Monitoring Fish Stocks and Aquatic Ecosystems

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Finally, we would like to say thanks to all who attended the 2005 ASFB workshop and conference for making it a very successful and enjoyable event.

Foreword

Dan Gaughan
ASFB President, 2005

A key objective of the Australian Society for Fish Biology is to foster the expertise of our nation's current and future aquatic resource scientists and managers. In achieving this objective, the Society's annual workshop and conference has become a leading forum for the exchange and discussion of information pertaining to all fields of aquatic science. The proceedings which have accompanied these events have also developed into an important tool for documenting the latest in aquatic research and management.

Each year, the Society explores a new topic in its annual workshop. In 2005, the workshop was hosted by the Northern Territory Department of Primary Industry, Fisheries and Mines. The theme for the workshop was '*Monitoring Fish Stocks and Aquatic Ecosystems*', which built on the previous year's theme of '*Ecosystem research and management*'. Reviewing the way we approach monitoring in Australia was considered timely given recent widespread changes in the way we utilise and manage aquatic resources.

A measure of the importance of this topic was the financial support provided by the Fisheries Research and Development Corporation, the Department of Primary Industry, Fisheries and Mines, the National Oceans Office, the Department of Environment and Heritage, Envirofund, the West Australian Department of Fisheries, the Murray-Darling Basin Commission, the South Australian Research and Development Institute, and the Tasmanian Department of Primary Industries, Water and Environment. All these organisations have expressed a key interest in the way we approach monitoring in this country.

Close to 150 people participated in the two day workshop, with delegates coming from across Australia, as well as from the United States, Uganda and New Zealand. Thirty-nine presentations explored a wide range of topics, encapsulated under the following sub-themes:

- Monitoring commercial, recreational and indigenous fisheries.
- Fisheries independent monitoring approaches
- Ecosystem-based approaches.

The 2005 workshop was planned to coincide with nine other related events hosted in Darwin over a two-week period. They included the Society's annual two-day conference, the Australian Marine Science Association's annual conference, a national barramundi workshop, several meetings and workshops hosted by Seafood Services Australia and the Australian Seafood Industry Council, and a series of seafood industry promotional events, including the Northern Territory Seafood Festival. It was estimated that these events attracted over 500 people to the Northern Territory and involved over 15 000 visitors and locals.

It goes without saying that a large amount of work occurred behind the scenes in order to ensure the smooth operation of this massive undertaking, which I believe we successfully achieved as shown by the tremendous amount of positive feedback. The voluntary contributions of a large number of people made this possible. Please see the acknowledgements for further details. While thanks are due equally to all, a debt of gratitude is owed in particular to Andria Handley (DPIFM), Paul de Lestang (DPIFM), Michael Phelan (DPIFM) and Gaye Messer (Best Conference and Events Company) for their fundamental role in presenting the 2005 workshop and conference.

About This Guide

Michael Phelan
Convener
2005 ASFB Workshop

The sustainable management of fish stocks and aquatic ecosystems is intrinsically dependent on effective monitoring. Reflecting the importance of monitoring, management agencies across Australia employ numerous programs to detect and profile change in aquatic environments. These monitoring programs are utilised to underpin and justify management decisions and funding allocations. It is therefore critical that the information they produce is as accurate as possible.

The aim of producing 'A Guide to Monitoring Fish Stocks and Aquatic Ecosystems' is to enhance the way we utilise monitoring programs in Australia. This guide was developed to assist scientists, students and volunteers in selecting appropriate monitoring methods and protocols. It is hoped that this guide will stimulate interest in new ideas and concepts, and will serve as a useful reference for anyone currently involved in the process of implementing a monitoring program.

The development of this guide capitalised on the congregation of almost 150 scientists, managers, students and stakeholder representatives who gathered in Darwin during July 2005 for a two-day workshop on '*Monitoring Fish Stocks and Aquatic Ecosystems'*. Leaders in the field of monitoring, participants in long-term and short-term monitoring programs, and the end users of monitoring data, gathered to exchange information on current and future techniques. This document conveys that information.

This guide opens with an 'Introduction to Monitoring Fish Stocks and Aquatic Ecosystems'. The paper is followed by an insightful discussion of the 'Essential Concepts of Effective Monitoring'. The paper was produced by James Scandol, the elected convenor of the 'ASFB Monitoring Committee' formed at the conclusion of the workshop. The final introductory paper 'Monitoring Fish Stocks and Aquatic Ecosystems: Guidelines for Consideration' was produced by the ASFB Monitoring Committee.

Following this are papers by the workshop's international keynote speakers, Ron Taylor and Oliva Mkumbo, which provide an insight into methods and progress of key monitoring programs currently undertaken in North America and Africa. A series of papers follow, each presented by delegates who participated in the monitoring workshop. The study areas and topics they cover are far-reaching and diverse. The abstracts of all presentations at the workshop are also provided, along with the contact details of each author.

While this document is not intended to be a complete, one-stop reference to monitoring fish stocks and aquatic environments, I hope you will find it thorough and diverse enough to provoke new thoughts and generate greater understanding. A series of comprehensive monitoring manuals already exist, covering almost every type of aquatic habitat. I strongly encourage you to refer to them. A selection of these manuals is listed in the appendices. Other excellent sources of information do exist and will undoubtedly continue to grow as we develop new and improved methods for monitoring.

Introduction to monitoring fish stocks and aquatic ecosystems

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Introduction

Monitoring may be defined simply as the process of collecting information for the purpose of detecting change. Those who monitor fish stocks or aquatic environments may gather information on particular species (see Lintermans in this publication), on components of ecosystems (see Buckle et al., in this publication), or on the people who utilise these natural resources (see Neil in this publication). Such information may reveal changes ranging from shifts in catch rates (see Ryan et al., in this publication) to shifts in predator-prey relationships (see Dunlop and Maxwell in this publication).

The papers presented at the 2005 ASFB Workshop on *'Monitoring Fish Stock and Aquatic Ecosystems'*, highlighted the diverse range of monitoring programs currently conducted in Australia. Through this diversity, monitoring programs are conducted to serve a wide variety of purposes, including gaining information for:

- Stock assessments or evaluations of management strategies (see Jebreen et al., in this publication).
- Meeting statutory obligations such as those introduced under the *Environment Protection and Biodiversity Conservation Act* (see Begg et al., in this publication).
- Cross-checking existing data, or providing an alternative source of information (see de Lestang in this publication).

The monitoring programs presented at the workshop ranged from continuous long-term projects spanning many years (see Dunning in this publication) to short-term snapshot projects (see Phelan in this publication). The scale of the projects also ranged from Australia-wide (see Coleman in this publication), to regional (see Williams et al., in this publication), to site-specific (see Karlov in this publication). Some projects tested the effectiveness of monitoring options (see Sumner in this publication), providing a template for other monitoring programs.

The information presented at the 2005 ASFB Workshop is summarised in the following pages. This summary covers presentations by delegates at the workshop (see Chapter 5 for abstracts) and session summaries provided by Dr. Norm Hall, Dr. Neil Gribble and Dr. Jeremy Lyle. While every effort was made to accurately report the views of the presenters in this discussion, the information may in some instances vary from that intended by the authors. Readers are therefore encouraged to either contact the authors (see Appendix 1) or refer to primary publications (see Appendix 2).

Monitoring of commercial, recreational and indigenous fisheries

Commercial, recreational or indigenous monitoring data may be sourced directly from fishing operations (see de Lestang et al., in this publication), or alternatively, it may be collected independently (see Chick in this publication). Fishery-dependent monitoring may gain information through such methods as logbooks (see Ziegler et al., in this publication) or observer programs (see Koopman et al., in this publication). Fisheries-independent monitoring may gain information through such methods as depletion studies (see Hay et al., in this publication) or visual surveys (see Buckle et al, in this publication).

Monitoring programs may also adopt a combination of fisheries-dependent and independent monitoring methods. Williams et al. (in this publication) provide a relevant example of a project that is successfully utilising both methods. They explain that in the eastern Torres Strait Coral Reef Finfish Fishery, a range of fisheries-dependent and independent monitoring methods are used to collect information from all sectors. They state that this approach caters for different needs, fishing practices and motives of each sector.

Begg et al. (in this publication) warn that because most fisheries tend to be multi-sector, multi-species and spatially heterogenous in both fishery and biological dynamics, designing an effective program of monitoring can be notoriously difficult. Hall, in his session summary, also cautioned that species structure and dynamics are often far more complex than is assumed. He strongly recommended that these factors be carefully considered when designing or reviewing a monitoring program.

Buckworth et al. (in this publication) provide a relevant example of the arguments presented by Begg et al. and Hall. They explain that stocks of narrow-barred Spanish mackerel (*Scomberomorus commerson*) form small functionally-distinct assemblages, connected by larval and juvenile interchange at low levels, and only a small degree of adult mixing. They argue that in order to avert the risk of small spatial scale depletions, monitoring and management of Spanish mackerel and similar fisheries must be sensitive and robust at a fine scale.

Ziegler et al. (in this publication) provide a similar example with Tasmanian scale fish fisheries. They claim because conventional data-intensive assessment techniques cannot be justified due to the high costs of data collection, simple analyses of fishery data is likely to remain intrinsic to future stock assessments. However, they caution that because many reef fish species exhibit spatial structuring, there is a mismatch between the relatively large-scale of data reporting and the fine-scale of population dynamics. They also indicate that in order to reduce the potential for masked serial depletion, the spatial resolution of data collection needs to match as closely as possible that of the stock processes.

Historically, commercial fisheries have provided much of the monitoring data collected in Australia, with relatively little being obtained from recreational fisheries (see Smith in this publication). The same applies to indigenous fisheries (see Sheppard et al., in this publication). Hence, the gradual withdrawal of commercial fishing effort in recent decades, especially from coastal and estuarine areas, has in some cases resulted in less data available for stock assessments (see Smith in this publication). To overcome this, focussed initiatives have been implemented across Australia (see Smith; Sheppard et al., in this publication), but it remains debatable whether enough is being achieved.

In his presentation regarding the monitoring needs of recreational fishers, Sawynok (in the publication) calls for a greater focus on social monitoring. He suggests this will lead to improved support for fisheries researchers and management change. He advises that knowing about recreational fishers' attitudes, practices, behavior and values can go a long way to implementing change in a less confrontational way. The scarcity of this type of data was highlighted at the workshop, with few examples being presented where social aspects were incorporated into current monitoring programs.

During the presentation on the monitoring needs of commercial fishers, Loveday (in this publication) also warns that the commercial fishing sector's substantial social and economic contribution is being undermined by knowledge gaps. He warns that management decisions which ignore the social and economic aspects of commercial fisheries risk being increasingly overridden, as industry is forced to seek relief through the political process. The views of Loveday and Sawynok warrant further consideration when designing or reviewing monitoring programs.

Morrison (in this publication) provided an insight into the monitoring needs of indigenous fishers. He explained that indigenous people differ in their views, the types of resources they exploit and the intensity with which they exploit them. Morrison (in this publication) claimed that these differences complicate interactions with others seeking to use the same resources. He also warns that due to the rapidly increasing indigenous population, its aspirations cannot be taken for granted. The views of Morrison also warrant further consideration.

While there are several current or recent examples of monitoring programs focusing on indigenous fisheries in Australia, the 2005 ASFB Workshop highlighted that the number of projects is limited. In many areas, subsistence indigenous fishers are likely to account for a significant component of the total harvest of a stock, and hence data on this sector is required to provide the basis of informed decision-making by resource custodians. There is a pressing need to correct the dearth of information available (to non-indigenous people at least) on indigenous fisheries.

Lyle (Session summary) appropriately cautioned that implementing monitoring programs in indigenous fishers requires recognition that typical recreational survey approaches are not appropriate. The workshop was presented with several options for conducting monitoring programs in indigenous fisheries. These ranged from large-scale programs (see Coleman in this publication) to community-scale programs (see Phelan in this publication). These programs, together with those presented by Busilacchi and Begg (in this publication) and Sheppard et al. (in this publication), provide a baseline for monitoring indigenous subsistence fisheries in Australia.

Monitoring ecosystems

The emphasis of most presentations during the ‘ecosystem monitoring session’ was on monitoring ecological sustainability of species or species assemblages, as a component of ecosystem monitoring, rather than attempting to monitor the system as a whole (Gribble, Session summary). The workshop highlighted that the concept of ecosystem monitoring is either too complex for conventional monitoring approaches or we still have difficulty in handling such complexity (Gribble, Session summary).

Lenanton (in this publication) suggested that cost precludes sampling all species and recommended that indicator species or species at risk should be identified for further attention. He explained that this must involve identifying species inherently venerable (biology) and vulnerable to the fishing or other impacts. He adds that the Department of Fisheries in Western Australia is currently prioritizing effort to determine sustainable harvest levels for key “indicator” species within each bioregion of the State. Lenanton (in this publication) indicated that at times, such determinations may need to be made in the absence of adequate data.

Griffiths et al. (in this publication) added that some fisheries may interact with hundreds of species with varying life strategies, many of which are rarely caught, are of low value and data-poor. They conclude that monitoring entire diverse communities would therefore be expensive, impractical or impossible. Griffiths et al. suggested that ecological risk assessment may be the only way to control the cost-effectiveness of monitoring programs without sacrificing the coverage of the most critical species. However, they warned that current approaches to risk assessments, such as productivity-sustainability analysis, may be inappropriate for the task.

Brewer et al. (in this publication), Griffiths et al. (in this publication), Zhou et al. (in this publication), and Heales et al. (in this publication) presented details of an alternative quantitative risk assessment that may be employed to prioritise species most likely to be at risk. Zhou et al. (in this publication) explained that this approach uses presence and absence data from scientific surveys to estimate the probability of detecting a species in a particular grid, and the probability that that species was present. The process involves monitoring the fishing mortality rate of the species. They advocate that this approach may be easily transferable to other fisheries due to its simplicity and requirement of only presence and absence data.

Clearly, understanding natural variation in species abundance will remain critical to establishing the extent of anthropogenic impacts. The workshop heard several examples where monitoring data is revealing information about the environmental drivers of species abundance. For example:

- Gribble et al., (in this publication) found that the abundance of a number of commercially fished species in northern Australia can be explained by climatic changes.

- Lintermans (in this publication) found Macquarie perch (*Macquaria australasica*) populations in southern Australia are subject to significant impacts from fire, sedimentation and river regulation.

- Douglas (in this publication) revealed that habitat stratification within lakes impacts on the size and location of brown trout (*Salmo trutta)* habitat.

Neil (in this publication) suggests that with the adoption of an ecosystem management approach there is a concomitant need to adopt an integrated approach to fisheries monitoring. He warned that monitoring programs in Australian have often been disjointed and have rarely considered or integrated data collected through other programs. He recommended that monitoring program managers integrate and rationalise data from existing monitoring programs before planning new surveys.

Ward and Goldsworthy (in this publication) identified the South Australian sardine fishery as an example of where a multidisciplinary program has been established to underpin the development of an ecosystem-based approach. They indicated that factors that have been critical to ensure the success of the program included a strong financial investment by governments and industry; seamless integration of monitoring and research programs, and strong collaboration with research agencies/scientists with complimentary skills or infrastructure.

Designing a monitoring program

All monitoring programs differ in design as they are all tailored for unique opportunities, such as avenues to gain and present information, and limitations, such as finite time and resources. A well-designed monitoring program may be defined as one which balances these opportunities and limitations. Due to the diversity of monitoring programs, there cannot be one standard approach to implementing a ‘well-designed monitoring program’ (Lyle, Session summary).

To implement an effective monitoring program, you will need to start by asking: “What is the problem or potential problem?” and “What do we need to know?” (Ohrel and Kathleen, 2001). No step is more critical in the planning process than establishing the objective or objectives of your monitoring program (Ohrel and Kathleen, 2001). Every phase that follows will depend on this initial decision. Hall (Session summary) stipulated that it is fundamental that the objectives of the monitoring program are clearly articulated before steps are taken to decide which monitoring strategy is most appropriate

Once a clear understanding of the objective has been established, it is possible to determine which monitoring method will be most appropriate (see Begg et al., in this publication). This, of course, depends on the question being asked, the resources available, and the qualifications of the persons who will do the work (Crosby and Reese, 1996). To determine which monitoring method will be most appropriate, you will need to consider questions such as:

- Why is the monitoring taking place?
- When and where are we going to monitor?
- What equipment do we have access to?
- Who will conduct the monitoring?
- Who will analyse the data?
- How will the data be used?

When designing a monitoring program, it is essential that the efficiency of the proposal is adequately tested (see Chick in this publication). To achieve this task, Rotherham et al., (see this publication) recommended conducting pilot studies incorporating manipulative experimental approaches to:

- Identify sampling gear suitable for the target species (see Lintermans et al., in this publication).
- Understand the spatial and temporal scales of variability across different strata (see Nicol et al., in this publication).
- Conduct cost-benefit analysis to determine optimal level of replication (see Ryan et al., in this publication).

Planners also need to be mindful of introducing errors that may bias results. A relevant example of this is presented by de Lestang et al (in this publication). They explain that in the case of fishery-dependent monitoring programs, advances in technology (e.g., colour sounders and GPS plotters) increase fishing efficiency, which is not incorporated into a measure derived solely from effort data. They warn biased data such as this can lead to overly optimistic estimates of stock abundance. Tucker (2004) listed key sources of error and described steps that can be taken to reduce the chances of collecting inaccurate data.

It is hard not? to overemphasize the importance of effectively analysing monitoring data and presenting results. Cresswell (in this publication) stated that what is required is for all sectors and jurisdictions (including government/industry partnerships) to better work together in data collection, analysis and refinement of data needs. He proposes that the ultimate goal of researchers and managers must be to provide a meaningful understanding of what is happening in our waters.

Managing the expanding sets of data, and catering for the growing applications and expectation of such information, is another important issue. Scandol and Ives (in this publication) introduced an effective method to tackle this challenge using a new electronic reporting system called the resource assessment system. This intranet-based system is an example of how contemporary web-based technologies can assist in the organization, presentation and maintenance of the information and data associated with stock monitoring and assessment. They suggest that the largest hurdle that most institutions will face in the implementation of such systems is the development of required data warehouses.

Hall (Session summary) strongly advocates that regular reviews of current monitoring strategies are essential. Helmke et al. (in this publication) also recommended that monitoring programs must also have the capacity to evolve over time in response to changes in legislation, management regimes and technological advances. Helmke et al (in this publication) and Staunton-Smith et al. (in this publication) presented the fisheries long term monitoring program of Queensland’s Department of Primary Industries and Fisheries as an example of a program that is able to address evolving issues.

Exploring all options for monitoring

The workshop revealed numerous examples of innovation in monitoring (Gribble, Session summary). Examples include:

- Estimating the detection probability for boat electro-fishing by analysing the number of fish caught from a population of radio-telemetry tagged fish (see Nicol et al., in this publication).
- The use of ‘trapping web’ depletion studies for monitoring the abundance of mud crabs (*Scylla serrata*) on tidal flats (see Hay et al., in this publication).
- The use of video footage to monitor interactions between trawl nets and dolphins (see Stephenson, in this publication).

Some of this innovative science is providing new, cost-effective options for monitoring. Dunlop and Maxwell (in this publication) provided a sound example. They utilise nitrogen and carbon stable isotope ratios in seabirds to monitor predator-prey relationships in baitfish fisheries. Gribble (Session summary) stated that isotope analysis of discarded feathers and tissues to track diet change appears to be an unobtrusive and relatively low-cost method of monitoring a trophic pathway. Gribble (Session summary) adds that this task is a necessary part of ecosystem monitoring.

Some innovative methods not only offer potential advantages in terms of efficiency, but also potentially increase the safety of staff; an issue which must be at the forefront of planning or conducting any monitoring program. A good example of this is provided by Buckle et al., (in this publication). To conduct ‘underwater visual surveys’, they installed a perspex bubble in the bow of a small dinghy. This allows effective monitoring of fish communities in waters potentially inhabited by sharks, crocodiles and other predators.

In another example, ‘gene-tagging’, the benefits apply to both the researchers and the fish (see Buckworth et al., in this publication). The basic concept of gene-tagging is to take a small tissue sample from a fish using a specially-developed hook/lure with a flexible shaft, and to use the DNA profile of the tissue sample as a ‘tag’ for that fish. To increase the chances of gaining recaptures, fin samples may be collected from fishers or processors (see Buckworth et al., in this publication). Gribble suggested gene-tagging was possibly the most innovative of the fishery ?dependent monitoring techniques described during the workshop.

The cost-efficiency and outputs of monitoring programs may be increased by involving fishers in the planning, implementation and review phases. Sawynok (in this publication) encourages an increase in the involvement of recreational fishers in monitoring programs. He suggests that if a monitoring program may ultimately lead to change for recreational fishers, they should be involved from the outset in the planning, data collection and distribution of information. He suggests that if this is done properly, those involved will become the greatest supporters of the outcome of the work, and probably the greatest advocates for change.

Likewise, Carne (in this publication) recommends involving indigenous people in monitoring programs. He suggests enlisting the assistance of indigenous people involved in such programs as the Northern Territory’s Indigenous Community Marine Ranger Program. He suggests partnerships of this nature would provide mutual benefits. The involvement of indigenous people in monitoring programs has been demonstrated to build a high level of understanding within the community, and may even result in community-driven management outcomes (see Phelan in this publication).

Smith (in this publication) provides an example of a program based on the involvement of fishers. The Research Angler Program of Western Australia’s Department of Fisheries focuses on the collection of scientific data, yet provides the added benefit of community education. The volunteer-based program conducts projects such as angler logbooks, biological sampling, tagging studies, and collection of fishing club and competition data. Smith says the advantages of the program include a single, continuous point of contact for volunteers, cost-effective volunteer administration and higher quality feedback to anglers.

Conclusion

The 2005 ASFB workshop provided a forum for the review of a diverse range of current and recent monitoring programs. Taking the time to stop and consider the direction of our monitoring efforts has untold benefits. A key take-home message that was continuously identified during the workshop was the importance of completing regular reviews of monitoring programs to ensure they remained relevant to the information required and that they continued to provide the best information with the resources available.

As a testament to the widespread desire to adopt optimal monitoring designs, a large proportion of the presentations during the workshop focused on efforts to test and improve the efficiency of sampling methods. The 2005 ASFB workshop also highlighted the extent of novel approaches that are emerging in Australia, providing further impetus for the expansion and refinement of monitoring programs. Undoubtedly, many of these novel approaches will become widely employed in the years to come.

Despite the breadth of monitoring projects presented at the 2005 ASFB workshop, there were areas where the collection of data was scarce. Most notably, the workshop highlighted the limited scope of projects collecting data on indigenous fisheries, and the scarcity of programs collecting social and economic data. There is an urgent need to rectify these deficits so that fishery and ecosystem management decisions are well founded. Ignoring these shortcomings may limit stakeholder acceptance of resource management changes.

Essential concepts of effective monitoring

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About monitoring

'Monitoring' is a term that is widely used in the management of natural resources. Published definitions include to 'observe, supervise, keep under review; measure or test at intervals, especially for the purpose of regulation or control' (ISO 9001:2000) and '… to establish a system of continued observation, measurement and evaluation for defined purposes' (Barrow, 1999). These definitions are useful as they capture the generic application of the concept across disciplines. Perhaps the most valuable concept associated with monitoring is the linkage between the state of a system with the human institutions responsible for the stewardship of that system. This linkage applies in examples as diverse as a physician monitoring the state of a patient's heart, or a fisheries management agency monitoring the abundance of a particular species.

Extending the concept of monitoring to include the institutional response associated with the information collected, forces consideration of the broader context and values associated with monitoring. A monitoring program with technically excellent data collection protocols, that provides the results to an agency that does not or cannot use the information, is just as compromised as sub-standard data collection coupled with a highly competent management agency.

When we consider the range and type of approaches to monitoring that are available for aquatic resource agencies, there must be an ongoing recognition that the mechanisms to interpret and act upon that information play a critical role. Part one of this introduction will: (i) briefly reflect upon monitoring in environmental law and policy; (ii) discuss the sometimes uncomfortable relationship monitoring has with science; and, (iii) consider the collection and utilisation of monitoring information in a socioeconomic context. Part Two will provide some guidelines for monitoring fisheries and aquatic ecosystems in Australia.

The concept of monitoring is embedded within international conventions as well as national and state environmental laws and policies. For example, the United Nations Fish Stocks Agreement, Article 5 (General Provisions) calls for states to 'implement and enforce conservation and management measures through effective monitoring, control and surveillance'. The Australian *Environmental Protection and Biodiversity Conservation Act* (1999), Part 12 includes provisions for identifying and monitoring biodiversity. Under the information requirements for Principle 1 of the Guidelines for the ecologically sustainable management of fisheries[1], it is required that 'There is a reliable information collection system in place appropriate to the scale of the fishery. The level of data collection should be based upon an appropriate mix of fishery independent and dependent research and monitoring'. Within the NSW *Fisheries Management Act* (1994), there are explicit provisions (s7E(e)) to 'include performance indicators to monitor whether the objectives of the [management] strategy … are being met'. The deficiencies of most environmental assessment legislation with respect to ongoing commitments to monitoring have been discussed by Thomas (2001).

Australian scientists and managers working in aquatic resource management have the responsibility of the next stage of monitoring. This involves designing, negotiating, implementing, interpreting and then acting upon the outcomes of practical monitoring programs. This is not a trivial task, as complex statistical and logistical issues are interwoven with important socio-economic considerations.

Science can have a somewhat uncomfortable relationship with monitoring because the hypotheses to be tested are not often explicitly stated (or, in some cases, completely disregarded). In most cases the null hypothesis for monitoring is that an indicator has not changed, though in the case of restoration or recovery it may be that an indicator is not significantly different from a control or reference state. Manly (2001) noted that monitoring programs to detect unexpected changes or trends are essentially repeated surveys. A range of statistical tools are available to interpret data collected from monitoring including generalised linear models, analysis of variance (or, more generally, deviance); control charts, methods to determine a change in the distribution of indicators; time series analysis; methods to detect change points and trends; and methods of spatial data analysis. Monitoring studies have also introduced concepts such as 'bioequivalence' and BACI (Before-After-Control-Impact) survey designs (also see Underwood (1990), Manly (2001), Quinn (2002)).

All of these approaches have the fundamental statistical trade-offs present including that between type I and type II errors, power and sample-size, power and effect-size and power and the variability of observations (Underwood 2000). Issues associated with type I and type II errors are so fundamental to the design of monitoring programs that they should underlie the wording of any null and alternative hypotheses that are being tested. For example, the United States Environmental Protection Agency (1989, in (Manly 2001)) recommends that hypotheses are worded so that, in essence, the environment 'gets the benefit of the doubt' associated with inadequate sample sizes and low statistical power. Although these issues of 'burden of proof' are linked with the precautionary principle and approach (Harding and Fisher 1999), it is often logistical constraints (such as costs) that drive monitoring designs, rather than abstract principles of sustainability. Alternative

[1] www.deh.gov.au/coasts/fisheries/guidelines.html

frameworks of statistical inference that are focussed upon parameter estimation rather than hypothesis testing do not eliminate tradeoffs in statistical inference; these tradeoffs are simply re-expressed in an alternative manner.

Monitoring associated with wild fisheries is sometimes opportunistic. Statistical concepts of replication, randomisation and control are difficult, expensive and sometimes logistically impossible to implement in actual fisheries. The notion of a ‘before state’ of an unfished population, where total mortality is equivalent to natural mortality, is a particularly rare phenomenon. To be able to find, or create, a ‘control site’ where human impacts are not present (or have been removed) can be particularly difficult. Stakeholder resistance to having their access compromised will likely out-weigh any purported benefits that might result from the study.

As a result of the constraints associated with monitoring fisheries, many datasets are particularly low in contrast (i.e. the information content of the data is such that any parameters of interest cannot be estimated precisely or accurately). This is a well-understood issue in fisheries science (Hilborn and Walters 1992) and can only be ameliorated by taking observations over a greater range of system states. There is thus logic in using ‘natural experiments’ to replace designed experiments if the latter are logistically impossible. Natural experiments may generate contrasting information and should provide superior evidence for scientific inference. This result cannot, however, be taken for granted. Often such situations are compromised by inadequate replication (i.e. often non-existent replication or pseudo-replication) or other uncontrolled confounding factors. Such natural experiments might include monitoring estuaries subject to different management regimes, monitoring systems recovering from natural events involving high natural mortality such as ‘fish kills’, observing systems before and after pulses of high fishing pressure (such as fishing tournaments), and monitoring changes associated with the implementation of aquatic protected areas. Fisheries scientists become pragmatic about the constraints they face when doing field research. Professional experience and judgement is usually required to classify an opportunity to monitor a system as one that is likely to yield useful inferences, as opposed to one that will probably be an expensive waste of time and effort.

Values of monitoring

Spellerburg (1991) noted that the values of monitoring could be derived from objectives such as ‘… the basis for managing biological resources for sustainable development and resource assessment’, and ‘…so that ecosystems and populations can be managed and conserved effectively.’. These suggestions are astute but perhaps a more parsimonious justification for monitoring is simply ‘accountability’.

The fundamental argument for the importance of monitoring is the accountability of natural resource management. Decision-making in any natural resource management agency is a complex mixture of public and private interests, many of which are neither particularly easy to understand nor quantify. Fisheries management agencies are constrained by political and legal boundaries as well as various layers of bureaucracy. Stakeholder-based advisory committees, that reflect composite sets of values, usually play a key role in consultation and decision-making. Yet, somewhere amongst this heady mixture of human-values and decision-making procedures, there exist environmental laws, government policy and the actual fish! Without some recourse to observations of the fish and their habitat it would be very easy for decision-makers to lose direction within this maze of value-laden issues. After an articulation of the objectives of aquatic resource management (usually expressed within legislative and policy frameworks) there must be some mechanism, which includes credible observations of the aquatic system, to see if aquatic management is actually achieving what it purports to achieve. This is why monitoring is so important.

The necessity of integrating the results of credible research into decision-making is a fundamental part of medicine and health. The nomenclature used in medicine and health is to apply an 'evidence based' approach. A search of articles in the *British Medical Journal* resulted in almost 3000 papers that included the phrase 'evidence based' in their title or abstract. Identical searches in the Aquatic Sciences and Fisheries Abstracts yielded only a handful of papers associated with aquatic resource management. This illustrates that, in contrast to natural resource management, health professionals dedicate a lot of effort into researching, understanding and then improving the application of evidence in decision-making.

Aquatic resource management should similarly be committed to evidence. This does not dismiss the essential importance of the socio-economic dimensions of environmental decision-making, but simply attempts to isolate the influences of objective and subjective information. Public and private interests associated with aquatic resources are complex (and will often overlap), but personal opinion should be able to be differentiated from objective observations of an aquatic system when it comes to the implementation of public policy.

There is an adage in control theory that 'you cannot control what you cannot measure', which is often re-used in management circles as 'you cannot manage what you do not measure'. There is, however, a corollary to this statement that could be stated as 'if you do not want something managed, ensure that it is not measured'. This re-interpretation introduces another dimension to monitoring: the constituency of error (Hammond 1996); i.e. stakeholders that benefit from type I errors are usually different from those who benefit from type II errors. Which leads us to another question: how should the costs of monitoring be allocated?

Costs of monitoring

When discussing some of the statistical issues associated with monitoring above, it was noted that the power of a statistical test to determine a significant change in a system was positively related to the number of observations taken. In general, observations take resources, therefore the more resources allocated, the more power to detect change. One strategy to reduce the ability to detect a change would be simply to allocate fewer resources to the monitoring study. As noted above, changes in the wording of the null hypothesis, or re-interpretation of the criteria for bioequivalence, is all that it takes to reverse this burden of proof, but such changes are only associated with an actual commitment to monitoring. The easier way to avoid such issues is simply to not provide any resources to monitor at all. Most natural resource scientists and managers have seen the withdrawal (and, in other cases, provision) of funding as the most effective design criteria for a monitoring program of them all.

There is, however, another twist to the socio-economics of monitoring and it is deeply embedded in the principles of ecological sustainability: the internalisation of environmental costs (or polluter pays). Using resources from the public purse to pay for monitoring may actually end up exacerbating environmental issues, not resolving them. If the general public pays for the monitoring of an activity that harvests an aquatic resource, then this can be interpreted as a subsidy to that activity (see WWF (2001) for a general overview of subsidies in fisheries). This concept is well understood in environmental economics and is often referred to as ‘privatising the profits and commonising the costs’ (Hardin 1985). Public resources expended on such subsidies may be better allocated to compensating stakeholders for changes to access rights or the structural adjustment of industry. Such arguments are economically straightforward for commercial fisheries, but are much more muddied for the recreational use of fish stocks and the non-use values of conservation-oriented stakeholders. In the latter cases, the benefits of monitoring are usually reasoned to be in the public, not private, interest.

Given the relative environmental impacts of resource-use (than non-use) it could be argued that commercial and recreational fisheries should contribute to the monitoring costs of their own activities. When dealing with large and profitable commercial fisheries or recreational fisheries with many hundreds of thousands of participants, the costs of monitoring would be a small and insignificant impost on the fishery. But when dealing with small and/or economically distressed commercial fisheries or recreational fisheries with few participants, the costs of large monitoring programs are likely to be prohibitive. In such circumstances, an effective compromise is for the fishery to supply in-kind rather than cash support. That is, participants in the fishery assist in the collection of data. Most fisheries in Australia are managed on the basis of monitoring programs that rely upon the goodwill and cooperation of industry (such as catch and effort logbooks). This is unlikely to change in the foreseeable future, simply because of the inability of most of the smaller fisheries in Australia to absorb the costs of independent monitoring programs.

The role of stakeholders in environmental monitoring programs was discussed by Harding (1998), where the values of such data, in contrast to data provided by ‘experts’ was noted. This was particularly relevant when the information was of a local character and was considered more relevant to an issue. When stakeholders within natural resource management hold the dual roles of advocate and data collector there is always the potential for a conflict of interest. The previous section noted that ‘personal opinion should be differentiated from objective observations of an aquatic system when it comes to public policy’. This statement must now be revisited in a more pragmatic light. In most circumstances in Australia, socio-economic realities will force the involvement of stakeholders in the monitoring of fisheries, with the alternative being that no or very little monitoring would occur at all. This involvement may be as simple as a recreational angler agreeing to be interviewed during a creel survey, or a commercial fisher diligently completing a logbook or permitting an observer to be on board. That involvement will, however, inevitably occur.

Given the trade-offs and compromises required for monitoring aquatic systems, public agencies have a responsibility to allocate their limited resources for such tasks with careful consideration. Programs that, by their statistical nature or socio-economic context, are unlikely to yield useable results should be identified and then discouraged. In contrast, monitoring programs that should be encouraged are those that have stakeholder support, are statistically robust and are directly linked to indicators that improve the accountability of science and management.

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Monitoring fish stocks and aquatic ecosystems: Guidelines for consideration

The Australian Society for Fish Biology Monitoring Committee

Background

This committee was convened after the 2005 ASFB workshop. The workshop brought together national experts in aquatic resource science and management and examined monitoring issues associated with commercial, recreational and indigenous fisheries. The workshop also included sessions on fisheries-independent monitoring and ecosystem-based approaches to monitoring aquatic resources.

The ecologically sustainable development of Australia's aquatic resources requires feedback mechanisms from the underlying ecological system to the management systems that make decisions. These decisions will, directly or indirectly, affect the human uses of, and values associated with, our aquatic resources. The term 'monitoring' now appears to be the preferred terminology to describe that feedback mechanism. This implies that monitoring not only includes the collection and interpretation of information about the ecological system, but monitoring also helps define, maintain and account for the systems of management that are expected respond to that information. Monitoring is a critical component of fisheries management science and there need to be improvements in the understanding and implementation of monitoring in Australia.

The ASFB Monitoring Committee has representation from all Australian jurisdictions and has identified the following guidelines for consideration when monitoring fish stocks and aquatic ecosystems. The Committee encourages all members of the Society to bear these guidelines in mind when designing, implementing, reviewing and interpreting monitoring programs:

1. Promote the role of monitoring within the ecologically sustainable development of Australia's aquatic resources. Specifically, promote the monitoring of target species, by-catch species and the other species and communities impacted by human activities.
2. Respect the privacy of all individuals providing data to monitoring projects and programs. Recognise and value the human dimensions of monitoring and, in particular, appreciate the unique role of indigenous people in Australian fisheries.

3. Encourage and recommend the application of best scientific practices to monitoring projects or programs, including the explicit testing of hypotheses with due consideration for type I and II errors, as well as the size of the change to be measured. Consideration should also be given to the precision and bias of estimated parameters.

4. Promote the importance of datasets containing high-contrast information. Monitoring programs or projects should therefore attempt to span changes to management (in space or time) and natural events (such as fish kills). This may require the collection of ‘baseline’ information and the development of logistical systems that can be rapidly implemented ‘after an event’. There should also be recognition that long-term datasets are fundamentally important because these are the datasets most likely to contain important temporal contrasts.

5. Critique the relationships between a monitoring project/program and the management agency. Are appropriate decision-making processes in place to use the information collected to achieve the management objectives for the resource?

6. Promote understanding of the relationship between the management risks, costs and benefits of the various options to manage and monitor aquatic resources.

7. Consider the costs and benefits of stakeholder-based monitoring, such as industry-based data collection and community-based monitoring by recreational-fishers and non-extractive users.

8. Encourage the provision of feedback and extension to stakeholders who provide information for monitoring programs.

9. Identify situations where a monitoring project/program is inadequate with respect to identified objectives. For various social, economic and ecological reasons there will be scenarios when monitoring is not the best option, and alternative management strategies may be more appropriate.

10. Identify situations where, for some stakeholder groups, there will be a disincentive for meaningful monitoring. Develop arguments for, and assist in the implementation of, incentive systems that value monitoring from the perspective of all stakeholder groups.

11. Critically examine the logistical methods used for monitoring. In particular consider gear selectivity issues and any associated biases.

12. Promote occupational health and safety issues associated with monitoring.

13. Explore and promote the application of new technology for monitoring, particularly when it will result in more cost-effective solutions. In particular the roles of acoustic technology, satellite imagery, and genetic methods should be objectively evaluated.

14. Encourage the development of new reporting systems to facilitate communication to management and stakeholder groups. These systems should be timely and efficient but not at the expense of an unacceptable rate of errors. Electronic reporting systems using intranets and the internet should be studied.

15. Encourage the electronic storage of monitoring datasets in robust relational database management systems. These systems should be described with meta-data statements consistent with national and international standards. The database systems should capture all relevant aspects of the source data. Metadata for monitoring projects/programs should be searchable within and between agencies.

16. Facilitate data exchange within and between institutions. This will include the usage of standard codes (such as the Codes for Australian Aquatic Biota or CAAB), lookup tables mapped to standard codes and use of easily interpretable fields and units (such as dates and latitude/longitude data). Appropriate linkages to online sources such as FishBase, CephBase and the Ocean Bio-geographic Information System (OBIS) should be developed.

17. Develop and review monitoring programs with input from fisheries managers, data analysts, on-ground staff, collaborating agencies and other stakeholders.

18. Provide sound documentation on monitoring protocols so that monitoring programs are repeatable, and could be adapted for other projects or programs.

The members of the ASFB Monitoring Committee for 2005-06 and 2006-07 are:

James Scandol (New South Wales – Convenor)
Rik Buckworth (Northern Territory – Co-convenor)
David Brewer (Commonwealth – North)
Kim Smith (Western Australia)
Matt Koopman (Victoria & Commonwealth – South)
Philippe Ziegler (Tasmania)
Sue Helmke (Queensland)
Tim Ward (South Australia).

History of monitoring, research and management of common snook in Florida, USA

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Abstract

Snook occur in Florida south of the 14º C winter water isotherm and sustain robust populations along both the southern Atlantic and gulf coasts. Prior to 1950, common snook supported recreational and heavily exploited commercial net fisheries that contributed to the precipitous decline of both populations. In 1957, snook were declared ‘game-fish’, their sale was prohibited, and bag and size limits were established. A hiatus in snook research occurred during the 1960s; however, anglers’ concern about snook populations precipitated an intensive tagging program during 1977-1987 along south-western Florida. This study revealed a declining, unstable population, which prompted a reduction of the bag limit, an increase of the legal minimum size, and harvest closure during the spawning season. Life-history studies conducted in Florida during 1986-2001 showed that common snook are protandric hermaphrodites and that major biological parameters of each coastal ‘stock’ differed significantly. In 1992, the Florida Marine Fisheries Commission (MFC) established a 40% spawning potential ratio (SPR) as a management goal for the entire population. Genetic studies in the mid 1990s validated the existence of separate Atlantic and gulf stocks, whereupon the MFC promulgated separate coastal regulations, with the gulf’s stock rules being more stringent because of higher exploitation. Since 1994, scientists have conducted nine stock assessments of snook and determined that despite systematic changes that curtailed harvest both stocks have consistently remained below the target SPR because of overfishing. About 60% of Florida’s daily influx of 1200 new residents settles along the coast, leading to increasing fishing effort and inexorable declines of coastal wetlands and mangrove forests. Thus, scientists and managers may be constrained to a permanent restrictive management posture.

Introduction

Fossil evidence suggests that the family of snooks, Centropomidae, arose in the Old World during the Eocene about 50 million years ago (MYA) and that the genus *Centropomus* and its most distantly related species (i.e., *C. undecimalis* and *C. armatus*) appeared in the New World about 30 MYA. The youngest species of *Centropomus*, the geminate species *C. viridis* and *C. medius*, speciated about 3 MYA when the Central American isthmus emerged (Greenwood 1976, Tringali et al. 1999). Four species of snook (fat snook, *C. parallelus*; tarpon snook, *C. pectinatus*; sword spine snook, *C. ensiferus*; and common snook, *C. undecimalis*; occur in Florida. Only common snook grow to legal harvestable size, and regulations developed to protect common snook are applied to all species of the genus because of the difficulty of species identification; consequently, the three diminutive species are totally protected from harvest. Snook occur along both coasts of Florida south of the winter 14º C water isotherm (Rivas 1986) which, approximates the terrestrial frost line; however, the adults migrate into more northern climes during periods of warm winters.

Because snook are an icon species of game-fish in Florida, the obligation to manage and conserve snook populations and to investigate and understand their biology remains a high priority for the Florida Fish and Wildlife Conservation Commission.

Main Text: Common snook have been heavily exploited in Florida since the 1920s because of their fighting nature and gastronomic value. Prior to 1947 common snook were exploited without regulation by both recreational hook-and-liners and commercial netters. Often local netters, using large-meshed seines, would land upwards of 10 000 pounds per trip when they targeted large aggregations of spawning fish. This unregulated harvest, combined with widespread alteration of coastal habitat, contributed to a precipitous decline of the population during the late 1940s and 1950s. During the 1950s, public outcry against this perceived disregard for the resource was brought to the attention of the fledgling Florida Board of Conservation, which consequently outlawed ‘snook haul seines’ in extreme southwest Florida. However, continued exploitation by commercial fishers using nets coupled with unrestricted harvest by recreational anglers precipitated milestone regulations enacted in 1957 that declared snook to be game-fish, banned their sale or barter, and reduced the daily bag limit to four snook larger than 18 in. fork length.

Total landings of snook in Florida varied from less than 100 000 lbs. in 1957 and 1982 to a record harvest of about 1 500 000 lbs. in 1997. Although the average annual harvest over the entire reporting history was about 500 000 lbs., the mean annual landings when commercial harvest was allowed were about 90 000 lbs. less than during the later period, which was composed solely of recreational landings (Figure 1.).

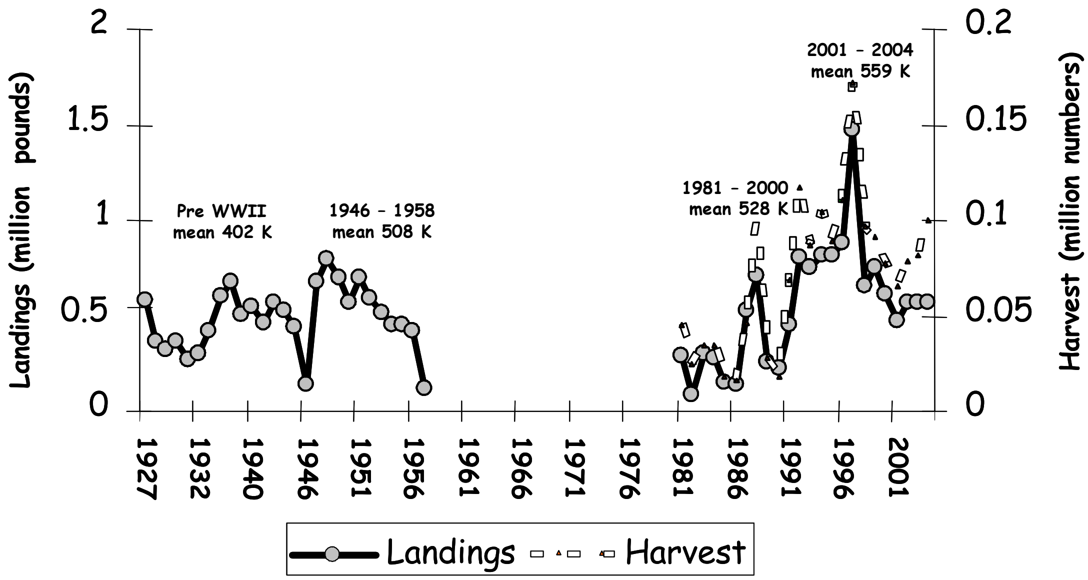

Figure 1. Historical landings of common snook in Florida Landings are in pounds, harvest is number of individuals. No data were collected during 1957- 1981.

The landmark legislation of 1957 renewed interest in snook and evoked fundamental biological research that resulted in two seminal studies that described the life history, distribution, movement, and basic ecology of snook (Marshall 1958, Volpe 1959).

Between 1959 and 1975 there was a hiatus in biological research and active fisheries management of common snook. In the mid 1970s, recreational anglers asked the Natural Resources Committee of the Florida Legislature to ascertain the condition of the snook population along the southwest Florida coast, the traditional centre of snook angling and exploitation. Accordingly, an extensive mark-and-recapture study designed to estimate fishing mortality and survival based on angler-reported recaptures and to estimate population abundance employing the Schnabel multiple-census technique was conducted along the Naples-Marco Island area during 1977-1987. Preliminary findings revealed that the snook population in the vicinity of Naples sustained high levels of fishing mortality and that annual survival rates were less than 50%. Concomitantly, the population was shown to be unstable and that it had declined about 70% between 1977 and 1981 (Fig. 2). Biologists feared that severe overfishing on a reduced population could result in recruitment failure.

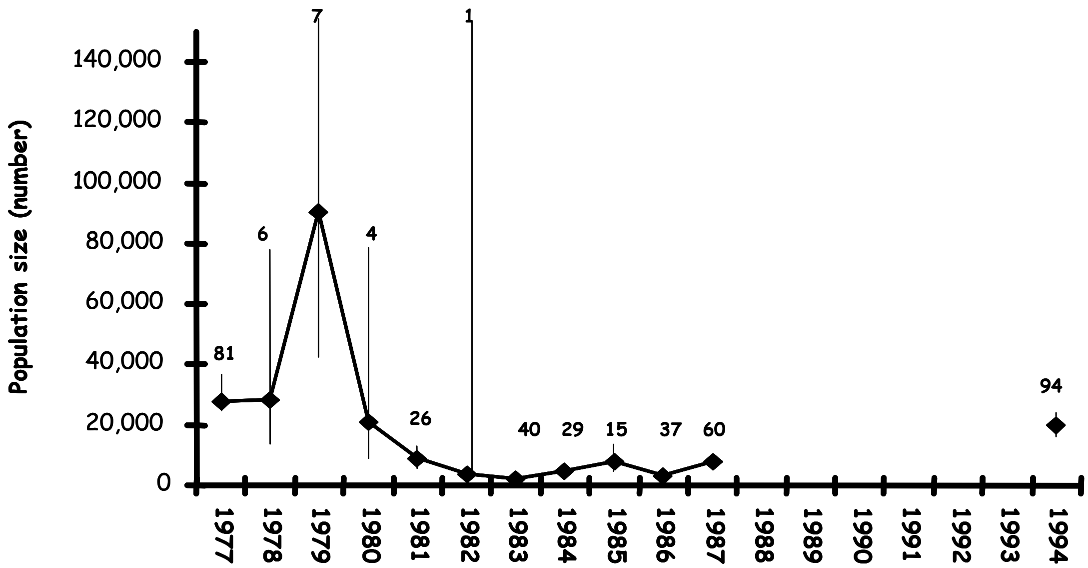

Figure 2. Schnabel estimates of population abundance for common snook in the Naples- Marco Island area of southwest Florida

Vertical bars are 95 % confidence limits. Numbers in parentheses are numbers of snook recaptured experimentally. (From Bruger 1987, unpublished)

Bruger's (1987, unpublished) tagging study provided scientific evidence of the condition of the stocks of snook in Florida in the early 1980s and served as the basis and catalyst for passage of additional snook fishery regulations in 1981, 1982, 1983, 1985, 1987, and 1989. These regulations reduced the bag limit to two snook/day; increased the minimum legal length to 26" total length (TL) with one snook > 34" TL allowed; closed January, February, June, July, and August to harvest; included all species of snook under these rules; and required a $2.00 harvest stamp for licensed anglers fishing in a boat (Table 1).

This tagging study was a poignant admonition of the condition of common snook stocks near Naples and possibly across their entire range in south Florida. Not only did this study identify overfishing as a cause for reduced population abundance, but it also implicated the deleterious loss of aquatic nursery wetlands and the insidious alteration of water quality and quantity by changing its natural sheet flow south of Lake Okeechobee and through the Everglades. The study documented that even though the sport fishery had been curtailed by regulations and the commercial harvest had been totally banned, snook populations continued to decline. Bruger recommended further limiting fishing pressure and also suggested that because of the stringent requirements for early life-stages of common snook, realizing an increase in numbers of snook to some historical abundance would require a massive habitat-restoration program across much of south Florida.

Because fish populations and the quality of the south Florida wetlands continue to be reduced 30 years later, the Federal and State governments have embarked on a massive multi billion dollar campaign aimed at restoring the vast aquatic habitat of south Florida.

Table 1. Chronology of regulations promulgated for the management of the common snook fishery in Florida during 1947- 2002

1947- Snook haul seines made illegal in Lee County.
1951 - Snook haul seines made illegal in Collier County.
1953 - Minimum size set at 18″ FL.
1957 - Snook made illegal to buy or sell; Bag limit set at four snook > 18″ FL.
1981 - Bag limit reduced to two snook/ day. No snook < 26″ FL may be taken in June or July during 1982-1986.
1982 - June and July of 1982 closed to snook possession.
- Snook designated 'species of special concern.'
1983 - January and February 1983-1986 closed to snook possession.
- June and July 1983-1986 closed to snook possession.
- Marine Fisheries Commission established.
1985 - January, February, June, and July closed permanently.
- August 1985-1986 closed.
- Minimum size increased to 24″ TL.
- Only one snook may be >34″TL
1987 - All species of Centropomus covered by the regulations.
- August is closed permanently.
- Use of treble hooks prohibited with natural baits.
1989- A $2.00 snook stamp required of boaters to retain legal snook.
1994 - Winter closed during 15 December -January 31.
- SPR goal set at 40%.
1997 – Population separated into Atlantic and Gulf stocks.
1999 – Harvest slot set at 26″ to 34 inches″ TL.
2001 - Snook removed from list of 'species of special concern'
2002 - Gulf stock: closed during May; daily bag reduced to one snook.

During the 1980s, common snook became the subject of intense biological and ecological research, and the fishery underwent extensive scrutiny by the Florida Marine Fisheries Commission (MFC), newly created in 1983 to manage and conserve Florida's aquatic resources. Several landmark studies were published during that time that described critical biological, ecological, and population attributes of snook from the Atlantic and gulf coasts of Florida (Gilmore 1983, McMichael et al.1989, Rivas 1986, Thue et al. 1982, Tucker and Campbell 1988). These new studies provided the MFC with information to critically evaluate the condition of the snook fishery and to conduct the first rudimentary assessment of the common snook fishery in Florida in 1992.

The initial assessment of 1992 established a protocol for collecting the necessary data and for providing the latest snook population parameters determined by the Florida Fish and Wildlife Research Institute (FWRI) to the MFC and served as a model for future assessments. This initial report contained summaries of population estimates, von Bertalanffy growth parameters, and reproductive characteristics of snook from microhabitats from the Atlantic and gulf coasts. Additional assessments submitted in 1993 and 1995 summarized results from ongoing mark-recapture projects that provided initial estimates of survival and fishing mortality rates and updated biological parameters for both coasts.

In 1990, FWRI began receiving Wallop-Breaux funds (monies collected and provided to the States by the Federal government from excise taxes levied on the sale of fishing-related goods and services) and monies earmarked for research from the newly enacted state fishing license. A myriad of sportfish investigations were conducted and reported during the late 1980s and 1990s, including age-and-growth, reproductive, and genetic studies of common snook (Grier and Taylor 1998, Peters et al. 1998, Taylor et al. 1998, Tringali and Bert 1996). Because these studies provided new and detailed information about the dynamics and genetic structure of snook in Florida, MFC enacted a series of new regulations intended to fine tune the management of snook on the Atlantic and gulf coasts. The new regulations adjusted the winter closed season to include the last two weeks of December and reopened February, established a 40% spawning potential ratio (SPR) as a management goal, and separated the Florida snook population into Atlantic and gulf stocks, although the harvest regulations remained the same for both stocks (Table 1).

In 1996, scientists from Canada’s Department of Oceans and the University of Miami participated in a workshop to review and critique the elements of FWRI’s Snook Research Program. Their final report found that although the overall research and assessments were “state of the art,” two critical techniques should be implemented to conduct future snook stock assessments-- virtual population analysis (VPA) and an improved creel census to collect catch and effort data from the fishery. In 1997, the FWRI snook research group began a broad regional survey to collect age and growth data from exploited cohorts of snook from both coasts of south Florida; however, the initiation of the fishery-dependent creel survey was postponed due to a lack of adequate funds and personnel.

Biological research continued to play an integral part in understanding the biological aspects of this enigmatic species. A long-term life-history study that was begun in 1988 and included material and data collected from the Atlantic and gulf coasts was completed and published in 2000 (Taylor et al. 2000). This study confirmed that the growth parameters of common snook from the Atlantic and gulf coasts were significantly different, supporting the separation of the two stocks. Common snook were also shown to be diandric protandric hermaphrodites, meaning that all snook hatch from the egg as males and that the only way to become female is for the males to reverse sex after their final episode of maturation. The previously assumed maximum age of snook (seven years) was shown to be significantly greater when a 21-year-old 1032-mm-total-length (TL) male was collected on the Atlantic coast; this also indicated that not all males reverse sex. The study convinced the commission that the minimum size and age at legal harvest should be set at the size and age when the sex ratio of the population becomes 1M:1F, not when 50% of the reversed females become mature.

The following year, a study was published that described the mortality of common snook that were caught and released (Taylor et al. 2001). A mortality rate of 2.13% is assumed to be the minimum rate because the study was conducted by biologists and cooperating anglers; the total number of snook that are estimated to be harvested annually include this fraction of snook that were caught but released. As more anglers practice catch-and-release ethics, it will become necessary to refine this estimate, especially because common snook may remain in the harvestable slot for six years and may be caught four times each year. Theoretically, catch-and-release angling may account for more snook killed than the number of snook actually harvested, especially in tournaments.

Through the technique of monitoring the mortality and survival of individual cohorts that compose the fishery, the stock assessments conducted in the late 1990s identified record-high levels of fishing mortality (F), especially in 1997 when F was estimated to be 1.21. It was determined that if the levels of fishing remained at the 1997 level during the late 1990s, then the snook stocks on both coasts would become severely overfished. To prevent the fishery from becoming over-harvested and to attempt to attain an SPR of 40%, an additional regulation was enacted in 1999 that created a harvest slot of 26”- 34” TL. As a result of the reduction in harvest by preventing harvest of snook < 26” and > 34”, in 2000, F was reduced from the 1997 level of 1.21 to 0.34; however, the SPR values remained below the target of 40% (Figure 3.).

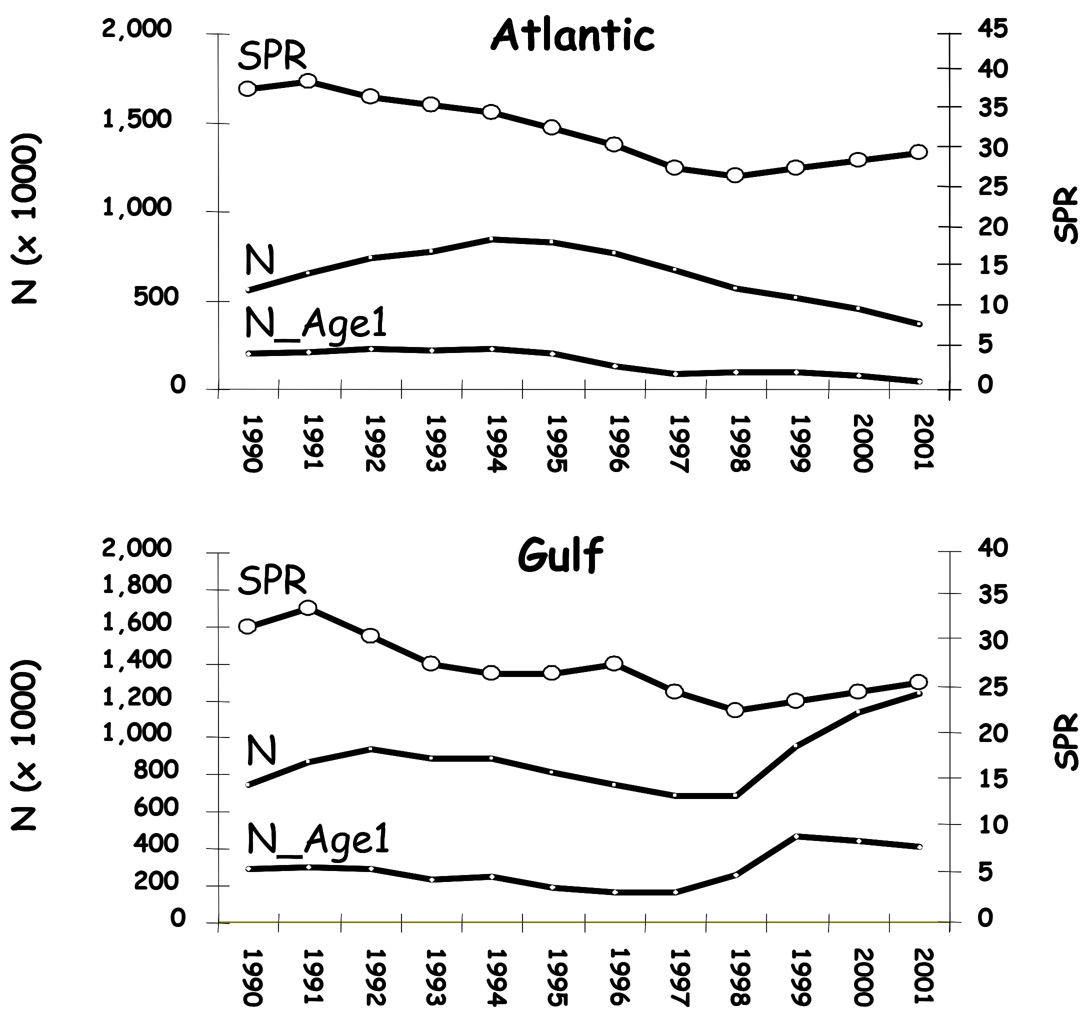

Figure 3. Estimates of SPR population estimates of adult snook (N), and one-year old recruits (N_age 1) for the Atlantic and Gulf stocks of common snook in Florida

In the 2000 stock assessment, the Atlantic stock was determined to be fully exploited because the ratio of F/Z (instantaneous rate of fishing mortality/ instantaneous rate of total mortality) was 0.17/0.37 = 46%. The gulf stock was determined to be moderately overfished: F/Z = 0.34/0.59 = 58%. To try to correct for this over-harvest of the gulf snook, in 2001 the MFC enacted additional regulations on the gulf stock that reduced the bag to one snook/day and closed the month of May to harvest. Even though the snook fishery is one of the most restricted fisheries in Florida, our fisheries-dependent monitoring program continues to report record-high levels of catch, effort, and harvest.

Conclusion

From the initial assessment in 1992 until the last assessment of 2001, the common snook stocks on both coasts of Florida have consistently remained below the target goal of 40% SPR, even though more restrictive harvest regulations have been enacted during this time. Hopefully, the stock assessment of November 2005 which will be based on the fishery data for the previous three years will result in measurable improvements because the fishery has been prosecuted under the most restrictive harvest regulations during this time. However, this may not occur because 60% of Florida’s daily influx of 1,200 new residents settles along the coast, which leads to increasing fishing effort and an inexorable loss of coastal wetlands and mangrove forests. Consequently, scientists and managers may be constrained to a permanently restrictive management policy for snook.

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Monitoring and management of the Nile perch fishery in Lake Victoria

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Abstract

Lake Victoria, the second largest freshwater lake in the world and the first in Africa, is the most productive inland fisheries in Africa and is very valuable to the three East African Community Partner States (Kenya, Uganda and Tanzania) sharing the Lake. The current annual catch is estimated at 554 986 t and the fishery is valued at US$ 544 million locally with exports values estimated at US$ 243 million by 2003 of which over 60% is from Nile perch (*Lates niloticus*).

Regional efforts to monitor and manage the fisheries of Lake Victoria started in the 1930s. However the regional efforts collapsed in the 1970s but joined management initiatives continued under FAO CIFA subcommittee to 1994 when a regional organisation –The Lake Victoria Fisheries Organization was formed by a convention. It was given the mandate to coordinate all management and development activities of the fisheries resources of the Lake. Under the regional body, institutional structures and processes are being strengthened. Fisheries research and management institutions are oriented to address management issues while working in collaboration with other stakeholders. Co-management institutions are established to the grass-root level where beach management units (BMUs) are empowered and facilitated to become partners in management. Efforts to revive the monitoring systems are underway. The benefits accrued from the fisheries have made all the stakeholders including governments to direct attention to the sustainability of the resources. The paper summarizes the initiatives undertaken in monitoring and management in ensuring the fisheries resources and their socio-economic benefits are sustainable.

Introduction

Lake Victoria, the second largest lake in the world has an area of 68 000 km² shared between Kenya (6%), Uganda (42%) and Tanzania (52%) (Figure1). The lake is situated in the central African depression at an elevation of 1122 m above sea level. It is shallow with a mean depth of 40 m and a maximum depth of 84 m. Evaporation is at the rate of 3330 m³/s, which accounts for the major loss of water from the lake while precipitation is the main source of water into Lake Victoria accounting for 82% and the rest coming in through rivers.

Prior to the 1960s, Lake Victoria boasted of a rich fish biodiversity with 400-500 species of fish most of which were cichlids and non-cichlids, consisting of about 50 species. Current observations from commercial catches indicate that the species composition of Lake Victoria fish stocks has been reduced to a three-species fishery, the Nile perch (*Lates niloticus* Linnaeus, 1758) and *Oreochromis niloticus* both introduced in the late 1950s and early 1960s and dagaa (*Rastreneobola argentea*) an endemic cyprinid. The effect of these introductions to the fishery and ecology of the lake was not immediately realised. Catch rates and the total yield continued to decrease for the following 20 years. It was only in late 1980s when Nile perch catches increased almost four-fold and changed the commercial fishery of Lake Victoria into an important export market supplier. Together with the changes, a number of challenges and opportunities exist in managing the resources for sustainability. The paper attempts to analyse the situation and provide some management recommendations.

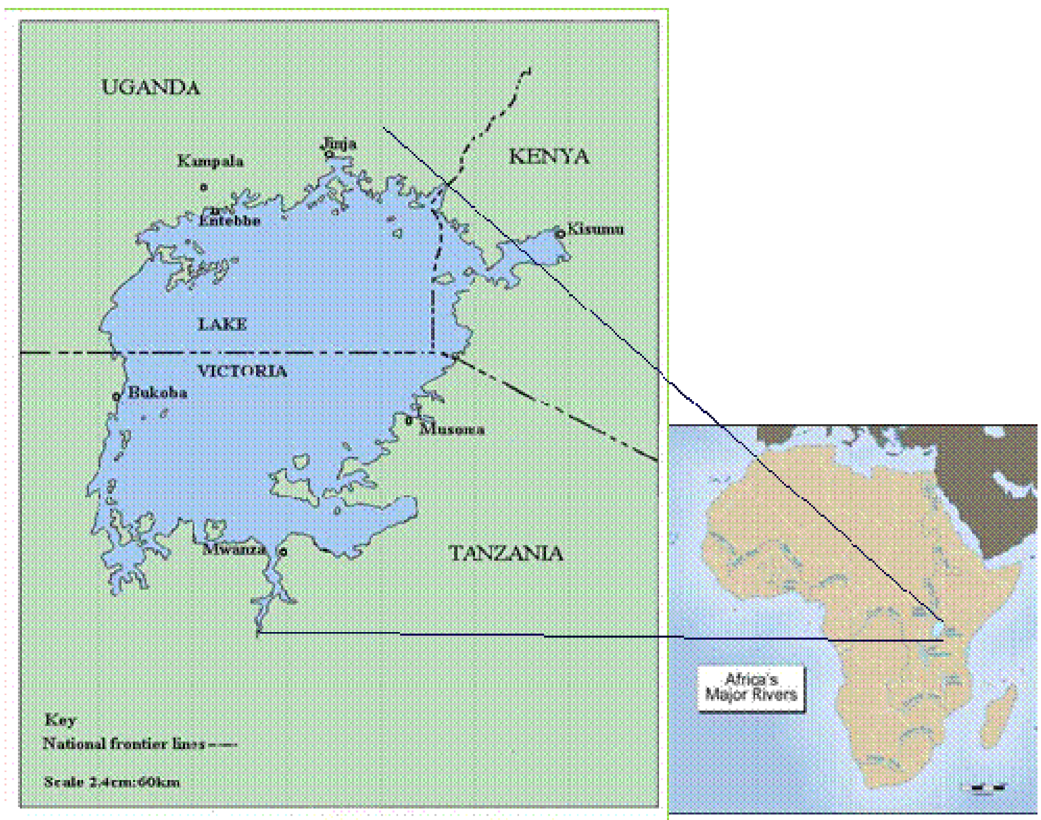

Figure 1. Lake Victoria, indicating the boundaries of the partner States sharing the lake and its location in the African continent

Status of the fishery

The first lake-wide bottom trawl survey in 1969 - 1971 estimated the total demersal fish biomass of Lake Victoria at 750 000 t, of which 80% (600 000 t) were haplochromines (Kudhongania and Cordone 1974). From these findings, bottom trawling and beach seining were advised to fully exploit the haplochromines. Exploratory fishing for haplochromines in the late 1970s showed that a 13.7 m trawler with 130 hp engine and with a bottom trawl net of 20 mm codend could catch an average of 1 000 kg hr^{-1} (Goudswaard and Ligtvoet, 1988). Highest yields of haplochromines were recorded in 1977 when annual landings were 36 158 t, 6264 t and 1560 t contributed 45%, 32% and 10% of the national catches for Tanzania, Kenya and Uganda, respectively. Nile perch at that time contributed only 0.04%, 0.1% and 0.3% to the national catches respectively (CIFA, 1982; Annual Fisheries Statistics, 1988).

In the 1980s an explosion of *Lates niloticus* stocks occurred and by 1987, it contributed 60% of the total yield (Ligtvoet et al., 1995; Bwathondi, 1990. As Nile perch increased to contribute up to 90% of the total catch in 1990 (Ligtvoet and Mkumbo, 1991), a declining trend in all the other species escalated except for dagaa which became the second and Nile tilapia (*Oreochromis* niloticus) the third in the commercial fisheries of the Lake. However, by the late 1990s, as investment and fishing effort in the Nile perch fishery increased, catch rates started to decrease (Figure 2). Currently Nile perch contributes 60%, dagaa *Rastrineobola argentea*) 30% and Nile tilapia 7%. With further decline in catch rates, fishermen have tended to move further offshore while concurrently reducing mesh sizes to catch juveniles or even changing fishing practices by vertically joining gillnets or towing them.

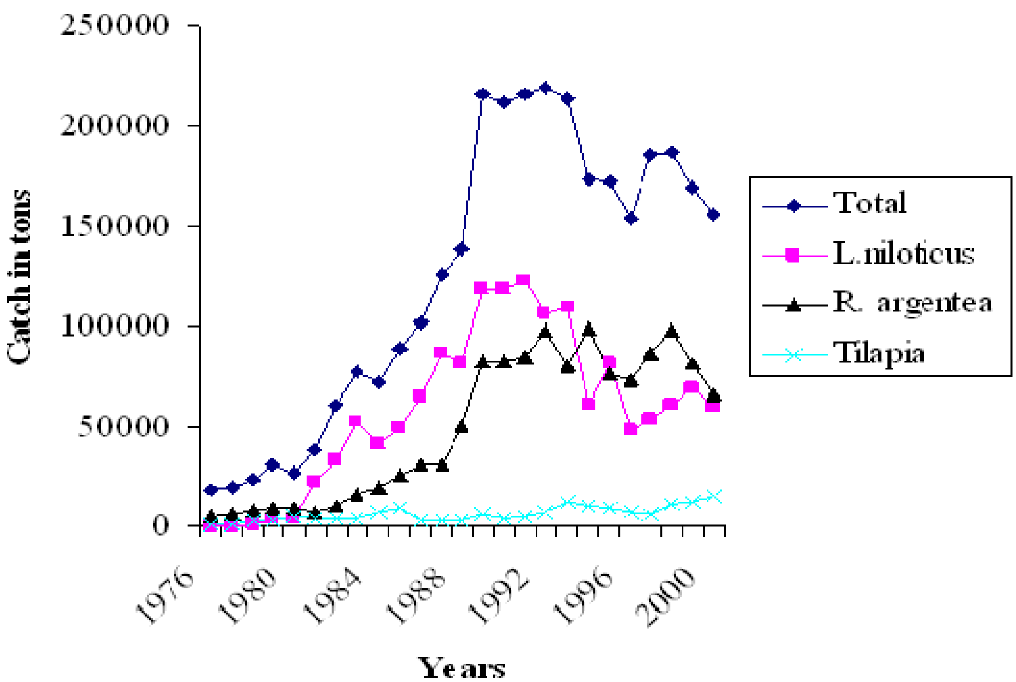

Figure 2. Lake Victoria landings of the three commercially important species and the total in tons indicating the increase in 1980s and the declining trend in 2000

The socio-economic importance of the Nile perch fishery

The Lake Victoria basin supports more than 30 million people of which 3 million depend directly or indirectly on fisheries. Gross domestic product (GDP) estimates from the three riparian states indicate fish contribute approximately 2% of national GDP on average. Fish is also a major source of animal protein and a major source of foreign currency. Records on fish exports from the three countries indicate that fish earns to the riparian states an estimate of over US$ 250 million annually. Fish exports from Lake Victoria in 2003 earned US$ 58.8 million for Kenya, US $ 112.1 million for Tanzania and over US $ 86 million for Uganda.

Nile perch is the most tradable fish species from Lake Victoria. Its importance therefore cannot be over-emphasized. The perch is the main exported fish from the Lake especially to the far markets of Europe, Asia, the Middle East and the United States. Nile perch is mainly exported as fresh and chilled fillet. Nile perch by-products on the other hand, constitute other major tradable items, such as swim bladders and skins which are processed and exported.

Nile perch is also processed locally and exported to and within the countries in the region including Uganda, Kenya, the Democratic Republic of Congo (DRC), Rwanda, Sudan, Zambia and even South Africa. A lot is also processed and consumed locally within the three countries. Nile perch frames are also consumed locally but are the major ingredient in fishmeal. Factory rejects and trimmings are used to make local chips while the skeletons and frames are also exportable within the region. The intestines-fats are boiled and purified to generate fish oil. The expansion in Nile perch markets has resulted in a number of challenges to the fishery especially issues relating to sustainability of the resource and its management.

Challenges facing the Nile perch fishery

A ready market and increasing demand have resulted in continuous increase in fishing effort, use of illegal and destructive gear resulting in over-capacity in fishing effort and signs of overexploitation. catch per unit of effort (CPUE) has been declining in the face of an increase in effort (number of fishing crafts) and increase of nets per boat (Mkumbo, 2002; Muhoozi, 2002). There has been a tendency of reducing gill net mesh sizes and thus landing fish below sizes at first maturity. Presently, factories operate at less than half the established capacity.

The open access system is encouraging most people to rush to fishing in expectations of quick gains at the expense of growth of other sectors. This poses ownership and access challenges. It has also resulted in over-dependence on the Nile perch fishery with many people migrating to the Lake Basin. This illustrates the declining economic opportunities elsewhere in the countries. The increasing population in the basin, therefore, is creating and increasing fishing pressure which is associated with an increase in fishing effort but ending up with the same or less amount of catch.

There are also concerns that the level of insecurity in the lake has increased due to unequal distribution of benefits from the fishery and differences in the level of investment. It is likely that richer traders push away local fishers causing conflict. Poverty assessment studies and other household surveys indicate that fishing communities are still poor despite increased fish trade (Abila, 1997). The challenge therefore is to design measures that will translate good prices received by fishermen into livelihood assets and poverty-reducing programs such as savings and investments.

The other challenge is availability of regular flow of funds for management and research in the fishery sector. There is no clear policy on the way resources are transferred from revenues or benefits accrued from the fish trade towards management of, and research on, fish stocks. More often, financial resources earmarked for fisheries management have been less that 10% of annual requirement. Although there has been significant support through donor programs, these programs are unsustainable. On the other hand, studies have demonstrated that fisheries resources in Lake Victoria are rich and the resource rents are so huge to the extent that, if a proportion is extracted, it can generate sufficient funds to finance fisheries management and research. This would be better than relying on insufficient, and often uncertain, transfers from central government treasuries. The challenge therefore is to make countries plough back part of the economic rents for fisheries management and research for the sustainability of the resource.

Monitoring and management efforts - existing opportunities

Fisheries being of high investment potentials, the East African Community (EAC) has designated the Lake Victoria Basin an economic growth zone. A number of interventions have been taking place in the fisheries with development objectives. To coordinate such initiatives effectively and ensure the sustainability of the resources the EAC partner states formed the Lake Victoria Fisheries Organisation, an intergovernmental organisation to foster cooperation and collaboration for sustainable management, development and utilisation of the fisheries of the lake.

Efforts to monitor and manage the resources jointly among the three East African countries sharing the lake started as early as 1929 when the first lake-wide survey using gill nets was conducted (Lowe-McConnell, 1997). With the establishment of the East African Freshwater Fisheries Research Organisation in 1947, a fisheries dependent monitoring system for data collection was established (Wanjala and Martens, 1974). With the collapse of the East African Community in 1977, each country operated independent systems for data collection under the assistance of the FAO Committee for Inland Fisheries of Africa (CIFA). National fisheries research institutes (TAFIRI- Tanzania Fisheries Research Institute, KMFRI- Kenya Marine and Freshwater Fisheries Institute and FIRRI- Uganda, Fisheries Resources Research Institute) were formed. With the establishment of the Lake Victoria Fisheries Organisation (LVFO) in 1996 the research institutions and the fisheries management institutions of the three states are now together developing and harmonizing research programs and procedures, formulating management decisions and harmonizing regulations and policies for implementation at the national level. There have been a number of projects implemented regionally with donor support, addressing research and management issues of the lake resources and the environment. Among these projects is the Lake Victoria Environmental Management Program (LVEMP), a world bank GEF funded project charged with biodiversity, conservation and management of the lake and its environment, which ended in 2004, while the Lake Victoria Fisheries Research Project (LVFRP), a European Union-funded project, ended in 2002. The latter was charged specifically to create and develop the knowledge base required for the rational management of the fisheries of Lake Victoria and assist the newly established Lake Victoria Fisheries Organisation in the creation and initial functioning of a viable management framework for the fisheries. Following the objectives, a fisheries management plan was developed and which is currently being implemented under a fisheries management plan project (IFMP). It is funded by the EU and implemented through the LVFO institutions of the three partner states – the fisheries management and the fisheries research institutions.

A number of initiatives are taking place under IFMP. There is an extensive monitoring, control and surveillance (MCS) program and a community development program which is addressing capacity building. The latter is also supporting establishment and empowerment of beach management units (BMUs) to be collaborators in managing the fisheries. Research programs are also supported, such as a harmonised and coordinated catch assessment survey, biannual frame survey and lake-wide hydro-acoustic surveys.

Way forward for sustainability of the Lake Victoria Nile perch fishery

The exploitation of Nile perch in Lake Victoria is primarily driven by high demand. The open access status of the lake is linked to increasing fishing effort and limited ability for enforcement of existing legislation. These are key issues to be considered for the sustainability of the fishery. Priority should be given to “enforcing” existing legislation on gear restrictions, i.e. the ban on beach seining and undersize nets should be enforced. More awareness programs to the fisher folk and all the stakeholders should go hand in hand with the surveillance programs. The legal gillnet mesh size of 127 mm (5”) is still catching a large proportion of immature fish and needs to be revised to be species specific. The minimum legal gillnet mesh size is appropriate for the Nile tilapia and other endemic species but not to Nile perch. Progressive efforts to increase mesh size to 152-177 mm (6-7”) for Nile perch should be considered.

To help support the implementation of the mesh size regulation, a size restriction was imposed on the fish processing factories. The size restriction recommended is an allowable slot size of 50-85 cm TL. This should be applied to all the landing sites and receiving centres and not only to fish processing factories.

Control of fishing effort in Lake Victoria has to be species-specific. Indicators of overfishing are more evident in the Nile perch fishery compared with the dagaa and Nile tilapia fisheries. Entry to the Nile perch fishery has to be limited and number and size of fishing craft has to be controlled. However, fishers have the tendency of increasing the number of gear/gillnets per canoe. This un-recorded effort has to be stopped and methods be considered to determine the appropriate number of gear per fishing craft. The region is currently developing a regional plan of action to control fishing capacity. A number of efforts are underway to investigate alternative employment to reduce fishing pressure on fisheries in Lake Victoria. Aquaculture is one area which is at present under research and development as an alternative source of livelihood.

The Lake Victoria fisheries organisation calls upon all stakeholders and development partners to pull efforts together to ensure the sustainability of the Nile perch fishery. The resource is crucial to the lake basin community who depend on it for livelihood, to the three partner states for foreign currency and to the rest of the world community who use it as an important source of protein. Let us work together for the sustainable exploitation of the valuable Nile perch from Lake Victoria.

Acknowledgement

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Integrated fisheries management in Western Australia - a significant challenge for fisheries scientists

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Keywords. Ecologically sustainable development, commercial, recreational, charter, indigenous fishing, prioritising expenditure of research effort, sustainable harvest levels, key indicator species, bioregional management

Abstract

The advent of the 21st century has seen the Department of Fisheries (DoF), Government of Western Australia (WA) embark on an ambitious initiative of integrated fisheries management (IFM) within the broad context of the principles of ecologically sustainable development (ESD). This initiative consolidates the outcome of earlier important initiatives such as the freeze on the issue of commercial fishing boat licences (FBLs) in 1983, and more recently the development of regionally based management strategies for recreational fishing, the formal management of charter fishing, and the recognition of the importance of indigenous fishing. An initial challenge for finfish scientists was the development of a means of prioritizing the expenditure of research effort directed at the important task of determining sustainable harvest levels for key "indicator" species within each Bioregion of the state. It is anticipated that at times, such determinations will need to be made in the absence of adequate data. The ongoing monitoring of the catch shares allocated to each sector also poses a significant challenge. Methods being developed to handle these challenges, and other important future needs identified as a consequence of embarking on this process, need to be addressed to ensure that the limited funds available for monitoring are well spent.

Introduction

Throughout the 20th century, the DoF in WA has achieved many significant milestones in its quest to manage the range of fishing activities that harvest the diversity of fish and shellfish that are found off the 12 000 km of the WA coastline. As a consequence, the commercial fisheries for all of the key invertebrate species and most of the finfish species were brought under formal management. However, as the state population rose along with the level of recreational use of these resources, and individuals became better informed about the need for conservation and sustainable fishing practices, it became apparent that the old framework of independently managing each sector would no longer meet the future needs of the community and government. Clearly, a more integrated approach to the management of fisheries and their supporting ecosystems, which also incorporated a means of allocating catch shares to all user groups within the broad context of ESD, (DoF, 2000; Fletcher et al., 2003), was needed for management in the 21st century.

The coastal environments in WA also support an extremely diverse finfish fauna, which has been the focus for a range of multi-sector, multi-method small-scale fisheries. Consequently, the research and management of the WA finfish resources has been chosen to illustrate how IFM is being implemented in WA, and to identify a number of the significant challenges IFM poses for fisheries scientists. The most crucial element for IFM is the need for an effective ongoing monitoring program of the status of key resources. Central to this is the development and use of risk assessment methodologies to enable the objective prioritisation of the limited research resources available to cover across all Bioregions, habitats and species.

Finfish fisheries over the last 30 years of the 20th century

Of the > 3500 finfish species which are found in the marine waters off WA only around 150 are retained by commercial fin-fishers and just 70 of these (90 stocks) are actually targeted. Given the oligotrophic nature of waters off the WA coast, the sustainable catches of these target species are generally small (<1000 t) by international standards (Penn et al., 2005).

The commercial fisheries located in the estuarine and coastal zone were the first to be formally managed during the late 1960s and early 1970s (Figure 1). Despite the significant increases in catches in the offshore zone between the 1970s and 1980s, the majority of these fisheries were not brought under formal management until the end of the 20th century. By this time, there were 23 managed commercial fin-fisheries. Only the general “wet-line” (primarily handline/dropline) fisheries remained largely unmanaged and “open access”[2]. Similarly, the recreational and charter sectors were considered to be only “loosely” regulated.

Beginning of a new era – 21st century

To address the growing demands for more specific allocation of catch shares for each sector, it was recognised that fishing activities could no longer be managed in isolation. Instead, they should be managed collectively as components of the broad range of community activities undertaken in waters off the coast to ensure the sustainable use of coastal ecosystems.

Fisheries management arrangements need to reflect any significant regional differences in resources and community values. Thus, early in the 21st century, DoF adopted a regional management focus by recognising four marine Bioregions, based on the current classification of faunal regional provinces, together with a northern and southern freshwater bioregion. At this time ESD reporting was also being formally introduced, driven primarily by the need to meet the Australian Government’s EPBC Act requirements. It was within this context, that the IFM strategy was first framed in 2002 (Justice J. Toohey, 2002), and finally initiated in 2004.

[2] It is still only open to the limited number of FBL holders.

One of the key pre-requisites for the introduction of IFM is an adequate management framework for each sector. Since the freeze in 1983 on the issue of new commercial FBLs, there has been a gradual, but consistent decline in the number of licensed fishing vessels and fishers in the industry. Moreover, “open-access” fishing is now well on the way to being formally managed in each bioregion. When this is completed, there will be 27 managed commercial fin fisheries, and a likely further decline in the overall number of participants in the commercial fishery, as the serious “wet-line” operators acquire additional access to remain economically viable in the newly managed fisheries.

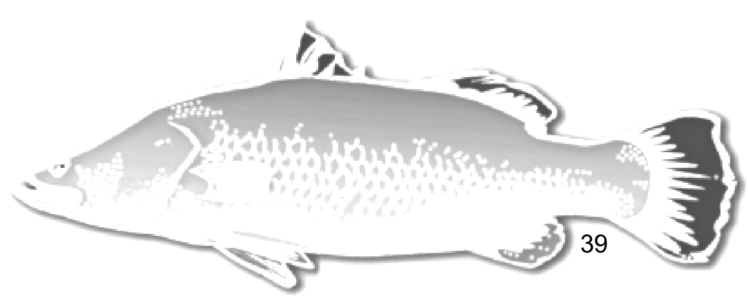
Formal management arrangements, including a ‘cap’ on the number of operators, were introduced into the Charter sector in 2001. There is also a legislative requirement for the access holder to submit trip-by-trip catch and effort records. Surveys of the level of participation in recreational fishing have revealed that while the annual rate of participation is not increasing, recreational fishing effort is continuing to increase. While bioregional management strategies for recreational fishing are now developed, there are still no formal plans of management for any recreational fin-fishery in WA.

The first national recreational and indigenous fishing survey has provided the first estimate of the magnitude of the indigenous catch. This information has complemented another major initiative in WA, which is to develop an indigenous fishing strategy (Hon. E.M. Franklyn QC, 2003).

Data from each of the fishing sectors is now available to estimate the total retained catch by each sector in each of the marine bioregions (Figure 2). A key challenge for the future is to develop a robust system for the collection of the relevant catch and effort data for each of the sectors, together with data needed to enable stock assessments of each of the “indicator species” with sufficient accuracy and precision to satisfy the needs of IFM.

There are six key steps involved in the implementation of IFM:

1. Set the sustainable harvest level for each resource (indicator species).
2. Allocate the explicit catch share for use by each sector (commercial, recreational, indigenous), noting that non-extractive users are likely to be accommodated through spatial management arrangements.
3. Monitor the level of catch within a sector.
4. Manage the operations of each sector within its allocation.
5. Develop mechanisms to enable the re-allocation of catch shares.
6. Monitor the status of the indicator species.



Steps 1, 3 and 6 are the priority tasks for the scientists. The extent to which the implementation of this new initiative will succeed is dependent to a large degree on how well the scientists deal with these three tasks, and others discussed below.

Challenges facing scientists

A strategic review of finfish research in WA has confirmed that Steps 1, 3 and 6 above are the three most important objectives to be dealt with by DoF finfish scientists. The majority of fisheries that exploit these species are multi-species fisheries. In those cases where individual species cannot be accurately targeted, these fisheries must thus each be managed on the basis of the most vulnerable species in the catch (i.e. an indicator species). However, because there are many more potential "indicator" species than the available DoF resources could accommodate effectively, there is a need to prioritise the use of research effort[3]. The strategy adopted to resolve this issue was to first document all information available on the status of all the stock(s) of each target species by the completion of a Resource Assessment Framework. This framework outlines what issues need to be examined, why and how they are to be examined, including the specific management objectives and performance measures for each target species. A "risk assessment" is then undertaken to prioritise monitoring needs. Priorities were determined for each stock by scoring against each of the following seven criteria (Fletcher and Lenanton, Prep).

1. Inherent vulnerability - basic biological characteristics, irrespective of current status.
2. Current risk to the stock - current and likely future status of stock.
3. Current management information requirements – levels of information needed to make management arrangements operate.
4. GVP – gross value of landed product.
5. Recreational significance – relative priority to all recreational fishers targeting stocks within bioregions.
6. Cultural significance – level of social concern or significance to the wider community.
7. Customary significance – relative level of customary use or significance.

The cumulative total of scores is used as the main basis for prioritisation amongst different stocks but the current risk to a stock is seen as critically important for long-term sustainability and this therefore warranted extra weighting. Each stock was also assessed separately on the basis of the current or likely future level of resource conflicts, which is important for the implementation of IFM.

[3] It is recognised that within the ESD framework (Fletcher et al. 2003), impacts on the broader fish community and supporting ecosystem also need to be sustainably managed. Thus by-catch issues and broader ecosystem impacts will be similarly addressed at a later date.

Following these evaluations, the risk scores for each target species in each of the habitats within each bioregion are presented in the form of a matrix (Figure 3). Using various combinations of these scores assists in determining the priorities amongst different species within a bioregion, within a habitat, or across all bioregions and habitats.

Determination of sustainable harvest levels

Following the objective identification of priority stocks, there is a need to ensure adequate data is available to facilitate the determination of the sustainable harvest level for each of these indicator species. Ongoing catch and catch-rate data are essential. While there is a legal requirement for commercial and charter operators to provide such data, this is not the case for the recreational sector. In the absence of a marine recreational fishing licence, comparable data for this sector needs to be gathered through regular expensive surveys. It is anticipated that lack of such data will severely compromise the capacity for scientists to determine sustainable catch for indicator species, particularly in areas where the relative levels of recreational participation are high. Thus management of those species within the estuarine/inshore and demersal shelf habitats of the west coast bioregion are at most risk of failing to meet management objectives /performance criteria. While recreational fishing remains unlicensed in WA, the task of gathering adequate data remains a significant challenge for scientists. Nevertheless, there are some promising projects underway to help overcome this problem. An FRDC-funded project has recently commenced to determine cost effective techniques to monitor recreational catch and effort in Western Australian demersal finfish fisheries. Similarly, a recreational angler program (RAP), which encourages research volunteers to provide the necessary data, has also been implemented.

At this early stage of the IFM process, it is difficult to anticipate the frequency for which sustainable harvest levels will need to be determined. The IFM Policy states that allocation decisions cannot be delayed as a consequence of a lack of suitable data; they must be made in a timely manner using the best available information. Thus, likely acceptable catch ranges with associated uncertainty may need to be used in a precautionary manner as a “proxy” for sustainable catch for a range of target species for which there is little information. In such instances, the acceptable catch range is equivalent to the historical catch range of functionally similar species that are considered to be fully exploited.

Monitoring catch levels and catch shares

Once there is an agreed set of management objectives, the sustainable harvest levels and catch shares have been determined, and the supporting operational management framework is in place, then ongoing harvest levels need to be carefully monitored to:

1. Provide data needed for ongoing stock assessment, thus enabling adjustments to harvest levels if and when required.

2. Manage catch shares of each sector within agreed limits.

The required outcomes, which will dictate the spatial/temporal resolution of the management arrangements, will have a very significant bearing on the design of the monitoring program, and the likelihood of its success.

Methods are beginning to be developed to address some of the issues that have already been identified. An FRDC-funded project (2004/042) which began in July 2004 is attempting to determine cost effective methodology for ongoing age monitoring needed for the management of finfish fisheries in WA.

Other issues

New management arrangements, particularly those developed to manage the recreational catch, will need careful and ongoing evaluation. State/Commonwealth/international jurisdictional issues also pose some significant risk to the management of some stocks, and to the equitable sharing of those resources. Topical examples are the management of coastal shark species throughout State waters, and the marlin and associated offshore recreational resources within the west coast and Gascoyne bioregions.

Finally, data required to make this IFM/ESD initiative succeed will increasingly need to come from the community, in particular the recreational sector. Thus it is critical to develop a process that optimises community comprehension of, and willing participation in, data gathering initiatives. Current FRDC-funded projects on demersal scale-fish in the west coast bioregion have key objectives to develop processes to maximise communication with all sectors.

Conclusions

DoF has embarked on the transition from single fishery/individual sector management arrangements to an integrated approach across all user groups. The broad challenge for scientists is twofold: to develop a system that (1) is capable of providing cost effective monitoring of biological and fishery data required for ongoing sustainable harvest estimation and monitoring of catch levels and catch shares for each sector, and (2) has the broad support of the community. In relation to the first challenge, effective monitoring is dependent on:

- A clear understanding of management objectives and required outcomes in each bioregion.
- The development of a Strategic Plan to guide the implementation and management of research endeavour.
- A risk assessment of priorities across bioregions, habitats and species.
- Management on the basis of the most vulnerable species.
- Survey designs that allow cost effective monitoring at the appropriate frequency.

These key factors are crucial to implementing and maintaining any long-term monitoring strategy that must be cost effective, transparent, and sufficiently flexible to meet new challenges.

Acknowledgements

The authors gratefully acknowledge the contributions made by Research Division staff, particularly Ms Susan Turner.

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Recreational fishing surveys: which method should I use?

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Keywords: bus route method, roving creel survey, national recreational and indigenous fishing survey, recall bias.

Various survey methods have been used to provide estimates of the recreational catch and fishing effort. These include on-site surveys, phone surveys and mail surveys. However, results from different survey methods are not always comparable due to inherent biases associated with each method. Results from the National Recreational and Indigenous Fishing Survey (NRIFS) were compared to estimates from on-site surveys to investigate the different biases associated with these methods. There was no indication of any systematic bias between the NRIFS and on-site surveys. For some locations NRIFS provided higher estimates of effort and for other locations the on-site surveys provided higher estimates of effort. There was no significant difference in the catch estimates between the NRIFS and on-site surveys. It was concluded that results from the NRIFS and on-site surveys were comparable. However, the NRIFS provided better coverage of shore-based fishing for the Pilbara region since the on-site survey did not include night fishing. It is shown that the NRIFS provided catch and fishing effort estimates for large regions within Western Australia with acceptable levels of precision. Estimates of fishing effort for two large embayments (Shark Bay and Cockburn Sound) were calculated from the NRIFS data with acceptable levels of precision however, the sample size was considered to be too small to calculate catch estimates for these areas. It was concluded that the NRIFS provided useful catch and effort information at national and state levels and for large regions within Western Australia. On-site methods should be used to estimate total catch and fishing effort for smaller areas within a state such as embayments or estuaries.

Introduction

There are many survey methods available to estimate catch and fishing effort for a recreational fishery. The choice of method is influenced by a number of factors including availability of a sampling frame, geographical size of the fishery and type of fishing activities (for example boat or shore). The method chosen must also provide an acceptable level of precision within budgetary constraints.

Different survey methods have been shown to provide conflicting results due to inherent biases. This was the case with phone-diary surveys and phone surveys that relied on the respondent's ability to recall information (Lyle, 2000). NRIFS (Henry and Lyle, 2003) attempted to address the problem with recall bias by providing respondents with a diary.

The NRIFS was developed to provide national, state and regional estimates of catch and fishing effort. The survey comprised a screening survey of Australian households to select fishing households. For each fishing household, this was followed by a 12-month phone survey with each household member sent a diary to record basic details of each fishing trip to overcome problems with recall bias. To avoid respondent fatigue the diary acted as a memory jogger rather than a place to record detailed catch and fishing effort information. The households were contacted by phone at the end of the month (or more often if necessary) to collect information on fishing trips. The diary survey continued from May 2000 to April 2001.

Catch and fishing effort estimates from the NRIFS had not previously been compared to estimates derived from on-site surveys. For this reason there have been concerns about the comparability of results from these different methods. This was particularly the case for Western Australia (WA) where on-site methods were widely used.

There has also been considerable conjecture about the useful resolution of the NRIFS. The survey was designed to provide broad estimates of catch and fishing effort at national and state levels and for large regions within a state. The catch and fishing effort estimates for these levels were available however; the errors had only been previously estimated and reported at the national and state levels. To assess these issues the errors associated with the catch and fishing effort have been estimated for the Pilbara region and two large embayments (Cockburn Sound and Shark Bay) to determine the useful resolution of the NRIFS within Western Australia.

Methods

NRIFS

Catch and fishing effort together with standard errors for regions within WA were estimated from the data collected for the NRIFS using the same methods of analysis documented by Henry and Lyle 2003. The data were expanded using the integrated weights provided. The calculations were validated against results for the whole of the state published in the report.

Initially the integrated household weights were used for expansions to estimate total catch and fishing effort (Henry and Lyle, 2003). However, neither the reported state catch, fishing effort nor associated errors could be reproduced. The expansions were then repeated using the person weights for household members and the results were shown to match those in the report.

The NRIFS catch and fishing effort estimates for regions of the state together with standard errors were then compared to results for these regions from on-site surveys. Standard errors were reported where available. All statistical tests were conducted at the 0.05 level of significance.

Pilbara region on-site survey

The on-site survey incorporated the area between 4 nm south of the Ashburton River (114°50′E) up to and including Broome, and included approximately 1200 kilometres of coastline (Figure 1). The region offers a range of fishing experiences including angling and fishing for crabs in the ocean and tidal creeks. The region is a popular tourist destination between April and October when residents of the south of WA travel north to escape the winter weather. Local residents and tourists participate in fishing activities.

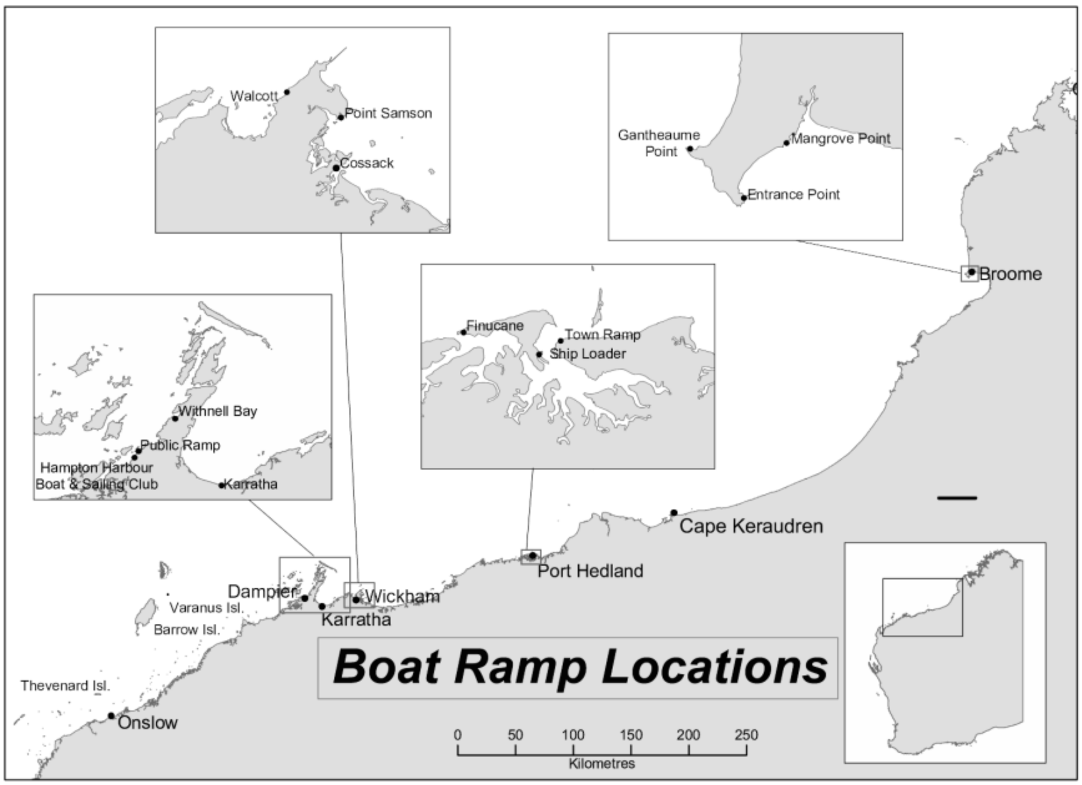

Figure 1. Map of Pilbara region of Western Australia showing boat ramps surveyed to record recreational fishing catch and effort 1999-2000

Separate on-site survey methods were used to estimate the recreational catch of all species for boat-based and shore-based fishers in the region from December 1999 to November 2000 (Williamson et al., in press). The bus route method (Robson and Jones 1989, Jones et al. 1990), where a survey interviewer visits all boat ramps in a district on the one day, was used for trailer boats launched from public boat ramps. Roving creel surveys were used to estimate the catch and fishing effort for shore-based fishers and fishers launching small boats from beaches.

The recreational catch and fishing effort from tourists on Thevenard Island (Figure 1) was estimated by surveying groups staying on the island. The Island Manager conducted the survey at the end of each day's fishing activities on a random sample of days. The survey continued while the island was open to tourists from March 1 to October 15, 2000.

Staff working on Barrow Island, Varanus Island and Thevenard Island (Figure 1) were asked to complete a questionnaire before leaving the islands. This component of the recreational catch and fishing effort for the region was estimated from the sample data collected from the survey together with information on the number of trips to these islands during the 12-month survey.

The catch estimates provided by the NRIFS and on-site surveys were compared by fitting a regression line to the predominant marine species.

The following alternate hypotheses were then tested:

H0: The biases associated with the two methods are equal (slope = 1).
H1: The biases associated with each method are not equal (slope <> 1).

Barramundi was not included since it was mostly caught in tidal creeks rather than the ocean. This type of fishing was not adequately covered by the on-site survey.

The catch and fishing effort from charter boats was not included in the study since a mandatory logbook program for tour operators was undergoing development at the time.

Cockburn Sound and Owen Anchorage on site survey

Cockburn Sound is an embayment south of Fremantle. Separate survey methods were used to estimate the recreational catch of all species for boat-based and shore-based fishers in the region from September 2001 to August 2002 (Sumner and Malseed, 2004). The bus route method was used for trailer boats launched from public boat ramps. Roving creel surveys were used to estimate the catch and fishing effort for shore-based fishing.

Shark Bay on site survey

Shark Bay is a large embayment south of Carnarvon. The boat-based and shore-based fishing effort between April 1998 and March 1999 was estimated by an on-site survey (Sumner et al., 2002)). Later surveys, from May 2000 to April 2001 (Sumner and Malseed, 2001) and May 2001 to April 2002 (Sumner and Malseed, 2003) estimated the catch and fishing effort from public boat ramps only.

Results

Pilbara region boat-based fishing

During the on-site survey 3,085 boat crews were interviewed at boat ramps. Of these 2,696 had been fishing (Williamson et al., in press). The boat-based fishing effort was estimated by the on-site surveys as 109,000 days of recreational fishing effort. The standard error (*se*=3,000) for the on-site boat-based fishing effort estimate was 3 percent of the total fishing effort for the region.

The sample for the NRIFS included 276 persons that fished from a boat in the region. The boat-based fishing effort was estimated as 114,000 fisher days from data collected by the NRIFS. The standard error (*se*=17,000) for the NRIFS boat-based fishing effort estimate was 15 percent of the total fishing effort for the region. The standard errors for the NRIFS were higher than for the on-site survey.

There was no significant difference between the fishing effort estimates from the NRIFS and Pilbara on-site survey for boat-based fishing (t (2,970) =0.498, p=0.618).

The regression relationship was significant (f (1, 13) =30.078, p=0.00) showing a linear relationship between the catch estimates for individual species for the two methods (Figure 2). The intercept was not significantly different from zero (t (13) =1.968, p=0.071) indicating that the line of best fit passed through the origin.

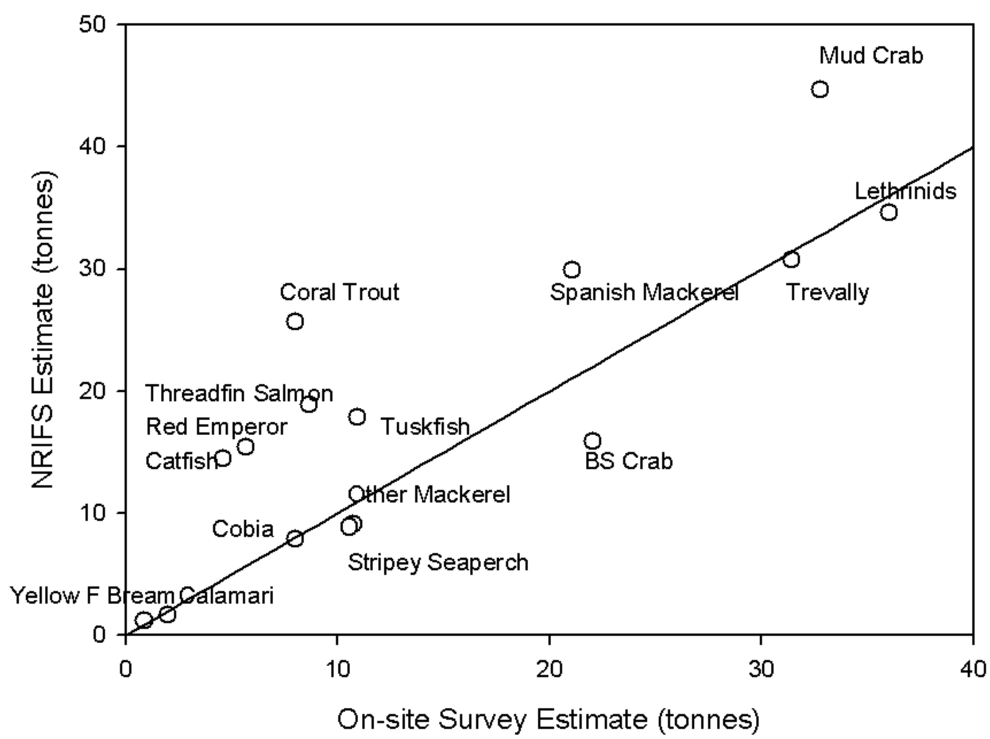

Figure 2. Comparison of boat-based catch estimates for NRIFS with on-site survey

H0 was accepted since the slope of the regression line (1.126) was not significantly different to one (t (14) =1.145, p=0.272). There was no overall significant difference between the catch estimates from the NRIFS and the on-site survey for boat-based fishing in the Pilbara region.

Pilbara region shore-based fishing

During the on-site survey 569 groups of shore-based fishers were interviewed at fishing locations and campsites. Of these 391 were fishing at the time of the interview, 110 had already finished fishing for the day, 45 had not been fishing and 23 were planning on fishing later that day (Williamson et al., in press).

The shore-based fishing effort was estimated by the on-site surveys as 67,000 days of recreational fishing effort. The standard error (*se*=3,000) for the on-site boat-based fishing effort estimate was 5 percent of the total fishing effort for the region.

The sample for the NRIFS included 364 persons that fished from the shore in the region. The shore-based fishing effort was estimated as 130,000 fisher days from data collected by the NRIFS. The standard error (*se*=17,000) for the NRIFS boat-based fishing effort estimate was 13 percent of the total fishing effort for the region. The standard errors for the NRIFS were higher than for the on-site survey.

There was a significant difference (t (886) =4.321, p=0.00) between the fishing effort estimates from the NRIFS and on-site survey for shore-based fishing. The estimate of fishing effort for the on-site survey was significantly lower than the estimate based on the NRIFS data.

Initially, the regression was not significant (f (1, 11) =3.479, p=0.089) and the intercept was not significantly different from zero (t (11) =0.670, p=0.517). Once the intercept was removed the regression was significant (f (1, 12) =17.084, p=0.001). Since the slope of the regression line (1.625) was not significantly different to 1 (t (12) =1.590, p=0.138), H0 was accepted (Figure 3). There was no significant difference between the catch estimates for individual species from the on-site survey and those for the NRIFS.

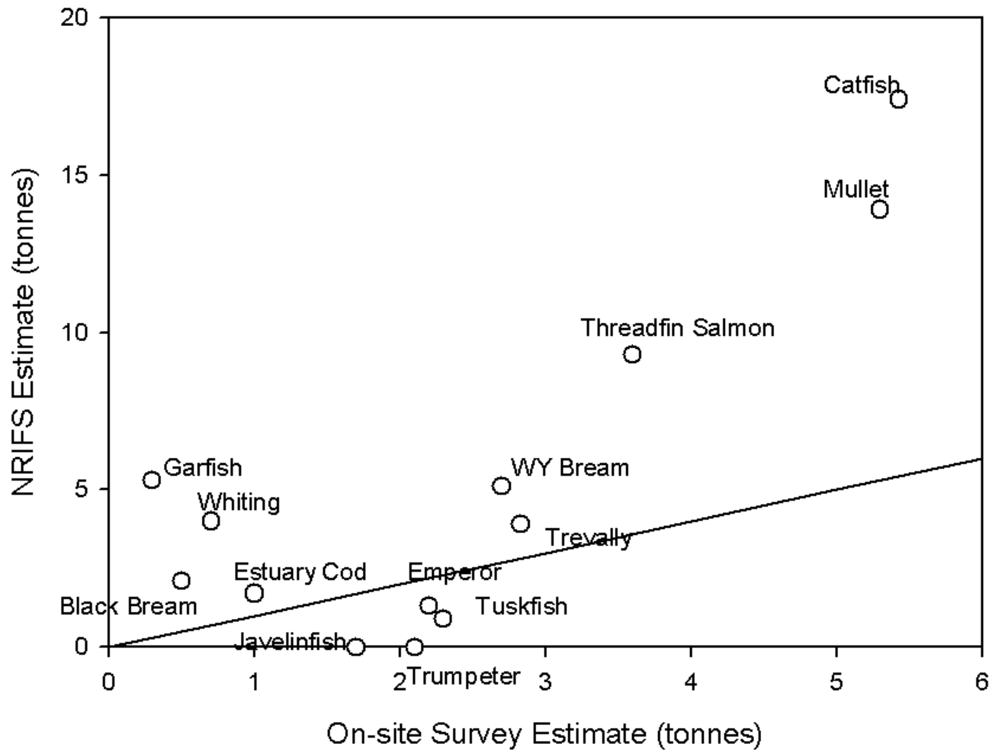

Figure 3. Comparison of shore-based catch estimates for NRIFS with on-site survey

Cockburn Sound and Owen Anchorage

During the on-site survey 1235 interviews were conducted at boat ramps. Of these, 692 boats had been fishing. An additional 619 shore based fishing parties were interviewed. The fishing effort was estimated by the on-site surveys as 200,000 days of recreational fishing effort (104 000 boat, 96,000 shore) (Sumner and Malseed, 2004).

The sample for the NRIFS included 49 persons that fished from a boat and 40 persons that fished from the shore. The boat-based fishing effort was estimated as 54,000 fisher days from data collected by the NRIFS. The standard error (*se*=8,000) for the NRIFS boat-based fishing effort estimate was 14 percent of the total fishing effort for the region.

The shore-based fishing effort was estimated as 28,000 fisher days from data collected by the NRIFS. The standard error (*se*=4,000) for the NRIFS boat-based fishing effort estimate was 14 percent of the total fishing effort for the region.

Shark Bay

Shark Bay is a large embayment south of Carnarvon. The fishing effort between April 1998 and March 1999 was estimated by an on-site survey as 89,000 fisher days (67,000 boat (of which 17,000 was boats launched from beaches) and 22,000 shore (Sumner *et al*., 2002)). A later survey (Sumner and Malseed, 2001) estimated the fishing effort from public boat ramps as 35,000 fisher days from May 2000 to April 2001; however, boats launched from beaches were not included. Another survey (Sumner and Malseed, 2003) estimated the fishing effort from public boat ramps as 31,000 fisher days from May 2001 to April 2002, however, boats launched from beaches were also not include in this survey. The later on-site surveys did not include shore-based fishing since the focus was on pink snapper, which were rarely caught from the shore.

The sample for the NRIFS included 33 persons that fished from a boat and 27 persons that fished from the shore. The boat-based fishing effort was estimated as 42,000 fisher days from data collected by the NRIFS. The standard error (*se*=7,000) for the NRIFS boat-based fishing effort estimate was 18 percent of the total fishing effort for the region.

The shore-based fishing effort was estimated as 16,000 fisher days from data collected by the NRIFS. The standard error (*se*=3,000) for the NRIFS shore-based fishing effort estimate was 18 percent of the total fishing effort for the region.

Discussion

Pilbara region boat-based fishing

There was good agreement between estimates of catch (Figure 2) and fishing effort from the NRIFS and on-site survey for boat-based fishing. There was no evidence of any systematic bias between results from the two survey methods.

Pilbara region shore-based fishing

The lower estimates of shore-based fishing effort for the on-site survey when compared to the NRIFS is most likely due to the on-site survey missing some of the shore-based fishing activity. Night fishing was not covered by the on-site survey and some of the locations where shore-based fishing occurred were missed on occasions.

Since there was a significant difference in fishing effort between the NRIFS and on-site methods a similar difference in catch estimates was expected. However, the difference in catch estimates between the two methods was not significant. Any difference between the two methods may not have been detected due to large errors, particularly for the NRIFS, and hence variability in the catch estimates. For some species there is reasonable agreement between the catch estimates for the NRIFS and on-site survey (Figure 3). However, the NRIFS catch estimates for catfish, mullet and threadfin salmon are higher than estimates from the on-site survey. It is likely that a type II error has been made.

Cockburn Sound and Owen Anchorage

The difference between the fishing effort estimates from the NRIFS and on-site surveys was considerable. The fishing effort estimated by the on-site surveys was greater for boat-based and shore-based fishing. This could not be completely explained by the two surveys being conducted during different years.

The catches for the two methods were not compared due to the small sample sizes for boat-based and shore-based fishing for the NRIFS.

Shark Bay

There is good agreement between the fishing effort estimates for the various on-site surveys and NRIFS. The reduction in fishing effort from 1998 to 2000 was most likely due to management measures such as the closure of the Eastern Gulf to fishing for pink snapper in July 1998 and reduced size and bag limits for the Western Gulf in August 2000.

The sample size for the NRIFS was considered too small to estimate total catch for this embayment.

Conclusion

The NRIFS provided useful catch and fishing effort information at national and state levels and for large regions within Western Australia. The estimates of fishing effort for smaller areas such as large embayments also had an acceptable level of precision. However, the catch estimates from the NRIFS were not useful for areas smaller than regions in Western Australia due to the small sample size. It is concluded that on-site methods should be used to estimate total catch for smaller areas within a state such as embayments or estuaries.

The errors associated with fishing effort estimates derived from the NRIFS data were small considering the number of people in the sample that fished in areas such as Cockburn Sound and Shark Bay. For this reason, the effort estimates for some locations may still be useful even if the precision of the catch estimates is not acceptable.

There was no indication of any systematic bias when comparing results from the NRIFS with on-site surveys. For some locations the NRIFS provided higher estimates of effort and for other locations the on-site surveys provided higher estimates of effort. There was no significant difference in the catch estimates between the NRIFS and on-site surveys. It can only be concluded that overall the results from the NRIFS and on-site surveys were comparable. However, the NRIFS provided better coverage of shore-based fishing for the Pilbara region since the on-site survey did not include night fishing.

The NRIFS provided broad estimates of catch and fishing effort. The precision of the NRIFS estimates could be improved by increasing the sample size (and cost). Although beyond the scope of this study, this could be investigated using the available data by changing the sample size and recalculating the standard errors. However, eventually the point of diminishing returns, where there is little improvement in precision associated with a large increase in cost, would be reached. It is likely that the cost of using the NRIFS approach to estimate the catch for small areas such as embayments and estuaries would be prohibitive.

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Evaluation of precision and cost effectiveness of bus route, creel and phone surveys for estimating recreational catch

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Keywords: recreational fishing, catch rate, Monte Carlo simulation, Port Phillip Bay, Victoria

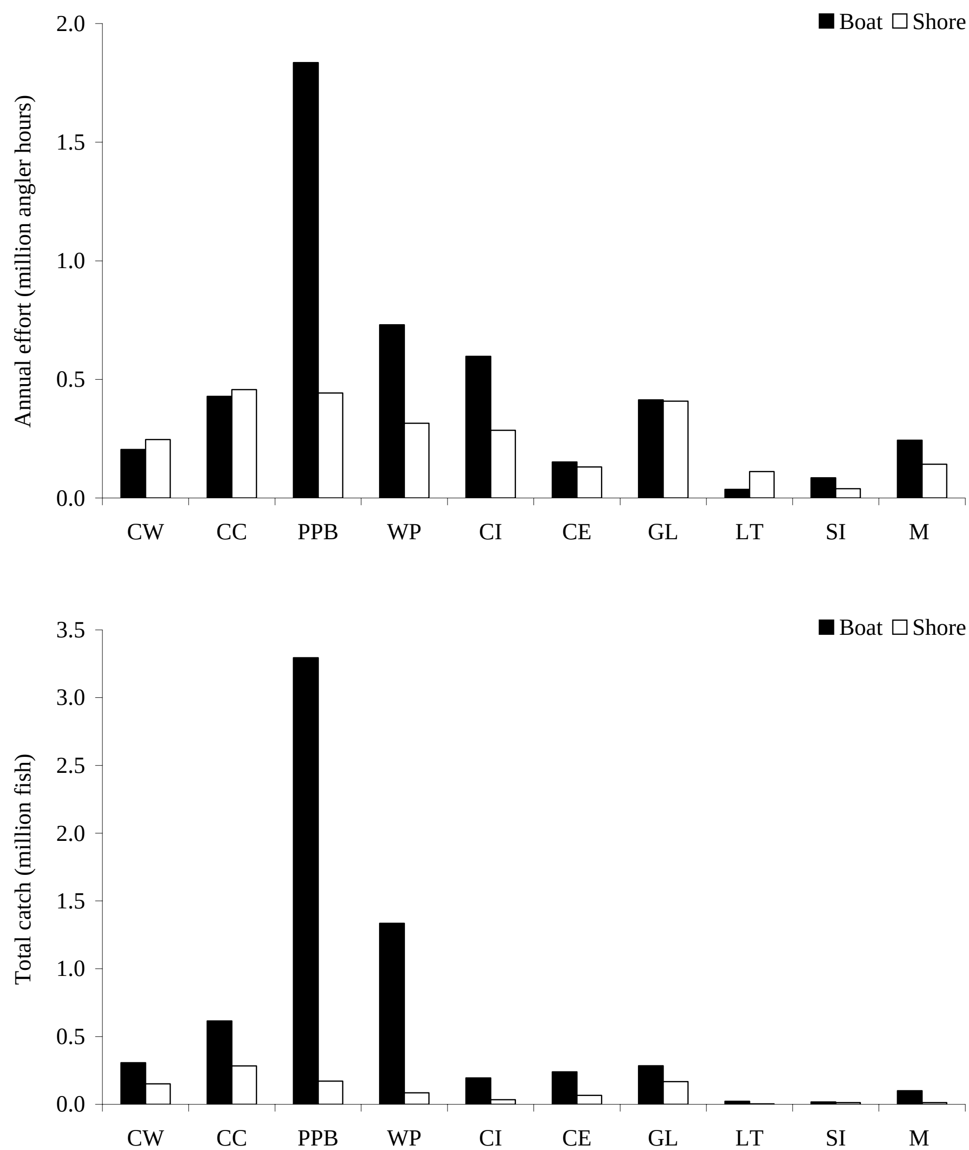
Abstract

The majority of recreational angling in Victorian bays and inlets occurs in Port Phillip Bay, where 95 % of the harvest is taken by anglers in boats. Three alternative survey methods have been used to estimate the total recreational catch of key species from boat-based angling in Port Phillip Bay: an off-site phone survey and on-site bus route and creel surveys. Ad hoc estimates indicate the total recreational catch of key species in Port Phillip Bay can exceed commercial catches. While logbook monitoring of commercial catches provides a continuous time series of catch data, routine estimates of total catch from the recreational sector have never been obtained. This project aims to evaluate survey methods for monitoring recreational harvest of key species. Monte Carlo simulations were used to estimate catch rates across a range of sample sizes using estimated probabilities and distributions from previous recreational fishing surveys. Estimated catch rates remained constant with increasing sample size for all survey methods; however, the precision increased with more samples. Assessment of the cost effectiveness of each survey method was made using the simulated precision and estimated survey costs. The cost of conducting a phone survey was considerably lower for the number of samples required to achieve reasonable precision, making this a cost effective survey method. The information obtained from the simulations will be used to design a precise and cost effective monitoring program to estimate recreational catch from Port Phillip Bay.

Introduction

Assessment of the impact of commercial and recreational fishing in Victoria's bays and inlets is hindered by the limited catch and effort data and the lack of rigorous stock assessments (Dragun 1991, Li 1999, Kearney 2002). A continuous time series of catch data is generally available for the commercial sector, but routine estimates of total catch from the recreational sector have never been obtained. Ad hoc estimates suggest the total recreational catch can exceed commercial catches for key species, such as snapper (*Pagrus auratus*), King George whiting (KGW) (*Sillaginodes* *punctata*) and sand flathead (*Platycephalus bassensis*). For example, the estimated total recreational catch of snapper (211 t), KGW (93 t) and flathead (395 t) in Port Phillip Bay during 2001 (Henry and Lyle 2003) exceeded commercial catches of 53, 85 and 23 t (Anon 2004), respectively. The aim of this project is to find a survey method that could be used every year to estimate the total recreational catch on a small spatial scale. The initial aims of this project were to review previous survey methods used to estimate total annual recreational catches in Victorian bays and inlets and statistically assess the costs and sampling requirements of different survey methods. Comparisons of bus route, creel and phone surveys are made for a range of sample sizes by computer simulations using data from previous surveys, developing a cost model and evaluating the trade-offs between precision and cost.

There have been 25 published recreational fishing surveys for coastal Victoria including studies in Port Phillip Bay, Western Port, Corner Inlet, Gippsland Lakes, Lake Tyers and Mallacoota. These have included off-site surveys, such as door to door, telephone, mail and diary surveys, and on-site surveys, such as bus route, creel and aerial surveys. The majority of fishing effort in Victoria occurs in estuarine habitats (42.8%) compared to offshore (0.8%), coastal (13.5%), rivers (21.7%) and lakes and dams (21.2%) (Henry and Lyle, 2003). Recreational fishing effort is also distributed among water body types according to population distribution and access. Port Phillip Bay and Western Port are within close proximity to Melbourne, the major urban population centre in Victoria (Henry and Lyle, 2003). Recreational fishing in Port Phillip Bay alone accounts for 30% of the state-wide effort and 50% of the state-wide catch (Figure 1). Within Port Phillip Bay, 80% of the effort and 95% of the catch is from boat anglers. Many surveys have been conducted in Port Phillip Bay where the recreational fishery provides an opportunity to compare survey methods using a simulation approach without repeating surveys simultaneously.



Abbreviations: CW – Coastal West, CC- Coastal Central, PPB – Port Phillip Bay, WP - Western Port, CI – Corner inlet, CE – Coastal East, GL – Gippsland Lakes, LT – Lake Tyers, SI – Sydenham Inlet, M – Mallacoota.

Figure 1. Total annual effort and catch for boat and shore anglers in Victorian bay and inlets

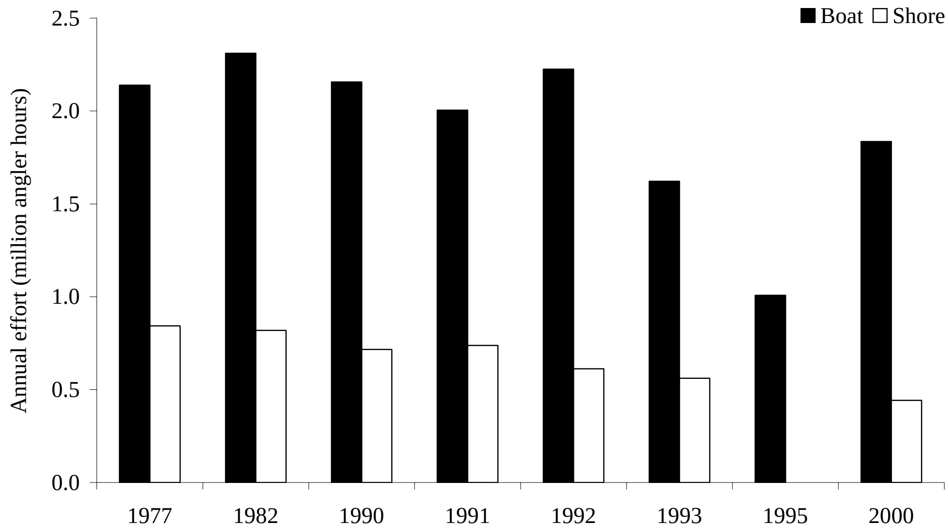
Previous estimates of total annual catch in Port Phillip Bay

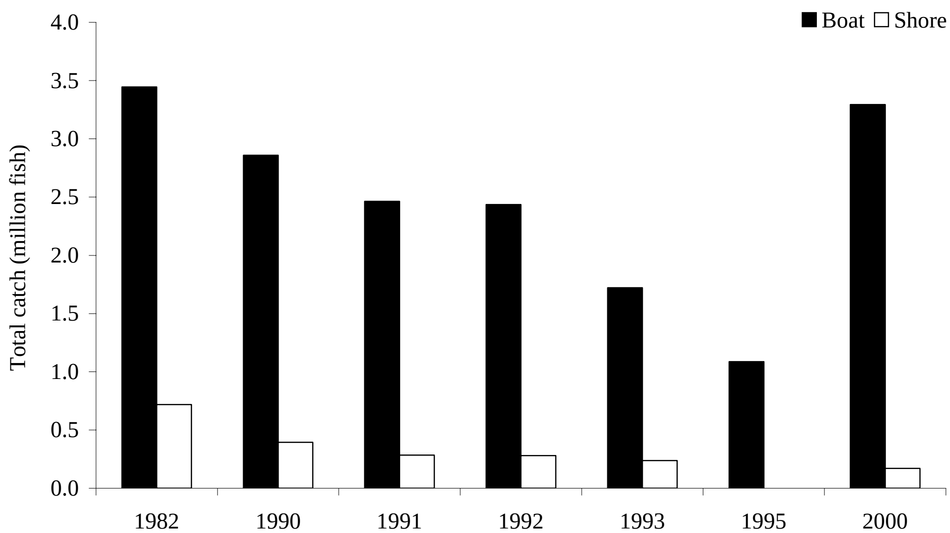
The methods of estimating total catch are different among survey methods (Pollock et al., 1994). For bus route and creel surveys, the total annual catch is estimated by multiplying the daily catch rate with the estimated total annual effort. Effort is determined by the amount of time trailers are observed at each ramp in a bus route survey and the number of boats engaged in fishing and the average number of anglers per boat from an aerial survey. In a phone survey, total annual effort does not need to be calculated. The total catch is estimated by expanding the total catch for each household with an expansion factor that uses population census data to scale. Catch in numbers is converted to catch in weight with appropriate weight conversion factors.

Three alternative survey methods have been used to estimate the total catch from the boat-based fishery in Port Phillip Bay: creel surveys (supported by aerial surveys) (Beinssen, 1977; MacDonald and Hall, 1987; Coutin et al., 1995), bus route surveys (Conron and Coutin, 1998) and a phone survey (Henry and Lyle, 2003). There was no shore estimate taken in the bus route survey. Effort from boats was extrapolated to an annual estimate and multiplied by the average number of anglers per boat to convert boat hours to angler hours. Data from these five previous surveys (Figure 2) indicates the proportion of effort for boat and shore anglers remained similar. There was a decline in shore effort, which halved from 0.8 million angler hours in 1982 to 0.4 million angler hours in 2000 that has been confirmed by anglers. The estimated boat effort has remained around two million angler hours between 1977 and 2000.

The total catch in Port Phillip Bay is compared from four previous surveys conducted between 1982 and 2000. The proportion of catch from boat and shore anglers remained similar with catches from boat anglers representing about 90% of the total catch (Figure 2). The estimated total annual catch from boat anglers appears to have declined between 1982 and 1995 and increased in 2000, but the larger catch in 2000 might also reflect the complete coverage of the NRIFS or recent recruitment. Total annual catch has averaged 2.5 million fish between 1982 and 2000.

The species catch composition from boat anglers in Port Phillip Bay indicates sand flathead, KGW and snapper have been the three main species from 1982 to 2000 (Figure 2). In the NRIFS, for example, flathead constituted 66% of the catch, KGW 13 % and snapper less than 10%.





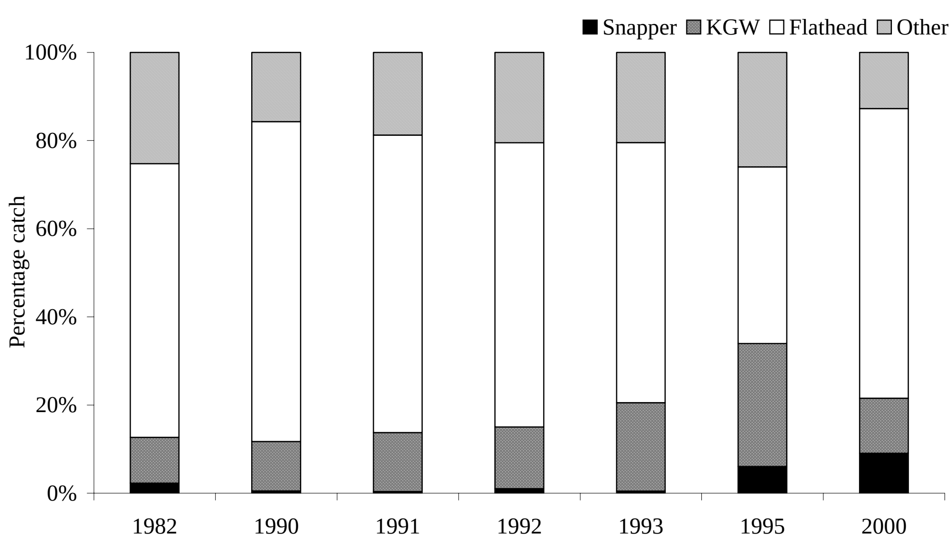


Figure 2. Boat based effort, catch and species composition in Port Phillip Bay

Simulations to compare survey methods

The sample frames and units are different among these three survey methods. The bus route and creel surveys use an area and time sampling frame, while the sampling frame for the NRIFS survey was a list of anglers obtained through a screening survey. The primary sample unit, which can be altered within a sample design, was sample day for the bus route or creel surveys and household for the phone survey. The secondary sample, which is the basis for each angler interview, is the number of fishing parties per sample day in the bus route and creel surveys or the number of recorded events per household in the phone survey.

Monte carol simulations were used to calculate the catch rates from bus route, creel and phone surveys. The simulations used data from three previous recreational fishing surveys of boat angling in Port Phillip Bay. Simulations were repeated for 50 to 650 primary sample units, indicating the number of sample days for bus route and creel surveys or households for a phone survey (with increments of 100). Simulations were repeated for snapper, KGW and flathead, but only the results for snapper are presented here. This approach required an assumption that the sampling frames were the same as the original surveys (ramps and waiting times for the bus route, list of anglers for the NRIFS and ramps for the creel survey). Anglers in the simulations were also assumed to be harvesting the same population, so the probability of a catch and distribution of non-zero catches were considered the same for all survey methods.

The first step in the simulations was to generate a secondary sample for each primary sample. This required allocating the number of fishing parties per sample day in a bus route or creel survey or the number of recorded events per household in the phone survey. The distributions that formed the basis for allocating the secondary samples were established from previous surveys (Figure 3). The number of secondary samples was generally small; 50% of sample days in the bus route and creel surveys had less than eight fishing parties. But 85% of households in the NRIFS had less than eight recorded events per household. In fact, 45% recorded only one fishing trip.

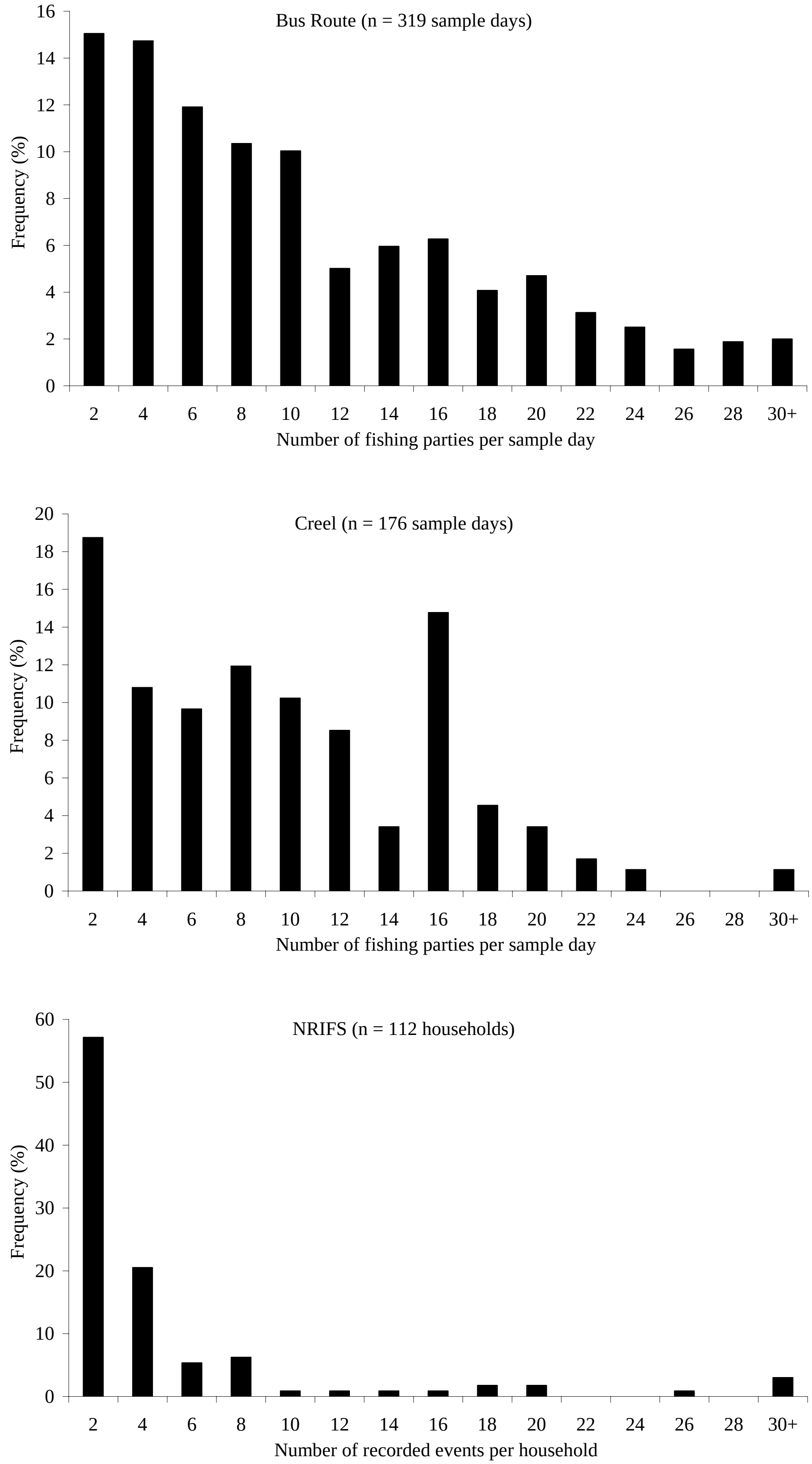


Figure 3. Distribution of secondary sample units for bus route, creel and phone surveys in Port Phillip Bay

The next step was to generate a catch for each interview. This involved firstly determining the likelihood of a catch according to the binary response of zero and non-zero catches (O'Neill and Faddy 2003) for eight species in Port Phillip Bay from the three different survey methods (Figure 4). There were differences between surveys, perhaps due to the survey methods or the different years that they were conducted, but similar trends in catch probability were observed among survey methods. For example, the probability of catching flathead was highest for all surveys, followed by KGW and snapper.

These pooled data from all surveys indicated that anglers in Port Phillip Bay were most likely to catch flathead (with a 51.49% chance). There was 21.69% chance of catching KGW and 13.02% chance of catching snapper. In the simulations, a random number between 0 and 1 was generated and if this was greater than the probability of a non-zero catch then zero catch was recorded, but if the random number was less than the probability of a non-zero catch, then a catch was generated.

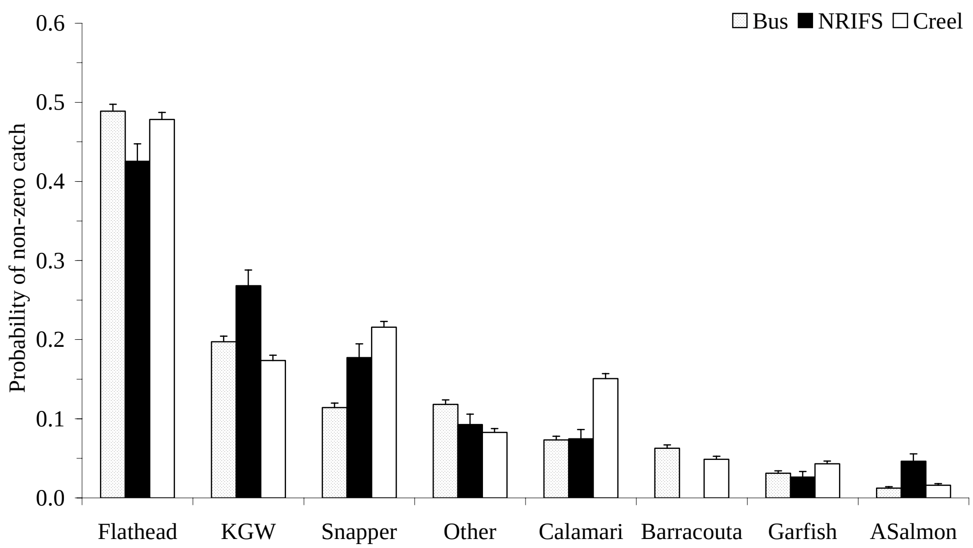


Figure 4. Probability of catching a fish in Port Phillip Bay

The distributions of (non-zero) catches were standardised to catch per angler for each interview; in most cases the average number of anglers was two. The range in catch reflects the maximum bag limit, which was 10 for snapper, 20 for KGW and 30 for flathead. Snapper were mostly caught in small numbers (61% of anglers caught only a single snapper), but 62% of anglers caught up to four KGW and 60% caught up to three flathead (Figure 5).

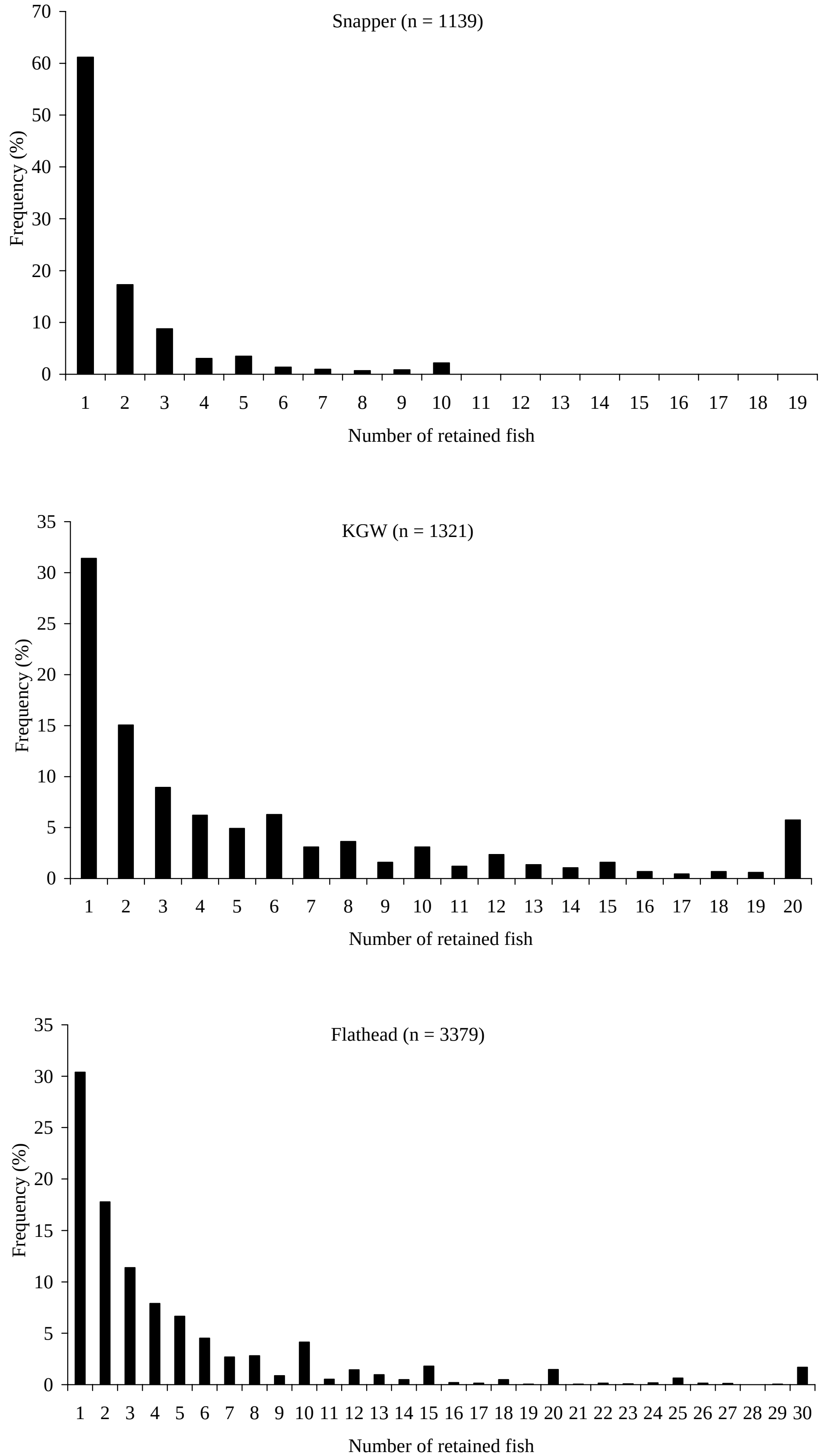


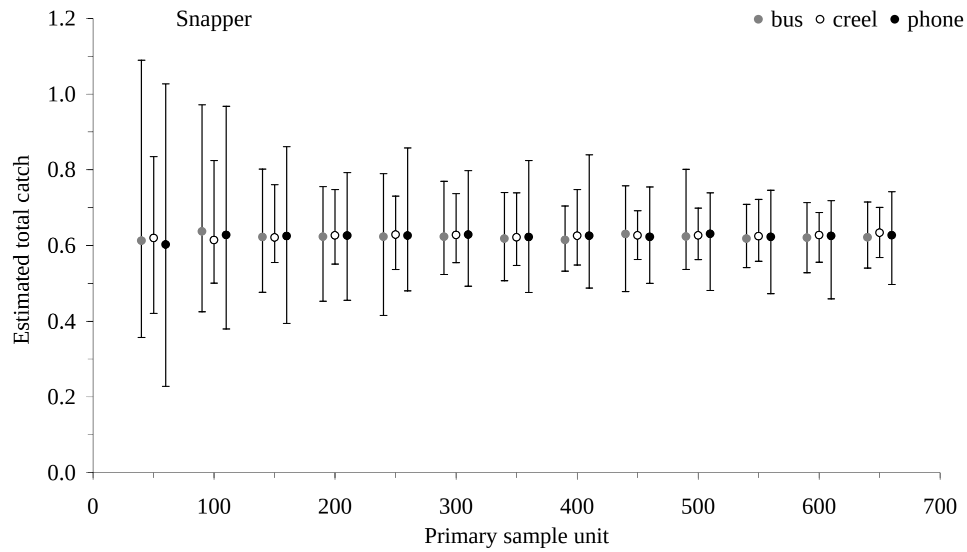
Figure 5. Distribution of non-zero catch of snapper, KGW and flathead in Port Phillip Bay

The simulated catches were repeated for 1000 iterations at each primary sample unit with catch rates estimated for each iteration. The mean catch rates were then estimated for each primary sample unit according to mean of ratios estimator (Jones et al., 1995; Pollock et al., 1997), where the sum of the catch rates for each angler was divided by the total number of anglers. This estimator accounts for the bias associated with roving creel surveys that target anglers whilst fishing where the probability of being sampled is proportional to their trip length (Pollock et al., 1997; Malvesestuto et al., 1978). The mean of ratios estimator can also be appropriate when equal weighting is given to each angler (Malvestuto, 1996) bus route and phone surveys that target anglers after completing their fishing activity with equal probability.

Results

The mean catch rate provides a comparison of the accuracy for a range of primary sample units (Figure 6). The mean catch rates remained constant with increasing sample size indicating the accuracy of the estimated catch rates did not change. But the range in maximum and minimum mean catches is larger for smaller samples indicating lower sample sizes are less likely to accurately estimate catch. These ranges are also different between survey methods, but it should be noted that the primary sample unit is not comparable between survey methods; one sample day is not the same as one household. What this does suggest is that the accuracy of the bus route and creel surveys increases rapidly between 50 and 150 sample days and accuracy of the phone survey improves more gradually between 150 and 250 households.

The standard error of the mean catch provides a comparison of the precision for different primary sample units (Figure 6). The standard error of the mean catch decreased as the number of samples increased. Higher samples had lower standard error and higher precision. The ranges between the lowest and highest standard error of the mean catch also decreased with increasing sample size. This is related to the nature of the survey method where eight or fewer recorded events were observed in 85 % of households in the phone survey, but eight or fewer fishing parties were observed in 50 % of sample days in the bus route and creel surveys.



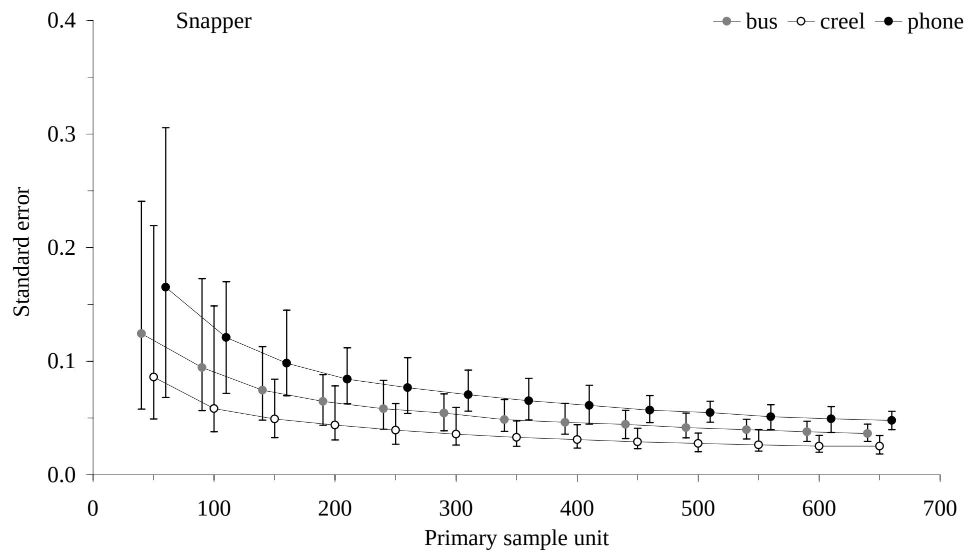


Figure 6. Predicted mean catch rate and standard error of snapper from bus route, creel and phone surveys in Port Phillip Bay

A cost model for the different survey methods, based on expenditure from previous recreational fishing surveys, was developed that was linear (costs increased with sample size), continuous and deterministic (there were no stochastic properties) (Figure 7). Costs were calculated as a combination of fixed and variable costs. The fixed costs for a phone survey ($130 000) were much higher than for bus route and creel surveys (both $50 000), reflecting the work required to establish a good sampling frame for a phone survey; however, the phone survey had much lower variable costs for collecting samples. These were estimated to be about $100 per household, compared to $700 for a sample day in the bus route and $900 for a sample day in the creel survey.

The standard error and cost for the three survey methods were compared to assess the sampling errors relative to the costs of collecting and processing the data (Figure 7). The initial high curve for a phone survey reflects the high fixed costs and low precision of small sample sizes, but the lower variable costs of a phone survey allow the precision and cost to become comparable with bus route and creel surveys. For example, at $190 000, there is a similar precision between 380 households for a phone survey or 190 sample days from a bus route survey. At $240 000, there is a similar precision between 750 households for a phone survey and 190 sample days from a creel survey. The cost effectiveness reaches a point at about $300 000 where the cost of taking additional samples produces minimal further decreases in standard error and has limited potential to increase precision for all survey methods.

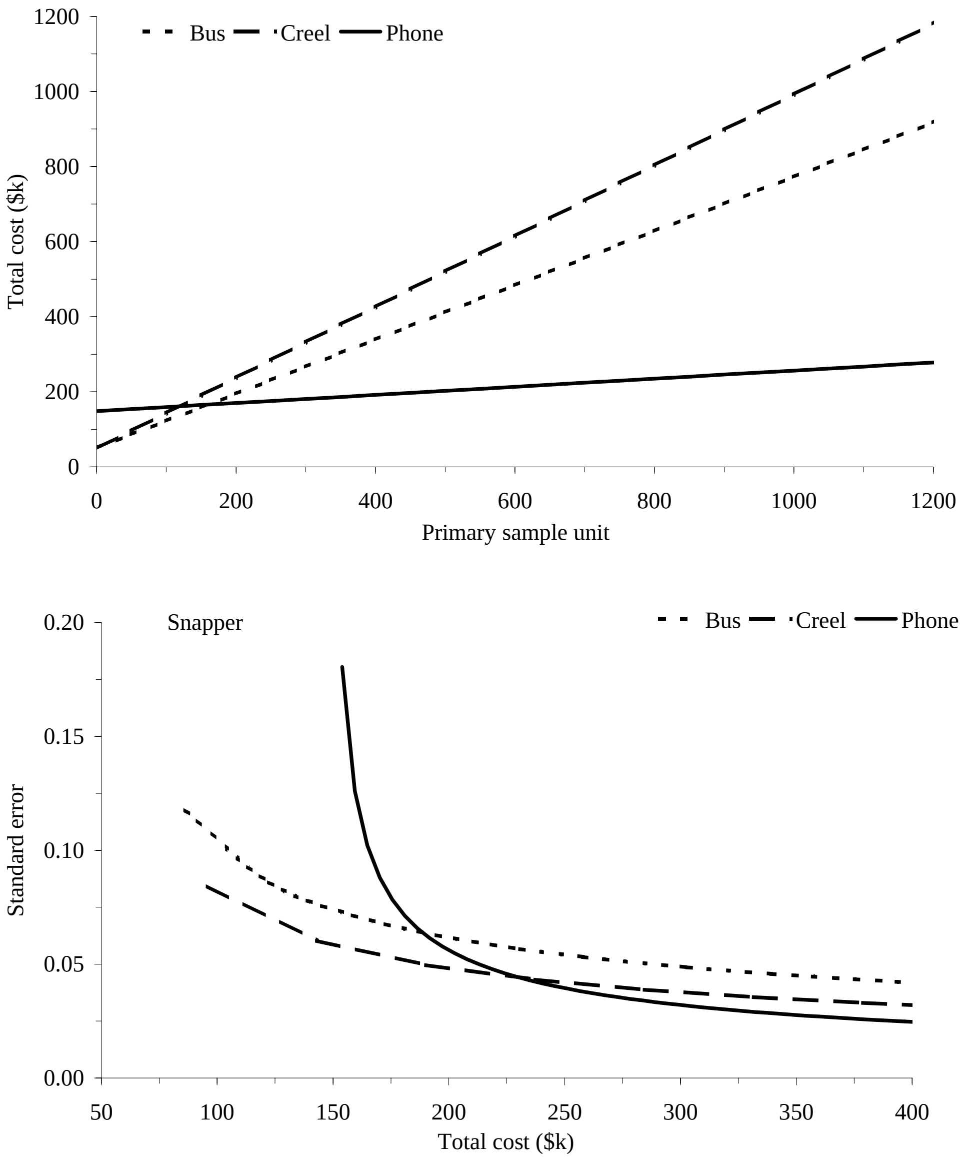


Figure 7. Comparison of total cost and precision for bus route, creel and phone surveys in Port Phillip Bay

Conclusions

The simulations create an estimate of catch rates, but further refinements are required to incorporate sampling errors associated with estimating annual effort. These are possibly higher for an aerial survey compared with the expansion procedure of the phone survey. The creel survey provided lower standard errors for the estimates of catch rate than the Bus Route survey at all levels of expenditure. The phone survey produced the most rapid decreases in standard error with increasing expenditure (but from a higher starting level) and produced the lowest standard errors at higher expenditure levels.

The recreational fishery in Port Phillip Bay is suitable for assessing the use of bus route, creel and phone surveys to estimate recreational catch within a small spatial scale. Ultimately the preferred survey method for estimating the recreational catch may depend on the ability and costs to reduce bias, objectives of the survey and available funds. If the survey objectives are purely to estimate catch by numbers, then a phone survey is most likely to provide this information at the lowest cost, particularly if the costs incurred with establishing a sampling frame can be reduced, for example, by using a database of fishing participants. Survey errors and survey costs are reflections of each other (increasing expenditure reduces uncertainty for all survey methods) and in planning a survey, effort should be directed toward both reducing the errors and producing the greatest usefulness with the funds available.

Acknowledgments

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The WA research angler program - getting volunteers involved in fisheries research

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Abstract

The demand for information about the status of fish stocks is ever increasing. In WA, annual data are required to sustainably manage recreational and commercial fisheries, monitor the allocation of catch between sectors, meet ESD guidelines and assess environmental impacts on stocks or ecosystems. Historically, commercial fishers provided much of these data, with relatively little being obtained from recreational fishers. Hence, the gradual withdrawal of commercial fishing effort from coastal and estuarine waters in WA over recent decades has resulted in less data available for stock assessments. The research angler program (RAP) aims to partly address this problem by increasing the contribution to fisheries research by recreational anglers and other volunteers. Unlike other volunteer support programs, RAP is focused on the collection of scientific data, rather than community education or liaison. RAP is an administrative framework that supports various volunteer-based research projects like angler logbooks, biological sampling, tagging studies and collection of fishing club and competition data. Advantages of RAP include a single continuous point of contact for volunteers who seek information about various research projects and how they can be involved, administrative support for research projects, more cost-effective volunteer administration and higher quality feedback to anglers, which is critical to the success of projects. Some recent WA volunteer-based projects and examples of collected data are discussed.

Introduction

Australian fisheries managers have traditionally relied heavily on data obtained from commercial fisheries to monitor and assess the status of fisheries resources. In many regions, the reallocation of fishery resources away from the commercial sector has caused a reduction in the availability of data for stock assessments. For example, in south-western Western Australia the total number of participants in the commercial estuarine and embayment fisheries has more than halved since the mid-1980s (Anon. 1999). In each case, the loss of commercial fishery data has not been replaced by recreational fishery or fishery-independent data. In the Leschenault Inlet, on the lower west coast, a total cessation of commercial fishing has resulted in virtually no data being available for stock assessments since 2002, despite a continuing need to monitor the status of recreational target species in that estuary. Prior to 2002, commercial fishers in the Leschenault Inlet had provided an unbroken time series of data spanning approximately 50 years.

The declining availability of fishery data has coincidently been accompanied by new environmental management standards that demand a higher quantity and quality of scientific information. The recent adoption of national ESD guidelines (Fletcher et al., 2002) and the current international trend towards ecosystem-based fishery management suggest that the amount of ecological data required by fishery managers in Australia will continue to increase. In addition, a national trend to reallocate fishery resources from commercial to non-commercial sectors will further decrease the availability of fishery data and probably increase the research effort required to monitor future allocations.

To fill the widening 'data gap', fisheries management agencies are increasingly relying on recreational anglers and other volunteers for provision of research data. Limited data on research volunteer participation rates suggest that current contributions by volunteers to fisheries research are substantial in most Australian States and Territories. For example, in 2003/04, volunteers contributed approximately 17 720 hours to finfish research at the WA Department of Fisheries, at an estimated value of $443 000, at $25 per hour (Guidelines for NHT funding applicants 2005). The growth of volunteer-based research has been accompanied by a willingness of the recreational fishing community to assume a greater responsibility for their industry and be more involved in fisheries research programs.

Despite the current and potential value of volunteer-based research and the willingness of volunteers to participate, few formal attempts have been made by fishery agencies to incorporate volunteers into strategic research plans or to develop efficient research volunteer management systems. This failure to formally acknowledge the role of volunteers in research may be partly due to limited awareness by managers of their value and scepticism about the quality of data collected by volunteers.

This paper aims to raise awareness of the significant contribution by volunteers to fisheries research. Some recent examples from Western Australia are included to illustrate the value of volunteer-based research. Finally, this paper outlines a new research volunteer management program recently developed in Western Australia to specifically meet the future needs of volunteers, researchers and other stakeholders.

Example 1. Structured research fishing

Tailor (*Pomatomus saltatrix*) is one of the most important recreational finfish resources in Western Australia (Malseed and Sumner, 2001). On the lower west coast, an estimated 94% of total landings are taken by recreational fishers (Smith et al., 2005).

Since 1994, a volunteer angling survey has been undertaken during summer at a specific location in the Swan River (Ayvazian et al., in prep.). At weekly fishing events, organised by the Department of Fisheries (DoF), volunteers target juvenile tailor aged 0^+ and 1^+ (the age at first capture) using a standardised angling method. All tailor caught are measured and released during a nominated time period in the early evening. The annual abundance of juvenile tailor is then expressed as a mean catch rate, calculated as number of 0^+ and 1^+ tailor caught per angler hour. From 1994 to 2001, DoF also sampled the abundance of juvenile tailor in coastal waters of the lower west coast using seine nets during annual fishery-independent surveys.

A juvenile recruitment index has been derived from the annual volunteer catch rate of juvenile tailor that is significantly correlated with the annual commercial catch rate of tailor in the Swan River two years later (Fig. 1). The correlation is slightly stronger than that between the fishery-independent catch rate and the commercial catch.

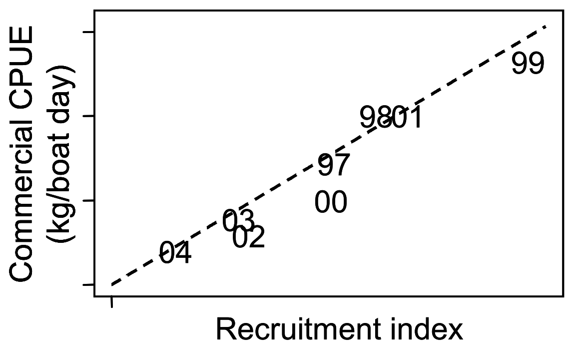

Figure 1. Relationship between annual recruitment index derived from volunteer angler catch rates and annual commercial catch lagged by two years in Swan River, Western Australia, from 1997 to 2004 (adapted from Ayvazian et al., in prep)

Catch rates by volunteer anglers provide a relatively low-cost tool to forecast recruitment levels to the recreational and commercial tailor fisheries on the lower west coast, and to assess stock levels and develop appropriate management options. The volunteer-based data provides a more accurate forecast of future catch levels than fishery-independent data collected by researchers. Furthermore, the involvement of volunteers has greatly assisted in generating public interest in this long-term study, and led to numerous opportunities to promote research and management issues associated with the tailor fishery.

Example 2. Fish tagging and biological data collection

Samson fish (*Seriola hippos*) form large aggregations near Rottnest Island, Western Australia, during summer/autumn and are strongly targeted by recreational game fishers and charter vessels at this time. In 2005, a tag/recapture study commenced to assess the importance of this aggregation and determine the stock structure of Samson fish in south-western WA (FRDC Project No. 2004/051).

In early 2005, approximately 2500 Samson fish were caught, measured, tagged and released in offshore waters near Rottnest Island within a 27 day period. This feat was achieved with the assistance of 270 volunteer anglers and 38 volunteer vessels. The publicity campaign to recruit these volunteers required significant resources, but ensured a high level of awareness in the angling community about the project and ensured a good rate of tag return. Also, the same group of volunteers will be invited to repeat the tagging exercise in 2006.

Twenty Samson fish were recaptured in the first month after tagging, having travelled up to 1000 km (M. Mackie, DoF, unpublished data). No fish were recaptured within the tagging area, suggesting that fish have a very brief residency at Rottnest Island and often undertake extensive migrations immediately after spawning.

Preliminary data from volunteers indicates that Rottnest Island is a key spawning site for a widely distributed, south-western stock of Samson fish and that future management action may be required to protect this aggregation. The similarity of lengths recorded by anglers during tagging and recapture (<1% difference between the first and second measurements for each recaptured fish in the first month after tagging) suggests that that volunteer anglers are able to provide precise biological data.

Example 3. Historic fishing club catch records

The Swan-Canning Estuary is located in the centre of the Perth metropolitan area. It hosts recreational and commercial fisheries and is also subject to various environmental impacts. Fishery stock assessments in the estuary have historically been based on catch and effort statistics and biological samples supplied by commercial fishers. However, between 1975 and 2005 the number of licensed commercial fishing boats declined from 42 to two, due to a voluntary licence buy-back scheme. An alternative source of ongoing data will be required in future to monitor the status of the recreational fishery and assess the general health of the estuary and its fish communities.

The Melville Amateur Angling Club has maintained records of their catch and effort in the estuary since 1987. There is good agreement between commercial and angling club catch trends for species common to both catches. For example, the commercial and angling club annual catch rates of cobbler (*Cnidoglanis macrocephalus*) declined at similar rates between 1987 and 2004 (Fig. 2) (Smith, in prep.). This trend is in agreement with research surveys (I. Potter, Murdoch University, unpubl. data) and anecdotal information suggesting a very large decline in cobbler abundance in the estuary, probably due to the combined effects of breeding habitat loss and fishing pressure. An increase in the average size of cobbler retained by the angling club over the same period is consistent with the possibility of recruitment failure by this stock. Cobbler exist as a discrete breeding stock in the Swan-Canning Estuary and have biological characteristics (low fecundity, aggregating behaviour, specific habitat requirements, etc.) that make them inherently vulnerable to, and slow to recover from, stock depletion.

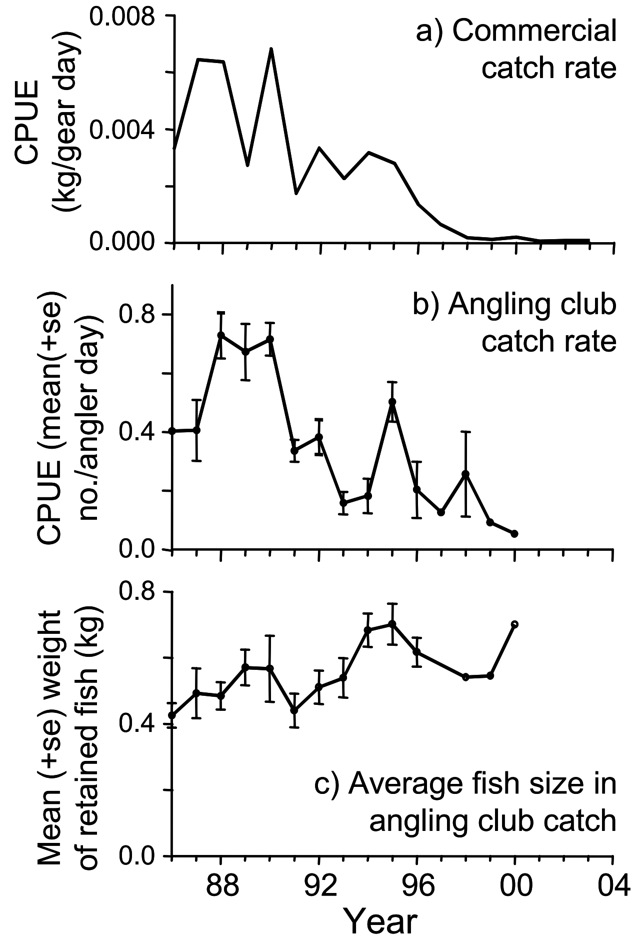

Figure 2. a) Commercial fishery catch rate, b) mean monthly angling club catch rate and c) mean weight of fish retained by angling club, for cobbler (*Cnidoglanis macrocephalus*) caught in the Swan-Canning Estuary, 1987 to 2004

The high level of agreement between commercial and angling club catch rates of cobbler is remarkable, given the relatively small 'sample sizes' in the angling club data. From 1987 to 2004, the angling club typically fished one day per month and caught 0-10 cobbler per month (0-200 per year). In contrast, the commercial catch of cobbler was 0-30 t per year.

Example 4. Daily angler logbooks

In early 2004, a recreational "angler daily logbook' program was launched by DoF in Western Australia. Two types of logbooks now exist – an 'estuary edition' for river/estuary fishing and an 'ocean edition' for ocean beach and offshore fishing. The logbooks allow fishers to record data on all fish and invertebrates caught, including retained and discarded catches. Logbook data includes date, location, fishing effort, fishing gear, bait, species caught, length, fish health, whether retained or discarded, and a reason for discarding. Logbook holders receive regular feedback via a quarterly newsletter, occasional reports and information on the Department's website.

Results from the first year of the program were very encouraging. In 2004-05, estuary logbook holders reported 33 species, including retained and discarded fish. Of these fish, 67% were released and 83% were measured. Sixteen percent of fishing sessions reported ‘nil catch’.

Preliminary data suggested that logbook anglers were providing precise length measurements and accurate catch rate information. For example, black bream (*Acanthopagrus butcheri*) is the main recreational target species in the Swan–Canning Estuary. Logbook data revealed a similar seasonal fishing pattern for this species in both parts of the estuary (different groups of anglers were collecting data in each part of the estuary) (Figure 3a). Catch rates suggested a higher abundance of bream in the Canning River although these fish were of a smaller size than those in the Swan River (Figure 3b). In fact, the majority of fish in the Canning River appeared to belong to a single year class. Overall, the size structure of the catch in both rivers suggested relatively high rates of fishing mortality for bream in this estuary, in agreement with the conclusions of several recent fishery-independent surveys in this estuary (Sarre and Potter 2000, N. Hall, Murdoch University, unpubl. data).

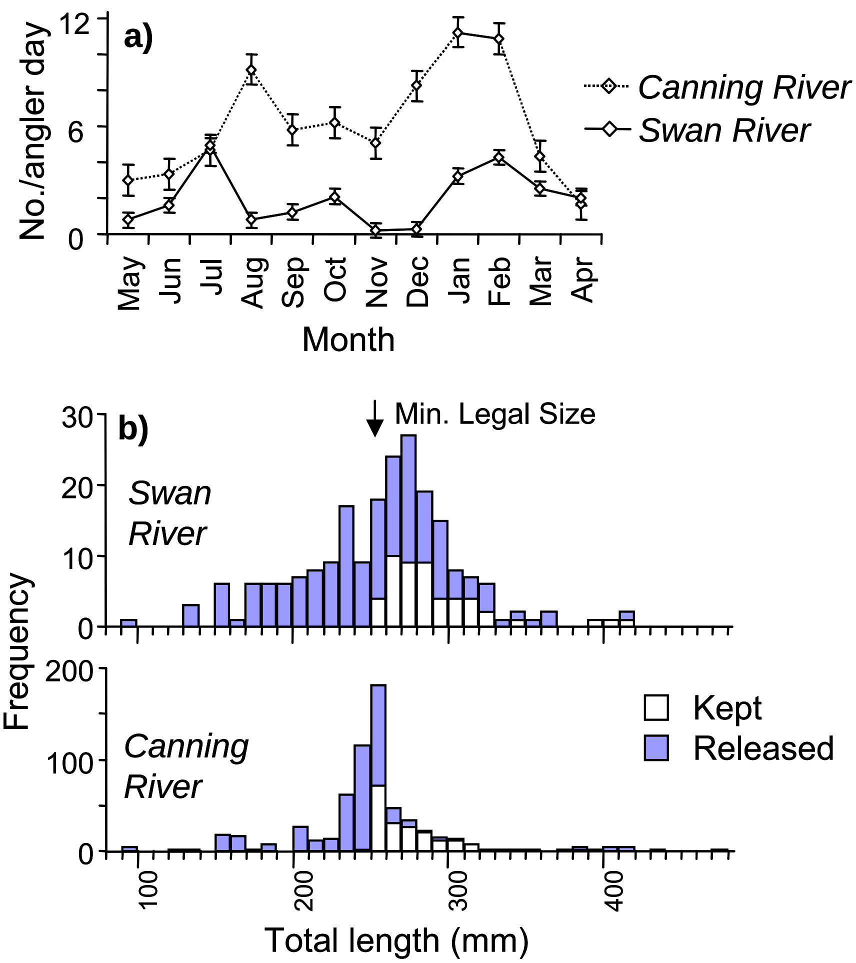

Figure 3. a) Mean monthly catch rate and b) annual length distribution of black bream (*Acanthopagrus butcheri*) recorded by recreational angler logbooks in the Swan-Canning Estuary, 2004-05.

Logbooks have also yielded data that can be applied in a broader environmental monitoring context. For example, a spring outbreak of 'red-spot' disease (epizootic ulcerative syndrome, (EUS)) in black bream in the Canning River was documented by logbook anglers in 2004-05 (Figure 4).

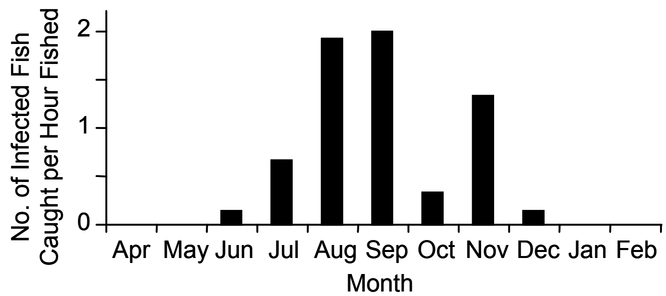

Figure 4. a) Monthly catch rate of black bream (*Acanthopagrus butcheri*) infected by 'red-spot' disease, recorded by recreational angler logbooks in the Swan-Canning Estuary, 2004/5.

Introducing RAP

The declining levels of commercial fishery data and the limited resources available to undertake fishery-independent monitoring have created an essential role for volunteers in Australian fisheries research. With appropriate training and support, volunteers can provide high quality data to meet various fisheries research and monitoring requirements. Volunteer-based research can be a cost-effective investment because it often generates more data, or a better quality outcome, for the same cost as non-volunteer research, especially when the benefits of community involvement (public interest, media coverage, education/promotion opportunities and stakeholder involvement) are considered. Community involvement in research encourages a sense of environmental stewardship and also satisfies the demands of external funding providers for a high level of stakeholder involvement in fishery research and management.

Until recently, fisheries volunteer programs, such as 'Fishcare' and 'Volunteer Fishery Liaison Officer' programs, were focused on community liaison and education and were not designed to administer research volunteers or collect research data. As the role of research volunteers continues to grow, fisheries agencies will need to adopt a more efficient approach to managing research volunteers and develop new methods that allow recreational fishers and non-extractive users to participate more effectively in research. These groups are now major users of many coastal, estuarine and freshwater fish resources and their input will be essential for sustainable management.

In 2004, RAP was established by DoF to cater for the specific needs of research volunteers. RAP will provide an integrated approach to the management of various groups of research volunteers through a single dedicated program with appropriately skilled staff.

From a management perspective, RAP is intended to provide efficient administration of multiple research projects and reduce the total cost of managing large numbers of research volunteers. To date, the management of research volunteers within fisheries agencies has tended to be inefficient, with separate groups of volunteers being variously managed by individual staff/projects. Despite the diversity of project types and management styles, all volunteer-based projects share common tasks, e.g. advertising for recruits, daily queries from volunteers, training, data entry, designing and publishing newsletters, organising seminars, etc. Also, RAP is an ongoing program that will coordinate a succession of short-term volunteer-based projects. This will prevent significant investments in volunteer-based research being lost because it will allow the systems/skills specific to managing research volunteers to be retained and enhanced, rather than being lost and re-built between projects or duplicated between concurrent projects.

An integrated approach to the management of volunteer-based research will enable greater planning and strategic deployment of volunteer effort to focus on meeting fishery management objectives. This will ensure that the cost of volunteer projects is justified and that only outcome-based projects are pursued.

From a volunteer perspective, RAP will provide a convenient contact point for prospective volunteers, a diversity of research projects to participate in, ongoing volunteering opportunities, long-term feedback and a sense of continuity for long-term volunteers that contribute to successive research projects. RAP will improve the communication of research results and information to volunteers and offer better access to information about research volunteering for volunteers and stakeholders.

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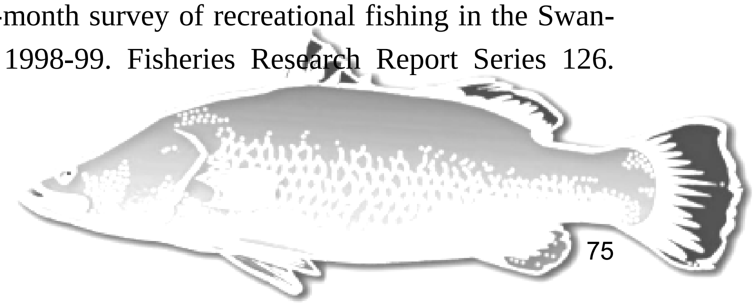

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The indigenous subsistence fishing survey kit: a tool for community-scale monitoring surveys

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Keywords: Monitoring, indigenous subsistence fishing, Cape York, Injinoo.

Abstract

The development of the national indigenous subsistence fishing survey ignited much debate regarding the scale of the monitoring survey. Many prominent indigenous organisations argued that the national survey would not generate community ownership of the survey results, but instead would remove community control of the monitoring data. In this paper I present an alternative approach by discussing the methods of a community-scale monitoring survey conducted in collaboration with the Injinoo Aboriginal Community in far north Cape York, Queensland. The survey adopted the Indigenous Subsistence Fishing Survey (ISFS) kit developed by the Balkanu Cape York Development Corporation, Queensland's Department of Primary Industries and Fisheries (QDPI&F), and the Queensland's Environment Protection Agency. The survey kit was designed to allow individual communities to undertake subsistence monitoring surveys in a manner deemed appropriate by members of the community.

Introduction

The traditional owners of northern Cape York Peninsula will always possess an inherent desire to ensure that the subsistence use of aquatic resources is sustainable (the late Daniel Ropeyarn, Anggamuthi Elder, Injinoo Aboriginal Community, pers comm., 1999). The people of the Injinoo Aboriginal community in far north Cape York possess a strong connection with the sea, and fishing is an important aspect of their culture. However, they feel that their knowledge and interests are not being clearly heard when decisions are made about their resources.

This situation is typical of that which led to the development of the ISFS kit. The survey kit was developed by the Balkanu Cape York Development Corporation, QDPI&F, and the Queensland

Environment Protection Agency. It was adopted in this study as it allows indigenous groups to collect their own fishing data in a manner deemed acceptable by the community. The survey kit is unique in that it starts by asking the community what it is they want to achieve and how the survey should be undertaken.

The present survey was initiated to overcome the lack of data on indigenous subsistence fishing in Australia. Prior to this study, QDPI&F had no reliable data on the type or quantity of aquatic resources harvested by indigenous fishers for traditional or cultural use (Tropical Fin Fish Management Advisory Committee, 1998). The importance of rectifying the situation is most evident in the Fisheries (Gulf of Carpentaria Inshore Fin Fish) Management Plan 1999. The Plan states that:

> 'The provision of a fishery that satisfies the traditional and customary fishing needs of Aborigines and Torres Strait Islanders is to be reviewed if surveys of participation in traditional or customary fishing that are accepted by the Authority show a significant decline in catches or participation'.

Background

In 1994, Elders from the Injinoo Aboriginal community publicly expressed their concern regarding an apparent increase in fishing effort targeting the aggregations of black jewfish (*Protonibea diacanthus*) that form annually in the waters of the northern Cape York Peninsula. This perceived increase in effort had prompted concerns from both the traditional owners of the area, who have custodial responsibilities for that stock, and from QDPI&F, which has statutory responsibilities for managing fisheries in Queensland on a sustainable basis.

Following raised awareness of the concerns held by the traditional owners of northern Cape York Peninsula, the Balkanu Cape York Development Corporation approached QDPI&F on behalf of the Injinoo Aboriginal community and the Injinoo Land Trust. In 1998, they obtained funding from the Fisheries Research Development Corporation (FRDC) to initiate a project to respond to the elders' concerns. The project (FRDC Project 1998/135) examined the biology and harvest of black jewfish (see Phelan 2002a, 2005) and the harvest of all aquatic resources by indigenous subsistence fishers of Injinoo (see Phelan 2002b).

This study represented the first time that RDC had funded research principally devoted to examining an indigenous fishery (Alex Wells, Projects Manager, FRDC, Canberra, pers comm. 1999). This study added to the limited range of indigenous subsistence fishing surveys conducted in Australia, and represents the first comprehensive survey of aquatic resource use in northern Cape York Peninsula.

Method

The present survey was based on the format suggested in the ISFS kit. The survey kit was developed specifically to guide monitoring surveys undertaken in indigenous communities.

Recognising the diverse needs of individual communities, the survey kit states that the methods suggested are provided as a guide only. However, following the format as close as possible is advantageous as this would allow future studies which also adopt the survey kit to be directly compared.

The survey methodology I adopted is outlined in the flow diagram below (see Table 1), and is explained in detail in the pages immediately following. The deviations that I made from the suggested format of the ISFS kit were:

The data sheets of the interview style survey, door to door survey, and monitoring surveys were customised to the needs of the Injinoo Aboriginal community. The needs of the community were identified in the steps prior to the development of each survey.

The decision to conduct either creel or access point monitoring surveys was made after the door to door survey was completed, not before as suggested in the survey kit. In this manner, the door to door survey complemented and confirmed the data profiled in the community survey assessment.

Opportunities were provided for the community to supply feed back at the completion of the surveys so that the success of the project could be gauged for the benefit of future studies.

Table1: Stages in the development and implementation of the ISFS conducted at the Injinoo Aboriginal community.

1. Identification of community needs
↓
2. Induction
↓
3. Community consultation
↓
Updates and feedback
↓
4. Community background survey
↓
5. Interview style survey
↓
6. Community survey assessment
↓
Update and feedback
↓
7. Project team selection
↓
8. Project team training
↓
9. Door to door survey
↓
10. Survey preparation
↓
Update and feedback
↓
11. Monitoring surveys
↓
Update and feedback
↓
12. Data access
↓
13. Findings presented to community
↓
Update and feedback

Identification of community needs (Stage 1)

Community concern for the sustainability of the harvest of black jewfish and other stocks in the local subsistence fishery were first raised publicly in 1994. Following the raised awareness of the concerns of indigenous fishers of Injinoo, the indigenous Subsistence Fishing Survey Kit was identified by Balkanu Cape York Development Corporation as an appropriate tool for profiling the subsistence fishery of Injinoo Aboriginal Community.

Induction (Stage 2)

In late 1998, I was invited by Balkanu Cape York Development Corporation to implement and coordinate ISFS at the Injinoo Aboriginal community. Prior to departing for Injinoo, I stayed in Cairns for three weeks in order to liaise with the agencies that developed the ISFS kit (the Queensland Environment Protection Authority, QDPI&F and Balkanu Cape York Development Corporation). During this period, I also consulted several Injinoo Aboriginal community members and representatives that resided in Cairns.

Community consultation (Stage 3)

Prior to the commencement of the initial surveys, I spent two weeks at Injinoo meeting the community residents and introducing the goals of the study. Although seemingly unproductive in terms of annotated returns, feedback generated at a later stage proved that this period was essential to gaining the understanding and trust of community members. To ensure the continuing support for the study, at all stages in the project's development I consulted the community council's Clerk (Mr Robbie Salee). The Clerk was appointed to the project steering committee to represent the interests of the community.

Community background survey (Stage 4)

The community background survey provided a description of the environment in which the present survey was conducted. The community background survey was conducted in December 1998 and collected information about the community's fishing activities and people's views about fishing. The survey also collected information about the community's fishing area, local fishing issues, fishing history, and the legislative regimen governing Indigenous subsistence fishing. I collected this information through: (1) interviews with representative community fishers and hunters, and (2) a review of all available literature relating to the region. This information was utilised in the planning of future surveys (i.e., the interview style survey, door to door survey, and monitoring surveys).

Interview style survey (Stage 5)

The interview style survey provided the means for the community to design where, when and how they preferred to be surveyed. I conducted the interview style survey in December 1998. Each household in the community was visited and volunteer representatives of the household were interviewed. Representatives were chosen by the residents, but were required to possess a clear knowledge of the subsistence fishing activities conducted by members of the household. Representatives of 44 households contributed to the interview style survey, representing 92% of the 48 households at Injinoo.

Community survey assessment (Stage 6)

The information gained from the community background survey (Stage 4) and the interview style survey (Stage 5) was collated to form a summary termed the ‘community survey assessment’.

Project team selection (Stage 7)

The community survey assessment (Stage 6) revealed that the people of Injinoo strongly preferred members of the local community to monitor their use of aquatic resources. This result had been anticipated in the project budget and funds were made available by FRDC to employ to two survey facilitators per day for seventy survey days. Subsequently, these positions were advertised within the Injinoo Aboriginal community. The ability to identify and local marine and freshwater species, and the ability to work with minimum supervision, were essential prerequisites for these positions. Those people that expressed an interest in the positions were invited to attend a formal meeting in which the applicants were informed in detail of the tasks required of the survey facilitators (Stages 8-11). All applicants were employed on a rotating roster.

Project team training (Stage 8)

The facilitators were trained to conduct the door to door survey (Stage 9) and the monitoring surveys (Stage 11) in accordance with the protocols proposed in the ISFS kit. The survey facilitators were each provided with a copy of the survey kit for their own reference, and were encouraged to provide suggestions to improve each stage of the project's development. I was present at Injinoo for most of the survey days and randomly accompanied team members to check and confirm that the results where a true representation of the local fishery.

Door to door survey (Stage 9)

The door to door survey provided information on when, where, and how the indigenous people of the community conducted subsistence activities utilising aquatic resources. The survey facilitators conducted the door to door survey in January 1999. The survey facilitators visited each household in the community and interviewed volunteer representatives. Over 90% of households in the community were interviewed.

Survey preparation (Stage 10)

The community background survey (Stage 4) and door to door survey (Stage 9) were conducted to provide the baseline information necessary to design the most appropriate method of conducting the monitoring surveys. The ISFS kit recommends that either creel surveys or access point surveys should be adopted to monitor marine and freshwater resource use. With the assistance of the survey facilitators, I determined that creel surveys were the most suitable means of monitoring subsistence fishing at the Injinoo Aboriginal community. Access point surveys were deemed less suitable because: (1) there were at least six access points within the survey area, and (2) a high volume of fishing activity was conducted during the night.

Monitoring surveys (Stage 11)

Monitoring surveys commenced in January 1999 and continued every month through to August 2000**.** Each month, between three to five days was randomly chosen for surveying. Catch details were collected on a total of 70 days. On the days of the survey, two survey facilitators conducted a minimum of three runs per day along a pre-selected route that incorporated the water access points identified in the background survey. Popular shore-based fishing spots between the access sites were also examined en route. The runs of the route were conducted early morning, again at midday, and finally close to dusk. At each site, survey facilitators interviewed all of the groups of fishers encountered. They recorded the type, quantity and length of the aquatic resources harvested, together with details of the fishing trips such as the number of people involved, the location and time of effort.

As the fishers may not have completed fishing at the time the survey facilitators visited the site, the catch data initially recorded may not have accurately represented the day’s total catch. Hence, at each site, the facilitators recorded the names of fishers who had not yet finished fishing for the day. Fishers out in vessels were identified by the cars or trailers at the boat ramp; the small population of the community (~350 people) made this possible. In the intervals between travelling the route, the facilitators visited the homes of these fishers, and whenever possible, the fishers were interviewed and the catch was examined.

Data access (Stage 12)

The data obtained through the monitoring surveys was recorded in a database developed by the agencies that designed the ISFS kit. The database stores quantitative information on the community use of aquatic resources for subsistence purposes. According to the project agreement between the Balkanu Cape York Development Corporation, QDPI&F and FRDC, this information is fully controlled by the Injinoo Aboriginal community.

Results

This paper focuses on the methods adopted in the community scale monitoring survey conducted at Injinoo Aboriginal community. The results of the survey are protected by an intellectual property agreement that ensures the distribution of the data is controlled by the Injinoo Aboriginal community. For further details, please contact Chris Roberts at Balkanu Cape York Development Corporation, Cairns.

Despite the profound lifestyle changes over the last 150 years, the act of utilising aquatic resources for subsistence purposes remained an important component in the lives and culture of the people of the Injinoo Aboriginal community. Harvesting aquatic resources not only fulfilled subsistence needs, but also contributed to the preservation of important cultural practices such as meeting kinship obligations and maintaining connections to country.

The survey revealed that the indigenous subsistence fishery of Injinoo was unique in many ways, and was shaped by the culture and beliefs of the local resource users and custodians, as well as the environment and the species it hosts. By examining the motives, needs, and activities of the indigenous subsistence fishers of the Injinoo Aboriginal community, the distinction between this fishing sector and others becomes evident. Understanding these characteristics will assist natural resources managers to make decisions that are relevant and acceptable to the community.

Conclusion

The Injinoo Aboriginal Community, like many other Australian indigenous communities, is the focus of numerous studies each year. Researchers in almost all these studies ‘fly-in and fly-out’, with the community often gaining little understanding of the study and its findings. The project benefited greatly from my decision to reside in the community (18 months in total) for the initial sampling period. By residing in the community I was able to build a strong personal and working relationship with the residents. With time, this improved our mutual understanding of each others needs.

The ISFS kit was not designed to provide information on the maritime culture of indigenous communities. Yet, I believed that to ignore the strong cultural values associated with the fishery at Injinoo, would underrate its importance to the indigenous groups. Hence, I documented the cultural values that comprise inseparable components of the subsistence fishery. Throughout the final report there are brief insights into the extensive maritime culture of the indigenous people of northern Cape York Peninsula.

As far as possible community members were involved in the design and implementation of this study, as well as in the interpretation of its results. The act of working together on all aspects of the project greatly enhanced the community’s understanding and trust, and hence their willingness to participate. At all stages this study adhered to the protocols established by the Balkanu Cape York Development Corporation for conducting research in indigenous environments (Balkanu Cape York Development Corporation 2005) These were designed to allow individual communities to participate in scientific research in a manner deemed culturally appropriate by the indigenous community.

Prior to the commencement of the surveys, I made a substantial commitment in time meeting the community residents and promoting a two-way discussion of the needs of the project. From feedback generated at later stages this initial consultation was deemed critical to the success of the study. Although unproductive in terms of formal results, this period was essential to: (1) identifying the issues of concern to ensure the relevance of the research, and (2) ensuring the transmission of salient objectives so that the direction of the study was clear to all.

To maintain the high level of community ownership of the project, I consulted with the community at all stages and presented the results in a transparent manner as soon as they became final. At regular intervals, I reported the progress of my studies to the community council’s Clerk, who also represented the interests of the community by serving on the project’s steering committee. The committee guided the progress and direction of the study, and served to ensure the transmission of the results to all stakeholder groups.

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Monitoring of fish communities in deep channel billabongs associated with the Ranger uranium mine, Northern Territory

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Abstract

Fish communities in two deep channel billabongs in the Alligator Rivers Region (NT) have been monitored by visual census using a boat with a customised bow-mounted, perspex-viewing dome. Mudginberri Billabong is located downstream of the Ranger uranium mine, and Sandy Billabong is a control site on a catchment with no mining activity. Fish data for Mudginberri Billabong have been gathered since 1989, while sampling of Sandy Billabong commenced in 1994.

The design of the monitoring program involves the pair-wise comparison of fish community data between Mudginberri and Sandy billabongs using multivariate dissimilarity measures. Shifts in fish community structure have been observed in both billabongs from year to year, with some semblance of 'tracking' of the sites in multivariate space. However, a decline in the paired-site dissimilarity measures over time requires additional analysis so that natural and mining-related changes can be correctly distinguished. Analysis of the datasets has shown that particularly high abundances of Chequered rainbow fish (*Melanotaenia splendida inornata*) in Mudginberri Billabong (and not Sandy Billabong) in the early years of the study are mainly responsible for the elevated paired-site dissimilarity measures in that period. This species undertakes very significant upstream migrations in Magela Creek in the late wet season after spawning and recruitment on the (downstream) floodplain. This paper considers floodplain grass communities, stream discharge patterns, stream solutes and net wet season input of mine site contaminants as possible factors affecting abundance of the chequered rainbow fish in Mudginberri Billabong. The potential and confounding effect of these factors on the ability to detect mine related changes is examined.

Introduction

Environmental monitoring for detection of off-site impacts on surface waters in Kakadu National Park (NT) by the Ranger uranium mine has been conducted since 1994 by the Supervising Scientist Division (SSD) of the Australian Department of Environment and Heritage. The current monitoring program comprises a 'multiple line of evidence' framework, incorporating an array of biological, chemical and radiological monitoring and assessment procedures focussed on the streams adjacent to the mine (Humphrey et al., 1999; van Dam et al., 2002). Studies of fish and macro-invertebrate community structure in streams have been conducted to enable an evaluation of the ecological significance of any impacts. The present study reports results from one of these biological monitoring procedures, namely, the study of fish community structure in deep channel billabongs. As most of the streams in this region cease to flow in the dry season, these permanent water bodies are very important as refuges for fish (Bishop et al., 1990). The main potential threat from mining to aquatic biota is altered water quality resulting from surface runoff and seepage from the mine site. The natural surface waters have very low levels of solutes and are poorly buffered. This enhances the potential for adverse effects from metal contamination.

Kakadu National Park contains the catchments of most of the rivers of the Alligator Rivers Region. Fish diversity, biology and ecology in the region's streams was well documented prior to the present study (Taylor, 1964; Midgley, 1973; Pollard, 1974; Bishop et al., 1986, 1990, 2001), providing a sound basis for development of the current fish community monitoring program. About 50 freshwater fish species occur in the region and around 37 of these species are commonly encountered (Walden and Pidgeon, 1998). The high species richness, high public profile, sensitivity to water quality changes and their high trophic position make fishes in the ARR a valuable and cost effective addition to monitoring long term ecosystem health (Humphrey and Dostine, 1994).

A major feature of the fish community is the large dispersal migrations that occur at the beginning and towards the end of the wet season. At the start of flow in the wet season, fish move from dry season refuges to occupy newly-inundated river channels and floodplain habitats (Bishop and Forbes, 1991). Most species also spawn around this time. In the later half of the wet season, fish migrate from seasonal floodplains and shallow billabongs and move upstream, recolonising refuges for the dry season (Bishop et al., 1995). One effect of the migration pattern is that the diversity and abundance of fish varies greatly at different times of year, even in permanent water bodies such as deep channel billabongs (Bishop et al., 1990). During migration phases, fish numbers can also vary greatly from day to day.

For long-term monitoring where changes from year to year were of concern, it was necessary to choose a sampling time that would be consistent from year to year in terms of fish behaviour. Consequently, in the present study, fish sampling was timed to occur as soon as flow decreased enough to prevent significant upstream migration. Fish species richness is also highest at this time. The long term nature of this study provides valuable information on the natural inter-annual variation in fish community structure in the wet–dry tropics of northern Australia. This information is important for evaluation of ecological significance of observed changes in fish communities and assessment of risks arising from mining and other human activities.

In the current monitoring program, potential mining-related changes in fish communities of Mudginberri Billabong need to be supported by results from other monitoring techniques in a multiple lines of evidence framework. This approach ensures correct inference of mining impact from the Ranger uranium mine.

Methods

Study sites

Fish communities in two channel billabongs were monitored. Mudginberri Billabong is located on Magela Creek (a tributary of the East Alligator River) about 12 km downstream of Ranger uranium mine. This billabong is the first large permanent water body downstream of the mine hence it is likely to provide the best indicator of any mine-related impacts. There are no equivalent billabongs upstream of the mine before permanent refugium pools below the sandstone escarpment, a distance of some 30 km. Sandy Billabong is located on Nourlangie Creek, a major tributary of the South Alligator River system. This catchment is independent of the East Alligator River system and is not impacted by mining. Consequently, Sandy Billabong acts as a control site for Mudginberri Billabong.

The channel billabongs are located in the lowland zone of their catchments, between the high gradient escarpment zone of the Arnhem Land plateau and the large coastal floodplain zone. Riparian vegetation is well developed and dominated by *Pandanus aquaticus* and *Melaleuca* and *Syzygium* spp. trees. The waterholes are well flushed during the wet season and, as a consequence, the stream bed is mostly sand. Macrophytes are sparse or absent except in sheltered shallow areas. In the early dry season, Mudginberri Billabong is 1 km long, 80 m wide with thalweg depth (the line defining the lowest point along the billabong) typically ranging from 2 to 5 m. Sandy Billabong is 2.5 km long, has an average width of 60 m and thalweg depth ranging from 2 m to 8 m.

Water quality in Mudginberri and Sandy billabongs is very similar (Table 1), with the exception of higher uranium concentrations in Mudginberri Billabong. (The trigger value for uranium in Magela Creek requiring further investigation by mine and government agencies is 0.3 μg/L.) The average annual discharge in Nourlangie Creek is almost four times greater than in Magela Creek. Despite the total discharge difference, the flow patterns are very similar and annual discharge between the Magela and Nourlangie creeks are highly correlated ($r = 0.911$, $P < 0.0001$).

Whilst the water in both streams is relatively clear for most of the wet season, high clarity suitable for visual fish counts occurs reliably in the late-wet to mid-dry season period (Table 1).

Table 1. Water quality and hydrological site characteristics of Mudginberri (M) and Sandy (S) billabongs based on spot samples collected in May (N = 4 years, from 2000). Discharge is the average annual discharge since 1994.

Site	pH	Conductivity µS/cm	Turbidity NTU	Magnesium mg/L Mg	Sulphate mg/L SO4	U µg/L	TP mg/L	Discharge ML
M	6.4	18.9	1.48	0.95	0.38	0.019	0.01	390,000
S	6.4	22.0	1.25	1.18	0.16	0.006	0.025	1,535,000

Fish sampling methods

Fish were sampled using a visual census technique to minimise sampling impact on fish in a World Heritage conservation area. Observations were made from a boat to reduce the risk of contact with crocodiles.

The sampling technique used observations of the fish inhabiting the littoral zones along a 50 m transect set parallel to the bank of the billabongs. Typically, the transect was set immediately adjacent to steep banks with dense, over-hanging or submerged pandanus palms and *Melaleuca* trees. Observations were made through the front of a boat with a custom-made, clear, Perspex-viewing dome. The observer lay in the boat with the bow weighted so as to submerge the dome and thereby provide clear vision along and below the surface of the water. A dark cloth covered the observer to prevent glare and reflection on the inside of the dome. The boat was manoeuvred along the transect over 15-25 minute periods. Observers relayed fish counts to a person recording data in a boat positioned close by.

Visual census surveying has been conducted in Mudginberri Billabong since 1989 and in Sandy Billabong since 1994, in the late-wet/early-dry season. On each annual sampling occasion, a transect at each of the five sites was sampled in each billabong. Each transect was surveyed repeatedly (five times) by alternating observers, and the average of these counts was used as the basis for further data analysis. The same transect locations were used each year.

Environmental variables

Habitat structure variables were recorded at all sites at five points, 10 m apart, along each transect. Variables recorded were water depth at 0.5 m and 2 m from the bank, distance of overhanging vegetation from the bank, and percentage cover of submerged vegetation (pandanus roots, tree roots, and aquatic macrophytes), percentage cover of riparian vegetation above and below 2 m height; and relative abundance of submerged logs and branches.

Secchi depth was measured in the middle of each day as an indicator of visibility for fish observation. Water level was measured daily from *in situ* gauge boards located at each billabong.

Water chemistry data in Magela Creek was derived from weekly water quality monitoring records of the Ranger uranium mine and the Supervising Scientist Division. The dominant contaminants associated with Ranger mine waste water discharges are uranium, magnesium and sulphate. Prior to 2000, most of the uranium (U) data from Magela Creek were gathered for compliance purposes by Ranger. These data are generally unreliable, due to contamination and instrumentation problems (i.e. poor detection limits). Therefore, magnesium (Mg) data have been used as a surrogate index for mine waste water contaminant concentrations in Magela Creek over the entire period of the study. Because Mg concentrations are inversely correlated with stream discharge in Magela Creek, the net input of Mg from Ranger was derived from the difference in median wet season concentration between downstream (compliance site) and upstream (control) locations. Median electrical conductivity data from upstream of Ranger were used as an indicator of background solute concentration in each wet season. No comparable data are available for Nourlangie Creek.

Discharge data were obtained from the NT Department of Natural Resources, Environment and the Arts (NRETA). Daily creek discharge data for Magela Creek were sourced from gauging station G8210009, about 7 km upstream of Mudginberri Billabong. Nourlangie Creek data were sourced from gauging station G8200112, about 11 km downstream of Sandy Billabong. Discharge values were calculated for November/December, April and for the whole of the year. Start and cease-to-flow dates were used to determine the length of previous dry season and length of wet season.

Experimental design

From 1989 to 1993 only Mudginberri Billabong was studied. Detection of any impact on fish community structure was then based on changes in the time series of community indices and the correlation with other lines of evidence. In 1994 the control site, Sandy Billabong, was added to the program. This has enabled the application of a BACI-P design (Before-After Control-Impact, Paired sites) (Stewart-Oaten et al. 1986, 1992). The monitoring objective is attained through a comparison of paired site, ‘difference’ data from a ‘baseline’ time series collected *before*, with data obtained *after* suspected contamination by mine waste water, or some other ‘event’ or particular period of interest. In the strict sense of pre-mining, the ‘Before’ term from BACI has no validity, because no comparable fish data from channel billabongs was gathered prior to 1979-82, when initial site disturbance and mining commenced. One of the important assumptions of the design is that the natural differences evident between any pair of sites should remain relatively constant in response to similar natural environmental variations.

The formal hypothesis is:

$$H_o: (C\text{-}I)_{Before} = (C\text{-}I)_{After}$$

where ‘C’ is the control site and ‘I’ is the exposed or impact site. The difference value, (C-I), can be derived from univariate parameters (eg abundance, taxa number), or multivariate dissimilarity measures derived from community data.

Dissimilarity measures reduce the differences between many different species of two sample or site communities to a single value. Comparison of billabong fish responses in this way has benefits over univariate measures in that dissimilarity represents and integrates the entire community response (Clarke, 1993). Community dissimilarity values have been demonstrated as a statistically powerful approach to detecting mining impacts (Faith et al., 1991, 1995).

The BACI-P test provides evidence of a change in community structure in one or both of the locations of interest when a significant difference is found to occur between a time-series of paired-site difference or dissimilarity measures (using, for example, a Student’s *t*-test). Further investigation is required to assess whether the change is associated with the ‘impact’ site and, if so, whether this is due to dispersed mine-waste waters that have reached Magela Creek, or whether the change is associated with other natural or anthropogenic factors. Inference about mining impact needs to be drawn from all other lines of evidence available from the monitoring program.

Statistical analysis

Correlation (Pearson's) and linear regression were conducted using the data analysis tools in Microsoft Office Excel 2003. One-way analysis of variance (ANOVA) and 2-sample *t* tests were conducted using Minitab release 14.13.

Multivariate analysis was conducted using the statistical package PRIMER (V5) (Clark and Gorley, 2001). Fish community data were analysed using the Bray-Curtis dissimilarity measure using a log(x+1) transformation to down-weight numerically dominant fish species. Species that were only represented once in the data set were excluded from the calculation. For each year of sampling, dissimilarity measures were calculated for five independent pairs of site comparisons of community structure data between billabongs (i.e. Mudginberri versus Sandy). The site ‘pairs’ were selected using a random, without-replacement procedure. The first pair for each year was selected at random from 25 possible pairings. Successive pairings were selected from a reduced array of options (i.e. 25, 16, 9, 4 and 1).

Multi-dimensional scaling (MDS) ordination was used to illustrate patterns in fish community structure.

The ANOSIM (2-way) function was used to analyse for significant differences in fish community patterns by the factors, ‘year’ and ‘billabong’.

The SIMPER routine was used to examine fish species contributing to the dissimilarities between and within Mudginberri and Sandy billabongs.

The BIOENV procedure was used to calculate the smallest subset of the 14 measured environmental variables that explained the greatest percentage of variation in the fish community dissimilarity pattern. The procedure uses Spearman rank correlation to compare the multivariate patterns in environmental data with those of fish community data. The maximum number of combination correlates was restricted to five to prevent long, meaningless correlating combinations. Environmental habitat variables used are listed above. Water chemistry variables were not included in the BIOENV analysis because of lack of equivalent data from Nourlangie Creek.

Results and discussion

Biodiversity

Since 1994, 30 species have been recorded in Mudginberri Billabong and 29 species recorded in Sandy Billabong with a combined total of 34 species (Table 2). Over this time, the technique has captured the majority of species expected in the billabongs from earlier studies (Bishop et al. 1990). This earlier work recorded only 20 species in Mudginberri from 6 samples obtained by netting.

However, another 10 species were recorded in samples from other locations on Magela Creek. Nourlangie Creek contains at least 3 species not found in Magela Creek. The catfishes *Anodontiglanis dahli* and *Arius graeffei* were seen only rarely while the Archerfish, *Toxotes lorentzi*, was not recorded in the present study.

Clearly the visual census procedure has provided a thorough species inventory of the fish community. However, it is biased against nektonic species such as ariid catfish (*Arius leptaspis*), plotosid catfish (*Neosiluris ater*), bony bream (*Nematalosa erebi*), and ox-eye herring (*Megalops cyprinoides*) which are more abundant in the open central waters of the billabongs.

Total fish density averaged 751 and 545 per 50 m transect in Mudginberri and Sandy billabongs, respectively (Table 2). More than 95% of the average fish abundance was derived from 6 small to medium-sized fish species. With one exception (mouth almighty, *Glossamia aprion*), these are all schooling species and this enhances their detection by visual census. Of the six species, the fly-specked hardy head (*Craterocephalus stercusmuscarum*) was the most abundant species in both billabongs and occurred in very similar densities: 397/50 m in Mudginberri and 357/50 m in Sandy. The chequered rainbow fish (*Melanotaenia splendida inornata*) was much more abundant in Mudginberri (255/50 m) than in Sandy (47/50 m) (Table 2). Rainbow fish were largely responsible for the higher overall abundance of fish recorded in Mudginberri. Penny fish (*Denariusa bandata*) were also more abundant in Mudginberri. Conversely, the abundances of Glassfish (*Ambassis agrammus* and *A. macleayi*), Banded grunter (*Amniataba percoides*) and mouth almighty (*G. aprion*) were greater in Sandy Billabong.

Fish community structure

MDS ordination analysis of the fish samples required three dimensions to obtain an acceptable stress level below 0.2 (Clark, 1993). The ordination pattern of the fish samples (Figure 1A) using axes 1 and 3 shows a separation of Mudginberri and Sandy fish communities throughout the sampling period, reflecting the differing proportions of the dominant fish species in each billabong. This was supported by the 2-way ANOSIM test which showed significant differences between billabongs (Global $R=0.74$, $p<0.001$) and between years (Global $R=0.65$, $P<0.001$).

Table 2. Mean abundance (no. fish/50 m) of fish species from Mudginberri and Sandy billabongs for the period 1994 to 2005, using visual boat census technique

Scientific Name	**Common name**	**Mudginberri**		**Sandy**	
		Abundance	% total	Abundance	% total
Craterocephalus stercusmuscarum	Fly-specked hardy head	397.2	52.9	356.8	65.4
Melanotaenia splendida inornata	Chequered rainbow fish	255.4	34.0	47.4	8.7
Ambassis spp.	Glassfish (*A. agrammus, A. macleayi)*	49.1	6.5	72.2	13.2
Amniataba percoides	Banded grunter	15.1	2.0	41.6	7.6
Denariusa bandata	Penny fish	12.9	1.7	2.16	0.4
Glossamia aprion	Mouth almighty	8.5	1.1	10.1	1.9
Nematalosa erebi	Bony bream	2.163	0.288	2.323	0.426
Leiopotherapon unicolor	Spangled grunter	2.050	0.273	0.713	0.131
Toxotes chatareus	Archer fish	2.037	0.271	4.927	0.904
Glossogobius spp.	Goby (*G. giurus & G aureus*)	1.470	0.196	1.260	0.231
Strongylura krefftii	Longtom	1.3	0.173	0.407	0.075
Syncomistes butleri	Sharp-nosed grunter	1.02	0.136	2.837	0.520
Lates calcarifer	Barramundi	0.987	0.131	0.283	0.052
Hypseleotris compressa	Carp gudgeon	0.563	0.075	0.0	0.0
Hephaestus fuliginosus	Sooty grunter	0.473	0.063	0.797	0.146
Neosilurus ater	Black catfish	0.337	0.045	0.780	0.143
Oxyeleotris spp.	Sleepy cod (*O. lineolata & O. selheimi*)	0.217	0.029	0.103	0.019
Pingalla midgleyi	Black-anal-fin grunter	0.15	0.020	0.143	0.026
Liza spp.	Mullet spp	0.103	0.014	0.133	0.024
Neosiluris hyrtlii	Hyrtl's catfish	0.097	0.013	0.013	0.002
Scleropages jardini	Saratoga	0.043	0.006	0.090	0.017
Melanotaenia nigrans	Black-striped rainbowfish	0.030	0.004	0.093	0.017
Mogurnda mogurnda	Purple-spotted gudgeon	0.023	0.003	0.003	0.001
Oxyeleotris nullipora	Dwarf gudgeon	0.023	0.003	0.0	0.0
Megalops cyprinoides	Ox-eye herring	0.013	0.002	0.017	0.003
Arius leptaspis	Salmon catfish	0.007	0.001	0.017	0.003
Pseudomugil tenellus	Delicate blue-eye	0.007	0.001	0.0	0.0
Scatophagus argus	Spotted scat	0.003	>0.001	0.0	0.0
Redigobius bikolanus	Speckled goby	0.003	>0.001	0.0	0.0
Craterocephalus marianae	Mariana's hardyhead	0.0	0.0	0.067	0.012
Pseudomugil gertrudae	Spotted blue-eye	0.0	0.0	0.003	0.001
Anodontiglanis dahli	Toothless catfish	0.0	0.0	0.02	0.004
Arius graeffei	Blue catfish	0.0	0.0	0.003	0.001
Total		751.26	100.0	545.277	100.0
Total no. taxa	34	30		29	
Species density (no./50m)		12.9		13.2	

The changes in position of the fish communities in ordination space in different years are shown in Fig.1B, this time using axes 1 and 2. For simplicity, the centroid of the five replicate sites for each billabong and year has been displayed. Changes through time are indicated by arrows. The patterns show some evidence of 'tracking' with the two communities often moving in the same direction between years. Of note, there is no evidence of a trend of movement in any particular direction that might indicate some long term change. This is emphasised by the closeness of the 2005 samples to the 1994 samples.

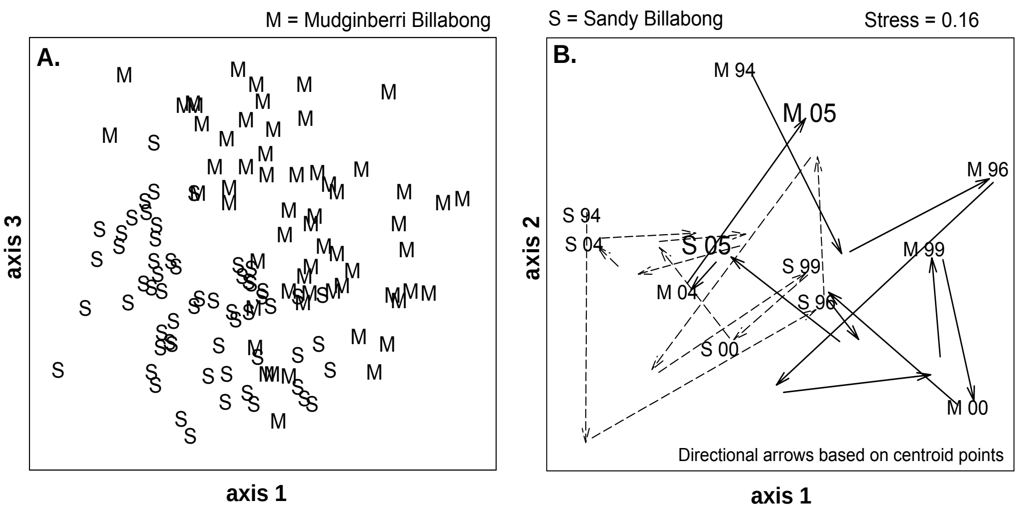

Figure 1. MDS ordination plot of fish communities sampled by visual census in Mudginberri (M) and Sandy (S) billabongs, 1994 to 2005. Patterns are based on three dimensional MDS.

A: Axis 1 and 3 depicts separation in fish communities over all samples;
B: Axis 1 and 2 shows centroid points for each year to depict changes of fish community structure in ordination space from year to year. Figure symbols indicate billabong and year of sampling (e.g. M 94 = Mudginberri 1994 etc).

The average dissimilarity between Mudginberri and Sandy billabongs was 27.4% (SIMPER analysis, Table 3). Eleven fish species contributed 79.2% to the dissimilarity between these billabongs. The fish species having the greatest influence on the between-billabong dissimilarity were species with proportionately large numerical differences. The chequered rainbow fish had the greatest influence with 12.6% followed by Glassfish 9.9% (Table 3). Overall, the fish communities of Mudginberri and Sandy billabongs have very similar species compositions (Table 2). Their relatively low dissimilarity to one another combined with significant separation in ordination space indicates the sensitivity of the visual fish census procedure in identifying small but consistent differences in fish community structure, as well as the sensitivity of the multivariate procedures used.

The average dissimilarity amongst years within Mudginberri and Sandy billabongs was 25.8% and 22.3%, respectively. This relatively low dissimilarity within each billabong was primarily influenced by the same five numerically-dominant fish species (fly-specked hardy-head, chequered rainbow fish, banded grunter, glassfish, and mouth almighty). The most abundant species, the fly-specked hardy-head, in contrast to the between-site dissimilarity, had the greatest contribution in both billabongs (28.6% in Mudginberri and 27.4% in Sandy). The low average dissimilarity amongst years and high proportion of the dissimilarity explained by the same five fish species suggests that, even though fish abundance was down-weighted by log transformation, yearly variation in fish community structures was driven primarily by abundance variations in the five numerically dominant fish species and probably by the same environmental factors.

Table 3. Contribution (%) of fish species to the average dissimilarity between (M v S) and within billabongs derived from SIMPER analysis. Only species making a contribution greater than 5% to the average dissimilarity are included. M indicates Mudginberri billabong and S indicates Sandy billabong.

		% contribution		
Scientific Name	**Common name**	**M v S**	**M**	**S**
Melanotaenia splendida inornata	Chequered rainbow fish	12.6	22.1	15.0
Ambassis spp.	Glassfish (*A. agrammus, A. macleayi)*	9.9	13.2	13.1
Denariusa bandata	Penny fish	8.3	*	*
Amniataba percoides	Banded grunter	8.2	12.0	17.8
Toxotes chatareus	Archer fish	7.0	*	5.9
Syncomistes butleri	Sharp-nosed grunter	6.3	*	*
Nematalosa erebi	Bony bream	5.9	*	*
Craterocephalus stercusmuscarum	Fly-specked hardy head	5.8	28.6	27.4
Leiopotherapon unicolor	Spangled grunter	5.2	*	*
Glossogobius spp.	Goby (*G. giurus & G aureus*)	5.0	*	*
Glossamia aprion	Mouth almighty	5.0	9.4	9.7
Total % contribution		79.16	85.3	88.9
Average dissimilarity		27.36	25.77	22.34

Comparison of measured habitat and hydrological variables for Mudginberri and Sandy billabongs with fish communities using the BIOENV procedure only explained a very small proportion of the fish community dissimilarities. The best Spearman rank correlation obtained was 0.325, based on four variables: % macrophyte cover, % riparian vegetation cover, length of previous dry season and annual creek discharge. The individual correlation coefficients for these variables were: length of previous dry season, 0.236; creek discharge, 0.187; % macrophyte cover, 0.14; and % riparian vegetation cover, 0.105. The low correlation of habitat structure variables is not surprising as the sites are structurally very similar and have changed very little over the sampling period.

Impact detection

The mean paired-site dissimilarity between Mudginberri and Sandy billabongs has ranged from 19.2% in 2003 to 33.7% in 1995. The consistently low dissimilarity value indicates the two communities have remained quite similar to one another. However, there has been a decline in this dissimilarity over the study period (Figure 2) which, in the absence of putative impact, violates an assumption of the BACI-P experimental design that requires the difference (or dissimilarity) values to be independent of one another. Linear regression of dissimilarity against time indicated this trend to be highly significant ($R^2 = 0.20$, $p<0.001$). In particular, however, the time series indicates that a more sudden change occurred in 1999 (Figure 2). There are significant differences in dissimilarity between 1994 and 1998 (one-way ANOVA, $p = 0.013$) but no significant trends. While no significant differences or trends have occurred between 1999 and 2005, the dissimilarities prior to 1999 are significantly higher than the post 1998 samples. A (BACIP) t-test using the time periods 1994-1998 as *before* impact data and 1999-2005 as *after* impact data was significant (DF 48, t = -4.49, p > 0.001).

Significant differences from 1994 to 1998 could indicate natural fluctuations in the dissimilarity value. However, the sudden decline in 1999 and significantly lower values from 1999 to 2005 could indicate some impact from mining unless it can be explained by other factors, unrelated to mining. Other lines of evidence from the monitoring program (water chemistry, creek-side toxicity testing, macro-invertebrate communities, fish communities in shallow lagoons) showed no evidence of a mining impact on Magela Creek in the 1999 wet season. The issue of potential mining impact is further examined below. Consequently, the apparent abrupt decline in dissimilarity in 1999 is concluded to be natural variation around the more general long-term decline.

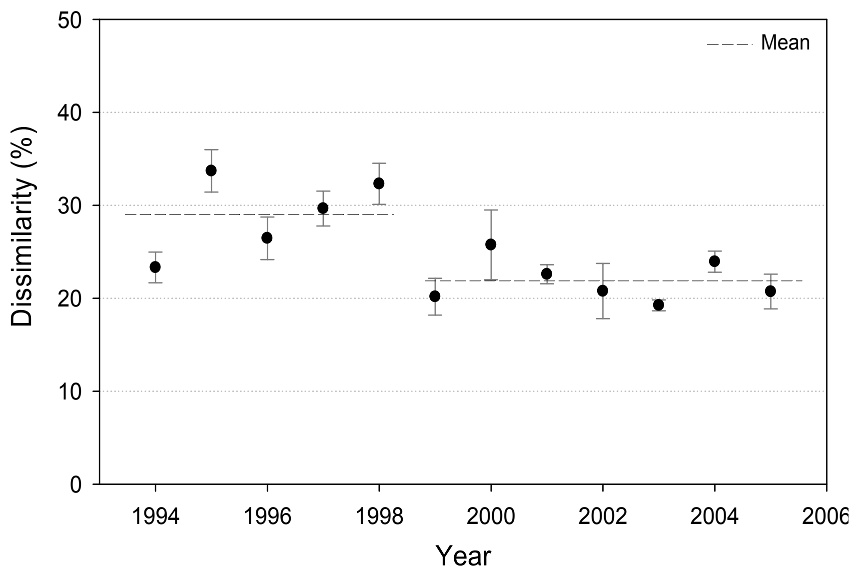

Figure 2. Paired control-exposed dissimilarity values calculated for community structure of fish in Mudginberri (‘exposed’) and Sandy (‘control’) billabongs. Values are means (± standard error) of the five possible (randomly-selected) pair-wise comparisons of transect data between the two billabongs. Dashed lines indicate the means of the annual values for the two time periods, 1994-1998 and 1999-2005.

SIMPER analysis showed that chequered rainbow fish and glassfish had the greatest influence on the between-billabong dissimilarity values. Re-analysis of the dissimilarity values for Mudginberri and Sandy billabong fish communities, with these species removed, enabled their influence on the correlation and regression analysis of the dissimilarity decline over time to be determined (Table 4). Removal of each species reduced both the level of correlation and the significance of the decline in the dissimilarity measure, although removal of rainbow fish had the greatest effect (R^2 reduced to 0.07, p = 0.02). Removal of both fish species resulted in a non-significant result (R^2 0.03), p = 0.20). Populations of rainbow fish were examined subsequently in more detail since this species had the greatest influence on the between-billabong dissimilarity (Table 3) and the decline in the dissimilarity values over time (Table 4).

Table 4. Influence of numerically-dominant fish species in channel billabongs upon correlation and regression results for paired-site dissimilarity value and time (1994 to 2005)

	Correlation (ρ)	Regression parameters	
		R^2	P
All taxa	-0.45	0.20	0.0003
Glassfish species removed	-0.36	0.13	0.005
Chequered rainbow fish removed	-0.31	0.07	0.02
Both taxa removed	-0.17	0.03	0.20

Investigation of the abundances of chequered rainbow fish in Mudginberri billabong (since 1989) and Sandy billabong (since 1994) showed considerable variations amongst years (Figure 3). In Mudginberri, densities ranged from 1341/ 50 m in 1996 to 23/ 50 m in 1997, and in Sandy billabong from 6.6/ 50 m in 1995 to 149/ 50 m in 1997. In 1996, exceptionally high abundances of chequered rainbow fish were recorded in Mudginberri billabong. For this year, sampling was conducted relatively early in the wet-dry recessional flow period and resulted in observations being conducted during a late migration of fish upstream through the billabong. Normally, sampling commenced after fish migration had ceased. Regression analysis of chequered rainbow fish with the removal of the anomalous 1996 data showed a significant decline in Mudginberri billabong since 1989 (r = 0.74, R^2 = 0.547, p = 0.001). There was no such decline in Sandy billabong (r = 0.14, R^2 = 0.019, p = 0.69).

Consequently, it is evident that a decline in the abundance of rainbow fish in Mudginberri billabong, without a corresponding decline in Sandy billabong, has largely been responsible for the decline in dissimilarity of the fish communities between the two sites.

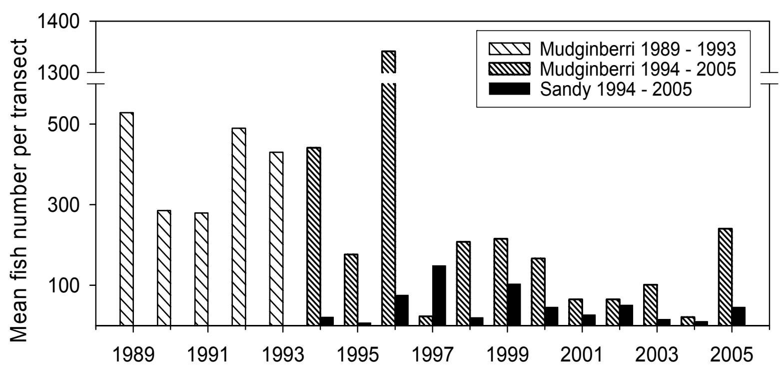

Figure 3. Relative abundance of chequered rainbow fish in Mudginberri and Sandy billabongs from 1989 to 2005

Environmental correlates of rainbow fish abundances

It is important to ascertain whether the decline in chequered rainbow fish in Mudginberri billabong is related to mining activity or not. Possible causes of the decline were sought from correlation with the natural environmental factors of annual discharge, length of previous dry season and natural water solute concentration of Magela Creek (as measured using electrical conductivity, EC, as a surrogate). Wet season concentrations in Magela Creek of solute characteristic of mine waste waters were used to infer possible mining impact (Table 5).

Table 5. Environmental correlates of rainbow fish abundance in Mudginberri billabong, 1989–2005 with 1996 data omitted

	Correlation	Regression parameters	
	(ρ)	R^2	P
Wet season stream discharge	-0.56	0.31	0.03
Wet season solutes	0.66	0.44	0.005
Length of previous dry season	0.57	0.32	0.021
Mine contaminants: Net wet season mine input of Mg (median, mg/L)	0.20	0.04	0.45

Natural environmental factors - length of dry season

The relationship between length of previous dry season and Mudginberri rainbow fish abundances (Table 5) was a highly significant, positive correlation ($r=0.57$, $R^2=0.32$, $p = 0.021$). The coastal floodplain of Magela Creek is a major source of recruitment of this species for upstream dispersal migrations (Bishop et al., 1995). This correlation may indicate that reduced drying of the floodplain (i.e. shorter dry season) reduces ensuing wet season production on the floodplain nursery zone for this species. Such a relationship would be consistent with the Flood-Pulse concept of Junk et al., (1989).

However, understanding the responses of fish communities to the processes involved in the flood pulse concept is complex (Welcomme and Halls, 2001; Welcomme and Halls, 2004) as responses can vary due to the size and duration of flooding, connectivity of systems and presence and intensities of fire on the floodplain (Junk and Wantzen, 2004). In this instance, the challenge may be to explain the absence of a similar correlation between fish communities of Sandy billabong and similar length of previous dry season data for Nourlangie Creek (data analysis results not shown here). However, and for possible reasons not discussed further in this paper, abundances of rainbow fish in Sandy billabong is generally much less than that in Mudginberri (Figure 3) and this low ‘signal’ makes it difficult to draw too much from this analysis.

Natural environmental factors - stream discharge and natural water quality

There has been a general increase in wet season rainfall and associated stream discharge in Magela Creek during the study period. Although rainfall declined from 2001 to 2005 it was still higher than in the early 1990s. Consequently, total wet season discharge in the creek is negatively correlated with rainbow fish numbers in Mudginberri billabong (Table 5).

How higher discharge *per se* would result in lower fish numbers is not immediately clear. However, greater discharge volumes result in greater dilution of wet season surface waters and their solute concentrations. Median wet season values of electrical conductivity (EC) of Magela Creek waters upstream of Ranger were used as a surrogate measure of solute concentrations. Not surprisingly, median wet season EC in Magela Creek is significantly and positively correlated with rainbow fish numbers (Table 5).

Magela Creek surface waters are extremely soft and poorly-buffered and this was accentuated at high wet season flows in the creek. The fry of both chequered rainbow fish (Humphrey, 1988) and the congener, the black-lined rainbow fish used in the SSD's creek-side testing program (Supervising Scientist 2005, Section 2.2.3), exhibit reduced survival when exposed to creek waters during high flow events. It is, therefore, possible that survival of early life stages of rainbow fish is reduced in years of high discharge.

Anthropogenic factors – floodplain grass expansion

A number of grass species have rapidly expanded in range and density on the Magela floodplain, due partly to removal of feral buffalo that once grazed on these grasses and acted as a form of control. A particularly aggressive species is the exotic Para grass (*Urochloa mutica*) which is currently expanding its area of coverage at 14% annually. Without management, it could dominate the floodplain in 15 to 20 years. With the presence of satellite patches of Para grass, this could be a conservative time frame (Bayliss, 2006). The recent rapid expansion of Para grass on Magela floodplain corresponds to the period of decline of chequered rainbow fish in Mudginberri billabong. It is quite possible that the expansion of this and other exotic and native grasses (especially native *Hymenachne*) has had some adverse effects on recruitment of rainbow fish.

Impacts of both Para grass and native *Hymenachne* on floodplain biota were studied by Douglas et al., (2002). Stands of both these grasses contained fewer fish species and lower fish abundance than areas of more open vegetation dominated by wild rice (*Oryza meridionalis*). Consequently, any increase in the area covered by either Para grass or Hymenachne could have adverse effects on the recruitment of fish that utilise floodplain habitats in the wet season into dry season refuges in floodplain billabongs and upstream channel billabongs.

Unfortunately, there is no comparable data on possible grass expansion for the floodplain of Nourlangie Creek downstream of Sandy billabong.

Anthropogenic factors – mine waste water contamination

Magnesium (Mg) data from the Magela Creek monitoring program was used as a surrogate measure of concentrations of mine waste water contaminants in Magela Creek. Correlation and regression analysis indicated no significant relationships between rainbow fish abundance and mine waste water input into Magela Creek (Table 5).

Consequently, there is no evidence of mining impact involved in the decline in abundance of rainbow fish. This is not surprising as concentrations of U and Mg in Magela Creek arising from mine waste water discharges are at least two orders of magnitude lower than those known to adversely affect larval fishes, including in the case of U, chequered rainbow fish (Supervising Scientist 2004, Section 3.4.1 and Supervising Scientist 2005, Section 3.4).

Long-term monitoring of macro-invertebrate communities in Magela Creek and of fish communities in shallow lowland billabongs downstream of the Ranger uranium mine are also employed in the SSD's stream monitoring program for Ranger. Neither component has shown community changes that indicate an influence of mining (Supervising Scientist, 2005).

Conclusions

This study has emphasised the importance of using multiple lines of evidence in monitoring for detection of environmental impacts. Whilst changes in some target indicators may be detected, supporting evidence from other indicators and a sound understanding of the influence of other natural and human environmental factors on indicators are necessary to have confidence in inferring an adverse environmental impact.

Since mining activities commenced at the Ranger mine in 1979, changes unrelated to mining have occurred in stream catchments that, if not well understood, have the potential to confound conclusions drawn about the environmental impact of mining. The most obvious changes have involved invasive species: exotic plants (Para grass, Mimosa, Salvinia), native Hymenachne and feral animals (water buffalo, pigs and, most recently, cane toads). Other changes unrelated to mining result from altered land management practices (especially fire management), increased tourism and increased infrastructure for this and indigenous communities. The effects of most of these factors on aquatic biota of the region are poorly understood or documented. Most modern knowledge is based on short term data sets. This study has shown the importance of longer term data sets in distinguishing between effects of natural and human factors on fish communities.

An important ongoing task for this program will be to gain an improved understanding of the dynamics and factors affecting populations and communities of the key biota in streams adjacent to mine sites. Over time with further monitoring and analysis, it may be possible to distinguish and identify natural stream water quality, discharge and/or floodplain habitat factors responsible for changes to fish populations in Magela Creek billabongs. These causal factors may then be modelled to account for variation in monitoring response variable(s).

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Quantitative estimates of fish abundance from boat electro-fishing

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Keywords: Capture probability; population estimation; koi carp

Abstract

Multiple removals by boat electro-fishing were used to estimate fish populations in non-wadeable habitats in New Zealand lakes and rivers. Mean capture probability was 0.47±0.10 (± 95% CI) from 35 population estimates made with 2-7 successive removals. The relationship between the population estimate from the Zippin method (*Y*) and the number of fish caught in the first removal (*X*) was significant (adjusted r^2=0.84, P<0.001; Figure 2). The least-squares regression was $Y = 1.55\ X^{1.23}$. Mean density ± 95% confidence interval for 13 fishing occasions was 30±27 fish 100 m^{-2}. Mean biomass of fish for sites was 78±39 g m^{-2} (range 29 to 245 g m^{-2}). Koi carp comprised the largest proportion of the fish biomass wherever they were present. The high biomasses of koi carp estimated in these results (mean 56±33 g m^{-2}) suggest that they can reach problematic abundances in New Zealand. Biomass of spawning koi carp can exceed 400 g m^{-2}.

Introduction

Passive capture techniques such as gill and trap nets have been used to capture a wide variety of fish species in shallow New Zealand lakes (e.g. Hayes, 1989), but they have the limitation that the area sampled is generally unknown. Thus any inferences that can be made about fish abundance relate only to relative abundance and not to estimates of absolute abundance. The objective of our study was to make quantitative estimates of fish abundance in non-wadeable habitats with removal population estimates from boat electro-fishing. Further, we sought to establish the relationship between the first removal and the total population estimate.

Previous studies have used multiple-removal boat electro-fishing (e.g., Meador 2005), and Mitro and Zale (2000) compared first removals to multiple removal population estimates. Jowett and

Richardson (1996) used electro-fishing to compare first-removal catches to population estimates in wadeable streams and rivers. Bayley and Austen (2002) estimated capture efficiency from comparisons of boat electro-fishing and independent population estimates by a combination of toxicants, explosives, and draining. One study used multiple removal boat fishing to estimate fish biomass directly, without first calculating density (Thompson et al., 2002).

Methods

We fished with a 4.5-m long electro-fishing boat. The boat had a rigid aluminium pontoon hull with a 2-m beam, and was fitted with a 6-kilowatt Honda-powered custom-wound generator and a 5-kilowatt gas-powered pulsator (Smith-Root, Inc., model 5.0 GPP); two anode poles created the fishing field at the bow. The pulsator emitted pulses of direct current at a frequency of 60 pulses per second, and the power output was normally 2-4 amps root mean square. The two adjustable anode arrays each had 1-m long stainless steel rat tails that dangled in the water, and the boat hull itself acted as the cathode. We estimated the length and area fished with a boat-mounted global positioning system (Lowrance GlobalMap® 2400). We assumed from the reactions of fish that were observed to undergo forced swimming at the surface that the effective fishing width of the field was 4 m, and we used this width to estimate the area fished.

To estimate fish population size, we tallied the fish from each of 2-7 successive removals separately, fishing without replacement. We calculated the population size and capture probability for two removals using the Zippin method (e.g. Hicks 2003). For three removals or more, we used the programme CAPTURE (White et al., 1982). Length fished ranged from 53 to 987 m (mean 312 m), and area ranged from 212 to 3948 m^2 (mean 1578 m^2). The study sites were all located in the North Island of New Zealand (Figure 1).

Results

We caught five native and seven introduced fish species at the eight sites fished (Table 1). In general, we caught all species present at any site on the first removal, and species richness did not increase with subsequent removals. Mean capture probability calculated from 35 population estimates was 0.47±0.10 (± 95% CI). Capture probabilities varied with fish species; koi carp generally had the highest capture probabilities (generally >0.5).

The relationship between the population estimate from the Zippin method (Y) and the number of fish caught in the first removal (X) was significant (adjusted r^2=0.84, N=35, P<0.001; Fig. 2). The least-squares regression was

$$Y = 1.55\ X^{1.23} \quad \text{equation 1.}$$

Mean density ± 95% confidence interval for 13 fishing occasions was 30±27 fish 100 m^{-2}. Mean biomass of fish for sites was 78±39 g m^{-2} (range 29 to 245 g m^{-2}). Mean koi carp weight was 357-3657 g, which implies a non-spawning biomass range for koi carp of 13.4-163.5 g m^{-2} (mean 56±33 g m^{-2}).

Koi carp reached huge biomasses in the shallow lakes and wetlands of the Waikato region during spawning. In Lake Whangape in September 2003, we caught 24 koi carp in 696 m^2 of edge habitat in 11 mins. Total fish weight was 87 kg. Estimating total abundance from this single-pass removal with equation 1 suggests that the population estimate was 77 fish, or 11.1 fish 100 m^{-2}. As mean individual fish weight was 3645 g, this estimated density implies that the true biomass could be 403 g m^{-2}, or 4030 kg ha^{-1}.

Seasonal abundance of koi carp in the outlet of the Kimihia Wetland was low in October, probably as a result of movement to spawning sites in August or September (Figure 3). Water clarity was generally poor; black disc (Davies-Colley 1988) ranged from 0.2-1.9 m.

Conclusion

Capture probabilities that we estimated from boat electro-fishing (mean 0.47±0.10, 95% CI) were somewhat lower than those for native fish species and brown trout in wadeable streams (range 0.54-0.86; Jowett and Richardson 1996). Water clarity was often poor in our study, which may account for our lower capture probabilities. In addition, water depth was 1-3 m, and fish that did not float on immobilisation may have remained unseen.

In the USA, catchability has been estimated by comparison of boat electro-fishing with independent population estimates by toxicants, explosives, and draining. The proportion of fish caught by electro-fishing was species and size dependent, with maximum catchability for each species about 0.03 to 0.08 in the presence of macrophytes, and 0.08 and 0.16 without macrophytes (Bayley and Austen 2002). Whether our capture probability truly represents catchability, in the sense of Bayley and Austen (2002), remains to be tested. We intend to combine mark-recapture with removal methods to carry out this test.

Where numerous fish species are present, multiple removal boat electro-fishing can be useful to fully characterise fish assemblages (Meador, 2005). The effectiveness of fishing was greatest when species richness was low (about 10 species), as was the case in our study, where a maximum of 8 species were caught at any site.

In our boat electro-fishing, the estimated population increased dramatically as number caught in the first removal increases. When $X = 5$, equation 1 predicts that $Y = 11$ (2.2 times the number caught in the first removal). However, when $X = 500$, $Y = 3236$, or about 6.5 times the first removal. Catchability of largemouth bass (*Micropterus salmoides*) sampled with boat electro-fishing also declined with increasing density (McInerny and Cross 2002). In wadeable habitats, this problem is less severe. The true population density (Y) has been estimated as

$$Y = 1.96\,X^{1.028} \text{ equation 2,}$$

where X = number of fish caught in the first removal for a number of native and introduced New Zealand fish species. This means that about half the fish present will be caught on the first pass almost regardless of the magnitude of the first removal. Reworking the results of Hayes and Baird (1994) for age 0 brown trout in wadeable streams yielded a remarkably similar result:

$$Y = 1.93\ X^{0.955} \text{ equation 3.}$$

Though we have yet to establish the ecological effects of koi carp in the Waikato, the high biomasses estimated in these results suggest that koi carp can reach problematic abundances in New Zealand. At the biomasses that we have estimated, koi are likely to compete with other benthic fish such as eels, bullies, and catfish, and to reduce water quality significantly.

Acknowledgments

We acknowledge the help of our skilled boat drivers, Dudley Bell and Alex Ring, and numerous field and laboratory helpers. The study and boat construction was funded by the Department of Biological Sciences of the University of Waikato, and the boat was built by Orca Engineering and Marine Ltd in Rotorua.

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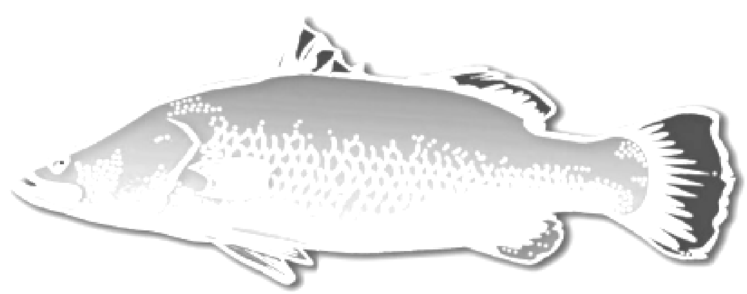
Table 1. Density and biomass of fish in lakes and rivers of the North Island, New Zealand, estimated from multiple removal boat electro-fishing

Site	Date	Common smelt	Common bullies	Grey mullet	Inanga	Shortfin eel	Catfish	Goldfish	Grass carp	Koi carp	Mosquitofish	Rainbow trout	Rudd	*N* removals	Length fished (m)	Area fished (m^2)	Density (fish 100 m^{-2})	Biomass (g m^{-2})
Huntly side channel, Waikato River	19 May 2004			●						●				2	636	2544	11	31
Kimihia Stream, Waikato River	29 Jan 2004							●		●			●	2	183	1281	21	175
Kimihia Stream, Waikato River	5 May 2004									●				4	167	1169	17	62
Kimihia Stream, Waikato River	8 Oct 2004									●				3	173	1730	7	83
Kimihia Stream, Waikato River	23 Jun 2005	●	●	●	●	●	●	●		●				3	196	1427	97	33
Lake Waiwhakareke, Waikato, site 1	13 May 2005					●								4	53	212	154	228
Lake Waiwhakareke, Waikato, site 2	13 May 2005						●							2	987	3948	2	17
Mokoia Pond, southern Taranaki	9 June 2005									●				7	215	951	2	62
Muddy Creek, Hawke Bay	6 Nov 2003								●					2	370	2959	2	125
Waikato River near Kimihia Stream	10 May 2004	●	●	●						●	●			3	264	1056	26	41
Waikato River near Kimihia Stream	8 Oct 2004									●				2	410	1640	5	99
Waikato River, Pukete	24 May 2005									●				4	199	796	16	35
Waikato River, Pukete	26 May 2005		●	●		●		●		●		●	●	5	199	796	33	29

Species identification Native species: common smelt (*Retropinna retropinna*), common bullies (*Gobiomorphus cotidianus*), grey mullet (*Mugil cephalus*), inanga (*Galaxias maculatus*), shortfin eel (*Anguilla australis*). Introduced species: catfish (*Ameiurus nebulosus*), goldfish (*Carassius auratus*), grass carp (*Ctenopharyngodon idella*), koi carp (*Cyprinus carpio*), mosquito fish (*Gambusia affinis*), rainbow trout (*Oncorhynchus mykiss*), rudd (*Scardinius erythrophthalmus*).



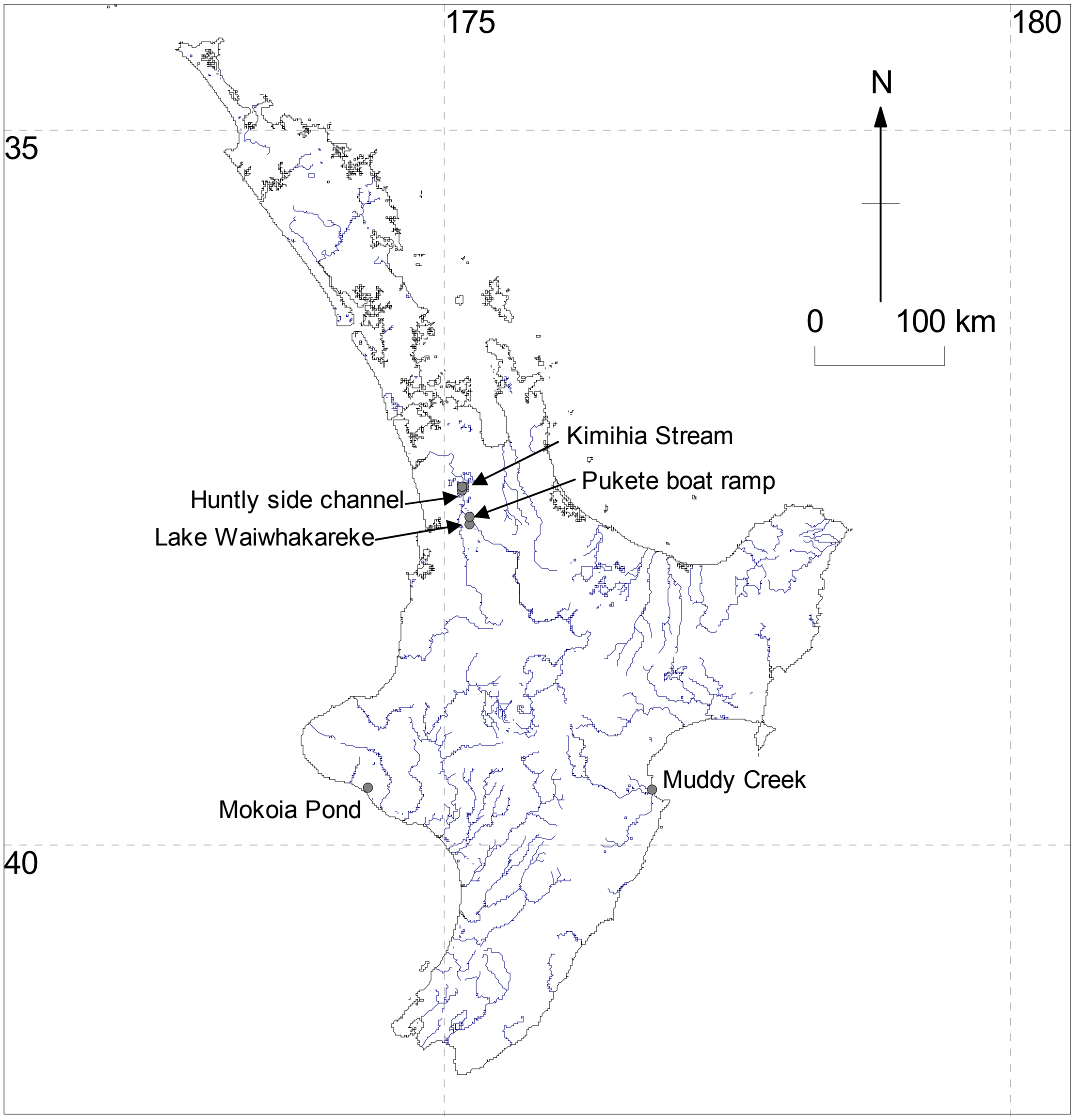


Figure 1. Location of study sites in the North Island of New Zealand

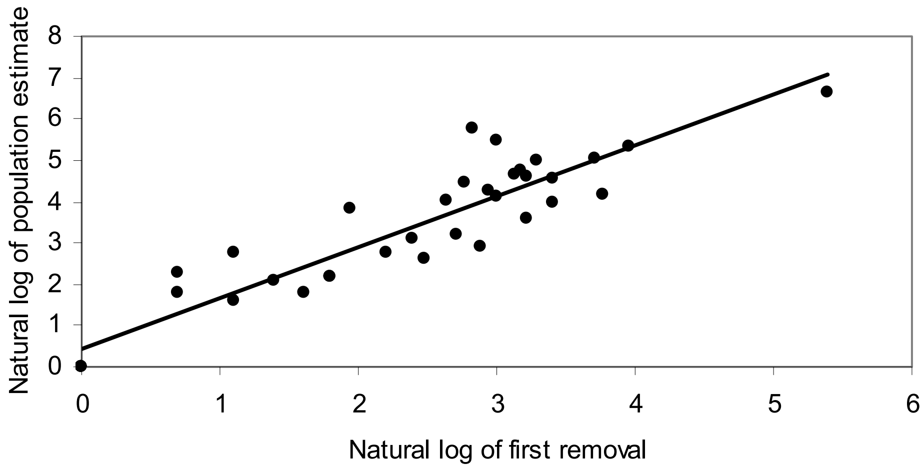


Figure 2. Relationship of first-pass removals to population estimates for all species in Table 1

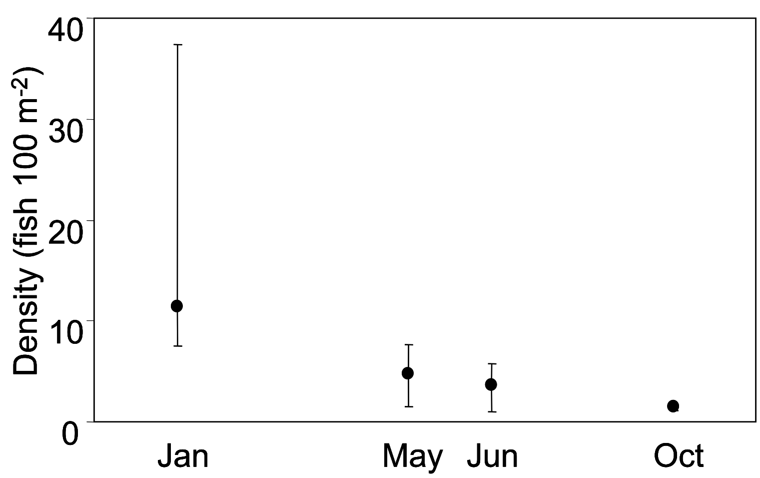
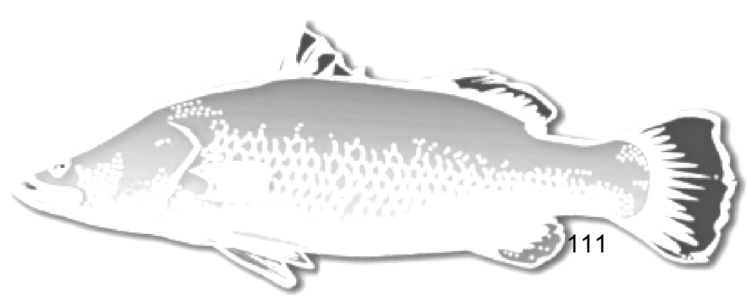


Figure 3. Removal population estimates of koi carp abundance in the Kimihia Wetland outlet and 95% confidence intervals



Comparing fishery-independent measures of snapper (*Pagrus auratus*) abundance in inner Shark Bay; daily egg production method vs. mark-recapture

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Keywords: biomass, spawning biomass, DEPM, egg survey, plankton, tagging.

Abstract

Snapper (*Pagrus auratus*) comprise about 50% of the catch landed by recreational boats in the inner gulfs of Shark Bay, Western Australia, and have been attracting large numbers of recreational fishers for many years, mostly during the winter tourist season when snapper are aggregated to spawn at predictable locations. Anecdotal information in the mid 1990s suggested recruitment overfishing in the Eastern Gulf, but no estimate of snapper abundance was available at that time. The daily egg production method (DEPM), where snapper eggs were collected in plankton surveys, has generated spawning biomass estimates annually since 1997. To confirm these estimates, a fishery-independent mark-recapture study was undertaken in 2004 (4,285 snapper tagged, 200+ recaptured), providing a biomass estimate (86 t, 95% ci 59 to 112) that was not significantly different to a comparable, concurrent DEPM estimate (38 t, 95% ci 22 to 73). The tagging study was assisted by about 70 volunteer fishers in 21 recreational vessels, who were trained in fish handling and tag application. Tagging generated an abundance estimate more quickly than the DEPM (one month v. six months, approx.), was relatively well received and understood by stakeholders, and helped foster a sense of custodianship of the resource and acceptance of management arrangements in the fishery. DEPM was less expensive, although tagging costs should decrease in future. The results help to develop a future research and management framework for inner Shark Bay snapper.

Introduction

Snapper (*Pagrus auratus*) is a recreationally important finfish species in the inner gulfs of Shark Bay, Western Australia (Figure 1), where much of the fishing effort focuses on spawning aggregations at predictable locations during the winter holiday season. Concerns over unsustainable recreational catches date back to the 1970s and 1980s, and by the mid 1990s anecdotal information suggested recruitment overfishing in the Eastern Gulf (Stephenson and Jackson, 2005). In 1997 various measures to limit recreational catches were introduced, and spawning biomass estimates were generated using the daily egg production method (DEPM) (Jackson and Cheng, 2001).

DEPM originally developed for small, schooling species (anchovies, sardines) with pelagic eggs and indeterminate annual fecundity (Lasker 1985, Alheit 1993), estimates spawning biomass from eggs produced each day over the area of spawning together with weight-specific daily batch fecundity. The technique requires ichthyoplankton surveys during the peak spawning period, combined with concurrent sampling of spawning fish. Since the DEPM was first proposed for benthic fishes with pelagic eggs and indeterminate fecundity (Zeldis, 1993), the technique has been used successfully on snapper in South Australia (McGlennon and Jones, 1999) and New Zealand (Zeldis and Francis, 1998).

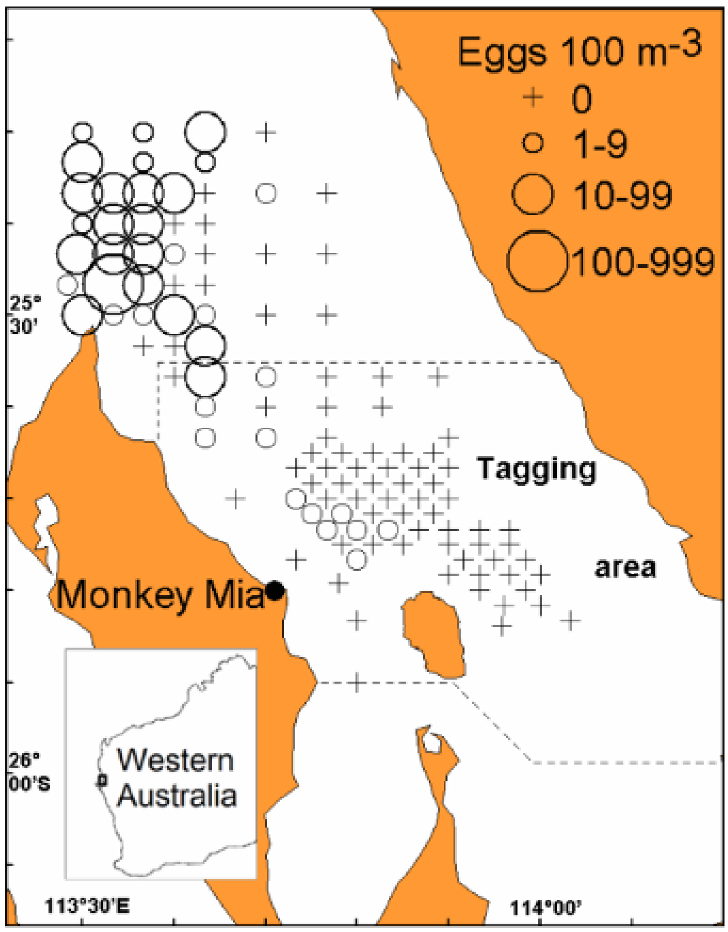

Figure 1. Map of Shark Bay's Eastern Gulf showing distribution and abundance of snapper eggs from the 2004 DEPM survey, and the area in which the concurrent tagging study was conducted

Spawning biomass estimates for inner Shark Bay snapper stocks have been imprecise in some years (i.e. 2000 and 2003), mainly due to large variances around estimates of egg production (Jackson and Cheng 2001), a feature common to DEPM surveys elsewhere (Borchers et al., 1997). DEPM requires significant resources, both in the field and laboratory, and is therefore relatively expensive for a recreational fishery with limited research funds. Microscopic plankton sorting, gonad histology and fecundity counts have required up to six months before biomass estimates can be formulated. The method is difficult for many recreational fishers and other stakeholders to comprehend, potentially undermining community confidence in research outcomes and management decisions. There is a need to investigate the potential of an alternative, independent measure of snapper stock size that may be incorporated into a future research framework for the inner Shark Bay snapper fishery.

The use of mark-recapture techniques to estimate the size of fish populations has a long history (Krebs, 1999). Tagging programs are routinely used for assessing snapper stocks in New Zealand (Maunder and Starr, 2001). In Australia, tagging of snapper has mainly been used to investigate movement and growth (Sanders, 1974; Sanders and Powell, 1979; Sumpton et al., 2003). In Shark Bay, Moran et al., (2003) conducted a major tagging study in the 1980s and found no evidence of mixing between the inner gulf populations, nor between oceanic waters and the inner bay, a conclusion supported by genetics (Johnson et al., 1986), otolith chemistry (Bastow et al., 2002; Gaughan et al., 2003) head morphology (Moran et al., 1998) and hydrodynamic modelling of snapper egg and larval dispersal (Nahas et al., 2003). Snapper stocks in inner Shark Bay are essentially closed populations, a necessary attribute for estimating abundance from tagging.

The objective of the current study was to undertake a DEPM survey in 2004 in Shark Bay’s Eastern Gulf to estimate the spawning biomass of snapper, and compare it to an abundance estimate generated by a tagging program, conducted in synchrony and utilizing volunteer recreational fishers. The time, cost and stakeholder receptivity of each method would also be compared in an effort to develop a future research framework for inner Shark Bay snapper.

Materials and Methods

DEPM

A DEPM survey was undertaken in the Eastern Gulf over five days around the new moon from 16 to 21 June 2004, coinciding with peak spawning. Snapper eggs were sampled at pre-determined stations (n = 114, Figure 1) based on a stratified design that utilised knowledge of the locations of main spawning grounds from previous surveys (Jackson and Cheng. 2001). A bongo-net fitted with a flow meter (each net mouth 60 cm diameter, net mesh 500 μm) was towed obliquely for 3 minutes at each station. Samples were fixed at sea in 5% buffered formaldehyde. In the laboratory eggs were counted and classified into 19 development stages, from which their age could be estimated based on water temperature and salinity (Norris and Jackson, 2002). Egg abundance at each station was converted to egg density m^{-2} of sea surface. Area of spawning (A, km^2) was estimated from the summation of area represented by each station at which eggs were collected. Egg densities were weighted in proportion to the area represented by each station with the sum of weighting factors equal to the total number of stations within A. Daily egg production (P, eggs m^{-2} day^{-1}) was estimated using an exponential mortality model by fitting a non-linear least squares regression to the weighted mean densities of all egg stages against egg age (days). A daily egg mortality value of

$Z = 0.64$ was used here, derived from analysis of DEPM data collected between 1997-2001 which provided more complete sampling of all egg stages. Non-parametric bootstrapping was used to calculate 95% confidence intervals for the estimate of *P* (Jackson and Cheng, 2001).

Spawning snapper were sampled at the same time as the egg survey, with the assistance of small number of local volunteer rod and line fishers in recreational vessels, simultaneously providing recapture data for the tagging program. Ashore, snapper were measured (fork length, *FL*, cm), from which the whole weight (w, g) of individual fish was estimated from a known length-weight relationship ($w = 0.0692FL^{2.6709}$). Ovaries were fixed in 10% formaldehyde for subsequent histological examination, upon which a reproductive stage was assigned: 1= mature, 2= resting, 3= developing, 4= developed, 5a= pre-spawning, 5b= spawning, 5c= recent spawning, 6= spent. The estimated weights of mature females (gonad stages 4, 5 and 6) were used to estimate average female weight (*W*). Only stage 5a ovaries (containing hydrated oocytes - indicative of daily spawning) were used to estimate batch fecundity using the standard gravimetric method (Hunter et al., 1985). The relationship between whole weight and batch fecundity, determined using linear regression, was then used to estimate mean batch fecundity (*F*) for all mature females sampled. Sex ratio (*R*) was estimated from the weight of all mature females divided by the total weight of all mature fish sampled. Spawning fraction (*S*) was estimated from the weight of females with stage 5a and 5b ovaries divided by the total weight of all mature females sampled. Non-parametric bootstrapping was used to calculate 95% confidence intervals (Jackson & Cheng, 2001).

Spawning biomass (*B*, tonnes) of snapper was estimated using the model of Parker (1985),

$$B = \frac{P\,A\,k\,W}{R\,F\,S}$$

where k = conversion factor (g to t, m^2 to km^2).

Tagging

Snapper were tagged between 28 May and 8 June 2004 while a prohibition on taking snapper was in force, with the assistance of volunteer recreational line fishers in 21 small vessels, trained in fish handling and tag application and accompanied at sea to ensure correct procedures were followed. Only fish greater than 38 cm FL were tagged, based on estimated length-at-50%-maturity (Jackson, unpubl. data), using uniquely numbered FloyTag dart tags (model FT-1-97, trimmed to about 6-7 cm in length to reduce tag movement), inserted about 2cm below the base of the dorsal fin. Tag shedding was assumed to only occur immediately upon release in this short term study, and was estimated by double tagging 60% of the fish. Tagging was confined to the central waters of the Eastern Gulf (Figure 1). The egg survey area, unchanged from previous years, extended beyond the tagging area, so a spawning biomass estimate for the tagging area was also generated to enable a valid comparison with the tagging biomass estimate.

A period of nine days to allow recovery, mixing and hence equal catchability between tagged and non-tagged fish was followed by the first recapture phase which coincided with sampling of spawning snapper for the DEPM. All recaptures during this 9 day period were omitted from the analysis. A second recapture phase occurred from 11 to 16 July 2004. The population size was estimated using the Petersen method (Seber, 1982), with some modifications to the maximum likelihood approach (see also Hilborn and Walters, 1992) necessary because a small number were tagged during the first recapture phase.

The log-likelihood function used was: $\log L = n2 \log (T/N) + (n1\text{-}n2) \log (1 - T/N)$

where T = the average number of tagged fish present during the two recapture phases weighted by the total fishing effort, $n1$ = the number of fish captured (tagged or not) during the two recapture phases and $n2$ = the number of these that were recaptures (tagged). The likelihood was maximised when the derivative of log L with respect to N was equal to zero, giving a population size $N = T\,(n1/n2)$. Confidence limits for this estimate of N were obtained by determining the coefficient of variation using $CV(N) = CV$ (Poisson ($n1$)) + CV (Poisson ($n2$)).

Time frame, cost and receptivity

The two methods were also compared for time taken to formulate biomass estimates, cost difference, and receptivity of stakeholders. The cost comparison was by determining the difference for each cost item under each method. Stakeholder receptivity was assessed through informal consultation.

Results

DEPM and tagging

DEPM estimated 195 tonnes (95% ci 110 to 368) of spawning snapper in the whole eastern gulf survey area, including 38 tonnes (95% ci 22 to 73) within the area where tagging was carried out (Table 1). The latter figure is not significantly different to the snapper biomass estimated by tagging: 86 tonnes (95% ci 59 to 112) (z-statistic 2.52. α= 0.05, p>0.05).

Table 1. Estimated DEPM parameters and bootstrapped 95% confidence intervals used to estimate snapper biomass in the Eastern Gulf of Shark Bay in 2004, including a biomass estimate for the tagging area only (*daily egg mortality based on estimates from surveys from 1997-2001)

Parameter	Sample size	Mean	95% ci
Adults -			
Average female weight, W (g)	46	3409	3106-3752
Batch fecundity, (,000) F	46	149.8	135.6-167.8
Sex ratio, R	113	0.45	0.33-0.53
Spawning fraction, S	46	0.39	0.25-0.55
Egg production -			
Total area surveyed (km2)		1,962	
Area of spawning, Eastern Gulf, A (km2)		700	
Area of spawning, Tagging Area only, A (km2)		138	
Daily egg mortality, Z (eggs dy-1)*		0.64	

Daily egg production, P (eggs m-2 dy-1)		1.95	1.32-2.87
Spawning biomass, B (tonnes)			
Eastern gulf		195	110-368
Tagging Area only		38	22-73

A total of 4735 snapper (≥380 mm, *FL*) were caught overall during the study. During initial mark-release phase, 4222 fish were tagged and released (98.5% of total tagged during the entire study). During the first recapture phase, 279 snapper were caught, 63 of which were tagged and released. In the second recapture phase, a further 234 snapper were caught but no further tagging occurred. A total of 75 tagged snapper were recaptured after the nine-day recovery phase.

Of the 105 double-tagged fish that were recaptured, 104 had retained both tags, suggesting a tag retention rate of 99.6% and the probability that double-tagged fish had lost both tags was estimated at 0.0016%. Combining these, the overall retention rate for the study was estimated at 99.8%. From this it was estimated that of the 4285 fish that were tagged, only seven fish would have been lost to the study due to tag shedding.

Because 63 fish were tagged during the first recapture phase, the numbers of tagged fish within the population varied slightly over the course of the study. The mean number of tagged fish within the population during this phase, weighted by fishing effort, was 4259. In our maximum likelihood estimation, we therefore adjusted the total number of tagged fish down from 4285 to 4259, to reflect the slightly reduced probability of recapture during this phase.

Time frame, cost and receptivity

The time taken to formulate the DEPM estimate, following plankton sorting, histology and fecundity work, was about 6 months, while the tagging estimate took approximately one month. The tagging program cost approximately $35 000 more than DEPM (Table 2), and used about 70 volunteer recreational fishers contributing about 480 person days during the tagging phase. Volunteers contributed approximately 14 person days sampling spawning stock for DEPM. The tagging was well received by the volunteers, who easily grasped the methodology and how it achieved the objective of estimating abundance. Many expressed a desire to participate in future tagging programs. Local business (bait, fuel and tackle suppliers) co-operated with the program and benefited from increased trade.

Table 2. Estimated additional cost of conducting the tagging program compared to DEPM, by cost item. For example, salaries (laboratory) were $1500 more for tagging, but histology $300 less. The figures do not reflect the absolute costs of running either program.

Cost item	**$**
Salaries (laboratory)	1500
Salaries (field)	10500
Staff field expenses	4000
Accommodation	1500
Fuel (car , boats)	5500
Bait, ice etc	1500
Fishing gear	1500
Reward t-shirts	3000
Tags	4500
Measuring boards	2000
Histology	-300
Chemical disposal	-200
Total	**$35,000**

Discussion

The DEPM spawning biomass estimate for the entire area of spawning in the Eastern Gulf (195 tonnes) is consistent with the trend of increasing stock size inferred from most DEPM estimates in previous years. This trend is unsurprising given the Eastern Gulf snapper fishery was closed in June 1998, until in 2003 it was reopened under highly restrictive management arrangements. The estimate and trend are also consistent with the trajectory of stock (mature) biomass predicted by an age-based assessment model (Stephenson and Jackson, 2005).

The DEPM and tagging biomass estimates for the same area of water were not significantly different, thereby independently confirming the reliability of each method. There is no evidence that either method provides a more reliable estimate, so the choice of method in future should encompass other criteria.

Although the tagging program was around $35 000 more expensive than the DEPM survey, it is important to note that we have undertaken DEPM surveys in the Eastern Gulf each year since 1997, and have become very efficient at field work (minimum of planning now required), sample processing and final analyses. The tagging study however, required much planning and organization, particularly the recruitment and management of the large number of volunteer recreational fishers. A relatively large (seven) number of Department of Fisheries WA research staff were required in the field to train and monitor volunteers. In future such tagging studies would require less planning, management and research staff input, potentially reducing the cost difference to around $20 000 in favour of DEPM.

The above cost comparison assumed all costs and benefits from tagging accrued in the same snapper spawning season, when aggregations were tagged and short term recaptures monitored. But future benefits can be derived from recaptures of tagged snapper in subsequent years, permitting biomass estimates without any further tagging. This was indeed the case in 2005, when no snapper were tagged but recaptures from the 2004 program were obtained.

The relatively longer time until results are available from the DEPM (up to six months) compared with the tagging biomass estimate (~ one month), delays the fishery management response if one is required. For the Eastern Gulf snapper fishery, information from an earlier time series of DEPM biomass estimates, catch, and a well developed age structure model suggest that sudden and urgent changes to management arrangements are unlikely. The longer delay for DEPM data is therefore not likely to be critical.

The tagging study was successful in terms of community involvement and interest. Incorporating volunteer recreational fishers into a genuine research program helped foster a sense of custodianship of the resource and an understanding of the research upon which management is based, and encouraged wider acceptance of management arrangements in the fishery. Some of the volunteers subsequently assisted the Department of Fisheries on other research projects.

In future, the research framework for inner gulf Shark Bay snapper may be modified in a way that assists management of the fishery. The use of tagging rather than the DEPM to estimate abundance enables faster management responses from earlier research results, which should be better received by stakeholders. Moreover, the stronger sense of custodianship of the resource among stakeholders that tagging fosters, may reduce the probability of unsustainably high recreational catches that could lead to highly restrictive management arrangements. Modifying the research framework in this way is financially slightly more expensive, however.

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Developing fishery-independent sampling tools for surveys of estuarine ichthyofauna in New South Wales: an experimental framework

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Abstract

Fishery-independent surveys are becoming a key tool in the scientific assessment of many important stocks of fish and invertebrates worldwide. However, before large-scale and long-term surveys can be implemented, it is necessary to do several pieces of important research. We present a logical framework for doing this research based on pilot studies incorporating manipulative experimental approaches. This includes (i) identifying suitable sampling gears for target species (ii) testing different gear configurations and sampling practices to ensure that samples of target species are optimal, representative and cost efficient (iii) understanding spatial and temporal scales of variability across different strata, and (iv) cost-benefit analyses to determine optimal levels of replication. We provide an example of this framework based on initial experiments from a research program currently developing fishery-independent surveys of estuarine fish in New South Wales. This approach can be applied elsewhere and we highlight the value of this type of pilot work as a precursor to fishery-independent monitoring studies in general.

Introduction

In the research and management of wild fisheries, the term ‘fishery-independent survey/s’ has become one of the catch phrases of the new millennium. However, the importance of these types of research surveys in assessing and monitoring the status of fish stocks, in conjunction with catch statistics from commercial fishing operations (i.e. fishery-dependent data), has long been recognised (Gunderson, 1993; Pennington and Stromme, 1998). The increased promotion and application of fishery-independent studies in recent years has probably occurred due to: (i) a demand for more rigorous and robust scientific assessments in accordance with the principles of ecologically-sustainable development (ESD), and (ii) an on-going need to assess stocks where commercial fishing has been reduced or eliminated; such as through the buying-out of fishing effort and declaration of marine parks, reserves and recreational-only fishing areas. While debate continues over the relative value of fishery-independent and -dependent data (Kesteven 1997; Horwood 1998), few could argue against cross-validating the different assessment methods wherever possible, or the use of an integrated approach towards reducing uncertainty in the management of marine resources (Kline 1996; Horwood 1998; Kennelly and Scandol 2002).

The perceived benefit of fishery-independent surveys is that they provide more robust information than commercial catch data because: (i) sampling is randomised over different strata rather than being concentrated where stocks are most abundant, (ii) there is potential to provide more representative data on the entire size-range of fish populations, rather than just the retained component, (iii) there is no reliance on fishers to accurately report their catches and effort, (iv) any biases should remain constant over time, and (v) data can be collected on species not normally retained in commercial and recreational fisheries, providing for broader biodiversity considerations. An important consideration, however, is that using poorly-designed or -developed sampling tools in research surveys may provide information as unreliable as that derived from some fishery-dependent sources. Therefore, the development of a standardized, representative and optimal sampling strategy, replicated over appropriate spatial and temporal scales, is a prerequisite to any fishery-independent study. This can be achieved through initial pilot studies using manipulative experiments to test specific hypotheses relating to the design and deployment of sampling gears.

There are numerous studies in the peer-reviewed literature that have (i) compared the utility and efficiency of different and/or competing methods for sampling fish and invertebrates (e.g. Gray and Bell, 1986; James and Fairweather, 1995; West, 2002; Guest et al., 2003; Olin and Malinen, 2003; Rotherham and West, 2003; Butcher et al., 2005), (ii) examined the effects of technical, biological and environmental factors on sampling gears (e.g. Miller, 1983; Jensen, 1990; Acosta, 1994; Misund et al.,1999; Petrakis et al., 2001), and (iii) measured spatial and temporal heterogeneity in abundances of organisms across a hierarchy of scales (e.g. Jones et al.,1990; Morrisey et al., 1992a,b; Ferrell et al., 1993). However, there are comparatively fewer published accounts of this type of experimental pilot work being done as a precursor to fishery-independent studies (but see Kennelly, 1989; Kennelly and Craig, 1992; Kennelly et al., 1993; Montgomery, 2000; Montgomery and Craig, 2003). This makes it difficult for researchers to become familiar with the basic approach in developing effective sampling tools prior to conducting large-scale and long-term surveys.

We reviewed previous fishery-independent surveys that have used pilot studies to develop and optimise sampling tools, and concluded that a simple and logical framework for this type of preliminary work could be constructed and adapted elsewhere. This framework involves: (i) identifying suitable sampling gears for the key target species, (ii) testing different gear configurations and sampling practices to ensure that samples of target species are optimal, representative and cost efficient, (iii) understanding spatial and temporal scales of variability across different strata, and (iv) cost-benefit analyses to determine optimal levels of replication (Figure 1).

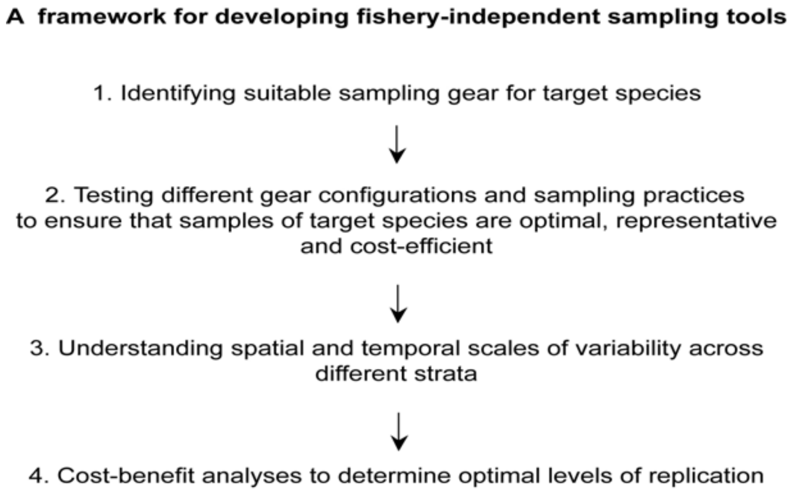

Figure 1. The framework used to develop fishery-independent sampling tools

Example of the framework: developing fishery-independent surveys for the adaptive management of NSW's estuarine fisheries

Estuaries in NSW contain a diverse and abundant ichthyofauna, which support complex multi-sector (i.e. commercial, recreational and indigenous), -species and -method fisheries. While more than 100 species of fish are landed from these estuaries, less than 10 species account for most (approximately 80%) of the commercial catch; valued at approximately $AUD 19 million in 2000-01 (5,043 t; Anon., 2003). At present, the status of estuarine fish stocks in NSW is mostly monitored using fishery-dependent data, including catch and effort information supplied by the commercial sector, biological sampling of 3 key species from commercial landings (e.g. Gray et al., 2002) and sporadic recreational creel surveys (e.g. Steffe and Chapman, 2003). Given the limitations of these types of data in assessing fish stocks (see above), the future use of fishery-independent surveys has been advocated to provide a vastly improved quality of information on the biology, ecology and status of NSW's estuarine fish resources.

A collaborative project between the NSW Department of Primary Industries, the Centre for Research on the Ecological Impacts of Coastal Cities at the University of Sydney, and the Fisheries Research and Development Corporation (FRDC 2002/059) is currently developing the necessary tools (i.e. the gears, methods, procedures and analyses) required to conduct these surveys in NSW estuaries. Once this work has been done, surveys based on a rigorous sampling design can be implemented. While commercial and scientific sampling gears are available, these gears typically are designed to be size-selective and species-specific. In many cases, such gears need to be modified so that they will sample wider size ranges and greater diversities of fish. It is clear, however, that given the range of different life histories and behaviours associated with the multi-species assemblages of fish in estuarine environments, no single sampling method will be effective in terms of providing an index of relative abundance for all species. Rather, a complementary 'suite' of both towed or 'active' (e.g. trawls, haul nets) and static or 'passive' sampling gears (e.g. gill nets, traps) is required to collect data on the relative abundances of fish and their demographic information (e.g. length, sex and age composition, reproduction and recruitment dynamics).

The remainder of this paper focuses on an example of how the above framework is being followed to develop one method in our future 'suite' of sampling tools for fishery-independent surveys of estuarine fish stocks in NSW. The sampling gear being considered is termed a multi-mesh gill net, and comprises a series of panels with different stretched mesh sizes (from small to large) designed to catch a wider range of fish species of differing sizes and morphologies.

Step 1. Identifying suitable sampling gears for target species

An experiment was done to compare the utility and efficiency of multi-mesh gill and trammel nets for sampling estuarine fishes. These two gears are similar in that they are both passive and so are influenced by fish activity and behaviour. The main difference is that gill nets comprise a single panel of netting, while trammel nets have three panels. The two outer panels of a trammel net are large-meshed and enclose a loosely-hung centre panel made from mesh typically 4 to 7 times smaller. When a fish swims through the large outer mesh into the loose interior mesh, a pocket is formed, which entangles the fish. Therefore, trammel nets primarily entangle fish, while gill nets rely more on 'meshing' fish behind their gills.

Replicate multi-mesh gill and trammel nets, each comprising five 30 m long panels made from different-sized mesh (38, 54, 70, 90 and 100 mm stretched mesh openings) were fished in a NSW barrier estuary to test the hypotheses of no differences in the catches of fish between net types and mesh size. For specific details see Gray et al. (in press). The results from this work showed that there were no differences in the compositions and structures of assemblages, mean abundance or diversity of catches between the two types of net. But, based on a greater precision of CPUE estimates, ease of use and less sampling effort, the multi-mesh gill net was considered to be the superior sampling unit.

Step 2. Testing different gear configurations and sampling practices to ensure that samples of target species are optimal, representative and cost efficient

Having identified multi-mesh gill nets as being a better sampling gear than trammel nets, the next step in the framework was to determine the most appropriate configuration and soak (i.e. length of time the gear is fished) and setting (i.e. time of night the gear is deployed) times. Experiments were done to test the hypotheses of no differences in catches and catch rates of fish between soak (1, 3 and 6 h) and setting times (18:00, 22:00 and 3:00 hrs) and net lengths (20, 50 and 120 m panels). Univariate and multivariate procedures revealed that 20-m panels soaked for 1 h at any time of the night were the most optimal (in terms of catch obtained and efficiency of sampling) and representative strategy for sampling populations, assemblages and the sizes of most species available for capture. Since previous studies employing gill nets often use much longer panels and soak times (e.g. nets are often set from dusk until dawn), the benefits of this strategy include greater replication, lower costs, and the potential for fewer mortalities (if the catch is processed and released as the gear is retrieved).

Step 3. Understanding spatial and temporal scales of variability across different strata

The third step in the framework involves examining variation in the abundance of organisms across a range of spatial and temporal scales, using hierarchical sampling designs. Data from these experiments typically are analysed using nested analysis of variance (see Morrisey et al. 1992a, b). This provides information on the need to sample at different scales and data for cost-benefit analysis (step 4). In the present example, two experiments (1 each on spatial and temporal variation, respectively) were designed..

The first experiment examined patterns of variability of fish fauna at a hierarchy of spatial scales in two different habitats (deep and shallow) within a NSW estuary. The design incorporated spatial scales ranging from zones within estuaries separated by 2 - 20 km, sites within zones separated by at least 1 km and replicate gill nets separated by 50 - 100 m (Fig. 2). The hypothesis tested was that abundances of estuarine fish fauna were not significantly different at each of the spatial scales investigated. To provide greater generality, the experiment was done in two large coastal lakes (Lake Macquarie and St Georges Basin lat long). These estuaries are relatively well-mixed and do not have large salinity gradients. Data from this experiment are being analysed and will form the basis of a future publication.

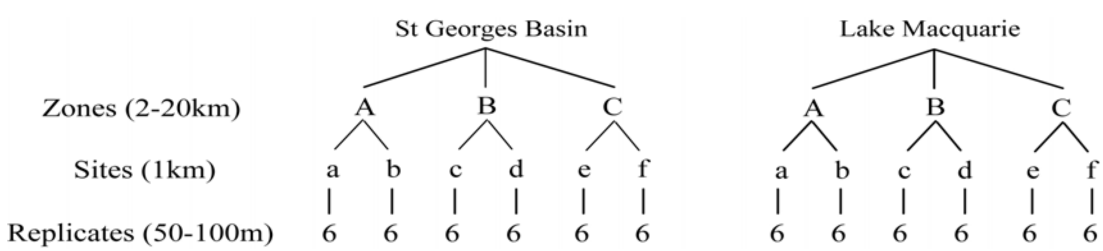

Figure 2. Diagrammatic representation of the hierarchical scales sampled with multi-mesh gill nets in an experiment on spatial variation of estuarine fish fauna

Experiments investigating spatial heterogeneity in the abundances of organisms across hierarchical scales are relatively common (e.g. Jones et al. 1990; Morrisey et al., 1992a; Olabarria and Chapman, 2001) and the problem of spatial pseudo-replication (after Hurlbert, 1984) is well understood. Few studies, however, have considered that short-term variation (day to day, week to week) has the potential to confound comparisons across longer scales (e.g. month to month, season to season) (but see Morrissey et al., 1992b; Olabarria and Chapman, 2002). Differences between one sampling time and the next cannot be interpreted as being associated with that particular scale (month to month or season to season), unless it has been demonstrated that differences at smaller scales (i.e. daily or weekly) are not as large (Morrisey et al., 1992b).

Given the above, a second experiment was designed to examine variation in the abundance of estuarine fish fauna at a hierarchy of temporal scales (weeks, months and seasons) and in two different habitats (deep and shallow water) in an estuary (St Georges Basin) (Figure 3). Sampling also incorporated two spatial scales within an estuary: two sites 1 km apart, nested within two zones (2 – 20 km) apart. The incorporation of replicate sites and zones was designed to measure temporal-spatial interactions and to provide greater generality. We did not attempt to measure nightly variation because it takes several nights (in this case four) to sample all four sites across the two zones. What we desired to know was whether weekly variation was greater than monthly or seasonal differences. Sites were sampled for two consecutive weeks, in each of two consecutive months in each of two consecutive 'seasons' (July-August and October-November 2005). At the time of writing, this experiment was still ongoing.

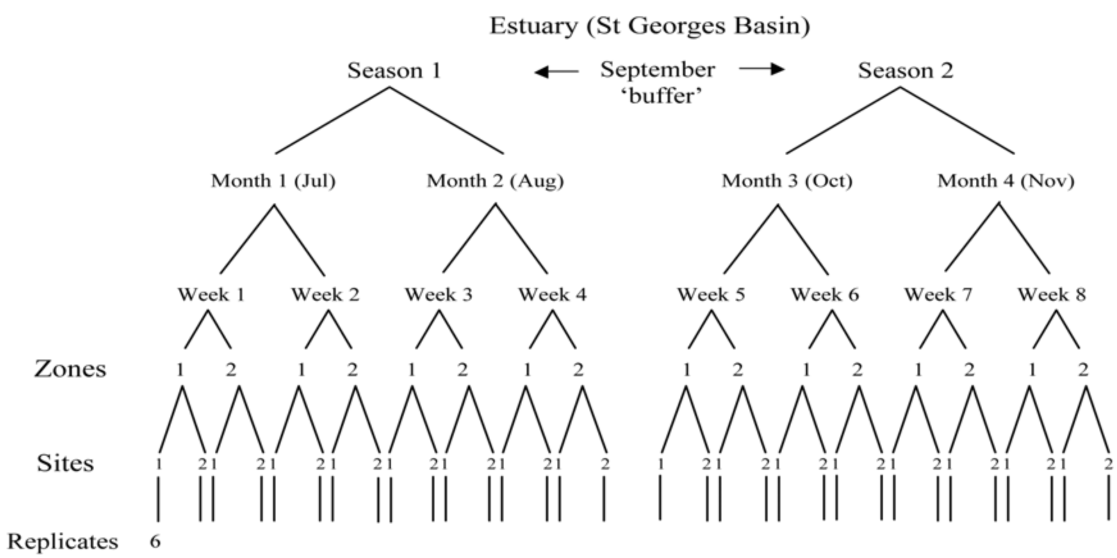

Figure 3. Diagrammatic representation of hierarchical scales sampled with multi-mesh gill nets in an experiment on temporal variation of estuarine fish fauna.

Step 4. Cost-benefit analyses to determine optimal levels of replication

The final step in the framework involves using information on variances from studies of spatial and temporal heterogeneity (step 3), to perform cost-benefit analyses. These techniques are well established (e.g. Snedecor and Cochran, 1967; Winer, 1971) and allow the determination of optimal levels of replication given restrictions of time, money or both. Applied to the present example, these analyses will answer questions relating to how many replicate nets, sites, zones and sampling times are required to provide reliable and representative results in future surveys. Since experiments from step 3 (see above) were in progress at the time of writing, cost-benefit analyses have yet to be completed. For applied examples focussing on marine biota see Kennelly and Underwood (1984, 1985), Kennelly (1989), Kennelly et al. (1993) and Montgomery (2000).

Conclusion

Fishery-independent surveys that employ poorly-developed gears and sampling protocols are likely to be non-representative, sub-optimal and costly. This paper has demonstrated that by using an experimental framework to develop sampling tools prior to implementing large-scale and long-term surveys of estuarine icthyofauna in NSW, more reliable and cost-effective information can be obtained. While generic frameworks have been used effectively in solving other fisheries-related problems (e.g. Kennelly, 1997; Broadhurst, 2000), the example provided here may not be applicable to all fishery-independent studies. Nevertheless, the present framework should at least be considered as a useful starting point in developing fishery-independent surveys or other types of biological and ecological sampling programs, elsewhere.

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The potential use of nitrogen and carbon stable isotope ratios in seabirds to monitor predator-prey relationships in baitfish fisheries

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Abstract

In the move towards ecosystem-based management there is need to find efficient methods of detecting changes in the availability of fish prey to predatory wildlife, where this could be influenced by a fishery. This is particularly important in fisheries targeting shoaling “forage” or bait fishes that are the main conduit of energy between primary production and higher order consumers in pelagic ecosystems. Seabird populations with point cantered foraging constraints may be sensitive to short term, local depletion of prey resources and therefore be an appropriate ecological indicator of trophic interactions with fisheries.

This paper reports on a small, partially self-funded, pilot project looking at the stable isotope ratios of carbon and nitrogen in baitfish and seabirds and their potential use as indicators of prey availability. The techniques employed involved little field time and were non-destructive, using discarded tissues such as the shell membranes from hatched eggs, mesoptile feathers from chicks and adult moult feathers. It will be argued that these techniques may offer a defensible and affordable method of monitoring the trophic impacts of fishing in some fisheries.

Introduction

It is now generally acknowledged that the assessment of sustainability in wild fisheries must encompass an understanding of the direct and indirect impact of the activity on the aquatic ecosystem (Fletcher et al., 2002). As a consequence there is an intention to shift management from decision-support systems focused entirely on target species, and supported solely by fishery dependent data, to monitoring a range of ecosystem components identified as being at risk (Ward et al., 2002).

Monitoring

In monitoring for ecosystem changes that may be induced by fishing, natural resource managers require indicators that are:

- measures of habitat quality or trophic state;
- focused on dependent or associated species;
- operating on appropriate spatial and temporal scales;
- capable of detecting incipient change (Goldsmith1991);
- suitable for reporting against pre-determined ecological objectives;
- both scientifically defensible and cost efficient.

Forage or baitfish fisheries

Springer and Speckman (1997) define forage fishes as "abundant schooling fishes preyed upon by many species of seabirds, marine mammals and other fish species. They provide important ecosystem functions by transferring energy from primary or secondary producers to higher trophic levels."

The pivotal role of forage fishes requires special consideration in the ecosystem-based management of marine fisheries. Globally, a range of teleost taxa have been identified as forage fishes in coastal waters including sardines (eg. *Sardinops*, *Sardinella*), anchovies (*Engraulis* spp), sand eels or lance (*Ammodytes* spp), scad (*Trachyurus* spp) herring (*Clupea harengus*) and mackerel (*Scomber* spp). Locally, off south-western Australia smaller near-shore and estuarine species such as the Sandy Sprat *Hyperlophus vittatus* and Hardeyheads (*Atherinidae*) also function as "bait" or forage species, the former being a mainstay of the recreational fishing bait market.

Two "baitfish fisheries" operate off the south-west coast of Western Australian, centred in the Perth metropolitan area. The central zone of the West Coast Purse Seine Fishery (WCPSF) targets pilchards *Sardinops sagax* on the inner continental shelf between 31 and 33^{o} south latitude. In strong Leeuwin Current periods, during *La Nina* events, the pilchards retreat from the fishing zone and the fishery then targets Scaly Mackerel *Sardinella lemura*, a tropical sardine that is otherwise concentrated further north in the Houtman Abrolhos region (Penn et al.2005, Gaughan and Mitchell 2000).

The West Coast Beach Bait Fishery targets whitebait using beach-seine nets on the lower west coast of WA between Guilderton north of Perth and Tim's Thicket to the south (Penn et. al., 2005). In recent years fishing has been more or less concentrated on the lower Swan River, Fremantle Harbour and the mouth of the Peel-Harvey Estuary. Coastal sites such as Warnbro Sound and Comet Bay have not been available to the fishers in recent years due to local government regulation of beach access to four-wheel drive vehicles. Another whitebait fishery operates further south in the Bunbury/Geographe Bay region and also supplies the local whitebait market.

Seabirds as potential ecological indicators of prey availability to predators

Seabirds have a number of advantages as dependent species that could be used as indicators of prey availability to the higher trophic levels.

- Their point centred foraging makes them potentially vulnerable to temporary local depletion which is an identified risk.
- Various species can provide a range of foraging (spatial) scales in most regions.
- Their foraging behaviour, breeding performance, demography and tissues can be used to monitor trends that occur on a variety of time-scales from diurnal to decadal.
- They must return to land when breeding making them predictable and accessible.
- They are relatively easy to sample.
- Their biology is relatively well understood.
- Sampling can be non-destructive.
- They are useful surrogates for medium sized predatory fish that may be more difficult to sample consistently.
- They are a protected species of public interest.

Stable Isotope Ratios in Marine Systems

In the avian literature, stable isotope ratios of nitrogen show stepwise enrichment with trophic level of 3-7^{o}/oo. Effectively, the nitrogen ratios represent a homogenized measure of "mean trophic level". Stable isotope ratios of carbon do not vary predictably with trophic level but most frequently show a slight increase in ^{13}C fractionation of less than 1^{o}/oo. The carbon isotope ratios are however broad indicators of the carbon source at the base of the food chain. (E.g. Hobson, 1993 and 1995; Hobson et al., 1994).

Methods

Selection of seabird species for sampling

Three species of seabird known to feed predominantly on forage fish, and with breeding colonies on islands between 31 and 33^{o} south latitude, were selected for this investigation.

The little penguin (*Eudyptula minor*) population on Penguin Island (at 32^{o} 18'S, 115^{o}41'E) has been shown over a number of study years to forage on adult whitebait (Klomp and Wooller, 1998; Wienecke et al., 1995; Wooller et al., 1991). Most foraging time is spent 15-20 km south of the colony in Comet Bay (Wooller et al., in review). The fish are thought to come predominantly from an insular stock with a nursery area in the swash zone at Becher Point, in the south-eastern quadrant of Warnbro Sound (Lenanton et al., 2003). It is estimated that the Penguin Island colony utilizes around 100 tonnes of white bait each season; this is more than 10% of the estimated breeding stock in the fishery (Wienecke et al., 1995; Gaughan et al., 1996). The risk to the penguins of whitebait removal from the Warnbro Sound / Comet Bay area of the fishery would therefore be significant if beach-seining continued or purse-seining for whitebait in this location were to be permitted.

Crested terns (*Sterna bergii*) in Australia and South Africa typically concentrate foraging on the dominant baitfish which is usually a sardine or anchovy (Dunlop, 1987; Nichollson, 2002; Surman and Wooller, 2002; Gaughan et al., 2002; Walter et al., 1987; Chairadia et al., 2002; Crawford, 2003). Pilchards were the dominant prey of crested terns in the study area during the early 1980s (Dunlop, 1987). During the 1998-2000 *La Nina* event the species appeared to switch onto scaly mackerel (pers.obs.).

The diet of the wedge-tailed shearwater has not been investigated within the study area. Further north at the Houtman Abrolhos the scaly mackerel is an important component of the diet and breeding may not occur if this species is not available (Gaughan et al., 2002). Observations at sea suggest that the species also targets pilchards and it is likely that, as in the Crested Tern, this species utilizes the dominant shoaling clupeid within foraging range.

Selection of tissues for stable isotope analysis

The stable isotopes ratios of ^{13}C and ^{15}N were determined in the seabirds using feather and egg-shell membrane materials. Scales were used to provide ratios in the baitfish (prey) species.

The stable isotope signatures in feathers reflect the dietary intake of the bird at or shortly before (weeks) their period of development. With some knowledge of the moult sequence one could find a feather containing trophic information on a particular stage in the life history or time of year. Fully grown feathers are inert, preventing the re-mobilization of constituents (Quillfelldt et al., 2005, Thompson and Furness, 1995, Cherel et al., 2000). Feathers are made of keratin, a protein based material and, as such, there was no need for costly lipid or calcium extraction for the analysis.

Egg shell membranes are laid down during egg formation reflecting the dietary intake of the seabird 3-6 days prior to laying (Hobson, 1995).

Fish scales are also collagen-based and laid down over the life of the animal.

All these materials could be stored indefinitely in a dried condition. Historical these materials may be available from specimens in museums and other collections to provide retrospective data on trophic conditions.

Sampling

Mesoptile (secondary down) feathers were collected from large (ruff stage) little penguin chicks in the Penguin Island colony on 6 November and 1 December 2004. Freshly moulted feathers from post-breeding adults were collected from artificial nest boxes and natural burrows on 19 December 2004. The adult moult feathers will provide information on the diet of the bird during the three week pre-moult fattening period in the preceding year (i.e. December/January 2003). The chick feathers are representative of recent foraging periods in the current season.

Moult feathers were also collected in December 2004 from six captive adult penguins housed at the Penguin Experience on the Island and fed a constant diet of whitebait (*Hyperlophus vittatus)* with occasional blue bait (*Sprattelloides robustus*). Blue bait frequently associate in whitebait schools and the mix fed to the penguins is probably representative of the "run-of- the-net" species composition in the fishery.

Egg-shell membranes were collected from recently hatched Crested Tern eggs at a colony on Carnac Island (32^{o}07'S, 115^{o}19'E) in early October 2004.

Egg-shell membranes were collected from hatched eggs located in the breeding burrows of wedge-tailed shearwater on Rottnest Island (Cape Vlaming – 32^{o}00'S, 115^{o}29'E) and Lancelin Island (31^{o}00'S, 115^{o}19'E) during January 2003. Protoptile and mesoptile feathers were collected from wedge-tailed shearwater chicks from the Lancelin Island colony in February, March and April 2003.

Homogenized samples of fish scales were collected from the fresh whitebait/blue bait product supplied to the captive penguins.

Pilchard scale samples were collected from a purse-seine vessel operating off Fremantle (within the fishery) in April 2005. Scaly Mackerel scale samples came from the Geraldton area in the same month.

Feather, egg-shell membrane and fish scale material was cut fine using dissecting scissors before being sent to the laboratory. Only 2-2.5 mg of feather and shell-membrane material was sufficient for stable isotope analysis. Slightly more (3-3.5 mg) of fish scale material was required. The analysis was conducted by the Western Australian Biogeochemistry Centre at the University of Western Australia.

Results

The change in ^{15}N and ^{13}C fractionation between the whitebait/blue bait fed to captive little penguins and the moulted feathers from the same birds is presented in Table 1. There was a significant enrichment for the nitrogen isotope between whitebait and little penguin of 4.79^o/oo ($t = 25.673$, $P < 0.001$). This was within the range for ^{15}N previously reported. There was a non-significant depletion of ^{13}C of 0.56^o/oo.

Table 1. The ^{15}N and ^{13}C stable isotope ratios (^{o}/oo) in whitebait feed samples (N=7) and the moult feathers of individual captive penguins (N =6)

In the subsequent analysis the enrichment increment derived from the captive penguins has been used to project ^{15}N values from baitfish prey to seabird predators. The ^{13}C values are taken to be constant between levels.

The mean ^{15}N and ^{13}C values obtained for all the baitfish and seabird samples collected are plotted against each other in Figure 1. The Shearwater egg-shell membrane and protoptile feather samples were combined as both are comprised of nutrients laid down during egg-formation during the pre-laying period. Seabird samples in Figure 1 are referred to as adult, pre-laying or chick to reflect the stage in the annual cycle during which the foraging occurred.

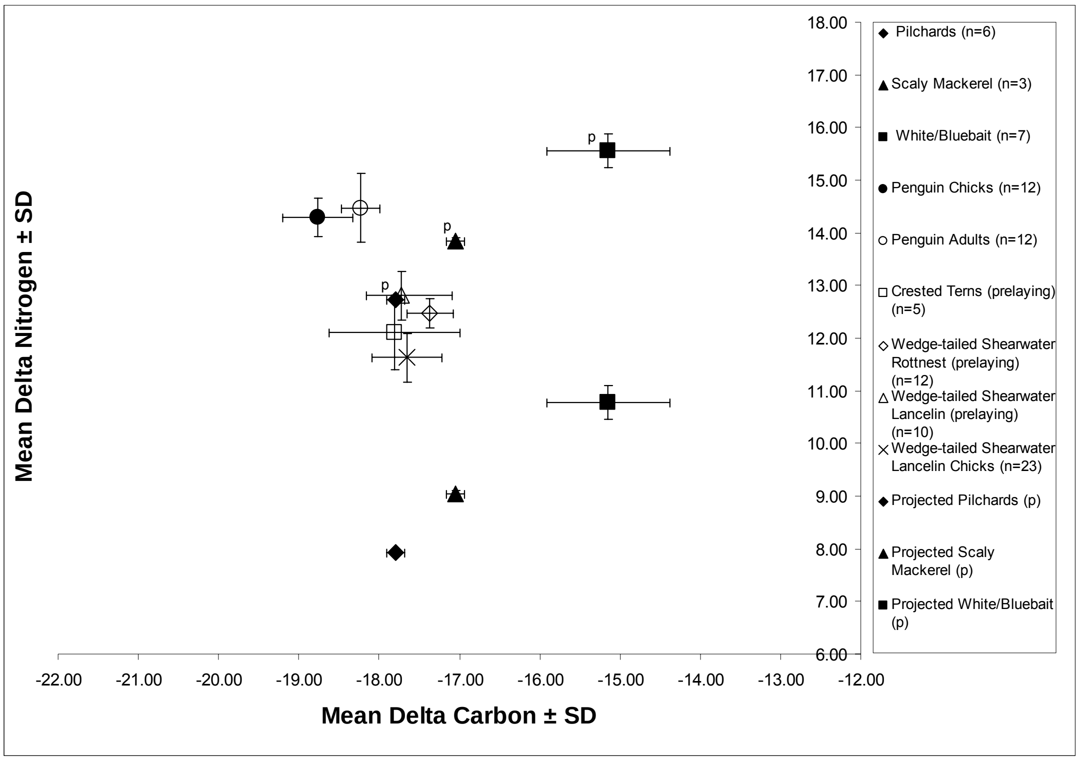

Figure 1. Plot of the mean ^{15}N and ^{13}C stable isotope ratios (with standard deviation error bars) from the analysis of baitfish and seabird tissues

The pilchard, scaly mackerel and whitebait samples produced distinct signatures that were clearly representative of different distributions for both ^{15}N ($F=267.2$, $P<0.001$) and ^{13}C ($F=63.17$, $P<0.001$). The ^{15}N values indicated that Whitebait feed at a higher mean trophic level than both Scaly Mackerel and Pilchards, with the latter most concentrated on producers (presumably phytoplankton).

The seabird samples formed two clusters, pilchard feeders and penguins, but overall the signature represented a number of distinct distributions for both ^{15}N ($F=84.954$, $P<0.001$) and ^{13}C ($F=16.513$, $P<0.001$).

The samples from the seabirds (wedge-tailed shearwater and Crested Tern) predicted from other studies to be focused on the sardines (*Sardinops* or *Sardinella*) clustered around the projected pilchard (*Sardinops*) signature. There were no significant differences for either ^{15}N or ^{13}C between the Crested Tern pre-laying, shearwater (Rottnest pre-laying) or shearwater (Lancelin pre-laying) samples or between these and projected pilchard signature. There was a significant difference between the combined shearwater pre-laying samples and Shearwater chicks for ^{15}N ($t = 6.693$, $P<0.001$).

The penguin adult (moult) samples clustered at a higher ^{15}N level to the other seabirds probably reflecting the higher trophic level of the whitebait component in the diet. There was a significant difference between adult and chick samples for ^{13}C ($t=3.716$, $P<0.01$). There was no correspondence between the penguin cluster and the projected whitebait signature, primarily due to a major discrepancy in the ^{13}C value.

Discussion

Baitfish-seabird trophic relationships

Bait fishes are planktivorous, consuming a varying mix of phytoplankton and zooplankton. Pilchards (*Sardinops sagax)* are considered to be mainly zooplankters when small but become grazers on phytoplankton when larger (King and MacLeod, 1976) and accruing most of their biomass. Scaly mackerel are predominately zooplankton foragers although phytoplankton and organic detritus are also consumed. (Gaughan et al.,1996). Whitebait (*Hyperlophus vittatus*) feed mainly on zooplankton (Blaber, 1980). The stepwise increase in ^{15}N values from pilchards, to scaly mackerel and then whitebait seems to broadly reflect our understanding of the trophic levels occupied by these species.

On the basis of previous diet and observational studies using other methods it was predicted the wedge-tailed shearwaters and Crested Terns would utilize the dominant sardine in the region. During 2002 and 2003 *Sardinella* outranked *Sardinops* in the catches of the central zone of the West Coast Purse-seine Fishery, however both species were available. Pilchards appear to be recovering after the catastrophic mortalities if 1995 and 1998 (Penn et al., 2005).

The wedge-tailed shearwater and crested tern samples clustered around the projected stable isotope signature for pilchards suggesting that both species were foraging predominately on this prey rather than scaly mackerel which may also have been available. This may suggest that pilchards are preferred over scaly mackerel when both are available.

Wedge-tailed shearwater chick mesoptile feathers were significantly lower in ^{15}N than egg-shell membranes. This indicates that the adults are able to feed at a higher trophic level during the three weeks of the fattening (honeymoon) period prior to laying, when they are free to range more widely. The necessity to feed chicks on an almost daily basis would be predicted to restrict foraging range and limit access to preferred prey types. The reduced trophic level during chick provisioning would suggest an increased intake of a herbivore such as krill rather than predators such as squid.

The adult and chick penguin samples are at a higher mean trophic level than in the pilchard feeders. This is consistent with a diet consisting predominantly of zooplankton foraging whitebait. The adult and chick samples differed significantly in the mean ^{13}C. This could indicate that adults forage further away from the colony in habitats with different carbon sources during the three-week pre-moult fattening exodus, when they are not tied to the colony. Alternatively it could indicate the foraging locations changed between summer 2003 and spring 2004. A time series of stable-isotope data would be necessary to improve the interpretation of this result.

The ^{13}C value for the whitebait is not consistent with the signature in the wild penguin samples. The indication from the whitebait fed to the captive penguins is that ^{13}C should remain virtually unchanged from prey to predator.

In recent years whitebait were not been caught from the coastal habitats in Warnbro Sound and Comet Bay due to the closure of the beaches to the fishers. It appears that the catch being fed to the captive penguins comes from a range of other locations including the mouth of the Swan – Canning Estuary, the mouth of the Peel Harvey Estuary and (outside the fishery) at the mouth of Leschenault Inlet. All these catching areas are in eutrophic, estuarine habitats with phytoplankton as the dominant producers. In the area fished by the wild penguins the main source of carbon is probably decaying seagrass and macro-algae from the inshore swash zones. The difference between the ^{13}C in wild penguins and the Whitebait probably reflects the external origin of the fish sampled. This range of sources in the Whitebait samples may also be reflected in the relatively large standard deviation compared to the other baitfish.

Monitoring prey availability using stable isotopes

The results of this study indicate that the stable isotope ratios on nitrogen and carbon in "discarded" seabird tissues could be used effectively to monitor the availability of baitfish prey to predators. Appropriate selection of tissues would provide a range a spatial scales and time (seasonal) periods. Sample sizes of 10-12 would appear to be sufficient to examine any particular time frame or stage in the seabird breeding cycle.

Historically the collection of data to produce similar (mean diet) parameters has involved field exercises taking several weeks or months. These activities would have involved direct observation of food delivery to mates or chicks, measurement of chick weights (provisioning) and growth, collection of regurgitates or sampling by water offloading, laboratory analysis of stomach contents and/or direct observation of seabirds foraging at sea.

None of these activities are likely to be cost-effective in the context of monitoring prey availability to predators in the management of a low value fishery. Using the stable isotope ratios in "discarded" seabird tissues the field activities can be limited to a few well timed days each season. The materials could in some cases be collected on the proximate islands by amateurs or by the fishers themselves. The main cost will be the laboratory analysis of stable isotopes (perhaps $1000-2000 per annum for the fisheries considered in this paper).

The critical steps in developing a monitoring plan using stable isotope ratios in seabirds as indicators of prey availability to predators would be:

1. Establish a time series encompassing sufficient inter-annual variation in baitfish abundance to identify “target prey availability” $^{15}N/^{13}C$ envelopes.

2. Develop fishery management objectives, reference and trigger points based on this indicator.

3. Identify potential management measures to ensure prey availability and establish the decision rules.

4. Develop an annual monitoring program to collect and process seabird material.

5. Report outcomes for the fishery and dependent wildlife.

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Wooller, R. D., Cannell, B. C., Bradley, J. S. Valeseni, S. J., Lenanton, R. C. J. and Connard, M. N. *in review* Do the needs of Little Penguins at sea limit their breeding sites? *Ibis*

Abstracts

Listed in alphabetical order by presenter's surname

Monitoring coral reef finfish fisheries - lessons learnt from the reef line fishery of the Great Barrier Reef

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Monitoring of tropical coral reef finfish fisheries is difficult because these tend to be multi-sector, multi-species, and spatially heterogeneous in both fishery and biological dynamics. Associated with these inherent difficulties is the need to consider the value and scale of the resource to be monitored, which traditionally for coral reef fisheries has been limited. The reef line fishery of the Great Barrier Reef, however, is the largest coral reef fishery in Australia, both in terms of value and magnitude of harvest. In recent years, the fishery has undergone significant management changes that have impacted on fisher behaviour and harvest characteristics. This has necessitated the development of a structured long-term monitoring program to assess these changes in relation to the sustainability of the resource. We provide insights into monitoring coral reef finfish fisheries based on our experiences from 10 years of monitoring the reef line fishery as part of a research project designed to understand the effects of line fishing on the productivity and conservation status of key target species of the Great Barrier Reef. We discuss lessons learnt in relation to monitoring, particularly with respect to fishery-dependent and -independent sampling methods, and how these relate to the multi-sector and multi-species characteristics of the fishery. Future monitoring of the fishery must be tuned to specific assessment requirements and decision rules to effectively inform feedback strategies and management decisions.

Cost-effective monitoring of by-catch species in Australia's northern prawn fishery

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In the past decade, management of non-target species in Australian fisheries has evolved substantially. Fisheries have recently come under pressure from legislation, new by-catch policies, external market forces and public perception; to not only reduce their impact on by-catch populations, but to demonstrate that all species are impacted at sustainable levels. In the NPF this is being achieved through new management initiatives and a staged research and development program. Management arrangements include the introduction of turtle excluder devices (TEDs) and by-catch reduction devices (BRDs) in 2000; a ban on shark finning in 2001; log book recording for protected species; effort reductions (e.g. 46% of boat days since 1990); spatial, seasonal and daytime closures; and the introduction of a by-catch monitoring program. The proposed by-catch monitoring program will be an important step in the fishery's move towards its demonstration of ecological sustainability. A collaborative research project between the CSIRO and AFMA is currently evaluating different monitoring methods in order to recommend the most cost-effective and acceptable strategy for conducting a long-term monitoring program for the diverse range of by-catch species caught in the NPF. This includes collecting data on the accuracy, reliability, feasibility, cost and acceptance of five different data collection methods; logbook data, requested industry collections, crew-member observers, scientific observers and fishery independent surveys. A new semi-quantitative risk assessment method has been developed in conjunction with this study to prioritise the monitoring program on species most likely to be at risk from the trawl fishery. This study describes the capacity and limitations of the different monitoring methods to collect suitable long-term data series and will recommend an adaptive strategy using a combination of data collection methods to begin in the tiger prawn season of 2005.

Monitoring of fish communities in deep channel billabongs associated with the Ranger Uranium Mine, Northern Territory

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The design of this monitoring technique entails the pair-wise comparison of fish community data between Mudginberri and Sandy billabongs using multivariate dissimilarity indices. Shifts in fish community structure have been observed in both billabongs from year to year, with some semblance of 'tracking' of the sites. However, a decline in the paired-site dissimilarity measures over time requires explanation and understanding so that natural and mining-related change can be correctly distinguished. Data analysis has shown that particularly high abundances of Chequered rainbow fish (*Melanotaenia splendida inornata*) in Mudginberri Billabong (and not Sandy Billabong) in the early years of the study are mainly responsible for the elevated paired-site dissimilarity measures in that period. This species undertakes very significant migrations in Magela Creek in the late wet season after spawning and recruitment on the (downstream) floodplain. This presentation considers floodplain morphology, floodplain grass communities and discharge patterns as possible factors affecting magnitude and variations in recruitment and migrations of numerically dominant fish species in the two catchments. The potential effect of these factors on the ability to detect mine related changes is examined.

Implications of complex stock structure for fisheries monitoring, management and stocking

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We examined several hypotheses about the stock structure of narrow-barred Spanish mackerel, *Scomberomorus commerson*. Different methods provided information over temporal scales: genetics, whole otolith isotope ratios, and parasite incidence, were analysed for samples taken across Australia's north, and west Timor (Indonesia). An understanding of stock structure is required to clarify responsibility for management of straddling stocks. Such work helps ensure that assessment and management actions match spatial dynamics in temporal and spatial scale. Fish were sampled in 1998 and 1999 from eight locations along northern Australia, between south-east Queensland and Shark Bay (WA). There was strong genetic separation between east coast and northern Australia stocks between which there may be a discrete stock, in Torres Strait. Spanish mackerel are large, active, schooling predators that are open water, broadcast spawners with high fecundity. These attributes might suggest a well-mixed stock over large spatial scales (eg. northern Australia). However, both the otolith isotope and parasite based methods indicated minimal mixing between adult *S. commerson* at scales ~ 100 km. Northern Australian *S. commerson* populations probably consist of a mosaic of small functionally distinct assemblages, connected by larval and juvenile interchange at low levels, and only a small degree of adult mixing: a meta-population.

Management questions and actions must apply to the appropriate spatial dynamic scales. Monitoring and management of *S. commerson* and similar fisheries must be designed to accommodate the risk of depletion of assemblages at small spatial scales. Rates of re-colonisation, the extent of retention of larvae and juveniles in natal populations, and genetic adaptation at the scales of the assemblages are probably unknowable in most management contexts. Monitoring and management sets must therefore be sensitive and robust to these fine-scale problems. Stocking must be only with considerable care.

Genetag: Monitoring fishing mortality rates and catchability using genetic mark –recapture

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Mark-recapture (tagging) is an attractive method of monitoring fisheries, directly measuring fishing rates and catchability, and abundance. Relatively few tags are necessary to provide effective monitoring. The method is fairly robust to spatial complexity and environmental variability. Unfortunately, tagging is usually hampered by three serious limitations: tag shedding, mortality due to capture, and under-reporting of recaptures. Costs of careful capture and tagging of sufficient individuals, plus experimentation to overcome the serious limitations, may be prohibitive. Genetag may overcome these problems. In genetic mark-recapture, individuals are first identified (marked or "tagged") from sampled tissue by micro-satellite DNA techniques (ms-DNA), and then a sample of the known total catch is screened for recaptures. An individual's genotype is permanent (no tag shedding); with very little tissue needed for ms-DNA, the approach is amenable to in situ sampling. Thus biopsies can be taken without imposing mortality risks in capturing the fish for tagging. Sampling a known fraction of the catch (given total catch) can be more tractable analytically than estimating a reporting fraction. A project to monitor a fishery using genetag may be comparable to otolith-based monitoring of age structure. Additionally, a combined genetag/conventional tag approach can be informative and can harness the energies of the catch-release sector of recreational fisheries. A joint NT-Qld-WA project with major FRDC backing is applying and refining the combined genetag approach as a monitoring method, at a fishery scale. We have demonstrated novel techniques for in situ tissue collection and efficient genetic processing and mark-recapture matching. The elements of the genetag approach are feasible. We have now genetagged more than 1000 Spanish mackerel around Darwin, and are poised at the point where, for the first time for this fishery, we might be able to make reasonably precise estimates of fishing rates.

Monitoring subsistence reef line fishing in the eastern Torres Strait

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Subsistence sectors, though rarely studied, account for a significant component of the total harvest of tropical reef fisheries, and should be considered when assessing and monitoring the impact of fishing on shared fish stocks. The eastern Torres Strait (ETS) reef line fishery is comprised of three sectors (Islander subsistence, Islander commercial, and non-Islander commercial) that share the same traditional fishing grounds and reef fish stocks. Lately, concerns about the status of reef fish in the ETS, increasing conflict between the sectors, and the necessity of resolving ever-increasing resource allocation and sustainable utilisation issues have arisen. Island communities have relied on the sea for subsistence and culture for centuries and continue to do so today. Nonetheless, there has been little monitoring of subsistence fishing practices in the region. Monitoring the indigenous subsistence sector in the ETS is critical since any assessment of the total harvest of reef fish in the region should consider all involved sectors, with their different management regulations and fishing practices. In addition, catch characteristics of the subsistence sector have changed in recent decades. Today, Islanders retain for consumption the non-saleable or non-regulated product, such as undersize fish, replacing the fish species that once were traditionally consumed. We discuss issues and protocols associated with monitoring the subsistence sector in the ETS, and analyse the catch composition to detect any spatial (among islands) or temporal patterns in fishing practices. Results from our study will provide protocols to monitor the ETS subsistence sector in the future, as well as, provide a baseline to monitor similar tropical subsistence fisheries. Our results will also be important in characterising a sector of the fishery about which little is known.

Indigenous community marine rangers

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Aboriginal land comprises approximately 84% of the Northern Territory coastline and about 27.5% of the NT population are indigenous Australians. The Northern Territory Government has identified the need for Aboriginal people to participate in managing and caring for marine resources and habitats. To date this has been done to a limited extent through Aboriginal Fishery Consultative Committees. Through these committees, senior Aboriginal traditional owners have advised that they would like more active involvement and participation by Aboriginal people in all aspects of coastal and marine management, particularly in marine surveillance and monitoring as part of an effort to protect marine resources through the deterrence of destructive and illegal fishing activities. Salt-water people are traditionally sea managers who are responsible for maintaining Aboriginal law in their respective areas. Traditional owners have expressed their desire to have Aboriginal people caring for their sea country in a manner that compliments customary management while fitting within contemporary management structures. Following concerns raised by traditional owners and those received by Police to have more surveillance on the water, the Northern Territory Government established the indigenous community marine ranger program through DPIFM – Fisheries Group.

Fishery independent surveys of black lip abalone (*Haliotis rubra*) in the western zone South Australian abalone fishery

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South Australia has three independently managed commercial fisheries for abalone. Within each of these fisheries two species of abalone, *Haliotis laevigata* (Green lip) and *H. rubra* (Black lip) are commercially harvested. Management Plans for each fishery outline performance indicators that are used to measure, amongst other things, the biological status of abalone populations. A number of biological performance indicators relate to measures of the abundance of abalone, with these measures to be taken independently of information gathered from abalone harvested by the commercial sector of the fishery. In the western zone abalone fishery, assessments of abalone populations and measures of performance indicators have primarily been reliant on fishery-dependent data. Due to the pressing need to obtain fishery-independent measures of the abundance of abalone populations within this fishery, areas from which large catches are currently harvested were targeted for survey. Results from initial surveys indicate that the density and pattern of distribution of black lip abalone within sites differs among locations. These results reflect the importance of understanding that the scale of detectable change in abundance of abalone between sampling times is likely to be different among locations. Moreover, these results highlight the need to use fishery-independent measures of abundance in concert with other indicators of the status of the stocks, such as changes in the size frequency of populations, in the formulation of appropriate trigger points within biological performance indicators of the Management Plan.

Queensland indigenous fisheries management and monitoring

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Indigenous people were the first custodians and managers of Australia’s fisheries resources (Coleman et al. 2003). Since the 1950’s, fisheries resources have been managed by State and Commonwealth governments and historically these organisations have not represented Indigenous people and traditional fishing. This is changing. Today, State and Commonwealth governments not only recognise the importance of traditional fishing for Indigenous people, but in many cases have re-written legislation and policies to allow Aboriginal and Islander people to exercise their traditional customs as they apply to sea country and fisheries resources. The Queensland Department of Primary Industries and Fisheries (DPI&F) is also encouraging sustainable economic development in Aboriginal communities through the development of commercial fishing, joint ventures, tourism, aquaculture, charter boat operations etc. An Indigenous Fishing Working Group helps Indigenous communities develop these projects and provides information through meetings, training, site visits and forums etc to ensure that all representatives have an opportunity to pursue their community fishing aspirations. In conjunction with the Indigenous Fishing Working Group and Indigenous communities, the DPI&F is also developing a process for collecting data and monitoring Indigenous take. Fisheries information and data will allow Indigenous people to actively participate in fisheries management with other stakeholder groups (i.e.: commercial and recreational fishing sectors). This paper will outline Queensland’s fisheries management in relation to traditional fishing, Indigenous fishing projects, a process for ongoing engagement with Indigenous communities and monitoring Indigenous take.

The value of an indigenous fishing survey

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A national survey of recreational and indigenous fishing was conducted in Australia during 2000-01. The survey was a joint initiative of Commonwealth and State governments. Grants from the Natural Heritage Trust, Fisheries Research and Development Corporation, State and Territory fisheries agencies supported the project. The national survey was a multifaceted project designed to provide a range of information about non-commercial fishing in Australia. Modified on-site survey techniques were used to collect information from indigenous fishers in northern Australia. An estimated 37 000 indigenous fishers, aged 5 years or older, living in communities in northern Australia fished at least once during the 12 months prior to July 2000. This represented a fishing participation rate of 91.7%. Indigenous fishers in northern Australia expended an estimated 420 000 fisher days of effort during the survey year, comprising 671 000 separate fishing events. Indigenous fishers harvested aquatic animals from a range of environments, but inshore waters accounted for more than half the fishing effort. Indigenous fishers used line fishing methods (53%), hand collection (26%), nets (12%) and spears (9%) as their primary fishing methods. Indigenous fishers harvested more than 3 million aquatic animals from the waters of northern Australia.

Monitoring our oceans – what do we need at the national level?

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A variety of national level performance monitoring and indicator frameworks for the state of our oceans currently exist. Some monitoring is required to satisfy specific sectoral regulatory requirements while other frameworks have been developed to try to monitor the broader “ecosystem health” of our marine systems. To date no single system or amalgam of different parts has been adopted to meet everyone’s needs. The recent combined efforts of Commonwealth, state and territory governments in natural resource management have lead to a more comprehensive and ongoing assessment of the effectiveness of management on land. Applying natural resource management monitoring and evaluation principles across our oceans present challenges. The Australian Government has espoused through Oceans Policy the need for an ecosystem-based approach to managing our oceans, centring on ecosystems and providing for integrated and adaptive management. Regional marine planning incorporates a framework for sustainability indicators for the assessment and long term monitoring against ecologically sustainable development objectives. Understanding and monitoring the oceans is no easy task and little is currently known about offshore conditions and processes. Agencies are building capabilities to efficiently acquire baseline data and to develop accessible and linked systems of physical, biological and socio-economic data. Government has embarked on initiatives to improve our state of knowledge such as through an integrated system for ocean observations and to improve access to information through the development of an Oceans Portal. What is required is for all sectors and jurisdictions (including government / industry partnerships) to better work together in data collection, analysis and refinement of our data needs. Together our ultimate goal must be to provide a meaningful understanding of what is happening in our oceans.

Monitoring the catch and effort in the western rock lobster commercial fishery

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The western rock lobster, *Panulirus cygnus George*, constitutes one of Australia's most valuable single-species fisheries (worth about AUD$ 350 million annually) with annual catches averaging 11 500 t. This species, which is endemic to Western Australia, is found predominantly in coastal waters from North West Cape (21^{o}45 S) to Cape Leeuwin (34^{o}22 S), in depths of less than 100 m. This fishery is fully exploited and its effective management relies heavily on annual stock assessments derived from data collected from different monitoring programs. These programs are conducted either by commercial fishers (e.g. fishers complete voluntary daily logbooks and fishers and processors both provide compulsory monthly returns), or research staff (e.g. annual breeding stock survey) or a combination of both (e.g. research staff monitor the composition of the catch on board commercial vessels). The main aim of all these monitoring programs is to produce accurate catch and effort data used to produce four abundance estimates, critical in the assessment of this fishery, namely the abundance of post-larval recruits, abundance of pre-recruits, abundance of recruits to the fishery and indices of egg production. Whether produced by fishery dependent or independent data sources, catch or effort estimates can often be subject to inconsistencies and bias. For example, in the case of fishery dependent monitoring programs, advances in technology (e.g. colour sounders and GPS plotters) increase fishing efficiency, which is not incorporated into an effort measure derived solely from pot lifts. Biased data such as this can lead to overly optimistic estimates of stock abundance. This paper examines the pros and cons of fishery dependent and independent monitoring programs and the approaches taken to obtain catch and effort estimates, the problems associated with inaccurate and biased data and the process employed to correct these data sets.

A strategic monitoring program for Queensland's fisheries

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Queensland's fisheries are diverse in terms of species harvested and fishing gear employed. They are, in many cases, multi-sectoral with commercial fisheries having an annual gross value of production of more than $350m and annual expenditure by recreational fishers estimated at more than $300m. The state-wide resource monitoring program which has developed since the late 1980s to provide advice to fisheries managers includes a mix of logbooks and diaries, fishery dependent and independent survey techniques. Beyond allowing trends in catches and catch rates for commercial fisheries to be assessed, biennial surveys of recreational fishers allow changes in participation to be monitored and trends in catches of major recreational species to be identified. In many cases, the same resource is harvested by multiple sectors and the monitoring strategy in place allows a comprehensive assessment of the sustainability of major resources to be made on a regular basis. The strategy is meeting the needs of Fishery Management Plans and assists in demonstrating that Queensland's fisheries are being managed in an ecologically sustainable manner as required by the Commonwealth Environment Protection and Biodiversity Conservation Act. An overview of the monitoring strategy and the basis on which its priorities have been set will be provided.

Drought impacts on marine ecosystems of the southern Gulf of Carpentaria

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Commercial logbook data for the pot, set-net, and line fisheries along the Queensland coast of the Gulf of Carpentaria have shown a cascading pattern of decrease then recovery in catch over 2000 to 2005 period. Climate information for the southern Gulf of Carpentaria shows a similar, although leading pattern that is apparently related to the severe drought over the 2001-2003 period. An immediate impact of the downstream effects of the drought can be seen in the dieback of inshore seagrass beds. Short-lived rapid turnover species and species that exhibit onshore-offshore migratory behaviour were the first to show the effect of this drop in coastal primary productivity, and conversely the first to recover as the seagrass returned. Longer lived resident fish species were slower to show a decrease in catch and conversely longer to show a recovery. For some recreationally important species it was possible to confirm these patterns via a fishery independent sampling program and through recreational/charter boat records. Although exact cause and effect relationships will be complex, a correlative analysis shows an interrelationship of the climate, coastal primary productivity, and productivity of a cascade of commercially fished species.

A review of current ecological risk assessments and their value in optimising monitoring programs

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The EPBC Act 1999 is relatively new legislation that has had a considerable influence on the management of Australian export fisheries, by regulating them to operate in a more ecologically sustainable manner. Consequently, several fisheries that interact with large numbers of species (e.g. by-catch in trawl fisheries) are considering ecosystem-based approaches to fisheries management to conform to this legislation. Sustainability of fished populations can be demonstrated via long-term monitoring programs having pre-defined reference and trigger points. However, some fisheries like Australia's northern prawn fishery (NPF) interact with hundreds of species with varying life strategies, including teleosts, elasmobranchs, turtles, sea snakes and a host of motile and sessile invertebrates; many of which are rarely caught, low value and data-poor. Monitoring of entire diverse communities would be expensive, impractical or impossible. We suggest that ecological risk assessment may be the only way to focus the design of monitoring programs to control their cost-effectiveness without sacrificing their coverage of the most critical species. We review the methods available to assess sustainability of individual species in diverse, data-limited communities. We employed a recently developed and widely used ecological risk assessment method called productivity-sustainability analysis (PSA), to assess elasmobranch sustainability in the NPF after the introduction of turtle excluder devices into the fishery. Our results demonstrate that the current PSA risk assessment method may be inadequate to identify the true species at risk of overfishing, and their inclusion in monitoring programs is limiting. We conclude that the current ecological risk assessment methods be improved to provide more quantitative measures of impact on individual species by fisheries.

Methods for monitoring the abundance and habitat of the northern Australian mud crab *Scylla serrata*

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The mud crab, *Scylla serrata*, is an important "icon" species, found in northern Australian estuarine and inshore waters throughout the year. Recreational, commercial and traditional fishers all covet the species and the combined interest results in consistently high fishing pressure. Stock assessment based on commercial and recreational catch statistics and estimates has not proved informative, as the underlying assumptions required for assessment methods and models based on such data cannot be met. High variability in growth rates and/or mortality has also made length-based assessment methods difficult and as yet, no stock estimates are available for Australian mud crab fisheries. The aims of this project are the identification and quantification of critical mud crab habitat; and to develop and assess techniques for estimating mud crab abundance. A significant achievement of this project has been the completion of mapping of coastal wetland habitats using remote sensing techniques, providing complete broad-scale coverage of mud crab habitat in the Northern Territory and Queensland. The identification and quantification of northern Australian coastal wetland habitats will benefit a broad range of northern Australian inshore fisheries and this work has been incorporated into a geographical information system permitting a much wider application across a variety of natural resource management agencies and issues. Animal abundance survey and analysis methodologies, based on mark-recapture techniques, have been developed to estimate mud crab density for two key habitat types in northern Australia. Density estimates for each habitat type were extrapolated up across adjacent regions in each state providing the first broad scale estimates of mud crab stock size.

Demonstrating sustainability for rarely-caught trawl by-catch species

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The challenge of demonstrating the long-term sustainability of rarely-caught trawl species presents an almost intractable problem for fisheries managers. The application of a robust quantitative risk assessment based on sound data can help to identify species requiring monitoring for demonstrating their long-term sustainability. However, it is likely that many of the species identified by this process will be the rarely-caught. In tropical prawn trawl fisheries with a large diverse by-catch, many species are rarely-caught, and require the sampling of thousands of trawls in order to detect meaningful short term changes in relative abundance. Many fisheries have only limited financial capacity to fund costly monitoring programs, and alternative strategies for demonstrating sustainability of rare species are needed. One monitoring option that can greatly reduce the high cost of large annual surveys is to accumulate data over a period of years from modest-sized annual surveys, and consequently can provide an ability to detect changes over time. The use of surrogates to represent either similar regions or similar species eco-morpho-types is another option that can improve the cost-effectiveness of monitoring programs. However, the effectiveness of such surrogates remains untested. A further option for trawl fisheries unable to fund costly annual surveys is to permanently protect the low effort trawl grounds in order to enhance the sustainability of rarely caught species. This should provide a non-impacted habitat for by-catch species to rebuild their populations, including the regrowth of macrobenthos in suitable habitats. Effects of fishing studies suffer from a universal absence of un-trawled control sites and the closed areas should recover in the long term, allowing useful comparisons of by-catch communities to be undertaken in the future. This option may well prove to be the most cost effective action that can be taken to ensure the long-term sustainability of the rarely- caught by-catch species. However, even this course of action will require monitoring to demonstrate its effectiveness as a long term sustainability strategy.

Responsiveness of monitoring programs to changes in management and legislation

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The design and implementation of fisheries monitoring programs to provide data for resource assessment is complicated. The ability for these monitoring programs to evolve over time in response to changes in legislation, management regimes and technological advances is a challenge. The Department of Primary Industries and Fisheries Queensland, Fisheries Long Term Monitoring Program was initiated in 1999 to monitor 11 Queensland Fisheries. Since the program was initiated, we have addressed the challenge of meeting additional needs of new and revised Fishery Management Plans and changes to government priorities. Recommendations for enhancement of existing fishery independent monitoring resulting from Ecological Assessments by the Commonwealth Environment Portfolio under the Environment Protection and Biodiversity Conservation Act are also being implemented. This presentation will discuss how the Long Term Monitoring Program has successfully been able to address evolving objectives, incorporate outcomes from research projects and some of the obstacles we have faced trying to meet our previous and new objectives. It will cover on-ground changes such as obtaining adequate resources, staff moral, support systems and monitoring review processes.

Evaluation of the effectiveness of spatial fishing closures in the Queensland saucer scallop (*Amusium japonicum balloti)* fishery 1997–2004

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In 1996 the catch rate and total catch of scallops (*Amusium japonicum balloti*) from the Queensland scallop fishery declined markedly. In response to the decline, the Department of Primary Industries and Fisheries (DPI&F) implemented 3 permanent spatial fishing closures, referred to as scallop replenishment areas (SRA's), to reduce fishing mortality on spawning stocks and enhance recruitment to the fishery. In 2001 the area covered by the SRA's was doubled and a rotational harvest strategy developed to permit fishers access to the high densities of scallops within the SRA's. In conjunction with the creation of the closures in 1997, DPI&F established a series of annual fishery independent surveys to monitor the effectiveness of this management strategy. The performance of the SRA's and the rotational harvest strategy was assessed by investigating the relationship between scallop abundance (i.e. standardised catch rate), and closure duration (weeks closed prior to survey) using generalised linear models. The results show a significant relationship between scallop abundance and closure duration, with abundance continuing to increase for closed periods up to 4 years. This result suggests the potential of these areas to act as a source of egg production increases with duration for periods of up to 4 years.

Abundance estimation of black lip abalone, *Haliotis rubra*, using a modified radial transect technique

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A considerable challenge faced when conducting benthic dive surveys, is that of effectively and efficiently arranging sample units in a random manner. A novel and efficient technique employed in the Victorian black lip abalone (*Haliotis rubra*) fishery uses transects that radiate outwards from a central point. In this arrangement however, individual transects lack independence due to their convergence toward the centre of the site, and strong spatial biases exist in the distribution of sampling effort. To overcome this problem, modifications to the radial transect technique were developed, which incorporate the efficiency benefits of the Victorian method whilst ensuring that sampling effort is distributed in an almost random manner. Intensive field trials were undertaken to compare the modified radial method to more conventional parallel transects. Both techniques yielded generally similar results in terms of density estimates, precision and levels of spatial autocorrelation. However, cost-benefit analysis showed that radial transects substantially outperformed those arranged in parallel, due to their efficiency of application. The new method was subsequently applied to abundance surveys of black lip abalone, *Haliotis rubra*, throughout a range of Tasmanian habitats, where it proved to be both effective and versatile. On the basis of these results, it is concluded that the modified radial transect method represents not only a valid abundance estimation technique, but one that is particularly efficient and suitably versatile when applied to dive surveys of benthic species.

The integrated scientific monitoring program (ISMP) – design and data

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Australia's southern and eastern scale fish and shark fishery is a complex multi-species, multi-gear fishery which operates in offshore waters off southern and eastern Australia. ISMP was developed to collect extensive information on the quantity, size and age composition of the retained and discarded catch of species caught in this fishery. To meet this objective, on-board field scientists sample the retained and discarded catches taken by vessels, and port-based fish measurers sample catches landed in the major ports. To ensure a statistically robust design, logbook and monitoring data were used to stratify the fishery based on species composition, gear and port groups with the aim of providing discard rates of all quota species and pooled non-quota species in each stratum. Efficiently allocated sampling effort ensures that the data are representative of the spatial and temporal dynamics of the fishery and its various fishing methods and that target precision in estimates of the total catch (retained and discarded) of quota and non-quota species are met. Annually, the ISMP covers about 60 vessels. Observers spend approximately 500 days at sea, and measure more than 100,000 fish. Port based staff measure about 100,000 fish per year. In addition, the ISMP collects about 10 000 otoliths per year. Data collected are used by industry and fisheries assessment groups, as well as various government agencies, consultancies and universities for a number of different purposes including catch summaries, ecological risk assessment, by-catch action plans and ecosystem modelling.

Integrated fisheries management in Western Australia – a significant challenge for fisheries scientists

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The advent of the 21st century has seen the Department of Fisheries, Government of Western Australia embark on an ambitious initiative of Integrated Fisheries Management (IFM) within the broad context of the principles of Ecologically Sustainable Development (ESD). This initiative consolidates the outcome of earlier important initiatives such as the freeze on the issue of commercial fishing boat licences in 1983, and more recently the development of regionally based management strategies for recreational fishing, the formal management of charter fishing, and the recognition of the importance of indigenous fishing. An initial challenge for finfish scientists was the development of a means of prioritizing the expenditure of research effort directed at the important task of determining sustainable harvest levels for key "indicator" species within each Bioregion of the state. It is anticipated that at times, such determinations will need to be made in the absence of adequate data. The ongoing monitoring of the catch shares allocated to each sector also poses a significant challenge. Methods being developed to handle these challenges, and other important future needs identified as a consequence of embarking on this process are discussed.

When should a manager start to worry: Monitoring the response of threatened fish (Macquarie perch) populations to fire, river regulation and drought

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Macquarie perch are listed as a threatened species in both national and State/Territory conservation listings, and are listed as endangered and vulnerable in the ACT and NSW, respectively. A monitoring program was established for the species in the Canberra region in 2001 with bushfires in 2003 and drought since late 2000 affecting the streams of the region. One of the Macquarie perch populations monitored is the result of a translocation of fish into the Queanbeyan River in 1980, with other monitored populations located in the Goodradigbee and upper Murrumbidgee rivers and Cotter Reservoir. This paper will present the results of a species-specific monitoring program conducted between 2001 and 2005, review the best methodology for sampling the species, and hi-light the disparate responses of this endangered fish to a range of environmental perturbations. Populations in the Cotter River have been subject to significant impacts from fire, sedimentation and river regulation, but are recruiting under an environment.

Fish and river health in the Murray-Darling Basin: the sustainable rivers audit

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In 2004, the Murray-Darling Basin Ministerial Council agreed to assess the Basin's river health through a sustainable rivers audit (SRA). A pilot audit was run to develop and test five themes (macro-invertebrates, fish, water quality, hydrology and habitat) across four valleys (Condamine, Lachlan, Ovens and Lower Murray). The fish theme's primary aim was to establish and trial standardised methods for fish bio-assessment across the Basin. Fish provide ideal assessment tools for long-term, broad-scale monitoring programs, as they are easily identified, relatively abundant, valued by the general community and sensitive to changes in river health. During the pilot audit, a total of 13 952 fish from 27 species were caught. Electro-fishing proved to be the most efficient of three different sampling methods. An SRA index of fish community health has been developed incorporating three sub-indices containing information on fish species richness, 'nativeness' of the fish community and diagnostic indicators. The indicators and sub-indices are combined using codified expert rules. Following the pilot audit, the fish theme was accepted by the independent sustainable rivers audit group as a cornerstone of the SRA. The first sampling season of the full SRA has since been completed, with eight of the Basin's 23 valleys sampled. The pilot audit has enabled refinement of the SRA's fish theme and validated its use for large-scale, consistent monitoring of river condition and trends in a diverse range of rivers.

Fishing industry data and monitoring priorities

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Industry and governments investment heavily in research aimed at fisheries sustainability. However, achieving a sustainable and internationally competitive seafood industry requires much more than sustainable fish stocks. There is a critical need to address knowledge gaps in areas not traditionally seen as a priority by fisheries researchers, or suffer ill-planned restrictions being imposed on the fishing industry, purportedly in the name of sustainability, with increasing frequency. As the industry's substantial social and economic contribution is undermined by such decisions, the scenario of healthy fish stocks but no fishermen to harvest them seem a distinct possibility in some areas. The importance of research to underpin appropriate stock assessments and the knowledge needed to achieve ecological sustainability of fisheries cannot be questioned. However, a more appropriate balance in R&D investments is needed. All industries need to be internationally competitive to survive in today's global economy. This also demands R&D investments throughout the seafood supply chain including production efficiency, market research and development, quality, food safety and environmental management systems, etc. Fisheries researchers opposed to R&D investments being directed towards these "supposedly" non-traditional areas of fisheries R&D, could contribute much more towards achieving sustainability by embracing it and leading the way. Fisheries management decisions which ignore the social and economic aspects of the industry and seek to mitigate negative impacts are also likely to be overridden with increasing frequency as the industry is forced to seek relief through the political process. A case study is used to demonstrate that substantial progress can be made through more effective use of existing data.

Monitoring and management initiatives for the sustainability of the fisheries of Lake Victoria, East Africa, with emphasis on the Nile perch (*Lates niloticus*) fishery

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Lake Victoria, is the largest freshwater lake in Africa (second globally to Lake Superior in USA), and has the most productive inland fisheries. The Lake is shared between the three East African

Community Partner States (Kenya, Uganda and Tanzania). The annual catch is estimated at 554,986 t of which over 60% is Nile perch (*Lates niloticus*). The fishery is valued at US$ 544 million locally with exports in 2003 at US$ 270 million. The lake supported a large number of endemic fish species before the mid 1980s when Nile perch, which was introduced in 1950s and 1960s, dominated. The fishery changed from multi-species to three species fishery: Nile perch, tilapia (*Oreochromis niloticus*) and dagaa (*Rastrineobola argentea*) with a five-fold increase in landings. The fishery attracted has been attracting very large numbers of fishers, so that the number of boats on the lake has grown from around 12 000 in 1983, to 52,479 in 2002. Nile perch processing factories that absorb over 170 000 t of wet fish per year were established and export market for the Nile perch worth about US$ 250 million annually developed. The economic benefits accruing from the fisheries have helped all the stakeholders including governments, to focus on the issues of the sustainability of resource use. The Lake Victoria Fisheries Organization is mandated to coordinate all management and development activities of the fisheries resources of the Lake. Under the regional body, Institutional structures and processes are being strengthened. Co-management institutions are being established at the grass-root level, where Beach Management Units (BMUs) are empowered and facilitated to become partners in management. Efforts to revive the monitoring systems are underway. The paper summarizes the initiatives undertaken in monitoring and management in ensuring the fisheries resources and their socio-economic benefits are sustainable.

Caring for country – indigenous customary values and use of aquatic and marine resources across north Australia

Joe Morrision

North Australian Indigenous Land and Sea Management Alliance (NAILSMA), Charles Darwin University.

Indigenous Australians have occupied the Australian continent and actively managed its resources for millennia. Active management of country and its resources by Indigenous people reflect dependence and most significantly of the purpose of this forum, understanding the complex array of values that constitute customary economic activity.

Today, across north Australia's wet-dry tropics, indigenous people play a significant role in population and land ownership, particularly away from heavily populated areas such as Broome, Darwin, Cairns and Townsville. During the 20th century, the passage of land rights and native title legislation has rapidly expanded the indigenous-owned estate. A rapidly expanding indigenous population means Indigenous people's aspirations cannot be taken for granted. In some remote parts, the Indigenous population comprise upwards of 90%. For example, in the Northern Territory indigenous people comprise 30% of the population and own 45% of the landmass, and 87% of the coast.

Indigenous people like those that comprise many other 'sectors' require access to native flora and fauna. However, they differ in their views, the types of resources they exploit and the intensity at which they exploit them. Their attachment to these resources is significantly different – they have attachments that have been conceived over much longer timeframes than most Australians. These differences complicate interactions with others seeking to make use of the same resources, and demand close engagement with other ‘sectoral’ groups in making decisions about use of natural resources. This presentation will articulate the importance Indigenous customary resource use and management to derive economic, social, spiritual and cultural benefits for people on country.

An integrated approach to monitoring fisheries stocks, fish habitats and threats to fisheries

Kerry Neil

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Effective management of fisheries and its assessment are underpinned by the collection of validated data through monitoring programs. Monitoring programs supporting the management of coral reef finfish stocks in Queensland have historically focused on collecting fishery independent and dependent data related primarily to the harvest dynamics, biological and ecological aspects of the fishery including species composition and size distribution of the fish populations available to the fishery, and catch and effort data for sectors influencing the coral reef finfish stocks. Monitoring strategies put in place to meet legislated management requirements have been limited in their capacity to collect data that describe threats to the fishery, such as habitat degradation, as well as socio-economic drivers of participation and behaviour in the fishery. Furthermore, monitoring programs have typically been disjointed and rarely consider or integrate data collected through other programs. With the adoption of an ecosystem management approach there is a concomitant need to adopt an integrated approach to fisheries monitoring. Assimilating data from related resource management and socio-economic programs would enable examination of key changes of a greater range of ecological and socio-economic influences on fish populations. This would provide greater support for the adoption and assessment of management strategies that can promote the ecologically sustainable use of the coral reef finfish stocks. The relevance of an integrated approach to the management of Queensland’s coral reef finfish stocks will be discussed.

Estimating the detection probability for boat electro-fishing

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Detection probability is an important parameter for estimating indices of abundance in fisheries monitoring. This probability is generally ignored and treated as a constant in analysis that comprises samples from boat electro-fishing. This is largely due to the inability to close a population in large water bodies where boat electro-fishing is deployed. We present a method whereby detection probabilities can be directly estimated. Marking fish with radio-telemetry tags creates an experimental design that satisfies the assumptions of a closed population. On each subsequent sampling occasion the number of radio-tagged fish available for capture in a particular area can be determined by tracking and locating all tagged fish. Detection probability is estimated by analysing the number of fish caught from the known available population. We trailed this method in the Murray River in 2004. Eighty-four fish were radio-tagged and the site was sampled on 4 independent occasions. Results showed that the probability of detection varied according to depth, fish size and there were dependencies between the sampling occasion (explained in terms of fishing time and woody debris). Fish movement was modelled and indicated that the study area needed to be large (approximately 2 km) and within site movement induces dependency on recapture probabilities. Individual effects were also observed. These results suggest that these factors would be a challenge to model in a standard mark-recapture analysis. The use of radio-tagged fish resolves these challenges by directly modelling the influence of the dependencies.

The potential use of nitrogen and carbon stable isotope ratios in seabirds to monitor predator-prey relationships in baitfish fisheries

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In the move towards ecosystem-based management there is need to find efficient methods of detecting changes in the availability of fish prey to predatory wildlife, where this could be influenced by a fishery. This is particularly important in fisheries targeting shoaling “forage” of bait fishes that are the main conduit of energy between primary production and higher order consumers in pelagic ecosystems. Seabird populations with point centred foraging constraints may be sensitive to short term, local depletion of prey resources and therefore be an appropriate ecological indicator of trophic interactions with fisheries. This presentation reports on a small, self- funded, pilot project looking at the stable isotope ratios of carbon and nitrogen in baitfish and seabirds and their potential use as indicators of prey availability. The techniques employed involved little field time and were non-destructive, using discarded tissues such as the shell membranes from hatched eggs, mesoptile feathers from chicks and adult moult feathers. It will be argued that these techniques may offer a defensible and affordable method of monitoring the trophic impacts of fishing in some fisheries.

The indigenous subsistence fishing survey kit: a tool for community scale monitoring surveys

Michael Phelan

DPIFM, GPO Box 3000, Darwin, NT, 0801. Formerly, Balkanu Cape York Development Corporation and Queensland Department of Primary Industries and Fisheries.

The development of the national indigenous subsistence fishing survey ignited much debate regarding the scale of the monitoring survey. Many prominent indigenous organisations argued that the national survey would not generate community ownership of the survey results, but instead would remove community control of the monitoring data. In this session, I discuss an alternative approach to monitoring indigenous subsistence fishing. I discuss a community scale monitoring survey conducted in collaboration with the Inijinoo Aboriginal community in far north Cape York, Queensland. The survey adopted the indigenous subsistence fishing survey kit developed by the Balkanu Cape York Development Corporation, the Queensland Department of Primary Industries and Queensland’s Environmental Protection Agency. The survey kit was designed to allow individual communities to undertake subsistence fishing monitoring surveys in a manner deemed appropriate by the members of the community.

Utilisation of GIS spatial statistical methods and fuzzy rule-based modelling to assist in the development of ecosystem based fishery management strategies

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In recent years the Australian fisheries organisations have moved towards ecosystem-based fisheries management in recognition of concerns about the impact of fishing activities on the marine environment and the need for a holistic management policy rather than emphasis on management of target species. However many commonly used assessment techniques are not suited to the complex interactions which occur within an ecosystem. We believe that GIS spatial statistical methods and fuzzy rule-based modelling are well suited to this type of analysis and can handle both quantitative and qualitative data. This presentation is an overview of an FRDC funded project, which is about to commence. This project is aimed at investigating the feasibility of these techniques in the Northern Territory's offshore snapper fisheries. The advantages of GIS spatial statistical methods and fuzzy rule-based modelling are, (1) the visual nature of GIS makes it an ideal tool to more effectively engage stakeholders and include them more fully in the decision making process, (2) GIS spatial statistical methods are well suited to handling data on different spatial scales, (3) fuzzy rule-based modelling is one alternative to binary logic that provides a mathematical framework to account for partially fulfilled properties, and is compatible to the quantification of human knowledge, (4) traditional statistical methods are not well suited to small data sets and the bias associated with them whereas fuzzy rule-based models rely on mapping of associations between dependent and independent variables and can handle small data sets and "vague" data.

Developing fishery-independent sampling tools for surveys of estuarine icthyo-fauna in New South Wales: an experimental framework

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Fishery-independent surveys are becoming a key-tool in the scientific assessment of many important stocks of fish and invertebrates worldwide. However, before large-scale and long-term surveys can be implemented, it is necessary to do several pieces of important research. We present a logical framework for doing this research based on pilot studies incorporating manipulative experimental approaches. This includes, (1) identification of suitable sampling gears for target species, (2) testing different gear configurations and sampling practices to ensure that samples of target species are optimal, representative and cost efficient, (3) understanding spatial and temporal scales of variability across different strata and (4) cost-benefit analyses to determine optimal levels of replication. We provide examples of the first two stages of this framework based on initial experiments from a research program currently developing fishery-independent surveys of estuarine fish in New South Wales. This approach can be applied elsewhere and we highlight the value of this type of pilot work as a precursor to fishery-independent monitoring studies in general.

Evaluation of precision and cost effectiveness of bus route, creel and phone surveys for estimating total recreational catch

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The majority of recreational angling in Victorian bays and inlets occurs in Port Phillip Bay, where 95 % of the harvest is taken by anglers in boats. Three alternative survey methods have been used to estimate the total recreational catch from boat-based angling in Port Phillip Bay: an off-site phone survey and on-site bus route and creel surveys. These ad hoc estimates demonstrate the total recreational harvest of snapper (211 t), King George whiting (93 t) and flathead (395 t) in Port Phillip Bay during 2000-01 exceeded commercial catches of 5385 and 23 t, respectively. While logbook monitoring of commercial catches provides a continuous time series of catch data, routine estimates of total catch from the recreational sector have never been obtained. This project aims to evaluate survey methods for monitoring recreational harvest. Monte Carlo simulations were used to calculate the total catch across a range of sample sizes using estimated probabilities and distributions from previous recreational fishing surveys. The estimated total catch remained constant with increasing sample size for all survey methods, however, the precision increased with more samples. Assessment of the cost effectiveness of each survey method was made using the simulated precision and estimated survey costs. The cost of conducting a phone survey was considerably lower for the number of samples required to achieve reasonable precision, making this a cost effective survey method. The information obtained from the simulations will be used to design a precise and cost effective monitoring program to estimate recreational catch from Port Phillip Bay.

What do our stakeholders want from us: a personal recreational perspective

William Sawynok

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At the heart of the change process are two things. What needs to change and how it should be done. Researchers provide data that is often the basis for change and there are two things that can be done to improve the current processes that involve researchers. Greater involvement of recreational fishers in monitoring and research and a greater focus on social research will lead to improved support for fisheries researchers and management change. When research is proposed that will ultimately lead to change for recreational fishers they should be involved from the outset in the planning, data collection and distribution of information. If this is done right those involved will become the greatest supporters of the outcome of the work, and probably the greatest advocates for change. There is a growing recognition of the success of this approach, not only in fisheries but also in the wider field of natural resources management. There needs to be an improved balance between research on the “what” and on the “how”. Fisheries researchers are very good at collecting data on

the "what", fish and fish stocks. While there is a growing recognition that communication and extension are integral parts of the "how" it goes beyond that. Less accepted is the need for social research into fishers themselves. Knowing about recreational fisher's attitudes, practices, behaviour and values can go a long way to implementing change in a less confrontational way.

RAS: an intranet-based resource assessment system for multi-species fisheries monitoring and management

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Strategic environmental assessment, and the associated improvements to fisheries management, has resulted in a greatly expanded program of stock status and stock assessment reporting in NSW. There are now 50 primary/target species and 36 key-secondary species that require various levels of internal deliberation and public reporting (excluding the recreational fishery). To tackle this challenge, a new electronic reporting system has been designed and implemented – RAS (the Resource Assessment System). RAS is an intranet application that provides simple access to 'pages' of structured information about a species. Pages are based upon templates and are composed of content generated from html strings, image files, SQL database queries or XML fragments. These content elements enable the construction of pages with free-form text, images (vector and raster), simple query-based tables and arbitrary html code (including more complex tables). Templates enable the same structure to be applied to different species, but with varying content. Pages can be authored by an individual or group, and have an associated status that could be used to constrain access. Content is exportable to Microsoft Word format or portable document files (PDF). Assessment scientists have been slow to recognise the ramifications in reporting that are likely to result from web-based technologies. RAS is an example of how such technologies can assist the organisation, presentation and maintenance of the information and data associated with stock monitoring and assessment. The largest hurdle that most institutions will face in the implementation of such systems is the commitment to database technology required.

WA Research Angler Program (RAP) - getting recreational anglers involved in fisheries science

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The demand for information about the status of fish stocks is ever increasing. In WA, annual data is required to sustainably manage commercial and recreational fisheries, monitor the allocation of catch between sectors, meet ESD guidelines and assess environmental impacts on stocks or ecosystems. Historically, commercial fishers provided much of this data, with relatively little being obtained from recreational fishers. Hence, the gradual withdrawal of commercial fishing effort in recent decades, especially from coastal and estuarine areas, has resulted in less data available for stock assessments. RAP aims to partly address this problem by increasing the involvement in fisheries research by recreational anglers. The focus of RAP is the collection of scientific data, and community education is a consequence rather than an objective. RAP is not a research project, but rather a framework that supports various volunteer-based research projects like angler logbooks, biological sampling, tagging studies and collection of fishing club and competition data. Advantages of RAP include a single, continuous point of contact for volunteers who seek information about various research projects and how they can be involved; administrative support for scientists (printing newsletters, organising seminars, data entry, etc); cost-effective volunteer administration; and higher quality feedback to anglers, (which is essential to the success of all projects). Some recent WA volunteer-based projects and examples of data collected will be discussed.

Outcomes of the national barramundi workshop

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The barramundi is considered the icon species of recreational fishing across northern Australia and also supports a valuable commercial fishery. The 2005 national barramundi workshop, which was held in Darwin from 6 to 8 July 2005, revisited the outcomes from the 1986 international ACIAR barramundi workshop and explored current and future issues affecting the barramundi resource. Discussion focussed on issues including the stocking of impoundments and natural waterways, the impact of recreational fishing, and the take of breeding female fish. The workshop conducted three discussion groups based on research, management and stocking. The research group developed a model for northern Australian barramundi stocks and used this model to explore the value of various fisheries dependant and independent data. The management group discussed methods of aligning the management arrangements across northern Australia as well as providing a list of research requirements to answer stock and allocation questions. Stocking members assessed stocking parameters in open and closed systems, identified areas requiring future research and agreed on complimentary policies between the states.

Fishery-dependent monitoring of two species of migratory schooling pelagic fish

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The Queensland Department of Primary Industries and Fisheries' Long-Term Monitoring Program collects data for many of the State's fisheries resources. These data are used to assess the status of stocks and the effects of different management strategies. In mid 2004, the monitoring of east-coast Spanish mackerel, which previously concentrated on collecting data from commercial fishers targeting spawning aggregations, now covers a wider spatial area (Cairns to Coolangatta) and collects information from the both the recreational and commercial sectors. In addition, the program now includes spotted mackerel. The expansion of the monitoring program for mackerel eventuated after recent management changes to both species. For example, bag limits have been reduced, commercial quotas have been introduced, commercial netting of spotted mackerel has been banned and the minimum legal size limit for spotted mackerel has been increased. In this presentation we concentrate on describing the objectives of the monitoring program and some of the main issues we faced when expanding the monitoring program for mackerel. In particular, we will outline strategies we have considered for collecting fishery-dependent data for these migratory, schooling, pelagic fishes.

By-catch mitigation in the Pilbara trawl fishery

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There have been interactions between dolphins and turtles and the fishing gear of the Pilbara trawl fishery for the last 30 years. The process of development of sustainability reports has resulted in increased public attention focussed on the deaths of an estimated 50 to 70 dolphins per year and a similar number of turtles. Acoustic pingers failed to reduce the number of dolphins sighted inside the trawl net when pingers were alternately deployed and not deployed during daylight hours. A flexible selection grid made from polypropylene pipe was trialled for five one-hour shots. The grid did not maintain its shape, resulting in no significant scale fish catch during trials. An oval shaped semi-rigid grid 1200 mm by 2000 mm with bar spacing of 155 mm was manufactured from stainless steel tube and braided stainless steel wire. Articulated joints enable the grid to be wound onto the net drum. Video footage showed dolphins backing down inside the net to within 3 m of the grid and then swimming out. One dolphin was caught when the grid was deployed, but this animal was in the cover net of the escape opening. No turtles were caught when the grid was deployed but this was not significant result. The catch of sharks over 1400 mm and rays of width over 40 cm was greatly reduced when the grid was deployed. The reduced catch of sandbar sharks, which are currently over-exploited in Northern WA, is an especially important result.

Recreational fishing surveys: which method should I use?

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Various survey methods have been used to provide estimates of the recreational catch. These include creel surveys, phone surveys and mail surveys. Different methods have been shown to provide conflicting results due to inherent biases. This was the case with phone-diary surveys and phone surveys that relied on the respondent's ability to recall information (Lyle, 2000). However, there have been few comparisons of other survey methods. Results from a roving creel survey and bus route survey were compared with a phone-diary survey – the national recreational and indigenous fishing survey (NRIFS) (Henry and Lyle, 2003)). The catch estimates from NRIFS and roving creel survey were significantly different at the 0.01 level, with NRIFS results over two times larger than the roving creel survey results. The roving creel survey under estimated catch and effort since some locations were missed and night fishing was not included. The estimates of fishing effort from the bus route method and NRIFS were similar and the catches were not significantly different at any reasonable level. More work is required to calculate errors associated with the NRIFS catch and fishing effort estimates. When choosing a survey method the researcher must take into account availability of a sampling frame, biases, cost, spatial scale of the fishery and times when fishing effort occurs. Furthermore, different survey methods will suit different recreational fisheries.

History of research and management of snook in Florida

Ronald Taylor

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Snook occur in Florida south of the 14º C winter isotherm and sustain robust populations along both coasts. Prior to 1950, common snook supported recreational and heavily exploited commercial net fisheries that contributed to precipitous declines of both populations. In 1957, snook were declared ‘game-fish’, (their sale prohibited), and bag and size limits were established. After a hiatus in research during the 60s, snook’s popularity precipitated an intensive tagging program during 1975-1986 along southwest Florida. This study revealed a declining unstable population which prompted a reduction of the bag limit, an increase of legal minimum size limit, and harvest closure during the spawning season. Life-history studies in Florida during 1986-1992 showed that common snook are protandric hermaphrodites and that each coastal ‘stock’ differed significantly in many biological parameters. In 1992, the Florida Marine Fisheries Commission (MFC) established a 40% spawning potential ratio (SPR) as a management goal for the entire population. Genetic studies in the mid 90s validated the existence of Atlantic and Gulf stocks, whereupon MFC promulgated separate coastal regulations with the Gulf’s stock rules being more stringent because of higher exploitation. Since 1994, scientists have conducted nine snook stock assessments and determined that both stocks have consistently remained below the target SPR because of overfishing, despite systematic changes regulating harvest. About 60% of Florida’s daily influx of 1200 new residents settles along the coast, leading to increasing fishing effort and inexorable declines of coastal wetlands and mangrove forests. Thus, scientists and managers may be constrained to a permanent restrictive management posture.

Ecological monitoring and research – a useful separation or a detrimental tautology – lessons from the South Australian sardine fishery

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The South Australian sardine fishery was established in 1991 and is now Australia’s largest fishery by weight, with a total allowable catch in 2005 of 51 100 t. The rapid development of this fishery has been underpinned by strong investment in research by industry and the Commonwealth and State Governments, and has resulted in the establishment of a sophisticated and precautionary management system, which is strongly supported by stakeholders, including commercial and recreational fishers, and conservation groups. The research program that has supported the development of the fishery has also provided significant scientific outcomes, including insights into the structure and function of the Flinders current ecosystem, and evidence to refute the existing paradigm that Australia’s seas are universally unproductive. This presentation provides an overview of the large multidisciplinary research program that has been established to underpin the development of an ecosystem-based approach to the management of the South Australian sardine fishery, and identifies the factors that have been critical to ensure both the success of the research program and the concurrent achievement of strong ecological and economic outcomes. These factors include strong investment by the Commonwealth Government (FRDC) prior to the development of an economically successful industry; State government commitment to ensure a good balance between funding for research and management (transparent cost recovery system); strong commitment by industry to funding the ongoing research program (vision, leadership and strategic investment); seamless integration of monitoring and research programs (to establish a coordinated and focused critical mass of capability); extensive involvement of students to maximise scientific outcomes (piggybacking); strong collaboration with research agencies/scientists with complimentary skills and/or infrastructure; and cultural emphasis on relationship development, quality assurance, peer review and scientific publication of results.

Monitoring the multi-sector eastern Torres Strait coral reef finfish fishery

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The eastern Torres Strait (ETS) coral reef finfish fishery (CRFFF) is a multi-sector fishery, which unlike most other reef fisheries in Australia, has a significant commercial indigenous sector. Other sectors of the fishery include a commercial non-indigenous sector and a subsistence indigenous sector. Recently, concerns have been expressed by all sectors in the fishery about the long-term sustainability of the fishery. These concerns have been exacerbated by the lack of detailed catch and effort information, particularly for the indigenous sectors. Historically, there has been no long-term monitoring of indigenous fishing in the ETS CRFFF, while compulsory logbooks, established in 1988, have provided the only mechanism available to monitor non-indigenous commercial fishing. The significant indigenous component of this fishery presents unique challenges for developing appropriate monitoring programs, as indigenous fishing practices and motivations vary substantially from non-indigenous fishing. We discuss a range of fishery-dependent and fishery-independent techniques that we have used to collect data from each sector of the fishery as part of two projects designed to evaluate the current status of the fishery. We highlight the strengths and weaknesses of each technique in relation to fulfilling data requirements for assessments and management objectives.

Monitoring and assessing low-value temperate reef fisheries: the role of spatial stock structure in commercial and biological monitoring

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Analysis of commercial catch and effort data forms the basis of the stock assessment for Tasmanian scale-fish fisheries. These fishery data derive from logbook returns at a spatial resolution of 30 nm blocks. Simple performance measures relating to reference years and trends of catch, effort and catch rates are used in the assessment, although their relationship to stock status or condition remains unclear. Because conventional data-intensive assessment techniques cannot be justified due to the high costs of data collection, simple analyses of fishery data, especially trends in standardised catch rates, are likely to remain the main source for future stock assessments. However, because many reef fish species such as banded morwong (*Cheilodactylus spectabilis*) or purple wrasse (*Notolabrus fucicola*) have spatial structuring of the stocks, there is a mismatch between the relatively large-scale data reporting and the fine-scale population dynamics. To reduce the potential for masked serial depletion, the spatial resolution of data collection needs to be increased to match that of the stock processes as closely as possible. Characterising some aspects of the biology of the species, including age, growth, maturity and movement rates through biological monitoring can improve the ability to interpret fishery data by identifying potential risk and determining appropriate spatial scales for reporting and assessment. However, biological monitoring will often have only poor spatial representation, restricting its use as a reliable measure of overall stock processes. Modelling also indicated that many indicators typically derived from biological monitoring are not indicative for the stock status over its full range.

A new approach to assess the impact of the northern prawn fishery on sustainability of by-catch

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We present a new quantitative approach to assess the long-term sustainability of diverse tropical faunal assemblages impacted by fishing in data-limited fisheries. This method is demonstrated by assessing the impact of the northern prawn fishery (NPF) on the sustainability of 56 elasmobranchs by-catch species. We used data from more than 70 fishery-independent surveys from 1979 to 2003 and NPF logbook data for the tiger prawn fishery from 1970 to 2003. Four indicators were used to evaluate the fishery's impact on by-catch sustainability: 1) fraction of bioregions being trawled, 2) proportion of a fish species' geographic distribution overlapping with trawled areas, 3) assumed fishing mortality, and 4) change in the distribution of impacted fish species over time. This approach uses presence and absence data from scientific surveys to estimate the probability of detecting a species in a particular grid and the probability that species was present. Fishing mortality is estimated from geographic overlap with trawled areas, catch probability, and the probability of escaping the fishing gear. We establish two reference points to guide fishery

management of by-catch species: maximum sustainable fishing mortality rate and minimum fishing mortality rate that renders the population unsustainable. The annual impacted area where fishing effort is greater than five boat-days increased from about 2% in the 1970s to a peak of 9% in the early 1980s and gradually declined to about 3% in the last several years. The proportion of a species' population distributed within the trawled areas ranges from 1% to 57% for the 56 elasmobranchs. Taking probabilities of capture and escape into account, our preliminary results indicate that fishing impacts may contribute to the maximum sustainable mortality being exceeded for only a few species. We demonstrate that this method is effective for determining the sustainability of species in diverse assemblages and may be easily transferable to other fisheries due to its simplicity and requirement of only presence and absence data.

APPENDIX 1: LIST OF PARTICIPANTS

Name	Organization
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Jeffrey Leis	Australian Museum
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Jason Lieschke	Arthur Rylah Institute
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Neil Loneragan	Murdoch University
Ted Loveday	Seafood Services Australia
Robert Luxon	Environmental Office Of The Supervising Scientist
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Jarod Lyon	ARI
Murray MacDonald	Fisheries Victoria
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David McKey	DPIFM
Mark Meekan	Australian Institute Marine Science
Samantha Meyer	National Oceans Office
Nathan Miles	University of Wollongong
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Oliva Mkumbo	Lake Victoria Fisheries Organisation
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Jason Morton	University Of Newcastle
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Thomas Mullaney	University of NSW
Andrew Munro	University Of Adelaide
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Stephen Newman	Department Of Fisheries
Gina Newton	Australian Marine Sciences Association
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David Schmarr	SARDI Aquatic Sciences
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David Semmens	Melbourne University
Ben Sharp	New Zealand Ministry of Fisheries
Stephen Sly	DPIFM
Kimberley Smith	Department of Fisheries WA
Annette Souter	DPIFM
Jonathan Staunton-Smith	Queensland DPI & F
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Rick Stuart-Smith	University of Tasmania
Neil Sumner	Department of Fisheries WA
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Stephen Swearer	University of Melbourne
Matt Taylor	School of Biological Earth & Environmental Science
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Philip Thomas	Flinders University
Richard Tilzey	Bureau of Rural Sciences
Renae Tobin	CRC Reef Research Centre
Zeb Tonkin	Arthur Rylah Institute
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Marten Wolter	University of Rostock
Kelly Wright	Australian Museum/University of NSW
Qifeng Ye	SARDI Aquatic Sciences
Ursula Zaar	DIPE/ DNREA
Shijie Zhou	CSIRO
Philippe Ziegler	Tasmanian Aquaculture & Fisheries Institute

APPENDIX 2: LIST OF RECOMMENDED MONITORING GUIDES

Coastal areas

Dartnall A.J. and Jone M. 1986. A Manual of Survey Methods for Living Resources in Coastal Areas. Australian Institute of Marine Science.

Coral reefs

Crosby M.P. and Reese E.S. 1996. A Manual for Monitoring Coral Reefs with Indicator Species: Butterfly fishes as Indicators of Change on Indo Pacific Reefs. Office of Ocean and Coastal Resource Management, National Oceanic and Atmospheric Administration, Washington, D.C.

Hill J. and Wilkinson C. 2004. Methods for Ecological Monitoring of Coral Reefs, Version 1. Australian Institute of Marine Science and Reef Cheek, Los Angeles.

Rogers C., Garrison G., Grober R., Hillis Z-M. and Franke, M.A. 1994. Coral Reef Monitoring Manual for the Caribbean and Western Atlantic. National Parks Service, Virgin Islands National Park.

Estuaries, streams and rivers

United States Environmental Protection Agency. 1997. Volunteer Stream Monitoring: A Methods Manual. November. Office of Water, Washington, DC.

Wheeler K. 2003. Marine Community Monitoring Manual. 2nd edition. Marine Conservation Branch, Department of Conservation and Land Management. Fremantle, Western Australia.

Fish stocks

Samoilys M. 1997. Manual for Assessing Fish Stock on Pacific Coral Reefs. Training Series QE97009. Department of Primary Industries, Queensland.

Freshwater areas and wetlands

U.S. Environmental Protection Agency. 1991. Volunteer Lake Monitoring: A Methods Manual. Office of Water, Washington, DC.

Tucker P. 2004. Your Wetland: Monitoring Manual – Data Collection, River Murray Catchment Water Management Board, Berri and Australian Landscape Trust, Renmark.