. The emphasis on perennial plants provided the “perma”(ient) aspect of the name permaculture.

Because humans need more than food and water to survive, permaculturists have developed seven principles, or domains, of sustainable design to guide development of larger systems:

- land and nature stewardship
- built environment
- tools and technology
- culture and education
- health and spiritual well-being
- finance and economics
- land tenure and community governance

Permaculture design gives priority to trees and forests. These long-lived, self-sustaining systems are essential to life on Earth. Through photosynthesis, trees naturally turn the energy of sunlight into food and energy for themselves and wood, fruit, medicine, and fiber that people can harvest, and at the same time they stabilize the landscape. They have had millions of years to perfect this process.

Permaculture design promotes the replication of natural systems to create human-made systems. These systems have to encompass social and economic systems that sustain humans, as well as systems that produce food, fiber, timber, and clean water and air.
Permaculture gardens or field crop areas are designed to make best use of relationships between species of plants and animals and to have multiple types and levels of production. Although some plants don’t grow well in close proximity, most can be planted near each other, and some have beneficial interactions. For example nitrogen-fixing bushes are planted around fruit trees—close enough that their roots intertwine. When the nitrogen-fixing bush is pruned or browsed, some of the nitrogen around the roots is given up and made available to the fruit tree, reducing or avoiding the need to apply fertilizer (Holmgren 1996, 46).

By including plants that grow at different heights, harvest can be obtained from various parts of the garden simultaneously. In a densely planted forest garden, root crops, ground cover, soft fruit from bushes, pome fruit from trees, and berries from vines can all be gathered from a small orchard planted with a variety of species. The soft berry fruit and ground cover such as strawberries are already adjusted to lower light conditions, so they are not restricted by the shade of fruit trees, as long as water and nutrients are available (Hart 1996).

In a permaculture design, both the soil and tanks or dams are used to make sure that there is adequate availability of rain water. Nutrients come from leaf litter and microbial activity—just as in any organic garden—and from making compost. With good fencing and management, animals can be beneficially introduced, and their manures can be composted or used directly by the plants when animals are introduced into the garden. Chickens under fruit trees is one example, but ducks, geese, sheep and pigs can all be used, from time to time, in permaculture systems to assist with nutrient cycling, control of grass, weeds or pests, and for their inherent products such as meat, milk, and eggs. The website of the Hunter & Central Coast Regional Environmental Management Strategy (HCCREMS 2010), a framework developed to address a range of environmental issues within New South Wales, Australia, hosts fact sheets full of information about the composting process.

This conscious design that links plants, animals, and structures such as water tanks has a natural appeal to the homesteader and hobby farmer but has been taken up on a broader scale, for example, where grain farmers and graziers increase the number of hedges and shelterbelts of trees, and where orchardists introduce animals to help manage their fruit crops (Mollison 1988, 60–61; Lillington 2007, 102–104).

The scientific basis of permaculture comes from both natural systems ecology (an ecological approach to agriculture) and from thermodynamics. The work of the US ecologists Howard and Elisabeth Odum, who incorporated the laws of thermodynamics and extended the concept of embodied energy, significantly influenced the Australian permaculture pioneer David Holmgren. In 2001, the Odums published A Prosperous Way Down: Principles and Policies, which proposes ways for the human race to better understand energy—where we get it from, how we use it, and what happens once we have used it—and then to be able to design sustainable systems. In particular, the Odums and permaculture practitioners point to the need to recognize and act upon the fact that people currently rely almost entirely on fossil fuel energy, which is both polluting and finite.

History and Development

The early thinking about permaculture combined ecology, landscape, and agriculture. In the 1970s, the Australian academics David Holmgren and Bill Mollison collaborated on what was to become permaculture. David Holmgren’s graduate thesis became the basis for the book Permaculture One, published in 1978. This book showed how ecology and agriculture could be combined, by conscious design, to create a landscape filled with sustainable food production systems. David Holmgren describes that time:

Permaculture arose from interaction between myself and Bill Mollison in the mid 1970s. We were two (very different) social radicals on the fringes of (different) education institutions, at the global fringes of western industrial society in Tasmania. Bill Mollison as bushman turned senior tutor, in the Psychology Dept.
of the Tasmanian University, attracted large student audiences to hear his radical and original (pre-permaculture) ideas while outraging the academic establishment.

I was a student in the Environmental Design School, a revolutionary “experiment” in tertiary education at the Tasmanian College of Advanced Education. This design school ran for ten years under the inspired leadership of Barry McNeil, a Hobart architect and education theorist. There was no fixed curriculum but a strong emphasis on decision making processes and problem solving. Self assessment, democratic organization and many other elements which radicals within tertiary institutions only dream about were reality within the school. (Holmgren 2006)

**An Ethical Approach**

As the world’s population diversifies from a fossil fuel economy to a renewables-based economy, many people are seeking rules or guidelines. Permaculture has always been an explicitly ethical approach. These ethics are not unique to permaculture and are similar to the precautionary principle (the idea that if the consequences of something—such as the use of nanotechnology—are unknown, it is best to study the issue before acting). As they studied early permaculture societies, Bill Mollison and David Holmgren observed similar ethical bases in them. These ethics are usually expressed as care for the Earth, care for people, and fair shares. (*Fair shares* captures the idea that no one is hoarding and that there are limits to consumption and population growth.)*

Care for the Earth implies that even as they use more of the sun’s energy, humans need to be careful with all the other resources of the Earth. People need more than energy—they also need forests, fish, good soil, minerals, and many other raw materials. The second and third ethics can be seen as derived from the first. Care for people and sharing surpluses spring from the understanding that abundant solar energy is useless unless there is a permanent civilization and a healthy, living planet for plants and animals, including humans.

Robert Hart, another ecological leader of the twentieth century whose work paralleled the development of modern permaculture, also emphasized ethical principles. He developed numerous ethical and sustainable projects and the idea that landscapes can be designed as edible food forests. In his book *Beyond the Forest Garden*, Hart (1996) wrote, “With our present knowledge, there is no technical reason why every woman, man and child on Gaia’s earth should not be adequately fed, clothed, housed and given the opportunity for self realization.”

**Principles**

The book *Introduction to Permaculture*, by Bill Mollison, built on earlier work to codify the basic principles of permaculture. David Holmgren’s subsequent *Permaculture: Principles & Pathways beyond Sustainability* lists twelve permaculture principles and adds the seven domains of sustainable design. The seven domains clarify that permaculture is more than organic gardening. Permaculture design principles are intended to assist in creative thinking—developing integrated solutions where landscape, buildings, people, commerce, technology, health, education, and governance come together.

Permaculture’s ethics and principles are considered when designing for any situation. In contrast, permaculture techniques and strategies have to be applied thoughtfully to each site-specific situation.

**Applied Permaculture**

People who study permaculture generally go on to take action, but the actions are of many kinds. Some privately owned land with energy-efficient housing and self-sustaining landscapes. Some develop ecovillages or
teach courses in permaculture. Others have developed the permablitz—an intensive working party of local volunteers to turn an unused or weedy garden into a food-producing system—and still others have gone on to establish or support city farms and community gardens.

Permaculture teachers and activists encourage others to examine where their food and essential needs come from and to measure (audit) their use of resources. Students of permaculture are encouraged to set targets to reduce their fossil fuel use. This is often done through growing at least some of the food they consume and obtaining most of their food from local and in-season sources to reduce the distance their food travels to their tables.

An audit is a simple self-check—for example, reading the household’s electric, water, and gas meters and setting targets to reduce the amount used or noting the distance driven by car each week and seeking to reduce it. Through these observations, individuals and households better understand and reduce their energy use. Energy is most obviously measured at the electric or gas meter or at the gas pump, but “waste” leaving a home, office, or factory is also a form of energy, and this can be measured as well. Waste is an unused resource—whether it is the very visible weekly household garbage collection or the unseen by-products of manufactured goods. Permaculture design seeks to turn any waste product (an output) into a resource (input) that can be used elsewhere in the system. Backyard chickens are symbolic of permaculture, because they provide important services to people as they convert kitchen scraps and garden waste into eggs, meat, and feathers. Because reducing the distance driven in a private car is a way to improve the local and global environment, permaculture designers aim to create livable cities and towns where more trips can be made by foot, bike, or public transportation, and where fewer trips need to be made in total as basic needs can be met closer to home.

Garden Agriculture

Although permaculture is not simply a gardening system, gardens are often the way that people become more conscious of the need for permaculture’s holistic approach to design. A significant part of being able to sustain our population is for each suburban garden to produce food or fiber, so that whole cities become a kind of farm. Many private gardens in every part of the world contain food plants that are maintained for enjoyment as well as production. David Holmgren and others call this garden agriculture. Redirecting the time and resources devoted to ornamental gardens into growing edible plants can achieve high levels of local production. Human labor rather than machines provide the major power input. There is room in every garden for some annual vegetables, and a large proportion of a household’s fresh food needs can be grown in a small area as long as the soil is fertile.

Transitioning to Resilience

From about 2002 onward, permaculture practitioners were drawing attention to the “peaking” of many key resources, such as oil and gas. American geo-scientist, M. King Hubbert (1903–1989) created and used the models behind peak oil in 1956 to accurately predict that US oil production would peak between 1965 and 1970 (American Heritage Center 2009). This model and its derivatives have described the peak and decline of oil production globally, and they have proved useful in forecasting the peak of other key resources, such as phosphorous (Heinburg 2007).

The first decades of the twenty-first century will be a period of transition to a wider range of energy sources. Permaculture principles are already assisting in the design for ways of living where energy comes from the renewables driven by the sun rather than from fossil fuels.

Permaculture was the starting point for the Transition Town movement that originated around 2005 in Ireland, for example. Those involved in the transition/permaculture movement have recognized that significant change is underway and that embracing change will allow people to be better prepared for an uncertain future.

Permaculture Education

Since the early days of permaculture, courses and workshops have spread permaculture’s ethics, principles, and techniques. Many students who took courses in the 1980s went on to become teachers, and growth has been exponential. The globally recognized educational standard is the Permaculture Design Course (PDC), which consists of at least seventy-two hours of study taught by experienced permaculture teachers. Some parts of the course cover the generic aspects of permaculture; some parts are specific to the region in which the course is taught.

The PDC is unusual for two reasons. First, it is resilient: thousands of such courses have been run in dozens of countries over three decades without the support of an institutional or financial backer. Second, the qualification is widely respected, with hundreds of thousands of graduates, perhaps more than a million. The longevity of the PDC is testimony to the dedication of permaculture teachers and networks of permaculture activists who ensure that those who are “doing permaculture” are also the quality control mechanism for permaculture.
Although the PDC is not recognized by many traditional institutions in most parts of the world, a few do have accredited courses in permaculture. Australia, where a diploma of permaculture is part of the Australian Qualifications Framework, is one country that does support this kind of education.

Can Permaculture Feed the World?

Permaculture is sometimes dismissed as being impractical and too labor intensive. Some critics have questioned whether there is enough scientific data to validate claims that a system of perennial plants will yield more per acre than a single crop such as wheat, maize, rice, or soy. In a 2001 article, Greg Williams, editor of HortIdeas magazine, presents a challenge to the argument that perennial is best. Williams argues that a garden based on a meadow is more productive than one based on a forest. Williams does not take account of extra energy that is needed in the annual immature system of the meadow compared with the perennial mature forest garden approach, however, or the soil loss that occurs whenever tillage happens.

Instead of debating what can be produced from a hypothetical acre, permaculturists prefer to assess how much food a town or city needs and whether it can be produced in or near that urban area. This is one of the reasons permaculture designers focus on cities as well as rural areas—there is much unproductive open space in the city that could be converted into food production.

Advocates of permaculture argue that to eat sustainably, you have to know how much energy goes into the food through the use of tractors, trucks, processing, and packaging. Most foods based on grains are very energy intensive. Permaculturists suggest that people do not need to rely on these annual crops; they simply have to work out what types of food they want to eat and ask if they can be produced in their own garden. Not all food can be produced from a forest garden—most permaculture practitioners who set out to grow a substantial amount of their own food end up with a mix of annual vegetables and tree crops. Beyond that, purchases in the permaculture system come from local growers in preference to food that has traveled a long distance. To be successful in permaculture, people are likely to change their diets substantially.

In addition, some researchers are concerned that permaculture advocates the spread of weeds. Permaculturists, however, stress the use of native plants whenever possible and express concern that genetically modified crops are more dangerous than any weed. Other practitioners say that modern agriculture has damaged the Earth to such an extent that including non-native plants to provide for a sustainable future is more important than preserving current ecosystems.

Toward a More Sustainable Society

Many contemporary thinkers and writers, including permaculturists, are convinced that there has to be some kind of descent from the heights of mass consumption. Since 1700, and especially since 1950, human population has grown enormously, along with a parallel growth in the consumption of the Earth’s natural resources—fossil fuels like oil, coal, and gas, and natural resources like fish, forests, and topsoil. This economic and population growth is a typical response of any group of plants, animals, or bacteria that has access to plenty of food and energy (WRI 2009).

Growth will, however, reach a limit. On a planet with finite physical resources, people will have to make smarter use of renewable energy sources, most of which originate with the sun. The world is constantly growing and changing. The years since 1900 can be compared with climbing a mountain—lots of energy is needed and new tasks have to be achieved at every step. At the top of the mountain, the peak, the climber has to find the way back down, which can often be the more dangerous part.

The sun continues to offer a significant and abundant alternative source of energy, but it comes in a different form from the concentrated supplies of oil, gas, and coal. Permaculture practitioners have shown how people can live in a renewable system powered by sun (the solar economy) rather than oil (the fossil fuel economy).

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See also Agricultural Intensification; Agroecology; Carrying Capacity; Home Ecology; Hydrology; Landscape Architecture; Resilience; Soil Conservation; Urban Agriculture; Urban Vegetation; Water Resource Management, Integrated (IWRM); Urban Agriculture; Urban Forestry

FURTHER READING
Broadly defined, any relationship occurring between organisms in the kingdoms Animalia and Plantae is classified as a plant-animal interaction. Plant-animal interactions are common features of virtually every environment, including all marine, freshwater, and terrestrial biomes. Many such interactions demonstrate evolutionary principles and the myriad ways that species interactions influence the functioning of the biosphere.

Biologists and naturalists have long been fascinated with plant-animal interactions (PAI), the relationships between organisms in the kingdoms Animalia and Plantae. The seeming simplicity of the formulation conceals an enormous number and diversity of ecological relationships and fundamental processes, ranging from the obscure to the ubiquitous. As a result, there is an extensive history of investigation into these often fascinating relationships. The preeminent biologist Charles Darwin wrote extensively about PAI in *On the Origin of Species by Natural Selection*, although many PAI were the focus of a considerable amount of descriptive ecology prior to Darwin’s text. Today, PAI remain a centerpiece within many of ecology’s central theories, including, among others, coevolution and consumer-resource theory.

General Categories of PAI

Plant-animal interactions range from the general to those that are highly specific and involve elaborate evolutionary adaptations. An example of a general PAI is a plant that provides shelter for an animal, such as a tree that provides critical habitat for a nesting bird. Some animals are flexible in their choice of plants; in contrast, some insects are highly specialized, living and laying eggs on only one plant species. In an attempt to categorize and describe the plethora of PAI, biologists further categorize PAI as being commensal, in which one partner benefits while the other is unaffected; antagonistic, in which the interaction is detrimental to at least one partner; or mutualistic, in which both the plant and animal partners both benefit. Interactions are classified by whether an individual partner has more, less, or the same number of offspring as a result of the relationship, in terms of higher or lower fitness. Although the ultimate value is the reproductive success (fitness) of the interacting plants and animals, this can be quite difficult to measure. Thus other metrics such as photosynthetic carbon gain, growth rate, longevity, and survival are often used as surrogate estimates of fitness.

Commensal Interactions

Commensal plant-animal interactions, while straightforward in theory, are somewhat difficult to demonstrate. This is because there is always some question whether an interaction has a completely neutral effect on one of the species involved. Take the previous example of a bird nesting in a tree, which clearly benefits the bird but may or may not influence the tree. If the presence of the nest has no effect on the tree’s growth and reproduction, then the relationship is truly commensal. The bird may eat herbivorous insects that feed on the tree, thus having a positive effect. Alternatively, the nest may block sunlight or weigh down branches away from sunlight exposure, thus having a negative effect. The task of conclusively demonstrating commensalisms in this type of interaction involves experimentally removing nesting birds from some trees and leaving others unchanged and comparing the fitness of the two groups.
Antagonistic Interactions

The most common plant-animal interactions are antagonistic and involve the direct consumption of plants by animals (called herbivores) for food. This general PAI of herbivores involves the direct consumption of plants by herbivores that preferentially use primary producers to the primary consumers. Carnivores, for example, feed on vertebrates and invertebrates that are herbivores. The most basic structural defense of plants is the production of cell walls and fibrous tissues composed of cellulose and lignin, a main component of wood, which are difficult for herbivores to chew and digest. More specialized structures include thorns, barbed spines, hooks, and hairs that protect especially the photosynthetic tissue of plants. Plant chemical defenses, also known as secondary compounds (or metabolites), are metabolic products not necessary for primary growth and reproduction.

The chemistry of plant secondary compounds is complex but well studied because of the deep historical connection with humans. For example, many species of carnivorous plants have evolved mechanisms to trap and digest insects. The Venus fly trap and pitcher plant, which trap and slowly digest insects, are examples of carnivorous plants. Currently there are more than six hundred species of carnivorous plants described, including the well-known Venus fly trap and pitcher plant, which trap and slowly extract nutrients from decomposing arthropods. The first popular scholarly text on carnivorous plants was written by none other than Charles Darwin in 1875.

Not all antagonistic relationships involve animals eating plants. One of the more interesting deviations from the typical pattern is that of the carnivorous plants. Currently there are more than six hundred species of carnivorous plants described, including the well-known Venus fly trap and pitcher plant, which trap and slowly extract nutrients from decomposing arthropods. The first popular scholarly text on carnivorous plants was written by none other than Charles Darwin in 1875.

Plants have evolved a variety of feeding styles to consume plants. For example, insects in the order Hemiptera, such as aphids, leafhoppers, and scale insects, have piercing and sucking mouth parts specialized to suck fluids directly from the vascular system (xylem water and phloem sugars) of the plant. Other insects, such as those belonging to the orders Orthoptera (grasshoppers and crickets) and the larvae of Lepidoptera (moths and butterflies), have chewing mouthparts that allow them to bite and tear leaf material. Vertebrate herbivores also come in a variety of types and sizes, including fresh- and saltwater fish that feed on algae, small rodents that eat parts of leaves, and large-bodied mammalian herbivores that forage on woody plant species (called browsers) or that eat more ground-dwelling herbaceous plants (grazers). Large-bodied mammal herbivores have high-crowned teeth and specialized digestion to facilitate the internal decomposition of plant material. Some ecosystems throughout the world, such as Serengeti National Park in Tanzania, Africa, and Yellowstone National Park in Wyoming, are famous for the abundance and diversity of these large mammal herbivores (also termed megaherbivores). These ecosystems have been labeled grazing ecosystems or browsing ecosystems because of the large proportion of energy transferred from primary producers to the primary consumers, grazers, and browsers.

Plants have evolved a broad spectrum of defenses against herbivory, ranging from tolerance to resistance of defoliation. Herbivory-tolerant plants have high growth rates and are able to reallocate stored carbohydrates to defoliated stems rapidly. Additionally, plants tolerant of herbivory often have architectures that protect carbohydrate-rich storage organs, found below ground or out of the bite range of herbivores. Plants that are resistant to herbivory employ either structural or chemical defenses that deter or even harm herbivores. The most basic structural defense of plants is the production of cell walls and fibrous tissues composed of cellulose and lignin, a main component of wood, which are difficult for herbivores to chew and digest. More specialized structures include thorns, barbed spines, hooks, and hairs that protect especially the photosynthetic tissue of plants. Plant chemical defenses, also known as secondary compounds (or metabolites), are metabolic products not necessary for primary growth and reproduction.
Mutualisms

Plants and animals also engage in a wide diversity of interactions that benefit both partners. One ubiquitous example is pollination, in which animals feed on nectar and pollen from flowers, transferring pollen to other plants, the foundation of the highly successful sexual reproduction of flowering plants. The vast majority of pollinators are insects, but the group also includes birds, bats, rodents, monkeys, and even lizards.

A second type of common mutualism between plants and animals is seed dispersal. Animals benefit by consuming fruits that house the seeds, while plants benefit by having their seeds dispersed long distances by animals, thus increasing their offspring survival probability. Especially for large-seeded plants, long-distance dispersal of seeds would be physically impossible without animal vectors. In this regard, the human domestication of fruits and vegetables may represent one of the most extensive plant-animal mutualisms on Earth. Another type of mutualism involves animals that protect plants from other animal herbivores.

**Ants and Acacias** The PAI involving ants and acacias is one of the best-known examples of mutualism. In tropical woodlands and savannas throughout the world, trees belonging to the genus *Acacia* produce hollow, swollen structures on their twigs that provide shelter for stinging ants. Moreover, these trees also have glands at the base of their leaves that secrete carbohydrate-rich nectar on which the ants feed. Thus, the ants benefit by receiving both a place to live and a source of energy-rich food. This relationship is mutualistic because the trees benefit in return: the ants swarm to attack leaf-eating mammal and insect herbivores. This relationship is very effective and even protects acacias from African elephants, the largest terrestrial herbivore on Earth. Interestingly, the ants do not attack bees that pollinate the acacia flowers because of a chemical released by the plant that somehow prevents ants from approaching during pollination.

**Yucca and Yucca Moths** Another classic example of mutualism is the close association between yucca plants and yucca moths. This interaction is a highly specialized relationship in that each species of yucca has only a few, and sometimes just one, species of moth with which it interacts. Also, the yucca moths are entirely dependent on the yucca plants for their own reproduction, while yucca plants require cross pollination between different individuals and rely completely on the yucca moths for pollen transfer. The moths pack the pollen into the stigma of the yucca plant, ensuring fertilization. While adult moths do not feed, female yucca moths lay eggs in the flowers, and the emerging larvae then feed on the developing seeds. This relationship is classic in that it provides an example of extreme specificity between partners and a relationship in which mutualism and antagonism are balanced in a strong coevolutionary relationship.

**Bees and Bee Orchids** A final example of PAI mutualism is the “deceptive” plant pollination that occurs between bees and bee orchids. Bee orchids have evolved a mechanism to deceive bees into pollinating their flowers through both visual and chemical mimicry of the female bee. The bee orchid produces floral structures that look like a female bee and emits volatile chemical compounds that mimic female reproductive pheromones. Bees are attracted to the plant and are deceived into “mating” with the flowers. In reality, the male bees distribute pollen between individual plants but receive nothing in return.

**Coevolution and the Antiquity of PAI**

The two most species-rich groups of macroscopic terrestrial organisms are plants and insects. One body of theory suggests that the global diversity of these groups is a consequence of their long coevolutionary history, which has persisted since the first invasion on land over 450 millions years ago. Evidence of ancient herbivory, in the form of fossilized insect dung containing plant pollen, begins to show up regularly at the transition between the Silurian and Devonian periods around 420 million years ago. The radiation of the modern angiosperms (the very plant group that is dominant on Earth today) and the modern insect fauna that prey upon them extends from the
Sustainability and Ecosystem Management

The sustainability of ecosystems throughout the world depends on an elaborate network of plant-animal interactions that facilitate ecosystem function (energy flow and nutrient cycling). Habitat destruction and the loss of biodiversity, brought about by rapidly expanding human populations and increased resource consumption, is threatening to unravel these core plant-animal interactions to the detriment of natural ecosystems and at great cost to human societies. The remarkable agricultural prosperity of the human race relies profoundly on the persistence of functioning plant-animal interactions. Chief among them is our dependence on insect-pollinated crops and food production for domestic livestock, such as cows, sheep, donkeys, and goats, which provide meat, natural fibers, and labor. Plant-animal interactions are also at the heart of natural processes that threaten human well-being and economic stability, such as the long history of cataclysmic crop damage by insect pests. Consequently, understanding and preserving the coevolutionary relationships between plants and animals is a critical component for a responsible stewardship of natural and agricultural ecosystems on which humans depend.

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See also Agricultural Intensification; Agroecology; Biodiversity; Boundary Ecotones; Community Ecology, Complexity Theory; Food Webs; Global Climate Change; Human Ecology; Microbial Ecosystem Processes; Mutualism; Population Dynamics; Refugia
Contamination of streams, lakes, reservoirs, estuaries, and coastal zones is increasingly due to nonpoint source (NPS) pollution—that is, pollution from diffuse sources. NPS pollution is typically generated over large land areas, and pollutants typically classified as being from nonpoint sources include nutrients, pathogens, pesticides, chemicals, and sediment. Few regulatory measures exist to reduce NPS pollution, and therefore public involvement and education is key to improving water quality of natural water bodies.

Increasing water contamination due to nonpoint source (NPS) pollution is a major concern throughout the world. Elevated pollutant levels in water bodies create a risk to human health due to exposure to chemicals and pathogenic organisms. The World Health Organization (WHO) has estimated that approximately 3.2 million deaths each year are associated with water contamination, accounting for about 6 percent of all deaths globally (WHO 2011b). An environmental sustainability Millennium Development Goal set by the United Nations is to reduce by 50 percent the number of people without access to safe drinking water by 2015. To achieve this, the World Bank (2011) has estimated that as much as US$23 billion per year will be necessary to develop the infrastructure to provide safe water to the public. Beyond monetary investments, public involvement and awareness of the sources and practices contributing to nonpoint source pollution is key to success in improving water quality.

According to the US Environmental Protection Agency (US EPA 2011a), nonpoint, or diffuse, sources are the leading cause of water quality impairments in the United States. (Impaired waters are those that are too polluted to meet state water quality standards.) NPS pollution is greatly influenced by a combination of hydrology and land management practices. Polluted runoff generated by rain, snowmelt, or irrigation has led to the contamination of many water bodies, including streams, lakes, reservoirs, estuaries, and coastal waters. In the United States (as of 2011), the EPA had assessed 1,550,689 kilometers of streams and found that 53 percent were impaired. More than 98 percent of the assessed areas of the Great Lakes, 81 percent of assessed coastal shorelines, 69 percent of assessed lakes and reservoirs, and 66 percent of assessed estuaries were impaired. (See figure 1 on page 303.) NPS pollution differs from point source pollution in multiple ways: it is variable in time, generated over extensive land areas, event driven, difficult to monitor and regulate, and best reduced through prevention rather than treatment strategies. These factors make mitigation and control of NPS pollution challenging.

Sources of Nonpoint Source Pollution

Nearly every form of land use has the potential to generate NPS pollution. Recent assessments, however, have identified agricultural lands as the leading cause of water quality impairments of rivers, streams, lakes, ponds, and reservoirs (US EPA 2011c), whereas pollution of estuaries is primarily generated by municipal point sources, urban areas, and industries (US EPA 2011d). Common sources of NPS pollution include excess fertilizer and pesticide applications to agricultural and urban lands; toxic chemicals, heavy metals, and hydrocarbons from urban areas; erosion of sediments from poorly vegetated lands (such as agricultural or construction sites) or other susceptible lands; and stream bank erosion. Subsurface
Figure 1. Contamination Level of US Water Bodies

![Contamination Level of US Water Bodies](image)

*Coastal (80.97%)*  
*Great Lakes Shorelines (98.24%)*  
*Streams (53%)*  
*Estuaries (66.14%)*  
*Lakes and Reservoirs (68.98%)*  
*Great Lakes Open Water (99.88%)*


*Shown are the percentages of EPA-assessed water bodies that were identified as impaired as of 2011.*

Drainage of hydric soils has the potential to export high nitrate loads when managed under intensive agricultural production. (The US government defines a hydric soil as one that formed under conditions of saturation, flooding, or ponding long enough during the growing season to develop anaerobic conditions in the upper part.) Additionally, pathogens from livestock, land application of animal manures, leaking septic systems, and wastes from domestic pets and wildlife contribute to water quality impairments and present an immediate risk to human and animal health in some cases.

Agriculture activities such as animal feeding operations, livestock grazing, overland flow or irrigation from cropped land, and tile drainage can degrade water quality. (See figure 2 on page 304.) (Tile drainage is a common practice used to remove excess water from subsurface soil.) Pollutants that result from agricultural NPS pollution are sediment, nutrients, pathogens, pesticides, metals, and
salts. These pollutants are typically released from the land surface and transported by runoff into water bodies. For example, erosion from agricultural land transports enormous amounts of sediment, and the influx of these particles into nearby streams, lakes, or wetlands can eliminate aquatic organisms through destruction of habitat or direct impacts such as clogging fish gills. Runoff with excessive nutrients also leads to degraded water quality. For example, to ensure maximum crop yields, land managers may apply abundant nutrients, including nitrogen, phosphorus, potassium, and manure. Excess nutrients that are not utilized by plants, or that are applied prior to rains, are potentially carried by the runoff from agricultural land to water bodies, where high nutrient levels cause algae blooms to proliferate, leading to an uninhabitable environment for aquatic life. Other NPS pollutants, such as pesticides, can also be toxic to aquatic life.

Livestock production is another source of NPS pollution. In the United States, animal feeding operations produce hundreds of millions of tons of manure each year. The animal production industry in the United States accounts for 55 percent of soil and sediment erosion, 80 percent of antibiotic usage, and more than 30 percent of the total nitrogen and phosphorous loading to national drinking water resources (Pew Commission on Industrial Animal Production 2011). For instance, effluent from animal feeding operations (AFO) contributes considerable amounts of pollutants such as pathogens, nitrogen, phosphorous, sediments, hormones, and antibiotics to ambient water bodies (US EPA 2011e). In addition, livestock overgrazing can lead to increased erosion, invasive plants, and degraded or eliminated riparian vegetation (that is, vegetation along the banks of waterways). An increase in livestock populations results in greater volumes of nitrogen- and phosphorus-rich animal waste, which is typically disposed of through application as fertilizer to crop and pasture-lands. When such manure is applied to the land based

Figure 2. Nonpoint Sources of Pollution in an Agricultural Watershed

Photo credits: Pramod Pandey; Charles Velasquez; Ray Sims; David Westhoff; Andrew Paxson; Kendal Agee.

Agricultural activity is a major source of nonpoint source pollution. Overland flow from agricultural land, the outflow from tile drains from cropped land, open feedlots, wildlife, and confined feeding operations all cause nonpoint source pollution of water bodies.
on the crop nitrate requirements, a buildup of soil phosphorus is often observed, increasing the risk for transport of nitrogen- and phosphorus-enriched water to surface waters.

In addition to agriculture, urban runoff can generate NPS pollution. More than 50 percent of the world’s population lives in urban areas, and the migration from rural areas to cities continues. Insufficient urban infrastructure makes cities a central point for environmental health concerns. According to the EPA (as of 2011), almost 56,000 kilometers of US streams and significant portions of US coastal shorelines were impaired due to storm water discharges and other urban runoff. An increase in impervious surfaces and storm water discharges has resulted in increased associated pollutant loads (such as solid waste and chemicals) into adjacent water bodies. Current trends show that future human populations will increasingly be concentrated within urban areas. Therefore, the urban environment will play a critical role in water quality and public health.

Another nonpoint source of pollution is abandoned mining operations. Runoff from these lands has great potential to impact water bodies. Mining operations in the mid-Atlantic region have led to acidification of surrounding water bodies. Runoff from abandoned mines can transport solids, oils, minerals, and metals such as zinc and arsenic to water bodies. Particles containing sulfur can lead to the formation of sulfuric acid and iron hydroxide, which can dissolve heavy metals such as copper, lead, and mercury. This acts to change water body chemistry, making the environment toxic to aquatic life and unsuitable for public and industrial uses. According to the EPA (as of 2011), about 8,240 kilometers of streams have been contaminated in the eastern United States by runoff from abandoned coal mines; coal mines in the mid-Atlantic region alone caused low pH values in 7,656 kilometers of streams.

Nonpoint source pollution from silviculture (the development and care of forests) can also cause significant water quality problems. The movement of heavy machinery in forested areas can remove vegetation, leading to increased erosion. Activities such as logging can generate considerable amounts of NPS pollution, particularly sediment, due to removal of streamside vegetation, road construction, and timber harvesting. For example, in the Lake Superior drainage basin, approximately 75 percent of the basin is forested, and 50 percent of the basin has highly erodible red clay soils. According to the EPA (as of 2011), approximately 9 percent of the water quality problems in assessed streams were caused by forestry activities. Road construction and road use contributed up to 90 percent of the total sediment from forestry operations. In addition to upland erosion, silviculture activities modify or remove riparian vegetation, which can also harm aquatic species and wildlife through limiting food and habitat.

**Impacts of NPS Pollution**

According to the EPA, nitrate is the most prevalent agriculture pollutant in drinking water, and approximately 1.5 million people are exposed to elevated levels in their drinking water wells (US EPA 2011f). In water bodies, algal growth from elevated nutrient levels results in low oxygen levels. This process, called eutrophication, can cause fish kills and reduced diversity in aquatic life. Nitrogen and phosphorus from agricultural land can contribute to eutrophication and associated algae blooms.

Nonpoint source pollution has been associated with over four hundred hypoxic zones in the world. The largest US hypoxic zone, in the Gulf of Mexico, is about 17,000 square kilometers. Hypoxia is due to low oxygen in water (less than 2 milligrams per liter), at which aquatic life cannot survive. Recent increases in the occurrence of hypoxic zones has been related to anthropogenic activities such as intensive use of fertilizers on agriculture land, erosion of soils with nutrients, and discharges from sewage treatment plants. For instance, the hypoxic zone in the Gulf of Mexico is primarily caused by excess nutrient loading to the Mississippi River. Studies have shown that in the last half of the twentieth century, nitrogen and phosphorus concentrations in the Lower Mississippi River have increased considerably, and this has been attributed to the increased use of nitrogen and phosphorus fertilizers on cropped land. The hypoxia in the Gulf of Mexico is of particular concern because Gulf fisheries generate billions of dollars annually.

Another potential health risk from NPS pollution is exposure to waterborne pathogens. The World Health Organization (WHO) has estimated that 88 percent of the global disease burden is attributable to contaminated waters, particularly pathogen-contaminated waters. About 62.5 million people suffer each day from diarrheal diseases, and intestinal worms infect about 10 percent of the population in developing counties. Water contamination by pathogens causes potential health risks to humans all over the world. For instance, more than 200 million people are infected with schistosomiasis, and more than 300 million suffer from malaria that is linked with contaminated waters (WHO 2011c). In developing countries in Africa, waterborne pathogens infect millions. Guinea worm disease, a parasitic infection caused by *Dracunculus medinensis*, begins when a person drinks contaminated water infested by the larvae of the guinea worm.

Even in developed countries such as the United States, pathogens are a major source of pollution in water bodies.
Approximately 900,000 illnesses and 900 deaths occur in the United States each year because of exposure to waterborne pathogens (Arnone and Walling 2007). Human diseases such as gastrointestinal illness, including vomiting, diarrhea, and fever, have been related to high levels of pathogenic bacteria in recreational waters. In the United States most beach closings result from elevated level of pathogens; the source is often untreated sewage. Waterborne pathogen contamination is a threat not only to human health but also to wildlife, livestock, and aquatic life. For example, massive frog die-offs in the US states of Colorado and Arizona and the Australian state of Queensland have been attributed to the chytrid fungus—a pathogen that lives in fresh water. In the US state of Massachusetts, bacterial contamination in shellfish and recreational waters is blamed for the loss of millions of dollars each year in the local economy.

Future Challenges

In addition to natural water bodies (i.e. streams, lakes, estuaries, coastal waters), designing new water resource structures has serious implications for water quality. Large reservoirs may increase pathogen contamination, particularly schistosomiasis. A study by the parasitologist Alan Fenwick (2006) reported that the development of water resources, particularly in Africa, has increased the transmission of waterborne diseases. For example, the Gezira Irrigation Scheme and the Sennar Dam, the first major dam across one of Africa’s great rivers (the Blue Nile), have been associated with increased schistosomiasis. Approximately 103 million out of 779 million people infected by schistosomiasis live in close proximity to large reservoirs and irrigation schemes (Daszak, Cunningham, and Hyatt 2000). It has been noted that such projects increase suitable aquatic snail species, which are the intermediate hosts of the larvae of schistosomiasis.

Contamination of water by high levels of nitrogen and phosphorus is an increasing concern in the United States. According to the EPA, about 50 percent of US water bodies are negatively impacted by nitrogen and phosphorus pollution (US EPA 2011h). Excessive levels of nitrate in drinking water over time can increase risk for thyroid cancer in women. Nitrate in drinking water also puts young children at risk for blue-baby syndrome (methemoglobinemia), where the oxygen-carrying capacity of the blood is lowered, resulting in bluish skin color. Conventional drinking water treatment processes cannot remove nitrate. Instead, expensive treatment methods are required such as reverse osmosis and biological denitrification (Elyanow and Persechino 2005).

According to the EPA, the costs to pollutant sources for implementing total maximum daily loads (TMDL) to control contamination are expected to be between approximately $1 billion and $3.4 billion per year (US EPA 2001).

Changing environmental conditions may alter the virulence of existing diseases and increase the possibility of new diseases. Many pathogens currently exist in a dormant stage in water bodies, but an increase in water temperature may alter this. Therefore, future work is needed to increase our understanding of how changes in climate and rainfall will impact NPS pollution. Only 2.5 percent of all water available on Earth is fresh, and approximately 15 percent of the world’s population lives in regions with limited water supply. Access to freshwater is critical to support life. Improving water quality is a major but necessary challenge to protect human and animal health and quality of life. In summary, reduction of nonpoint sources of pollution is needed to protect public health and provide clean water to humans and animals.

Controlling NPS will require interdisciplinary collaboration among various disciplines such as hydrology, soil science, agriculture, ecology, environmental science, and engineering in order to develop land and water management practices which are eco-friendly and sustainable. In addition to scientists, it will require public involvement and education, which is a key to improving the water quality of our ambient water bodies.

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See also Agricultural Intensification; Agroecology; Ecosystem Services; Eutrophication; Groundwater Management; Irrigation; Large Marine Ecosystem (LME) Management and Assessment; Marine Protected Areas (MPAs); Pollution, Point Source; Rain Gardens; Road Ecology; Soil Conservation; Stormwater Management; Water Resource Management, Integrated (IWRM)
FURTHER READING


Point source pollution is an identified source from which contamination enters air, water, and soil. These sources may be natural, such as volcanoes, which release hazardous ash and gases into the air. Anthropogenic, or human-made, point source pollution can take forms as diverse as chemicals, greenhouse gases, noise, light, vibration, nuclear material, viruses, odors, or cigarette fumes, and may be stationary, mobile, temperate, or constant.

The Industrial Revolution significantly improved the welfare of people around the world. The trade-off, however, is increased air, water, and land pollution. The word *pollution* brings to mind images of a chemical plant dumping contaminated water into a river or the large chimney of an industrial enterprise spewing smoke into the air, but pollution may be natural, as well. Pollution that comes from a known source (natural or not) is called *point source pollution* (PSP).

Between 1850 and 1900 industry pumped into the atmosphere large amounts of metal-containing emissions carrying nickel, cadmium, copper, zinc, and lead. From 1900 to 1980 the rates increased dramatically. (See table 1.)

Human activities dating back to ancient times have caused PSP (Borsos et al. 2003). Natural objects such as volcanoes can be point source pollutants, too. Different substances, sources, and harmful agents make up point source pollution. A few key concepts and definitions are helpful in understanding PSP.

*Pollution* is contamination of soil, air, water, and space (i.e., outside the Earth’s atmosphere). Scientists consider pollution harmful when it exceeds normal (natural) background levels or accumulates harmful substance(s) in an organism over time. Pollution can be incautious (accidental) or intentional contamination.

A *point source of pollution* is a spot that produces harmful substances that can damage living organisms. These substances impact human health and disturb natural rhythms of plant growth and bird migration. A point source releases pollution into the atmosphere, into marine and freshwater environments, on the land’s surface, or into the ground. The USSR State Committee on Standards (1977) describes point source pollution as “the source that emits air pollutants from [a] fixed aperture.” A drain pipe or channel may discharge sewage into the water. Effluents from an industrial setting may come through pipes, ditches, sediment bowls (traps that collect pollutants), holes or fissures, or waste oil collectors. A point source pollutant can also be an individually identified source of pollution, as disparate as an aircraft carrier, an oil tanker, or a mine.

*Nonpoint source pollution* (NPS pollution) is a source of diffusive contamination that cannot be detected precisely. According to the US Environmental Protection Agency (2011b), “NPS pollution occurs when rainfall, snowmelt, 

<table>
<thead>
<tr>
<th>Metal</th>
<th>Amount Produced from 1850 to 1900 (in metric tons)</th>
<th>Amount Produced from 1900 to 1980 (in metric tons)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nickel</td>
<td>218</td>
<td>11,000 (51 times)</td>
</tr>
<tr>
<td>Cadmium</td>
<td>345</td>
<td>2,760 (8 times)</td>
</tr>
<tr>
<td>Copper</td>
<td>1,633</td>
<td>9,800 (6 times)</td>
</tr>
<tr>
<td>Zinc</td>
<td>15,422</td>
<td>123,000 (8 times)</td>
</tr>
<tr>
<td>Lead</td>
<td>20,000</td>
<td>180,000 (9 times)</td>
</tr>
</tbody>
</table>

*Sources: Nriagu (1979 and 1994), cited in Yarime (2003).*
or irrigation runs over land or through the ground, picks up pollutants, and deposits them into rivers, lakes, and coastal waters or introduces them into ground water." NPS substances may include sediments of soil erosion as well as fertilizers, pesticides, salt, and manufacturing by-products.

Categories

Classification of point sources of pollution is a complex task because of the many factors and spheres involved. Point source pollutants can be natural or anthropogenic (human made). They can come from industrial, agricultural, residential, or individual sources. They may originate as light, noise, or temperature changes. Dust, vibrations, gases, heavy metals, pesticides, viruses, bacteria, or radioactivity are point source pollutants. Air, water, and soil/land may emit them. The time scale of the activity may be permanent or temporary. It may occur outdoors or indoors. It may be stationary or mobile.

Most point source pollutants are harmful to humans but substantially affect the environment, too. Cigarette smoke is a point source pollutant that emits fumes, odors, and heavy metals or chemically active radicals. Snoring is an indoor point source of noise pollution. Noise pollution created by felling equipment at forest harvesting sites disturbs wildlife habitats; oil spills harm water wildlife, birds, animals, and even the microclimate because of their effects on temperature and gas exchange between the water and the atmosphere (Reimers 1990).

Point source pollution breaks down into two major types: natural, such as a volcanic eruption, and anthropogenic, such as oil spills and nuclear reactor radioactivity.

Volcanic Eruptions

Major volcanic eruptions are a well-known source of pollution (Smith 2004). Several small volcanic eruptions in the nineteenth century produced a substantial amount of ash and dust in the atmosphere (Heidorn 2000). These events started with La Soufrière on Saint Vincent in the Caribbean and Awu on Sangihe Islands, Indonesia, in 1812. Volcanoes erupted on Suwanose-jima on the Ryukyu Islands, Japan (1813), and Mayon in the Philippines (1814). On Sumbawa Island, Indonesia, the Mount Tambora eruption of 1815, the largest eruption in recorded history, created the "year without a summer" in 1816. The eruption significantly affected global weather that year, especially in the Atlantic part of North America and western parts of Europe, causing crop failures and food shortages. The culmination of these volcanic eruptions created an environmental disaster that destroyed rice, buckwheat, and other crops that led to a famine in China's Yunnan Province (Yang, Man, and Zheng 2005). Krakatoa, a volcanic island between Sumatra and Java, erupted in 1883, making what is believed to be the loudest sound in recorded history. A volcanic eruption came from the Icelandic ice cap Eyjafjallajökull in 2010. The eruption disrupted airline travel schedules for several days because the ash tail stretched over many kilometers (BBC News 2010; O’Sullivan 2010).

Major air pollutants from volcanoes are both solid, like tephra (ash or dust particles and small pyroclastic rocks, made of volcanic ash), and gaseous. Volcanoes emit a spectrum of volcanic gases, including sulfur dioxide, that affect air quality. Ash fall causes breathing problems and damages crops.

Oil Spills

Oil spills are human-made pollution in marine, freshwater, and land spheres. The point source pollutants come from tankers, platforms, oil wells, or refinery plants. The Kuwait oil fires during the Gulf War in 1991 are a notable example (although they could be considered a nonpoint source, too). Oil spills from the Torrey Canyon in 1967, the Atlantic Empress in 1979, and the Exxon Valdez in Alaska’s Prince William Sound in 1989 created environmental havoc, releasing hundreds of thousands of tons of oil, killing bird and marine life, and leaving lasting ecological and habitat damage. The 1994 pipeline accident in the Komi Republic of Russia caused one of the largest terrestrial oil spills of all time when more than 102,000 metric tons of oil covered a large area of land (Kireeva 2007). The most recent large spill occurred in the Gulf of Mexico in 2010. Instead of a ship colliding with an object, an explosion caused a huge fire and sank the Deepwater Horizon platform (Business Insider 2010; Robertson and Krauss 2010).
Light

City and town lights interfere with astronomical observations, disorient migratory birds so that they fly into buildings, and affect darkness-related night-foraging wildlife such as bats and mice. This “permanent full moon” makes animals vulnerable to predators and reduces time for their foraging. Chicago and other cities employ “lights out” programs, saving more than ten thousand birds each year by weakening building lights during bird migratory seasons (Hirji 2010).

Thermal

Thermal point source pollutants such as factories or ships release too much waste heat into fresh- or marine water. Heat reduces the amount of oxygen in the water, threatening or killing aquatic life and stimulating plant growth.

Nuclear

Nuclear power plants are a potential source of point source pollution. Plants at Three Mile Island (United States, 1979), Chernobyl (USSR [Ukraine], 1986), and Fukushima (Japan, 2011) have all had accidents, although the accident at Three Mile Island was far less serious than the latter two. The major risk from nuclear pollution is radioactivity, which has no smell, color, or taste but is extremely damaging to humans and animals and can lead to radiation poisoning, cancer, or death.

Vibration and Noise

Many phenomena cause vibration and noise. People often do not consider noise to be a source of pollution, but continuous noise from construction work and traffic, for example, may cause such physical effects on humans as high blood pressure, sleep disturbance, and hearing loss (US EPA 2011a). Ireland (1992), India (2000), and Singapore (2007), among other countries, have passed noise pollution legislation that regulates working hours and construction noise. These laws mostly protect residential areas; they address societal needs but do not protect the natural environment.

Even renewable energy production can cause noise pollution. Wind farms create noise and vibrations that are transferred into the ground and impact biodiversity, including birds, bats, squirrels, and even aquatic biodiversity if located on the seashore. Such noise can also widely result in “habitat degradation, and can affect regional ecological integrity” (Government of Alberta 2011, 1).

Chemical

Chemical point source pollution is the most widely spread pollution and has deep roots in the history of industrial development. The invention of boilers to power industry as well as of internal combustion engines created early chemical point source pollution. Chemical point source polluters are enterprises that emit gases, oxides, and by-products including heavy metals. Such pollutants are well recognized in this era of climate change discussion. Major greenhouse gas emitters include power-generating plants burning fossil fuels, as well as manufacturers of chemical ingredients for fertilizers, detergents, and the like. Chemical enterprises leak heavy metals, acids, or related pollutants. Most modes of transport, including cars and airplanes, are mobile point source polluters. Smog is nonpoint source pollution, but coal combustion for heating houses and producing power for manufacturers makes every chimney a point source pollutant.

Odor

Dumps, farms, and manufacturers emit odors such as ammonia, animal waste, or other unpleasant smelling by-products. Slaughtering animals and tanning hides create strong odors. Tourists who wish to see the process of leather goods production in Marrakech, Morocco, hold a mint stem under their noses in order to reduce the strong smells from the tannery.

Responses

The Netherlands, the United States, Korea, Japan, and China, among other countries, have passed legislation to regulate point source pollution. Legislative, economic, educational, and health care approaches can prevent or mitigate the problems created by point sources of pollution. So called end-of-pipe technology controls environmental contamination by the installation of the appropriate equipment at the polluting source to reduce the volume of emissions into the air and water during manufacturing. Absorption, adsorption, condensation, incineration, and
selective diffusion through membranes constitute other industrial technologies (Yarime 2003). Scientists and manufacturers have created “clean” or “green” technologies to avoid the creation of pollutants during manufacturing processes (McMeekin and Green 1995; Caprotti 2009; Cleantech Group 2010; UNEP 2011).

The Future

An interesting historical observation sheds light on point source pollution:

One of the reasons the tribes of early history were nomadic was to move periodically away from the stench of the animal, vegetable, and human wastes they generated. When the tribesmen learned to use fire, they used it for millennia in a way that filled the air inside their living quarters with the products of incomplete combustion. . . . After its invention, the chimney removed the combustion products and cooking smells from the living quarters, but for centuries the open fire in the fireplace caused its emission to be smoky. (Boubel et al. 1994, 3)

As a species, we have struggled with anthropogenic point source pollution for millennia. Because of the many sources of point source pollution, governments, industry, and consumers must make a worldwide effort to address the concerns and provide the answers. The changes wrought by point source pollution, including global climate change, make these issues all the more pressing.

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See also Agricultural Intensification; Brownfield Redevelopment; Eutrophication; Extreme Episodic Events; Global Climate Change; Groundwater Management; Irrigation; Light Pollution and Biological Systems; Marine Protected Areas (MPAs); Ocean Acidification—Management; Pollution, Nonpoint Source; Safe Minimum Standard (SMS); Shale Gas Extraction; Waste Management; Water Resource Management, Integrated (IWRM)

FURTHER READING


Population dynamics reflects changes in the numbers of a single species living in a specific area over time. Population dynamics is complex; in reality, it cannot be understood without knowing the processes and context at species, ecosystem, and landscape levels. Models of population dynamics are widely used as sound bases for ecosystem management to achieve the goal of environmental sustainability.

A population is a group of individuals of a single species who live in the same place and interact with one another. Ecologists use the term population dynamics to refer to changes in population abundance through time and space under natural or anthropogenic influences. Population dynamics are measured by changes in population size (the total number of individuals), population density (the number of individuals in a certain space at a given time), population dispersion (the spatial arrangement of individuals within a population), and population age distribution (the proportion of individuals in each age group).

At a basic level, population dynamics can be grouped into three major types: exponential growth, logistic growth, and metapopulation dynamics. These categories are not mutually exclusive, and the same population can experience each of them at different times. Why populations grow or decline in size provides insight that is key to sustainable resources management and conservation of biodiversity.

**Population Change**

Natural growth of a population is affected by four components: birth rate, death rate, immigration (the movement of members into a population), and emigration (movement of members from a population). The change in population size from one time period \( t \) to another \( t + 1 \) can be described by a mathematical equation that embodies these four components:

\[
N_{t+1} = N_t + (B - D) + (I - E)
\]

where \( N_t \) is a population size at time \( t \), \( B \) is the number of individuals born, \( D \) is the number of individuals dying, \( I \) is the number of immigrants, and \( E \) is the number of emigrants between time \( t \) and time \( t + 1 \).

Population growth is regulated by limiting density-dependent factors. Density dependent factors typically involve biotic factors, such as the availability of food, parasitism, predation, disease, and migration. In many cases density-independent factors can determine population size. Density-independent factors limit the growth of the population regardless of the density of the population. These factors include changes in environment (temperature, sunlight, etc.), natural disasters, or human interference.

If conditions are favorable, a population can increase rapidly until density-dependent factors inhibit its growth. No population can continue growing forever. Even organisms that reproduce very slowly (elephants, rhinos, and whales, among others), would outstrip their resources if they reproduced indefinitely. Most often density-dependent regulation works through competition for survival.

**Exponential Growth**

This model of population growth assumes that essential resources (food, space, water, etc.), are unlimited and the environment is constant. When biotic and abiotic conditions are favorable and individuals in a
population reproduce without synchrony at varying times, changes in its size occur continuously, and a population is said to be undergoing exponential growth. Based on density independence, rate of growth remains constant and can be described by the following logistic equation (note that “d” is basically shorthand for “the change in”):

\[ \frac{dN}{dt} = rN \]

The rate of growth (\( \frac{dN}{dt} \)) is directly proportional to the size of population (\( N \)) at any instant time; \( r \) is the intrinsic rate of increase, and it provides a measure of how rapidly a population can grow. This equation predicts the size (\( N \)) of an exponentially growing population at any time (\( t \)), if we have an estimate for \( r \) (different values of \( r \) describe different exponential curves) and know the initial population size. In exponential growth models, births, deaths, emigration, and immigration take place continuously. This is a good approximation for the growth of most biological populations. When plotted on a graph, exponential growth pattern forms a J-shaped curve.

In contrast to the exponential growth, geometric growth occurs when individuals in a population reproduce in synchrony at regular time intervals. Most populations (including people) experience discrete reproductive pulses with overlapping generations. In such case a population grows in size by a constant proportion of individuals during discrete time periods. Geometric growth also can be represented in an equation:

\[ N_{t+1} = \lambda N_t \]

where \( \lambda \) (lambda) is the finite rate of increase; and \( t \) represents discrete time periods. Depending on the value of \( \lambda \) or \( r \), a population growth pattern might be different (1) when population is stable (represented by \( \lambda = 1 \) and \( r = 1 \)); (2) when population is shrinking (represented by \( \lambda < 1 \) and \( r < 1 \)); and (3) when population is growing (represented by \( \lambda > 1 \) and \( r > 1 \)). When plotted on a graph, the geometric growth pattern, like the exponential growth pattern, forms a J-shaped set of points (slow to rise, then increasing sharply on the graph). Because geometric and exponential curves overlap and are almost similar in form, both types of growth are occasionally lumped together for simplicity as "exponential growth."

A population explosion (sometimes called population bomb) may result when there is a rapid growth of a species by multiplication. Following a population explosion, the abundant species may take over the ecosystem and change its dynamics. Populations exhibiting exponential and geometric growth under conditions of unrestricted resources, absent competition, or other limitations are most common among invasive and expanding species. An example of such population dynamics might be the rise of the gray seal (Halichoerus grypus) pup production on Sable Island in Nova Scotia, Canada. It has been monitored since the early 1960s, and the recent estimate indicates that pup production in this population, now the largest gray seal colony in the world, has been increasing exponentially at an annual rate of 12.8 percent for four decades (Bowen, McMillan, and Mohn 2003).

Knowledge of the described patterns of exponential population growth (see the last two equations above) is very useful in natural resource management and biodiversity conservation, particularly when age-structured models, which include information about both the rate of population growth and the distribution of individuals of different ages, are utilized. They help us to calculate the potential sustainable harvest rate (number of individuals removed from a population relative to the number available) and determine minimum population size (the minimum number of individuals in a given locality that could be expected to survive in the long-term).

**Logistic Growth**

No population continues to grow indefinitely. Logistic growth is found in a population that increases initially and then stabilizes at a maximum population size. Logistic growth is based on several assumptions: age distribution is stable, there is no immigration or emigration, relationship between size and the rate of growth is linear, and density depresses the rate of growth. The logistic growth model accounts for carrying capacity. Each population has its own carrying capacity—the maximum number of individuals a given environment can sustain. The logistic model of population growth can be defined as an equation:

\[ \frac{dN}{dt} = rN \left(1 - \frac{N}{K}\right) \]

where \( \frac{dN}{dt} \) is the rate of change in population size at time \( t \); \( N \) is the number of individuals in the population...
at a given time; \( r \) is per capita population growth rate under ideal conditions; and \( K \) is carrying capacity.

The rate of population growth is based on density-dependent effects and is susceptible to change. When the number of individuals in a population is small, most resources are unutilized (\( dN/dt \approx r \)), with “\( \approx \)” meaning approximately equal to. A population increases until \( N \) is equal to \( 0.5K \), after which the rate of growth declines as density increases because resources (food, space, water, etc.), begin to be in short supply. Once a population reaches its carrying capacity, most resources are utilized, growth rate is zero, and population size does not change (\( N = K, dN/dt = 0 \)). The sigmoidal or S-shape curve is a characteristic feature of logistic growth.

A population extinction (sometimes called population collapse) occurs when mortality, for whatever reason, is higher than natality (birthrate). If a population is too small (\( N > K \)), it may decline, and eventually it can become extinct. Decrease in population size can further lead to the emergence of poor genetic pools, causing weakness and a further decline in population; this process is known as the Allee effect. A population can be driven to extinction not only by the demographic and genetic factors described above, but also by environmental fluctuation (floods, fires, windstorms, etc.) and disease outbreaks.

A classic example of population collapse is the decline of the northern Atlantic cod (\( Gadus morhua \)) in Newfoundland and Labrador, Canada. For centuries it made up the most lucrative fish trade between North American and European countries, but since 1984 cod stock has declined by 97 percent, with a loss of 17 percent of the harvest per year, due primarily to overexploitation (Hutchings and Myers 1994). In 1992, the Canadian government declared a moratorium on cod fishing because of devastatingly low cod stocks. In fact, even twenty years later cod had not achieved a discernable recovery in abundance (Hutchings and Reynolds 2004). The risk was posed by allowing populations to decline to extraordinarily low levels. This clearly demonstrates the need for the accurate estimation of per capita population growth rates, which can have important socioeconomic and ecological consequences, even though ecosystems managers often ignore it.

Ecological footprint, which is the total area of productive ecosystems required to support a population, was suggested to measure the environmental impact of population (Rees 1992). This concept is being used in relation to human population growth. The study of human demography goes back to the eighteenth century. Thomas Robert Malthus (1766–1834) was an early economist who reached the conclusion that human population tends to grow until it outstrips its available food supply. Historically the human population has increased even more rapidly than expected from exponential growth. Using the ecological footprint approach, it has been estimated that Earth could support about 4.6 billion people for a long time at a moderate level of consumption (Ewing et al. 2009), but human population will approach 7 billion in 2011, and it is projected to increase from 8 to 10.5 billion by 2050 (UNDESA 2009). The degree to which the growing human population affects the environment depends on the amount of resources used by each individual and the total number of individuals. The ability to sustain exploitation of natural resources by the human population to meet the growing needs of current and future generations is a main focus of environmental sustainability.

**Metapopulation Dynamics**

A metapopulation is a group of spatially separated populations of the same species that are linked by dispersal, or species movement, and interact at some level. The classic concept of metapopulation dynamics, which is a balance between colonization and extinction, was formulated mathematically by Richard Levins (1969):

\[
dP/dt = cP(1 - P) - eP
\]

where \( P \) is fraction of habitat patches that are occupied at any given time (\( t \)); \( c \) is a rate of patch colonization; and \( e \) is a rate of patch extinction. Patch is a relatively homogeneous unit of the landscape that changes and fluctuates over time.

The metapopulation concept derived from thinking about the influence of area and isolation on colonization and extinction. Habitat islands suffer periodic, predictable extinction; however, they might be recolonized by dispersers from neighboring islands. If migration is greater than extinction, population persists. This formed the current framework for source-sink system and metapopulation dynamics (Hanski and Gilpin 1991). Source is a high-quality habitat that produces surplus and supports long-term population (growth rate \( > 0 \)); sink is a very low-quality habitat that, on its own, is not able to support a population that cannot replace itself without immigration (growth rate \( < 0 \)). Therefore, source-sink dynamics means that a source population grows to maximum density, and surplus individuals migrate to a sink population. Growth models indicate that source populations display increased growth rates during establishment and increased stability around their carrying capacities (Neal 2004).

Metapopulation theory suggests that habitat variability is important for population persistence. Movement between sources and sinks of a metapopulation could be as vital as resource distribution, which is
currently a primary focus for landscape managers and conservationists. This idea was developed further in studies on a highly endangered species, the Asian tiger (*Panthera tigris*). A group of ecologists used long-term demographic data combined with a GIS-based model (a geographic information system merges cartography, statistical analysis, and database technology) to identify potential corridors to increase dispersal, develop strategically placed transit refuges, and provide recommendations for off-reserve land management (Wikramanayake et al. 2004).

Protecting source populations while subsequently increasing dispersal rates between sources and sinks through constructing conservation landscapes was proved to be an effective conservation method.

**Implications**

Population dynamics is the branch of ecology that studies changes in population abundance under the influence of biological and environmental processes. Population dynamics forms a theoretical basis for understanding and managing many environmental issues, particularly related to sustainable resources use and conservation of biodiversity.

Population models are very important for management of commercial wildlife species. They also play a key role in creating management plans for both invasive and endangered species. Population dynamics helped to shift the view of ecosystems from a nonequilibrium model to one that is in constant change. As a result, population dynamics influenced many approaches to biodiversity conservation from preserving natural areas to influencing ecosystem processes such as fire, herbivory, water regimes, and nutrient flow. Overall, understanding population dynamics helps us make decisions on how to best meet an overarching goal of environmental sustainability.

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See also Carrying Capacity; Complexity Theory; Disturbance; Extreme Episodic Events; Food Webs; Global Climate Change; Keystone Species; Outbreak Species; Plant-Animal Interactions; Regime Shifts; Resilience; Shifting Baselines Syndrome

**FURTHER READING**


Rain Gardens

Stormwater runoff that flows over impervious surfaces picks up contaminants that find their way into lakes and streams. Stormwater runoff can cause flooding, erosion, destroy fish and wildlife habitat, and cause sewer overflows. Rain gardens, which are landscaped depressions filled with plants, bioswales, and water gardens, can be used to reduce the damaging effect of urban stormwater runoff while creating beauty and wildlife habitats.

Snow and rainwater falling from the clouds and onto the ground pick up oils, grease, and heavy metals; salt used to maintain roadways in winter; fertilizers and pesticides from urban lawns; and silt from eroding soils. Precipitation transports these contaminants to bodies of water by way of storm drains. Stormwater runoff is the number one threat to lake and stream water quality.

Numerous innovative stormwater management techniques create beauty, sustain wildlife habitat, and provide groundwater recharge while also improving water quality, storing floodwater, and slowing the flow of stormwater runoff. Emerging best practices for addressing stormwater concerns are moving away from piping stormwater to water bodies and moving toward slowing and soaking water into the adjacent soils. One of these emerging stormwater management techniques is the rain garden.

Ecosystem Services

Natural ecosystems, including wetlands and the porosity of undeveloped green spaces, provide a natural way to address stormwater. Wetlands act like giant sponges with the ability to absorb and hold stormwater and provide the ecological service of reducing floods. Wetlands also work as natural filters; wetland plants take up or break down pollutants, keeping them out of receiving waters. Green space, whether forests, grass, or wildflowers, likewise allows rainwater to soak into the ground, replenishing groundwater and reducing erosion and subsequent sedimentation of lakes and streams. Microbes in the soil help to break down pollutants carried by stormwater. These types of ecosystem services can be created as part of the landscaping during construction of buildings, homes, or even parking areas. By building rain gardens, homeowners can help restore some of the functions lost due to destruction of native wetlands and other vegetation.

Rain Gardens

Rain gardens are simply landscaped depressions and are usually planted with native wildflowers and wetland plants that soak up rainwater and snowmelt from driveways, parking lots, sidewalks, sump pumps, and roof downspouts. Rain gardens are created as shallow, low areas and are designed and engineered to catch and absorb stormwater. Rain gardens provide a mechanism for stormwater to filter into the ground instead of being transported by storm sewers to rivers and lakes, where it carries pollutants from streets, parking lots, and lawns, affecting fish populations and water quality. Development of housing, shopping areas, and freeways removes the earth’s ability to cleanse stormwater naturally and creates impervious surfaces that direct stormwater into water bodies.

Constructing a Rain Garden

Rain gardens can be very large, designed by engineering firms, and incorporated into construction projects. For example, the University of Minnesota’s Duluth campus created a one-third-acre rain garden that can hold up to
227 liters of water coming off a parking lot. Rain gardens also can be as small as a 3 square meter area designed to catch rainwater from a home gutter. Home rain gardens are usually in the 3 square meter range and from 15 to 30 centimeters deep. These shallow ponds hold water from several hours to several days depending on the soil type, and then dry out in between rain events. Rain gardens are not ponds, and because they dry out, they are not breeding grounds for mosquitoes. A home rain garden can be constructed easily in a day without professional help. Plants are the main expense of a rain garden.

The objective of a rain garden is to encourage infiltration of rainwater, and therefore it should not be placed in an already wet location. A homeowner should locate the rain garden at least 3 meters away from the house so that the rainwater does not seep into the foundation. Considerations include other landscaped areas around the property, the view from both inside and outside of the house, and the proximity to the house. A rope or garden hose can create the outline of the garden. Soil type is also a factor in determining the location and size of the garden. Clay soils have the slowest infiltration rate, and rain gardens need to be larger to accommodate this, or the soil needs to be amended by adding sand or a sandy-loam mix.

A truly rewarding aspect of constructing a rain garden is that when it fills with water, it attracts birds and other animals that come to drink and bathe in it.

Plants

Choosing plants for a rain garden is similar to selecting them for other gardens; height, sun requirements, and color need to be taken into consideration. A rain garden is unique, however, in that the plants must be able to tolerate both saturation and dry periods, and if collecting runoff from streets, must also be salt tolerant. Herbaceous wildflowers, wetland plants, grass, bushes, and trees can all be part of the landscape of a rain garden. Some rain gardens are planted with species that will attract and feed butterflies, creating twice the benefit of such landscaping. Once these plants become established, the garden requires little weeding. Moss-covered rocks can conceal the pipes that bring the water in to the garden. Local extension service offices or garden centers can provide a list of plants that will work for a specific location.

Making an Impact with Rain Gardens

Creating an individual rain garden may seem like a tiny step toward correcting environmental damage and human impacts, but collectively they can provide great benefit to a community. A project in Kansas City, Missouri, called 10,000 Rain Gardens, has brought citizens together to capture stormwater in rain gardens as well as in rain barrels for later use in their gardens. To comply with the US Environmental Protection Agency’s mandate to address stormwater issues, Kansas City chose rain gardens as one of their strategies. The city’s goal of building 10,000 rain gardens has mobilized the community to voluntarily build backyard rain gardens. Rain gardens have been planted at the city hall as well as in other public and private properties. Engaging citizens in a community-wide rain-garden project has the dual benefits of educating the citizenry about stormwater issues and combating the problem with hands-on construction.

There are many benefits of building rain gardens. For example, they are far less expensive than traditional stormwater infrastructures such as stormwater detention basins (often seen in freeway interchanges and shopping centers) and curb-and-gutter street construction. Rain gardens slow overland water flow resulting in less soil and stream-bank erosion, related stream sedimentation, and improved water quality of local lakes and streams. Because rain gardens allow water to soak into the ground, there is greater recharge of groundwater aquifers. Rain gardens help restore native ecologic functioning, increase biological diversity, and provide habitat for birds, butterflies, and insects. Finally, rain gardens improve neighborhood beauty, increase civic involvement, and provide opportunities for environmental education and community building.

Water Gardens

Water gardens differ from rain gardens in that they do not dry out in between rain events. Water gardens are constructed ponds of varying sizes. They range in depth from a few centimeters at the edges to a half meter (or deeper) in the middle and support plants that live in water. Water gardens can be designed to trap sediment carried by stormwater, and the plants can take up or break down pollutants in the same ways rain gardens do. A water garden differs from a rain garden in its constant source of water and use of aquatic plants.

The restoration of ecosystem services, stormwater cleansing, and public education are goals behind the Bayfront Stormwater Garden project in Duluth, Minnesota. The project was designed through a collaborative process among local artists, engineers, plant ecologists, and the Sweetwater Alliance, a nonprofit organization with a mission to raise water awareness through the arts and science. The designs for this project integrate art, sculpture, native aquatic and wetland plants, cultural history, and environmental education focused on the ecosystem functions of wetlands in stormwater management. The project is moving forward but is tied into another development on the site, which has not yet been approved.
by the city council. Upon completion, this project will cleanse stormwater that enters the St. Louis River from Interstate 35 as well as the surrounding downtown areas. Stormwater pumped to the top of a nearly 4-meter-high turtle (locally designed and created) will trickle down to water ferns and wetland plants. The water then moves through constructed concrete wetland plant ponds that create the outline of the Great Lakes. Each Great Lake is planted in one wetland plant species that when in flower will be visible from the hills above the waterfront. A dragonfly's wings forms both a bridge and a plaza for people to gather. Local artists will create the aesthetic elements of the Bayfront Stormwater Garden, and children from local schools will help make clay tiles to line the walkways and give voice to their concerns for water quality.

Ecological art restores degraded habitat while also educating the community about environmental degradation. Such community-based projects expand the voices engaged with habitat restoration projects and integrate many different professionals and laypersons into the art and restoration process.

**Outlook**

As people become more aware of the dangers of stormwater runoff and the pollutants it carries to bodies of water, they will find ways to mitigate the damage. This action may take the form of community-wide projects or of individual choices such as building home rain gardens. Whatever they do, people in groups or by themselves can make an impact in providing clean, fresh water for their environment.

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*See also* Ecosystem Services; Groundwater Management; Human Ecology; Hydrology; Irrigation; Landscape Architecture; Pollution, Nonpoint Source; Road Ecology; Stormwater Management; Urban Agriculture; Urban Forestry; Urban Vegetation; Water Resource Management, Integrated (IWRM)

**FURTHER READING**


Reforestation is considered as a means of managing climate change, both by helping reduce carbon dioxide emissions and creating sustainable forests that contribute generally to healthy ecosystems. Reforestation occurs throughout the world through natural and artificial techniques, and involves complex management issues and controversies involving land use and viability. Nonetheless, it promises to be an important arena of action into the next century.

At its most basic, reforestation is defined as the reestablishment of a forest in a land area that was formerly forested. In technical terms, reforestation is the natural and/or artificial restocking and regeneration of trees in recently depleted forests and woodlands as a result of natural or man-made activities, such as fires, storms, flooding, landslides, insect infestations, volcanic eruptions, slash-and-burn clearing, logging, or clear-cutting.

Natural reforestation or natural regeneration arises from stump sprouts (shoots), root sprouts (suckers), and natural seeding. Artificial reforestation is the direct seeding or planting of trees through ground and aerial seeding via machine or by hand, or both. Reforestation should not be confused with afforestation, which is the restoring and re-creating of non-forest land to a new forest, or a forest that was deforested many years ago from human activities, such as agriculture or habitation.

Reforestation is beneficial to humans and ecosystems because it improves air quality by reducing dust and capturing and storing the greenhouse gas carbon dioxide through a biological process called biosequestration. Planting trees is critical in preventing flooding and erosion by reducing the loss of topsoil and preventing increased sedimentation in streams, rivers, and wetlands. Reforestation benefits water bodies as well through aquifer recharge, storage, and recovery, and recycling inland rainfall. Tree planting and maintaining topsoil are essential for rebuilding natural habitats for plants and wildlife.

Reforestation around the World

Globally, there are 4 billion hectares (9,884,215,240 acres) of forest area. Five countries—Brazil, Canada, China, the Russian Federation, and the United States—account for approximately half of the forested area; there are ten countries that have no forest, and fifty-four that have less than 10 percent forestland (FAO 2010). Globally, from 1998 to 2007, the annual average rate of afforestation and reforestation was 10 million hectares per year (24,710,538 acres/year). These involved predominately indigenous species that were both afforested (approximately 71 percent) and reforested (approximately 64 percent). According to the Food and Agriculture Organization (FAO) of the United Nations (2010), the total global reforested area was 5.3 million hectares (13,096,585 acres) in 2005. These statistics include 163 countries, which account for approximately 95 percent of the forested area and 98 percent of the planted forest area. (These figures are noted by the FAO report as being from the years 2003 to 2007, but are referred to as 2005 figures.) There is considerable variability among the 5.3 million reforested hectares efforts across the planet. Table 1 on page 322 shows the reforested hectares by region. (As a point of comparison, Europe's nearly 1 million reforested hectares, or nearly 10,000 square kilometers, is approximately the size of Lebanon.)

Worldwide, ten countries account for 82 percent of reforested hectares, or 4,392,412 of the 5,348,017 million
Table 1. Reforested Land by Region (2005)

<table>
<thead>
<tr>
<th>Continent</th>
<th>Hectares</th>
<th>Acres</th>
</tr>
</thead>
<tbody>
<tr>
<td>Africa</td>
<td>237,123</td>
<td>585,943</td>
</tr>
<tr>
<td>Asia</td>
<td>2,478,801</td>
<td>6,125,250</td>
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<tr>
<td>Europe</td>
<td>992,540</td>
<td>2,452,619</td>
</tr>
<tr>
<td>North America</td>
<td>835,815</td>
<td>2,109,822</td>
</tr>
<tr>
<td>Central America</td>
<td>22,392</td>
<td>55,331</td>
</tr>
<tr>
<td>Oceania</td>
<td>37,423</td>
<td>92,474</td>
</tr>
<tr>
<td>South America</td>
<td>722,527</td>
<td>1,785,403</td>
</tr>
</tbody>
</table>

Source: FAO 2010.

Table 2. Top Ten Reforested Areas by Country

<table>
<thead>
<tr>
<th>Country</th>
<th>Hectares</th>
<th>Acres</th>
</tr>
</thead>
<tbody>
<tr>
<td>India*</td>
<td>1,480,000</td>
<td>3,657,159</td>
</tr>
<tr>
<td>United States</td>
<td>606,000</td>
<td>1,497,458</td>
</tr>
<tr>
<td>Brazil*</td>
<td>553,000</td>
<td>1,366,492</td>
</tr>
<tr>
<td>Russian Federation</td>
<td>422,856</td>
<td>1,044,899</td>
</tr>
<tr>
<td>China</td>
<td>337,000</td>
<td>832,745</td>
</tr>
<tr>
<td>Vietnam</td>
<td>327,785</td>
<td>809,974</td>
</tr>
<tr>
<td>Mexico</td>
<td>247,600</td>
<td>611,832</td>
</tr>
<tr>
<td>Indonesia</td>
<td>153,941</td>
<td>380,396</td>
</tr>
<tr>
<td>Finland</td>
<td>133,680</td>
<td>330,330</td>
</tr>
<tr>
<td>Sweden</td>
<td>130,550</td>
<td>322,596</td>
</tr>
</tbody>
</table>

Source: FAO 2010.

*India and Brazil figures include afforestation (the practice of planting trees in areas where trees have not existed before).

hectares in 2005. Table 2 shows the distribution of the reforested areas by the top ten countries.

Comparing Deforestation, Reforestation, and Afforestation

Although reforestation is a significant global activity, deforestation has been outpacing it for decades. Deforestation accounted for an annual loss of 16.0 million hectares (39 million acres) of forest through the 1990s and dropped to 13.0 million hectares (32 million acres) annually in the first decade of the twenty-first century. The 5.3 million reforested hectares (13 million acres) minus the 13.0 million hectares (32 million acres) loss to deforestation results in a net loss of 7.7 million hectares (19 million acres). Even if including the 5.6 million afforested hectares (13.9 million acres), there are about 2.0 million more hectares (4.9 million acres) deforested than reforested or afforested. The global South accounts for the majority of forest loss, with South America (4.0 million hectares [8.9 million acres]) and Africa (3.4 million hectares [8.4 million acres]) accounting for a little over half of global forest loss. Forest utilization in the less-developed countries is a result of rapid urbanization, population growth, and economic demands for forest products. European forests have continued to expand, while those in North and Central America have remained stable. Australia has been losing forest since 2000 as a result of considerable drought-related issues and forest fires (FAO 2010).

Management Challenges of Reforestation

A main reforestation management challenge is to be cognizant of the reasons for the forest loss or degradation. For example, forest loss due to natural events (e.g., fires, floods, insect infestation) or human-caused events (e.g., logging, livestock grazing) result in different management challenges. Fully understanding the reasons for the forest loss or degradation allows better policy and management decisions. Current emphasis is on adding more trees rather than addressing the underlying causes of the forest loss and degradation. The science of forestry management should drive government and agency policy and work in conjunction with local populations as much as possible to address their land needs. The management challenge is to balance biodiversity and economic benefits and to reduce ecological degradation, such as flooding and erosion, all of which are factors in creating sustainable and healthy forests and ecosystems.

Policy makers’ decisions to plant trees for primarily economic benefits without considering other forest health variables can result in significant management challenges. Reforestation illustrates the manifest and latent effects of large-scale policy decisions, which can have both positive and negative effects on the local populations, ecosystems, and forests. The manifest effect of reforestation policy is the reforestation of a new or previously forested area. The number of trees planted can be quantified to demonstrate a successful policy, if the number of trees is the statistic of interest. The methods used to plant or replant trees can affect the forest and ecosystem health, however, if not managed carefully. The latent effects of a trees-only reforestation policy can result in the creation of a monoculture and reduced biodiversity.
A monoculture can result in high yields, growth, and returns if reforestation efforts use offspring of genetically superior trees. Properties of monocultures differ, however, depending on whether they result from natural or artificial reforestation. Natural regeneration takes place over many years and results in a mix of young, mature, old, and dead trees. This allows a polyculture of diverse plant and animal species to flourish, in contrast to the artificial method of seeding or planting one species in a short time window, which prevents other vegetation from growing to support the biodiversity needed for a healthy forest. Lack of biodiversity makes trees vulnerable to insect infestations, pathogens, undesirable environment conditions, or a combination of all three. In a monoculture, a single undesirable event can completely degrade or even destroy the forest.

On the positive side, monocultures allow logging operations to be industrialized, because the trees are the same size and can be clear-cut. But the artificially produced monoculture and subsequent logging can hinder the growth of vegetation needed for a healthy forest that supports birds, plants, and wildlife. So the challenge in managing a forest is in creating a sustainable forest rather than just a tree plot. Small-scale clear-cuts, followed by a burn, provide a process for a greater variety of trees and species to thrive, thus preventing tree monoculture and increasing biodiversity.

**Reforestation Methods and Approaches**

Natural regeneration varies depending on the species of trees, but it is a predictable process. The three most common natural methods of tree regeneration are stump sprouts (shoots), root sprouts (suckers), and natural seeding. Stump sprouts or shoots are common among most hardwoods, such as oak, maple, and yellow poplars, and require other vegetation to be part of the forest environment. Stump sprouts arise out of burned or cut stumps and generally form in clumps. A root sprout is a clone that sprouts from the roots of the parent tree.

There are many trees that regenerate by root sprouts, such as willows, cottonwoods, black locust, beech, and wild cherries. Pines reproduce through natural seeding or when seeds from the parent tree fall from the cones to the ground and possibly regenerate, depending on habitat, temperature, and soil conditions. If the parent pine tree is left from a logging operation, it will need to be harvested after the seedlings have been established, because pines’ optimal growth is when they receive the same amount of sunlight and are approximately the same age. Although natural regeneration is a predictable process from a reproductive standpoint, the success of natural seed trees is less predictable.

Artificial reforestation is commonly what people perceive as reforestation. Relative to natural reforestation, artificial methods provide more control or active management of tree regeneration. The artificial process also allows for matching particular tree species to the available soil and site conditions. Artificial reforestation is based on direct seeding and planting. Seed planting is generally considered more successful than other natural or artificial seeding methods because planting allows for greater monitoring and control over the seed type (e.g., genetic characteristics), soil conditions, geographic location, and germination conditions. Successful seed planting, however, is dependent on having the correct or optimal soil conditions, geographic location, and germination conditions. Without these optimal conditions, results of seed planting are considered marginal.

Direct seeding is the planting of tree seeds by hand or by machine. It is considered to be less successful and requires more seeds than planting. Direct seeding is not widely used and is mostly limited to specific areas where natural regeneration or planting is not possible due to cost, poor soil conditions, or inaccessible terrain. Direct seeding is, however, quick and inexpensive. Mechanical row, cyclone, or spot seeders are used for direct seeding small areas. Seeds can be hand sown with a hand-cranked seeder or a cyclone seeder or by planting rows at regular intervals. Heavy seeds are placed in the ground by hand. Large areas of several hundreds of acres can be sown by airplanes or helicopters.

For direct seeding, germination conditions must be carefully considered. Direct seeding works best with lightweight seeds in lowlands with moist soil. In dry or sandy soils, seeds need to be covered with about...
1.3 centimeters of soil. In grazing areas, the livestock can trample the seedlings, and seeds in steep-sloped areas are prone to being washed away by rainfall.

The second method of artificial reforestation is planting live trees. This involves the transplanting of one-year-old nursery-grown seedlings to a specified location. The species of trees used are based on genetic qualities of growth characteristics and rates of growth. Planting is more labor intensive and expensive than seeding, because it involves more ground preparation, such as burning and chemical treatment, as well as planting the seedlings. A mix of manual and mechanical methods can be employed for planting, including hand-scalping, chemical treatment, and using a tractor. All these methods seek to reduce competitive vegetation in the transplant area. Hand-scalping involves clearing a 3- to 4-square-foot space and placing a mulch mat around the seedling. Chemical treatment involves spraying an herbicide and sometimes a pesticide prior to transplanting, to eliminate competitive vegetation. The mechanical method generally involves a tractor that plows, disk, or rototills the area prior to planting. The drawbacks here are that tractors can compact and remove topsoil and encourage competitive vegetation.

In any natural or artificial method of reforestation, seeded trees and seedlings compete with other vegetation for water, sunlight, and nutrients. Therefore, for the first three to five years, regular removal of undesired vegetation should be part of a successful reforestation management effort. This requires monitoring seed and seedling growth as well, since tree seeds are vulnerable to being eaten by birds and other wildlife, while seedlings are vulnerable to being eaten by rodents, rabbits, deer, and more.

Reforestation Controversies and Debates

The controversies and debates surrounding reforestation focus on whether, or to what extent, reforestation (1) reduces greenhouse gases; (2) competes with food production, livestock grazing, and human habitation; and (3) results in increased fire risk.

The main debate within the environmental and scientific communities is to what extent reforestation reduces greenhouse gases. Although there is little doubt that reforestation can reduce carbon dioxide (CO₂) in certain areas, the debate centers on how effective trees are at removing CO₂ gas in the short term and long term. On the positive side of the debate, reforestation creates immediate land carbon sinks that reduce CO₂. For example, a mature apple tree lowers CO₂ by approximately 161 kilograms annually, while an oak tree 46 centimeters in diameter lowers CO₂ by 282 kilograms annually. The US Environmental Protection Agency (2009), however, notes that a two-person household annually releases 18,820 kilograms of CO₂ into the atmosphere. So a two-person household would need to plant around 138 of those apple trees or 67 of those oak trees to offset their annual CO₂ release of direct and indirect fossil-fuel burning.

Some researchers and scientists argue that reforestation will have only limited effects on climate mitigation because of the volume of CO₂ still released by households. In addition, reforestation and its associated carbon sink vary by geographic location, and reforesting certain areas can reduce or enhance the planet’s albedo—its ability to reflect sunlight. For example, large-scale reforesting of boreal forests would result in a forest canopy overtopping the previously snow-covered area, and this would reduce the reflection of sunlight (decrease albedo) and trap more heat that would otherwise be reflected by snow-covered surface. In contrast, reforesting a tropical region would result in more cloud formations or cloudiness, which would reflect more sunlight into the atmosphere (increase albedo) rather than penetrating the ground surface. Many researchers have claimed that increased reforestation in the tropical regions has the greatest effect on reducing CO₂.

Some people have argued that the scientific community is not accurately addressing this greenhouse gas. Specifically, they argue that the CO₂ measurement equipment is limited in measuring plant biomass, there is lack of a standard instrument or process for determining CO₂ levels, and methodological and sampling errors have resulted in inconsistent results from year to year. Thus, creating better scientific equipment, instruments, and methodologies is important in helping policy makers formulate effective reforestation and forest management plans. These tools can also help resolve the debate of whether reforestation reduces greenhouse gases.

Reforestation programs may also compete for land with food production, livestock grazing, and human habitation. The controversy revolves around appropriate utilization of the land. Some people argue that reforestation lets an area remain idle when it could be used for economic activities such as agriculture and livestock production. Letting a forest grow also competes with development of human habitation and settlements. In less-developed countries and regions with limited economic activities, the practice of reforestation is not viable, because communities are not paid for the services provided by standing forests.

Finally, some forest managers and policy makers are concerned that reforestation will lead to a higher risk of forest fires; the reasons for this are that tree harvest rates are lowered and fire-suppression activities, such as brush thinning, will not occur. The increased forest biomass
Reforestation efforts should focus on building a healthy forest or a biologically diverse polyculture rather than a monoculture tree plot. In addition, forests need better fire management plans that include reduction efforts, such as thinning and limited or controlled burns. In all, reforestation is an important tool in preventing or controlling climate change and is technically simpler than many other means. The countries with the largest forests are expected to be world leaders, but will require motivations, including public support, to drive policy changes and proactive solutions.

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See also Best Management Practices (BMP); Biodiversity; Ecological Restoration; Ecosystem Services; Fire Management; Global Climate Change; Natural Capital; Nutrient and Biogeochemical Cycling; Plant-Animal Interactions; Population Dynamics; Refugia; Resilience; Rewilding; Soil Conservation; Species Reintroduction; Tree Planting; Wilderness Areas

FURTHER READING


and prolonged annual drier seasons in some regions have significantly increased the frequency of and potential for fires. In addition to their danger, forest fires release large amounts of CO₂ into the atmosphere. Typically, the new vegetation following the fire offsets the CO₂ release. It should be noted, however, that the frequency of forest fires coupled with warmer temperatures and reduced precipitation can lead to more carbon being released than is recovered by the new vegetation after the burn.

Reforestation Outlook

The reforestation outlook for the next five to ten years is mostly positive because of the policies, projects, and increased awareness campaigns continually being developed. Projects include global-scale programs such as the United Nations’ Billion Tree Campaign and countrywide programs such as Peru’s National Reforestation Campaign, which seeks to plant 60 million trees along the Rimac River. As well, corporate-sponsored initiatives, such as the PRAIS Corporate Communications and PRAIS Foundation “Millions of People, Millions of Trees” campaign, are increasing. These efforts will increase awareness of reforestation beyond the next decade.

At the same time, reforestation efforts face challenges at the local, state, and global levels. First, many reforestation efforts center on the economic benefits of tree regeneration—a focus that generally comes at the expense of sustainable forests. But if sustainability was the focus of the policies instead of economics, the current deforestation levels would come into balance with the reforestation/afforestation efforts. This significant change of focusing on sustainability in forestry would require considerable political and social will.

Next, the view of reforestation as single solution for climate mitigation is not possible given the CO₂ per capita utilization in more-developed countries. It is important for policy makers to initiate and enforce policies that reduce the amount of CO₂ entering the atmosphere from all sources, so as not to rely exclusively on reforestation to reduce CO₂.
Refugia

A refugium is a geographical area where a population, species, or community has survived environmental instabilities over long periods of time. The refugium theory suggests that existing genetic and species diversity is shaped by historical environmental changes and predicts that the long-term persistence of biodiversity is dependent on refugia. Future studies should focus on substantiating the general biological significance of refugia, identifying climate change refugia, and studying the evolutionary potential of refugial and nonrefugial populations.

Biological diversity (ecosystems, species, and their genes) is not randomly distributed on Earth. Both contemporary ecological processes and historical ones, especially changing climates resulting in changed environments, have shaped biodiversity. The refugium theory predicts that long-term survival of biodiversity depends on areas of environmental stability (refugia). The refugium theory thus offers a framework for understanding and studying biodiversity in relation to the history of populations and geographical areas. The concept of refugia is being refined, and it has potential in promoting sustainable landscape management and conservation. Unfortunately, refugia will undoubtedly be affected by climate change in the twenty-first century.

Theory

Since life first appeared, new organisms diversified and colonized the Earth starting from the place of their origin. Although processes were interrupted by major catastrophic extinction events, speciation (species formation) resulted in species being more specialized and adapted to new ecological conditions. Adaptation to specific geographical areas resulted in both higher numbers of coexisting species and also smaller potential geographical distributions, making highly adapted species vulnerable to fast environmental and ecological change.

Hypothetically, given environmental stability over a long time period, a species should be able to disperse from its place of origin to all potential habitats to which it is adapted. Many species distributions, however, are geographically rather limited, although potential habitats are widespread. This might be a result of (1) unstable environments resulting in local and regional extinction over time, (2) new species failing to evolve as fast as the environments change, and (3) species needing long time periods to disperse to and colonize all potential habitats (Bennet 2004).

Refugia are based on the observations that areas that are environmentally similar have quite variable species numbers, and that areas with many endemic species (i.e., species restricted to small geographical areas) do exist. The refugium theory tries to explain the persistence of biodiversity in hot spots with historical factors. Assuming that species do not evolve faster than the environment is changing, this theory predicts species stability in areas that are relatively less affected by changes to the environment.

History

The German ornithologist Jürgen Haffer (1969) proposed that among Amazonian rain forest birds, “most species probably originated in forest refuges during dry climatic periods” and suggested that species development in geographical isolation in several areas was the primary explanation for high species diversity. This argument was based on earlier ideas about speciation in ecological refugia (Haffer and Prance 2001), and...
since then many studies have adopted the refugium concept to explain other patterns of global diversity. Phylogeographers—researchers studying the genetic relationships of populations throughout their geographic ranges—later demonstrated the validity of the refugium theory (Hewitt 1996; Taberlet et al. 1998; Tribsch and Schönswetter 2003). Many studies support the hypothesis that species evolved and adapted in refugia that were geographically isolated. They showed that the lack of or reduction in gene flow among populations in once isolated areas during the ice ages can still be measured in present populations and that hybrid zones (zones where genetically differentiated populations came together after their range expanded beyond refugia) are found in many species. Progress in methodology in paleontology and the availability of historical pollen deposit data and macrofossils are two factors that allowed for pinpointing where refugia were located (Willis and Niklas 2004).

Several terms have been used to describe the idea that species survived in discontinuous ecologically stable areas within shifting but overlapping ranges caused by climatic changes. The vegetation ecologist J. H. Tallis (1991), for example, proposed the term reservoir instead of refugium. This term was intended to take account of the fact that populations in refugia were able to spread from the refugium, rather than being restricted to it. Scientific opinion, however, is settling on the term refugium (Rull 2009).

Toward a Unified Theory

A refugium can be generally defined as a geographical area of ecological and environmental stability over a period of time where a population, species, or community has survived long-term instabilities. The identification of a refugium could be based on independent, abiotic data or environmental modeling approaches (see Hugall et al. 2002 and Schönswetter et al. 2005). The term realized refugium might be used in cases where biological data (species endemism, phylogeographical pattern, and continuous paleontological records) provide evidence of the continuous presence of a species or population.

Several assumptions are associated with the refugium theory. The main process of diversification is assumed to be allopatric speciation (speciation under geographical isolation) or genetic drift (the random process of genetic differentiation of more or less small populations in isolation). Moreover, it is assumed that refugia are geographically stable. The refugium theory is applicable to single species or clades (related groups of species) as well as to assemblages of organisms or even ecosystems and biomes (ecological communities), depending on the point of view of the researcher.

Given that refugia have different shapes and sizes, the Spanish biologist Valenti Rull (2009) has suggested some refinements to refugium concepts. He defined microrefugia as small areas with local favorable environmental features, in which small populations can survive outside their main macrorefugium (larger areas). He also suggested distinguishing between glacial and interglacial microrefugia, which remain associated with a geographically stable macrorefugium over longer times (Rull 2010). A similar approach is to distinguish in situ and ex situ refugia; the latter do not assume a geographical overlap of geographical distributions of a population, species, or community over time. Depending on environmental conditions, ex situ refugia thus move around in the landscape.

Refugium theory makes several predictions. For example, refugia are predicted to maintain species, even when extinction is predicted for areas outside of them. Development of new species would only be possible in cases where a new species (or evolutionary lineage) has evolved in a refugium or where a species rapidly disperses to a refugium where it survives. Because genetic diversity diminishes only once a species moves out of a refugium, genetic variation, in particular allelic richness and the number of rare alleles (Widmer and Lexer 2001), is expected to be higher in refugia than in areas outside of them. Since the evolutionary potential of a population or species is higher when genetic variation is higher (see Frankham et al. 1999 for empirical evidence), refugial populations have a higher evolutionary potential than nonrefugial populations. Similar results hold on the species level: endemic species (especially paleoendemics) are expected to be frequent in refugia, and areas of endemism (areas where the geographical distribution of endemics significantly overlap) must coincide with refugia. Genetic diversity (in terms of heterozygosity) as well as species diversity (in terms of species richness), however, might be higher outside refugia (see Paun et al. 2008 and Tribsch et al. 2010 for empirical evidence), especially in general hybrid and suture zones (that is, areas where populations and species originating from refugia merge).

Critique and Modifications

Paleontologists frequently use the refugium concept (Bennet 2004, Willis and Whittaker 2000), but some have complicated it by defining many different types of refugia to explain observed data (Bennet and Provan 2008; Stewart et al. 2010). Based on emerging data that species responded individualistically to the ice ages and taking into account the importance of different vegetation types, several types of refugia have been proposed. For example, one researcher has proposed cryptic refugia within the microrefugium concept (Rull 2010).
Others take into account the different responses of cold- and warm-adapted species and propose glacial and interglacial refugia (Stewart et al. 2010). Such distinctions, however, are incompatible with the generally accepted understanding of refugium, because the definitions do not include geographical stability. A clear distinction should be made between refugium theory and a (testable) refugium hypothesis on the one hand and post hoc (after the fact) identifications used in many paleontological studies on the other hand.

The validity of the refugia theory has been proven for many temperate and boreal areas, for example, the European Alps (Schönswetter et al. 2005), southern South America (Sérsic et al. 2011), and North America (Soltis et al. 2006). Studies for the tropical biome, especially for Amazonia, have been more critical (see Knapp and Mallet 2003 and citations therein; but see also Haffer and Prance 2001). General criticism of the refugium theory might also question the underlying assumptions that allopatric speciation and differentiation are the main processes of diversification (Bennet 2004).

Ecological landscape genetic or population genetic studies use the refugium concept for shorter timescales, naming as refugia certain locations where individuals or populations survive perturbation and disturbance. Such so-called refugia help scientists understand ecosystem resilience (Ashcroft 2010; Sedell et al. 1990). Because the refugium theory involves assumptions of speciation and diversification at the gene or species level, however, this use of the term refugia may be misleading.

Refugia, Conservation, and Climate Change

Refugia that have high concentrations of native species, that have populations with unique genetic structures, or that contain important evolutionary lineages within species deserve to be considered in nature conservation strategies (Noss 2001). Conservation planning based on the location of refugia would effectively complement the traditional measures planners use, which include species and ecosystem diversity, uniqueness, and anthropogenic (human caused) threat.

The refugium theory has implications for global climate change research. Predictions of future climates and future distributions of organisms may allow for predicting the location of climate change refugia. The spatial ecologist M. B. Ashcroft (2010) pointed out that a loose definition of the term refugium is an obstacle for identification of refugia from predicted climate change. He points out, for example, that a strictly local view of refugia might exaggerate the extinction risk for species that can easily move over long distances. Although biology and behavior of species are very important in understanding how species are distributed, future studies nevertheless should use a clear and simple refugium definition, with only few assumptions, at least as a starting point for climate change studies at large scales (the macrorefugium level). At the microrefugium scale, for example, in small mountain ranges with a focus on certain species, studies could be also based on more complex refugium definitions (for example, a dynamic microrefugium concept) and should aim for considering as many species-specific biological and ecological properties as feasible. Such approaches have high potential for contributing to global and local conservation planning and for reducing the extinction risk of species in the future.

Why Study Refugia?

Comparative ecological and biogeographical studies show that refugia are of general importance for biodiversity. This offers the opportunity to compare refugial populations, species assemblages, and even whole ecosystems with recently established nonrefugial ones. In the face of global climate change, such studies might involve (1) identifying global change refugia, (2) quantifying the threat to species and genetic diversity in refugia, (3) studying the evolutionary potential of refugial populations, and (4) developing conservation strategies including refugial information worldwide.

Refugia are natural centers of long-term sustainability. Nature, however, is dynamic, and scientists still do not understand completely the types of organisms and timescales for which the refugium theory is valid. Sustainable ecosystem management may benefit in many ways from including refugia in management strategies. Considering
refugia as a predictor of unique biodiversity and unique evolutionary capacity should allow for an effective conservation strategy complementing, but not replacing traditional approaches.

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See also Agricultural Intensification; Biodiversity; Biodiversity Hotspots; Biological Corridors; Boundary Ecotones; Buffers; Community Ecology; Complexity Theory; Edge Effects; Food Webs; Global Climate Change; Landscape Planning, Large-Scale; Permaculture; Population Dynamics; Succession

FURTHER READING


Regime Shifts

A regime shift is a large, persistent change in the structure and function of a system. In ecosystems, regime shifts substantially affect the flow of services that societies rely on; they often occur unexpectedly and are difficult or impossible to reverse. Understanding the mechanisms that lead to and sustain regime shifts and predicting regime shifts before they happen are major challenges for ecosystem managers concerned with long-term sustainable resource management.

Regime shifts are large, abrupt, persistent changes in the structure and function of a system. The terms abrupt and persistent are relative, referring to the time period over which the shift occurs in comparison to the duration of the regime. A particular regime is not itself an unchanging condition, but is characterized by dynamic fluctuations while maintaining the same basic system structure and function. Each regime is maintained by mutually reinforcing processes that produce a characteristic dynamic behavior. Regime shifts occur when a smooth change or a single disturbance triggers a dramatically different system behavior. In ecosystems, abrupt changes can disrupt the sustainable provision of services that societies rely upon, such as freshwater, food production, or nutrient cycling. Regime shifts operate and impact societies at various scales, ranging from local to global. At the local scale, one of the best documented examples of a regime shift is bush encroachment—or vegetation progression—where small changes in herbivory can lead to distinct shifts in habitat, such as from grassy-dominated to woody-dominated savannas. Encroachment has been documented in wet savannas in Africa and South America, and the change severely impacts the use of the grassland ecosystems, such as for cattle ranching. At the regional scale, forests can shift to savannas if the moisture recycling is weakened by deforestation. Rainforest areas in the Amazon and East Asia are thought to be at risk from this type of regime change. At the global scale, climate warming is leading to the retreat of arctic sea ice during summer, with impacts on sea water levels, climate regulation, and ecosystem regimes worldwide.

Historical Background

Mathematicians were intrigued by the phenomenon of abrupt shifts in systems behavior long before ecologists. They developed catastrophe and bifurcation theory to explain and classify different types of nonlinear dynamics, analyzing how the effects of small changes in circumstances can sometimes result in dramatic shifts in the behavior of a system. This theory has had applications in a broad range of fields, from atoms to climate dynamics, as well as social systems.

In ecology, the idea of populations or ecosystems exhibiting regime shifts arose from reflections about the meaning of stability and change in ecosystems. The work of the US biologist Richard C. Lewontin and the Canadian ecologist C. S. Holling established the first references to alternative stable states or regime shifts in the late 1960s. During the 1970s, theoretical models were developed to explore Lewontin and Holling’s ideas in grazing and fisheries systems, as well as insect outbreaks. Early work on regime shifts was criticized for the lack of empirical evidence and long-term quality data. These critiques slowed research on regime shifts until the 1990s when evidence was collected for abrupt changes in kelp forests, shallow lakes, drylands, and coral reefs. This empirical evidence led to theoretical revisions in the early 2000s, which led to a boom in regime shifts research in the last decade in such fields as oceanography, fisheries science, terrestrial ecology, and community ecology.
Despite this boom in research activity, there is as yet no agreement on a definition of regime shifts. Differences among definitions center on the meaning of stability and the meaning of abrupt—how long a time span is required for a situation to be considered a regime, and what span of time qualifies as abrupt? In the end it is a matter of scale, and scholars in different disciplines are using slightly different definitions. For example, in oceanography a regime must persist for at least several decades and include climate variability as a driver, while for marine biologists a regime might last only five years and be induced simply by changes in population dynamics.

In social sciences, regime shift ideas have been applied using a slightly different framework. Parallel concepts have evolved referring to abrupt change in society to explain phenomena where an attractor is reinforced by feedbacks and the output is generally determined by initial conditions and the system history. For example, the norms that rule our societies—also known as institutions—change over time. The possibility of change is reduced, however, once a set of norms is established and accepted by a community. Such is the case of driving on the right versus driving on the left. Once a norm is established, infrastructure such as roads and cars are built to fit this option, and then it becomes difficult to shift to the alternative option. In a broader sense, each modus operandi or arrangement of institutions can be considered a regime in social systems.

Nevertheless, the application of bifurcation and catastrophe theory in ecology and social sciences remains controversial. Difficulties include collecting enough data, performing experiments with real systems, and distinguishing true regime shift dynamics from environmental noise. In order to apply the regime shift concept to a particular problem, one necessarily has to limit the scales and range of dynamics being studied. For example, mass extinctions of species can be seen as abrupt change on geological time scales, while understanding a financial downturn within the human economic sphere requires focusing on much shorter time scales.

**Theoretical Basis**

Regime shifts are triggered either by large system shocks, for example droughts, earthquakes, and floods, or when internal processes become weakened to the point that the system reorganizes itself into a different dynamic structure and function. In both cases the resulting change can be smooth, abrupt, or discontinuous, where the difference is characterized by the interaction among fast and slow processes in the system.

In analyzing regime change, it is important to distinguish between state variables (also called fast variables or response variables) and conditions (also referred to as parameters, forcing variables, control variables, or slow variables). The former always represents a fast dynamic in the system, and the latter refers to factors that often are assumed to be constant but in fact change slowly. Smooth or gradual change can be described by a quasi-linear relationship in fast and slow processes. It implies that a small disturbance will deliver a small consequence and a big disturbance a big consequence. In contrast, abrupt change shows a nonlinear relationship among fast and slow variables leading to a threshold-like response, where a small change can lead to a big consequence. Discontinuous change is characterized by the difference in the trajectory of the fast variable when the slow one increases compared to when it decreases.

The difference associated with discontinuous change is termed hysteresis and indicates that the behavior of the system is dependent on the history of the system. It means that once the system has shifted from one regime to another, it is not sufficient to restore the conditions present when the system originally shifted. Instead, if one wants to restore the original regime, one needs to reduce the factors that led to the shift (e.g., nutrient pollution that led to eutrophication of a water habitat) to a much lower level.

In some cases, crossing the threshold brings about a dramatic change in the response variables while in others the transition is more gradual. Abrupt change has been documented in eutrophication in lakes and coastal ecosystems, which leads to toxic algae blooms and reduction in fish productivity. Another example is that of coral reefs where algae overgrows coral and inhibits its further development, reducing ecological complexity in habitat diversity, as well as limiting opportunities for fishing and tourism industries. Smooth or gradual change has been documented for bush encroachment, a regime shift that reduces the ecosystem services related to grasslands, such as cattle ranching.

Strong hysteresis effects are often caused by a change in the strength of internal system feedbacks or by the formation of new feedbacks. For example, regime shifts in coral reefs are often mediated by the strength of the herbivory feedback that controls algae growth. On the other hand, in a case such as lake eutrophication, once the system has shifted to the murky water regime, a new phosphorous recycling feedback maintains the regime even if the driver (nutrient input) is reduced.

**Evidence and Prediction**

Evidence of regime shifts derives from observations, models, and experiments. Numerous examples in ecological systems are now considered well documented. (See table 1 on page 332.) More evidence is available for small-scale
# Table 1. Well-Known Examples of Ecological Regime Shifts

<table>
<thead>
<tr>
<th>Regime Shift</th>
<th>Regime A</th>
<th>Regime B</th>
<th>Impacts on Ecosystem Services</th>
<th>Evidence</th>
<th>Source of Evidence</th>
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<td>Freshwater eutrophication</td>
<td>Non-eutrophic</td>
<td>Eutrophic</td>
<td>Reduced access to recreation, reduced drinking water quality, risk of fish loss</td>
<td>Strong</td>
<td>Observations</td>
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<td>Fisheries collapse</td>
<td>Abundant stocks</td>
<td>Collapsed stocks</td>
<td>Reduced food production, employment and ecosystem degradation</td>
<td>Strong</td>
<td>Observations</td>
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<tr>
<td>Soil salinization</td>
<td>High productivity</td>
<td>Low productivity</td>
<td>Yield declines, salt damage to infrastructure and ecosystems, contamination of drinking water</td>
<td>Strong</td>
<td>Observations</td>
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<tr>
<td>Coral reef degradation</td>
<td>Diverse coral reef</td>
<td>Reef dominated by macro-algae</td>
<td>Reduced tourism, fisheries, biodiversity</td>
<td>Strong</td>
<td>Observations</td>
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<tr>
<td>Coastal hypoxia</td>
<td>Non-hypoxic</td>
<td>Hypoxic</td>
<td>Fishery decline, loss of marine biodiversity, toxic algae</td>
<td>Strong</td>
<td>Observations</td>
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<tr>
<td>River channel position</td>
<td>Old channel</td>
<td>New channel</td>
<td>Damage to trade and infrastructure</td>
<td>Strong</td>
<td>Observations</td>
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<td>models</td>
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<tr>
<td>Forest—Savanna</td>
<td>Forest</td>
<td>Savanna</td>
<td>Loss of biodiversity, moisture cycle, and rain,</td>
<td>Strong</td>
<td>Observations</td>
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<tr>
<td>Kelp transitions</td>
<td>Kelp dominated seascape</td>
<td>Turf algae dominated or urchin barrens</td>
<td>Reduced biodiversity and fisheries</td>
<td>Strong</td>
<td>Observations</td>
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<tr>
<td>Monsoon circulation</td>
<td>Strong monsoon</td>
<td>Weak monsoon</td>
<td>Reduced water cycling, less rain, and less productivity.</td>
<td>Medium</td>
<td>Models</td>
</tr>
<tr>
<td>Vegetation patchiness</td>
<td>Spatial pattern</td>
<td>No spatial pattern</td>
<td>Productivity declines, erosion</td>
<td>Medium</td>
<td>Observations</td>
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<tr>
<td>Wet savanna-Dry savanna</td>
<td>Wet savanna or desert</td>
<td>Dry savanna</td>
<td>Loss of productivity, yield declines, droughts/dry spells</td>
<td>Medium</td>
<td>Models</td>
</tr>
<tr>
<td>Cloud forest</td>
<td>Cloud forest</td>
<td>Woodland</td>
<td>Loss of productivity, reduced runoff, biodiversity loss</td>
<td>Medium</td>
<td>Observations</td>
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<td>models</td>
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<tr>
<td>Bush encroachment</td>
<td>Open grassland</td>
<td>Closed woodland</td>
<td>Reduced grazing for cattle, reduced mobility, increased fuelwood</td>
<td>Medium</td>
<td>Observations</td>
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<tr>
<td>Greenland Icesheet</td>
<td>Constant icesheet</td>
<td>Periodic icesheet</td>
<td>Reduced climate regulation, salinity regulation in the sea, and increase of sea level.</td>
<td>Medium</td>
<td>Observations</td>
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<td>model</td>
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<tr>
<td>Arctic sea ice</td>
<td>Constant icesheet</td>
<td>Periodic icesheet</td>
<td>Reduced climate regulation, salinity regulation in the sea, and increase of sea level.</td>
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*Source: Modified from Gordon, Peterson, and Bennett (2008, 215); Scheffer et al. (2001, 595); and the Regime Shifts DataBase (n.d.).*
systems that have fast dynamics and are relatively easy to manipulate, such as lakes; for large-scale systems with slow dynamics, less evidence is available. Very little is known about interactions between regime shifts, especially at different scales.

Observation of abrupt change in time series data constitutes the main evidence of a regime shift. When time series are available, one can identify regime shifts by looking for abrupt jumps in the data. An iconic example is the record of ocean sediments in the Sahara region some 5,500 years ago, clearly signaling the transition from a moist, vegetated condition to the current desert state.

Most time series data are statistically “noisy,” and statistical methods are required to identify different regimes (see Andersen et al. 2009). Among the most commonly used tools are ordination methods such as principal component analysis (PCA), chronological clustering, and sequential t-tests. These methods, however, can only be used to identify regime shifts that have already happened.

Given the difficulty of obtaining observations, much of the work on regime shifts has been theoretical, using models to explore under which conditions regime shifts are likely to occur. Modeling is a good tool for gaining understanding of causal relationships and the strength of feedback mechanisms underlying regime shifts; it can also serve to identify critical thresholds and assess the interaction of key variables. The variability in threshold level can be studied through modeling to identify threshold zones or values where a regime shift becomes more likely: for example, grazing has been modeled as an option for managing bush encroachment (Anderies, Janssen, and Walker 2002); irrigation systems have been modeled to understand soil salinization (Saysel and Barlas 2001); and lake eutrophication has been studied in various experiments and models (Carpenter 2003).

A key research frontier is the development of early warning signals for regime shifts. Among the methods being explored are changes in the statistical properties of a system, such as an increase in variability and autocorrelation that occurs ahead of an approaching threshold. Such studies, however, require extended time series that often are not available for ecological data. Where deep knowledge of the feedback dynamics is available, application of the statistical method known as Bayesian networks can be used to understand and predict potential interacting thresholds.

Management Challenges

Managers focused on sustainable use of resources aim to optimize the efficiency of the systems they manage, in other words increasing the well-being obtained from ecosystems now without compromising the ability of future generations to do so. As managers pursue optimization, however, change in key variables can accumulate slowly and bring about abrupt transitions. Awareness of the nature of regime shifts can help managers identify key feedback processes to build resilience to such transitions—the ability to cope with change while maintaining the basic structure and function of the system.

Identifying the main drivers of change or possible interactions among drivers represents major challenges for current research. It has been suggested that cross-scale interaction among local and regional regime shifts are possible. Vertical interactions refer to linkages among regime shifts that happen at different scales. Horizontal interactions happen roughly on the same scale, when processes happening at the same rate interact in time or space.

Managers face the challenge of identifying systems vulnerable to regime shifts as well as possible actions to maintain or restore desired regimes. To gain such understanding, feedback loops underlying change need to be assessed for each specific case. Understanding of slow and fast processes helps in assessing managerial options and identifying what is manageable and what is not. Also helpful is assessment of the robustness of policies and learning from the counterintuitive nature of system dynamics. Since regime shift dynamics are characterized by the possibility that a small disturbance will cause big effects, understanding regime shifts also allows managers to look for windows of opportunity to minimize or reverse catastrophic dynamics. For example, improved understanding of events associated with El Niño, the warm Pacific Ocean current disruptive to ocean life and weather patterns, can be used to restore ecosystems from degraded to vegetated states (Holmgren and Scheffer 2001). On the other hand, detection methods need to be further developed to fit the reality of
resource management: limited in-depth knowledge and limited time series data regarding the complex functioning of environmental systems.

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See also Biodiversity; Community Ecology; Complexity Theory; Ecological Forecasting; Edge Effects; Eutrophication; Extreme Episodic Events; Food Webs; Global Climate Change; Indicator Species; Invasive Species; Keystone Species; Light Pollution and Biological Systems; Outbreak Species; Plant-Animal Interactions; Population Dynamics; Resilience; Wilderness Areas

FURTHER READING
Ecological resilience is the capacity of a system to withstand disturbance without collapsing and shifting into a different regime. Humankind relies upon a consistent production of ecological goods and services. When an ecosystem's resilience is exceeded and the system shifts into a new regime, the system may be less favorable from a human perspective. Understanding and managing for resilience is thus essential for sustainability.

Humankind relies upon the goods and services provided by ecosystems: clean water and air are two such examples. Resilience is a measure of the disturbance that an ecosystem can withstand before shifting into a different ecological regime, which may provide fewer goods and services. Because it is in humankind's interest to maintain ecosystems in regimes that provide vital ecological goods and services, it is critical to understand resilience.

Background

Resilience is the capacity of an ecosystem to withstand disturbance without collapsing and shifting into an alternate regime, or a different type of ecological system organized around different processes and structures. Examples of alternate regimes are a clear, low-nutrient, low-algae, oxygen-rich lake (oligotrophic) or a turbid, high-nutrient, high-algae, oxygen-poor lake (eutrophic); a coral reef dominated by corals or by macroalgae; and a grassland or a woody-plant-dominated shrubland. Resilience is an emergent phenomenon of complex systems, which means that it cannot be deduced from the behavior of the parts of a system. In other words, a detailed understanding of the wolf and elk populations in Yellowstone does not tell us how the ecosystem as a whole operates or if it is resilient. The behavior of a system cannot be understood by merely adding together what we know about the parts.

Ecologists understand complex systems to be self-organizing and to have inherent uncertainty, nonlinear dynamics, and emergent phenomena. Complex systems are self-organizing because there is no central entity responsible for directing the processes and functions of the ecosystem. An ecosystem arises instead from the nondirected interaction of the parts; complexity arises over time from many simple interactions. An ecosystem is complex because the whole is more than the aggregation of the parts. Nonlinear dynamics occur when small changes have a disproportionately large effect. Phosphorus levels in a lake may steadily rise over time, with no apparent consequence to the lake, for example, until they rise just a bit more and the lake suddenly tips into a new regime, becoming eutrophic and prone to algae blooms. Ecosystems are also complex adaptive systems; the interaction between the parts and the emergent properties of the whole leads to dynamic changes in the system. These changes have consequences for how scientists understand and manage ecosystems. Understanding the resilience of ecosystems is important, in part because humankind relies upon a consistent production of ecological goods and services, such as drinking water, crop pollination, soil renewal and regeneration, abundant marine life to eat, carbon dioxide storage, and so forth. When an ecosystem's resilience is exceeded and the system shifts into a new regime, the system may become less favorable from a human perspective and produce fewer goods and services.

Ecologists have developed resilience theory since the 1970s to explain the nonlinear dynamics of complex adaptive systems. When the resilience of an ecosystem has been exceeded, the system discernibly changes, such
systems are characterized by a single equilibrium state. This assumption is inappropriate for complex adaptive systems such as ecological systems.

Scale is a critical concept to understand when discussing complex systems. Scale typically refers to the spatial extent and temporal frequency of the object or process of interest. In ecosystems, different processes dominate at different spatial and temporal scales. Small and fast processes such as the turnover of leaves on trees are orders of magnitude different from the large and slow processes, such as climate, that drive the location of boreal forest on a continent. Because all these processes occur at discrete spatial and temporal scales, there are thresholds between scales of structuring processes, which are called discontinuities. Neither processes nor structure in complex systems are continuous. Processes operate over domains of scale. These processes are separated by abrupt thresholds that are the transition to a new set of structuring processes. This discontinuous structure is critical to the resilience of ecological systems. Ecologists propose that resilience, or the ability of a system to buffer disturbance as when a lake shifts from a clear to a turbid state. (See figure 1.) The nonlinear dynamics of complex systems make it difficult to predict when that shift might occur, though advances have been made in this area. Small changes to a system may have disproportionately large consequences, and vice versa.

Ecological resilience should not be confused with engineering resilience, which emphasizes the ability of a system to perform a specific task consistently and predictably and to reestablish performance quickly should a disturbance occur. Applying this type of thinking to the management of ecosystems has been extremely problematic. The harvesting of renewable resources such as trees or fish cannot be treated as an engineered system, with predictable and consistent outputs. Ecosystems do not have an equilibrium state, where opposing forces are in balance, as assumed by an engineering definition. An ecosystem exists within a regime. Within a particular regime, the abundance and composition of the species that constitute that regime may change quite dynamically over time. Engineering resilience assumes that ecological

**Figure 1. Resilience**

*Source: US Environmental Protection Agency.*

The figure depicts a conceptual diagram of the basins of attraction for two possible ecosystem states. The position of the ball in the left basin of the upper diagram represents the current state of the system. Ecologists measure the resilience of the system as the amount of disturbance required to push the system from one basin of attraction to another (bottom diagram). The two basins represent two possible alternative stable states, characterized by two different regimes.
and stay in the same regime, results in part from the distribution of function within and across the domains of scales of an ecosystem. The relationship between resilience, discontinuities, and the distribution of function within and across scale has led to formal propositions that allow for quantifiable measures of the relative resilience of different ecosystems.

One such proposition is that the resilience of ecological processes, and therefore of ecosystems, depends partly upon the distribution of function within and across scales. Many ecological functions, such as pollination or seed dispersal, are provided by species. If species that are members of the same functional group operate at different scales, they provide mutual reinforcement that contributes to the resilience of a function, while at the same time minimizing competition among species within the same functional group. Seed dispersal, for example, is an important function that occurs at multiple scales, ranging from the very small in the form of ant dispersal of spring ephemerals to the very large in the form of large vertebrates such as tapirs. Scientists believe resilience is enhanced by the diversity of functions present within a scale and the redundancy of functions distributed across multiple scales. Note that resilience is not driven by the identity of any given element of the system, but rather by the functions those elements provide and their distribution within and across scales.

As of 2011, ecologists detect discontinuities in ecosystems by analyzing animal body-mass distributions. Direct evidence for discontinuities in ecological structure is mounting. Scientists have found discontinuities in animal body-mass distributions in virtually every ecological system they have assessed. They have uncovered patterns that correlate a species location within their particular domain of scale (based on log body mass) to biological phenomena such as invasion, extinction, high population variability, migration, and nomadism. The clustering of these phenomena at the thresholds between domains of scale suggests that variability in resource distribution or availability is greatest at these locations (i.e., discontinuities). These observations support the contention that ecological communities structured by self-organizing dynamics will tend to maintain a similar pattern of discontinuities in animal body-mass distributions despite changes in species composition, at least as long as the processes structuring the system are unchanged.

Origins

The Canadian ecologist C. S. Holling first proposed the concept of ecological resilience in 1973. He recognized that systems perturbed beyond their capacity to recover could shift into an alternative state or regime. Holling preferred the term regime because it emphasizes the controlling processes of a given state of a system. The emphasis on alternative regimes was at odds with prevailing ecological theory of the time, which considered the relevant measure to be return time following perturbation (i.e., engineering resilience). The emphasis on return time was based on the premise that most systems can exist in only one stable state. Holling provided an overview of the origins of the concept in his memoirs:

Up to that time, a concentration on a single equilibrium and assumptions of global stability had made ecology, as well as economics, focus on near equilibrium behavior, and on fixed carrying capacity with a goal of minimizing variability. Command and control was the policy for managing fish, fowl, trees, herds. . . .

The multi-stable state reality, in contrast, opened an entirely different direction that focused on behavior far from equilibrium and on stability boundaries. High variability, not low variability, became an attribute necessary to maintain existence and learning. Surprise and inherent unpredictability was the inevitable consequence for ecological systems. (Holling 2006)

One of the earliest responses to Holling’s original publication was one that refuted it. Wayne Sousa and Joseph H. Connell, both US ecologists, searched the ecological literature to determine if there was evidence for the existence of multistable states in nature. They analyzed published data on time series of population changes of organisms to determine if there was any indication of multiple stable states and found no supporting evidence. This reinforced the single-equilibrium paradigm that dominated in population ecology, and the concept of resilience became, in effect, scientifically dormant.

Over time, as ecologists had access to longer-duration data sets and developed the notion of modeling system behavior (as opposed to the behavior of a part), Holling’s concept of resilience began to reemerge. In the late 1990s a grant from the MacArthur Foundation helped create what is currently known as the Resilience Alliance (2011): a “global consortium of institutions that seeks novel ways to integrate science and policy in order to discover foundations for sustainability.” The alliance includes universities, government, and nongovernment agencies as partners in a program of research and communications with the goal of integrated social, economic, and ecological sustainability. Since the Resilience Alliance created a global research network focused on developing theory and understanding case studies, publications focused on resilience in social-ecological
systems have grown exponentially. Holling’s concept of ecological resilience has overshadowed competing definitions focused on return time.

Impact and Application

A premise of any complex system is that surprise and uncertainty are inherent to the system. Conventional ecosystem management has been slow to address surprise and uncertainty in system behavior and has, to a large extent, struggled over the long term to ensure the consistent delivery of ecological goods and services. On the other hand, scientists’ ability to replicate a complex system within a lab constrains experimentation on ecosystems. One outcome of the development of resilience theory has been the recognition that we need ecosystem management frameworks that explicitly incorporate planning for and managing uncertainty in ecological systems, as well as for emergent phenomena such as resilience. Ecologists developed adaptive management as a way to conduct safe-to-fail experiments for ecosystems and a way to allow management to occur in the face of uncertainty. Managing for resilience therefore consists of actively maintaining a diversity of functions within and across scales, accounting for thresholds and the nonlinear dynamics that occur at thresholds, and implementing adaptive management and governance. Managing for resilience requires an improved understanding of system-level behavior, in addition to specific, detailed knowledge of parts of the system. Systems in undesirable states can also be highly resilient, however. In such cases the manager’s goal is to reduce the resilience of the system and help shift the system to a more desirable regime.

The following propositions constitute the core of managing for resilience:

• Identify the conditions that indicate loss of resilience for the particular system. Recent research demonstrates that there are system-specific conditions that indicate a system is losing resilience and approaching a regime shift. These indicators are measurable and will differ between ecosystems.
• Identify and maintain a diversity of system elements and feedbacks that help keep a system within a desired regime. Maintain the distribution of ecological functions within and across scales that contribute to system resilience.
• Use adaptive management and governance, which are critical to managing for resilience. They treat policy and management options as hypotheses to be put at risk, and thus enhance learning and reduce uncertainty.
• Take management actions to reduce the likelihood of shifts into a different regime. Control invasive species, for example, or monitor and maintain important structuring processes (e.g., fire, hydrology).

Outlook

The success of concepts such as resilience brings with them threats of overuse and misinterpretation. These pitfalls arguably have befallen the concepts of adaptive management and sustainability. Close adherence to the scientific definition and advances in our understanding of regime shifts (and therefore resilience) can prevent this. A challenge to resilience theory is that it is very easy to recognize a system that has undergone a regime shift, but very difficult to recognize when the resilience of a system has been compromised.

Quantification of the resilience of systems is in its infancy and remains poorly developed. Ecologists in the twenty-first century have made advances in detecting early warning of impending regime shifts, usually by focusing on rising variance in key parameters of the ecological system in question. Scientists need to develop leading indicators to manage for resilience, and therefore to develop sound environmental management.

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See also Biodiversity; Complexity Theory; Community Ecology; Disturbance; Eutrophication; Extreme Episodic Events; Fire Management; Fisheries Management; Food Webs; Forest Management; Global Climate Change; Hydrology; Groundwater Management; Landscape Planning, Large-Scale; Mutualism; Natural Capital; Regime Shifts; Shifting Baselines Syndrome; Succession

Further Readings


Rewilding builds on many approaches to conservation as part of a broad vision to restore ecologically disturbed areas to balanced wilderness. It involves connecting large core reserves, thus allowing wildlife to travel unhindered across continents and maintaining populations of top-level predators to regulate prey populations. Rewilding is already occurring on its own in abandoned farm fields, but proponents seek success on a continental scale.

It has been said that prior to the loss of the American chestnut tree in North America, a squirrel could travel the entire length of the eastern United States without ever leaving the treetops. This story may be more ecological fable than scientific fact, but the conservation strategy of rewilding proposes to again allow distance-roving wildlife such as wolves, elephants, and jaguars to travel unhindered across continents.

Rewilding is a conservation vision that emerged in the 1990s aimed at creating systems of connected core reserves on a continental scale that allows large-scale ecological processes and healthy wildlife populations to carry on unhindered or recover from setbacks. The approach focuses on the continued presence—and reintroduction where necessary—of large carnivores in order to maintain top-down species and habitat regulation in ecosystems. Proponents argue that a system of reserves large and connected enough to support viable populations of wide-ranging carnivores would also protect many other species with smaller habitat requirements and would be a sustainable method of maintaining global biodiversity. The approach is based on sound science, researched and promulgated by academics, and it is also supported by environmentalists who are closely aligned with the ethical arguments of wilderness preservation.

Roots of Rewilding

Shifting cultivation, or swidden agriculture (also known as slash-and-burn), could be considered the first small-scale application of rewilding. This agricultural system is based on moving cropped areas frequently so that soils can recover and natural vegetation can regrow in the abandoned fields. It has been used by traditional peoples around the world for centuries and continues to be practiced by subsistence farmers in the developing world.

Both deliberate and passive abandonment of many farm fields has contributed to the rewilding of large areas. Following European settlement of North America, much of the forest in the northeastern United States was cleared for timber and then converted to croplands. Then many farm fields were abandoned during the westward expansion of the country and agriculture, and natural succession led to recolonization of the abandoned fields by trees and shrubs.

In the twenty-first century, some areas in the northeastern United States are reaching presettlement levels of forest cover. And along with the forests comes habitat for wildlife. Populations of species such as white-tailed deer and beaver are rebounding from extreme lows at the peak of deforestation in the early 1900s. Although shifting cultivation and cropland abandonment return formerly utilized land to a more natural and wild condition, neither has the purpose, scale, or planned coordination to achieve rewilding’s goals.

Undeveloped natural areas will always be important for the conservation of biodiversity, but rewilding goes a step further, seeking to restore ecologically disturbed areas to balanced wilderness. The science of ecological restoration began in the early twentieth century and was heavily influenced by the work of Aldo Leopold.
(1887–1948), who taught wildlife ecology at the University of Wisconsin. Along with others at the university, he was involved with starting one of the world’s first restoration projects at the University of Wisconsin–Madison Arboretum by attempting to re-create a tallgrass prairie on an old horse pasture. Leopold is even better known for his work in restoring a worn-out farm in central Wisconsin and his seminal writings about the experience. His essays introduced many of the ethical underpinnings of environmental preservation and restoration, as well as scientific principles that are central to rewilding, such as top-down regulation in ecosystems by carnivores.

The techniques used to restore an ecosystem vary depending on the existing plant and animal communities, past uses of and disturbances to the area, and the restoration goal. Successful restoration projects have clearly stated goals, such as reestablishing a predisturbance vegetation community or creating conditions similar to a chosen reference site if predisturbance conditions are unknown or impossible to reproduce. Restoration ecology has focused more on re-creating lost or degraded plant communities than on establishing large-scale wildlife habitat or reintroducing animal species. Rewilding expands the scale and scope of restoration ecology to the level of continent-wide wildlife populations and naturally functioning ecological processes.

Principles of Rewilding

Rewilding as a conservation strategy blends many scientific disciplines, including ecological restoration, wildlife biology, and ecosystem management, together with public policy tools such as environmental law and international diplomacy. It draws deepest from the field of conservation biology, a synthesis discipline that emerged in the late 1970s in response to a phenomenon many scientists believe to be in progress and largely caused by human activities—the sixth mass extinction of species on Earth. Conservation biology studies the loss and conservation of global biodiversity, from the genetic level up through populations and communities, expanding out to ecosystems across entire landscapes. For rewilding to be successful on a grand scale, it needs input from scientists working at all ecological levels to support its concentration on large animals and ecosystems.

At its heart is the design of large and interconnected networks of protected reserves built around existing core reserves, such as national parks and forests, which are crucial because of the limited number of roads within them. Roads can fragment habitat, introduce non-native species, and enable exploitation of natural resources. To allow wildlife populations to move, the core reserves must be connected by habitat corridors and linkages.

This connectivity is critical to the rewilding strategy of maintaining populations of top-level carnivores. Such wide-ranging species require vast wild areas for persistence; yet even the largest core reserves may not, individually, be big enough to support viable populations. As well, connected core reserves would allow large-scale ecological processes, such as natural disturbance regimes and metapopulation dynamics, to function normally. While connectivity can be achieved directly by corridors of suitable habitat between core reserves, the rewilding strategy also proposes to make the entire landscape more permeable to wildlife movements by identifying barriers, such as major highways, large areas of intensive agriculture, or dams along river ecosystems, and finding ways to mitigate their impact.

The need for large, interconnected core reserves is supported by the theory of island biogeography, which predicts that larger islands that are closer to the mainland will have higher and more persistent levels of biodiversity. A core reserve surrounded by inhospitable habitat is like an island surrounded by water, preventing species travel and limiting diversity. Scientists have debated for decades over what’s required for effective conservation of biodiversity in terms of size and shape of protected areas and level of connectedness between them. Most scientists have come to agree that larger and more-connected reserves better preserve biodiversity than smaller, isolated ones. Rewilding builds on this foundation, adding focus on top-level carnivores, such as gray wolf, brown bear, or sea otter—various “keystone species” at the apex of food
From the shipping lane to the grasslands of the Pantanal with the small but ecologically important Paseo del Jaguar. Farther south there is the Cerrado–Pantanal Ecological Corridors project in Brazil to link the world’s largest northern tip of Columbia. This initiative has struggled with controversy and opposition from indigenous peoples, but it lives on in the renamed Paseo del Jaguar. Large carnivores are important not only for ecological reasons, but also for the ethical rationale of restoring the lost wilderness that they symbolize. Reintroducing large carnivores across continent-wide landscapes may be rewilding’s most ambitious goal.

Rewilding in Practice

Before the term rewilding was coined, large-scale conservation was underway through efforts like the United Nations Educational, Scientific and Cultural Organization (UNESCO) Man and the Biosphere Programme started in the 1970s that now encompasses hundreds of reserves around the world. A recently created and similar program in Africa, called the Transboundary Protected Areas for Peace and Co-operation, provides the framework for establishing parks that straddle political boundaries; for example, the Great Limpopo Transfrontier Conservation Area, which includes the famous Kruger National Park in South Africa, Gonarezhou National Park in Zimbabwe, and three national parks in Mozambique. The largest of these “peace parks” is the Kavango–Zambezi Transfrontier Conservation Area at almost 300,000 square kilometers across five countries.

In Central America, the Paseo Pantera, or Path of the Panther, was proposed in the early 1990s so that a jaguar could wander from Mexico’s Yucatán Peninsula to the northern tip of Columbia. This initiative has struggled with controversy and opposition from indigenous peoples, but it lives on in the renamed Paseo del Jaguar. Farther south there is the Cerrado–Pantanal Ecological Corridors project in Brazil to link the world’s largest grasslands of the Pantanal with the small but ecologically rich Emas National Park. And in southern Asia, the Terai Arc Landscape Project runs along the border of India and Nepal.

These projects all embrace the principles of rewilding while confronting the realities of large-scale conservation. The legacies of colonialism, corrupt governments, widespread poverty, and sometimes warfare all have hindered conservation efforts and led to further environmental degradation, loss of biodiversity, and human suffering.

Several founders of the rewilding concept are working in North America on the Yellowstone to Yukon Initiative (Y2Y) aimed at connecting the world’s first national park, Yellowstone National Park in the United States, with the vast Yukon Territory in northwestern Canada through a massive wildlife corridor of protected areas and roadless lands. Progress on this project is slowed when reintroduction attempts and delisting the gray wolf as an endangered species are bitterly contested in lengthy legal battles, and bison roaming outside the protection of Yellowstone National Park are shot by state wildlife agencies to protect private property and grazing contracts. The Y2Y is the best-known rewilding effort in North America, but smaller projects also exist, such as the Algonquin to Adirondack Conservation Association (A2A) in the Northeast and the Sky Island Alliance in the Southwest.

Even in affluent North America, and even without the added challenges of conservation projects in the developing world, rewilding can create tensions between native cultures, local residents, and outside interests, such as environmental organizations and government agencies. The Malpai Borderlands Group, which spans the Arizona–New Mexico border, seeks to bridge gaps and find common ground among diverse stakeholders that will lead to sustainable use of the region’s natural resources for supporting human communities while maintaining a healthy ecosystem for wildlife.

Impact of Rewilding

Since the idea of rewilding was conceived, several large conservation planning efforts around the world have incorporated its major aspects (cores, corridors, and carnivores), but few on-the-ground projects have achieved the success needed to justify the scale and scope that a full rewilding strategy entails. Rewilding is best applied to landscapes that are not densely populated or developed yet still retain large tracts of wildlands to function as core reserves. Sometimes the focus on protected core reserves and large carnivores can place rewilding at odds with economic and sustainable development projects that attempt to benefit local people. There needs to be further study on how humans will impact (and benefit
From) large-scale conservation projects before rewilding can be an effective and sustainable conservation strategy around the world.

Even without complete implementation of the rewilding approach, however, including its daring and hopeful vision as part of the dialogue in conservation planning can lead to better and more sustainable solutions for preserving global biodiversity.

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See also Agroecology; Biological Corridors; Biodiversity; Boundary Ecotones; Charismatic Megafauna; Community Ecology; Ecological Restoration; Edge Effects; Food Webs; Habitat Fragmentation; Keystone Species; Population Dynamics; Reforestation; Resilience; Species Reintroduction; Succession; Wilderness Areas

Further Reading
Environmental Effects

The negative effects of roads and traffic on animal abundance by far outnumber the positive. Roads and traffic decrease habitat amount and quality. Vehicles collide with and kill wildlife. Roads keep animals from accessing resources on the other side of the road and subdivide animal populations into smaller and more vulnerable fractions. Roads not only take up habitat; edge effects reduce core habitat by an even higher amount. The “road effect zone” describes how far from the road its effects extend. Declines in species abundances range between 40 meters and 2,800 meters for birds, between 250 meters and 1,000 meters and possibly more for amphibians, and up to 17 kilometers for mammals (Forman et al. 2003; Benítez-López, Alkemade, and Verweij 2010).

Many species need to access different types of habitat in the various phases of their lives (e.g., feeding, breeding, overwintering). The subdivision and isolation of subpopulations reduces genetic variability and interrupts metapopulation dynamics—that is, organisms cannot move between different subpopulations and between habitat patches where conditions have become unfavorable and new empty habitat patches. Roads increase the risk of extinction and increase small and isolated populations’ vulnerability to natural stress factors (e.g., adverse weather conditions, fires, diseases). Some species are particularly vulnerable to roads and traffic: many populations of large terrestrial mammals are endangered or live in small numbers, and most of these species have large habitat needs and require movement over long distances. Traffic often kills amphibians, turtles, snakes and other reptiles, and birds. Researchers estimate that annual road-kill numbers vary from a...
ECOSYSTEMS AND PERSIAN RUGS

Let’s start by imagining a fine Persian carpet and a hunting knife. The carpet is twelve feet by eighteen, say. That gives us 216 square feet of continuous woven material. We set about cutting the carpet into thirty-six equal pieces, each one a rectangle, two feet by three. The severing fibers release small tweaky noises, like the muted yelps of outraged Persian weavers. When we’re finished cutting, we measure the individual pieces, total them up—and find that, lo, there’s still nearly 216 square feet of recognizably carpetlike stuff. But what does it amount to? Have we got thirty-six nice Persian throw rugs? No. All we’re left with is three dozen ragged fragments, each one worthless and commencing to come apart.

Now take the same logic outdoors and it begins to explain why the tiger, Panthera tigris, has disappeared from the island of Bali. It suggests why the jaguar, the puma, and forty-five species of birds have been extirpated from a place called Barro Colorado Island, and why myriad other creatures are mysteriously absent from myriad other sites. An ecosystem is a tapestry of species and relationships. Chop away a section, isolate that section, and there arises the problem of unraveling.


hundred thousand in some countries to several 100 million in others. Vehicles annually kill 500,000 large ungulates, or hoofed mammals, in Europe (excluding Russia) and 8.5 million birds in Sweden alone (Seiler 2003). Those species most strongly affected generally reproduce at low rates or have long generation times, occur at low densities (e.g., lynx, wolverine), have large home range sizes, or frequently move over long ranges of the landscape (e.g., various large mammals, some large birds). They may be larger, move slowly (e.g., amphibians), exhibit low car avoidance or low road avoidance, or may be attracted by roads (e.g., some turtles and snakes). Although birds can fly easily across roads, they often take off from vegetation next to the road and do not gain height fast enough to escape passing vehicles (Jaeger et al. 2005; Fährig and Rytwinski 2009).

The more roads traverse their habitat, the less likely these populations are to survive. Several studies found that at road densities above certain levels, some species do not occur anymore (summarized in Robinson, Duinker, and Beazley 2010). When the threshold in road density has been reached, the next new road is likely to bring about the populations’ extinction. (See figure 1 on page 346.) Even worse, when road density crosses the “point of no return” and the population is already in decline, even relatively drastic measures will not reverse the trend, and it will be impossible to rescue the population.

Roads facilitate the spread of invasive species and enhance human access to wildlife habitats. Consequently, humans hunt, poach, convert, and deforest land, extract resources, and create other forms of disturbance in these areas more intensely. Roads significantly affect entire communities, ecosystems, and various ecosystem services. (See table 1 on page 347.) Species movement, water-related services, and erosion prevention are examples of regulation and maintenance services that roads affect. In addition, roads alter provisional services by creating small land parcels that reduce profits and lower the quality of agricultural products grown alongside them. Roads also influence cultural services. Although roads make recreational areas more accessible, they permeate the landscape with technology. Their fragmentation alters people’s perception of recreational areas as less connected. Noise and air pollution are spread more widely by roads and affect landscape quality and human well-being. These findings apply to other transportation infrastructure as well, such as railways, pipelines, and ski lifts.

Conservation and Planning

Ecologists find it difficult to quantify the effects of roads on animal and plant populations because they do not know the full extent of the ecological effects of landscape alterations until decades after they are implemented. Even
introducing obligatory insurance for such ecological risks to increase the level of accountability for uncertain ecological effects and effects with long time lags. This would consist of a funding mechanism to pay for the monitoring and repair or compensation of unanticipated ecological damages observed years after the road was constructed.

Roads and rising traffic volumes increasingly fragment landscapes all over the world, and this trend will continue in the future, particularly in eastern Europe, China, India, and Latin America. The 2,594-kilometer transcontinental Inter-Oceanic Highway in Brazil and Peru, completed in 2011, is a dramatic example. It connects the Atlantic to the Pacific and dissects the Amazonian rain forest. In 2010, the Tanzanian government proposed the construction of a major 480-kilometer highway through Serengeti National Park. The highway would devastate many ecosystems and wildlife populations, such as migrating herds of wildebeest and zebra (Dobson et al. 2010). These plans directly contradict the purpose of the renowned park.

Policy makers and planners should protect remaining large unfragmented areas with high priority. The scientific literature emphasizes the importance of large roadless areas to conserve biodiversity (Selva et al. 2011).
Table 1. Effects of Roads and Traffic on Environment and Ecosystem Services

<table>
<thead>
<tr>
<th>Theme</th>
<th>Consequences of Roads</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land cover</td>
<td>Land occupation for road surface and shoulders</td>
</tr>
<tr>
<td></td>
<td>Soil compaction, sealing of soil surface</td>
</tr>
<tr>
<td></td>
<td>Alterations to geomorphology (e.g., cuts, embankments, dams, stabilization of slopes)</td>
</tr>
<tr>
<td></td>
<td>Removal of vegetation, alteration of vegetation</td>
</tr>
<tr>
<td>Local climate</td>
<td>Modification of temperature conditions (e.g., heating of roads, increased variability in temperature)</td>
</tr>
<tr>
<td></td>
<td>Accumulation of cold air at embankments of roads (cold-air buildups)</td>
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<tr>
<td></td>
<td>Modification of humidity conditions (e.g., lower moisture content in the air due to higher solar radiation, stagnant moisture on road shoulders due to soil compaction)</td>
</tr>
<tr>
<td></td>
<td>Modification of light conditions</td>
</tr>
<tr>
<td></td>
<td>Modification of wind conditions (e.g., due to aisles in forests)</td>
</tr>
<tr>
<td></td>
<td>Climatic thresholds</td>
</tr>
<tr>
<td>Emissions</td>
<td>Vehicle exhaust, pollutants, fertilizing substances leading to eutrophication (excess nutrients in water bodies lead to excessive plant growth)</td>
</tr>
<tr>
<td></td>
<td>Dust, particles (abrasion from tires and brake linings)</td>
</tr>
<tr>
<td></td>
<td>Oil, fuel, etc. (e.g., results of traffic accidents)</td>
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<tr>
<td></td>
<td>Road salt</td>
</tr>
<tr>
<td></td>
<td>Noise</td>
</tr>
<tr>
<td></td>
<td>Visual stimuli, lighting</td>
</tr>
<tr>
<td>Water</td>
<td>Drainage, faster removal of water</td>
</tr>
<tr>
<td></td>
<td>Modification of surface water courses</td>
</tr>
<tr>
<td></td>
<td>Lifting or lowering of groundwater table</td>
</tr>
<tr>
<td></td>
<td>Water pollution</td>
</tr>
<tr>
<td>Flora/Fauna</td>
<td>Death of animals caused by road mortality (partially due to animals’ attraction to roads, i.e., “trap effect”)</td>
</tr>
<tr>
<td></td>
<td>Higher levels of disturbance and stress, loss of refuges</td>
</tr>
<tr>
<td></td>
<td>Reduction or loss of habitat; sometimes creation of new habitat</td>
</tr>
<tr>
<td></td>
<td>Modifications of food availability and diet composition (e.g., reduced food availability for bats due to cold-air buildups along road embankments at night)</td>
</tr>
<tr>
<td></td>
<td>Barrier effect, filter effect to animal movement (reduced connectivity)</td>
</tr>
<tr>
<td></td>
<td>Disruption of seasonal migration pathways, impediment of dispersal, restriction of recolonization of empty habitats</td>
</tr>
<tr>
<td></td>
<td>Subdivision and isolation of habitats and resources, breaking up of populations</td>
</tr>
<tr>
<td></td>
<td>Disruption of metapopulation dynamics, genetic isolation, inbreeding effects and increased genetic drift, interruption of the processes of evolutionary development</td>
</tr>
<tr>
<td></td>
<td>Reduction of habitat below required minimal areas, loss of species, reduction of biodiversity</td>
</tr>
<tr>
<td></td>
<td>Increased intrusion and distribution of invasive species, pathways facilitating infection with diseases</td>
</tr>
<tr>
<td></td>
<td>Reduced effectiveness of natural predators of pests in agriculture and forestry (i.e., biological control of pests more difficult)</td>
</tr>
<tr>
<td>Landscape scenery</td>
<td>Visual stimuli, noise</td>
</tr>
<tr>
<td></td>
<td>Increasing penetration of landscapes by roads, posts, and wires</td>
</tr>
<tr>
<td></td>
<td>Visual breaks, contrasts between nature and technology; occasionally vivification of landscapes (e.g., by avenues with trees)</td>
</tr>
<tr>
<td></td>
<td>Change of landscape character and identity</td>
</tr>
<tr>
<td>Land use</td>
<td>Consequences of increased accessibility of landscapes for humans due to roads, increase in traffic volumes, increased pressure for urban development and mobility</td>
</tr>
<tr>
<td></td>
<td>Farm consolidation (mostly in relation to construction of new transport infrastructure)</td>
</tr>
<tr>
<td></td>
<td>Reduced quality of agricultural products harvested along roads</td>
</tr>
<tr>
<td></td>
<td>Reduced quality of recreational areas due to shrinkage, dissection, and noise</td>
</tr>
</tbody>
</table>

Note: Excludes effects of construction sites such as soil excavation and deposition, vibrations, and acoustic and visual disturbances.

Source: Jaeger (2003), based on various sources.
Planners also need to prevent further loss and fragmentation of habitats in landscapes that are already fragmented by identifying areas where further fragmentation is an imminent threat and rapidly preserving them. This task is particularly urgent in regions with a rapid pace of development, such as large parts of eastern and central European countries. Urban sprawl leads to more road construction and higher traffic volumes, and roads attract urban development. Therefore, regional planning legislation should require local and regional authorities to treat land sparingly. Settlement boundaries and green belts can ensure that built-up areas leave clear open spaces.

Mitigation Measures

Current trends of landscape fragmentation clearly contradict the principles of sustainability. Scientists and nongovernmental organizations should inform decision makers and the general public about the problems of landscape fragmentation and habitat loss and about suitable measures to combat them. Planners need to consider four types of measures: (1) minimize negative impacts during the planning and construction stages of new roads, (2) restore connectivity across existing roads, (3) protect roadless areas and prevent further increase of the density of the road network, and (4) remove existing roads. Restoring damaged or severed wildlife corridors re-creates opportunities for species to move. National and international defragmentation strategies should coordinate these efforts and identify regionally important unfragmented areas and priority areas for defragmentation. The Pan-European Ecological Network (PEEN), under the aegis of the Council of Europe, the United Nations Environment Programme, and the European Centre of Nature Conservation (ECNC), is an example of an international organization working on mitigation issues. Climate change adaptation networks are likely to gain importance in the future as well.

Tunnels, wildlife overpasses, underpasses, and pillars that raise the road are the most common measures that allow wildlife to cross. They take advantage of the relief of the landscape, for example, through the use of wider bridges across streams. The story of the badger in the Netherlands is an encouraging example. A national defragmentation program established in 1984 addressed the decline of the badger populations observed since the 1970s. Culverts (so-called badger pipes) combined with other measures to stop the decline, and the populations since have recovered (Dekker and Bekker 2010).

Wildlife-crossing structures often need to be combined with fences to reduce traffic mortality. Although fences increase traffic safety and protect animals from collisions, they increase the barrier effect of roads if they are used without crossing structures. The outcome of this trade-off for the population depends on traffic volume and the behavior of the animals at the road (Jaeger and Fahrig 2004). It is often unclear in what situations fences are an advantage or a disadvantage for wildlife populations. Fences slow the decrease of wildlife populations, but they need to be combined with wildlife passages. (See figure 2 on page 349.) These measures are standard for new roads in some parts of the world, but retrofitting existing roads is not as common. There must be enough habitat left in the landscape for passages to be effective, though. Decision makers may consider road construction unproblematic if they combine new roads with wildlife passages and fences. This attitude ignores the other negative effects of roads and the critical importance of habitat amount. The conservation and restoration of wildlife habitats thus must be the first priority. Wildlife passages will be useless if there is not enough habitat left to connect.

It is better to upgrade existing roads than to construct new roads at another location, even though the widening will increase their barrier effects. Siting bypasses closer to urban areas preserves larger unfragmented areas. Governments should remove roads not urgently needed or reduce the width of roads on which traffic volumes have decreased.

Continued landscape fragmentation will increase the cost of reconnecting isolated habitats, restoring wildlife corridors, and rescuing endangered wildlife populations. Wise policy therefore avoids an increase of fragmentation from the start, in particular because even ecologists do not know when wildlife populations will reach the point of no return. Future regulations should link funds for road construction to the analysis of the cumulative effects of new roads on landscape fragmentation and the protection of unfragmented areas.

Targets and limits for the future degree of landscape fragmentation would allow governments to justify their decisions and actions toward better protection of the environment. The German Federal Environment Agency, for example, has proposed limits to curtail landscape fragmentation. Governments also require a management approach that addresses the remaining uncertainties that are irreducible to a large degree. Such a precautionary approach would open up promising new lines of action for landscape management. Decision makers need to communicate and educate the public, create economic or market-based instruments, and promote changes in travel behavior.

The Future

Scientists monitor the environment to discover and better understand changes. The level to which roads fragment
the landscape is an essential indicator of various threats to biodiversity, to the sustainability of human land use, and to landscape quality. Planners should implement these indicators in their monitoring systems of biodiversity, sustainable development, and landscape quality. Scientists need to track the changes in landscape fragmentation to diagnose the rate of increase and changes in trends. Studies should more carefully observe the effects of new roads as well as wildlife passages and other mitigation measures using the before-after-control-impact study design, which compares data from before and after the construction of the road with other locations where no road was built (Roedenbeck et al. 2007).

In addition, planners need to consider more seriously the uncertain effects of roads. Many species’ long response times to increases in road density present a particular challenge. Scientists are unlikely to know the exact thresholds for wildlife populations any time soon. Any hopes planners have for a general hard number for the maximum acceptable level of fragmentation are likely to be disappointed. The fast pace of road development by far exceeds our understanding of the effects of roads on the environment and biodiversity, which makes appropriate adaptive management impossible. This dilemma makes it all the more essential that decision makers adopt a precautionary approach that curtails landscape fragmentation while continuing research fills the remaining knowledge gaps. Implementation of targets and limits requires a consultation process like that for environmental standards for water and air quality.

Research has been unable to catch up with the ecological effects of the rapid increase in road densities. Decision makers often argue that they need more research results before they include more substantive mitigation measures or slow down road construction. This attitude flies in the face of the principles of sustainability and is contrary to the precautionary principle.

The relationship between the level of landscape fragmentation and biodiversity urgently requires further research. This work needs to consider the response times of species to the deterioration of their environment (extinction debt) and therefore needs to include historic states of the landscape. Ecologists and transportation agencies need to establish collaborative links. Many road agencies have environmental sustainability as one of their goals. The only way to achieve such goals is for them to support long-term scientific research. Independent research undertaking numerous small-scale projects cannot provide the information required to quantify the negative effects of roads and traffic and the positive

Figure 2. Wildlife Overpass across a Four-Lane Motorway in Hungary

Wildlife overpasses such as this one in Hungary are standard for new roads in some parts of the world, but retrofitting existing roads is still rare. In order to be effective, there must be enough intact habitat left in the landscape.
effects of mitigation measures. Researchers will enhance the future of road ecology when they combine and integrate multiple road projects in different states or countries as part of integrated, well-replicated studies (van der Ree et al. 2011).

Large unfragmented areas are a limited and nonrenewable resource. This fact is particularly important to consider in regions where high human population densities compete with biodiversity for land. Land and soils are finite, and their destruction is irreversible within human life spans. This issue is going to increase considerably in the future. Renewable energy supply requires large tracts of land, food production necessitates arable and pasture land with suitable soils, and urban-industrial purposes, transport, resource extraction, refuse deposition, and recreation all compete for land. No form of adaptation can circumvent these growing demands. The German landscape ecologist Wolfgang Haber has called humankind’s increasing needs for energy, food, and land the three major “ecological traps” that threaten humans probably more severely than any other environmental problem (Haber 2007). If endeavors for promoting sustainable development disregard these three ecological traps, they will inevitably miss their goals. Policy makers must make much greater efforts now to conserve unfragmented landscapes.

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See also Biological Corridors; Boundary Ecotones; Buffers; Edge Effects; Fencing; Forest Management; Habitat Fragmentation; Landscape Planning, Large-Scale; Refugia; Rewilding; Stormwater Management; Wilderness Areas

FURTHER READING


Dobson, Andrew P., et al. (2010). Road will ruin Serengeti.


The safe minimum standard is a guide for managing catastrophic risk. Originally designed for the conservation of natural resources, the rule can be applied to any area when management decisions entail a probability of catastrophic outcome and when actions to be taken can be categorized as risky or non-risky. The rule specifies that decisions should be chosen from the non-risky set of possible actions.

The safe minimum standard (SMS) is a policy that attempts to eliminate risk of catastrophic outcomes in the management of natural resources. SMS has been advocated as an applicable rule when the probabilities associated with different actions can be taken are difficult or impossible to quantify and standard economic calculations of costs and benefits are unreliable. The SMS was originally proposed by the German natural resource economist Sigfried von Ciriacy-Wantrup (1906–1980) as a policy that could be applied to conservation issues. Proponents argue that the SMS is a rational way of responding to the risk of catastrophes; others argue that the SMS is too blunt an instrument to be applied as a general rule.

The Concept of SMS

The SMS is an intuitively obvious and simple way of dealing with catastrophic risk: if there is a possibility that an action will trigger unacceptable consequences, do not take that action. The concept does, however, demand a few prerequisites for practical implementation. First of all, it must be possible to specify which actions imply the potential for catastrophic risk and which actions do not. For example, in managing greenhouse gas emissions it is unclear to many what the threshold is for emissions that will cause irreversible climate change; this makes setting a SMS very difficult. Second, the term unacceptable consequences must be properly defined. For example, is the loss of a species of fish an unacceptable consequence if a fish of greater economic value is able to thrive in its place? Third, even if risky actions are deemed unacceptable, it may very well be that the cost of restricting the choice of actions to the ones without risk may also be “unacceptably” high, in which case a cost-benefit calculation of some sort becomes unavoidable. SMSs specify allowable actions only for a specific risk, so if another risk is to be considered, there can be contradictory signals for what action is acceptable. The term unacceptable consequences is subjective, so SMS is not an objective policy tool that can be mechanically applied on a case-by-case basis.

An alternative way of thinking about SMSs is that they may function as temporary policy rules until a better understanding of risks, costs, and benefits of available actions are available. This “stop sign” interpretation of SMS makes the concept similar to the concept of the precautionary principle, which is widely accepted as a sensible and more flexible policy rule when managing potentially catastrophic risk. (The precautionary principle posits that until a risk—such as the potential ramification of a new technology—has been studied, it is best to avoid that risk.)

Resource Management Problems and SMSs

There are many situations for which SMSs may be an appropriate policy guide. In ecology, for example, there are minimum viable populations (MVPs): if a population of organisms declines below a certain level, population growth becomes negative, and the population becomes
extinct. If the MVP is known, then that value is a candidate for a SMS. If the MVP is unknown, a SMS may then be prescribed as the maximum value of the range of possible MVPs. Eutrophication provides another example. If accumulated nutrient deposition in a lake from agriculture crosses a certain threshold, this may trigger algae blooms, with associated deterioration of water quality and species composition. This process may be irreversible, so that even if nutrient depositions are stopped, the lake will not revert to its original state. This threshold is also a candidate for a SMS. A tipping point with consequences on a larger scale includes the possibility of greenhouse gas emissions inducing runaway climate change that causes irreversible sea level rise and temperature change.

In all these examples, it is possible to deduce the existence of critical boundaries that, if they exist and are in fact crossed, lead to a fundamental change in the system dynamics. They thus satisfy the criterion for implementation of a SMS. Nonetheless, defining an unacceptable consequence is arguably a subjective choice and in some—but not all—cases is debatable. For example, intensive agriculture practices can deplete soil to irreparable levels, causing desertification; some might argue, however, that intensive agricultural practices are necessary to provide food for starving people. While a SMS might be put in place to prevent desertification, those who seek to combat current food challenges might argue that the SMS limits their ability to manage another unacceptable risk—starvation.

Criticisms of SMSs

SMSs have been criticized on many grounds, but mostly because the economic rationale for SMSs is considered poor. The criticism may be summarized as follows (Randall 2011, 174): even if a particular ecosystem collapses, there are substitutes. If one lake eutrophies, that is not such a big disaster as long as other lakes are preserved. Imposing an SMS may thus imply a decision to avoid risky outcomes that are not truly catastrophic. The revenue from harvesting trees may, for example, be determined a greater benefit than leaving the forest ecosystem intact even though it will be a long time before the forest is regenerated. Finally, economic resource management tends to be incremental in nature and rarely prescribes that a resource should not be used at all. SMSs implicitly assign a value that reduces the risk to zero, which most economic management practices find arbitrary. Economists have been reluctant to accept SMSs as a policy tool because of the contrast with traditional economic methods of cost-benefit calculations and improved economic techniques for evaluating catastrophic risk. When there are suspected tipping points, however, they often prescribe SMSs as the optimal policy from an economic perspective, since tipping points indicate the possibility of irreversible change that can have indeterminate economic costs (Margolis and Nævdal 2008).

Proponents of economic cost-benefit analysis often cite the inherent subjectivity in defining “unacceptable costs” when deciding on the adoption of a SMS as an argument against its use. Advocates of SMS, however, counter that economic cost-benefit analysis is founded on a theory of what is valuable (utilitarianism), which is also a subjective moral philosophy. Therefore, advocates of SMSs claim that cost-benefit analysis is not in any sense a more objective policy tool than SMSs.

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See also Adaptive Resource Management; Administrative Law; Best Management Practices (BMP); Carrying Capacity; Complexity Theory; Community Ecology; Ecological Forecasting; Forest Management; Keystone Species; Mutualism; Population Dynamics; Regime Shifts; Resilience

FURTHER READING


Technologies for extracting gas from shale by using high-volume hydraulic fracturing (commonly known as “fracking”), a process that breaks up the rock and releases the gas, have developed since about the year 2000. Although many see shale gas as a viable alternative to other fossil fuels, especially natural gas from conventional sources, the environmental costs are high. Particular concerns include water and air pollution as well as emissions of greenhouse gases.

Natural gas makes up some 20 percent of all energy use globally (IEA 2011) and 24 percent in the United States (US EIA 2011). Most natural gas is obtained by drilling a well into a pocket of gas trapped beneath an impermeable layer within the Earth. The gas simply flows through the well to escape and reach the surface. People have gradually been depleting such sources of “conventional” natural gas, however, and are increasingly turning to “unconventional” sources: gas that is tightly held in rocks with very low permeability such as shale, some sandstones, and coal seams. High-volume hydraulic fracturing (fracking) is a controversial method that is used to extract natural gas from unconventional sources.

The fracking process involves forcing water or other fracturing fluid into a gas or oil wellbore to create fractures and small fissures in the underlying rock, which will release the gas and thus increase the production of the wells. The technique has been used since the 1940s in conventional gas and oil wells, but until recently, only with modest quantities of water—a few hundred thousand liters of water per fracking event per well at most. (Exactly how often industry may re-frack wells, or if they do so at all at significant levels, remains unknown, as the technology is simply too new.) The industry started experimenting with high-volume hydraulic fracturing of shales in Texas in the mid-1990s, but using relatively low volumes, and relatively few wells. By 2003 or so, they were using larger volumes, and starting to use the approach for significant production of shale gas, but still only in Texas. They started later in that decade to move into other states, and to increase the volumes used. Mostly, this practice is still quite new. For instance, in the Marcellus shale in Pennsylvania, significant shale gas extraction began only in 2009.

To get unconventional gas from shales, producers began to use much larger volumes of water as well as many chemical additives, combined with high-precision horizontal drilling of wells. With high-precision drilling, workers can bore down into the Earth to depths of 3 kilometers or more, then curve the well and drill sideways for another 2 kilometers or more, closely following within a vein of a particular gas-rich shale. (See figure 1 on page 355.) The shale rock is then fractured by forcing large volumes of water—20 million liters of water per well on average—and additives through the well at high pressure. In addition to the chemical additives discussed in the section “What Goes Down / What Comes Back Up,” many of which have not been identified to the public, sand or other fine particles are included in the mixture to help keep the new fissures open to aid in the flow of gas from the rock and up the well.

Shale Gas Is New

The development of unconventional gas by fracking is fairly new, having started slowly with gas in tight sandstones in the 1980s, then in coalbeds in the early 1990s, and in shale in the late 1990s. In the United States, production of gas from tight sandstones and coalbeds has already peaked, and only shale gas production is expected
to increase over the coming decades (US EIA 2011). Shale gas development began in Texas, and Texas still dominates the production of shale gas globally, although some commercial production came online in Arkansas, Louisiana, and Pennsylvania between 2007 and 2009. In 2007, shale gas contributed only 1 percent of the supply of natural gas to the United States. By 2009, this had grown to 14 percent, and the US Department of Energy has projected that shale gas will supply 45 percent of the country’s natural gas by 2035 (US EIA 2011), although many researchers believe this is too optimistic (Howarth and Ingraffea 2011). Outside of the United States, exploratory shale gas wells have been drilled in Quebec and British Columbia, Canada, and in a few European countries, but no shale gas wells have gone into commercial production in any of these places. Many parts of the world contain possible shale gas resources, and some researchers predict a massive global explosion in shale gas development (Engelder 2011). Geologists do not know precisely how large the world’s reserves of shale gas are, however, and a report by the US Geological Survey in August 2011 (Coleman, J. L. et al. 2011) cast doubt on the more optimistic estimates (Howarth and Ingraffea 2011).

Scientific study of shale gas extraction’s environmental effects is in the early stages. In fact, the first studies of the environmental consequences of shale gas were published in 2011 (Howarth and Ingraffea 2011). In 2005, the US Congress exempted fracking from most federal environmental oversight and regulation, which has made it difficult to obtain information on fracking’s effects from industry sources. Nonetheless, a growing body of evidence indicates reason for concern.

What Goes Down/What Comes Back Up

In addition to the huge volume of water used, roughly 200,000 liters of chemicals are added to a well during the fracking process. These include acids to assist with
opening up fissures, biocides to prevent microorganisms from growing and clogging up the fissures, scale inhibitors to reduce corrosion of the pipes, and surfactants to reduce the friction of the high volume of water at high pressure traveling through the long pipe runs of the well. Many of these chemical additives are toxic, mutagenic (causing birth defects), or carcinogenic (causing cancer). The exact composition of the additives used has not been disclosed to the public because the 2005 congressional fracking exemption allows the industry to keep the chemical list secret (Howarth and Ingraffea 2011). Similarly, Canada has no requirements about disclosing the composition of fracking chemicals (De Souza 2011). Nonetheless, although disclosure is not the industry standard, more drilling companies are voluntarily reporting their chemical mixtures as the public becomes more vocal about obtaining the information.

When injected into the well, the fracturing water and chemicals extract additional materials from the rock formations, including toxic heavy metals, organic materials (including some such as benzene that are toxic and carcinogenic), and radioactive substances such as thorium, radium, and uranium. The fracking of a well generally takes less than a day. During the following two weeks or so, some of the water (approximately one-fifth of what was added) together with the additives and extracted materials, flows back to the surface. These mixtures are called flow-back fluids.

Treatment and Disposal of Flow-Back Fluids

When the flow-back fluids come to the surface, they are stored in open pits or in tanks until they are treated or disposed of. In Texas, the industry disposes of most of the flow-back fluids by injecting them into old, abandoned conventional oil and gas wells. Elsewhere in the United States, there are not enough abandoned wells to provide sufficient disposal capacity, and other approaches are necessary. In Pennsylvania, for example, most flow-back fluids have been trucked to municipal sewage treatment plants. Unfortunately, these facilities are not designed to handle the toxic materials, and much of the waste has simply flowed through the sewage plants and been discharged into rivers (Howarth and Ingraffea 2011; Urbina 2011a). In the summer of 2011, the State of Pennsylvania outlawed using sewage treatment plants for flow-back fluid disposal. The natural gas industry is attempting to develop effective and nonhazardous disposal methods, such as to recycle the fluids and reuse them in fracking. To date, only small percentages of flow-back fluids have actually been recycled (Urbina 2011b), and the future for waste disposal is highly uncertain.

Water Pollution

Improper disposal of flow-back fluids can lead to surface water pollution, and, in addition, the development of unconventional gas may contaminate groundwater. Freshwater aquifers are usually at shallow depths underground, within the top 100 meters or so, while the shale gas is at depths of a kilometer or more. Despite this distance, evidence indicates that in at least some cases, fracking fluids actually have entered surface aquifers (Urbina 2011d). One mechanism for the contamination may be leaks in the well pipes as they pass through the aquifer. Another possibility is that the high pressure used in fracking forces the fluids up through nearby older, abandoned wells, and these in turn leak into the groundwater aquifer. The contamination of groundwater by fracking fluids has received little study or scrutiny, however, in part because information about it has been sealed and kept from the public when drilling companies settle lawsuits with landowners whose water has been contaminated (Urbina 2011d).

More common than contamination with fracking fluids is contamination with methane gas. A team of scientists from Duke University demonstrated high levels of methane contamination in many private drinking water wells within one kilometer of gas wells in Pennsylvania (Osborn et al. 2011). Water wells at greater distances from gas wells sometimes had methane contamination too, but at much lower concentrations. Natural gas is composed mostly of methane, and this study proved that the high levels of methane contamination came from the deep shale gas, and not from other sources of methane closer to the surface (such as bacteria in waterlogged soils). The Duke team did not find fracking fluid contamination in the water wells they sampled, and methane is not toxic. The methane did occur at levels that pose a major risk of explosion, however. Furthermore, the fact that methane could migrate from the deep shale formation into surface water wells suggests that other gases from the shale, such as benzene vapor, may also be migrating and contaminating the wells.

Air Pollution

Shale gas development is a major industrial enterprise that results in sometimes severe air pollution (Howarth and Ingraffea 2011). Large numbers of trucks haul water to the wells and flow-back fluids away. Massive diesel engines are used to drive the drills through kilometers of rock, and ten-thousand-horsepower diesel
engines drive the pumps for the actual fracking. More engines run compressors to deliver the gas through pipelines. Toxic organic gases and vapors—compounds such as benzene and toluene—are routinely vented and leaked into the air. Cumulatively, these emissions can lead to high levels of ozone, which pose a risk to human health but also adversely affect the vegetation of natural ecosystems. Since shale gas drilling began in rural Colorado, ozone concentrations in the once pristine air have often approached or exceeded the regulatory standard set by the US Environmental Protection Agency (CDPHE 2010).

In Texas and Pennsylvania, state regulatory agencies routinely measure benzene concentrations in the air at levels that pose a significant risk of cancer from chronic exposure, and at times in Texas the concentrations exceed the acute public health standard (Howarth and Ingraffea 2011).

Greenhouse Gas Emissions

Shale gas has been widely promoted as a clean fuel, one with fewer greenhouse gas emissions than coal or oil, and therefore suitable as a bridge fuel that would let society continue to rely on fossil fuels while reducing global warming to some extent. Shale gas does indeed produce less carbon dioxide than coal or oil for an equivalent amount of energy, but this is only part of the emissions story. Methane is an incredibly powerful greenhouse gas—105 times more potent than carbon dioxide over a twenty-year period following emission (Shindell et al. 2009). Consequently, even small leakages of shale gas, which is mostly methane, have a huge influence on the greenhouse gas footprint of shale gas.

In April 2011, the first comprehensive analysis of emissions of all greenhouse gases from shale gas development, including methane as well as carbon dioxide (Howarth, Santoro, and Ingraffea 2011), evaluated the venting (purposeful emission) and leakage (accidental) of methane from the time of fracking and well completion through the processing of gas and delivery to the final consumer. The analysis found that shale gas development emits more methane than does conventional natural gas, due to a large venting of gas during the two-week flow-back period following fracking. Methane emissions dominate the greenhouse gas footprint of shale gas, giving this fuel a larger footprint than any other fossil fuel when considered over a twenty-year period. (See figure 2 for a comparison of greenhouse gas emissions for different types of fuel.) As time goes on, though, the influence of methane is diminished, as methane is removed from the atmosphere some ten times faster than is carbon dioxide. Nonetheless, the greenhouse gas footprint of shale gas is comparable to that of other fossil fuels over periods of up to one hundred years or more when used to generate heat (the primary use of natural gas) and over periods of up to fifty years when used to generate electricity (Howarth and Ingraffea 2011; Howarth, Santoro, and Ingraffea 2011, 2012; Hughes 2011). That is, shale gas extracted by fracking should not be used as a bridging fuel if society is to reduce global warming and avoid tipping points in the global climate system over the coming decades.

Figure 2 on page 358 illustrates the total greenhouse gas footprint of shale gas in comparison to other fossil fuels, considered at the integrated time scale for the 20 years following emissions. The footprints for each fossil fuel represented by the six columns in figure 2 are divided into three segments: (1) the direct emission of carbon dioxide from burning the fuel (as indicated by the bottom portion of each column); (2) indirect emissions of carbon dioxide necessary to develop and use the fuel, including for example trucking water to a fracking site and carrying coal in trains (as indicated by the small sliver in the middle of each column); and (3) methane emissions, converted to equivalents of carbon dioxide over a twenty-year integrated time period (as indicated by the top segment of each column). The figure provides both low and high estimates for methane emission rates from shale gas and from conventional gas as well as for surface and deep-mined coal. Note that while methane is emitted from coal mining and oil wells, the amount is small compared to the leakage from natural gas.
Shale Gas and the Environment

Shale gas development is relatively new, and the industry is still developing techniques and technologies. Research into reducing the environmental consequences of the process is ongoing. For instance, there is an urgent need to develop appropriate methods for treating and disposing of flow-back fluids. Whether this will happen, and whether new treatment technologies such as recycling the wastes can be done in a manner industry considers economical, remains to be determined. An important factor is that the cost of extracting the shale gas is high compared to the market price of natural gas (Howarth, Santoro, and Ingraffea 2011b; Urbina 2011c). From 2009 through 2011, natural gas prices hovered near four dollars per thousand cubic feet of gas, yet the break-even point for companies to turn a profit in developing shale gas is probably greater than six dollars per thousand cubic feet, but “if history is a guide, the cost of production of any new resource always drops over time” (RTEC n.d.), which means initial high costs may not be a deterrent to producers.

The technology for reducing methane emissions, thereby reducing the greenhouse gas footprint of shale gas, is well developed. The gas vented during the flow-back period can be captured and sold, rather than released to the atmosphere. But at current prices the value of the gas is small compared to the cost of the capture, giving industry at best a small return on investment. As a result, industry captures the gas less than 15 percent of the time when they complete wells (Howarth, Santoro, and Ingraffea 2012). Further, much of the methane emissions

Figure 2. The Greenhouse Gas Footprint of Shale Gas


The greenhouse effects of shale gas are predicted to exceed those of conventional gas, oil, and coal.
cannot be captured, because it is in the form of leaks from pipelines as the gas is pumped to consumers. In the United States, the average pipeline is more than fifty years old (CEQ 2004), and both long-distance transmission pipelines and distribution pipelines within cities can be quite leaky. The price of replacing these pipelines with modern technology is very high and of questionable value if the goal is to use shale gas as a bridge fuel for two or three decades before moving to truly green, renewable sources of energy.

**Outlook**

Shale gas is widely distributed across the planet. As society depletes conventional sources of natural gas and other fossil fuels, shale gas suggests to some the potential to continue to rely on fossil fuels over the coming decades. The environmental consequences are high, however, with widespread water and air pollution. Further, shale gas has a larger greenhouse gas footprint than any other fossil fuel, when evaluated over a period of fifty years or less. The environmental consequences are high, however, with widespread water and air pollution. Further, shale gas has a larger greenhouse gas footprint than any other fossil fuel, when evaluated over a period of fifty years or less following emission. As a result, reliance on this resource using existing extraction technology will tax the planet. Improvements in technology and better capture of wastes and emissions are necessary if shale gas is to be part of a sustainable future.

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See also Groundwater Management; Hydrology; Pollution, Point Source; Waste Management; Water Resource Management, Integrated (IWRM)

**FURTHER READING**


In 1995, fisheries scientist Daniel Pauly speculated that baselines used to measure change in his field shifted with every generation of researcher, and this “shifting baselines syndrome of fisheries” contributed to ruinous fisheries management, fish stock collapse, and continued unsustainable fishing. Since then, global research has shown that many marine species and ecosystems have declined by 90 percent from historical conditions, and that the notion of sustainability itself depends upon baseline choice.

In 1995, Daniel Pauly, an eminent French fisheries scientist and foe of overfishing, coined the term shifting baselines syndrome of fisheries to describe the tendency of each generation of fisheries scientists (as well as managers and policy makers) to compare stock conditions to baselines determined by their own first observations. Because institutional memory resets every twenty or thirty years, subsequent definitions of normal conditions incorporated an imperceptible but steady decline in the standards for assessing fish populations and marine ecosystems. Over long time periods, significant degradation in the oceans accumulated without detection and was not accounted for in mathematical models that scientists and managers used to regulate fisheries.

The collapse of Newfoundland cod stocks in 1992, arguably the worst fisheries catastrophe in history, profoundly influenced Pauly’s thinking. Although warning signs were evident in retrospect, the collapse took almost everyone by surprise. Scientists labored to discover what had happened. Then at the University of British Columbia (UBC) in Vancouver, Pauly proposed that the shifting baselines syndrome prevented fisheries scientists and managers from realizing the magnitude and trajectory of declining fish stocks. By incorporating anecdotes and historical perspectives into fisheries science, Pauly reasoned, this conceptual blind spot might be avoided.

The shifting baselines syndrome challenged science and management to rethink the nature of evidence and find a way to bring historical information into a modern scientific framework. One scientist to take this on was marine ecologist Jeremy B. C. Jackson of the Scripps Institution of Oceanography, who worked on Caribbean coral reefs and had long been fascinated by historical descriptions of the Caribbean and the different sea they seemed to describe. For example, in the present day all species of sea turtles are endangered worldwide, but historical descriptions of the Caribbean indicate that sea turtles were once “so numerous that it seemed that the ships would run aground on them” (Jackson 1997).

Jackson put together an international working group of scientists who examined overfishing on kelp forests, coral reefs, and estuaries using data from paleontological remains going back 125,000 years, archaeological records going back 10,000 years, historical documents going back to the 1500s, and oceanographic observations approximately 100 years old. Published in 2001, their paper showed that overfishing was neither recent nor rare but had occurred since prehistory and had steadily increased in amount, extent, and severity until the present. Widespread pollution, habitat change, and global warming also affect coastal ecosystems, but overfishing was the earliest and worst offender. Subsequent papers on coral reefs and estuaries also profoundly challenged the existing working definitions of healthy fish stocks and ocean ecosystems. They directly linked historical degradation with human population growth and increasing resource consumption.

Soon media attention brought the terms shifting baselines and overfishing to public attention. In 2003 the late Ransom Myers and Boris Worm of Dalhousie University
SHIFTING BASELINES ARE NOT ALWAYS BAD

As the French fisheries scientist Daniel Pauly explains, a shift in a baseline—the “yardstick” against which we may monitor change in ecosystems and cultural norms—is not always a bad thing.

Shifts in baselines need not be associated with losses. Indeed, forgetting can be a good thing. When people who have suffered under the load of a long, stifling tradition emigrate and thus are enabled to distance themselves, both geographically and emotionally, from the ancestral conflicts that in their home countries confined them within balkanized camps, a positive shifting baseline occurs in the generations that follow.

Positive shifts in baselines also occur after social change. One example is smoking in enclosed public spaces, which was ubiquitous in the 1960s. At the time, change seemed impossible, and the stranglehold that the tobacco industry had on our legislators seemed unbreakable. Then, somehow, anti-tobacco activism, medical science, and common sense coalesced into an unstoppable force—let’s call it the Zeitgeist—which overcame all resistance, first in the United States, then in Europe, including France (France!). Now we look back, and our baseline—and especially that of young people—has so shifted that we do not understand how we ever accepted smoking in tight public places. We have collectively forgotten how it felt (and smelled) and how we could even tolerate it—just as we have collectively forgotten how it was when the majority of people were farmers or, even earlier, hunter-gatherers surrounded by nature that teemed with a diverse animal and plant life.


in Halifax, Canada, showed that overfishing had caused populations of large predatory fish in the oceans to decline by 90 percent from a baseline set about fifty years in the past. This research was featured in late night talk show host David Letterman’s Top Ten List of Dumb Things to Do. Such work and the work of like-minded scientists and historians challenged current definitions of healthy fish stocks, marine ecosystem function, and the mutual influence of human activity and change in the ocean, but not without attracting opposition and generating controversy.

Abundance Baselines

In 2001, Myers and colleagues calculated the carrying capacity—the maximum population size an ecosystem could support—for cod populations in the North Atlantic. Finding the carrying capacity was one way of estimating unfished cod abundance. Four years later, Andrew Rosenberg and colleagues at the University of New Hampshire modeled catch data from nineteenth-century fishery logbooks to estimate the size of the cod population on the Scotian Shelf, an important fishing ground off the coast of Nova Scotia, Canada, in 1852. Their estimate—1,260,000 million metric tons of cod—was statistically equivalent to the carrying capacity Myers and colleagues calculated for the same region. Both studies showed that Scotian Shelf cod population had declined by 90 percent or more.

Agreement in findings from the two different methods added credibility to the concept of the shifting baselines syndrome; however, abundance estimates of unfished whale populations have not agreed. Models using catch data in whaling logs yielded significantly smaller population baselines than estimates derived by analyzing genetic diversity within whale populations.

Historical Baselines

Paleontology, archaeology, history, sociology, ecology, oceanography, geography, and molecular chemistry have all generated historical baselines describing past ocean conditions. Time series of data going back decades, centuries, and millennia link historical baselines to the present. They can reveal cyclical processes, processes that operate at different geographical and chronological scales that are otherwise difficult to detect, including the long-term effects of human activity and climate change.
Work to explain the divergence is ongoing, but this discrepancy serves as a caution that all methods are sensitive to assumptions about biological, ecological, and sociological processes, as well as uncertainties in data.

**Distribution Baselines**

Because contracted range can signal population decline, geographical distribution can indicate species abundance. Maine fisherman and scientist Ted Ames compared the geographic distribution of cod spawning grounds collected by old fishermen with locations of spawning grounds today. He found that almost half of the grounds have been lost since World War II. Marine ecologist Loren McClenachan and colleagues estimated endangered sea turtle populations using the number and distribution of nesting beaches and historical descriptions of nesting female abundance. Turtle numbers appear to have declined 80 percent or more from levels in the 1500s. Archaeologist Ian Smith found that the Maoris had exterminated fur seals from 90 percent of their range in New Zealand even before the arrival of Europeans.

**Average Animal Size**

A drop in animal size over time can also indicate population decline. McClenachan also examined photos of trophy fish taken by one Key West charter boat company. She found that the 2-meter groupers and sharks on the trophy board in the 1950s have been replaced by 34-centimeter snappers today, an 83 percent decline in the length of trophy fish. (See figure 1.) Similarly, while abundance estimates for cod and ling in the North Sea in 1872 derived by Danish historian René Paulsen and fisheries scientist Andrew Cooper showed little decline, smaller average fish size and contracted range suggest that overfishing had occurred.

**Ecosystem Baselines**

As fisheries management moves toward place-based ecosystem approaches, analyzing long-term local variations in the diversity of species, geographical and oceanographic characteristics, and ecological processes becomes increasingly important to establish accurate baselines. Marine ecologist Heike Lotze and others described historical changes in Passamaquoddy Bay in the Gulf of Maine and in the Wadden Sea, stretching along shore from the Netherlands to Denmark, to show how European-style fishing and agriculture led to loss of species, decline in water quality, and changes in the ocean floor. Overfishing at all levels of the food web, from seaweeds and shellfish to swordfish and whales, simplified food webs; siltation, chemical pollution, and lack of oxygen resulting from excessive nutrient runoff altered the physical characteristics of the environment in both places. Both ecosystems became simpler and more vulnerable to collapse as food web interactions between animals broke down and ecosystem components were lost. The rate of decline was slower in Passamaquoddy Bay than it was in the Wadden Sea, and the loss of biodiversity less extreme.

**Figure 1. Trophy Fish Caught on Key West Charter Boats: (1) 1957, (2) early 1980s, and (3) 2007**

Coral reef systems are among the most endangered ecosystems in the world, and human interference has been shown to heighten their vulnerability. The marine ecologist Enric Sala and colleagues found that relatively untouched coral reefs near atolls in the central Pacific replicate the marine complexity described in historical records and found in paleontological evidence. Large sharks and other reef fishes were conspicuous, and corals were widespread and healthy. Because unfished reefs exhibit less coral bleaching (an indication of change in water temperature, usually warming, but also other environmental stress) than heavily fished reefs, biodiversity apparently conveys resistance to climate change.

Climate

Fisheries research begun more than half a century ago showed that fish species could be affected by climate. Time series of anchovy and sardine scales in sediment going back two thousand years showed that the abundance of these fish was affected by Pacific Oscillations, cycles of sea surface temperature change that influence weather patterns. The sardine and anchovy fisheries that span the Pacific have been vulnerable to collapse since industrial expansion in the 1940s. Boom and bust years affected fishermen and coastal communities around the Pacific Rim. Research has shown that high fishing pressure can also cause anchovies and sardines to collapse when their numbers are already depressed by fluctuations in climate.

Recent research on time series of Atlantic salmon catches going back to the 1600s near the White and Barents seas in northwest Russia revealed no evidence of overfishing until the twentieth century. The Russian scientist Dmitry Lajus and historians Julia Lajus and Alexey Kraykovskiy collaborated to discover that it was climate fluctuations that have played the greatest role in catch variability on this sparsely populated Arctic frontier.

Sustainability and Baseline Choice

Baseline choice determines how difficult it will be to achieve sustainability. Often, baselines derived from more recent data will be premised on smaller populations than those derived from historical data. Karin Limburg and John Waldman, marine scientists in the State University of New York system (SUNY), examined government catch statistics since the 1860s to show that catch of diadromous fish, which live in fresh- and saltwater at different stages of their lives, have declined on both sides of the Atlantic more than 90 percent from maximum catch. SUNY scientists Carolyn Hall, Adrian Jordaan, and Mike Frisk examined dam building in Maine since 1634 to investigate its effect on alewives and blueback herring, anadromous fish that spend much of their lives in the sea but spawn in freshwater. Dams control population size by preventing fish from spawning. Hall, Jordaan, and Frisk (2011) found that, by 1860, dams blocked passage to 99 percent of spawning habitats on a sample of Maine watersheds.

This decline is corroborated by historical anecdote. According to Puritan settler William Wood, in 1634, 100,000 river herring (alewives and shad) were taken at one weir on the Charles River in Massachusetts in two tides. By estimating the weight of seasonal catch and dividing by the area fished, it is possible to compare the density of river herring in 1634 (1.28) to the density of today’s catch (0.005). Thus, using descriptions from the early 1600s, it appears that alewife and shad populations have declined by 1,000 percent in the past four hundred years.

Most baselines used in fisheries management date back only to the early 1980s. Sustainability measured from a baseline alewife population in 1980 would be easier to achieve than from a baseline set in 1860, and that goal would be easier to achieve than one set from a baseline based in the 1600s.

The Future of Shifting Baselines

Historical marine ecology has been criticized for being “faith-based science” and for creating “bad science and bad history.” To some fisheries scientists, the uncertainties, irregularities, and modeling limitations of historical data make them unsuitable for serious scientific consideration. Some historians fear that the singular character of historical events will be lost in scientific standardization, and that historians will become “serfs to science” in the quest for historical baselines. These debates are ongoing, but increasing numbers of publications show that the shifting baselines syndrome has become an accepted, although still controversial, paradigm.

Historical baselines disclose how much change has taken place, but they do not offer a roadmap for recovery. One major criticism of the shifting baselines syndrome is that former abundance levels are impossible to restore because ecosystems supporting such abundance have changed. Now protected, gray whales have recovered in parts of their range, and alewives have recovered on the lower part of Maine’s Kennebec River since the removal of dams that blocked spawning grounds. But these partial recoveries do not mean that the California coast and the Kennebec River have reverted to former conditions. Ecosystems may recover complexity despite supporting significantly different species composition than was historically the case.

Historical baselines have been incorporated in the management of marine sanctuaries at Stellwagen Bank...
in New England, the Florida Keys, and the Great Barrier Reef, and for some whales, seals, halibut, and alewives. On land, historical baselines are used in park and refuge design and in reintroducing species to regions they once inhabited. Wolves introduced into Yellowstone National Park helped restore cottonwoods to the Soda Butte Creek and the Lamar River by suppressing browsing animals, but here and elsewhere the reintroduction of predators has come into conflict with the interests of local people. How to go from an abundant past through today’s瓶颈 to an equally abundant future remains an open question that is key to achieving a sustainable future featuring both complex ecosystems and high levels of human well being.

In examining change in the Gulf of California marine ecosystem, the marine ecologist Andrea Sáenz Arroyo acknowledged the difficulty in fusing history and fisheries science. Perhaps the real value of incorporating historical knowledge into fisheries management lies in revealing anecdotes that recreate compelling snapshots of past conditions. Displaying marine ecosystems teeming with fish, shellfish, seals, and whales, and the benefits people derived from them, these snapshots inspire managers and govern-ments to reconsider sustainability in light of present impoverished conditions. Historical perspectives counteract the shifting baselines syndrome by encouraging people to value the oceans and set recovery standards high.

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See also Carrying Capacity; Comanagement; Community Ecology; Dam Removal; Ecological Restoration; Ecosystem Services; Fish Hatcheries; Fisheries Management; Food Webs; Large Marine Ecosystem (LME) Management and Assessment; Marine Protected Areas (MPAs); Natural Capital; Ocean Resource Management; Population Dynamics; Species Reintroduction; Succession

FURTHER READING


Soil is a multifunctional natural resource because it provides innumerable ecosystem services. Among others, it produces food, fodder, and fiber; is a sink of atmospheric carbon; and filters pollutants. Particularly in resource-poor regions of the world, soils are becoming increasingly degraded. Soils must be restored and conserved through judicious management practices to address global climate change, food insecurity, energy supply insecurity, and overall ecosystem sustainability.

Soil is the basic natural resource that sustains life in all terrestrial ecosystems. It is probably the single most fundamental natural resource because it contains the three phases essential to life sustenance: solid (inorganic and organic) components, water, and air. Soil is a complex and dynamic medium that supports biota and fauna. Despite its vital importance to life, soil is often neglected and even touted as “dirt.” Soil is neither “dirt” nor inert material but a precious resource. Soil is not only a medium for plant growth as traditionally perceived but a multifunctional entity because it (1) produces food and fiber to meet the increasing global demands, (2) buffers and filters nonpoint source pollutants that impair water quality, (3) promotes habitat and diversity of wildlife by providing food and cover, (4) sequesters soil organic carbon and acts as a sink for atmospheric greenhouse gases (carbon dioxide, methane, and nitrous oxide), and (5) supplies feedstocks for renewable energy production. As a multifunctional system, soils play a major role in the mitigation of the projected global climate change, food insecurity, energy supply insecurity, and environmental quality degradation. These innumerable ecosystem services provide essential reasons to conserve the soil.

Agents of Soil Degradation

Soil is fragile and prone to rapid degradation with mismanagement. It is not a renewable resource over the human time scale. Continued degradation threatens the multifunctionality of soils. Water and wind erosion is the leading agent of soil degradation. Compaction, alkalinization and salinization, and acidification are also agents of degradation. Tillage erosion in sloping cultivated lands, which gradually moves soil downslope with time, is often unnoticed and ignored but it can significantly reduce soil productivity.

Anthropogenic activities set the stage for water and wind erosion through deforestation, slash-and-burn agriculture, overgrazing, intensive plowing, use of fallow systems with limited vegetative cover, cultivation of steep slopes, biomass burning and removal, and unplanned urbanization. Croplands are more erodible than lands under permanent grass and trees because croplands are often disturbed and left bare or with little residue cover. Cultivation of sloping and marginal lands is a major cause of soil erosion in mountainous regions.

Accelerated Erosion

Slight soil erosion is an important process of soil formation and ecosystem dynamics. Soil erosion becomes a problem when it accelerates. The extent of soil erosion varies among regions and continents. Water and wind erosion rates in the United States have decreased by about 35 percent between 1982 and 2003 (USDA 2010) thanks to land stewardship, soil conservation efforts, and policies such as the Conservation Reserve Program under the Food Security Act of 1985. About one-third of US...
croplands, particularly under intensive tillage and monocropping or crop-fallow systems, are, however, eroding faster than the tolerable rate (Pimentel and Lal 2007). Tolerable soil loss is when the rate of soil loss equals the rate of soil formation. Soil-loss tolerance will vary with climate, topography, vegetation, soil type, and management (Troeh, Hobbs, and Donahue 2004).

Whereas soil erosion may not be an imminent crisis in the United States and other developed countries, it is major problem in impoverished regions of the world with high population pressure, scarcity of agricultural lands, predominance of resource-poor farmers, and lack of effective soil erosion control practices and policies. Most agricultural soils are being eroded at a rate faster than the rate of soil formation. The current soil loss ranges between 13 and 40 metric tons per year worldwide (Pimentel and Kounang 1998), but the rate of soil formation is less than 0.5 millimeters per year or less than 5 metric tons. Erosion such as interrill (removal of a uniform layer of soil by raindrop splash and sheet flow) is not readily noticeable but often causes soil loss greater than the formation rates. For example, 1 millimeter of soil surface loss is equivalent to about 10 metric tons per hectare.

Similarly, wind erosion can be extremely high in arid and semiarid regions of the world. In western Africa, northern China, the Andes and the Pampas in South America, southwestern Australia, and the US Great Plains, wind erosion exceeds water erosion. The Dust Bowl of the 1930s in the United States illustrates the severity of wind erosion when proper soil conservation practices are not in place. Wind erosion has intensified in recent years as agriculture has expanded to marginal lands and droughts have increased. In the west African Sahel, intensively cultivated croplands erode at a rate of 20 to 50 metric tons per year, resulting in a severe decline in crop yields (Bidders, Karlheinz, and Rajot 2000; Sterk 2003). Water erosion causes approximately 60 percent of the total degraded land worldwide, and wind erosion causes 30 percent. Despite increased erosion remediation efforts, knowledge of erosion factors and causes, and technological advances, erosion remains high, a situation that deserves more attention (Uri 2000).

Implications of Accelerated Erosion

Lessons from the past are crucial to understanding the potential implications of accelerated erosion. Accelerated erosion resulting from soil mismanagement led to the downfall of old civilizations in the Middle East (Bennett 1939). Severe erosion has drastic on-site and off-site implications on long-term agricultural production as well as on soil and the quality of the environment. It deteriorates the physical, chemical, and biological properties of soil. It induces soil surface crusting and sealing, degrades soil structure, reduces water infiltration, and increases the soil’s susceptibility to compaction. Water erosion can pollute downstream water bodies with sediment and chemicals (e.g., nutrients, pesticides) and cause hypoxia, or decreased oxygen in a body of water that affects aerobic organisms (e.g., in the Gulf of Mexico). The main on-site consequence of severe erosion is reduction in crop yields through the reduction of top-soil thickness and loss of soil fertility. In shallow soils with low fertility, even small losses of soil can reduce crop production, whereas in deep and fertile soils, the same amount of soil loss can have less adverse effects.

Soil erosion alters the ecosystem sustainability. It affects not only agricultural lands but also the quality of forests, pastures, and rangelands. It influences diversity and habitat of wildlife by reducing vegetative cover. Eroded materials accumulate in alluvial plains and cause flooding of downstream croplands and water reservoirs. Similarly, wind erosion pollutes the air with dust particles, reduces atmospheric radiation and fluxes of energy, and threatens human and animal health. Wind transports fine particles hundreds or even thousands of miles from the source. Continued severe soil erosion results in a downward spiral of reduced crop and biomass yields and degraded soil properties. Contemporary issues such as global climate change, food insecurity, and renewable energy production directly link to soil erosion.

Erosion and Global Climate Change

Soil erosion is expected to increase under the projected global climate change. Abrupt fluctuations in climatic conditions including erratic and intense rainstorms under the new climate can reduce soil resilience and accelerate soil erosion. In the United States, total amounts of precipitation increased in the last century, and 53 percent of rainstorms were intense or extreme (O’Neal et al. 2005). Rainfall intensity is more critical than rainfall amount. A few rainstorms of high intensity cause dramatic losses of soil (Nearing et al. 2005). A 10 to 20 percent increase in precipitation under the new climate may change soil loss and runoff by as much as 300 percent (O’Neal et al. 2005). Likewise, frequent and long droughts, particularly in semiarid regions, will increase soil erosion by wind. Ice melting, flooding, abrupt river flow fluctuations, and storms near rivers and coastal areas can also increase soil erosion.

After the oceans, the soil contains the largest pool of carbon. It contains twice as much carbon as the atmosphere does, and thrice more carbon than all vegetation
Accelerated soil erosion can reduce this pool of carbon by rapidly oxidizing soil organic matter. As a result, erosion contributes to global climate change through the release of greenhouse gases such as carbon dioxide, methane, and nitrous oxide during erosion (Polyakov and Lal 2008). In semiarid regions, increased soil temperature coupled with limited precipitation can rapidly oxidize soil organic matter, further increasing carbon emissions. Soil degradation by water and wind erosion under the new climate change is expected to be greater in arid and semiarid regions than in humid and cool regions because of lower biomass production and soil organic matter concentration.

Erosion and Food Security

Soil erosion is directly linked to food security. Crop yield correlates with erosion and decreases in a curvilinear or exponential function with an increase in the erosion rate. Erosion preferentially removes the most fertile layer of soil containing organic matter and essential nutrients, which are normally concentrated near the surface layers. Loss of soil fertility reduces biomass and grain yields. Food insecurity is becoming particularly apparent in developing countries, where resource-poor or subsistence farmers lack financial resources for the establishment of effective soil conservation practices to reverse soil degradation. Sub-Saharan Africa, the Caribbean (e.g., Haiti), Central Asia, and some countries in Latin America are experiencing increased soil erosion and food insecurity (Kaiser 2004).

Whereas in some countries the introduction of new crop varieties, fertilizers, and other technologies has partially reduced food insecurity, food production has generally either stagnated or declined in the poorest regions of the world while populations grow (Stocking 2003). For example, in sub-Saharan Africa, crop production has been reduced by about 50 percent due to increased soil erosion and loss of soil fertility. Excessive exploitation of soil and mining of nutrients threaten food security (Bekunda, Sanginga, and Woomer 2010). Although in the past subsistence farmers often used shifting cultivation, now they are forced to overexploit the same piece of land, often hilly and erodible, as productive lands become scarce.

Erosion and Biofuel Development

The increasing demands for alternative renewable energy will most likely exert further pressure on soils. In addition to food and fiber production, soils may have to supply feedstocks for biofuel production. The conversion of Conservation Reserve Program lands to corn ethanol production in the United States and the conversion of rainforests to produce soybean biodiesel, palm biodiesel, and sugarcane ethanol in tropical countries can adversely affect terrestrial ecosystems and further accelerate water and wind erosion (Fargione et al. 2008). Slash-and-burn clearing of forest or pasturelands to create new croplands is a common practice in tropical forests. Large-scale production of crops for biofuel and biodiesel production thus could change the whole ecosystem. It will increase soil erosion, stress soil and water resources, reduce soil productivity and fertility, and increase risks of water pollution. It can also exacerbate the projected global climate change by releasing large quantities of carbon to the atmosphere from land clearing and biomass burning. Biofuel and biodiesel production from corn and soybeans has already increased food prices and could further accelerate food insecurity.

Cellulosic ethanol or second-generation biofuel is also receiving increased attention as an alternative to grain-based biofuel. Crop residues, dedicated energy crops (e.g., perennial warm-season grasses), wood, prairie grass, and other biomass materials are candidates for cellulosic biofuel production. Because large quantities of biomass will be needed to meet the goals of renewable energy production, excessive removal of biomass would also accelerate water and wind erosion. Indeed, recent studies have shown an indiscriminate removal of crop residues influences soil erosion, soil properties, soil carbon sequestration potential, and the overall agricultural productivity (Wilhelm et al. 2004; Blanco-Canqui and Lal 2007; Lal 2009). Crop residue removal may increase soil erosion by ten to one hundred times (Pimentel 2010). Crop residue mulch provides a protective layer against water and wind erosion, and its removal induces surface sealing and crusting, reducing water
infiltration and increasing runoff. Residue removal also compacts soil and reduces soil aggregate stability and strength, soil organic carbon pools, water retention capacity, biological diversity, and soil fertility. In other words, indiscriminate crop residue removal for off-farm uses is against the principles of soil and water conservation.

Growing dedicated energy crops such as perennial grasses in marginal and degraded lands as biofuel feedstocks may provide an alternative to crop residue removal (Blanco-Canqui 2010). This strategy may have fewer negative impacts on soil and water conservation than crop residue removal. A proper balance between perennial grass biomass removal (e.g., cutting height and frequency) and retention may control soil erosion and maintain soil properties and soil carbon pools over row crops. Soils will need to be managed differently, though, if dedicated energy crops are grown and harvested at large scales. Some argue that even production of cellulosic ethanol from perennial biomass feedstocks may adversely affect soil and the environment (Pimentel 2010). Biofuels may not be as carbon negative as initially thought and may not significantly reduce fossil fuel consumption or reduce greenhouse gas emissions (Fargione et al. 2008; Tilman, Hill, and Lehman 2006). Therefore, the impacts of the cellulosic biofuel production on soil and water conservation warrant experimental verification and objective analysis.

Soil Restoration and Conservation

Soil restoration and conservation are critical to enhance soil multifunctionality and meet the challenges of ecosystem sustainability. Degraded soils must be restored and properly managed, while productive soils must be conserved. Management comes before conservation for degraded agricultural soils. Soil conservation not only keeps the soil in place but enhances soil resilience and ability to meet increasing needs. Soil resilience is intrinsically related to ecosystem resilience. Soils are inherently resilient and able to recover from degradative forces. Highly degraded soils, however, will need extended periods of time before they recover to a state similar to predegradation levels.

Soils can be both a problem and solution to the projected climate change and food insecurity, depending on how they are managed. Soils affect the projected global climate change by sequestering atmospheric carbon and reducing net greenhouse gas fluxes. They can be a sink rather than a source of atmospheric carbon if wisely managed and conserved. For example, soils with annual biomass input coupled with reduced disturbance can sequester carbon. In contrast, intensively plowed soils with limited vegetative cover accelerate greenhouse gas emissions. Similarly, conserving soil and water and improving soil productivity through judicious management help avoid food insecurity.

A combination of mechanical and biological practices should be used to conserve soil. Subsistence farmers in resource-poor countries who cannot afford expensive mechanical structures for erosion control can more readily access biological conservation practices. Unlike mechanical structures, biological practices not only keep the soil in place but also improve natural fertility and resilience of degraded soils with time.

Conservation Strategies

Conservation tillage, improved cropping systems (continuous cropping systems and crop rotations), cover cropping, green manure, residue mulching, alley cropping, agroforestry, and conservation buffers are some of the best management strategies for soil conservation. No-tillage farming, where the seeds are deposited directly into untilled soil, is one of the most effective soil management and conservation strategies for reducing concerns over the projected global climate change, food insecurity, and environmental quality degradation.

Soil loss by water and wind erosion from no-till fields is much lower than from conventionally or intensively plowed fields because no-tillage farming reduces soil disturbance and provides permanent crop residue cover. The crop residue mulch intercepts and buffers the erosive forces of raindrops and wind. No-till effectiveness for reducing water and wind erosion depends on the amount of residue input. No-till systems with little or no annual residue input may be no better than conventional tillage for conserving soil and water, sequestering carbon, and enhancing soil resilience and productivity.

Adoption of no-till farming has resulted in better soil management in many regions (e.g., the United States, Brazil, and Australia). It reduces soil
erosion and improves near-surface soil structural properties such as aggregate stability and strength. Despite the numerous benefits and ecosystem services that no-till provides, only about 5 percent of cultivated land worldwide and about 37 percent in the United States is under no-till farming (Lal et al. 2004). Soil-specific strategies of no-till management will expand this technology.

No-till works best when combined with continuous cropping systems, crop rotations, and cover crops, conservation buffers, and other conservation practices. If no-till alone does not work, companion practices should be used to enhance its performance. Continuous cropping systems provide permanent canopy or residue cover relative to crop-fallow systems. Similarly, complex crop rotations with legumes, deep-rooted crop species, close-growing crops, and sod- or grass-based rotations enhance no-till performance to improve soil properties and reduce soil erosion. Cover crops are potential companion practices for no-till; they are planted between main growing seasons to provide additional residue input and protect soil from erosion (Blanco-Canqui et al. 2011). They also increase crop yields by fixing large amounts of atmospheric nitrogen (N) and improving soil fertility (Blanco-Canqui, Claassen, Presley forthcoming 2012). Likewise, conservation buffers placed at the bottom of no-till fields reduce runoff, filter sediment and sediment-bound nutrients, prevent gully erosion, absorb nutrients, and improve wildlife habitat and diversity. Grass barriers, filter strips, field borders, grass waterways, riparian buffers, and windbreaks are potential buffers. Combined with mechanical conservation practices (e.g., terraces), these practices further control soil erosion.

Soil management practices that increase soil organic carbon concentration are a key to improving soil properties and reducing soil erodibility. No-till systems generally store more soil organic carbon near the surface layers than do plowed soils. The greater near-surface accumulation of carbon in no-till soils improves soil aggregation and increases macroporosity and water retention and transmission characteristics compared with plowed soils. Soil organic materials bind soil particles into stable aggregates, provide elasticity to the whole soil, and increase the resilience of soils against erosive forces. High-biomass-producing cropping systems such as complex rotations, cover crops, and continuous cropping systems combined with no-till also increase carbon concentration. Accumulation of soil organic carbon renders many ecosystem services, including mitigation of greenhouse gas emissions, improvement in soil properties, reduction of soil erodibility, and filtration and absorption of pollutants in water. Most importantly, soil carbon accumulation addresses concerns over food insecurity because soil organic carbon directly improves soil productivity. Most degraded soils have lower soil organic carbon concentration. Restoring soil carbon is essential to improve soil productivity and resiliency while reducing soil erodibility.

Forward Steps

Conserving soil for present and future generations must be a top priority to alleviate food insecurity, adapt to climate change, and achieve ecosystem sustainability. Protecting soil as the most basic resource is an investment. Food production, soil and environmental quality, and sustainability of all terrestrial ecosystems are at stake. Thus, efforts should focus on developing region-specific conservation strategies to enhance the multifunctionality of soils. Conservation practices that restore soil carbon are potential strategies to improve soil productivity and reduce carbon emissions to the atmosphere.

Soil management and conservation require a multidisciplinary participation among land owners, farmers, policy makers, and the general public at different local, regional, and national levels. Soil erosion is linked to political, social, and economic conditions. Further soil degradation can be reduced only through land stewardship, technological input, and implementation of judicious soil management strategies and conservation policies.

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See also Adaptive Resource Management (ARM); Agricultural Intensification; Agroecology; Best Management Practices (BMP); Buffers; Desertification; Ecosystem Services; Irrigation; Microbial Ecosystem Processes; Mutualism; Nitrogen Saturation; Nutrient and Biogeochemical Cycling; Permaculture

Further Reading


Reintroductions are attempts to return species to parts of their historic ranges where they were extirpated. Whereas reintroduction in the context of biodiversity conservation was historically used for species preservation, a relatively new use of reintroduction is to restore ecosystem function, especially in the face of species extinction and global climate change.

Humans have moved domesticated or captive animals from one place to another for millennia. There is a well-documented history of wildlife releases to establish new food resources, for biological pest control, and for aesthetic reasons. These movements have frequently entailed release of species outside their natural ranges, however.

Movement of native species may involve the release of animals within their natural ranges to restock hunted populations, to solve human-wildlife conflicts, or to supply nonconsumptive industries such as nature-based tourism. Reintroducing species to fulfill a biodiversity preservation or restoration objective is a relatively recent activity that has developed as a consequence of increasing global awareness of the need to conserve biological diversity in the face of species extinctions.

The high-profile reintroductions of a few charismatic vertebrates in the 1970s and 1980s increased awareness of reintroduction as a viable conservation option. The Arabian oryx (Oryx leucoryx) (pictured in the image above, photographed by Janet Tropp of the Phoenix Zoo) was reintroduced in Oman, golden lion tamarins (Leontopithecus rosalia) in Brazil, and Peregrine falcons (Falco peregrinus) in North America. Reintroductions are an attractive option for generating publicity, particularly because handling, transport, and release of animals are media-friendly events and show concrete action being taken by concerned authorities. In addition, changing public attitudes toward captive wildlife have encouraged zoos to expand their activities to wider conservation measures including reintroductions.

The Species Survival Commission of the International Union for Conservation of Nature (IUCN) created the Reintroduction Specialist Group in 1988 to provide guidance for increasing numbers of wildlife restoration projects globally. The Reintroduction Specialist Group held their first strategic planning workshop in 1992, which formulated the IUCN guidelines for reintroductions (IUCN 1998; see IUCN 2007).

Reintroduction biologists often use the terminology related to reintroduction inconsistently, resulting in considerable confusion. The original terminology outlined in the IUCN position statement on the translocation of living organisms defined “translocation” as the movement of living organisms from one area to another (IUCN 1987). That document recognized three types of translocation: (1) introduction, or movement of an organism outside its historically known native range; (2) reintroduction, that is, intentional movement of an organism into a part of its native range from which it has disappeared or become extirpated (locally extinct) in historic times; and (3) restocking, which is the movement of individuals to build up an existing population.

The term “reintroduction biology” refers to research undertaken to improve the outcomes of reintroductions and other translocations carried out for conservation purposes. In the journal Trends in Ecology and Evolution, New Zealand conservation biologist Doug P. Armstrong and zoologist Philip J. Seddon (2008) proposed ten key questions for reintroduction biologists to address, with different questions focusing on the population level, metapopulation level, and ecosystem level.
effects). The first key question is, “How is establishment low reproduction or survival rates at low densities (Allee because of either chance (demographic stochasticity) or Small release groups can fail to establish populations that would enable long-term persistence once the population is established. The dichotomy is useful because reintroduced populations can fail to establish themselves for several reasons. Regardless of the strategy used to establish a population, habitat effects on population persistence are important. Factors that affect postrelease survival and dispersal can include predation, competition, disease, and habitat disturbance. The factors that affect postrelease survival and dispersal can be divided into those affecting the establishment and spread of populations, following the division often made in invasion biology. Distinguishing between establishment and persistence is more appropriate for reintroduction. Persistence is more general, because it applies to populations that are graphically bounded, and that therefore grow through increasing density rather than increasing range. It also refers to populations that have reached carrying capacity. The dichotomy is useful because reintroduced populations can fail to survive the establishment phase in conditions that would enable long-term persistence once the population is established. Small release groups can fail to establish populations because of either chance (demographic stochasticity) or low reproduction or survival rates at low densities (Allee effects). The first key question is, “How is establishment probability affected by size and composition of the release group?” (See figure 1 for all key questions.) Populations might also fail to establish themselves because they disperse at high rates. Their survival or reproduction rates may be low if they suffer from translocation stress or if they fail to acclimate to the release site. High postrelease dispersal and mortality create a disparity between release population size and the effective initial population size. The second key question is, “How are postrelease survival and dispersal affected by pre- and postrelease management?” Questions about the effects of specific management practices naturally follow. Release strategies intended to facilitate acclimatization are often termed soft release. These strategies may not have the desired effect. Although some reintroduction biologists believe that dispersal and/or mortality can be reduced by holding animals at the release site for some period, not all studies have supported this notion. The factors that affect postrelease survival and dispersal lend themselves to experimental investigation. Reintroduction biologists have experimentally manipulated the size of the release group. Such experiments require multiple reintroduction attempts and are unlikely to be feasible with threatened species. A better approach models the relationship based on the available data on survival, reproduction, and dispersal rates for the species and system in question. Comparative analyses of reintroduction success rates for multiple species and systems probably give a misleading indication of the relationship between release group size and establishment success. There is a bias toward success with large release groups because reintroduction biologists generally release low numbers when they perceive reintroductions to have a low probability of success. Reintroduction programs may be poorly resourced, as well. Where postrelease dispersal and mortality are low, however, populations can potentially establish successfully from fewer than ten released individuals. Habitat Effects on Population Persistence Regardless of the strategy used to establish a population, a reintroduction will fail if the habitat at the release site
cannot support the species. Consequently, the first key question about population persistence is, “What habitat conditions are needed for persistence of the reintroduced population?” Assuming the number of organisms released is below carrying capacity, the essential prerequisite for persistence is positive growth. This growth should be the main target of reintroduction programs. The IUCN reintroduction guidelines emphasize that the original cause of decline must be identified and eliminated before a species can be reintroduced to a site. This may sound simple, but assessing the conditions needed for growth is seldom trivial. No data are usually available for the species at the site. Invasion biologists face the same challenge when trying to predict habitats that can be invaded. Biologists can thus apply similar habitat modeling methods to project the fates of invasions and reintroductions. After release of the species, biologists can model data on survival and reproduction to estimate the rate of population growth and to quantify uncertainty in this relationship.

Reintroduction biologists might also use an adaptive management approach. Habitat conditions could be manipulated over time and/or space to determine requirements for population growth. Such adaptive management could be used to develop criteria for future reintroduction sites, as well as protocols for the population under management.

Genetic Effects on Population Persistence

Although habitat conditions will be the main drivers of population growth, the intrinsic nature of the organisms also affects it. The next key question is, “How will genetic makeup affect persistence of the reintroduced population?” A population could fail to grow from the outset if the founder group were highly inbred or of inappropriate provenance—that is, genetically adapted to conditions different from those at the release site. A more likely problem, however, is that if genetic diversity reduces over time, the species may suffer from inbreeding depression and declining immunocompetence. If populations remain small, such effects are probable. Reintroduction biologists may need to continue managing the reintroduction. Such management can potentially prevent local adaptation if the population is supplemented with individuals of non-native genetic provenance, however, or waste resources that could be invested elsewhere. Research in this area may not only predict how management will affect genetic diversity of reintroduced populations; it may also predict effects on population growth and persistence. Reintroduction biologists who make such predictions must estimate the effects of genetic diversity on survival and reproduction of reintroduced populations, then project the impacts using population modeling.

Questions at the Metapopulation Level

Metapopulation questions deal with multiple populations of species. The term *metapopulation* traditionally described networks of semi-isolated populations connected by natural dispersal. Reintroduction biologists also use the term to describe networks of populations that can be connected by translocation. Any translocation involves at least a simple metapopulation consisting of the source population and recipient population.

Impact on Source Populations

Although reintroduction biology has traditionally focused on the fates of the reintroduced populations, the potential benefits of establishing these populations need to be balanced against the impact to source populations, regardless of whether they are captive or wild. The first key question at the metapopulation level is, “How heavily should source populations be harvested?” Population modeling is one way to address this question, but accurate projections require a good understanding of populations’ regulatory mechanisms—that is, the compensatory increases in survival and/or reproduction following density reduction. Harvesting provides density manipulations that are invaluable for understanding these mechanisms, so lends itself to adaptive management.

Allocation of Translocated Individuals

Moving beyond a single source and recipient population, species recovery programs often involve multiple reintroductions and many potential reintroduction sites. What is the optimal allocation of translocated individuals among sites? Meetings of stakeholder groups involved in a species’ recovery often decide such allocations on an ad hoc basis. Reintroduction biologists could potentially plan them using theoretically derived optimization strategies. Similar methods could also decide the optimal allocation of management effort among sites.

Translocation to Compensate for Isolation

The final question at the metapopulation level is, “Should translocation be used to compensate for isolation in fragmented landscapes?” Reintroduction biologists almost always consider this factor, at least implicitly, because reintroduction is unnecessary if the species is likely to recolonize the site naturally. What they rarely consider is that some local extinctions could be primarily a consequence of metapopulation dynamics following habitat fragmentation—that is, the emergence of discontinuities in a species’ preferred environment. This effect means
that translocation could be used to restore distributions by connecting populations without management of habitat. If the local extinctions were owing to local declines in habitat quality, however, this strategy would be disastrous. Reintroduction biologists require methods for resolving the roles of habitat quality and metapopulation dynamics in species declines, an endeavor that is by no means trivial.

Key Questions at the Ecosystem Level

Although the goal of reintroduction has traditionally been species recovery, reintroductions increasingly occur within the context of ecosystem restoration programs. Despite this, there has been surprisingly little overlap between the disciplines of reintroduction biology and restoration ecology. The reintroduction literature has taken a single-species perspective and focused on animals, whereas the restoration literature has focused on abiotic factors—that is, nonliving factors that affect biotic, or living species—and vegetation. Armstrong and Seddon (2008) suggest that there are three key questions for reintroduction biology at the ecosystem level, and all of these bridge the two disciplines.

Target Species and Parasites

The first key question at the ecosystem level is, “Are the target species and its parasites native to the ecosystem?” The IUCN reintroduction guidelines stress that the organisms used for reintroduction should be as close as possible genetically to those originally found in the area, and that introducing a species outside its historic range should be considered only if there is no suitable habitat available within that range. Assessing the historic ranges and genetic provenances of species proposed for translocation is, therefore, a fundamental part of reintroduction biology. The parasites carried by a species are considered from a veterinary perspective, however, usually with no consideration as to whether those parasites already occur at the release site or occurred there historically. The key focuses of disease-screening procedures should be to restore host-parasite relationships and prevent introduction of non-native parasites. This process needs to be accompanied by research designed to reconstruct historic distributions of parasites.

These issues are likely to become increasingly complex in the future. Global climate change is shifting the distributions of suitable habitat for many species. At least some programs will have to focus less on restoring what was originally found at a site and instead focus on facilitating development of ecosystems suitable for new climatic regimes. According to Seddon (2010), the movement of living organisms from one area to another for conservation purposes can be viewed along a spectrum, characterized by a decreasing reliance on documented historic distribution and ranging in scope from population restoration to conservation introduction. (See table 1.)

Ecosystem Effects of Releases

The next key question is, “How will the target species affect the ecosystem and its parasites?” This question is closely related to the previous question of whether the parasites are native to the ecosystem. The primary goal of translocations should be to restore ecosystem function rather than species composition. Although the IUCN reintroduction guidelines provide for introducing species to new areas to satisfy species recovery goals, a better justification might be to restore the functional roles of extinct species. Scientists need research designed to project impacts of translocated species to justify such introductions, to determine which parasites it is most important to screen for, and to prioritize reintroduction of ecosystem engineers—that is, species that create or modify habitats—and other species crucial to ecosystem function. Studies documenting previous ecosystem-level impacts of reintroductions are relevant here, as are methods invasion biologists developed for projecting impacts of new species on ecosystems.

Order of Reintroductions

The final key question is, “How does the order of reintroductions affect the ultimate species composition?” This question comes up frequently in the course of restoration programs, but reintroduction biologists often make decisions based on intuition. Because the question often concerns species at different trophic levels—that is, the level a species occupies in the food chain—one promising area for research is the functional responses between predator and prey species that are likely to be reintroduced. These responses could determine the ability of the predator and prey to coexist in relation to their initial densities.

Outlook

Reintroduction biology will always strongly emphasize case studies because there is no substitute for local knowledge of species and systems. Although future research might give a greater role to meta-analyses, useful meta-analyses depend on good data from individual case studies. Comparative analyses of simple statistics—such as success rates—will produce misleading or trivial results in the absence of such data.
Table 1. The Conservation Translocation Spectrum

<table>
<thead>
<tr>
<th>Reliance on Documented Historic Distribution</th>
<th>Primary Focus</th>
<th>Term</th>
<th>Definition</th>
<th>Synonyms</th>
<th>Scope</th>
</tr>
</thead>
<tbody>
<tr>
<td>High</td>
<td>Single species</td>
<td>Reintroduction</td>
<td>Intentional movement of an organism into part of its native range from which it has disappeared or become extirpated in historic times</td>
<td></td>
<td>Population restoration (release outside known range)</td>
</tr>
<tr>
<td>Medium</td>
<td></td>
<td>Restocking</td>
<td>Movement of individuals to build up an existing population</td>
<td>Supplementation, augmentation, reinforcement, enhancement (plants only)</td>
<td></td>
</tr>
<tr>
<td>Low</td>
<td>Ecosystem</td>
<td>Ecological replacement</td>
<td>Introduction of the most suitable extant form to fill the ecological niche left vacant by the extinction of a species</td>
<td>Subspecific substitution; taxon substitution; ecological substitutes/proxies/surrogates</td>
<td>Benign/conservation introduction (release outside known range)</td>
</tr>
<tr>
<td></td>
<td>Assisted colonization</td>
<td>Translocation of species beyond their natural range to protect them from human-induced threats</td>
<td>Assisted migration, managed relocation</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Community construction</td>
<td>Introduction of suites of species to create new species assemblages</td>
<td>Futuristic restoration; designer/novel/invented ecosystems</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Source: Seddon (2010).

Following his proposal of a conservation translocation spectrum, Seddon (2010) outlined three key implications for the future of species reintroduction. (See table 1.) Historic distribution records will always provide a good starting point for identifying translocation release sites. Global climate change and the dynamic nature of ecosystems mean, however, that historic species ranges have only limited use. Reintroduction biologists will need to use even prehistoric reference points and should consider species-specific habitat suitability assessments.

Single-species conservation actions in the core of historic range will remain the backbone of many conservation efforts, but increasingly we need to adopt an ecosystem focus and consider the translocation of suites of species to restore key ecological functions. Ecological functions once performed by now-extinct taxa can be restored through the introduction of ecological replacements, which may themselves be threatened in their native range.

Reintroduction biologists and restoration ecologists should join forces in selected projects to create novel...
ecosystems, including, where appropriate, ecological community construction through conservation introductions, to serve both single-species conservation and ecosystem management objectives.

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See also Biodiversity; Carrying Capacity; Charismatic Megafauna; Community Ecology; Ecological Restoration; Forest Management; Global Climate Change; Hunting; Indicator Species; Keystone Species; Plant-Animal Interactions; Population Dynamics; Reforestation; Refugia; Regime Shifts; Wilderness Areas

FURTHER READING


Techniques using green infrastructure are emerging as a viable alternative to traditional gray water infrastructure approaches for managing stormwater. Rather than attempting to manage stormwater by conveying it elsewhere, green infrastructure techniques more closely mimic how water is managed in the natural world, capturing and infiltrating as much water as possible on site through rain gardens, bioswales, native landscaping, porous pavement, constructed wetlands, and other techniques.

Stormwater runoff is a substantial water quality problem, especially in urban areas where rainwater running down streets and across parking lots picks up pollutants before flowing into streams, rivers, and lakes, and occasionally overburdening municipal sewage treatment systems. Historically, traditional (or “gray”) stormwater management approaches were developed to manage the impacts of urbanization, including localized flooding. Large-scale solutions were engineered using storm sewers, deep tunnels, stormwater retention ponds, and other “gray infrastructure” technologies.

The results of gray infrastructure approaches to stormwater management have been mixed. While they have helped mitigate the impacts of rainfall events and flooding, traditional approaches have come with considerable economic, social, and environmental costs. For example, many urban streams and rivers were buried or diverted into channelized concrete structures in the name of stormwater management. As a result, plant and animal biodiversity has suffered, while citizens have lost access to recreational landscapes. Meanwhile, communities have had to spend large amounts of money building and maintaining extensive gray water infrastructure. In Milwaukee, Wisconsin, for example, the estimated cost for managing a gallon of stormwater using traditional approaches is estimated to be $2.42 per gallon, significantly higher than all green infrastructure approaches except for green roofs and rain gardens (Milwaukee Metropolitan Sewerage District 2009).

Unlike this traditional gray infrastructure, green infrastructure captures and infiltrates water where it falls. Green infrastructure is being adopted by communities around the world as a cost-effective means to provide more environmentally sensitive solutions to stormwater management. For example, in communities with combined storm and sanitary sewers, green infrastructure solutions can help minimize the number of overflow events where untreated water is discharged into rivers and lakes. Green infrastructure provides social benefits to communities as well, for example, by improving aesthetics, providing habitat, and helping minimize the “urban heat island effect” that is caused in part by the buildup of hard infrastructure in cities, which captures solar energy and radiates heat. Some of the more common green infrastructure practices, including the use of green roofs, the planting of trees, bioretention and infiltration mechanisms, the use of permeable pavement, and water harvesting, are described below.

Green Infrastructure Approaches

Green roofs are rooftops covered in part or wholly by vegetation. (A waterproof barrier is used to protect the roof structure from damage such as water, soil, and/or root penetration.) By absorbing rainfall, green roofs help reduce stormwater runoff. Rooftop vegetation also improves air quality and provides habitat for plants and wildlife, including butterflies and birds. Finally, green roofs offer people opportunities for environmental education and recreation,
while providing aesthetic benefits. They do add considerable weight to building rooftops, however, and therefore their use is limited to those buildings that are structurally engineered to support the additional weight. The green roof approach can be quite expensive, with capital costs ranging between eight and twenty-five dollars per square foot (Milwaukee Metropolitan Sewerage District 2009).

Trees reduce stormwater runoff by intercepting rainfall. They also take up soil moisture, which allows the soil to absorb more water during rainfall events, which in turn helps minimize runoff. Trees also improve air quality, reduce energy consumption, provide habitat, and enhance quality of life. While trees provide multiple benefits, older mature trees function best as green infrastructure; therefore, trees are not a quick fix for addressing immediate stormwater issues (Milwaukee Metropolitan Sewerage District 2009).

Bioretention and infiltration practices help retain water from rainfall and allow it to infiltrate the ground gradually. These include the use of rain gardens, bioswales, and wetlands. Rain gardens help infiltrate stormwater runoff and often include native plantings, which also evaporate water (remove it from soil by both evaporation and transpiration from the leaves of plants). Bioswales are shallow, usually linear depressions often located along roadways and parking lots that collect and infiltrate runoff. Wetlands serve a similar function and act like sponges to absorb runoff during rain events and slowly release stored water over time. Bioretention and infiltration practices not only reduce runoff; they also improve air quality and mitigate against climate change by sequestering carbon dioxide (i.e., capturing it rather than releasing it into the Earth’s atmosphere). In addition, they improve aesthetics, create plant and animal habitat, and provide educational opportunities. Successful bioretention and infiltration mechanisms require good management practices to maintain the desired species mix, avoid the introduction of invasive species, and minimize litter accumulation.

Large swaths of the Earth’s surface, particularly in urban areas, are now covered with pavement, which can dramatically increase the amount of polluted runoff in these areas. Permeable pavement provides the benefits of pavement while reducing the negative impacts of associated runoff. Permeable, or porous, pavement contains small spaces that allow water to percolate through. Once below the pavement surface, the water is either absorbed into the ground, stored in a catchment system, or conveyed off-site. Chicago has received attention for its use of permeable pavement in its alleyways, allowing water to percolate on-site, thereby reducing stormwater runoff flows into the sewer system and eventually into Lake Michigan, which is Chicago’s source of drinking water (Solsby 2010).

Water harvesting techniques redirect rainwater away from stormwater sewers and into vessels, including rain barrels and cisterns, so that the stored water can then be used on-site for irrigation and residential or commercial gray water needs, including toilet flushing. Water harvesting techniques can be deployed adjacent to virtually any existing building or structure with a roof system. Downspouts can be disconnected and the water redirected into storage systems, ranging from small 50-gallon (190-liter) rain barrels to large underground cisterns capable of holding thousands of gallons. Water harvesting techniques are especially suitable for arid regions where water is often in short supply and can be quite costly. By reducing the demand for irrigation water, which is often treated, harvesting systems can help households, businesses, and governments save both water and energy.

Implementation of Green Infrastructure

Communities wishing to promote green infrastructure practices can do so in a number of different ways. They can lead by example by installing rain gardens, bioswales, and green roofs on government-owned property. Communities wanting to encourage green infrastructure practices more broadly, however, have a wide range of policy options
available to them. Several North American communities—including Toronto, Chicago, and Portland (Oregon)—have encouraged the private development of green infrastructure through incentives, subsidies, consultation services, fee reductions, and regulations (Bitting and Kloss 2008).

Stormwater management is an area of public policy and private development that can take advantage of green infrastructure approaches to allow households, businesses, and communities to manage their water resources more effectively while promoting sustainable environmental health, improving quality of life, and saving money. Green infrastructure also mitigates against urban heat island effects, offsets irrigation and water demand, improves air quality, and enhances the overall quality of life in communities.

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See also Buffers; Groundwater Management; Hydrology; Landscape Architecture; Permaculture; Rain Gardens; Road Ecology; Tree Planting; Urban Agriculture; Urban Forestry; Urban Vegetation; Waste Management; Water Resource Management, Integrated (IWRM)

FURTHER READING


Succession is temporal change in ecosystem structure that can be initiated either naturally or by humans. Ecologists and ecosystem managers use different models to understand and predict this change in order to promote sustainability. The traditional linear models for succession have been challenged by newer nonequilibrium models that emphasize abiotic controls and multiple endpoints.

Ecological succession refers to the change in ecosystems and their constituent plant communities that occurs as the organisms in the ecosystem respond to and modify that ecosystem over time. Succession models are theoretical frameworks for describing and interpreting the development of plant communities and ecosystems, and they have had profound impact on the way that people interpret and manage the land. Assumptions derived from ideas about how ecosystems develop and change also influence how people think about sustainability. The classic succession model posits that soils, plants, and the animals associated with them go through various stages of development until they reach a stable equilibrium with their physical environment. This final stage is termed a climax state, achievable in the absence of disturbance that sets the community or ecosystem back. For many, ecological sustainability has come to mean achieving and maintaining some sort of equilibrial state. More recent theories of ecosystem change, however, cast doubt on the viability of an equilibrial state as a goal or measure of the sustainability of a system. Newer models, based on nonequilibrium theories, offer a more broadly applicable framework for understanding and evaluating the sustainability of managed ecosystems.

Linear Succession Model

The classic model of succession views ecosystem development as a linear process, driven by competition among living organisms. The thinking behind this model essentially began in the early twentieth century and was elaborated in the 1920s by the ecologist Frederick Clements while studying plowed fields in the Midwest. He fully developed the concept of linear, deterministic succession to describe vegetation response to human disturbance. This concept gained such traction that it is now often referred to as the Clementian succession model. In the decades following his work, ecologists have debated the particulars of the model, and whether succession follows from the organic development of ecosystems, the individual characteristics of plants, the facilitation or suppression of new plants by existing plants, or the initial floristic composition on the site.

The standard illustration of primary succession begins with a volcanic eruption and fresh lava that eventually weathers. Plant seeds brought in by wind or animals take root. Soils form, and a predictable succession of plant communities or ecosystem states—from grass and shrubs to forest—ensues. Canopy closure shades out the herbaceous and shrubby pioneer plants, and the forest matures and eventually reaches a stable state of equilibrium when it becomes "old growth." Secondary succession takes place after less traumatic events or disturbances that do not require a new soil to form, such as when a forest is burned or cut down and then recovers, progressing from grass and shrubs back to trees.

The environmental characteristics and species interactions at each site influence the community at the endpoint, whether forest or desert scrub. The collective stages of plant community succession at a site, referred to
as *successional series*, may provide a series of habitats for distinct wildlife species. For example, the spotted owl of California depends on an old-growth forest habitat and is often described as a climax species. Various attempts have been made to correlate maximum biodiversity and other features with climax stages. Some observers and ecosystem managers still evaluate the condition of an ecosystem or plant community based on how close its current state is to the predicted climax, although many ecologists consider this to be a misleading and inaccurate method of assessment.

Succession models based on succession toward an equilibrium state imply that any force that drives an ecosystem away from climax is detrimental. This idea has influenced the way ecologists and ecosystem managers have interpreted landscapes and assessed sustainability. For example, in much of the world it has provided a rationale for suppressing indigenous and traditional patterns of natural-resource management and use. The effect of humans on the ecosystem, seen as a form of disturbance, has often been assumed to be detrimental to the ecosystem’s condition because it moves the ecosystem further from a posited climax state. Ecosystem managers and conservationists have treated natural and human-made fire as a detrimental disturbance to ecosystems for decades, and only recently have begun to fully accept managed fire as a shaper of sustainable ecosystems. The anthropologists James Fairhead and Melissa Leach argued in a 1995 article that environmental historians misinterpreted a landscape in Africa because they assumed that human actions were inherently degrading to ecosystems and would cause a loss rather than a gain of forest cover. The use of fire by indigenous people was eventually found, in fact, to contribute to forest renewal and human sustenance. Underlying this misinterpretation is the classic succession model now largely rejected by ecologists but still widely influential among managers.

**Alternative Succession Models**

An alternative to the classic view, nonequilibrium ecology holds that ecosystem characteristics are more often influenced by disturbance and abiotic (nonliving) factors than by the biotic (organic) interactions that are used to explain the linear pattern of succession. When disturbance is frequent, severe, and unpredictable, stochastic models may be a better fit to the observed ecological changes. Stochastic models consider the outcomes of succession to be unpredictable, though attempts at prediction may be based on probabilities derived from historical outcomes. Most ecosystem dynamics, however, fall somewhere between an undisturbed progression from one set of species to another, driven by competition and other biotic factors, and a completely unpredictable system that responds ad hoc to unanticipated disturbance. Instead, a model that recognizes the persistence of relatively stable ecosystem configurations and acknowledges that there are multiple possible states and pathways among them has proven useful for understanding ecological dynamics. Such models are termed state and transition models. Table 1 on page 382 compares the three approaches.

For those interested in sustainable management of ecosystems, the simple linear deterministic model of succession is easy to apply when evaluating the potential sustainability of managed systems, but it has low predictive power in many systems. Lands used for agriculture, grazing, timber, or many other types of ecosystem service production are inherently managed for vegetation conditions that would be considered nonclimax, or distanced from the supposed equilibrial state. State and transition models, on the other hand, suggest that management is better understood as a process of choosing among possible stable states and maintaining those states rather than seeking a single equilibrial climax. They accommodate the role of stochastic processes in ecosystem change, as well as the driver of classic succession, plant competition. State and transition models can also evaluate the role of human interaction in ecological change, unlike the linear succession model.

A state and transition model is research intensive. It allows the manager to identify ecosystem states that are stable within management horizons, and to gather data that identifies states and explains shifts from one state to another. Management practices that change or maintain sites can be identified, tested, and incorporated into the model, which makes it amenable to adaptive management. In a forest, for example, fire frequency and intensity, natural or human caused, may maintain the stability of a state or cause a transition among the possible stable states on a site, affecting the characteristics of the forest indefinitely. A linear succession model predicts a single path after fire from “weedy species” like grasses to a climax state of mature trees, while a state and transition model accommodates different stable endpoints or outcomes depending on management practices or natural events. States and transitions are defined by data collected within a site of well-defined environmental characteristics, including soils, climate, slope, and other factors. As more is learned about the ecosystem, states and the transitions among them may be better defined and understood. State and transition models incorporate new information and can be corrected and elaborated as more data is provided. Because they do not rely on a singular pattern of development to explain ecosystem
Table 1. Comparison of Three Types of Models for Explaining Species Composition and Ecological Change

<table>
<thead>
<tr>
<th>Model</th>
<th>System Characteristics</th>
<th>Site Conditions Under Which Model Might Work</th>
</tr>
</thead>
<tbody>
<tr>
<td>Linear deterministic succession</td>
<td>• Long- and short-term prediction</td>
<td>• Abiotic conditions are stable</td>
</tr>
<tr>
<td>(Clementian succession)</td>
<td>• Initial conditions are known but have minor influence on predicting succession</td>
<td>• Disturbance is rare and extrinsic to the community</td>
</tr>
<tr>
<td></td>
<td>• Competition and other plant-animal interactions are drivers within given site conditions</td>
<td>• Low level of environmental heterogeneity</td>
</tr>
<tr>
<td></td>
<td>• A single state commonly develops in the absence of disturbance</td>
<td></td>
</tr>
<tr>
<td>Stochastic</td>
<td>• Short-term predictions are more accurate</td>
<td>• Frequent and relatively intense unpredictable disturbance</td>
</tr>
<tr>
<td></td>
<td>• Initial conditions are often unknown and important predictors</td>
<td>• Small spatial and temporal scales</td>
</tr>
<tr>
<td></td>
<td>• Random events are an important influence on succession</td>
<td></td>
</tr>
<tr>
<td></td>
<td>• Order of propagule arrival may have strong influence</td>
<td></td>
</tr>
<tr>
<td>Alternative stable states (states and transitions)</td>
<td>• Initial conditions known or unknown</td>
<td>• Adaptable to any spatial and temporal scale or pattern</td>
</tr>
<tr>
<td></td>
<td>• Environmental conditions may be important drivers of succession</td>
<td></td>
</tr>
<tr>
<td></td>
<td>• Random events may be important</td>
<td></td>
</tr>
<tr>
<td></td>
<td>• The contributions of resilience and thresholds to state stability are important</td>
<td></td>
</tr>
<tr>
<td></td>
<td>• Disturbance may be important factor, intrinsic or extrinsic to the ecosystem</td>
<td></td>
</tr>
<tr>
<td></td>
<td>• Abiotic factors may or may not mediate competition</td>
<td></td>
</tr>
</tbody>
</table>

Source: authors.

change, they can define research needs for understanding the dynamics of an ecosystem. The major disadvantage to state and transition models is the up-front need for more information about states and transitions, but because they are based on data rather than solely on theory, their predictive capacity is high compared to Clementian models in ecosystems where abiotic factors are important drivers of vegetation change. Specific methods for better incorporating reliable information into management practices are now well developed and known as adaptive management.

Ecosystem Stability

While the state and transition model of ecological succession focuses on multiple possible outcomes, ecosystem managers remain concerned with establishing stability, and certain theoretical frameworks are important to finding the keys to sustainability. Understanding the factors that predict the stability of an ecosystem state can result in management practices that will sustain that state within the bounds of likely climate, social, and economic change. The concept of resilience is used to describe an ecosystem state with strong feedbacks or responses that help maintain the state. For example, openings in a grassland may stimulate more seed production in nearby grasses because increased nutrients are available, so the openings will eventually close again. Feedbacks can also be destabilizing, such as when an invasive grass increases the likelihood of fire in sagebrush steppe, leading to more frequent fire, even more grass, and a state change, or transition, to grassland. Ecosystem managers aim to identify and support feedbacks that promote stability to enhance ecosystem sustainability.

The concept of thresholds is also important in understanding what contributes to stability in the state and transition model. Changing from one ecosystem state to another may involve crossing a threshold that has directionality—it is easier to go in one direction than another. For example, a state and transition model constructed for Australian eucalypt forest predicted that
when overharvest completely removes trees, a subsequent influx of saline groundwater shifts the site from a eucalyptus-dominated lowland to a salty flat that can support only salt-tolerant plants. Returning to the eucalypt-dominated state, if feasible, would require major and costly interventions. In other words, the overharvest causes a transition that crosses a threshold. Understanding transitions, resilience, and thresholds can be invaluable when assessing opportunities for restoration and their likelihood of success.

A simplified “ball and cup” diagram can be used to illustrate the ideas of resilience, thresholds, transitions, and stable states. (See figure 1.) The bowl of the cup is a stable state, and in order to shift to another state, the ball must transition over a threshold, represented as the rim of the cup. The ball can be thought of as moving within the cup due to natural environmental variations and even succession. In the diagram, which illustrates changes in a California oak woodland, oaks may be removed by fire or harvest, a transition with a relatively low threshold. If the oaks are completely removed and there is no resprouting, however, it will take planting and perhaps protection of regrowth to restore the oak woodland, a difficult transition that is unlikely to occur naturally. (See figure 1 below.) It must be noted that these models need validation by empirical tests, and they fit specific sites with particular environmental configurations.

**Outlook and Challenges**

There are three areas of major challenges and debates concerning new theories of succession or ecosystem change.

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**Figure 1. Resilience, Thresholds, Transitions, and Stable States**

**T1: Transition 1**  Oaks harvested

**T2: Transition 2**  Oaks restored by planting

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*Source:* authors.

This “ball and cup” model illustrates the role of thresholds and resilience in transitions among states. Resilience operates like gravity to keep the ball in the cup. Thresholds and resilience support the stability of a state, keeping it within recognizable bounds. The ball represents the ecosystem status, and the depth of the cup illustrates the state’s resilience to disturbance.
First, many managers prefer the linear succession model because it can greatly simplify management planning and monitoring and provides an implicit goal. The models work fairly well in many ecosystems, such as temperate hardwood forests. Even though their use has become a professional norm, however, linear succession models are open to criticism as premised on a way of looking at the world that leads to normative judgments and the use of pseudoscientific terms like “degradation” that imply a linear, reversible pattern to ecosystem change; alternative nonequilibrium models take into consideration multiple variables and outcomes and focus on processes rather than endpoints. States and transition models may include multiple possible stable states, and the manager must choose among them as a management, or sustainability, goal. Second, the move to broader use of state and transition models is limited by the scarcity of available information and the need for intensive research. While organizations like the US Department of the Interior’s Natural Resources Conservation Service are linking state and transition models to soil surveys and ecological sites throughout the United States, there is still a paucity of data from which to build and rigorously test these models. That said, the few tests of state and transition models have shown the predictive value of the models when supported by site- and time-specific data. Third, adequate monitoring of ecosystem response to environment and management may be too costly without the development of new and efficient methodologies.

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See also Biodiversity; Biogeography; Community Ecology; Complexity Theory; Dam Removal; Disturbance; Ecological Forecasting; Fire Management; Food Webs; Forest Management; Plant-Animal Interactions; Population Dynamics; Regime Shifts; Resilience; Rewilding

FURTHER READING


Tree Planting

Trees and their ecosystem services—for example, oxygen (through photosynthesis), clean water, soil protection—are essential for all animal life. Human activity and natural disasters reduce the forest cover on the planet. It is vital that people replace lost trees, and many organizations around the world are attempting to do this.

Trees are an invaluable natural resource. In the landscape, they retain water, provide shade, enrich soil, calm winds, and provide wildlife habitat. In urban areas they increase property values, provide noise and odor barriers, and moderate microclimates around homes and businesses.

Trees serve a variety of purposes in different settings. Farmers and landowners may want trees that can produce fruit crops or high-value timber. People in suburban or urban areas may choose ornamental trees. Homeowners may be interested in trees that will shade their homes, reducing energy needs for both heating and cooling. Farmers in plains states and other open landscapes may want shelterbelts of trees to protect their buildings, livestock, and crops. They may also choose certain tree species, notably nitrogen-fixing trees, for their foliage that makes good livestock fodder, especially for smaller animals like goats.

Humans deplete the world’s stock of trees. In addition, weather conditions and global warming contribute to the loss of this valuable resource. Around the world, more severe storms are occurring throughout the year, and as local climates change, trees that once thrived are now threatened. Some storms, like ice storms, hurricanes, and tornadoes, do physical damage to trees, and some create flood conditions that can drown existing trees. Replanting lost trees or introducing new trees to damaged lands maintains the many services trees provide for people.

Planting Conditions

Whenever trees are being introduced or reintroduced into a landscape, there are many factors to consider, including species, soil type, and availability of water. Native species or species that grow in similar climates and at similar latitudes in other countries are likely to flourish, although some trees can become invasive when planted in a new environment. An analysis of soil types will match native tree species to appropriate sites. Since trees can be very vulnerable while they become established, availability of water is critical, either naturally through precipitation or artificially through an irrigation system. In plains or other open landscapes, strong winds will make it more difficult for new plantings to establish themselves successfully. Local topography can also influence the success of tree planting. Low-lying lands and valleys may be frost pockets that are unusually cold in the winter, subjecting vulnerable newly planted trees to the possibility of freezing. Aspect—the compass direction in which a slope faces—can also influence the success or failure of tree planting. In the Northern Hemisphere, southern- and western-facing slopes tend to be hotter and drier; northern- and eastern-facing slopes tend to be cooler and moister. Different types of trees prefer these different microclimates, and it is important to accommodate those preferences. In the Southern Hemisphere, of course, these aspect characteristics are reversed.

Planting

The purposes for the trees may dictate whether saplings, seedlings, or cuttings will be used. Saplings are trees that are a few to several years old (often, three to eight years), and they are suitable for planting in many sites—rural,
urban, desert, and wetlands. Seedlings are usually one to three years old; the seedlings of conifers are often older than the seedlings of broadleaved trees due to normal differences in the rapidity of early growth. Seedlings are the least expensive and are often used for reforestation projects, where large numbers of them are required, for that reason. Cuttings are small saplings (about 3 or 4 cm in diameter at the root collar and often 2 to 3 m in height) from which both roots and tops are removed in the early growing season. The cuttings are then immediately planted along river banks or in other moist sites to stabilize soil. Species such as true poplars (Populus spp.), willows (Salix spp.), and other species commonly found along waterways are particularly useful for this purpose. When planted in the spring, these species can readily produce new roots and shoots and establish themselves as small trees within one growing season.

Newly planted trees benefit from a blanket of mulch around them to conserve moisture and eventually provide nutritious organic matter. In very rocky and arid locations, trees can be mulched with rocks or stones rather than with organic matter. Piled loosely near the base of the tree (not touching the bark), rock mulch can maximize the amount of water the trees receive. The rocks slow the movement of the water, allowing it to filter slowly into the soil. The physical barrier of the rocks can also minimize the impact of direct sunlight on the soil around the tree base and thereby minimize the loss of moisture through evaporation.

In desert locations where there is little water, trees may need to be spaced at least twice as far apart as is recommended for planting trees in more temperate or Mediterranean climates. Wider spacing gives the tree roots a wider area in which to access limited water resources. Where waste oil is readily available, concave or funnel-shaped planting sites for the trees can be “painted” with oil to direct water to the base and roots of the trees, preventing it from running off or filtering away quickly in the sandy soil.

**Agroforestry**

Agroforestry combines tree planting with crops and/or livestock. This science is practiced widely in tropical regions and is attracting interest in more temperate climates. All agroforestry techniques are designed to improve land use and make food or fiber production more sustainable. The major techniques are alley cropping, riparian buffer strips, windbreaks, shelterbelts, silvopasture, and forest farming. Alley cropping, buffer strips, windbreaks, and shelterbelts involve planting trees singly or in rows in fields of annual crops or along rivers and streams; silvopasture adds livestock to the forage and tree crops. Forest farming manages existing forests for crops that can be harvested either annually (for example, bee products or medicinal plants) or in short-term rotations of a few years rather than the decades of growth needed for timber production.

**Reforestation**

In the 1920s, a British forestry officer serving in the colonial service in Kenya, Richard St. Barbe Baker, was employed to select timber for harvest. Although his employers were interested in maximizing the amount of timber cut, St. Barbe Baker carefully assessed the probable sustained yield of the forests and refused to authorize timbering licenses for unsustainable cuts. When he reached the northern highlands of Kenya, St. Barbe Baker found mostly scrubland rather than forests. The local Kikuyu tribes were cutting and burning the forests to provide cropland, an unsustainable practice. St. Barbe Baker held a council with the tribal elders and encouraged them to replant trees to reclaim the denuded land. He selected fifty volunteers to become “Men of the Trees.” These volunteers took a solemn oath to plant and nurture trees to save their land from desert encroachment. From this beginning in Kenya, the organization spread to other parts of the British Commonwealth, especially to Australia, where it is now thriving. The mission of the Men of the Trees is “to bring people together to plant and grow trees, and to achieve healthy, productive, sustainable landscapes.”

Interestingly, although it was the warriors who swore to plant and protect trees, it was the women of the Kenyan tribe who actually did the work (especially establishing tree nurseries for the planting stock). The Nobel Peace Prize winner Wangari Maathai (1940–2011) went on to develop the Green Belt Movement in Kenya. Professor Maathai’s purpose was to reduce poverty and enhance environmental conservation by planting trees. The organization she founded has been responsible for planting more than 40 million trees on community lands including farms and school and other community property. In 1986, the Green Belt Movement established a Pan African Green Belt Network that has spread tree planting initiatives to other African countries, including Ethiopia, Lesotho, Malawi, Tanzania, Uganda, and Zimbabwe.

Other parts of the world have also seen notable tree-planting successes. After the Cultural Revolution in the People’s Republic of China (PRC), for example, each person was encouraged to plant one hundred trees in “four-around” plantations (around houses, villages, along rivers and canals, and along roads). Since about 2000, the PRC has committed nearly 9 billion dollars annually to its “greening” campaign to raise the country’s forest coverage...
to 20 percent. This will require nearly 17 million hectares planted to trees (Xinhua News Agency 2009). The country stopped cutting natural forests in 1998, which has helped preserve more than 95 million hectares of forest land.

Before the tsunami of 2004, nongovernmental organizations in Aceh, Indonesia, had encouraged some villages to replant mangrove (*Rhizophora spp.*) forests along their coastline; those that did were far less impacted by the tsunami than coastal areas without tree protection. The devastation of Hurricane Katrina in the United States in 2005 also highlighted the need to restore mangrove forests to protect coastlines. In 2011, the government of Kabul, Afghanistan, encouraged shopkeepers and residents to plant trees along the streets that had been denuded by decades of war—the trees are free, but homeowners and shopkeepers are responsible for their care and maintenance. And in the *plano* of Colombia, a solar energy—based village named Gaviotas established a native pine (*Pinus spp.*) plantation which, untended, developed into a nascent rain forest.

In the Indian state of Tamil Nadu, Sadhguru Jaggi Vasudev has created Project GreenHands within his Isha Foundation, with goals similar to the Green Belt Movement in Africa. Since the project’s inception in 2006, some 2 million people have planted more than 8 million trees in more than 1,800 communities. The first mobilized planting there was recognized by Guinness World Records for the greatest number of trees planted in a three-day period. The longer-term goal is to plant more than 14 million trees in Tamil Nadu to bring the state up to about 33 percent tree cover.

In South Korea, where the Japanese occupation (1910–1945) and Korean War (1950–1953) resulted in enormous deforestation, the government established Arbor Day immediately after the Korean War. Celebrated in April and called *simgogil*, “tree-planting day,” it is a day on which people in workplaces, army bases, offices, schools, and villages are encouraged to plant trees. The government’s recent slogan “low carbon, green growth” encourages “green” industrial development as well as tree planting.

**Outlook**

On a global scale, the Green World Campaign plans to reforest the planet, raise living standards for the rural poor, and combat global climate change. This kind of effort, supported by people all around the world, will positively impact the well-being of the planet.

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*Further Reading*


Urban Agriculture

Urban agriculture is the growing, processing, and distribution of food inside the borders of cities and towns. The number of its supporters has been expanding since the 1970s, and adherents view it as a more sustainable way to feed population centers, improve nutrition, and reduce hunger. Urban agriculture takes many forms in practice, including community gardens, urban farms, edible schoolyards, roof gardens, and skyscraper farms.

For some old-time agrarians and modern urbanites, the idea of urban agriculture is an oxymoron: tomatoes and traffic are notions that collide for mind-sets locked in traditional views of each. Yet within the growing food-justice movement (a movement that supports a more equitable distribution of food as a means to end hunger and malnutrition), the concept of farming in the city is resonating deeply—especially among new agrarians and the urban poor across the globe.

The industrial age and global capitalism have led to the progressive decline of small-scale agriculture over the last hundred years, as farmers and farm workers worldwide have migrated to cities, drawn by the promise of new job opportunities and the allure of city life. As the twenty-first century begins, over half of the world lives in cities, and across the globe the shortcomings of our food system as well as the increasing number of “food deserts” in inner-city neighborhoods have become very apparent. As a result, growing practices that were standard prior to World War I are being revived and adapted to local conditions. Today it is common to see small urban plots between office buildings, in alleys, near freeways, and as replacements for once-sacred front lawns. There also has been a revival of Eleanor Roosevelt’s World War II–era victory gardens, and Michelle Obama, the US First Lady, is experimenting with a 102-square-meter garden plot on the lawn of the White House. Meanwhile, a multitude of groups like Food First are working to help local communities establish sustainable food systems.

Catalysts for Growth

Imaginative ideas for redesigned cities that support a sustainable way of living are increasingly being proposed. The ecovillage (a small-scale, intentional community) is a global phenomenon that is in the forefront of this movement, and although these villages emphasize sustainable food production, they are frequently urban based. The ecocities of the US urban ecologist Richard Register (2002) incorporate agricultural plots within walking distance of residents. This design supports the local economy while reducing automobile use and the need for food to be trucked long distances from farm to table. Transition Initiatives and the Cittaslow (Slow Cities) international network are both community mobilization efforts launched from Europe that promote localized edible landscaping.

These movements are inspirational, but more urgent concerns have motivated some people to adopt an urban-agriculture model. In the early 1990s, Cuba found itself isolated when trade with the Soviet Union came to an abrupt end. Since chemical and technological inputs were no longer available, the nation had to rethink the industrialized approach to agriculture that it had relied on. In 1993 the government ended its oversight of state-run farms, and the old ways of sustainable farming were reinstated as biological methods of cultivation replaced technological ones. There are now thousands of decentralized garden spaces across Havana, ranging in size from small parcels to large-scale operations. These plots are worked...
by farmers who grow and sell their harvest, as well as by cooperatives whose purpose it is to grow food for its member-families. Organoponico Vivero Alamar, for example, is a successful garden of 11 hectares in a city that now devotes 35,000 hectares to urban agriculture (Koontz 2009).

For Taja Sevelle, executive director of the nonprofit organization Urban Farming, the goal is to end hunger by using unused land to grow food. Many large cities have areas with limited access to healthy food, and mapping these “food deserts” or “grocery gaps” (Shaffer 2002) typically reveals low-income neighborhoods to be the ones that lack fresh produce or full-service grocery stores. Currently, the city of Detroit, Michigan, may have a full-scale, citywide food desert. Yet, as the investigative historian Mark Dowie (2009) notes, “There is open land, fertile soil, ample water, willing labor, and a desperate demand for decent food. And there is plenty of community will behind the idea of turning the capital of American industry into an agrarian paradise. In fact, of all the cities in the world, Detroit may be best positioned to become the world’s first 100 percent food self-sufficient city.”

Personal need may not be the primary motivation for middle-class suburbanites to grow food, but this population still may find that growing its own food at home is a pleasant hobby with satisfying rewards. Farmscape, a business venture started in Southern California, helps local residents set up and maintain attractive and productive raised-bed vegetable gardens, and teaches them the basics of garden care.

Models of Development

One of the most basic forms of urban agriculture is the burgeoning practice of urban homesteading, in which front lawns are replaced by food gardens and backyard tours reveal egg-laying hens and even mini-breed goats. Worldwide movements like Food Not Lawns International and Edible Estates are proving that growing food near the sidewalk can be beautiful and productive. In some cities, homesteaders are challenging ordinances that might quash their agrarian-minded inclinations.

Community gardens and urban farms are also forms of urban agriculture. Gardens as places to meet neighbors were, ironically, first “organized by the mayor of Detroit in the 1890s to help families cope with the effects of economic depression of that era” (Lyson 2004, 96). A common model for both is for landowners to either loan, rent, or place in long-term trusts blighted spaces that can then be cultivated by community members. The land is divided into plots that are assigned to individuals and families who are interested in cultivating them, and who may pay a nominal fee for water, tools, and maintenance of the space. This model may have its drawbacks, however. Homestead spaces are more likely to thrive if the people working them are growing food for themselves or to sell at market, as opposed to hobby gardeners who may neglect regular maintenance. Financing, planting, tending, and harvesting the whole garden or portions of it collaboratively may correct for this problem by creating accountability and mutual support among people who depend on each other’s participation. This approach may also breed conflict, however, if members of the group disagree on what methods to use for pests and weeds.

Entrepreneurial farmers represent another growing segment within urban agriculture. Wally Satzewich, for example, validated his SPIN (Small Plot Intensive) Farming strategy on twenty-five rented backyards in Saskatoon, Saskatchewan. Although the total land mass of the yards was only one-half acre, Wally was able to earn an annual income of $50,000 from what he grew, and a number of urban farmers across the United States and Canada are now following his model. On a larger scale, companies like Lufa Farms, Gotham Greens, and Sky Vegetables have been successful in using greenhouses and Dutch-style hydroponics on North American rooftops.

Significant synergy occurs when urban farmers bring their harvest to one or more of the six-thousand-plus farmers’ markets in the United States. Organizations like the Food Trust in Philadelphia have been working to open such markets; in that city alone there are close to a half-million people without access to fresh produce. Other urban farmers have developed Community Supported Agriculture (CSA) programs, which provide food to subscribers.
Educational gardening provides an excellent way to beautify an urban schoolyard while teaching city kids about healthy food. Chef Ann Cooper’s work in Berkeley, California, and Boulder, Colorado, is aimed at bringing healthy food and food education back into urban schools (Cooper 2011). With obesity, diabetes, and mysterious food allergies on the rise, people like chef Alice Waters are committed to supporting the establishment of “edible schoolyards,” and her foundation has been instrumental in creating one at an elementary school near her home in Berkeley (Waters 2008). There are also more and more universities like Yale in New Haven, Connecticut, that are getting on the sustainability bandwagon and creating campus-based farms or supporting local garden initiatives.

Near downtown Milwaukee, Wisconsin, Will Allen is teaching his community to grow its own food. In fact, he is calling for 50 million people to get busy doing just that (Royte 2009). When he started Growing Power in 1993, he learned to make compost with red wiggler worms, and soon local youth were eager to join the former pro basketball player. Twenty years later, Allen has worked out numerous agricultural innovations, including a symbiotic system for harvesting fish known as aquaponics.

There are roof gardens showing up globally atop homes, schools, community centers, apartments, and restaurants. Food is being grown in kiddie pools, self-watering EarthBoxes, small containers, and raised beds. There are challenges—getting water up to the roof and the weight of the soil, for example—but the benefits, including improved air quality and cooler buildings, are significant. Meanwhile, Dickson Despommier (2010) and his students at Columbia University have decided that rooftop gardens don’t go far enough, and that the dense population of cities calls for food to be grown vertically in “skyscraper farms.” In cities with limited land available for farming, tall buildings with greenhouses on every floor could be a valuable agricultural solution.

Urban farming is also being practiced in clandestine ways. Guerrilla gardening began in New York in 1973 with the covert conversion of a neglected private lot into a garden that is still tended by volunteers. Tossing prepared seed balls into empty urban spaces has caught on in England and Australia. In 1996, one thousand people in Copenhagen worked all night setting up a garden. For some groups, such as the Abahlali baseMjondolo, a public-housing movement in South Africa, the motivation is to establish sources of food in poor communities.

Kirk Anderson tells an emergent following of urban beekeepers to resist commercial beekeeping methods, saying, “backwards is the new forwards” (Backwards Beekeepers 2011). The influence of Anderson in Los Angeles County, and that of other organic beekeepers worldwide, is being demonstrated as an increasing number of city dwellers participate in bee rescues and establish hives in urban farms and homesteads.

Brenda Palms-Barber started Sweet Beginnings, a retail operation that turns urban honey into skincare products and puts formerly incarcerated men and women to work in the business of beekeeping, manufacturing, and distribution (Sweet Beginnings 2011). The Bronx Environmental Stewardship Training program trains community members (some of them post-prison) for “green collar” jobs, including green-roof installation and urban forestry (SSBx 2011).

**Challenges and Resources**

The movement faces its own unique challenges and has had its share of defeats. The battle between Los Angeles residents who worked the 5-hectare South Central Garden, and the landowner who in 2006 refused to sell them the land that for twelve years had been their source of
food, was heartbreaking. Tragically, this land still sits empty in 2011. Meanwhile, as Walmart enters the organic market, some may feel that growing food locally is not as urgent.

There are also costs that need to be considered when engaging in urban agriculture. Water bills may go down without a lawn, but homeowners may need to replace a sprinkler system with an irrigation system. Inadequate soil quality, industrial pollution, theft, vandalism, and lack of expertise are other issues that could affect urban-agriculture initiatives. Community vision and the attitude of urban planners, redevelopment agencies, and elected officials will help to determine the level of support for agriculture within a city’s limits. Often, the potential for tax revenue from new businesses may limit the ability of municipal leaders to value the cultivation potential of vacant lots. The policies of government at the national level also can affect the adoption of urban agriculture. In the United States, for example, there was concern that the Food Safety Modernization Act of 2010 would include backyard gardeners in its regulatory framework. As it turns out, amendments were added that purportedly protect small farmers; nevertheless, urban agrarians will be watching closely to see how the law works in practice.

There are thousands of vacant lots in urban centers across the globe, waiting for compost and good seed. Think tanks like the Public Health Institute (Stair, Wooten, and Raimi 2008) are creating resources for policy makers that encourage the development of various forms of urban agriculture designed to provide higher levels of food access across the city. The challenge of securing land for agricultural use may be the most important hurdle for urban-agriculture advocates.

**Future Outlook**

The urban-agriculture movement is growing. In the summer of 2009, the publishers of *Hobby Farms* magazine launched *Urban Farm* magazine. There are now many blogs devoted to urban agriculture, such as *Root Simple*, an urban homesteader do-it-yourself site, and Novella Carpenter’s *Ghost Town Farm* blog, which tells her joyous, off-beat, and honest story of raising livestock in an inner-city Oakland apartment. Other stimulating stories, such as that of Michael Ableman (1998) at Fairview Gardens in Santa Barbara, California, and Brian Donahue (1999) at Land’s Sake community farm in New England, provide further insight into the realities and possibilities of urban agriculture.

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**FURTHER READING**


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See also Agroecology; Brownfield Redevelopment; Buffers; Disturbance; Home Ecology; Human Ecology; Indigenous Peoples and Traditional Knowledge; Landscape Architecture; Rain Gardens; Rewilding; Tree Planting; Urban Forestry; Urban Vegetation


The field of urban forestry encompasses the planning, design, establishment, and maintenance of trees and forests in and around cities. Green spaces in urban areas can combat air pollution, provide habitats to support biodiversity, improve the physical and mental well-being of people living near them, and enhance local economies. Since the 1960s, communities have increasingly used urban forestry as a tool to aid in these and other sustainable goals.

Urban forestry is the art, science, and technology of managing trees and forest resources in and around urban communities for the environmental, social, and economic benefits trees provide to people (see Miller 1997; Helms 1998). In its broadest sense, urban forestry embraces a system that includes the management of municipal watersheds, wildlife habitats, outdoor recreation opportunities, landscape design, recycling of municipal wastes, general tree care, and the production of wood fiber as a raw material. Urban forestry includes activities carried out in the city center, suburban areas, and the areas that lie between suburbs and rural lands.

Urban forestry is a relatively new research field that is still developing. It began to emerge in North America during the 1960s as people became more aware of threats to urban tree populations caused by pests and diseases (like Dutch elm disease), and as they recognized the need to include green space in the planning and management of urban areas. Europe and other parts of the world followed, and by the late 1990s an international research network had been established (Miller 1997; Konijnendijk 2003; Konijnendijk et al. 2005). Most recently, communities have come to realize the role urban forestry plays in developing vital green infrastructures (Jensen et al. 2000). (See figure 1 on page 396.)

Benefits Provided by Urban Forestry

Green space helps to make urban areas more healthy and pleasant places to live, and it can increase property values. It also enhances the physical and mental well-being of the people who live there.

Environmental and Ecological Functions

Trees intercept solid and liquid particles and absorb gaseous pollutants like ozone, sulphur dioxide, and nitrogen dioxide, thus removing them from the atmosphere. This suggests that planting trees along heavily trafficked roads and around industrial areas will reduce air pollution, and, in fact, studies conducted in the United States, Germany, and China have confirmed that mass plantings have the potential to improve air quality (Bernatzky 1994; Nowak et al. 2002; Yang et al. 2005).

Trees also lower air temperatures through the shade that they provide and through the water vapor emitted through their leaves. They help to reduce energy consumption and pollution from power plants by shading buildings in the summer and blocking winds in the winter (McPherson and Rountree 1993). Furthermore, green areas can play an important role in limiting the effects of rising temperatures caused by the “urban heat island” effect that heat-absorbing pavement and buildings create (Gill et al. 2007).

Finally, urban green spaces retain and slow the flow of storm-water runoff that threatens the quality of ground-water and wetlands. Green spaces also serve as sites for the germination, establishment, and colonization of different species (Zipperer et al. 2000; Williams et al. 2008). These new habitats support biodiversity by providing a suite of benefits unavailable in the urban-built (human-made) environment.
Social Functions

Urban forests provide important outdoor recreational opportunities to citizens. City life is stressful, and researchers show that urban green areas improve the health and well-being of urban populations. Several studies conducted in Europe and the United States show that visits to green areas can reduce stress, increase energy, and help people heal faster (Nilsson et al. 2011). The environmental psychologists Rachel and Stephen Kaplan (1989) suggest that the constant stress of urban living is relieved by vegetation and nature, which allows us to relax and sharpen our concentration.

Nature in urban areas is also important to increase environmental awareness, since more and more people live in cities and get their ideas about nature from urban environments. The US journalist Richard Louv (2008), author of Last Child in the Woods, coined the term nature-deficit disorder to describe a range of behavioral problems in children—including childhood obesity, attention
disorders, and depression—that result from a lack of exposure to nature. Children who are allowed outdoors during the school day are likely to be more attentive and creative, less anxious, and have better cognitive development.

**Future Directions**

People have become more interested in urban ecology and the environment in recent years, leading communities to invest in parks and green areas. An example is the “million trees” programs that have been instituted in a number of cities all over the world. This trend is likely to continue in the coming decades. Planning to ensure that population growth is sustainable and that all people have access to green space will be especially important in the larger and rapidly expanding cities of Asia, Africa, and Latin America. As the climate changes and energy costs increase, compact cities with landscapes that can retain water, filter air pollution, and provide vital amenities such as food, fuel, and recreational opportunities may become the norm. The art and science of urban forestry will help build cities that are less vulnerable to climate change—bringing nature back to where people live.

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See also Brownfield Redevelopment; Ecological Restoration; Groundwater Management; Human Ecology; Landscape Architecture; Rain Gardens; Road Ecology; Stormwater Management; Tree Planting; Urban Agriculture; Urban Vegetation; Waste Management; Water Resource Management, Integrated (IWRM)

**Further Reading**


Urban vegetation includes all types of plant life found in city environments, from preexisting native species to those introduced to improve landscapes. It is found in urban forests and parks, along roadsides, around ponds and streams, and even in vacant lots. Properly managed urban vegetation can help reduce air pollution, noise, and dust, and add oxygen and visual appeal to otherwise bland cityscapes.

Urban vegetation refers to all types of plants that grow in urban environments, such as forests, parks, roadways, and wasteland areas (Jiang 1993). As a significant part of urban ecosystems, urban vegetation can not only help clean and freshen air quality by reducing dust and environment pollution, but it can also help maintain the ecological balance of urban environments. Urban vegetation also plays an important role in indicating and monitoring environmental pollution.

Categories

Researchers have different ways of categorizing urban vegetation. Some classify it as urban forest, parks and green spaces, gardens and lawns, wall or roof plants, and wetlands (Guntenspergen 1998); others identify roadside trees, greenbelts in streets, green areas in parks, grasslands, and aquatic green spaces (Huang et al. 1990). More simply, some have divided urban vegetation into three types: relict (or remnant) natural communities retained as they were before urbanization, weed communities occupying new urban habitats, and artificial green spaces (Ohsawa and Da 1988). Another way of looking at urban vegetation is according to its three main types: natural plants, seminatural plants, and introduced plants. Natural plants are those that existed before city construction. The major component of seminatural vegetation in cities is the anthropochory community (comprising companion plants), which relies closely on anthropogenic (human) interference under urban habitats and plays a special role in composing urban vegetation, mainly grasses (Jiang 1989). Introduced plants can be categorized into roadside trees, urban forests, parks, gardens, street greenbelts, and so on.

Function

Urban vegetation has multiple functions. The main role, however, is to help maintain the urban environment, which is easily affected by all kinds of pollutants, thereby improving human living conditions. For instance, urban vegetation can adjust microclimatic conditions, clean up air pollutants, reduce dust, dampen noise, and maintain ecological balance. Urban vegetation can also serve aesthetic and educational purposes. Generally, the function of urban vegetation is closely related to the type, for example, forests, grasses, and/or wetlands. Urban forests improve urban environments more dramatically than other types. For instance, the temperature in urban forests can be roughly 6°C–16°C lower than urban open space on sunny summer days. In a case study in the city of Beijing, the air passing through a fruit-bearing forest 80–100 meters wide reduced the concentration of hydrogen fluoride in the atmosphere 22 percent compared to open space of the same width (Wang 1998). A forest belt 40 meters wide also reduced noise about 10–15 decibels (Wang 1998).

Development

Along with global urbanization, the distribution of urban vegetation is being further specialized. According to the
United Nations Educational, Scientific and Cultural Organization (UNESCO), 60 percent of the world’s population will live in cities by 2030 (Wibly and Perry 2006). On one hand, people tend to change urban habitats into areas of development, which exposes them to human disturbance. Adverse effects such as urban pollution become more and more serious, thus putting the urban vegetation into a state of strong instability. A case study conducted in Chiba City, Japan, showed that from 1952 to 1981 forest coverage declined from 51 percent to 8 percent. Meanwhile, land-use patterns had changed dramatically: farmlands and forests had been turned into residential areas and once-natural hills had been covered with buildings (Ohsawa and Da 1998).

**Exotic Species**

On the other hand, various types of urban plant communities appear in cities, such as anthropochory communities and introduced plant communities. Besides, human beings bring numerous exotic species to cities, while they destroy and discard a huge variety of native species. Whether those influences or interferences are conscious or unconscious, direct or indirect, they ultimately alter the natural features of urban vegetation, its composition, structure, and function. As a result, much urban vegetation has completely lost its natural traits (Huang et al. 1990). For instance, urban vegetation depends largely on fertilization, pesticides, and irrigation to live, similar characteristics to crops in agricultural systems.

People like to bring exotic plants to cities for various purposes, but they often pay no attention to local dominant species. Such activities can destroy native urban vegetation. Although relict communities can reflect the distribution of zonal vegetation, the dominant species will gradually disappear and be replaced by those that adapt to urban habitats. As a result, the dominant species in urban plant communities are often not obvious (Jiang 1993). People often neglect comprehensive assessments before widely adopting exotic plants, leading to the uncontrolled spread of invasive species. Such a phenomenon is more likely to occur during the process of introducing herbaceous plants. *Eichhornia crassipes*, or common water hyacinth (native to the Amazon basin), and *Eupatorium adenophorum*, a flowering shrub native to Mexico (also called snakeroot), have brought great harm to both urban and rural environments in China (Bao 2008). In the beginning, people may bring in the invasive species for economic purposes (e.g., *Eichhornia crassipes* was supplied as feed for pigs in the 1950s) or simply to beautify the urban environment (e.g., the ornamental tree *Rhus typhina*), but they later realize the harmful effects when those species become dominant in new spaces such as urban landscapes.

**Proposals**

Since numerous plant species are immigrants to urban habitats, urbanites have become increasingly disconnected from indigenous species and natural ecosystems (McKinney 2006). To counter this, ecological principles should be abided in the selection of urban plants. Increasing awareness of the niche that each species occupies would provide indigenous species a better chance for survival. Choosing and making good use of native species for the greening of urban habitats should be fully considered in the future.

There are still many deficiencies in urban greening, such as a decrease of native species, scarcity of plant diversity, and a lack of ecological background features (Bao 2008). A comprehensive investigation of native species and a study of the genetic diversity of dominant species should be conducted. In addition, selection of indigenous species, especially trees, should be emphasized because of their large biomass and ability to provide habitats for birds and other urban creatures. Introduction of exotic species should be appropriately considered. Furthermore, city designers need to conduct environment impact assessments to avoid the malignant spread of invasive species before introducing exotic species into urban settings.

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See also Best Management Practices (BMP); Brownfield Redevelopment; Disturbance; Ecological Restoration;
Ecosystem Services; Invasive Species; Landscape Architecture; Light Pollution and Biological Systems; Nitrogen Saturation; Permaculture; Pollution, Nonpoint Source; Pollution, Point Source; Rain Gardens; Road Ecology; Tree Planting; Urban Agriculture; Urban Forestry

**FURTHER READING**


Visual consequences due to rapid and extensive growth during the 1900s set the stage for more stringent environmental planning ordinances, guidelines, and controls on development. To protect spectacular viewsheds, the use of analytical techniques and visual quality assessments grew from the 1970s on. In the era of ecosystem management, determining relationships between ecological quality and scenic quality becomes complex and remains controversial.

A viewshed is made up of the areas of land, water, and other environmental elements that can be seen from a fixed vantage point. The visual consequences of rapid and extensive growth and development in the post–World War II era—and the subsequent damage to viewsheds—became ever more apparent during the 1960s and 1970s. During the 1960s, the English landscape architect Sylvia Crowe discussed the importance of the visual character of landscape in her book Forestry in the Landscape. In order to maintain a good pattern of landscape, she says, “there must be both contrasts between areas of open ground and of planting of tree species, of farm crops and of other vegetation” (Crowe 1966, 6). Crowe was instrumental in the use of pencil and pen sketches to reveal proper design of viewsheds and in the manipulation of those viewsheds to improve the visual character of the landscape.

Throughout the late-nineteenth and twentieth centuries, the US landscape designer Fredrick Law Olmsted, British-born landscape designer Calvert Vaux, and US landscape painter Frederic Edwin Church sketched landscapes, carved out important viewsheds, and even borrowed distant views as part of their design, planning, and evaluation of landscapes. Charles A. Birnbaum, the US founder and president of the Cultural Landscape Foundation, provides insight into some of the origins of the term viewshed. He concludes that “as a painter, Church depicted views—at Olana, his Persian-inspired home along New York’s Hudson River, where he ‘borrowed’ views, meaning the design of Olana’s picturesque landscape incorporated the dramatic vistas beyond the property’s boundaries.” (The viewshed from Olana is seen in the above photograph by Stan Ries, looking south across the Hudson valley.) Birnbaum goes on to say that the “significance of Olana’s viewshed is tied to a larger concept, the origins of the American attitudes about scenic conservation—the idea, for example, that it was important to preserve a place like Niagara Falls.” Birnbaum credits Church, who in 1869 worked privately with the pioneering landscape architect Frederick Law Olmsted and the architect H. H. Richardson to build public support for preserving viewsheds (Birnbaum 2011).

**Development**

The term viewshed was used as early as the late 1800s. The National Environmental Policy Act (NEPA) of 1968 addressed large-scale planning, ushering in a new era of environmental planning in which, according to the US landscape architect Ervin Zube, “visual values could be included in the planning and design decision-making process.” He adds, “NEPA made it clear that visual values, the visual quality of the landscape, was not only of concern with reference to uniquely beautiful or ugly landscapes but to all landscapes that were affected by federal design, planning, or management activities” (Zube 1986, 13).

According to Warren Bacon, a landscape architect from the United States Forest Service (an agency of the United States Department of Agriculture, or USDA), “the National Forest Landscape Management program began, as a formal program, at a service-wide meeting in St. Louis...
in 1969 in response to growing agency (Forest Service) and public concern for the visual resource. . . . Because of this concern, it has become appropriate to establish the ‘visual landscape’ as a basic resource, to be treated as an essential part of and receive equal consideration with the other basic resources of the land” (Bacon 1979, 660).

The Visual Management System (VMS), developed by the USDA in 1974 and subsequently modified in 1995, set the standard for integration of aesthetic and visual considerations in large-scale resource management decisions. The VMS included objective criteria such as distance of view and visual magnitude. At the same time, it relied on somewhat subjective definitions of aesthetic landscapes and expert assessment of impacts based on classical principles of art and beauty.

The 1979 conference “Our National Landscape” dealt with applied techniques for analysis and management of the visual resource. Held at Incline Village, Nevada, this gathering brought together more than five hundred practicing landscape architects, environmental and recreational planners, professors, recreationists, industrialists, land and resource managers, researchers, and environmental consultants to focus on visual management problems. This pioneering conference and the simultaneous growth in the use of computer technology were the beginning of interdisciplinary discussions and the sharing of ideas instrumental to a focus on visual landscape assessment.

In 1995, a significant refocus of the VMS developed the Scenery Management System (USDA Forest Service 1995). The Scenery Management System (SMS) presents a vocabulary for managing scenery and a systematic approach for determining the relative value and importance of scenery in a national forest. The handbook Landscape Aesthetics (1995) was written for national forest resource managers, landscape architects, and others interested in landscape aesthetics and scenery. Both students and the general public benefit from the system’s straightforward approach to a complex art and science. Ecosystems provide the environmental context for this scenery management system. This significant revision shifted the focus from a static viewshed perspective to evaluate scenery to a more integrative system. In the context of ecosystem management, the system is used to inventory and analyze scenery in a national forest, to assist in establishment of overall resource goals and objectives, to monitor the scenic resource, and to ensure high-quality scenery for future generations.

The management and protection of scenic resources became a worldwide concern during this same period. Scenic Solutions is a company based in Adelaide, Australia, that provides visual or scenic assessment solutions for land management. They evaluate the visual impact of wind farms in coastal and inland locations in South Australia and measure and map the landscape quality of the South Australian coast (4,800 km) and the scenic quality of the Flinders Ranges in South Australia. The company provides elegant solutions to preserve and protect visual values. New Zealand has developed a comprehensive network of national parks and reserves to manage scenic resources. The European community has been instrumental in associating visual values with cultural or countryside landscapes. The European Pathways to the Cultural Landscape (EPCL) is a project that covers ten countries from Ireland to Estonia and twelve areas spanning a wide range of different landscape and cultural types. Part of this project’s mission is to protect visual values (Déjeant-Pons 2005).

Parks Canada encourages the preservation of scenic values in its national park system. The historic Rideau Canal, managed by Parks Canada to encourage respect for the natural, cultural, and scenic values of the canal’s waterfront lands, is a noted example. Management and protection of scenic resources is a widespread mandate.

The Art of Analysis

A rich history dates to the late-eighteenth and early-nineteenth centuries. Landscape architects used pencil, watercolors, and pen sketches to reveal proper design of viewsheds and the manipulation of those viewsheds to improve the visual character of the landscape. This site-specific use of viewshed depiction was instrumental in helping the designer convey a design idea to a client or the general public.

Computer-based techniques have grown to accommodate the demand to evaluate large-scale landscape...
problems, including scenic quality and visual values. The Incline Village, Nevada, conference in 1979 brought together practitioners from many disciplines to focus on applied techniques for analysis and management of visual resources. Ervin Zube (1986, 16) remarks that computer and quantitative approaches were developed initially for working with large landscapes that may be difficult of access and for which there are available quantitative and spatial geographic data such as slope, vegetation type, elevation, and percentage of tree cover. They provide descriptive information about variability in landscape character and can identify at different viewing points, for example, which area can be seen and which are hidden from view because of topographic configuration or vegetative cover.

Spatial algorithms, graphical interfaces, and output display are more sophisticated than those developed in the late 1970s, but the basic functionality of viewshed analysis remains the same: the study of intervisibility between points on a terrain surface. The concept and most fundamental principal of intervisibility studies is that “if I can see you, you can see me.” The USDA Forest Service and other public land agencies around the world have used visual impact assessment and measures of intervisibility for years.

Intervisibility involves measuring the tangent from the viewer’s eye to each cell. (The size of a cell is totally dependent on the scale of the analysis. The more cells used in a given area means a higher resolution map, like pixels on a screen.) Starting with cells closest to the viewer, a line-of-sight process calculates and maps whether the cell can or cannot be seen, creating a raster map. As long as the tangent increases in the line of sight from the viewer, the cell on a raster or digital map surface is visible. If the tangent decreases, the cell is not visible. For example the backside of a mountain range would not be visible from a viewer’s vantage point. Such spatial analytical techniques provide the ability to assess the effects of various design, planning, and management decisions prior to implementation.

Viewshed analysis is used to site transmission or hydro power lines and to evaluate the visual impact of development and forestry management practices. Fred Henley and Frank Hunsaker, both US Forest Service landscape architects, used mapping techniques to establish visual quality objectives for evaluating forest management practices such as clear-cutting and thinning operations (Henley and Hunsaker 1979). Viewshed analysis has become a standard part of any geographic information system. A geographic information system is a system designed to capture, store, manipulate, analyze, manage, and present all types of geographically referenced data. Advanced visual simulation techniques coupled with psychometric and social science techniques allow the public to see the visual consequences of any proposed development on the landscape and to evaluate stakeholders’ judgments to determine the degree of impact and acceptability of the proposed solution.

Protection Ordinances

Federal land management agencies have been employing visual management techniques since the 1970s. A growing number of towns, cities, and communities are concerned about the visual impact of development and other intrusive features on a landscape. They have been steadily developing viewshed ordinances with the purpose of protecting, preserving, and enhancing scenic views and vistas. Scenic vistas can be a major asset to communities as a draw for new industry. The protection of scenic beauty and the natural environment of hillside areas is vital to the preservation of a high quality of life and continued economic development.

Viewshed protection districts regulate building height restrictions and vegetation loss. These districts are overlay districts used primarily for unique situations regarding views and vistas that are not adequately covered by the standard zoning districts. These ordinances
generally require that no part of a new structure, sign, tower, rooftop equipment, or other appurtenance encroaches on any designated viewshed. If the maximum height allowed in any zoning district within the city differs from the height permitted by a protection district, the more restrictive height limitation applies. New development is usually beneficial, but when construction overwhelms or intrudes, in scale and mass, on the main view or vista, the viewshed should be protected. Viewshed analysis has many implications for cities. Aside from siting housing developments, determining locations for power lines, and protecting natural corridors, it has been used by architects and planners to explore proper orientation of high-rise buildings to improve sunlight effects to buildings, streetscapes, and landscapes that are in their immediate shadow. In addition, proper orientation can improve the availability of skyline effects and improve the marketability of such developments.

Pima County, Arizona, developed the Hillside Development Overlay Zone Ordinance to conserve and maintain the character, identity, and image of Pima County and promote the public health, safety, convenience, and general welfare. It accomplishes these aims in the following ways (Pima County 1985):

• Conserves the unique natural resources of hillside areas
• Permits intensity of development (density) compatible with the natural characteristics of hillside terrain, such as steepness of slope and significant land forms
• Reduces the physical impact of hillside development by encouraging innovative site and architectural design, minimizing grading, and requiring more intense restoration of graded areas
• Minimizes disturbances incurred in development alteration of hillside terrain

Viewshed protection and ordinances have not developed without controversy. Viewshed ordinances aim to preserve scenic beauty, a high quality of life, and continued economic development. Debate centers around the way the ordinances are applied based on the questionable accuracy of the viewshed maps. They may require that property be developed to minimize its visual impact from the major corridors. The legal need to preserve and protect foliage and trees on property can become contentious.

Much of the debate centers around the landowners’ property rights. A one-size-fits-all set of rules may not suit every neighborhood in an area. Viewshed ordinances impose property restrictions and may take away property rights from people who bought their homes in good faith, telling them the trees and privacy they paid for are no longer theirs by placing controls on what can and cannot be done with these trees. Environmental activists say that developers and builders have too free a hand in constructing homes visible from great distances, while property owners who want to build houses along ridge lines say the law is unnecessarily stringent.

In the Age of Ecosystem Management

The Scenery Management System was an attempt to move from a single Visual Management System approach to one that took a more integrative systematic approach to incorporating ecosystem values. Ecosystem management poses serious challenges to assessment and implementation of viewshed protection. The US environmental psychologist Terry Daniel concludes that “ecosystem management has posed significant challenges to forest managers, particularly revolving around the fact that ecosystems are a complex interaction of physical, biological and social processes that are highly interactive over multiple scales of time and space” (Daniel 2001, 275). Perception-based landscape quality assessments are insufficient for determining relationships between ecological quality and visual landscape aesthetic quality.

Advanced computer modeling techniques can evaluate the changes to visual quality from ecological changes. The Australian landscape planners Payam Ghadirian and Ian Bishop explore the changes in visual character in changing environments by combining a geographic information systems–based environmental process modeling with the use of augmented reality technology to evaluate environmental change in an immersive environment. (Immersive environments are virtual environments where an immersant’s awareness of physical self is diminished by being surrounded in the total environment.) These virtual environments allow one to explore, manipulate, and test solutions without affecting the real environment. Landscape planners are using virtual reality environments to address ecological change and to evaluate such spatial-dynamic changes of environments and responses from the public (Ghadirian and Bishop 2002).

Viewshed protection challenges many communities. Aside from the difficulties of ecological change, local citizens drive concerns about viewshed protection. People may be generally in favor of an ecological improvement, such as wind energy, but do not want the necessary infrastructure built anywhere they can see it. They cite noise and the towering appearance of the wind turbines as an issue and strongly feel that they lower home owner property values. Planners may seek alternative locations outside of the community. In 2011, the US Bureau of Land Management entertained numerous right-of-way applications for wind testing in Oregon and Washington.
Environmental groups oppose any such development on public lands. As development and landscape change continues at a rapid pace, saving what remains of our outstanding scenery and its value to the public will continue to be a high priority. Viewshed protection that responds to ecological conditions will continue to be a challenge and a controversy and will inevitably be at the forefront of landscape research.

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See also Comanagement; Landscape Architecture; Landscape Planning, Large-Scale; Light Pollution and Biological Systems; Natural Capital

FURTHER READINGS


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In a nutshell, the ultimate goal of sustainable waste management is to create the conditions where waste is not generated in the first place. Waste is also, however, an unavoidable condition of contemporary market systems that produce short-lived products and urge consumers to purchase more and more goods. Complete elimination of waste—both physically and conceptually—would require structural change in capitalist markets that currently favor so-called planned obsolescence. Short of that transformation, the immediate goal of sustainable waste management is to create economic and regulatory contexts in which waste is reused or recycled rather than discarded in a way that threatens the environment that supports us as a species.

The various categories of waste are principally defined by their origin. Common types include agriculture and forestry waste; waste from mining and mineral processing; nuclear or radioactive waste; construction and demolition waste; non-nuclear hazardous industrial waste; nonhazardous industrial solid waste; and municipal solid waste (MSW) from households, institutions, and commercial establishments. The following sections examine MSW policy and regulation in the United States and then move on to waste management in other industrialized regions before concluding with consideration of the definition and management of MSW in the developing world.

Regulation in the United States

The 1976 Resource Conservation and Recovery Act (RCRA) is the principal federal act regulating waste management in the United States. RCRA covers both hazardous and municipal solid waste, the two separated according to their origin. Hazardous waste is discarded material that
is ignitable, corrosive, explosive, or toxic (including infectious) and is only produced by industry; solid waste may be produced by industry but also includes all waste generated by households or commercial establishments (MSW), even if that household waste includes materials that are chemically hazardous such as batteries, electronic waste, or products such as discarded paint thinner (which collectively make up 1 to 3 percent of the MSW stream by weight).

The differentiation between hazardous and solid waste is important because RCRA requires much more detailed tracking and regulation of treatment, storage, and disposal of categorized hazardous waste than it does for MSW. This results in a cost typically ten times greater (or more) per ton generated for treatment or disposal of hazardous waste than for solid waste—currently thirty to ninety dollars per ton in the United States (van Haaren, Themelis, and Goldstein 2010). (Exact fees vary by region. Most data are proprietary and available via subscription from sources such as the Waste Business Journal.)

Hazardous waste fees vary greatly depending on the actual waste composition, size of the generator, and the method of disposal. The high cost of disposing of the hazardous waste is the main incentive for hazardous waste generators to reduce toxic chemical use (pollution prevention), reuse or recycle their waste into the production process, or attempt to have the residual hazardous waste reclassified as nonhazardous solid waste. Due in part to the political power of vested interests, several types of waste, including agriculture and mining waste, are not classified at all under the RCRA. According to the US Environmental Protection Agency (EPA), mining wastes are considered “special wastes” and have been exempted by the Mining Waste Exclusion from federal hazardous waste regulations (US EPA 2011a). Agricultural waste is not regulated by the federal government except for separated hazardous waste (pesticides) and contaminated irrigation runoff. Concentrated animal feeding operation (CAFO) waste is regulated under the Clean Water Act. According to the EPA, “unless prohibited by other State or local laws, agricultural producers can dispose of solid, non-hazardous agricultural wastes (including manure and crop residues returned to the soil as fertilizers or soil conditioners, and solid or dissolved materials in irrigation return flows) on their own property” (US EPA 2011b).

Sewage sludge, as well as power plant and solid waste incinerator ash containing chlorinated organic and metal concentrations below EPA hazardous standards, is typically treated as solid waste, with the dried sludge or ash disposed of at licensed MSW landfills if it is not recycled into fertilizer or other useful products (for example, building materials).

The amount and type of waste generated by households varies dramatically by income and geography. For example, we tend to find more yard waste in suburban areas, paper office waste in urban waste streams, and beverage containers (often recycled) in college towns. Daily per capita US production of MSW averaged 2 kilograms in 2009, nearly double that for western Europe and more than three times the amount generated in Japan. Though after 2000 per capita production of solid waste decreased, the daily amount is still more than the 1.22 kilograms per person generated in 1960, an increase driven largely by an expansion in packaging waste (US EPA 2010). Medical waste, which is treated as MSW once it has been sterilized, has also dramatically increased, the increase driven in part by concerns over HIV infection and a push for disposables rather than reusable sterilized materials.

Although actual content varies by region, on average, prior to any reuse or recycling, 28.2 percent of the MSW stream in 2009 by weight consisted of paper products and cardboard, followed by food scraps (14.1 percent) and yard trimmings (13.7 percent) (US EPA 2010). Plastics, while only 12.3 percent by weight, make up a much larger quantity by volume due to their light weight.

RCRA encourages states (responsible for implementing federal regulations) and municipalities (responsible for waste collection) to follow a four-step hierarchy:

1. Source reduction through changes in the manufacturing process to eliminate waste, reuse of the waste back into the original production process, and on-site composting of yard trimmings

2. Recycling into new products (including off-site composting)

3. Combustion (incineration) with energy recovery (using the heat from burned waste to create electricity or heat buildings)

4. Landfilling for stabilized residuals—the least preferred method of treatment

In practice, however, while reuse and recycling have increased since the late 1980s, landfilling still accounted for 54.3 percent of the MSW handled in 2009, while 33.8 percent of the generated MSW was recovered, composted, or recycled (US EPA 2010). Although the 11.9 percent that was combusted with energy recovery in 2009 reduced the volume of waste to be landfilled as ash or left as noncombustible by 70–90 percent, the laws of conservation of matter and energy dictate that much of the original waste material was released to the environment as air and water emissions.

Recycling

Recycling is the preferred method for handling waste once it has been produced. It provides benefits across a product’s entire life cycle. Compared with virgin raw material extraction and product formation, recycling uses fewer raw materials, generates lower air and water contamination,
results in reduced greenhouse gas emissions, and generates less demand for waste disposal and landfill space. **Closed loop recycling** refers to post-consumer waste that is used to make the same type of product, as, for example, crushed beverage containers melted to make new containers. **Open loop recycling**, on the other hand, occurs when the waste is recycled into a new product, as with plastic bags and bottles being ground up and reformulated as park benches, plastic flooring, and synthetic carpet fibers.

Approximately one-third of the MSW generated in the United States by weight was recycled or composted in 2009, with the highest rate recorded for paper and paperboard (62 percent) and for yard trimmings (60 percent) (US EPA 2010). Plastics, on the other hand, had only a 7 percent recovery rate, due in part to the complexity of mixed plastics in the waste stream and the comparatively low price for virgin plastic compared with recycled plastic.

The volatile secondary material market (the market for recycled materials) is a major factor influencing recycling rates once consumers separate material for recycling. Hence post-consumer waste put out for recycling might still end up being landfilled if there is no market for the materials. While recycling generally saves on the cost for raw materials and the energy used for production, recycle rates also depend on the commitment to run a separate, and often costly, recycling program, whether materials are separated by the consumer or at a collection and sorting facility. As a result, it is often cheaper for manufacturers to pay for virgin materials rather than work with recycled materials as production inputs.

Despite its “green” allure, recycling is hardly pollution free because melting and reforming metal, plastic, and other material components itself causes contamination. Moreover, local resistance to situating recycling plants nearby tends to push recycling centers, along with garbage incinerators and landfills, down the path of least political and economic resistance, often into low-income areas and communities of color, generating in turn a potential for environmental discrimination and injustice.

Several methods have been used successfully to encourage recycling while cutting back on disposed residual garbage. These include “pay-as-you-throw” garbage collection with higher rates for trash not separated for recycling, recoverable deposit fees (common with beverage containers), development of robust secondary material markets through government purchase, and better product labeling and packaging laws allowing consumers to clearly understand the pre- and post-consumer recycled content of the goods and packaging they are purchasing. Other more politically sensitive measures include mandatory manufacturer “cradle-to-grave” responsibility to take back post-consumer waste, as is common in Europe, and ending tax breaks and various subsidies that lower the cost for virgin materials, as is found with oil and gas depletion allowances or low-revenue mineral and timber removal from public lands. Tightening regulatory requirements, such as stricter toxicity tests for incinerator ash that would limit lower-cost disposal as solid and not hazardous waste, mandatory recovery and reuse of methane from landfills, or stricter standards for treating substances that leach from landfills, would raise the cost for garbage disposal, encouraging source reduction, reuse, and recycling.

In short, while preferable to waste disposal, recycling still focuses on waste after it has been made rather than creating the conditions whereby waste is not produced in the first place. Moreover, judging from heavily promoted advertising campaigns, recycling is particularly popular with producers, because it shifts primary responsibility for waste generation and disposal to consumer demand and choice, rather than requiring them to confront up-front production of products that tend to be discarded as waste.

**Incineration**

Although growing worldwide since 1990, the percentage and absolute weight of waste combusted with energy recovery in the United States has fallen along with amounts landfilled, while composting and recycling have expanded (US EPA 2010). Commonly referred to as **resource recovery or waste-to-energy**, incineration with energy recovery is more common in areas with limited space for landfills, notably the urbanized northeastern United States and across Europe and east Asia. Compared with landfilling, incineration is typically twice as expensive per unit of waste handled due to high costs for air pollution control—principally for volatilized metals, acids, nitrogen and sulfur oxides, particulates, and dioxins produced from the burning of plastic chlorinated hydrocarbons—while it still relies on a landfill for final ash disposal.

Often described as a high-tech solution to a low-tech problem, incineration has fallen out of favor in the United States; only a few new plants have been built since 1990. In addition to the high costs associated with pollution control, another issue is that incineration may compete with recycling for high-thermal-value paper and plastics found in the waste stream, although many communities with advanced waste management programs do maintain high recycle rates despite the presence of waste-to-energy facilities. Notwithstanding, municipalities typically sign long-term contracts with developers and operators in which they guarantee waste streams to keep the boilers operating, thereby limiting future options as waste management techniques evolve. Furthermore, while energy
recovery is common, the limited amount of electricity generated is relatively expensive given the structural modifications required to generate electricity, and typically pales in comparison with energy saved through waste reduction and reuse.

Landfills

Although lowest on RCRA’s hierarchy of waste management options, landfills continue to be the destination for more than half of the MSW generated, principally as they remain the lowest cost option. Prior to the second half of the twentieth century, most public landfills were municipally operated, accepted industrial hazardous waste, and were often unlined. With RCRA’s passage in 1976, thousands of these older landfills were closed and replaced by much larger, regulated landfills that often receive waste from a multistate region and are capable of accepting waste from a million households. Yet the legacy of past mismanagement remains, as locations that accepted municipal waste in the past make up one-third of the priority sites under the federal Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, or Superfund).

As provided under the 1980 Superfund act, the waste generator may ultimately be responsible for a leaking landfill should landfill operators or intermediaries (waste transporters) default on their obligation. For this reason, generators of both hazardous and solid waste tend to send their waste to companies they believe have enough resources (aka the “deepest pockets”) to maintain landfills in acceptable operating condition, and provide bonding (financial assurance) for post-closure monitoring and maintenance. This supports a natural monopoly where a few large waste management companies now dominate the US market. Frequently, these firms take responsibility for an existing landfill threatened with state or federal Superfund status in exchange for a permit to expand the facility, often in areas where local resistance would have prohibited new landfill sites.

Today RCRA-regulated landfills must contain five basic elements:

1. A geologically appropriate substrate with a bottom clay liner to minimize loss of leachate (the liquid component in an open landfill consisting of organic waste and dissolved solids, e.g., hazardous metals leached out by rainwater)
2. A flexible liner system, principally composed of high-density polyethylene plastic along with other impact-absorbing liners
3. A leachate collection system connected to a treatment plant for contaminant removal prior to surface water or local sewer discharge
4. A system to capture and flare methane (a major greenhouse gas produced by anaerobic decomposition of organic waste) preferably with energy recovery
5. Adjacent groundwater monitoring wells

Closing a landfill under RCRA requires the site to be capped with clay and plastic, with leachate treatment, methane capture, and monitoring for an additional thirty years backed by sufficient bonding security. These requirements notwithstanding, landfills basically remain elaborate “baggies” in the ground. They will eventually release their remaining contents (including hazardous household waste) to the environment—a situation that, along with the large amount of space they require, contributes to their low status on the waste management hierarchy.

Global Efforts

While the first documented public waste dump was in Athens c. 500 ce, the practice of hauling organic waste out of urban centers, often to feed pigs and fertilize fields, only picked up in sixteenth-century Europe once waste became more closely associated with vermin and disease (Vehlow et al. 2007). Modern waste collection, separation, and recycling during the twentieth century were linked in part to conservation efforts during World Wars I and II. Today the twenty-seven members of the European Union have a common binding target of 45 percent waste recovery or recycling by 2025, up from the average of 38 percent in 2011 (Fischer 2011). While actual practice varies among member states—with recycling and incineration more common in western Europe and continued reliance on landfills in the new eastern and southern European member states—the trend is away from landfilling (down from 62 percent to 40 percent between 1995 and 2008) toward increased reliance on source reduction, composting, recycling, and incineration (Fischer 2011).

Still popular in land-constrained nations such as Denmark and the Netherlands, European incineration is tightly controlled. Strict up-front source separation eliminates most of the chlorinated waste that contributes to dioxin formation when it is burned, while also allowing higher reuse/recycle rates than are typical in the United States. Landfill for biological waste (including paper) is scheduled to be phased out altogether (Fischer 2011). Moreover, because poorly designed landfills and incineration of carbon-containing waste are major contributors to greenhouse gas emissions, improved waste management is expected to contribute 17 to 18 percent of the European Union’s 2012 commitment to greenhouse gas reduction under the Kyoto Protocol (Fischer 2011).

In the developing world, most rural populations are not served by organized waste collection. Even in urban
areas, only half the population typically has access to pick up and removal. While the vast majority of waste in the developing world is organic in nature and is informally handled through local composting and recycled as fertilizer or fuel, a serious problem remains both with infectious waste and the growing amount of hazardous waste, particularly that linked to the manufacturing of electronic goods with their heavy metals and chlorinated plastics, and with other toxin-containing consumer items produced for a world market.

China provides an instructive study for a nation undergoing rapid industrialization with accompanying waste problems. Economic growth combined with a high rate of urbanization has led China to overtake the United States as the largest MSW generator in the world by total weight. In the process, the composition of the waste stream has changed, for example, from ash due to a previous reliance on coal for home cooking and heating to the plastic packaging characteristic of urban households relying on electricity and gas. Responsibility for waste management is now moving toward producers who may be required to take waste back for reuse or recycling, while at the same time the private market for advanced waste management is expanding.

Although Chinese regulations also give priority to source reduction, reuse, and recycling, the immediate focus is on reducing the hazarousness of waste first, with subsequent incineration and landfill viewed as equal options, though landfills still accounted for the majority of the discarded waste in 2006 (Chen, Geng, and Fujita 2010). By comparison, as might be expected for a dense island nation, Japan, with over half the world’s total installed combustion capacity, relies heavily on incineration for waste disposal, with landfills reserved primarily for the ash produced. Heavy emphasis on source reduction limits that nation’s per capita MSW generation to only one-third that of the United States, though Japan is still a world leader in the amount of plastic waste discarded—much of it incinerated rather than recycled.

With the United States the only industrialized nation that did not ratify the 1989 Basel Convention regulating the global trade in toxic waste, the majority of electronic consumer waste collected for recycling there ends up in India, China, Pakistan, and African nations, often with no guarantee of sound recycling management. In these developing nations, waste—both imported and domestic—remains an important part of the local economy. China still has several million people employed in waste scavenging, a notoriously dangerous enterprise involving “backyard” melting of toxic metals and plastics to produce materials for the secondary materials market.

Outside of urban manufacturing centers in India and across the developing world, the waste generated is largely organic and composted locally or disposed of in open dumps, with scavenging common for recoverable plastics and metals. The urban waste scavengers of India—typically women and children—are important for sustainable waste management, but their exposure to very dangerous working conditions, low social status, and bare subsistence have inspired social reforms involving improved access to social services and medical care.

Sustainable waste management in the developing world, as elsewhere, requires the development of the cultural, political, social, and economic conditions that will minimize the generation of waste and encourage reuse or recycling. These reforms must be sensitive to local conditions, however. The current trend toward modern waste collection and management, for example, threatens the livelihoods of millions of impoverished individuals still dependent on the informal waste sector for their material existence. Improving waste management may have unexpected social, cultural, and economic effects.

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See also Agroecology; Brownfield Redevelopment; Carrying Capacity; Human Ecology; Microbial Ecosystem Processes; Permaculture; Pollution, Nonpoint Source; Pollution, Point Source; Urban Agriculture; Urban Forestry; Urban Vegetation

**Further Reading**


Integrated water resource management (IWRM) is an approach in which water is managed in conjunction with land and other natural resources. Environmental, economic, and social aspects of a watershed area are considered together, and stakeholders are engaged in the management process. Effective IWRM is based on a clear vision regarding a desirable, sustainable future; a focus on key variables; integration of management efforts at different spatial scales; and stakeholder engagement.

Water resource management is challenging because usually more than water needs to be considered. For example, basic causes of flooding can include removal of forest cover and wetlands to facilitate increased agricultural production or expansion of cities and related economic activities. Therefore, to reduce flooding, managers cannot focus only on controlling water in river channels; associated land and other resource systems with implications for water flow need to be considered. An integrated approach that considers a full watershed area, the human activities and needs within that area, and the local ecology can contribute much to long-term sustainability of resources and ecosystems.

Another challenge for water resource management is that the responsibility and authority of government agencies related to water resources usually apply to areas defined by political or administrative boundaries, while the natural boundaries of surface water systems, delineated by watersheds, or by aquifers for groundwater, rarely align with these political or administrative units. Such misalignment raises governance challenges that government agencies normally are not well equipped to handle. In addition, water is essential for life. No alternatives exist. As a result, decisions about water allocation and use often focus on meeting the needs of human beings and, increasingly, of livestock and wildlife. Water also has cultural and economic significance and needs to be considered in light of an area’s history and its fishing, recreational, and tourism industries.

Integrated water resource management (IWRM) has emerged to allow water managers to address the types of issues noted above from an integrated or systems perspective. IWRM therefore is a tool or a means for better water management, which includes protecting the sustainability of water resources for human use and for ecosystem stability. For IWRM to be effective, however, managers and societies need a clear vision or direction regarding what water future they desire. IWRM by itself cannot deliver a desirable future, but rather it operates as one means to help achieve a desired end state.

Development and Definition

Many would agree that IWRM emerged from the United Nations Conference on Environment and Development (UNCED), known as the Earth Summit, in Rio de Janeiro in 1992. Its roots extend back many decades, however. In 1914, in the US state of Ohio, the Ohio Conservancy Act provided opportunity for “conservancy districts” to be created to prevent floods, regulate stream channels, reclaim waterlogged land, provide irrigation, and control stream flows. The following year, the Miami Conservancy District was created in Ohio, becoming one of the first river basin management agencies in the United States. It was followed in 1933 by the Muskingum Watershed Conservancy District in eastern Ohio, which focused on flood control, soil conservation, recreation,
and parks. Also in 1933, the Tennessee Valley Authority (TVA) was established with responsibility for the entire Tennessee River basin, the fifth largest river system in the United States. The legislation enabled the TVA to manage the Tennessee River system for flood control, hydroelectric power, and navigation. In addition, the TVA was intended to stimulate economic development in a region that then was one of the poorest in the nation. As a result, the TVA became another early multipurpose watershed management agency designed to deliver economic, environmental, and social benefits.

In other nations, similar initiatives were adopted. For example, in 1946, the province of Ontario in Canada passed the Conservation Authorities Act, which allowed the creation of river basin agencies that would be responsible for integrated water and land management; by 2011, thirty-three conservation authorities were in place in Ontario. In 1941 legislation was approved in New Zealand that focused on soil conservation and river control, placing New Zealand among the first nations to recognize the need to manage water and land in an interrelated manner on the basis of catchments. During the same decade, in England and Wales, a Rivers Board Act made provision for river basin agencies in most areas of those two countries. By the end of the 1940s, water managers had recognized the advantages of managing water, land, and other resources together, as well as addressing environmental, economic, and social matters together.

One outcome of the 1992 Earth Summit was endorsement of a principle for water management (developed at a pre-summit meeting in Dublin): “Since water sustains life, effective management of water resources demands a holistic approach, linking social and economic development with protection of natural ecosystems. Effective management links land and water use across the whole of a catchment area or groundwater aquifer” (ICWE 1992). This principle often is identified as the foundation for articulation of IWRM. Later, in 2000, the Global Water Partnership (GWP), an organization based in Sweden involving a mix of groups interested in water, published a definition of IWRM that has become the most frequently cited. According to the GWP (2000), IWRM is “a process which promotes the coordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems.”

Implementation

For IWRM to be implemented successfully, at least four elements need to be present.

First, water managers need to work with managers responsible for other natural resources to ensure consensus about a vision regarding the desired future for a watershed or region. Such a vision, or goal, has to be determined before IWRM, a means, can be applied to help achieve the desired future.

Second, care is needed to differentiate between comprehensive/holistic and integrated approaches. By definition, in a comprehensive or holistic approach managers must consider every variable in water and related natural resource systems, as well as every relationship among them. This interpretation underlay what was called “comprehensive river basin management” in the 1950s and 1960s. Trying to examine all variables and relationships resulted in too much time being taken to complete comprehensive river basin plans, to the extent that they were often out of date by the time they were finished (Mitchell 1983). In addition, they often provided so many un prioritized recommendations that it was not clear which agencies were responsible for specific recommendations. In contrast, an integrated approach focuses on key (not all) variables and relationships, with particular attention to those for which action is possible. Determining key variables and relations is achieved by drawing on previous research as well as local experiential knowledge.

Third, IWRM needs to be conducted at different spatial scales. Ideally, a first IWRM plan should be developed for an entire watershed or river basin, with attention to goals and targets for the entire basin. Following such an overall strategic plan, more specific plans then should be developed for subcatchments or subbasins, followed by increasingly detailed plans for tributaries and even specific sites. Recognizing different spatial scales helps managers determine which aspects need attention at each
scale, and helps them avoid going into too much detail at any one spatial scale.

Fourth, a defining feature of IWRM is a commitment to partnerships or engagement with stakeholders: managers must recognize that government agencies with responsibility and authority for water management are not sufficient by themselves to achieve effective governance. Although a collaborative approach may, in the short run, add time to the development of a watershed plan, the sharing of views about problems and solutions will, in the long run, be more effective than a government agency moving forward on its own to decide which action is needed.

Issues

Despite general use of the GWP definition, other definitions of IWRM continue to be put forward. As a result, it has been questioned whether it is possible to know what IWRM is or when it has been achieved (Biswas 2004).

While different definitions do exist, they almost all share several concepts: water must be considered in conjunction with land and other natural resources; the watershed or catchment is a more appropriate spatial scale and governance area for water resources management than an administrative or political unit; attention should be given to environmental, economic, and social considerations; and opportunity should be provided for stakeholder engagement. These common elements represent a consensus view of the critical features of IWRM and hence adequately define it as a water resources management tool.

Some critics have argued that no examples exist of successful implementation of IWRM and that no “objective assessments” have been conducted of IWRM experiences (Tortajada 2010).

IWRM has, however, under various definitions been introduced and implemented in both developed and developing countries. Objective evidence of accomplishments has been provided since 1999, when the International River Foundation (sic), based in Australia, began to award its International River Prize to recognize outstanding long-term outcomes. Awards have been given to organizations in the United Kingdom (1999, 2010), the United States (2004, 2008), Canada (2000, 2009), Australia (2001), China (2006), France (2005), and Israel (2003), and to the Mekong River Commission (2002) and the International Commission for the Protection of the Danube River in Central Europe (2007). The UK Environment Agency received the award in 2010 for its work focused on London's River Thames. Many of these awards have explicitly recognized accomplishments through IWRM programs. At the same time, implementation of IWRM can be challenging, and in some countries modest or little progress has been made.

Critics have also suggested that if a comprehensive or holistic approach is advocated by IWRM, then still other areas of human society should be taken into consideration, such as energy development and use, and poverty and health (Biswas 2004; 2008a). Supporters of IWRM observe that indeed many interrelationships occur with respect to water, but, as in all problem-solving situations, boundaries or limits have to be established or the scope becomes overwhelming and little is accomplished (Kidd and Shaw 2007).

Very few people consider IWRM to be the sole consideration or tool that should be used in water resource management.

As with any policy implementation, the adoption of IWRM requires creation of some new structures and processes to support it. Critics argue that its implementation will be futile unless there is strong commitment in political, administrative, and financial arenas. It is even alleged that too often politicians or senior bureaucrats only give lip service to IWRM because they do not want to relinquish responsibility for aspects of water management now under their control or authority. Furthermore, it has been argued, in some jurisdictions the presence of corruption ensures that IWRM will not move forward because it would threaten well-established interests (Biswas 2008b).

These criticisms may reflect reality, but resource management will benefit if constructive solutions to these and other issues surrounding an integrated approach can be developed.

Outlook

IWRM is not a silver bullet that can “fix” all problems regarding water and related natural resources. It was developed as one means to ensure that the connections between water and other natural resources are considered together, that upstream and downstream aspects of water systems are recognized, that water quantity and quality are addressed together, and that ground- and surface water receive attention at the same time. IWRM also aims to ensure that attention is given to environmental, economic, and social considerations, and that stakeholders have the opportunity to shape decisions. If all of these aspects are considered and acted upon, prospects are good for enhanced water management.

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See also Best Management Practices (BMP); Catchment Management; Coastal Management; Complexity Theory; Dam Removal; Ecosystem Services; Groundwater Management; Human Ecology; Hydrology; Irrigation; Rain Gardens; Stormwater Management; Waste Management

**FURTHER READING**


In the United States and several other countries, the word wilderness has a legal definition. These wilderness areas, protected by legislation, have many purposes, from ecological preservation to recreational use. By 2010, the National Wilderness Preservation System in the United States included 791 management units and more than 109 million acres of publicly owned lands managed by four federal agencies.

There are few places in the world that have not been under human control, habitation, cultivation, or influence at some point in history. The impact of humans ranges from urban centers that are very heavily influenced, through rural areas, to wilderness that is much less developed. The so-called human footprint on the world is large and extending rapidly with population growth, road building, food production, power generation, and industrialization (Sanderson et al. 2002). Some identifiable “last of the wild places” exist on each continent and may continue to do so with careful conservation of resources and global protection.

Wilderness Defined

In the early years of the United States, European immigrants cultivated and tamed the wild places and took dominion over the land for human habitation. Wilderness was seen as a place for exploration and primitive travel and was often feared and avoided by most of the population (Nash 2001). As people realized that the amount of remaining land that preserved wild conditions was diminishing, they began to appreciate the wilderness as a contrast to urban development. The public’s interest in wild places evolved as wild areas became scarce. In the late nineteenth century, a few notable places in the United States were set aside as national parks: Yellowstone, Yosemite, and the Grand Tetons. These parks were seen as areas for recreation and tourism rather than as ecological preserves. After World War II, a greater public interest in preserving areas for wilderness emerged. Some of the desire to designate areas for protection as wilderness was due to people’s increasing leisure time for recreational pursuits coupled with their growing concern about the way rapid industrialization and population growth was transforming the landscape.

Some scientists argue that there are few or no places left in the world that are wilderness in the strictest sense of the word, because the world’s ecosystems have all been affected by human activity, particularly the warming due to increased atmospheric carbon dioxide. For this reason, the term wilderness is usually used to refer to areas that are little known or predominantly under the influence of natural processes and forces. Although the term was once commonly applied to any large, remote area with natural characteristics, conditions, and processes, by 1964 the term gained a legal definition that was applied to federally owned land designated as wilderness by congressional action in the United States.

Policy

Legislative protection was needed to create a more permanent and coordinated national system for wilderness preservation and management. To get a wilderness bill passed into legislation, political compromises were necessary, and so the law was drafted to permit certain human activities in some areas, even though those activities would not conform to the intent of the wilderness legislation. These included activities such as mining, grazing, aircraft landings, and water resources development.
In 1964, the US Congress passed the Wilderness Act (US Public Law 88-577) and created the National Wilderness Preservation System (NWPS). The Wilderness Act defines a broad statement of policy for designating wilderness:

In order to assure that an increasing population, accompanied by expanding settlement and growing mechanization, does not occupy and modify all areas within the United States and its possessions, leaving no lands designated for preservation and protection in their natural condition, it is hereby declared to be the policy of the Congress to secure for the American people of present and future generations the benefits of an enduring resource of wilderness. (US Public Law 88-577, section 2a)

Section 2c of the Wilderness Act includes an important and often-quoted definition of wilderness that has led to much controversy and debate because it has left room for administrative and legal interpretation. The definition has been interpreted in a variety of ways to make it practical and applicable:

A wilderness, in contrast with those areas where man and his own works dominate the landscape, is hereby recognized as an area where the earth and its community of life are untrammeled by man, where man himself is a visitor who does not remain. An area of wilderness is further defined to mean . . . an area of underdeveloped Federal land retaining its primeval character and influence, without permanent improvements or human habitation, which is protected and managed so as to preserve its natural conditions and which (1) generally appears to have been affected primarily by the forces of nature, with the imprint of man’s work substantially unnoticeable; (2) has outstanding opportunities for solitude or a primitive and unconfined type of recreation; (3) has at least five thousand acres of land or is of sufficient size as to make practicable its preservation and use in an unimpaired condition; and (4) may also contain ecological, geological, or other features of scientific, educational, scenic, or historical value. (US Public Law 88-577, section 2c)

As more and more visitors use wilderness resources, the areas have to be managed in different ways. Managing an area that is intended to be free of the influences of modern human activities may appear paradoxical. Wilderness stewardship, however, protects and preserves an area’s solitude and natural features (Dawson and Hendee 2009).

The NWPS has selected and established wilderness areas that represent different ecosystem types in order to preserve natural conditions and processes. These wilderness areas serve as the genetic pool necessary to support sustainability both inside them and in surrounding areas, especially where they are connected to other necessary natural features, such as high mountain landscapes that are the habitat of grizzly bears.

**Potential Threats**

After an area is designated as wilderness, the area must be maintained to preserve the ecosystem. Numerous types of internal and external conditions, influences, and changes threaten wilderness resources and values, now and in the future. Two examples of nineteen specified threats that highlight the concern about the future sustainability of wilderness conditions and processes are habitat fragmentation and the introduction of exotic (non-native) plants and animals (Dawson and Hendee 2009).

Wilderness areas are increasingly isolated fragments or remnants of historic ecosystems. As more and more people move into the surrounding landscape, wilderness areas become ecologic islands. These islands can continue to thrive only if they are of a substantial size or are connected to other natural areas. Fragmentation is most pronounced in the eastern United States, where wilderness areas are relatively small, but the threat is felt throughout the country. Exotic and non-native species of plants and animals are a direct threat to naturalness and wildness. The effort to control these invasive species can itself have undesirable impacts on wilderness conditions. Invasive plant species like knapweed, cheatgrass, and purple loosestrife can rapidly change an ecosystem and fundamentally alter its native plant and animal species. Few of the enumerated threats to wilderness areas will diminish, and most are projected to increase in the coming decades. Land managers will need to monitor these potential threats and develop management plans to steward wilderness areas and minimize, mitigate, or remove the threats.

**Management Agencies**

At the national level, four federal agencies are responsible for wilderness planning and management activities. They are the National Park Service (NPS), the Bureau of Land Management (BLM) and the Fish and Wildlife Service (FWS) in the Department of Interior, and the Forest Service (FS) in the Department of Agriculture. The four agencies have devised regulations based on legislation and have developed policy and management documents to steward the lands under their jurisdiction. In addition, all four continue to evaluate and manage additional land for potential inclusion in the NWPS. Although the NWPS is a national system, each agency has developed
its own procedures and organizational approach to protecting the “enduring resource of wilderness,” based on its own administrative mission and structure. Some of the different approaches to visitor and resource management can be confusing to a public that does not understand that each agency has a different mission. For example, the FWS has a unique wildlife management mission that incorporates a national wildlife refuge system.

The 109-million-acre NWPS represents just over 4.5 percent of the US land area, in contrast to the more than 6 percent of total acreage in urban and suburban land area and more than 20 percent of the total in agricultural cropland (Dawson and Hendee 2009). Although the NWPS attempts to protect areas that represent different geographic regions and ecosystems, not all US ecosystems are included (less than 50 percent of types are represented), and more arid lands and mountain ecosystems of the west are included than coastal lowlands, grasslands, and eastern hardwood forests.

In addition to the National Wilderness Preservation System, based on federal land ownership, twelve states have designated state wilderness areas on state-owned lands since the 1970s and they protect more than 3.2 million acres (Propst and Dawson 2008). These are managed by the state land managing agencies and are not part of the NWPS.

Management Principles

The guiding principle for managing wilderness areas is that wilderness should be managed as a pristine extreme in the landscape to maintain the distinctive qualities that define and separate wilderness from other land uses (Dawson and Hendee 2009). Wilderness management is biologically centered, that is, environmental integrity and primeval conditions of wilderness are the foundation for human enjoyment, values, and benefits.

Management that focuses on wilderness as an ecosystem, not as a separate set of resource types (for example, water, forests, wildlife), provides a comprehensive view of the protected area. Most wilderness areas represent the remnants of ecosystems, or entire ecosystems, which need to be protected for present and future generations to experience and enjoy. In addition, it is imperative that human impact be managed to preserve wilderness, because without such stewardship, these remaining areas will lose their unique value in the US landscape.

If wilderness is to be managed to maintain or improve wilderness conditions, an understanding of a particular area’s carrying capacity to sustain recreational use is essential. One of the major components in overseeing such recreational use is to manage in favor of activities that depend on wilderness conditions to achieve their goals, while not degrading the wilderness conditions. Only those activities that require such conditions should be allowed in wilderness, and only as much activity as the area can sustain while maintaining its wilderness conditions and processes should be permitted.

National and International Movements

The US public is strongly supportive of wilderness designation and the NWPS (Cordell, Bergstrom, and Bowker 2005; Cordell, Tarrant, and Green 2003). A summary of seven different surveys in the United States from 1999 through 2002 showed that 48 to 81 percent of respondents supported designating more wilderness land in the United States (Scott 2004). Although there is widespread public support for wilderness, views diverge on how to define wilderness, ranging from extreme protectionists who believe that humans have no place in wilderness to the utilitarian interests that believe that wilderness is a setting for future economic development of recreation and tourism activities.

Membership in organizations that promote wilderness designations, stewardship, information, and education, such as the Wilderness Society and the Sierra Club, has grown dramatically over the last forty years. People have mobilized to protect wilderness at all levels, international, national, state, and local.

The US legislative model has influenced wilderness protection globally, although the variation in level and type of protection is based on the cultural and legislative history in each country (Kormos 2008). The concept of wilderness is universal, and the national legislative approach used in the United States has been
widely adopted by other countries such as Canada, Australia, Finland, Russia, and South Africa (Martin and Watson 2009). Many countries have both strong public support for wilderness and active related organizations that support wilderness designation and stewardship.

Wilderness preservation is a national and international movement comprising grass roots and membership organizations interested in protection and stewardship of dwindling wild areas. The value of wilderness is supported by the general population of the United States and many other countries, but the continued support and work of many people and organizations will be needed to stimulate the legislative and administrative branches of governments to continue their efforts to maintain wilderness for present and future generations.

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See also Adaptive Resource Management (ARM); Administrative Law; Best Management Practices (BMP); Biological Corridors; Boundary Ecotones; Carrying Capacity; Ecosystem Services; Edge Effects; Forest Management; Habitat Fragmentation; Human Ecology; Hunting; Refugia; Road Ecology

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Ecosystem Management and Sustainability analyzes myriad human-initiated processes and tools developed to foster sustainable natural resource use, preservation, and restoration. It also examines how humans interact with plant, marine, and animal life in both natural and human-altered environments. Experts explain the complex ecosystem relationships that result from invasive species, roads, fencing, and even our homes—by addressing topics such as fire and groundwater management, disturbance, and ecosystem resilience. Because most people in the twenty-first century live in urban environments, the volume pays special attention to the ecology of cities, with detailed coverage on topics ranging from urban agriculture to landscape architecture. The volume focuses on how ecosystems across the world can be restored, maintained, and used productively and sustainably.