Air Pollution and Health in Ulaanbaatar

Final Project Report
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This final report benefits from comments made by a number of participants at a workshop presenting preliminary results held at the Health Sciences University of Mongolia on February 7, 2014.

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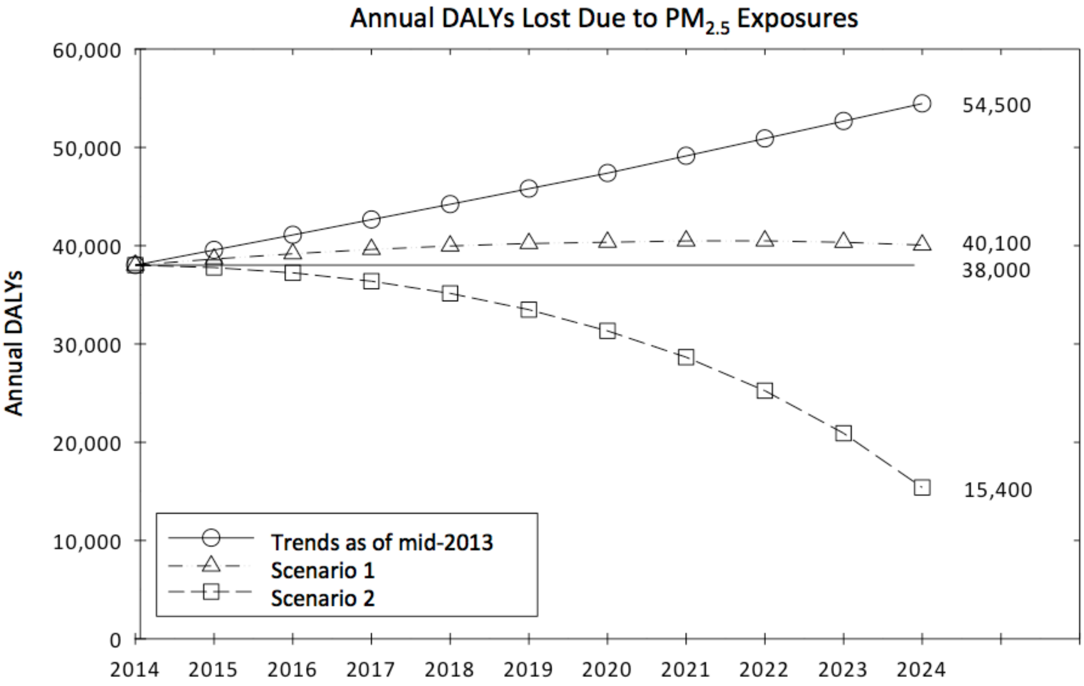
Executive Summary and Recommendation

Annual $PM_{2.5}$ (fine particle) air pollution averages nearly 70 micrograms per cubic meter ($\mu g/m^3$) for the Ulaanbaatar (UB) population today and causes a substantial amount of ill health, mainly as pneumonia in children (~130 annual premature deaths) and cardiovascular disease in adults (~1440 annual premature deaths). $PM_{2.5}$ exposures will grow slowly under current emissions control trends, which include some planned reductions in emissions in most sectors. The total health impact will thus rise somewhat over the next decade – see Trends-2013 in the figure below, which uses the preferred impact metric for health assessments – lost disability-adjusted life years (DALYs).



Executive Summary Figure is found as Figure 3 in the full summary

Acceleration of emissions reductions in household coal stoves and modest improvements in other sectors, however, have the potential to considerably lower outdoor pollution and reduce total exposures to about 70% of those today (Scenario 1).

At ~45 $\mu g/m^3$, however, these annual levels would still be well above World Health Organization (WHO) Air Quality Guidelines (10 $\mu g/m^3$) or national standards in the United States (12 $\mu g/m^3$). Reducing total exposures closer to these international benchmark levels will require moving away from coal and wood as household fuels and even more control on other sources (Scenario 2). In addition, current anti-smoking campaigns should be enhanced so that non-smokers are not exposed to environmental tobacco smoke in homes. Though advanced coal stoves will contribute to lower levels of $PM_{2.5}$, UB should consider plans to move fully to household gaseous/electric fuels and district heating.

The first package of moderate control measures (Scenario 1) considered in this assessment will result in a slow decline in impacts (Figure) and a cumulative health savings over trends in 2013, but leave annual per capita health impacts only about 25% lower than today after ten years. A more aggressive set of control measures (Scenario 2), however, will result in more health protection over the period and reduce annual impacts by approximately 60% from current levels in 2024 (Figure). In terms of impact per capita, this would represent nearly a 70% reduction over the period taking population growth into account.

The mandate for this assessment did not include an evaluation of the costs of reducing pollution under the two scenarios, which would have to be considered in any policy strategy. The team points out, however, that all of the measures have been implemented in other countries and are technically feasible in UB. Given the major health impacts that exist today, and the results of this assessment that document the need to take aggressive actions to reduce them substantially, we urge UB authorities to seriously consider the aggressive actions outlined in Scenario 2 of this assessment.

Extended Summary [1]

Ulaanbaatar (UB) has some of the worst winter outdoor air pollution in the world, a problem of growing concern among the public, media, and policy makers. To a considerable extent, this pollution is due to coal heating in the residential sector, although other sources such as power plants, vehicles, and industry play roles. Most observers agree with the need to eventually reduce emissions across the board, but the benefits of doing so quickly rather than more slowly have remained unclear, a choice with substantial differences in costs and strategic approaches.

Although there are other important impacts of air pollution including visibility, property values, cleanliness, and climate, perhaps the most important is that on health. To provide better evidence of the benefits of different strategies for reducing UB's air pollution, we have undertaken an assessment focused on the aim:

"What health benefits could be expected from cleaner household stoves and fuels and associated emissions reductions in other sectors by 2024?"

Description of scenarios

Our assessment uses a comparative framework across three alternative scenarios to 2024:

- **Trends as of mid-2013 (T-13):** No major changes in emissions trends from those currently underway, which include universal use of reduced emissions coal stoves in households and other improvements.
- **Moderately accelerated improvements (Scenario 1**): Improvements in all sectors beyond those in T-13, including full deployment of even cleaner coal stoves.
- **Maximum rate of improvement (Scenario 2):** Health benefits based on feasible but ambitious rates of change in all sectors including elimination of solid fuels in households.

Appendix A describes the control measures considered in each scenario.

Methods

Details on data sources, assumptions, uncertainties, and other methodological issues are provided in the appendices. As explained below, this assessment benefits from the application of several kinds of new information previously unavailable for assessments in UB or elsewhere.

We take advantage of the results of the Comparative Risk Assessments of the Global Burden of Disease Project (CRA/GBD), the details of which have just been published (Burnett et al. 2014; Lim et al. 2012; Smith et al. 2014). Among other innovations is the development of exposure-response curves that allow scientists to determine the health effects of any level of air pollution exposure separately for all major types of disease associated with air pollution. This, in turn, allows us to accurately estimate the change of health impact by disease category from one level of exposure to another. The UB assessment is one

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[1] Details of the assumptions, data sources, evidence base, and methods used in this assessment are found in Appendices A-H.

of the first in the world to take advantage of this new information in a policy-relevant assessment of options.

The CRA/GBD assessment involved extensive review of the world scientific literature on the health effects of air pollution. For five diseases, it found compelling evidence to derive quantitative relationships between disease and pollution exposure:

Children under 5 years

Pneumonia as acute lower respiratory infections (ALRI) – Ranked #2

Adults

Ischemic heart disease (IHD) – Ranked #1
Cerebrovascular disease (stroke) – Ranked #3
Chronic obstructive pulmonary disease (COPD) – Ranked #17
Lung cancer – Ranked #18

Three of these diseases are very important in the country. Shown is their ranking among all diseases in terms of lost life years in 2010 as determined by the Global Burden of Disease Project where IHD, ALRI, and stroke are ranked 1, 2, and 3, respectively in Mongolia. Together they account for nearly one-third of the lost life years in 2010 (Institute for Health Metrics & Evaluation (IHME) 2014).[2] Air pollution is not the only cause of these diseases, but this report shows it is an important one.

Although many other diseases are associated with air pollution, including tuberculosis[3] and low birth weight, the CRA/GBD assessment did not find sufficient evidence to include them as effects of air pollution. Future assessment, of course, will likely add new diseases as the evidence base improves. Thus, in this study for UB, we restrict our analysis to these five diseases, which are also the most important health outcomes for smoking and other types of combustion particle pollution.

A major innovation in the UB assessment is a focus not only on outdoor or indoor pollution, but on total exposure of the population, which is driven by outdoor emissions, indoor emissions, and how the two interact, i.e., how much outdoor air pollution penetrates into living environments. As health impacts are driven not by pollution in any one place, but by people's exposure to pollution in all the places they spend time during the day (total exposure), this more accurately reflects the health benefits of changes in emissions in different locations. This also led us to include an additional source of indoor pollution not originally planned, secondhand tobacco smoke. A focus on total exposure is an innovation in policy-oriented assessments. More explanation and details of the calculations are shown in Appendix D.

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[2] See full ranking in the first table of Appendix E.

[3] The impact of air pollution on tuberculosis (TB) is not as well established compared to these other diseases, and there is also no published relationship between exposure and TB. Thus, although the appendices discuss TB and provide preliminary estimates, TB is not included in the primary results of our report.

Local sources of information used in this assessment

We were able to use several sources of information that were not available to previous assessments of UB air pollution, which make the results more relevant to local conditions. :

- Local statistic were used to estimate demographic and health trends, the latter by disease category, to enable more accurate estimates of impacts. See Appendix G.
- Recent measurements of household indoor air pollution, stove emissions factors and other household parameters for UB as part of assessment of the Millennium Challenge Account (MCA) impact evaluation of the Energy and Environment Projects (SI, 2013).
- Stove usage and indoor pollution levels measured in gers for this project.
- New outdoor air pollution modeling conducted for this project that, for the first time, estimates future changes under different emissions scenarios.

Note on metrics

Although there are a number of health-damaging air pollutants, the primary indicator of health effects for combustion-related pollution is considered to be $PM_{2.5}$, particles less than 2.5 µm in size that can penetrate deep into the lung. More information on the health effects of this pollutant exists than for any other, although it is recognized that observed effects for some diseases may be partly due to it serving as an indicator of combustion pollution in general and that other pollutants play roles. Only in the case of $PM_{2.5}$, however, are there exposure-response relationships available for the major disease outcomes and thus we focus entirely on this pollutant in our analysis.

We report the primary health results of the three scenarios across disease categories and age groups in two ways: Premature Deaths and lost Disability Adjusted Life Years (DALYs). DALYs are the international unit used in health studies to take into account both the age distribution of premature morality and the severity of non-fatal diseases. DALYs from air pollution, or any other risk factor, are thus a combination of two factors:

YLL – years of life lost by premature mortality from the risk factor compared to the world's best life expectancy of ~86 years

YLD – years lived with disability, which multiplies years lived with an illness or injury by a severity factor specific to each type

These are added to obtain the DALY total for the risk factor:

DALYs = YLLs + YLDs

DALYs thus allow for combining the impacts of different types of diseases (pneumonia and lung, cancer, for example) among different ages (children and adults, for example) into a common metric. DALYs offer the best metric for cost-effectiveness comparisons by contrasting investments in health protection across sectors in terms of the health benefit per dollar spent. See Appendix E for more information.

Primary Results

Here, we discuss how population-weighted exposures to $PM_{2.5}$ might change with different scenarios of pollution control. We start with the current situation (Baseline) and then examine each of the three scenarios. Then, we present the implications for health for each scenario.

Estimated exposures by scenario

This assessment estimates total exposures based on projections of outdoor and indoor air pollution levels, combined with time-activity profiles for different populations that estimated how much time they spend in each location. Indoor levels are affected by indoor sources, such as smoking and leakage from stoves, as well as partial penetration of outdoor pollution indoors, which depends on house type.[4] We estimate outdoor air pollution using modeling methods described in Appendix B and indoor pollution levels using methods described in Appendix C. In Appendix D we describe the methods used to combine the results with time-activity information to estimate total exposures. In all cases, both indoor and outdoor sources (via penetration) contribute to indoor $PM_{2.5}$ concentrations.

Figure 1 shows the estimated outdoor pollution levels over the assessment period for the three scenarios used in the analysis of exposures and health effects. It does not take into account any new sources of pollution that might be introduced over the period. As noted, although Scenario 1 would greatly improve outdoor levels, it would not bring the city close to international norms in the same way as Scenario 2.

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[4] Occupational exposures were not considered in this assessment.

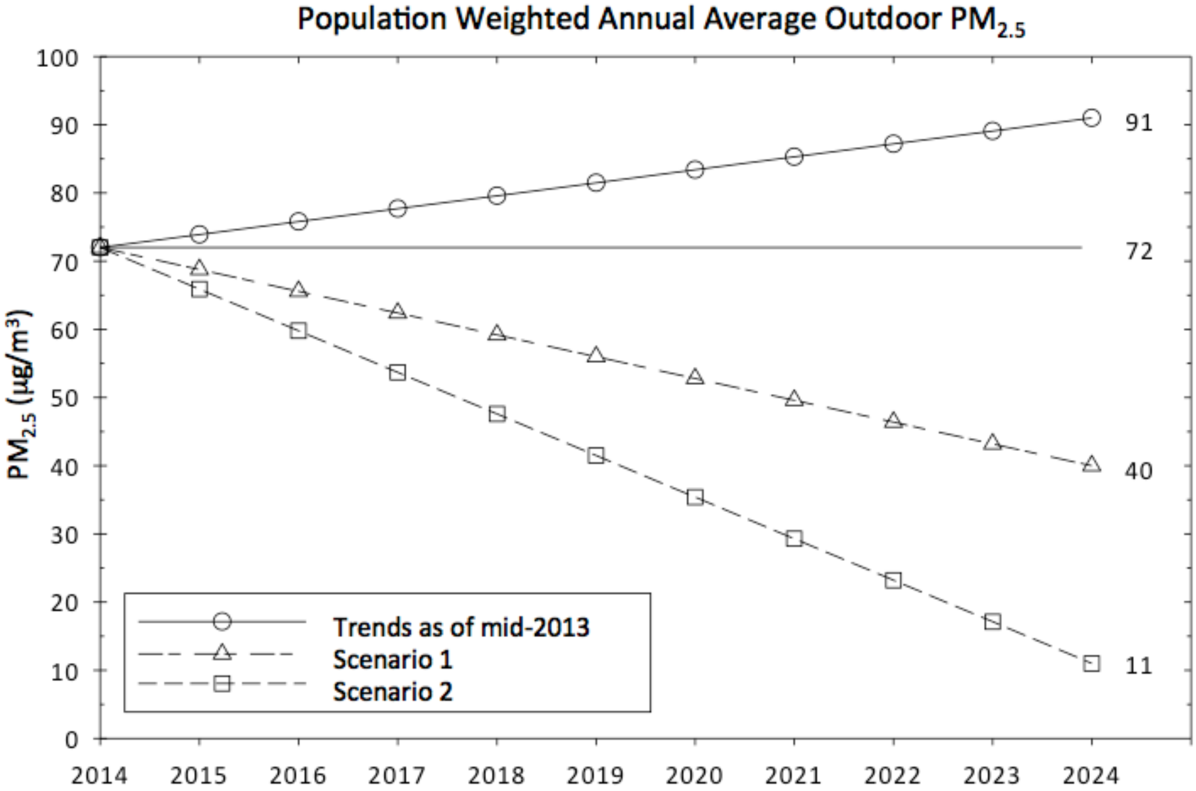

Figure 1. Estimated annual average outdoor $PM_{2.5}$ levels in UB. Shown is the average level outside people's homes, i.e., weighted by population distribution. This graph shows a straight-line change between the Baseline (2014) and the three 2024 scenario values, which were the years for which outdoor modeling was completed.[5] Current trends in control (T-13) are estimated to slow the rise, but not result in a decline in pollution levels over the period considering population growth and other changes.

Table 1 shows the population-weighted annual average exposures for Ulaanbaatar at the end of the assessment period under the range of modeled scenarios. It also shows averages specific to populations residing in each home type, which were weighted by population residing in each type to comprise the city-wide population-weighted annual average. The table also shows the percentages contributed by exposure in different microenvironments and smoking. (Numbers illustrating differences across residential populations and scenarios are found in Appendix D and may not add to 100% because of rounding.)

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[5] Modeled values for 2014 include a multiplicative factor to account for non-modeled sources and general model calibration which was applied to all of the modeling. Modeled 2024 population weighted averages in this figure were adjusted upward by 10 µg/m³ to broadly account for sources not included in the modeling such as kilns and other industrial sources, fugitive road dust, and other re-suspended dust (e.g., windblown). This 10 µg/m³ adjustment for the 2024 scenarios was not included in the exposure modeling.

Population Weighted Exposure Estimates -- Annual Average				
	Annual Average ($\mu g/m^3$)	Percentage of Annual Average from:		
		Indoor Exposures	Outdoor Exposures	Environmental Smoking Exposures
BASELINE - All Population	68.0	72	7.3	21
Ger Population	78.6	76	6.2	18
House Population	78.0	75	6.7	18
Apartment Population	57.0	67	8.5	25
T-13 - All Population	74.7	72	9.2	19
Ger Population	85.8	76	7.8	16
House Population	83.0	74	8.5	17
Apartment Population	69.8	70	10	20
Scenario 1 - All Population	46.9	63	6.8	30
Ger Population	70.3	76	4.4	20
House Population	58.1	68	6.2	26
Apartment Population	38.4	55	7.9	37
Scenario 2 - All Population	21.5	30	4.4	66
Ger Population	23.5	36	4.0	60
House Population	21.0	28	4.5	67
Apartment Population	21.2	28	4.6	67

Table 1. Population weighted exposure estimates.

Baseline: Currently, annual average modeled exposure for the population of Ulaanbaatar (after full installation of MCA improved stoves) is 68.0 $\mu g/m^3$. Indoor environments contribute approximately 72% of this exposure while outdoor exposures contribute 7.3%. The presence of smoking in homes contributes 21%. (This is not the contribution to active smokers, but the contribution of tobacco smoke in the environment that affects children and adults who do not smoke). Overall indoor environments contribute the largest fraction to the population exposures that lead to health impacts. The exposures, as expected, are higher in houses and gers compared to apartment buildings due to the presence of district heating in apartment buildings (i.e., heating does not contribute directly to indoor $PM_{2.5}$ in apartment buildings).

Trends as of mid-2013: Under the T-13 scenario, during the next 10 years exposures will increase modestly and retain a similar proportion from indoor, outdoor, and smoking. Higher ambient air concentrations are largely the result of population increases in the city. Thus, the health impacts of air pollution will likely increase.

Scenario 1: Modest additional emissions reductions from T-13 levels under this scenario would result in a significant 31% decrease from Baseline exposures. These results are achievable by continuing to distribute more advanced stoves with lower emissions in gers and houses, a 70% reduction in Heat Only Boiler (“HOB”) emissions (50% decrease in the number of HOBs as well as the implementation of considerable emissions controls) and a 3% increase in the population living in apartments with district heating, combined with retrofitting all currently existing power plants with emissions controls, and a 75% reduction in transportation emissions and meeting any additional electricity demand with renewables and/or imports. Although indoor air contributions to exposure still remain relatively high, there are substantial improvements in ambient air, which results in reduced exposures both outdoors and indoors (via penetration). In addition, as the other sources decrease, household smoking becomes a more dominant source of exposure. UB is to be praised for recently establishing a ban on indoor smoking in public places, which is assumed in this study to be completely effective. More public education may reduce household smoking in the population of Ulaanbaatar; however, this assessment assumes it will remain at current levels during the period of analysis.

Scenario 2: More vigorous measures to reduce air pollution result in a 68% reduction in population exposures from Baseline. Scenario 2 is based on a shift of all population in the ger districts to gaseous fuels or district heating, decommissioning of all HOBs, a reduction in transportation emissions to roughly 13% of 2014 baseline emissions, decommissioning of power plant #2 (CPP-2), and meeting any additional electricity demand with renewables and/or imports.

Estimated health effects by scenario

Details of the evidence base of air pollution health effects and the assumptions and calculation methods used to make health burden estimates are found in Appendix E. Here, we first discuss the estimated health impacts today from air pollution exposures in UB and how they are expected to change over the next decade with current plans for air pollution control as we understand them in 2013 (T-13). Then we summarize the expected reductions in health effects from additional moderate and aggressive control strategies (Scenarios 1 & 2).[6]

Baseline: Air pollution exposures are responsible for a considerable amount of ill health in UB today as shown in the first column of Table 2 (Baseline) in terms of premature morality and DALYs. The distribution by disease displayed in Figure 2 shows premature mortality is dominated by cardiovascular disease (hearth disease and stroke), while lost DALYs are primarily impacted by cardiovascular disease and ALRI (pneumonia) in children.

Another way to consider the importance of a risk factor is the “population attributable fraction” (PAF), which is the proportion of a disease that is attributed to it. In Table 3, we show the PAFs for each of the five diseases and how we project them to change over the period of the analysis. At baseline, the PAFs range from 42% for stroke to 17% for COPD. This implies that eliminating air pollution as a risk factor would lower each disease by approximately these amounts.

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[6] For background health conditions, we use local datasets provided by the Ministry of Health and adjust them to match international datasets provided in the International Institute of Health Metrics website (Institute for Health Metrics & Evaluation (IHME) 2013). These international datasets were created using methods developed for the Global Burden of Disease and applied in many countries to estimate underlying distribution of causes of death when local death records are incomplete or inconsistent. Appendix G shows details and a summary of differences from local statistics in UB.

Excess Deaths & DALYs, T-13, Scenario 1, & Scenario 2 (rounded to 2 significant digits)

	Accrued, 2014-2024 (per 1000 Capita)	Incurred in First Year of Scenario, 2014 (per 1000 Capita)	Incurred in Final Year of Scenario, 2024 (per 1000 Capita)
Deaths			
T-13	16,200 (10.1)	1,250 (0.92)	1,710 (0.92)
Scenario 1	14,400 (9.0)		1,340 (0.72)
Scenario 2	11,600 (7.4)		580 (0.31)
DALYs			
T-13	510,000 (310)	38,000 (28)	54,000 (29)
Scenario 1	440,000 (270)		40,000 (22)
Scenario 2	340,000 (220)		15,400 (8.3)

Table 2. Excess deaths and DALYs for the scenarios.

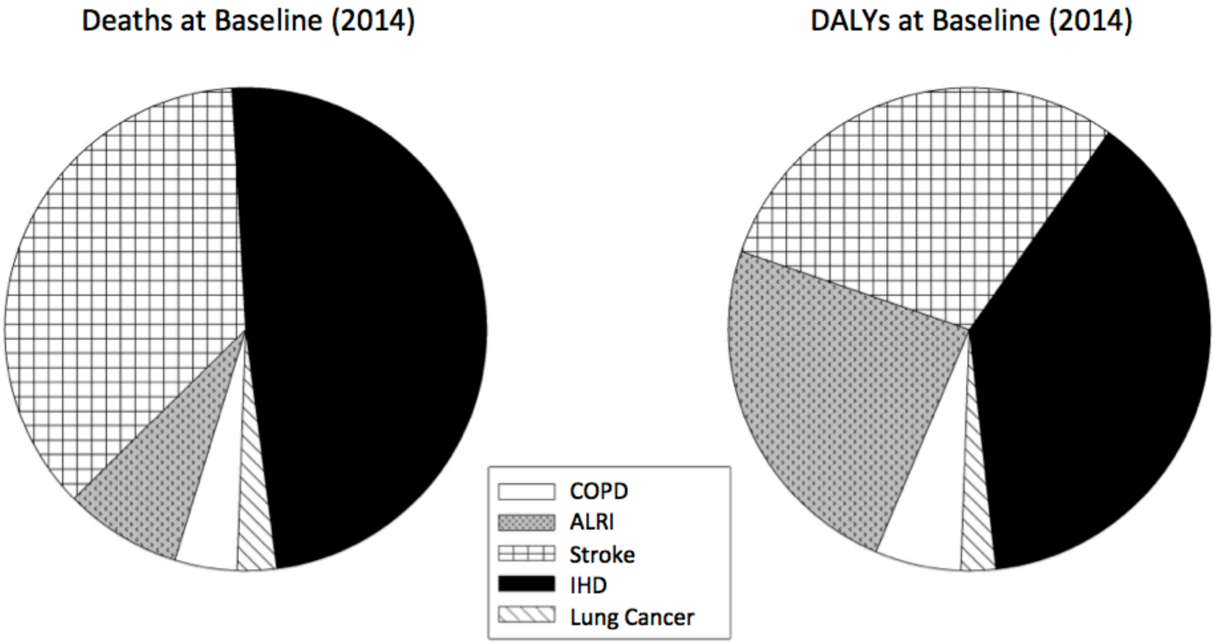

Figure 2: Distribution by disease of premature deaths (left) and lost DALYs (right) due to air pollution in UB at present (Baseline). Note the higher importance for ALRI in the DALY distribution because it affects young children.

Estimated population attributable fractions (PAFs), reported as percentages, for the five air-pollution-related diseases at baseline and at the end of the study period for each scenario.

	Baseline - 2014	T-13 - 2024	Scenario 1 - 2024	Scenario 2 - 2024
Lung Cancer	19.5	21.2	13.6	4.30
IHD	18.8	19.7	15.3	6.80
Stroke	41.7	42.6	37.2	18.0
ALRI (0-4 yrs old)	32.0	34.7	21.1	4.60
COPD	16.6	17.8	12.0	4.20

Table 3. Estimated population attributable fractions.

Trends as of mid-2013: The T-13 trend shows a slow rise in health effects over the next 10 years measured in DALYs (Figure 3). This is due partly to slowly rising pollution exposure (Table 1), but mainly to the increasing population in the city – about 35% increase in 10 years (See Appendix G). The PAFs in Table 3 drift slowly upwards with air pollution accounting for slightly more of each disease in 2024 than in 2014.

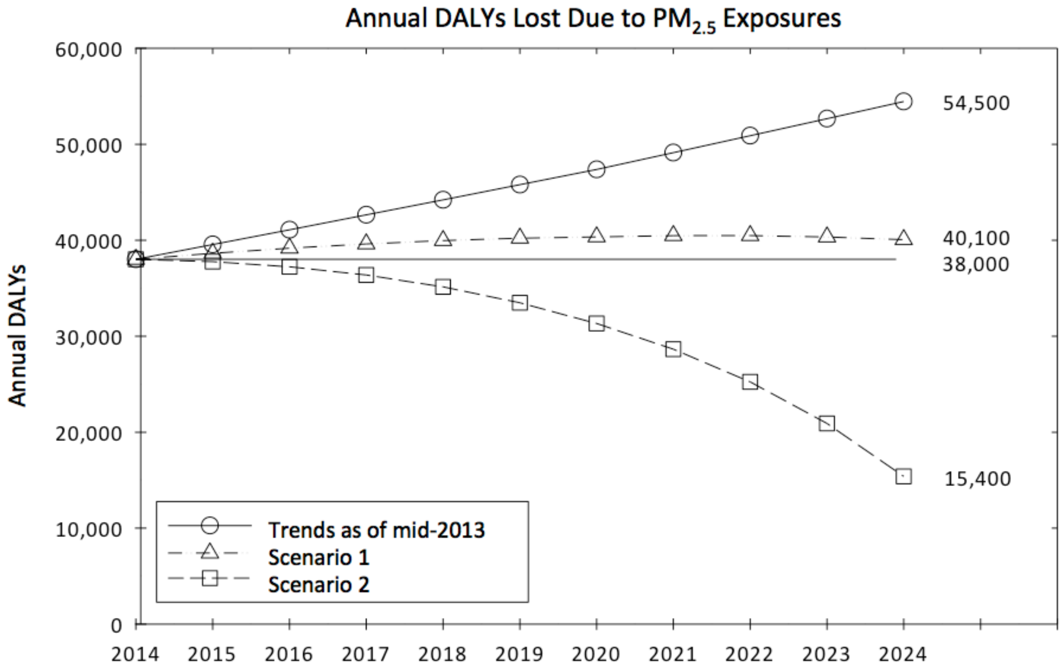

Figure 3: Estimated health impacts of air pollution over the assessment period for the three scenarios: DALYS/year. Note: the impact declines only in Scenario 2.

Scenario 1: Figure 3 illustrates that moderate programs to reduce pollution would have only modestly reduced the annual health burden after 10 years compared to the T-13. Figure 4 shows that a total of nearly 2000 premature deaths would be avoided as a result. Figure 5 shows that most of the benefit would accrue to children under 5 years of age in the form of fewer DALYs due to pneumonia, but important benefits would accrue to adults as well. It is worth noting, however, that although Scenario 1 results in important reductions in ill health compared the T-13, it does not reduce the annual impact in the city from the current level. In other words, it essentially just keeps up with population growth.

The PAFs in Table 3 lower appreciably; meaning that by 2024 air pollution will have become a less important risk factor for the population for all diseases. By 2024, the PAFs range from 12% for COPD to 37% for stroke.

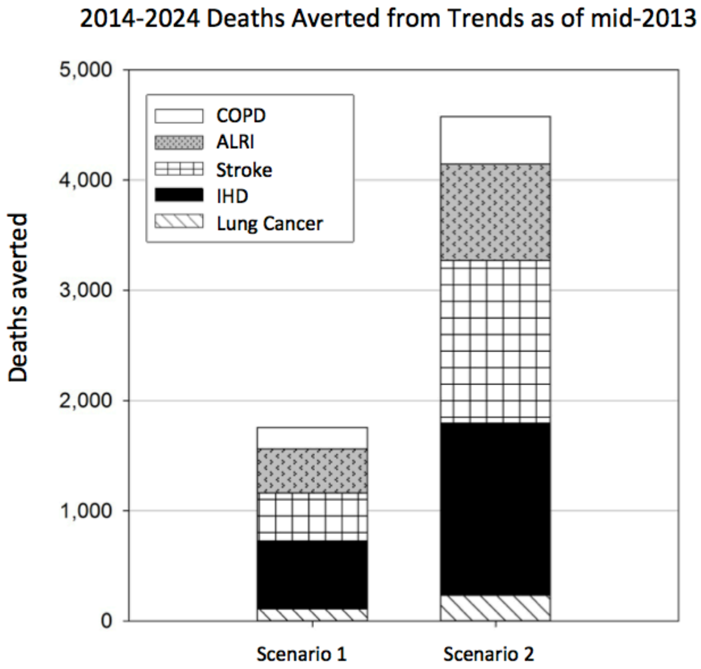

Figure 4: Estimate cumulative premature deaths averted by Scenarios 1 and 2 compared to T-13 over the assessment period. Scenario 2 would save more than 2.6 times more lives than Scenario 1.

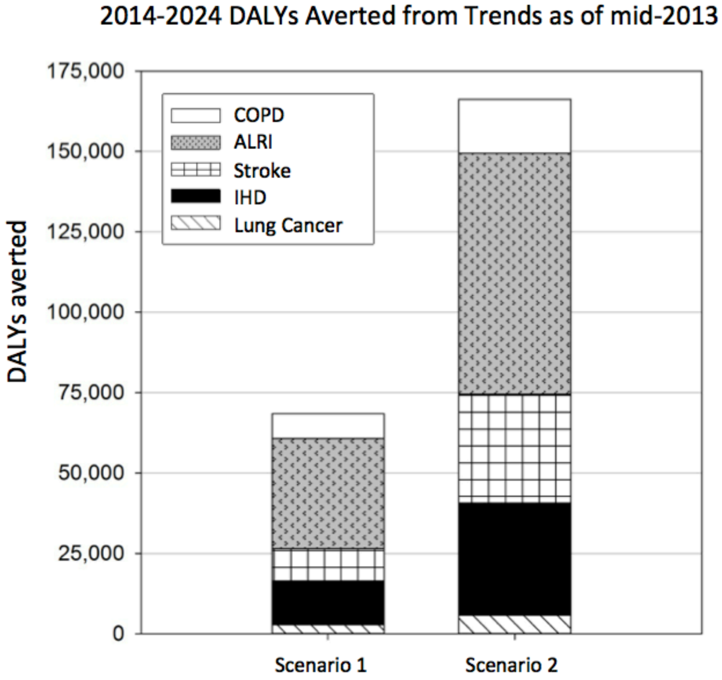

Figure 5: Estimate cumulative DALYs averted by Scenarios 1 and 2 compared to T-13 over the assessment period. Note the greater importance of child ALRI in Figure 5 compared to Figure 4.

Scenario 2: More aggressive pollution control, including the complete elimination of solid fuel use in households, has a much greater health benefit than moderate control. Figure 3 shows that, in spite of population growth, the annual impact of air pollution exposure would be reduced by 60% from current levels. As shown in Figure 5, the greatest benefits include reductions in child pneumonia and adult cardiovascular disease.

Another way to express the results of this assessment is as per capita rather than absolute levels (see Table 2). Figure 6 shows the trends in health impacts (DALYs) per capita as well as the estimated city population size over the study period, which is projected to grow to about 1.9 million by 2025 compared to 1.4 million in 2015. It shows T-13 basically retains the same impact per capita, Scenario 1 reduces impacts per capita at roughly the same rate as population growth, and Scenario 2 produces a major reduction in per capita effects.

The PAFs in 2024 lower substantially in Scenario (Table 3). Not only is there less air pollution exposure and less absolute impact from air pollution, it becomes less important overall as a health hazard in UB. It would account for less than 7% of ALRI, COPD, lung cancer, and IHD, although it is still estimated to account for about 18% of stroke. These PAF calculations assume there is no major change in other important risk factors for these diseases, for example salt intake, smoking, and physical activity.

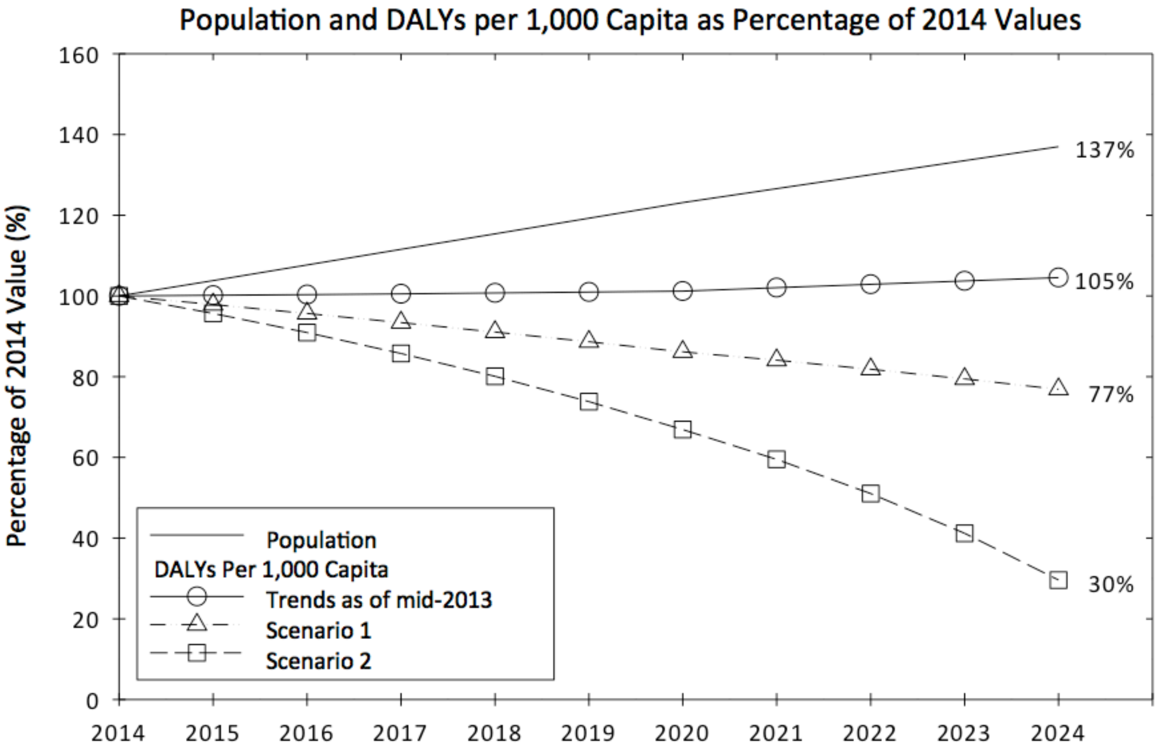

Figure 6: Relative projected urban population and estimated health impacts (DALYs) per capita by scenario over the assessment period with 2014 values = 100%. Scenario 1 reduces impacts per capita slowly over time, while Scenario 2 would reduce them to about one-third of current levels.

Conclusion

This assessment incorporated information from many sources into estimates of trends in population, household size, household type, health status, indoor air pollution, and outdoor air pollution for Ulaanbaatar over the study period – 2014-2024. These trends are then incorporated into the health impact estimates using state-of-the-art, internationally recognized exposure-response relationships by disease category. Information from each of these data sources is subject to different kinds of uncertainties and limitations both as to accuracy and to representativeness. As a result, we are not able to assign formal uncertainty bounds to our final estimates, but do list the major limitations of our results in Appendix F. We do believe that, although not too much emphasis should be placed on trends from year to year or exact numbers of premature deaths and DALYs, the overall differences represent reasonable estimates of the differences between the three scenarios over the study period for health.

The people of Ulaanbaatar currently experience mean exposures to $PM_{2.5}$ air pollution that far exceed international health-based guidelines and standards. Unless major new initiatives are taken, these exposures will increase slowly over the ten years with little progress in health impacts in total or on a per capita basis. A package of moderate pollution emission reductions (Scenario 1), however, would reduce the total impacts over the decade by about 25% in per capita terms, but still leave mean exposures well above international norms. A package of more aggressive controls including the elimination of household solid fuel use (Scenario 2), however, would bring annual health effects down by about 60% in absolute and 70% in per capita terms and exposures close to international norms at the end of the assessment period.

The mandate for this assessment did not include an evaluation of the costs of reducing pollution under the two scenarios, which would have to be considered in any policy strategy. The team points out, however, that all of the measures considered in the two control scenarios have been implemented in other countries and are technically feasible in UB. Given the major health impacts that exist today, and the results of this assessment that document the need to take aggressive actions to reduce them substantially, we urge UB authorities to seriously consider the aggressive actions outlined in this assessment. By doing so, UB could follow the lead of other polluted cities such as London in the 1950s, which although still suffering the infamous London smog episodes at the start became one of the cleanest cities in the world after a series of policy actions starting with banning raw coal use in households.

Appendix A. Summary of Scenario Assumptions and Data Sources

An estimate of the health benefits associated with varying degrees of air quality mitigation depends upon both current and future trends in home fuel use, and cannot be isolated from trends in other major sources of air pollution like power plants and transportation. Based on interviews with senior officials, who all expressed the need for and feasibility of fairly rapid transformation of the energy/pollution situation in the country, our study includes a 10-year scenario period, starting from a baseline of Jan 2014. Baseline heating practices, power plant function, and vehicle activity are estimated from government reports and recently published journal articles, while T-13 (trends as of mid-2013) trends are projected from reasonable assumptions informed by government projections and the expertise of project members. Two alternative scenarios are also formulated to demonstrate how moderate and strong improvements over T-13 to the three aforementioned sectors are likely to affect air quality.

Baseline and scenario activities were then translated into outdoor concentrations of respirable particulate matter ($PM_{2.5}$) as well as indoor $PM_{2.5}$ concentrations by stove and house types. Using time-activity estimates, total exposures are then estimated for adults and children separately. Methods for concentration, exposure, and health burden assessments are discussed in their respective appendices.

Household heating scenarios [7]

Houses and gers were originally heated by burning raw coal in simple metal heating stoves. Although a number of projects over the last 10 years distributed stoves designed to burn coal more efficiently and reduce emissions, the coverage by these projects was very low, and the vast majority of dwellings still use traditional heating stoves. The U.S. Millennium Challenge Corporation (MCC) through its compact with the Government of Mongolia introduced a program in 2011 to encourage replacement of traditional stoves with subsidized top-lit updraft coal burning lower-emission stoves. As part of this compact approximately 97,230 stoves were distributed consisting of the Ulzii, the Khas, and the Dul stove types ("MCA stoves"). An independent impact evaluation indicated the program resulted in an overall reduction in emissions in grams $PM_{2.5}$ per day of approximately 67% when weighted by the overall distribution of the different stove types in the 97,230 stoves subsidized in Ulaanbaatar, with a corresponding 16% reduction in indoor air pollution levels (see model in Appendix C below).

Subsequently government officials and local air quality experts indicate the Government of Mongolia had a near term goal of replacing all traditional stoves with these lower emission models, with subsidies for an additional 45,000 planned for 2013, and full coverage shortly afterward. [8] Other heating types for dwellings include: general household heating stoves fired with semi-coking fuels ("semi-coking stoves"); low pressure boilers ("LPB") burning coal used to heat household radiator systems; heat only boilers

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[7] Estimated emissions from stoves, HOB, and vehicles were scaled upward by a factor of 2.85 to reconcile with current ambient measurements. This scaling was retained for all scenarios.

[8] While traditional coal stoves were in existence during the winter of 2013, we assume the Clean Air Project meets its goal of replacing 100% of such stoves with MCA/TLUD stoves by 2014.

(“HOB”), which create outdoor emissions but negligible indoor emissions; and centrally-distributed steam that is produced during combined electricity and heat generation (Social Impact (SI) 2013a).

We also include in our projections a separate category for “other clean heating types.” These produce a negligible amount of $PM_{2.5}$ and are expected to become more accessible as the government’s energy and development plans progress. We also include a hypothetical “future tech stove” that provides an additional 60% emissions reduction and approximately 16% indoor concentration reduction over the MCA TLUD stoves.[9] “Stoves” are relevant only to house and ger household types and are applied one stove per household. LPB are relevant to houses and are applied one stove per household, and HOB are relevant only to apartments. Distributed steam and other clean heating types can be employed in all home types, though central steam is considered unlikely in gers.

Baseline MCA stove prevalence is estimated from government implementation plans and 2013 bank figures of MCA stove sales (Social Impact (SI) 2013a). Government implementation plans as conveyed to us by government officials suggest a near-term goal of installing MCA stoves in 100% of ger and house homes that do not otherwise have LPB, clean heat, or semi-coking coal-fired stoves. For this reason it is assumed that MCA stoves are employed in 100% of gers located outside of Bayangol,[10] and 100% of non-LPB houses outside of Bayangol.[11] We assume 20,000 LPB households (19% of houses) with one LPB per home, based on information provided to us by government officials at the Clean Air Fund. This is consistent with recent data, which show 14,186 LPB households in 2010 (National Statistics Office of Mongolia (NSOM) 2012). The National Bureau of Statistics identified 86% of apartment dwellings relying on steam heat from cogeneration in 2012, a percentage we conservatively assume remains constant through 2014.[12] Remaining apartment households (14%) are assigned heating from HOB, which are distributed to the 189 HOBs that government officials indicated were in use as of winter 2013-2014.[13]

Trends as of mid-2013 (T-13) will see all ger and non-LPB households’ transition to MCA stoves — this is distinct from baseline in that homes in Bayangol in 2014 are assumed to rely on semi-coking coal stoves. The number of LPB houses will remain constant at 20,000. An increase in the number of HOB is unlikely as it is an outdated technology, and we assume the same 189 HOB service the same number of households in 2024 (T-13) as they serviced in 2014. At 14% of 2014 apartment households, this number works out to be 25,488 apartment households, or 6% of 2024 apartment households. The remainder of apartment households will rely on clean heating or distributed steam. In Scenario 1, 100% of ger transition to the Future Tech stove, 50% houses transition to the Future Tech stove, and 50% of houses

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[9] The intent of the Future Tech stove is to simulate the introduction of a stove type that provides a reduction in $PM_{2.5}$ emissions and exposures from the MCA stoves that is of the same magnitude as the reduction in emissions and exposures that the MCA stoves provided over the traditional coal stove.

[10] A raw-coal ban that was in effect in Bayangol as of the publishing of this report means that only semi-coking coal is allowed to be burned. From 2012 census data, we identified 7% of ger households and 6% of house households in these districts.

[11] For simplicity and a lack of data suggesting otherwise, the 20,000 LPB houses are assumed to be located outside of Bayangol.

[12] No change in the percentage of residents using distributed steam is conservative, because an increase, which would move households away from the dirtier alternative of HOB, is actually most likely. This will keep our baseline emissions estimates slightly lower than would an increase in this percentage.

[13] Personal communication with Dr. Battogtokh Zagd, Senior Lecturer, School of Power Engineering, MUST - Fall 2013. Ulaanbaatar, Mongolia.

transition to “clean” heat. Half of all HOB operating at Baseline are decommissioned, leaving 50% fewer apartment households relying on HOB (12,744). These homes change to clean heating and distributed steam, bringing the number of apartments with clean heat up to 402,451. In Scenario 2, 100% of ger and house households transition to clean heat. All HOB are abolished, and 100% of apartment households rely on distributed steam or clean heat. Summary tables of the number of each type of stove used by household type for baseline, T-13, and each scenario can be found below (Tables A-1, A-2).

Household heating types at baseline (2014) and in the projected scenarios						
	Percent of Gers	*Number of Gers*	*Percent of Houses*	*Number of Houses*	*Percent of Apartment Households*	*Number of Apartment Households*
2014 - Baseline						
MCA Stove	93	80,122	75	79972	0	0
Stove w/Semi-Coking Fuel	7	6123	6	6381	0	0
Low Pressure Boiler	0	0	19	20000	0	0
HOB	0	0	0	0	14	25488
Clean: distributed steam, gas, elec	0	0	0	0	86	154230
2024 – T-13						
MCA Stove	100	94834	83	96943	0	0
Low Pressure Boiler	0	0	17	20000	0	0
HOB	0	0	0	0	6	25488
Clean: distributed steam, gas, elec	0	0	0	0	94	389707
2024- Scenario 1						
Future Tech Stove	100	94834	41	48472	0	0
Low Pressure Boiler	0	0	17	20000	0	0
HOB	0	0	0	0	3	12744
Clean: distributed steam, gas, elec	0	0	41	48472	97	402451
2024- Scenario 2						
Clean: distributed steam, gas, elec	100	94834	100	116943	100	415195

Table A-1. Household heating types at baseline and for the scenarios.

MCA stove emissions profiles weighted by the prevalence of Ulzii, Khas, and Dul stove sales are detailed in the SI impact evaluation final results report (Social Impact (SI) 2013a). Data on the emissions profiles of low-pressure boilers and semi-coking coal stoves in UB were unavailable, and so they were conservatively assigned the emissions profiles of MCA stoves. “Future Tech” stove emissions are assigned by applying to the MCA emissions profile the same reduction seen in the transition from traditional coal stoves to MCA stoves during the SI impact evaluation. HOB emissions are informed by a

2013 Japan International Cooperation Agency (JICA) PM_{10} emissions inventory prepared for HOBs. All stack emissions are assumed to be in the $PM_{2.5}$ size range. These inventories are used with no modifications for the year 2014 and year 2024 T-13 $PM_{2.5}$ emission inventories. Scenario 1 (2024) assumes an overall 70% reduction in HOB emissions, which is consistent with adoption of high efficiency cyclone as a control strategy (JICA 2013). 2024 Scenario 2 assumes all HOBs are decommissioned. Overall emissions are summarized in Table A-3.

Estimated Household Numbers in 2014 and 2024		
	2014	**2024**
Total Households	372317	626972
Ger Households	*86246*	*94834*
House Households	*106353*	*116943*
Apartment Households	*179718*	*415195*

Table A-2. Estimated household numbers in 2014 and 2024

HOB Emissions for the Scenarios	
Scenario	HOB Emissions (tons $PM_{2.5}$/year)
2014 Baseline	1,300
2024 T-13	1,300
2024 Scenario 1	390
2024 Scenario 2	0

Table A-3. HOB emissions for baseline and the scenarios.

Power plant emissions scenarios

JICA (2013) prepared a year 2010 PM_{10} emission inventory for each of the four existing combined heat and power (CHP) plants (CHP-2, CHP-3 (two units), and CHP-4). All stack emissions are assumed to be in the $PM_{2.5}$ size range and these inventories are used with no modifications for the year 2014 $PM_{2.5}$ emission inventories. The year 2024 Trends as of mid-2013 (T-13) inventory retained these emissions and included a new power plant (CHP-5). In summer 2013 a Memorandum of Understanding was signed between the Mongolian government and GDF Suez SA group, which will build the plant. The current design is for a 450 MW plant. There has been discussion of eventually expanding the capacity to 820 MW and this larger size was for the 2024 scenarios (Ulaanbaatar Clean Air Project (UB CAP) 2012). Few details are currently available for this facility, which is planned to start operation by 2017 with a location about 15km east of the UB Central Business District. Emission rates depend on the plant efficiency and coal quality. A generic contemporary design is assumed with an electricity generation rate of 1870 kWh/ton coal and a coal heat content of 19.53 MMBtu/ton. The plant is assumed to meet the U.S. New Source Performance Standard (NSPS) for electric utility power plants which is 0.015 lb PM/MMBtu. Assuming the plant operates continuously throughout the year, the estimated PM emissions are 511 tons/year and all emissions are assumed to be in the $PM_{2.5}$ size range. Scenario 1 assumed all the above but high efficiency control devices, such as electrostatic precipitators (ESP), installed on units CHP-2, -3, and -4. The PM capture rate is assumed to be 98%, which is a conservative estimate because properly designed and operated ESPs typically have capture efficiencies exceeding 99%. This would be a significant upgrade to the existing infrastructure at the UB CHPs, which includes wet scrubbers or ESPs, depending on the facility. The 98% capture rate is applied to an assumed uncontrolled emission factor of 16.6 kg PM/ton coal. Scenario 2 assumes all the above but with CHP-2 decommissioned. Any additional electricity demand in 2024 that is not met by the UB power plants is assumed to be supplied by renewables and/or imports with no impacts on UB air quality. Overall emissions are summarized in Table A-4.

Power Plant Emissions for the Four Scenarios

Scenario	Power Plant Emissions (tons $PM_{2.5}$/year)
2014 Baseline	11,500
2024 T-13	12,000
2024 Scenario 1	1,900
2024 Scenario 2	1,830

Table A-4. Power plant emissions scenarios

Motor vehicle emissions scenarios

Bottom-up emission inventories for motor vehicles require detailed information about the vehicle fleet, annual miles traveled by various vehicle classes in the fleet, and emissions per vehicle mile traveled. This can depend on several factors including vehicle emission controls, vehicle age and maintenance, and roadway congestion. The development of a comprehensive bottom-up inventory is beyond the scope of this project. Instead, the approach is based on simple scaling of an existing inventory. JICA (2013) prepared a year 2010 PM_{10} emission inventory for motor vehicles traveling on major and minor roads. All vehicle exhaust emissions – this inventory does not include brake wear, tire wear, or resuspended dust – are assumed to be in the $PM_{2.5}$ size range. Year 2014 $PM_{2.5}$ emissions are estimated as 1.7 times the 2010 inventory to account for growth in the number of registered vehicles – a nearly 100% increase between 2010 and 2013 – that is partially offset by the vehicle travel day ban program. This simple scaling approach does not account for changes in the fleet composition over time, but insufficient details for the JICA 2010 inventory are available to make more sophisticated projections. The year 2024 Trends as of mid-2013 (T-13) Scenario inventory is assumed to be 1.3 times the 2014 inventory. This estimate is based on an emissions growth rate of 2.5%/year over the 10 year period. While this growth rate might seem low given the recent trends in vehicle registration, a higher growth rate seems intractable given the existing transportation network infrastructure. Scenario 1 (2024) assumes all the above but emission reductions consistent with adoption of Euro V emission standards compared to Euro III standards. For diesel vehicles the Euro V PM emission standards are 80%-93% lower than the Euro III standards depending on vehicle class. There are no Euro standards for PM emissions from gasoline-fueled vehicles and thus 90% overall reduction would not be realized. However, gasoline vehicle Total Hydrocarbon (THC) standards are 50% lower for Euro V compared to Euro III. This may result in some PM reductions for the cold wintertime conditions, which favor semivolatile gaseous compounds entering the particle phase. Overall, a 75% reduction in emissions compared to year 2014 is used. Scenario 2 (2024) assumes all the above but with an additional 50% reduction compared to Scenario 1. Opportunities for such reductions include but are not limited to higher adoption rates for mass transit use, transportation network enhancements to improve traffic flow, and adoption of Euro VI standards, which include an additional 50% reduction in PM emission rates from heavy duty diesel vehicles compared to Euro V standards. Overall emissions are summarized in Table A-5.

Motor Vehicle Emissions for the Four Scenarios

Scenario	Motor Vehicle Emissions (tons $PM_{2.5}$/year)
2014 Baseline	384
2024 T-13	500
2024 Scenario 1	96
2024 Scenario 2	48

Table A-5. Motor vehicle emissions for the four scenarios.

Summary of Emissions Assumptions and Data Sources

2014 (Baseline)[14]

Stoves – 100% of stoves are either MCA, semi-coking, or low pressure boiler (all assume emissions profile of MCA stove, with emissions data from MCC/SI)

HOB – 189 HOB service 14% of apartments, emissions from JICA 2010 PM10 inventory (assume all $PM_{2.5}$)

Vehicles – emissions 1.7 times JICA PM10 inventory (assume all $PM_{2.5}$), 1.7 multiplier to account for dramatic increase in vehicles and congestion since 2010

Power Plants – emissions from JICA 2010 PM10 inventory (assume all $PM_{2.5}$)

2024 T-13 – Continued Trends as of 2013

Stoves – no technology change (there is a small increase in number of stoves)

HOB – no change

Vehicles – 130% of Baseline emissions from emissions growth rate of 2.5%/year

Power Plants – no change in existing plants, bring CPP-5 online and any additional electricity demand met by renewables and/or imports, emissions at 104% of Baseline

2024 Scenario 1 – Moderately accelerated improvements

Stoves – 20,000 stoves remain as low pressure boilers (low pressure boilers assume emissions of MCA stove), the balance have Future Tech emissions (40% of MCA emissions, i.e. 60% reduction)

HOB – 70% reduction from Baseline emissions, installation of high efficiency cyclones, reduction in number of HOBs by 50%

Vehicles – 25% of Baseline emissions, adoption of Euro V

Power Plants – 17% of Baseline emissions, installation of 98% efficient control devices (e.g. electrostatic precipitators) on CPP-2,-3, and -4

2024 Scenario 2 – Maximum rate of improvement

Stoves – no emissions from stoves (all converted to gas, clean liquids, or electricity)

HOB – no emissions from HOB (all decommissioned)

Vehicles – 13% of Baseline emissions, adoption of Euro VI

Power Plants – 16% of Baseline emissions, Scenario 1 assumptions plus decommission CPP-2

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[14] Estimated emissions from stoves, HOB, and vehicles were scaled upward by a factor of 2.85 to reconcile with current ambient measurements. This scaling was retained for all scenarios. This analysis neglects kiosks stoves, small HOB, industrial emissions (e.g., kilns), and re-suspended road dust.

Appendix B. Outdoor Ambient Air Quality Modeling

Ambient air quality modeling methods

Air quality modeling was conducted to estimate outdoor $PM_{2.5}$ mass concentrations. The modeling methodology closely follows that used for an impact evaluation of the MCA Mongolia Energy and Environment Project Energy Efficient Stove Subsidy Program conducted by Social Impact (SI) for the Millennium Challenge Corporation, in this case expanded to include additional emission source categories and emissions scenarios (Social Impact (SI) 2013a). The modeling framework was described in the SI impact evaluation final results report and thus is only briefly summarized here. Ground-level outdoor $PM_{2.5}$ concentrations were estimated using the Industrial Source Complex Short-Term, version 3 (ISCST3) dispersion model (US EPA 1995). Modeling was conducted at hourly resolution for the period June 2012 – May 2013 using temperature, wind speed, and wind direction from air quality monitoring station #4 (UB04) in the National Agency for Meteorology Hydrology and Environmental Monitoring (NAMHEM) network, and mixing layer height and solar radiation estimates from the NOAA HYSPLIT (NOAA Hybrid Single Particle Lagrangian Integrated Trajectory) model (Draxler and Hess 1997, 1998; Draxler 1999). Average concentrations were generated for daytime (8 AM to 6 PM) and nighttime (6 PM to 8 AM) and for winter (October through March) and summer (April through September).

Emissions from residential heating stoves, heat only boilers (HOB), motor vehicles, and combined heat and power (CHP) plants were included in the model. Other sources not included in the model are heating stoves in kiosks, industrial emissions including kilns, resuspended road dust, and windblown dust. Existing power plants were modeled as point sources using available geographic location and stack properties data (JICA 2013). Residential heating stoves, HOB, and motor vehicle emissions were modeled as area sources. The greater UB region was discretized into 6,298 grid cells, each with dimension 1 km × 1 km. Emissions were allocated to these grid cells and the center of each grid cell was used as a receptor site for which modeled $PM_{2.5}$ concentrations were generated. Two districts – Baganuur and Bagakhangai – were excluded from the modeling and exposures to outdoor PM for these areas were handled outside the model.

The SI impact evaluation final results report provides a detailed description of the approach to estimate residential heating stove emissions and to allocate these emissions in space and time. Briefly, emission testing conducted in more than 200 dwellings included measurement of PM in the stove flue. These data were used to estimate emission factors (g $PM_{2.5}$/kg coal) for traditional stoves and different manufacturers and models of MCA program stoves with separate factors for stoves in gers and houses. Daily average coal consumption rates during three phases of the heating season (late fall, winter, and early spring) were estimated from a survey of 1096 households conducted for the impact evaluation. These rates were also stratified by stove type and dwelling type. The hour of day profile for stove emissions was estimated from the survey data with emissions assigned to the hour of fueling events.

Stove distribution lists provided by Khan Bank and Khasbank were used to determine the sales number of each MCA stove type in each khoroo. Khoroo-level data for MCA stoves were allocated to the 1 km × 1 km grids using Geographical information System (GIS) software. Ger area khoroo boundaries were clipped to include only the populated areas. Stoves were allocated to the grid using area-weighted sums. Total residential heating stoves in each khoroo were estimated using the 2012 household census and initially assuming one stove for ger area household (whether ger or regular hose) (National Statistics Office of Mongolia (NSOM) 2012). The number of residential heating stoves in each khoroo was increased by 20% to account for multiple dwellings in a household (e.g. a house and a ger) (Japan International Cooperation Agency (JICA) 2013). The year 2014 scenario treats all non-"Future Tech" stoves as MCA stoves.

Projected stoves sales in each grid cell (i.e. stoves to dwellings not purchasing a stove during the MCA program) were allocated at the same proportion of stove sales, by house and ger, as the actual MCA stove sales. For grid cells with five or fewer MCA stoves, the projected stove sales were allocated using the overall program sales fractions. Data on the emissions profiles of low-pressure boilers in UB were unavailable, and so they were conservatively assigned the emissions profiles of MCA stoves. "Future Tech" stove emissions are assigned by applying to the MCA emissions profile using the same reduction seen in the transition from traditional stoves to MCA stoves during the SI impact evaluation. Projected changes in the number of ger stoves between 2012 and the 2014 and 2024 scenario years were distributed across grid cells in proportion to the number of 2012 ger households. The same approach was taken for houses Low-pressure boilers and Future Tech stoves were allocated in proportion to the number of houses in each grid cell. Residential heating stove emissions were assumed to be zero during the summer period.

Dr. Sarath Guttikunda provided 0.01°× 0.01° emissions fields for the source categories included in his year 2010 inventory (Guttikunda et al. 2013). These fields were re-projected in GIS for this project by first contouring the 1°×1° data and then calculating area weighted means for the 1 km × 1 km grids. HOB and motor vehicle emissions used for this project were spatially allocated in proportion to the re-projected fields obtained from Dr. Guttikunda. HOB emissions were allocated by month using the weights obtained from Dr. Guttikunda and were assumed to be zero during the summer period. Motor vehicle emissions were held constant for each season and were allocated to hour of day using a typical urban profile with morning and afternoon rush hour peaks.

Khoroo-level year 2012 population by dwelling type (ger, houses, and apartment) was also allocated to the 1 km × 1 km grids using area weighted sums. Projected changes in the ger household population between 2012 and the 2014 and 2024 scenario years were distributed across grid cells in proportion to the 2012 ger household population. We took the same approach for the population residing in houses. Projected changes in the population residing in apartments were allocated in proportion to the total population in each grid.

In addition to the four scenarios we examined, air quality modeling was conducted to evaluate model performance through model-to-monitor comparisons. Limited outdoor $PM_{2.5}$ data are available for this comparison. For example, during the 2012-2013 heating season $PM_{2.5}$ mass concentration data were collected with high data completeness by NAHMEM at one location - air quality monitoring station #2 (UB02). This site is next to a major roadway and likely has high impacts from traffic that cannot be resolved by the model. Thus, outdoor $PM_{2.5}$ data collected by Ecography and Ecoworld under contract from MCA were used for the comparison. The sampling locations, methodology, and key results are presented in the SI impact evaluation final results report (Social Impact (SI) 2013a). Briefly, measurements were conducted from January 22 to April 22, 2013, at four locations within ger areas or at the boundary of ger areas. $PM_{2.5}$ samples were collected onto filters for 24-hour periods (noon-to-noon) every two-to-three days and subsequently analyzed for $PM_{2.5}$ mass concentration and chemical composition. Data from January 22 to March 2, 2013, were used for the model-to-monitor comparison with 19 samples per site. The 2014 projected inventory was used except that residential stove emissions were calculated using the actual MCA stoves sales with the remaining residential stoves assumed to be of traditional design.

The measured average $PM_{2.5}$ concentrations at each site, shown by the single-crossed bars in Figure B-1, demonstrate high spatial variability with up to a 50% difference between sites. The sample time period was also modeled with the average concentrations, including all days between January 22 and March 2, shown by the red bars in Figure B-1. Modeled concentrations are much lower than the measured values and are less variable between sites. There are several possible reasons for these differences including, but not limited to, the emissions for these sources being underestimated and the model not being able to account for the trapping and accumulation of emissions from one hour to the next. The model was reconciled to the measurement data by increasing the residential stove, HOB, and motor vehicle emissions by a factor of 2.85, which is the value of the four measured-to-modeled concentration ratios. The green bars in Figure B-1 show the modeled $PM_{2.5}$ concentrations after this scaling. Assuming the only error was in the emissions inventory, the nearly threefold increase of the projected JICA 2010 inventory is still lower than the inventory projected by Guttikunda et al. for 2010 for each of these source categories (Guttikunda et al. 2013; JICA 2013). While the scaling increases the emission inventory for these sources by about a factor of three, this places the effective emissions between those projected from the year 2010 inventories prepared by JICA and Guttikunda et al. Thus, the scaled emissions were deemed reasonable because they are bounded by the best available inventories. Power plant emissions were not scaled because the JICA and Guttikunda et al. inventories are relatively similar and emissions from tall stacks are less likely to be trapped and accumulate at ground level. This residential stove, HOB, and motor vehicle emissions scaling was applied to all of the modeled scenarios.

Ambient air quality modeling results

Figure B-2 shows the winter season (October-March) average outdoor $PM_{2.5}$ concentrations for the scenarios. Scenario 2 values for 2024 are excluded because the highest average concentration attributed to the modeled sources was ~3 $\mu g/m^3$. For the three scenarios shown in Figure B-2, the model predicts large variations in $PM_{2.5}$ mass concentrations across UB with highest concentrations in the ger areas where residential stoves and HOBs have the largest impact. Given these large spatial variations, outdoor concentration levels between scenarios are compared using population-weighted measures. Table B-1 presents the population-weighted mean outdoor $PM_{2.5}$

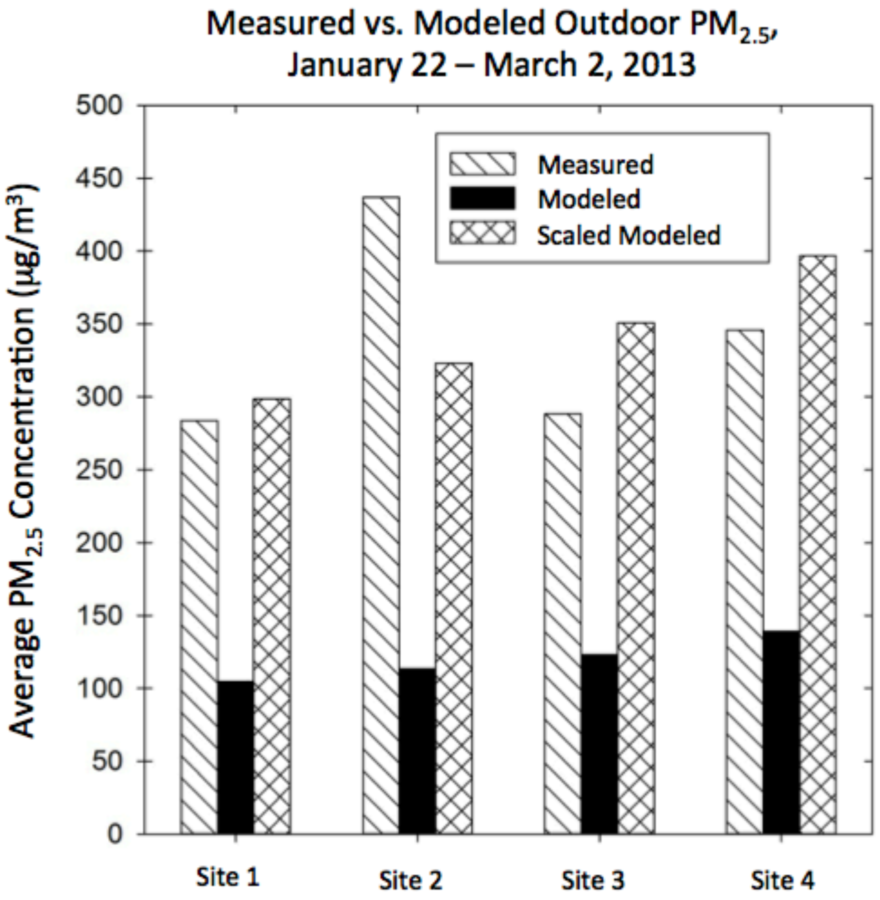

Figure B-1. Measured and Modeled $PM_{2.5}$ Concentrations at Four Sites for the Period January 22 – March 2, 2013

Scenario 1 reduces 2024 wintertime population mean concentration by 60% compared to the 2024 T-13 scenario, but the mean concentration value of 69 µg/m^3 is still quite high. Figure B-3 shows box plots for the population-weighted distribution of wintertime outdoor concentrations. For each of the three scenarios, 10% of the population resides in areas with $PM_{2.5}$ outdoor concentrations ~50% higher than the mean scenario-specific outdoor concentration reported in Table B-1.

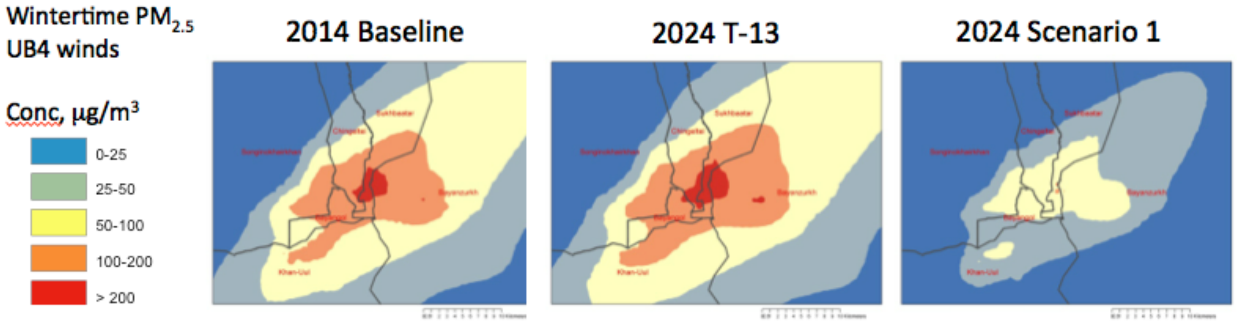

Figure B-2. Winter Average Outdoor $PM_{2.5}$ Concentrations for the Scenarios

Population Weighted Mean $PM_{2.5}$ Outdoor Concentrations ($\mu g/m^3$) by Season

Scenario [15]	Summer	Winter			
	Total Pop.	Total Pop.	Ger Pop.	House Pop.	Apt. Pop.
2014	6	138	137	144	133
2024 T-13	19	163	162	169	161
2024 Scenario 1	12	69	70	72	68
2024 Scenario 2	10	12	12	12	12

Table B-1. Population weighted mean $PM_{2.5}$ outdoor concentrations by season.

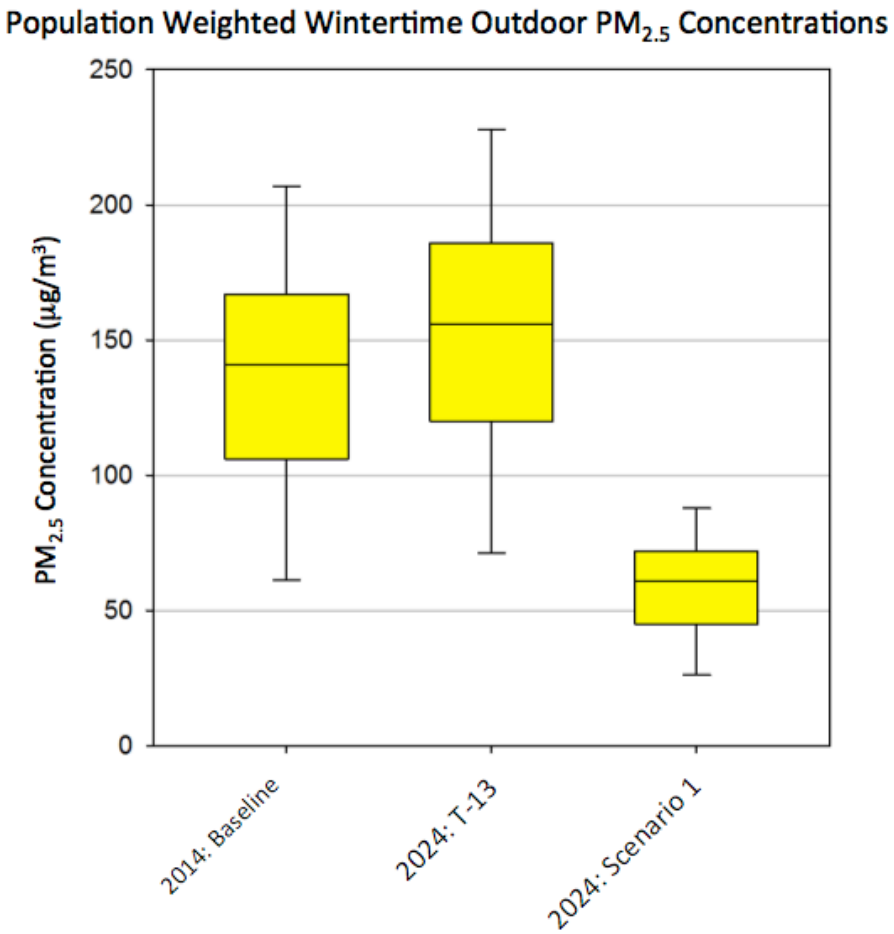

Figure B-3. Population Weighted Wintertime Outdoor $PM_{2.5}$ Concentrations

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[15] Modeled values for T-13 2024 summer total pop., winter total pop., winter ger pop., winter house pop., and winter apt. pop. were 9, 153, 152, 159, and 151 µg/m3, respectively. Modeled values for Scenario 1 2024 summer total pop., winter total pop., winter ger pop., winter house pop., and winter apt. pop. were <2, 59, 60, 62, and 58 µg/m3, respectively. Modeled values for Scenario 2 2024 summer total pop., winter total pop., winter ger pop., winter house pop., and winter apt. pop. were <1, <2, <2, <2, and <2 µg/m3, respectively. This is quite reasonable for the sources we included, and these outdoor values are included in the exposure assessment. However, in this table only, we upwardly adjust by 10 $\mu g/m^3$ in order to broadly account for sources not included in the modeling such as kilns and other industrial sources, fugitive road dust, and other resuspended (e.g. windblown) dust.

Appendix C: Indoor $PM_{2.5}$ Air Pollution Estimates

Indoor concentrations of airborne particles smaller than 2.5 um in diameter ($PM_{2.5}$) are estimated by home type, household heating source, smoking status, and season. This analysis includes all home types most relevant to the Ulaanbaatar context: gers, single family houses, and multi-family apartments. Home types are described in further detail in Appendix A.

Heat source, smoking, and indoor concentrations

Government officials and local air quality experts have indicated that the heating types most common in Ulaanbaatar include traditional coal-fired household stoves ("traditional stoves"), top-lit updraft stoves (Ulzii, Khas, and Dul) that have been shown to collectively reduce emissions compared to traditional stoves distributed as a component of the Millennium Challenge Energy and Environment Project ("MCA stoves") (Social Impact (SI) 2013b), general household heating stoves fired with semi-coking fuels ("semi-coking stoves"), Heat Only Boilers ("HOB"), Low Pressure Boilers ("LPB"), and centrally-distributed steam that is produced during combined electricity and heat generation ("distributed steam"). We also include in our projections a separate category for "other clean heating types" (e.g., gas and electricity) that we expect will become more accessible as the government's energy and development plans progress, as well as a hypothetical "future tech stove" that provides an additional reduction over the MCA TLUD stove and is described in more detail in Appendix B. "Stoves" are relevant only to house and ger household types, LPB are relevant to houses, and HOB are relevant only to apartments. Distributed steam and other clean heating types can be employed in all home types, though central steam is considered unlikely in gers.

Observation suggests that indoor $PM_{2.5}$ concentrations in UB are driven primarily by heating appliances and environmental tobacco smoke (ETS). The vast majority of ger and houses rely on the combustion of solid fuels like wood and coal in small chimney stoves to meet their heating needs. Such stoves are known to emit considerable amounts of $PM_{2.5}$ both indoors and out (Cowlin et al. 2005). Moreover, smoking rates in Mongolia are among the highest in the world, with recent national rates identified as high as 65% in males 15 years and older and 21% among corresponding women (Baigalmaa et al. 2006). The 2012-2013 survey of Ulaanbaatar households performed by the Social Impact project team identified 58% of houses and 62% of gers as dwellings with at least one smoker, though no data were available on apartment households (Social Impact (SI) 2013a). While these results are not comprehensive, they are likely a fair approximation of the general population. Our indoor concentration estimates thereby include a citywide average smoking prevalence of 60 percent of households, which may be conservatively high (WHO 2010). Recent nation-wide bans on indoor smoking in public areas suggest indoor ETS may only make considerable contributions to exposure in personal, private indoor environments. The indoor concentration estimates thereby assume ETS occurs only at home indoors. The conservative assumption is made that smoking rates remain constant during 2014-2024.

Indoor $PM_{2.5}$ concentrations measured in gers and houses as part of the impact evaluation of the MCA Mongolia Energy and Environment Project Energy Efficient Stove Subsidy Program conducted by Social Impact for the Millennium Challenge Corporation are used to estimate the contribution of various stove types and ETS to indoor concentrations. The sample of homes was drawn from a larger household survey of 1096 homes conducted 3 times over the winter heating season 2012/2013. The sample population for the household survey included residents of Ulaanbaatar's Bayangol, Bayanzurkh, Chingeltei, Khan-Uul, Songino Khairkhan, and Sukhbaatar Districts. With the exception of Bayangol, these were target areas for stove distribution in order to achieve the highest reductions in PM throughout the city at the earliest possible date, as they are the most heavily polluted areas in the city. MCA stove owners were randomly selected from complete stove distribution lists kept by Khan Bank and Khasbank. Traditional stove owners were randomly selected from the Ministry of Labor and Social Welfare's 2010-2011 Proxy Means Test (PMT) data, a census of all Ulaanbaatar ger area households that was designed to assess poverty.[16] Indoor air measurements were assessed over an approximate 14-hour period from early evening through the next morning in a subsample of 216 dwellings from the household survey. Homes were randomly selected from each stove/dwelling combination selected to evaluate statistical comparisons between MCA stoves and traditional stoves (see Table C-1 below).

Indoor air quality sample distribution – number monitored					
	Traditional	**Ulzii**	**Khas**	**Dul**	**Total**
Gers	34	36	0	25	95
Houses	32	32	36	21	121
Total	66	68	36	46	**216**

Table C-1. Indoor air quality sample distributions by the number that were monitored.

Indoor measurements were conducted in both homes and gers by placing a TSI DustTrak II Aerosol Monitor and TSI Q-Trak CO/CO2 monitor in the main living space of the home, usually close to the wall at approximately 1.5m high to reflect breathing zones. $PM_{2.5}$ concentrations were assessed using simultaneous gravimetric and semi continuous $PM_{2.5}$ measurements with a TSI DustTrak II Aerosol Monitor using 37mm PTFE (Teflon) 2.0µm Pore Size Filters (preweighed and loaded into cassettes). The gravimetric samples were then used to calibrate the DustTrak semi continuous data response. Semi continuous CO/CO_2 measurements were conducted using TSI Q-Trak 7565/7575 CO & CO_2 Monitor. All flow rates were set using Dry Cal flow meter primary standard. Pre and post weights of filters for particulate matter were weighed in an environmentally controlled microbalance room after equilibration for at least 48 hours. Nine field blanks were collected.

From these data, the following model is employed to estimate contributions of stove type and ETS to indoor concentrations (Table C-2):

$$Indoor\ PM2.5 = \beta_0 + \beta_1 * Traditional + \beta_2 * ETS$$

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[16] Joint order of the Chairman of the National Statistical Office and the Minister for Social Welfare, Labor. "To approve methodology, questionnaire form, and fill-in instructions." 5 April, 2010.

Where Indoor $PM_{2.5}$ is the overnight average indoor concentration of $PM_{2.5}$ ($\mu g/m^3$), β_0 is the average concentration in homes with an MCA stove, β_1 is the difference in average concentration between homes with traditional stoves and MCA, and β_2 is the difference in concentration between homes with smokers and homes with non-smokers. Presence of traditional stove or ETS is binary (0 = no; 1 = yes). Longitudinal indoor $PM_{2.5}$ measurements taken in a small subset of overlapping households as part of another study were used to test the long-term representativeness of the aforementioned overnight measurements.[17] The overnight averages included in this regression were similar to the corresponding 24-hour averages, as well as longer averages taken between January and April. Results of the longitudinal data are planned for publication.

Ultimately, reductions in indoor air concentrations derived from regression results of only ger households are applied equally over both gers and houses (Tables C-2 and C-3). Average nighttime indoor concentrations measured are generally in the same range in houses and gers for PM (140 - 180 and 130 - 170 $\mu g/m^3$, respectively), but the average indoor concentration is anomalously lowest in houses with traditional stoves. Actual values are highly variable, and exclusion of homes with smokers present results in very similar concentrations in houses and gers (130 and 140 $\mu g/m^3$, respectively). The idea that houses with traditional stoves would have lower indoor air concentrations than gers with traditional stoves and houses with MCA stoves seems anomalous for the following reasons:

1. Emissions of $PM_{2.5}$ per day from houses with traditional stoves are larger than gers, and significantly higher than the MCA stoves; and

2. Indoor CO concentrations in houses are substantially higher than in gers for all stove types, and show reductions between homes with traditional and MCA stoves. $PM_{2.5}$ would be expected to show a similar direction in the relationships, although not necessarily the same magnitudes.

Although we are unsure why the indoor measurements in houses with traditional stoves would show a lower indoor air concentration than shown in gers or other stove types, it is possible that the positioning of the monitors in adjacent but connected rooms, and whether the houses studied were single or multiple room, might have impacted the $PM_{2.5}$ results and requires further examination. Given that the emissions show similar reductions between houses and gers, and regression models accounting for the impact of smoking in gers indicated a 16% reduction in indoor air concentrations as a result of the installation of MCA stoves, for the purposes of the modeling, we determined that the reductions in indoor air concentrations would be applied equally over both gers and houses.

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[17] This study is still in progress by a collection of the authors of this report: A longitudinal assessment of coal stove use and $PM_{2.5}$ in the ger district of Ulaanbaatar, Mongolia.

Indoor Concentration Model - MCA vs. Traditional Stove - Ger			
	Coefficient (µg/m^3)	St. Error (µg/m^3)	p-value
MCA Stove	124.7	19.8	1.102 e-8
Traditional Stove	23.4	24.7	0.346
Smoker in Household	40.4	24.6	0.104
Total p-value of model: 0.113 , n = 94 ger households			

Table C-2. Coefficients derived from indoor $PM_{2.5}$ concentration data collected in ger homes in the winter of 2012-2013. Data represent overnight average concentrations (approximately 18:00 – 08:00).

Estimated Average Wintertime Indoor $PM_{2.5}$ Concentrations – MCA vs. Traditional Stove Dwelling		
	Average Wintertime Indoor Concentration, No Smoker in Home (µg/m^3)	Average Wintertime Indoor Concentration, with Smoker in Home (µg/m^3)
MCA Stove Dwelling	124.7	165.1
Traditional Stove Dwelling	148.1	188.5

Table C-3. Estimates of average wintertime indoor $PM_{2.5}$ concentrations by smoking status and stove type. Estimates are derived from the model described in Table C-2.

Low pressure boilers and semi-coking coal stoves are widely heralded by the public for their improvements in efficiency and functionality over traditional coal stoves, but little data exist to verify or quantify actual reductions in related contributions to indoor $PM_{2.5}$. It is unreasonable to suggest completely clean function, so contributions from these stoves are assumed somewhere between traditional stoves and "clean" heat sources. For the sake of simplicity and a lack of data, we assign the same indoor contribution factors to low pressure boilers and semi-coking coal stoves as are assigned to MCA stoves. "Future Tech" stoves were hypothesized as new stove technology coming to market in the near future, which will produce the same fractional reduction in indoor $PM_{2.5}$ concentrations as the transition from traditional stoves to MCA stoves. Contributions from Future Tech stoves are thereby assigned at a level ~ 16% lower than MCA stoves (105 µg/m^3).

Penetration of outdoor particles

Penetration of outdoor particulate matter into the indoor environment is becoming increasingly acknowledged as a considerable contribution to indoor concentrations (Allen et al. 2012). Studies indicate that the contribution of outdoor $PM_{2.5}$ to the indoor environment is quantified primarily through a factor called the infiltration efficiency (F_{inf}) of an enclosure (Dockery et al. 1981). Infiltration efficiency is the fraction of outdoor $PM_{2.5}$ concentration that penetrates indoors and remains suspended. F_{inf} can be an important source of indoor $PM_{2.5}$, and thereby must be accounted for when attempting to accurately assess PM exposures and the related health burden.

An annual average F_{inf} for $PM_{2.5}$ in apartment buildings and houses was taken from EXPOLIS, a study that included the calculation of infiltration efficiency in several major cities (Hänninen et al. 2004). We use the average F_{inf} specifically reported for Helsinki (F_{inf} = 59%), as it is the EXPOLIS city with a climate most similar to that of Ulaanbaatar. EXPOLIS reports 48-hour estimates of infiltration taken throughout the year in a multitude of homes. Because this study draws from a combination of houses and multistory apartments (58% multistory) at various seasons, its estimates are generally representative of annual average penetration efficiencies in apartment dwellings and houses in Ulaanbaatar (Jantunen et al. 1998). For this reason, our study applies this infiltration efficiency to both houses and apartments throughout the entire year. This number is consistent with F_{inf} found in similar home types in similar climates (Long et al. 2001). The F_{inf} overestimates the penetration of outdoor particles during the winter months, however, as it is more reflective of an annual $F_{inf.}$

Separate $PM_{2.5}$ infiltration efficiencies for ger dwellings are employed for winter and summer months. The literature shows the greatest equalizers of indoor and outdoor $PM_{2.5}$ levels are increased ventilation from open windows and doors and removal of indoor emission sources (Allen et al. 2012). We assume that ger doors and ceiling flaps are left open for much of the summer months, which results in high ventilation rates and a virtual elimination of filtration related to airflow through building casings. We also assume that no major indoor sources aside from cigarettes exist in ger in the summertime, as stoves are decommissioned. This is consistent with comparisons of wintertime and summertime infiltration in other regions (Long et al. 2001). We thereby employ a summer F_{inf} of 100%. Our winter F_{inf}, 70%, is taken from blower door tests performed at the Mongolian University of Science and Technology.[18] Blower door test results provide an n_{50} air change rate for gers with modest insulation and fly cover, which are converted to natural air change rates using the methods described in Sherman et al (1987). Air change rates are converted to crude estimates of $PM_{2.5}$ infiltration efficiency using a curve elucidated by Williams et al (2003) in which the number of air changes per hour measured during a blower door test (ACH50) is used to calculate an estimate of the natural air change rate, which is, in turn, used to estimate an infiltration factor. Results of each step are reported in Table C-4. This is not an ideal conversion process, but provides a reasonable estimate of wintertime penetration efficiency in gers in the absence of better data.

A PM infiltration component is applied to all apartments as well as apartments and gers with clean heat sources. We do not apply a distinct PM infiltration component in our estimates of indoor concentrations in houses and gers with MCA stoves, traditional stoves, semi-coking coal stoves, or low pressure boilers. Instead, we assume infiltrated PM is accounted for in the average concentrations identified in the model from which we take our indoor $PM_{2.5}$ concentrations for these home types.

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[18] This study is not yet publicly available: Munkhbayar, B. (not yet published). Blower door tests in ger structures. Retrieved December, 2013.

Inputs, Equations, and Outputs for the Estimation of Wintertime Ger Infiltration Factors		
Air changes/hr @ n50 ([19])	Natural air change estimate: ACH = (ACH @n50)/20 (Sherman et al, 1987)	Estimated conversion to Infiltration Factor: Finf = 0.5571 + 0.1726ln(natural air change rate) (Williams et al, 2003)
45 air changes / hr	2.25 air changes / hr	0.697

Table C-4. Inputs and outputs of the estimation of wintertime ger infiltration factor from blower door data and conversion equations (Sherman 1987; Williams et al. 2003). [20]

Final estimates of average indoor concentrations

From the aforementioned figures assumptions, 24-hour average indoor $PM_{2.5}$ concentrations are estimated for gers, houses, and apartments in each season of T-13 and Scenario periods. Infiltration factors are applied to average outdoor $PM_{2.5}$ concentrations specific to each season and semi-diurnal period. Outdoor model and averaging methods are detailed elsewhere in the report. Briefly, average outdoor $PM_{2.5}$ concentrations are taken from outdoor models and are the average of grid cell values weighted by home type populations (ger, house, apartment). Heating sources are assumed off in summer periods, and the contribution to indoor $PM_{2.5}$ from ETS indicated in the above model is applied to both summer and winter periods. Winter and summer time concentration estimates by home type, stove type, and smoking status are reported in Tables C-5 and C-6.

Although significant reductions in emissions were made by MCA stove, the reductions in indoor air concentrations were more modest. As a result wintertime indoor air concentrations in gers and houses are still high – a factor of 3.6 higher than WHO interim targets for indoor air (35 $\mu g/m^3$) – and considerably higher than the WHO Air Quality Guideline (10 $\mu g/m^3$). Under Scenario 1, where an additional program of more advanced stoves is undertaken that incorporates a 16% reduction in wintertime indoor $PM_{2.5}$ concentrations, without increased attention paid to emissions into the indoor environment, the indoor air concentrations still remain high. Clearly, however, if the stoves selected for the program are designed to minimize emissions into the indoor environment, more substantial gains in exposure and reduced air pollution disease can be made. Scenario 2 represents the adoption of gas and electric stoves and district heating in homes with very low emissions, combined with measures to reduce ambient air pollution leads to modeled indoor air concentrations that are substantially below WHO air quality guidelines. These indoor air pollution estimates only refer to emissions from stoves and smoking and penetration from outdoor pollution and do not include emissions from the food itself and other indoor sources (candles, incense, etc.), which also contribute to low levels of indoor air pollution. They do, however, represent the ideal scenario where emissions are largely removed from the home, resulting in concentrations that are unlikely to cause adverse health effects in the population, except for those homes that still have smokers present.

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[19] This study is not yet publicly available: Munkhbayar, B. (not yet published). Blower door tests in ger structures. Retrieved December, 2013.

[20] This study is not yet publicly available: Munkhbayar, B. (not yet published). Blower door tests in ger structures. Retrieved December, 2013.

Wintertime Indoor Concentration Estimates - T-13			
		Indoor Concentration – non-Smoking ($\mu g/m^3$)	Indoor Concentration – Smoking ($\mu g/m^3$)
MCA Stove		125	165
Low Pressure Boiler		125	165
"Clean" Heating			
	Ger	*112*	*153*
	House	*99*	*139*
	Apt	*95*	*135*
Wintertime Indoor Concentration Estimates – Scenario 1			
MCA Stove		125	165
Low Pressure Boiler		125	165
Future Tech Stove		105	145
"Clean" Heating			
	Ger	*48*	*89*
	House	*42*	*83*
	Apt	*40*	*80*
Wintertime Indoor Concentration Estimates – Scenario 2			
"Clean" Heating			
	Ger	*8*	*48*
	House	*7*	*47*
	Apt	*7*	*47*

Table C-5. Wintertime concentration estimates by home and heating type, and smoking status.

Summertime Indoor Concentration Estimates - T-13		
	Indoor Concentration – non-Smoking ($\mu g/m^3$)	Indoor Concentration – Smoking ($\mu g/m^3$)
Ger	18	58
House	10	51
Apt	12	52
Summertime Indoor Concentration Estimates – Scenario 1		
Ger	11	52
House	7	47
Apt	7	47
Summertime Indoor Concentration Estimates – Scenario 2		
Ger	11	51
House	6	47
Apt	6	47

Table C-6. Summertime concentration estimates by home and heating type, and smoking status

Appendix D. $PM_{2.5}$ Exposure Estimation

A basic multi-level model was defined to facilitate the estimation of citywide mean average annual $PM_{2.5}$ exposure concentrations for each of our scenario years. This model accounts for the major sources specific to indoor and outdoor microenvironments, time spent in each of these microenvironments, and particle infiltration (or penetration). It also accounts for variations by age in the amount of time spent in each microenvironment. This section outlines the processes, equations, and key assumptions that are employed in each scenario in the estimation of population weighted exposures to $PM_{2.5}$.

Sources

In depth information on the indoor and outdoor sources included in our exposure assessment can be found in Appendices B and C. Briefly, the major indoor sources identified include heating stoves and cigarettes (environmental tobacco smoke), while the major outdoor sources considered include household heating stoves, low pressure boilers, semi-coking coal stoves heat only boilers (HOB), vehicles, and combined heat and power (CHP) plants. Assumptions specific to those sources are discussed in the aforementioned sections. Contributions to indoor exposures from ETS are distinguished using the models discussed below and reported as fractional contributions to seasonal and annual average concentrations.

Modeling exposure

We employ a multi-level series of models to estimate a population-weighted annual average for each of the scenarios that we examine. These figures are derived from microenvironment exposures estimated at the seasonal (summer and winter) and semi-diurnal (nighttime and daytime) periods.[21]

At the highest level, this is accomplished by taking a weighted average of the annual average exposures of each of the 21 key sub-populations (Model A). The 21 key sub-populations include children (< 4 years old), caretakers (age >5, assumed 1 per child), and all others (ages 5 and up) in each of the seven home types considered: gers with MCA stoves or semi-coking coal stoves, "future tech" stoves, or clean heat; [22] houses with MCA stoves or semi-coking coal stoves or low pressure boilers, future tech stoves, or clean heat; and apartments with clean heat. Specific sub-population annual averages are arrived at by averaging their seasonal average exposures (Model B). Sub-population seasonal averages are the summation of time-activity weighted average indoor and outdoor exposures (Models C and D).

For each of the models, "A" denotes the age category for which the exposure is being tabulated (child or non-child), "H" denotes one of the home-stove type combinations (e.g. ger-MCA), "ETS" indicates that environmental tobacco smoke is the source of interest, "S" denotes the season (summer or winter), "D" denotes a daytime period, "N" denotes a nighttime period, "in" denotes an indoor micro-environment, "out" denotes an outdoor microenvironment, "PM" is the concentration of $PM_{2.5}$ in the denoted

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[21] Winter months are October through March. Summer months are April through September. Daytime is defined as 8:00-18:00. Nighttime is defined as 18:00- 8:00. More detailed is described in the subsection on Population weighting and Time Activity.

[22] In this context, "clean" denotes a heating method that does not result in indoor $PM_{2.5}$ emissions, and does not imply negligible lifecycle emissions. Examples include gas, electricity, or steam distributed from CHP or HOB.

microenvironment (μg/m^3), "%t" is the percentage of period "D,S" or "N,S" spent in the denoted microenvironment, and "IF" is the infiltration factor (seasonally specific for gers) of the home type in question (indicated by home-stove combination "H").

$$\begin{matrix} Pop - Weighted\ Annual \\ Avg. Exposure \end{matrix} = \sum \left(\%Pop_{A,H} * \begin{matrix} Annual\ Average \\ Exposure \end{matrix}_{A,H} \right)$$

Model A. The highest-level model used to tabulate population-weighted annual average exposure for the entire city of UB. "Annual Average Exposure $_{A,\,H}$" is a function of the other models listed below.

$$\begin{matrix} Annual\ Average \\ Exposure \end{matrix}_{A,H} = \frac{\left(Seasonal\ \ Average\ Exposure_{A,H,Winter}\right) + \left(Seasonal\ Average\ Exposure_{A,H,Summer}\right)}{2}$$

Model B. The model used for tabulating annual average concentration for each age and home-stove type combination.

$$\begin{aligned} \begin{matrix} Seasonal\ Average \\ Exposure \end{matrix}_{A,H,S} &= \left(\%t_{in,D,S} * \begin{matrix} Seasonal\ Daytime\ Indoor \\ Average\ PM2.5\ Exposure \end{matrix}_{A,H,S} \right) \\ &+ \left(\%t_{in,N,S} * \begin{matrix} Seasonal\ Nightime\ Indoor \\ Average\ PM2.5\ Exposure \end{matrix}_{A,H,S} \right) \\ &+ \left(\%t_{out,D,S} * \begin{matrix} Seasonal\ Daytime\ Outdoor \\ Average\ PM2.5\ Exposure \end{matrix}_{A,H,S} \right) \\ &+ \left(\%t_{out,N,S} * \begin{matrix} Seasonal\ Nightime\ Outdoor \\ Average\ PM2.5\ Exposure \end{matrix}_{A,H,S} \right) \end{aligned}$$

Model C. The general model for estimating seasonal average concentrations in sub-population of age "A" and home-stove combination "H" in season "S". Seasonal averages are calculated for summer and winter. "%t" is the percentage of period "D,S" or "N,S" spent in the denoted microenvironment.

i. $$\begin{matrix} Seasonal, Daytime\ Indoor \\ Average\ PM2.5\ Exposure \end{matrix}_{A,H,S} = PM_{in,H,S} + \left(IF_{H,S} * \ PM_{out,H,D,S}\right)$$

ii. $$\begin{matrix} Seasonal, Nighttime\ Indoor \\ Average\ PM2.5\ Exposure \end{matrix}_{A,H,S} = PM_{in,H,S} + \left(IF_{H,S} * \ PM_{out,H,N,S}\right) + \left(PM_{in,ETS,S}\right)$$

iii. $$\begin{matrix} Seasonal, Daytime\ Outdoor \\ Average\ PM2.5\ Exposure \end{matrix}_{A,H,S} = PM_{out,H,D,S}$$

iv. $$\begin{matrix} Seasonal, Nighttime\ Outdoor \\ Average\ PM2.5\ Exposure \end{matrix}_{A,H,S} = PM_{out,H,D,S}$$

Model D. The general models for estimating daytime and nighttime exposures in the indoor and outdoor microenvironments in sub-population of age "A" and home-stove combination "H" in season "S". In this context, "IF" is the infiltration factor of home-stove combination "H" (seasonally specific for gers).[23]

Population weighting and time activity

Annual average $PM_{2.5}$ exposures incorporate population weighting at two levels. In model series D, average outdoor ambient concentrations are calculated separately for populations residing in the three generic home types (ger population, house population, and apartment population), by season, and by semi-diurnal period. These concentrations are taken as outdoor averages of each cell in the outdoor concentration model weighted by home-type-specific population, with the exception of two districts – Baganuur and Bagakhangai. These districts were excluded from the outdoor $PM_{2.5}$ modeling, and so average outdoor $PM_{2.5}$ concentrations for these areas are simply assumed at the 10th percentile (decile) of population-weighted values for each season (Table D-1).[24] Then in Model A, sub-population annual averages are weighted by the fraction of the total population for which they account in order to arrive at the city-wide weighted average annual exposure.

Decile Outdoor Average Concentrations (μg/m^3) – Weighted by Total Population				
	10th %ile Winter Day	10th %ile Winter Night	10th %ile Summer Day	10th %ile Summer Night
Baseline	49.8	69.2	1.3	1.9
T-13	59.9	79.4	2.4	3.3
Scenario 1	22.9	29.8	0.4	0.6
Scenario 2	0.6	0.7	0.2	0.3

Table D-1. First decile of outdoor average $PM_{2.5}$ concentrations as weighted by total population for each season and scenario. Values are assumed for outdoor ambient concentrations in the two districts excluded from the Outdoor Model.

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[23] For gers and houses with stove or low-pressure boiler heating systems, infiltration is accounted for in the estimate of "PM_{in}", and so wintertime infiltration factors are set to 0.

[24] These districts are located in rural areas relatively far from the city center, and so are assumed to have lower outdoor ambient $PM_{2.5}$ concentrations.

Semi-diurnal averages are also weighted, but by time spent indoors vs. outdoors, rather than by population. Little information exists on the time-activity patterns of Mongolians. However, a recent wintertime survey administered by Social Impact (SI) takes account of the number of hours that children and adults living in Ulaanbaatar in gers or houses with MCA stoves spend inside and outside of the household (Social Impact (SI) 2013a). The results of this survey provide insight into the behavioral patterns of UB residents in the winter, but are not fully representative of each district, home type, stove type, and season. They thereby cannot be directly applied to our study. Instead, we combine these survey results with reasonable judgment and expert opinion to make wintertime time-activity assumptions for the entire population of Ulaanbaatar. From wintertime time-activity patterns, we make reasonable assumptions about summertime time-activity patterns. Time activity numbers are estimated for non-children (> 5 years old), children, and caretakers (Table D-2). It is expected that children and their caretakers spend the majority of their time indoors, and so they are assigned an indoor time activity of 100% for the entire year.[25] For non-children, our observations suggest two semi-diurnal phases of activity: a ten hour daytime during which many non-children leave the home to tend to work or school (8:00 to 18:00), and a fourteen hour nighttime period spent at the household (18:00 to 8:00).

Daytime Time Activity - Winter		
	Indoors	**Outdoors**
Child	1	0
Non-Child	0.75	0.25
Caretaker	1	0

Nighttime Time Activity - Winter		
	Indoors	**Outdoors**
Child	1	0
Non-Child	1	0
Caretaker	1	0

Daytime Time Activity - Summer		
	Indoors	**Outdoors**
Children	1	0
Non-Child	0.75	0.25
Caretaker	1	0

Nighttime Time Activity - Summer		
	Indoors	**Outdoors**
Child	1	0
Non-Child	1	0
Caretaker	1	0

Table D-2. Assumed fraction of daytime and nighttime periods spent indoors and outdoors by season and age category.

It is expected that the microenvironments in which residents spend their daytime periods differ by socio-demographics and season, but because no data on this breakdown are readily apparent, we make a conservative assumption to account for such variation: at a population level, the average amount of daytime hours spent indoors is 75%, and the average amount of daytime hours spent outdoors is 25%.[26,27] Because no data are available on the amount of evening and morning hours spent indoors, we make a simple assumption that 100% of children's daytime and nighttime hours are spent indoors. It is

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[25] Even if this is not the case, increased summertime ventilation due to open windows, doors, and ger flaps would result in very similar summertime $PM_{2.5}$ concentrations between indoor and outdoor environments.

[26] This assumption is conservative, because the true amount of daytime spent outdoors in UB is likely greater than 25% as employment in UB is dominated by sectors involving much outdoor work: construction, agricultural service, mining, quarrying, and transportation (Chilkhaasuren and Baasankhuu 2010).

[27] For simplicity, the concentration profiles of the indoor environments in which the population spends their time away from home are assumed the same as those of their home indoor environments.

expected that a subset of the non-child population is charged with taking care of the children, and so we assign one non-child as a caretaker per every child. Caretakers are assigned the same time activity profile as children. It is assumed that children, caretakers, and non-children are distributed evenly to each household, so that, for example, the fraction of the total population that lives in gers is also the fraction of caretakers, children, or non-children that live in gers.

Representations of exposures by residential population type and scenario

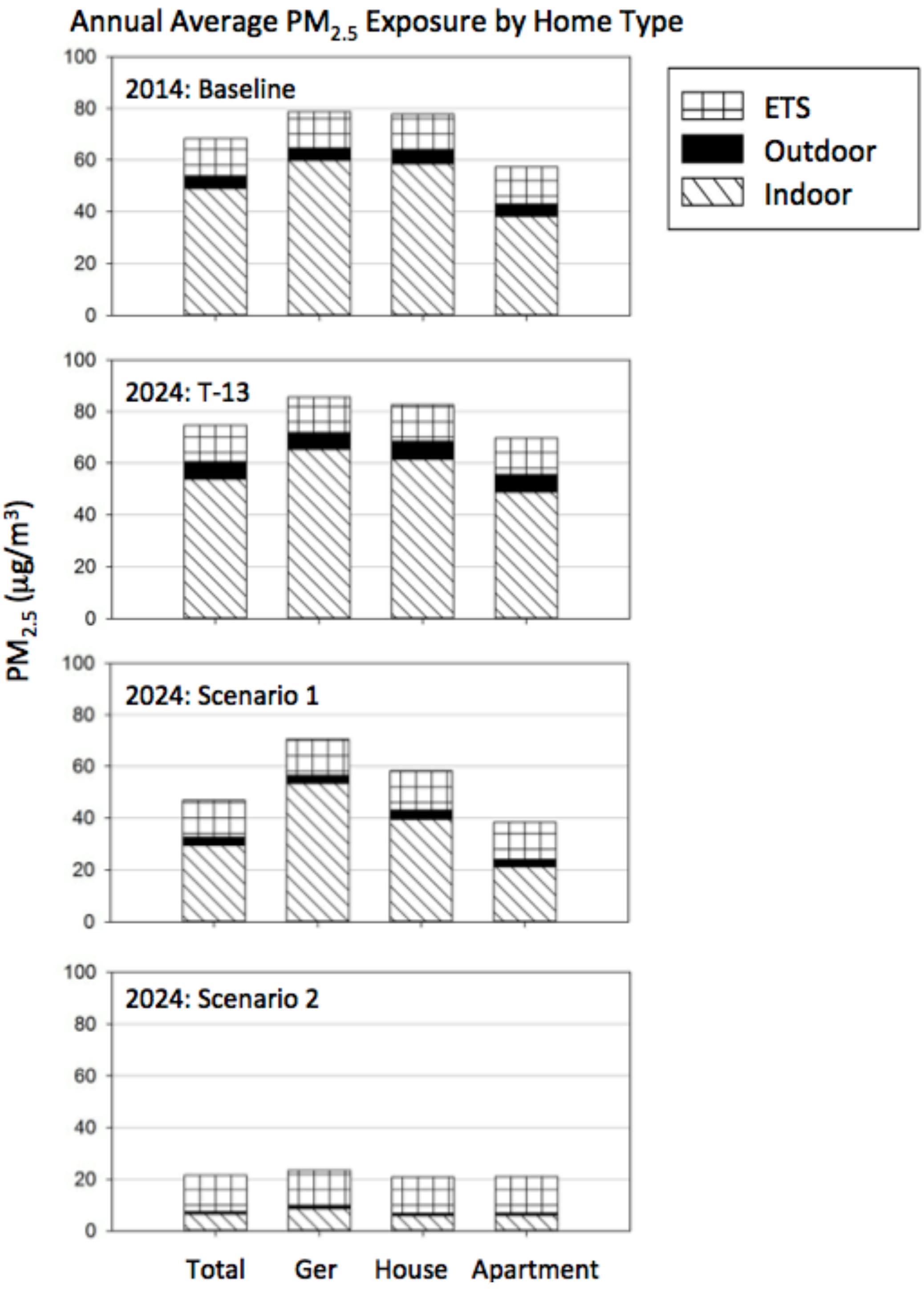

Figure D-1. Graphical representation of exposures by residential population type, scenario, and exposure type.

Appendix E. Health Burden and Benefit Calculation

Current national pattern of ill-health

The most important diseases in Mongolia in terms of lost life years in thousands are shown in Table E-1. Note the importance of IHD, Acute LRI, and stroke, which together account for nearly one-third of all lost life years. Of course, many other small causes contribute as well. From the Global Burden of Disease website (IHME, 2014):

Disease in Mongolia Ranked by Life-Years Lost in 2010

	Lost life Years (1000s)	Percent of Total
1. Ischemic Heart Disease*	96	11.8
2. Lower Respiratory Infections*	87	10.6
3. Stroke*	78	9.5
4. Neonatal Encephalopathy	41	5
5. Road Injury	34	4.2
6. Cirrhosis	32	3.9
7. Liver Cancer	30	3.6
8. Preterm Birth Complications	28	3.5
9. Congenital Anomalies	25	3.1
10. Tuberculosis	20	2.4
11. Chronic Kidney Disease	19	2.3
12. Stomach Cancer	16	2
13. Self-harm	16	2
14. Rheumatic Heart Disease	14	1.8
15. Interpersonal Violence	14	1.7
16. Mechanical Forces	14	1.7
17. COPD*	13	1.5
18. Lung Cancer*	12	1.5

Table E-1. Diseases in Mongolia as ranked by the number of life-years lost, 2010. From: http://www.healthmetricsandevaluation.org/gbd/country-profiles

Explanation of methods used in this assessment

Primary health results of the three scenarios are reported across disease categories and age groups. Health results are tabulated using a modified version of HAPIT (Household Air Pollution Intervention Tool) currently being developed by colleagues at the University of California, Berkeley, and the United Nations Foundation (Pillarisetti et al. 2013). This modified HAPIT allows for the projection of scenario-specific population attributable risk fractions for Lung Cancer, Ischemic Heart Disease, Stroke, and Chronic Obstructive Pulmonary Disorder in UB residents as well as Acute Lower Respiratory Tract Infection in UB children (ages 0-4 years) for 2014 through 2024. These five primary health indicators were chosen to allow for comparison with the national, regional, and global burden estimates published in the 2010 Institute for Health Metrics and Evaluation Global Burden of Disease Study, which uses these health outcomes as the indicators of air pollution related mortality and morbidity (Murray et al. 2012). While these are the only diseases to currently meet the rigorous standards of a reliable integrated exposure function, the literature relating $PM_{2.5}$ exposures to increased risk of Pulmonary Tuberculosis mortality are reaching a critical mass. For this reason, the health burden of tuberculosis is calculated as a secondary outcome.[28]

Deaths and DALYs

Burden of disease in UB is calculated in comparison to the expected burden under a 12.0 $\mu g/m^3$ exposure level (counterfactual).[29] Burden in excess of that incurred under the counterfactual is then calculated for 2014- 2024 under "Trends as of mid-2013" (T-13) policies as well as each of the defined scenarios. Note the equations used in the modified-HAPIT tool employ a counterfactual of 7.3 $\mu g/m^3$. To estimate accrued burden against a counterfactual of 12.0 $\mu g/m^3$, we used the modified-HAPIT to estimate the burdens that would accrue under a 12.0 $\mu g/m^3$ exposure level, and then subtracted those values from the modified-HAPIT estimates of accrued burden for T-13, Scenario 1, and Scenario 2 (see Appendix H for raw modified-HAPIT output). Exposures employed are population-weighted averages, explained in greater detail elsewhere in the report. A linear path is assumed between baseline (2014) and 2024 $PM_{2.5}$ exposure concentrations. Exposure levels for the baseline year are estimated and set equal between T-13 and the scenarios. All exposure paths are assumed to take the form of step function changes occurring on the first day of each year, the first of which occurs on January 1, 2015. Exposure changes occur for 10 years, finally halting on Jan 1, 2024. Health burden and related benefit over T-13 levels are calculated in terms of deaths and DALYs for each of these 11 years (2014 – 2024). Appendix H shows burden estimates for each scenario by year.

Morbidity is calculated in the form of disability adjusted life years (DALYs). DALYs are widely used around the globe to take into account both the age distribution of premature mortality and the severity of non-fatal diseases. DALYs also provide the best metric for cost-effectiveness in terms of the health benefit per dollar spent. Morbidity in the form of DALYs is calculated using the national disease-specific Death: DALY ratio defined in the IHME Global Burden of Disease study. The ratios used are derived from the 2010 GBD and are assumed constant throughout the projection period (Institute for Health Metrics & Evaluation (IHME) 2013).

Deaths Averted and DALYs Averted for each scenario are calculated as the difference between the number of deaths or DALYs accrued in that scenario and the number accrued in T-13. Remaining Burden of Attributable Disease in 2024 is calculated for both deaths and DALYs, and is the percentage of the amount of burden expected in 2024 under T-13 that is estimated in 2024 in each scenario. It is defined as the following:

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[28] One of the goals of this study was to estimate the health impact on three major secondary outcomes: tuberculosis, low birth weight, and cataracts. However, historical health records for the background disease rate modeling period (2006-2012) for low birth weight and cataracts were too low to infer any trends, which is an essential part of the impact evaluation process. We thereby exclude these diseases from our evaluation.

[29] This level represents the US Environmental Protection Agency's annual air quality standard for $PM_{2.5}$, and is the strictest national $PM_{2.5}$ standard in the world at present.

$$Remaining\ Burden\ in\ 2024 = \left(\frac{Estimated\ Burden\ in\ 2024\ in\ Scenario}{Estimated\ Burden\ in\ 2024\ in\ T13}\right) * 100$$

Relative risk

Morbidity and mortality are calculated as a function of relative risk. Specifically, relative risk is used to find the population attributable risk fraction (PAF) for each disease. PAF in this instance is the fraction of background disease due $PM_{2.5}$ exposure, and is defined in as:

$$PAF\ = \frac{RR - 1}{(RR - 1) + 1}$$

Where RR is the relative risk of the disease in question at the exposure level of interest (Desai et al. 2004). Each disease-specific PAF is then multiplied by the background rate of that disease to arrive at the quantity of disease attributable to $PM_{2.5}$ in that year. Upper and lower bounds for each PAF are also calculated.

Dose-response curves for mortality risk are derived from integrated exposure response data provided by the authors of the Global Burden of Disease 2010 (Burnett et al., 2014). A functional form was fit to these data using Eureqa Formulize (Schmidt and Lipson 2013). Upper and lower uncertainty bounds were also fit at the level of 95% confidence. We use these uncertainty limits to estimate the upper and lower bounds for disease-specific PAFs. The counterfactual exposure at which RR = 1.00 is assumed in these functions to be 7.3 μg/m^3. The models derived for these curves are reported in Table E-2.

Tuberculosis (TB) estimates

The Terms of Reference for the project included estimates for the impact on TB, but the results shown in Appendix H should be treated with caution because, unlike the other diseases, there is no published exposure-response information for TB and air pollution. The relative risk equation for pulmonary tuberculosis mortality was derived from estimates of relative risk (RR) for developing pulmonary tuberculosis. The RR of developing tuberculosis was arrived at by applying linear methods to odds ratios (OR) and exposure levels reported in the recent literature (Table E-2). We rely on a single study, rather than a meta-analytical estimate, to maintain consistency in the exposure circumstances among the “exposed” and “unexposed.” The most pertinent risk information appears to come from Lakshmi et al, who identify an odds ratio of 3.14 (95%CI: 1.15, 8.56) between female biomass users and female gas users in rural Nepal (Lakshmi et al. 2012). This is consistent with the current body of literature (Smith et al., 2014). Because the total TB risk in Nepal is less than 10%, this OR is considered a close approximation of RR (Kakchapati et al. 2010). While $PM_{2.5}$ exposures for the two groups were not directly measured, we assume a literature-reported average $PM_{2.5}$ exposure in female gas users of 70 μg/m^3, and in female biomass users of 300 μg/m^3, which was recently reported as a reasonable set of assumptions in nearby India (Burnett et al. 2014). A line was drawn between OR=1.00 at 70μg/m^3 and OR=3.14 at 300 μg/m^3. This line was extended to estimate the OR at the counterfactual exposure of 7.3

$\mu g/m^3$. The resulting linear equation was then normalized so that the OR at the counterfactual exposure was set to 1.00. This normalized equation is the RR equation for tuberculosis. We simulate a conservative uncertainty factor of ± 50% by adding 50% to premature death and DALY estimates (upper sensitivity) and subtracting 50% from premature death and DALY estimates (lower sensitivity).

Relative Risk Equations	
IHD	(IHD RR) = 0.9564*log(log(12.98 + PM))
Lower Uncertainty Bound	(Low IHD) = (41.12 + PM - 421.9/(3.363 + PM))^0.09225 - 0.2773
Upper Uncertainty Bound	(Low IHD) = (2.281 + (-422.7 - 20.98*PM)/(435.4 + PM^2)
Stroke	(Stroke RR) = 2.125 + (-172.9 - 19.42*PM)/(227.9 + PM^2)
Lower Uncertainty Bound	(Low Stroke) = (1.199 + 0.03729*PM)^0.13
Upper Uncertainty Bound	(Stroke RR) = 2.075 + 5.204e-5*PM + (-710 - 8.422*PM)/(643.9 + PM^2 - PM)
COPD	(COPD RR) = sqrt(1.398 + 0.00638*PM + -24.08/(54.74 + PM))
Lower Uncertainty Bound	(Low COPD) = 1.007 + 0.001675*PM + 2.848e-10*PM^3 - 7.654e-7*PM^2
Upper Uncertainty Bound	(Up COPD) = sqrt(1.538 + 0.01125*PM + -17.93/(27.84 + PM))
Lung Cancer	(LC RR) = 1.015 + 0.00596*PM + 1.781e-8*PM^3 + 5.276e-15*PM^5 - 1.047e-5*PM^2 - 1.557e-11*PM^4
Lower Uncertainty Bound	(Low LC) = 0.9849 + 0.001803*PM - 2.049e-8*PM^2
Upper Uncertainty Bound	(Up LC) = 1.032 + 0.01001*PM + 7.365e-8*PM^3 + 7.736e-14*PM^5 - 2.268e-17*PM^6 - 1.053e-10*PM^4 - 2.927e-5*PM^2
ALRI	(ALRE RR) = (9304 + 27.36*PM + 3.328*PM^2)/(8704 + 38.75*PM + PM^2)
Lower Uncertainty Bound	(Lower CI) = exp((23.51*PM + PM^2)/(6063 + 138*PM + PM^2))
Upper Uncertainty Bound	(Upper CI) = (53.9 + 3.371*PM)/(65.26 + PM) - 0.02199*PM*exp(-0.01778*PM)
Tuberculosis*	(TB RR) = (0.0225*PM) + 0.8426
Lower Uncertainty Bound	na
Upper Uncertainty Bound	na

*See text for explanation

Table E-2. Relative risk equations used in calculation of health burden from $PM_{2.5}$.

Non-Linear Exposure Response

The shapes of the dose-response curves derived from Burnett et al. (2014) provide new insight into the urgency of reducing $PM_{2.5}$ exposures to very low levels. Figure E-2 demonstrates the relationships between $PM_{2.5}$ exposure and mortality from IHD, stroke, and ALRI are all quite supralinear, with those for lung cancer and COPD closer to linear. This indicates larger health benefit from equal exposure reductions at the lower end of the exposure range than at the upper end, and thereby suggests that substantial health benefits may only be seen after large reductions in exposure (Smith et al., 2014).

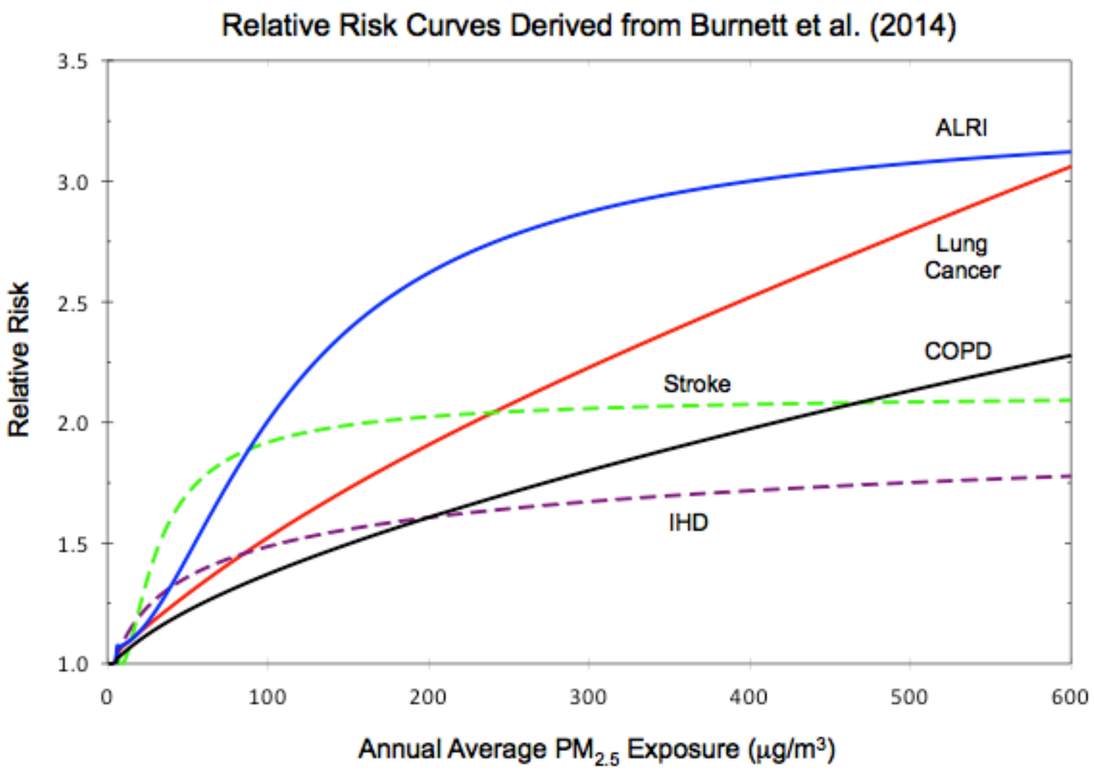

Figure E-1. Supra-linear relative risk curves derived from Burnett et al (2014) and use to tabulate health burden and benefits related to the five primary diseases assessed in this study.

Cessation lag

Health burdens and benefits are reported as committed or committed-averted deaths and DALYs due to exposure reductions between 2014 - 2024, which means the total impacts that are likely to ever be seen due to exposure reductions in only that period. However, because many $PM_{2.5}$-related illnesses take several years to develop in the population and because exposure effects can be long lasting (i.e., impacts can be experienced long after initial exposure), it is more likely that related health risks reduce gradually after $PM_{2.5}$ exposures are diminished and that some of the deaths and DALYs averted from exposure reductions between 2014-2024 will actually be incurred after that period (Doll et al. 2004; Stapleton and West 2012).[30] Our assumption of committed averted DALYs is thereby likely to accelerate the accrual of health benefits in our calculations, and result in slightly inflated estimates for the time period given. For comparison, we have tabulated in the appended modified HAPIT spreadsheets the deaths, DALYs, averted deaths, and averted DALYs under the assumption of no cessation lag and under the standard 20-year Distributed Lag scheme suggested by the Science Advisory Board to the US EPA (Cameron and Ostro 2004). In this scheme 30% of total burden reduction occurs in the year of exposure reduction, 50% of burden reduction is distributed evenly among years 2-5, and the final 20% of reductions are distributed evenly among years 6-20. Due to its acute nature, no lag period is applied to ALRI in this tabulation.

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[30] The major exception to this rule is acute lower respiratory tract in children, which is thought to be caused by recent exposures.

Cost effectiveness

This assessment does not provide specific analysis for cost-effectiveness because the estimation of program costs for each scenario is beyond our team's mandate. The modified HAPIT spreadsheet tool, however, has been designed to allow for such calculations to be performed by the Ministry as information becomes available. Cost-effectiveness analysis using the tool requires three additional inputs: total startup cost, average annual cost during the program period, and the estimated average GDP/capita over the program period, all of which must be reported in the same constant currency rates.[31] Cost-effectiveness is determined by comparing the expected annual cost of the intervention per DALY to the GDP/Capita (PPP) in international dollars. The WHO's Choosing Interventions that are Cost-Effective (WHO CHOICE) effort advises that interventions costing less than the GDP/capita (PPP) are very cost-effective, those costing one to three times the GDP/capita (PPP) are cost-effective, and those costing more than three times the GDP/capita (PPP) are not cost-effective.

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[31] Constant international 2005 dollars are recommended.

Appendix F: Limitations in Methods and Data

PM Emissions Estimation: Emission inventories prepared for UB have dramatic differences. For example, year 2010 inventories prepared by Guttikunda et al. (2013) and JICA (2013) differ by as much as a factor of ten depending on the emission source category. For the case of HOB these inventories include different groups of sources, but in other cases the source categories appear similar yet the emission estimates are quite different. Emission inventory improvements are needed and in their absence any modeling results will have large uncertainties. The inventory methodology needs to be transparent and of sufficient detail for users to be able to understand the strengths, limitations, and areas of needed improvement. Emissions data must also reflect the extreme wintertime conditions in UB. For example, emission testing of a stove by sampling from the flue and diluting with clean air at indoor temperature may dramatically underestimate the outdoor PM emissions because co-emitted gaseous semivolatile compounds will enter the particle phase upon mixing with the cold outdoor air. This dynamic process will depend on combustion efficiency and environmental conditions.

Air Quality Model Selection: A more sophisticated air quality model is needed to improve the accuracy of outdoor particulate matter concentrations. The Gaussian plume model ISCST-3 was used for this analysis. All Gaussian plume models will fail to capture important physical processes that affect the estimated concentrations. For the case of UB, the complex terrain (such as valleys) will trap emissions and lead to concentrations higher than model predictions. The model predicts concentrations for a given hour from the emissions in that hour. It has no memory of emissions in previous hours and therefore does not account for accumulation of pollutants during stagnant conditions that prevail during the UB wintertime. The model also assumes the wind conditions (speed and direction) are uniform across the city. The complex terrain channels air flow, however, and at any moment there can be large differences in wind conditions across the city. Puff models can account for some but not all of the limitations and a chemical transport model is recommended. However, it is recognized that considerable resources will be needed to develop the input parameters and optimize model performance for UB.

Ambient Air Quality Data: Model evaluation requires high-quality outdoor PM data collected at sites that are well characterized and ideally not strongly influenced by emissions sources that are very close to the monitoring site(s). The network must be carefully designed to meet the measurement objectives. For example, a roadside monitor provides information about pollution levels in the near-road environment but has less utility in urban- and larger-scale air quality model evaluation. Also, special care is needed to minimize measurement artifacts which may have different characteristics for UB wintertime conditions than typically observed in more temperate areas. For example, the extent to which semivolatile compounds partition to the particle phase in the outdoor air will differ under the warmer conditions of the shelters used to house automated instruments or in the laboratory after collection onto filters. These dynamics must be evaluated to properly interpret and use the data.

Health Effects: The health burden calculations in this study rely on projections of future trends in background disease rates and thus are more uncertain the further they are projected into the future. This is one of the main reasons we limit all our projections to 10 years. In addition:

- The models used in this study to extrapolate background burdens are relatively simple and may benefit from more sophisticated techniques in future assessments.
- Due to the principles of cessation lag, the 2006-2012 death records may not adequately account for the recent trends in increasing air pollution.
- Thus, the background health models used in this assessment, which are based on 2006-2012 health statistics, may underestimate the rate of increase in background death rates, and thus be conservative.

For each scenario, a linear annual progression in $PM_{2.5}$ exposure concentrations is assumed from baseline (2014) to the final scenario year (2024). This overlooks a number of likely nuances — such as rapid introduction of interventions, abrupt population changes, and non-uniform changes in grid cell population densities during outdoor ambient modeling — but allows us to make the fewest assumptions in the face of considerable uncertainty. With several of the IER curves used in this study defined as convex functions, such an assumption could overlook the true magnitudes of health impacts resulting from any dramatic increases (or decreases) in year-to-year exposures.

Morbidity in the form of DALYs is calculated using national disease-specific Death: DALY ratios defined in the IHME Global Burden of Disease study. The ratios used are (a) estimated at the National level by IHM rather than the city level, and (b) are assumed constant throughout the projection period. These ratios are likely to change as age structure alters, but are difficult to predict in which direction they will change.

Scenarios for City Growth: The scenarios for development of the urban area of Ulaanbaatar were taken from the Master Plan for the city, including the transition to apartment buildings, etc. The exposure and health estimates will be affected by rates of change that differ from those outlined in the Master Plan. Similarly the rates of population migration to the city of Ulaanbaatar are based on historical growth, which may change significantly over the next 10 years. Such changes are hard to predict and incorporate into this assessment.

Smoking: Disease estimates do not incorporate the health impacts of active smoking, although these are present in background disease prevalence rates. Increased smoking rates can significantly affect the estimation of health benefits, as they may outweigh the gains made by reducing emissions in other sectors. In addition, prevalence of smoking in workplace environments can significantly increase exposures. This is not adequately accounted for in these models.

Indoor Concentrations: Indoor concentrations of air pollution are currently dominated by stove emissions and smoking. As concentrations decrease, however, other sources become more important in the home, including contributions from ambient air including traffic and other outdoor sources, and indoor sources from cooking, incense burning, cleaning, resuspended dust, etc. Here we focus on the principal stove emissions and smoking, but acknowledge indoor concentrations are unlikely to decrease to the low concentrations identified under gas/district heating.

Industrial Emissions: Ambient models in Ulaanbaatar do not incorporate emissions from industrial sources as there is very limited information about the number of, or the emissions from, these sources.

It is clear they play a role in air quality in Ulaanbaatar, but it is not currently possible to incorporate these emissions sources into the models.

Coal Types: There is large diversity in the coal used in Ulaanbaatar. Currently, most household consumption is a mix of Nalaikh, Baganuur and Alagtolgoi. The ratio of these, and the influx of other coal types, will likely change the emissions from household stoves and indoor air concentrations as they are dependent on a fuel stove combination. Since the performance of improved combustion stoves is dependent on the fuel used in the stoves, the potential gains of stove dissemination projects is highly dependent on Ulaanbaatar fuel policies. At minimum we recommend that policies do not promote coal types that lead to greater emissions from stoves as that is counterproductive to the other measures to improve air quality in Ulaanbaatar. Similarly, processed coal testing should be undertaken before promotion of the fuel to ensure that emissions in household stoves are not increased. In Scenario 1, we incorporate stoves that result in a similar decrease in emissions as that seen in the MCA program. As mentioned above, however, greater health benefits can be achieved with a focus on stoves that also reduce emissions into indoor air. Emissions may also increase, however, if fuel policies promote lower quality fuels.

Appendix G. Population and Health Projections

Population data and estimates

Total citywide population data for 2006 - 2012 were available from official city and federal statistics publications (Mendsaikhan et al. 2011; National Statistics Office of Mongolia (NSOM) 2007; Statistics Department of Ulaanbaatar 2013). Population data for residents < 5 years old were available for 2006, 2007, 2011, and 2012, and are linearly interpolated for years 2008-2010 from 2007 and 2011 estimates. Population projections (total population and population < 5 years old) for Ulaanbaatar for 2010, 2015, 2020, 2025, and 2030 were available from the National Statistics Office. Annual population estimates for future years (2014-2030) are arrived at by linearly interpolating from the 5-year "medium" growth projections provided by the 2010 Population and Housing Census of Mongolia Report (Figure G-1) (National Statistics Office of Mongolia (NSOM) 2012). District-specific populations in 2012 were taken from city census data (Statistics Department of Ulaanbaatar 2013). Tabulated population estimates are detailed in Table H-1 of Appendix H.

Household type projections

Projections for the number of total households in each year were unavailable. We estimate this figure from Population and Family Size as identified in census data. As previously discussed, population figures are interpolated from NSOM projections of total city population (rather than district level populations). Family size is extrapolated by fitting a curve to historical trends (2000-2010) obtained from Mongolia’s Annual Statistical Yearbook series. Because a linear extrapolation results in atypically low family sizes, we use trends in the Total Fertility Rate (TFR) to set a reasonable lower limit.

TFR is defined by the UN as the average number of children a hypothetical cohort of women would have at the end of their reproductive period if they were subject during their whole lives to the fertility rates of a given period and if they were not subject to mortality. While modeling the UN’s 5-year TFR estimates against the 10 available years of household size data is unhelpful for identifying a relationship between the two, we make the assumption that trends in family size will follow national TFR.

Mongolia’s TFR has experienced a dramatic decrease in recent decades, but is expected to level off. The UN suggests that Mongolia’s TFR will become stable at near-Western levels through 2030 (United Nations Department of Economic and Social Affairs (UN DESA): Population Division 2013). This suggests that while the estimated rate of decline in household size observed between 2000-2010 was steep and linear, it is likely to level off in the near future. For this reason, we used Eureqa Formulize to fit a curve to past data that would approximate a near-term approach of 2.6 persons/home, the average US household size in 2010 (Schmidt and Lipson 2009, Bureau 2012).[32] Household sizes for individual years during 2010-2030 were then taken from this curve (Figure G-2). Household number is estimated by dividing the expected size of Ulaanbaatar’s population in each year by the average household size.

Household types most relevant to the Ulaanbataar context were identified as gers, single family houses, and multi-family apartments. A ger is defined as a traditional yurt-like dwelling most commonly

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[32] We assume one family per “household,” thus family size and household size are equivalent in our calculations.

constructed from wood and felt; a house is defined as a non-ger dwelling in which one family resides, are typically wood, cement, or brick structures in the style of traditional western houses; and apartments are identified as buildings within which two or more families are living, which, in most cases, are large complexes that house dozens of families.

Gers and houses are most common to regions identified by the Statistics Department of Ulaanbaatar as “ger areas,” while apartment households most commonly occupy what are known as “apartment areas.” Projections of the proportion of Ulaanbaatar residents living in the "ger areas" (%g) were provided by the 2010 Ulaanbaatar City Plan for 2010, 2020, and 2030. Annual trends are linearly interpolated (Figure G-1). The proportion of households located in ger areas is taken as the proportion of residents living in the ger area (%g). The proportion of Ulaanbaatar families living in the “apartment areas” (%a) is calculated for each year as 1 - %g (i.e. any home that is not in the ger area). Approximately 99% of apartment area households were classified by the Statistics Department of Ulaanbaatar as “apartment” households in 2012, and so other household types are not considered an important contribution to our apartment district estimates.[33] However, Statistics Department figures show a fairly even distribution between “ger” and “house” households in the ger areas. In 2012, ger households accounted for 44.2% of households in the ger areas (%g_g), while house households accounted for approximately 54.5% of ger area households (%g_h). Because the remaining household types make up a proportion of ger district households that is likely within the range of uncertainty, we exclude them from our analysis and calculate the proportion of ger district households as “houses” (%g_h) as 1 - %g_g. Reliable trend projections for %g_g and %g_h are unavailable, and so they are assumed constant between 2013 and 2030. Total number of households by area and type are graphically displayed in Figure G-3.

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[33] Household heating emissions estimates are not calculated for what the City Statistical Office classifies as “luxury house” and “homeless,” as they only make up about 0.9% and 0.01% of total apartment district households, respectively, and 1% and 0.2 % of total ger district households, respectively.

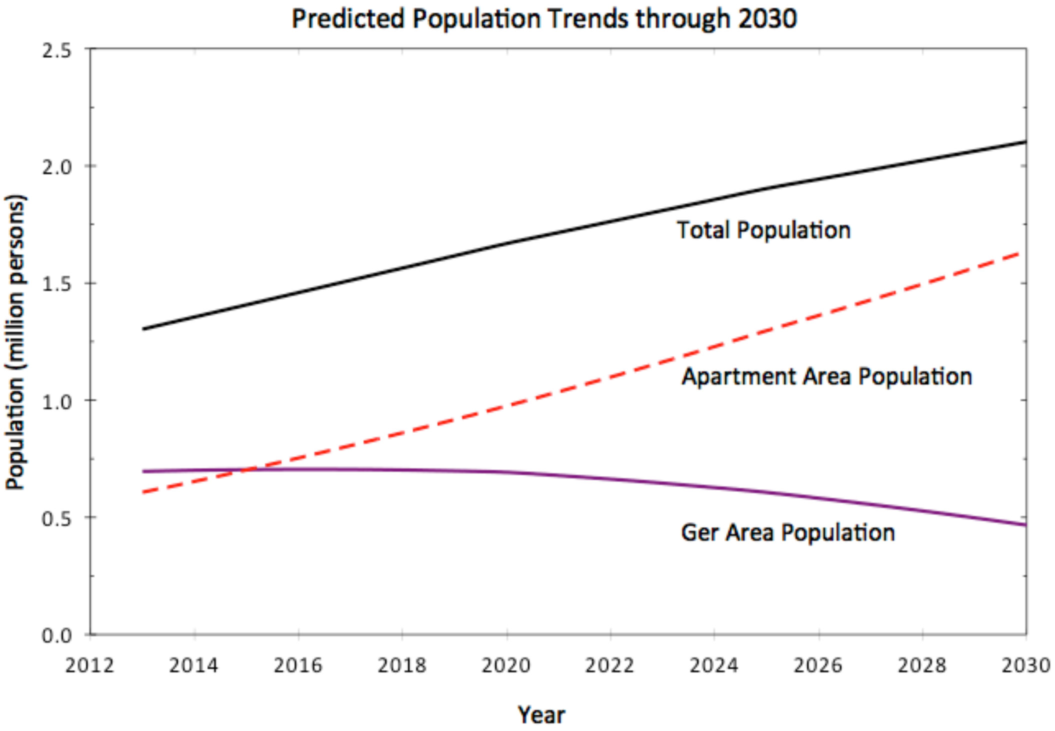

Figure G-1. Predicted trends in population numbers for 2013-2030 and distribution of population among apartment areas and ger areas. Total population is expected to increase steadily, while ger area populations are expected decline.

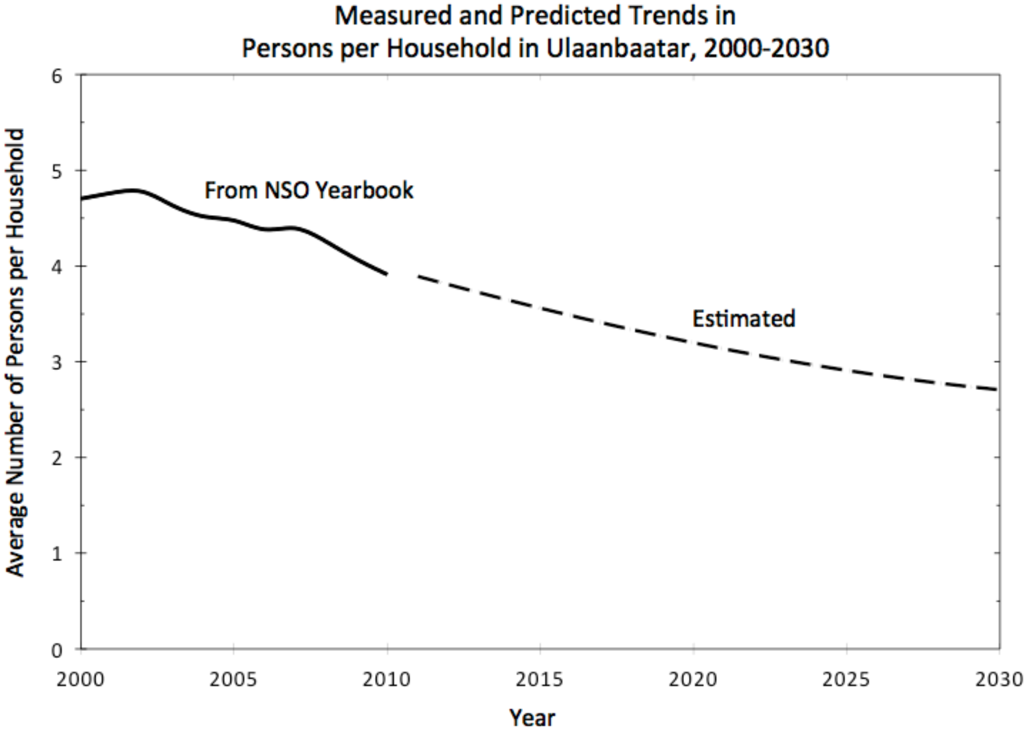

Figure G-2. Trends in family size (persons per household) as identified by the National Statistics Office of Mongolia for 2000 – 2010, and as estimated using extrapolation and assumptions of the Total Fertility Rate for 2011 - 2030.

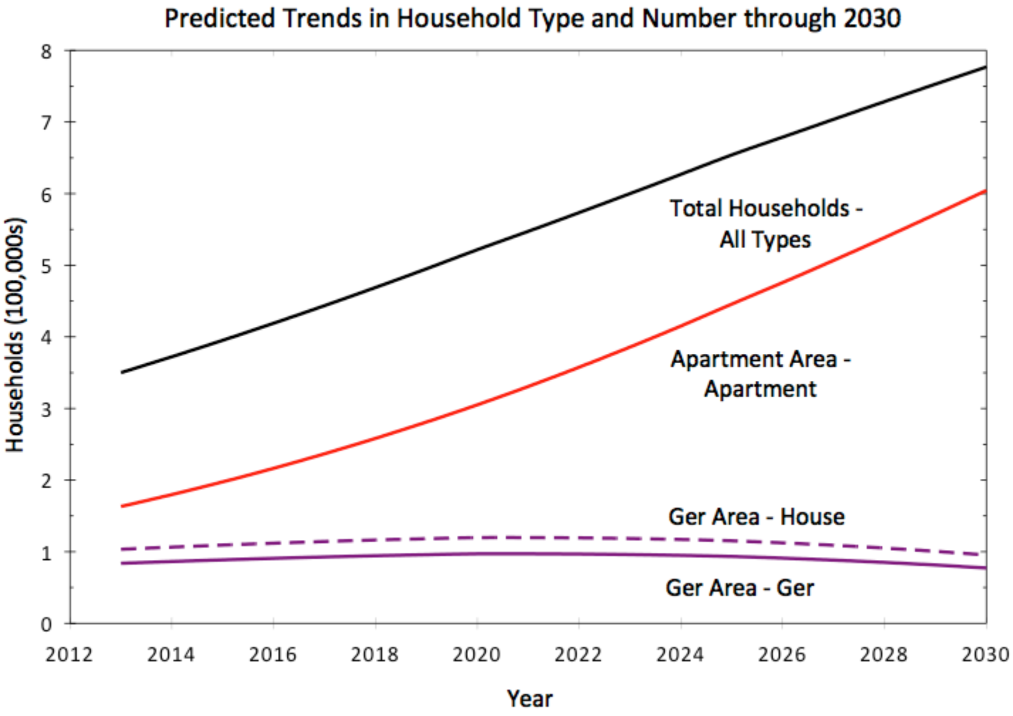

Figure G-3. Predicted trends in household number by area and by household type for 2013-2030. The number of households in apartment areas is expected to increase dramatically.

Background disease rate projections

Mortality burdens accrued during T-13 and each scenario are reported. Background death rates are essential for such calculations, and so future death rates for Ulaanbaatar are projected for 2014-2024 by carefully extrapolating mortality trends from all available years of historical data, and then modifying them to correspond with more rigorously adjusted international datasets.

Mortality data for the capital city for 2006-2012 were provided by the Health Development Center of the Ministry of Health in conjunction with the Health Sciences University of Mongolia. Deaths matching the ICD-10 codes reported in the 2010 IHME GBD study were requested (Table G-1). Trends in disease-specific death rates over this period are modeled as a linear function of time (year) and population (Table G-2). Each equation takes the general form of:

$$\#\, Deaths\ in\ Year\ X = \beta_0 + \beta_1 * Year + \beta_2 * AllPop$$

Where “AllPop” is the population in year X, and “Year” is the four number representation of year X (e.g. 2009). Ulaanbaatar population data (all population and child) for 2006- 2012 were obtained from annual national Statistics Yearbook reports (Mendsaikhan et al. 2011; National Statistics Office of Mongolia (NSOM) 2007). Tabulated background disease estimates are detailed in Table H-1 of Appendix H. Population levels through 2024 are projected using the various demographic tools identified previously in this appendix.

Due to a considerable discrepancy discovered between locally identified death rates and internationally identified death rates, disease specific models are not applied directly to arrive at the number of deaths in years 2014-2024. Recent international estimates of Mongolia-wide disease-specific mortality rates (deaths / 100,000 population) show values that are between 5% and 500% higher than those identified by the Ministry of Health for Ulaanbaatar (Lim et. al., 2012). While national disease rates should not exactly mirror those of the capital city, they are expected to be similar.[34] It is suspected these discrepancies arise from the inclusion of “garbage codes,” or improperly coded deaths, in the raw Ministry of Health dataset.[35] Garbage codes are a well-known phenomenon and occur frequently all over the world. International teams like the Institute for Health Metrics and Evaluation (IHME) employ rigorous statistical and diagnostic methods to redistribute such deaths to their probable underlying causes. This results in disease-specific mortality estimates that more accurately represent true rates.

To account for the impact of garbage codes on health burden results, our disease-specific models were adjusted using data provided by IHME (Institute for Health Metrics & Evaluation (IHME) 2013). Specifically, a ratio of UB-specific death rates (reported by the Ministry of Health) to national death rates (Lim et. al., 2012) was prepared for each disease (Table G-3). Ratios were created for 2010, which was the only year for which both IHME and Ministry of Health estimates were available. Citywide rates from the Ministry of Health were compared with IHME nationwide rates because IHME does not provide death rates at the city level. To arrive at the background disease profile expected under national disease rates for 2014-2024, 2010 ratios were applied to the background disease projections that were originally estimated from the mortality provided by the Ministry of Health. Health burden analyses were then run with the ratio-adjusted background disease profile.

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[34] IHME does not provide death rates at the city level, and so nationwide rates were used. As Ulaanbaatar accounts for approximately half of the national population, these rates should approximate each other.

[35] The Institute for Health Metrics and Evaluation identifies garbage codes as “causes of death that should not be identified as underlying causes of death but have been entered as the underlying cause of death on death certificates” (Lozano et al. 2012).

ICD Codes of Disease Categories Included in the Health Impacts Evaluation		
Disease Name	**ICD-10 Code Group**	**ICD-9 Code Group**
Cardiovascular Diseases		
Ischemic heart disease	I20 – I25	410-414
Ischemic Stroke	I63, I65 - I67 (except I67.4), I69.3	433-434, 437.0-437.2, 437.5-437.8
Hemorrhagic and other non-ischemic stroke	I60-I62, I69.0 - I69.2, I67.4	430-432, 437.2
Lung Cancer (trachea, bronchus, & lung cancer)	C33- C34, D02.1-D02.2, D38.1	162-162.9, 231.1, 231.2, 231.8, 235.7
All Chronic Respiratory Diseases		
Chronic Obstructive Pulmonary Disease	J40 – J44, J47	490-492.8, 494, 496
All Lower Respiratory Infections, < 5 years old		
Influenza	J09 – J11	487
Pneumococcal pneumonia	J13	481
H influenza type B pneumonia	J14	482.2
Respiratory syncytial virus pneumonia	J12.1	480.1
Other lower respiratory infections	J12 (except J12.1), J15 – J22, J85, P23	466, 480.0-480.9(except 480.1), 482.0-482.9(except 482.2), 483-486, 513, 770.0
Secondary Outcomes		
Tuberculosis	A15 - A19, B90, P37.0	010 – 018, 137.0 – 137.4, 320.4, 730.4 – 730.6

Table G-1. Names and related ICD-10 and ICD-9 codes of the disease categories included in the health impacts estimation. ICD codes were selected to match the 2010 Global Burden of Disease Study.

Disease Rate Projection Models

	β0	SE	β1 (Year)	SE	β2 (All Pop)	SE	*Total Model p-value*
Lung Cancer	-1.28E+04	4.21E+04	6.458	2.12E+01	-4.82E-05	3.96E-04	*6.50E-01*
CVD	-1.86E+05	3.18E+05	9.38E+01	1.60E+02	-7.06E-04	2.99E-03	*2.83E-01*
ALRI	-2.37E+04	4.21E+04	1.19E+01	2.12E+01	-1.27E-04	3.96E-04	*4.60E-01*
COPD	-3.23E+04	1.70E+04	1.62E+01	8.44E+00	-1.99E-04	1.60E-04	*4.94E-02*
Isch. Heart Disease	-1.49E+05	2.07E+05	7.47E+01	1.04E+02	-5.66E-04	1.94E-03	*1.87E-01*
Isch. Stroke & Other	-3.76E+04	1.17E+05	1.92E+01	5.86E+01	-1.41E-04	5.86E+01	*6.08E-01*
Tuberculosis	3.43E+04	6.28E+04	-1.72E+01	3.16E+01	3.33E-04	5.91E-04	*8.57E-01*

Table G-2. Disease rate projection models, as employed in the background disease burden estimation.

Background Mortality Rates for 2010, Ministry of Health Rates for UB vs. Lim et al (2012) National Rates

Disease	**2010 Deaths per 100,000 (Source: MOH)**	**2010 Deaths per 100,000 (Source: Lim et al, 2012)**	**Ratio of MOH UB rate to Lim (2012) National Rate**
Lung Cancer	12.07	18.14	0.67
ALRI (ages 0-4)	76.92	276.82	0.28
COPD	4.43	21.66	0.20
Ischemic Heart Disease	85.80	164.19	0.52
Stroke	76.60	124.22	0.62
Tuberculosis	15.72	16.14	0.97

Table G-3. Background mortality rates for 2010, Ministry of Health rates for UB vs. Lim et al (2012) national rates.

Appendix H. Supplementary Health Burden Tables

Table H-1 (Part 1). Estimated background disease rates through the scenario period.

Year	Population	Population 0-4	Trachea, Bronchus, Lung Cancers Deaths	Trachea, Bronchus, Lung Cancers DALYs	Lower Respiratory Infections Deaths (0-4)	Lower Respiratory Infections DALYs (0-4)	COPD Deaths	COPD DALYs
2014	1,355,176	148,219	227	5,678	416	35,640	425	16,621
2015	1,407,196	155,551	233	5,826	435	37,290	453	17,739
2016	1,459,516	158,438	239	5,974	454	38,927	482	18,845
2017	1,511,836	161,325	245	6,122	473	40,565	510	19,952
2018	1,564,157	164,212	251	6,270	493	42,203	538	21,059
2019	1,616,477	167,099	257	6,418	512	43,841	566	22,165
2020	1,668,797	169,986	263	6,566	531	45,478	595	23,272
2021	1,715,748	168,427	269	6,723	552	47,325	628	24,583
2022	1,762,700	166,869	275	6,881	574	49,173	662	25,895
2023	1,809,651	165,310	281	7,039	595	51,020	695	27,206
2024	1,856,603	163,752	288	7,196	617	52,867	729	28,517

Table H-1 (Part 2). Estimated background disease rates through the scenario period.

Year	IHD Deaths (Ischemic Heart Disease)	IHD DALYs (Ischemic Heart Disease)	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke Deaths	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke DALYs	Tuberculosis Deaths	Tuberculosis DALYs
2014	2,064	45,975	1,471	33,808	155	8,963
2015	2,151	47,903	1,493	34,321	156	8,973
2016	2,237	49,823	1,516	34,832	156	8,989
2017	2,323	51,744	1,538	35,343	156	9,005
2018	2,409	53,665	1,560	35,855	156	9,021
2019	2,495	55,585	1,582	36,366	157	9,037
2020	2,582	57,506	1,605	36,877	157	9,053
2021	2,674	59,556	1,628	37,410	155	8,963
2022	2,766	61,606	1,651	37,942	154	8,873
2023	2,858	63,656	1,674	38,474	152	8,783
2024	2,950	65,706	1,697	39,006	151	8,693

Table H-2 (Part 1). Estimated health burden from $PM_{2.5}$ under "Counterfactual" exposure (12.0 $\mu g/m^3$)

Year	Trachea, Bronchus, Lung Cancers Deaths from $PM_{2.5}$	Trachea, Bronchus, Lung Cancers Deaths from $PM_{2.5}$, per 1000 capita	Trachea, Bronchus, Lung Cancers DALYs from $PM_{2.5}$	Trachea, Bronchus, Lung Cancers DALYs from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$ per 1000 children affected	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$ per 1000 children affected	COPD Deaths from $PM_{2.5}$	COPD Deaths from $PM_{2.5}$ per 1000 capita	COPD DALYs from $PM_{2.5}$	COPD DALYs from $PM_{2.5}$ per 1000 capita
2014	18	0.013	445	0.328	33	0.024	0.222	2815	2.077	18.990	22	0.016	872	0.643
2015	18	0.013	457	0.324	34	0.024	0.221	2945	2.093	18.932	24	0.017	930	0.661
2016	19	0.013	468	0.321	36	0.025	0.226	3074	2.106	19.404	25	0.017	988	0.677
2017	19	0.013	480	0.317	37	0.025	0.232	3204	2.119	19.858	27	0.018	1046	0.692
2018	20	0.013	491	0.314	39	0.025	0.237	3333	2.131	20.297	28	0.018	1104	0.706
2019	20	0.012	503	0.311	40	0.025	0.242	3462	2.142	20.720	30	0.018	1162	0.719
2020	21	0.012	515	0.308	42	0.025	0.247	3592	2.152	21.129	31	0.019	1220	0.731
2021	21	0.012	527	0.307	44	0.025	0.259	3737	2.178	22.190	33	0.019	1289	0.751
2022	22	0.012	539	0.306	45	0.026	0.272	3883	2.203	23.272	35	0.020	1358	0.770
2023	22	0.012	552	0.305	47	0.026	0.284	4029	2.227	24.374	36	0.020	1427	0.788
2024	23	0.012	564	0.304	49	0.026	0.298	4175	2.249	25.497	38	0.021	1496	0.806

Table H-2 (Part 2). Estimated health burden from $PM_{2.5}$ under "Counterfactual" exposure (12.0 $\mu g/m^3$)

Year	IHD Deaths (Ischemic Heart Disease) from $PM_{2.5}$	IHD Deaths (Ischemic Heart Disease) from $PM_{2.5}$ per 1000 capita	IHD DALYs (Ischemic Heart Disease) from $PM_{2.5}$	IHD DALYs (Ischemic Heart Disease) from $PM_{2.5}$ per 1000 capita	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke Deaths from $PM_{2.5}$	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke Deaths from $PM_{2.5}$ per 1000 capita	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke DALYs from $PM_{2.5}$	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke DALYs from $PM_{2.5}$ per 1000 capita	Tuberculosis Deaths from $PM_{2.5}$	Tuberculosis Deaths from $PM_{2.5}$ per 1000 Capita	Tuberculosis DALYs from $PM_{2.5}$	Tuberculosis DALYs from $PM_{2.5}$ per 1000 Capita
2014	218	0.161	4846	3.576	48	0.035	1095	0.808	16	0.012	907	0.669
2015	227	0.161	5049	3.588	48	0.034	1112	0.790	16	0.011	908	0.645
2016	236	0.162	5252	3.598	49	0.034	1128	0.773	16	0.011	910	0.623
2017	245	0.162	5454	3.608	50	0.033	1145	0.757	16	0.010	911	0.603
2018	254	0.162	5657	3.616	51	0.032	1161	0.742	16	0.010	913	0.584
2019	263	0.163	5859	3.625	51	0.032	1178	0.729	16	0.010	915	0.566
2020	272	0.163	6061	3.632	52	0.031	1194	0.716	16	0.010	916	0.549
2021	282	0.164	6277	3.659	53	0.031	1212	0.706	16	0.009	907	0.529
2022	292	0.165	6494	3.684	53	0.030	1229	0.697	16	0.009	898	0.509
2023	301	0.166	6710	3.708	54	0.030	1246	0.689	15	0.009	889	0.491
2024	311	0.167	6926	3.730	55	0.030	1263	0.680	15	0.008	880	0.474

Table H-3 (Part 1). Estimated health burden from $PM_{2.5}$ under Trends as of mid-2013 (T-13)

Year	Trachea, Bronchus, Lung Cancers Deaths from $PM_{2.5}$	Trachea, Bronchus, Lung Cancers Deaths from $PM_{2.5}$, per 1000 capita	Trachea, Bronchus, Lung Cancers DALYs from $PM_{2.5}$	Trachea, Bronchus, Lung Cancers DALYs from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$ per 1000 children	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$ per 1000 children	COPD Deaths from $PM_{2.5}$	COPD Deaths from $PM_{2.5}$ per 1000 capita	COPD DALYs from $PM_{2.5}$	COPD DALYs from $PM_{2.5}$ per 1000 capita
2014	62	0.046	1555	1.147	166	0.122	1.119	14214	10.489	95.901	93	0.068	3625	2.675
2015	64	0.046	1605	1.141	175	0.124	1.124	14979	10.644	96.294	99	0.071	3891	2.765
2016	66	0.045	1656	1.135	184	0.126	1.160	15746	10.789	99.384	106	0.073	4158	2.849
2017	68	0.045	1707	1.129	193	0.128	1.195	16521	10.928	102.411	113	0.075	4427	2.928
2018	70	0.045	1759	1.124	202	0.129	1.230	17304	11.063	105.379	120	0.077	4699	3.004
2019	72	0.045	1811	1.120	211	0.131	1.264	18095	11.194	108.289	127	0.079	4973	3.077
2020	75	0.045	1863	1.116	221	0.132	1.297	18893	11.321	111.143	134	0.080	5250	3.146
2021	77	0.045	1919	1.118	231	0.135	1.371	19785	11.532	117.471	142	0.083	5576	3.250
2022	79	0.045	1975	1.120	241	0.137	1.447	20686	11.735	123.966	151	0.086	5905	3.350
2023	81	0.045	2031	1.122	252	0.139	1.525	21595	11.933	130.632	159	0.088	6237	3.446
2024	83	0.045	2088	1.125	263	0.142	1.605	22511	12.125	137.473	168	0.090	6572	3.540

Table H-3 (Part 2). Estimated health burden from $PM_{2.5}$ under Trends as of mid-2013 (T-13)

Year	IHD Deaths (Ischemic Heart Disease) from $PM_{2.5}$	IHD Deaths (Ischemic Heart Disease) from $PM_{2.5}$ per 1000 capita	IHD DALYs (Ischemic Heart Disease) from $PM_{2.5}$	IHD DALYs (Ischemic Heart Disease) from $PM_{2.5}$ per 1000 capita	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke Deaths from $PM_{2.5}$	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke Deaths from $PM_{2.5}$ per 1000 capita	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke DALYs from $PM_{2.5}$	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke DALYs from $PM_{2.5}$ per 1000 capita	Tuberculosis Deaths from $PM_{2.5}$	Tuberculosis Deaths from $PM_{2.5}$ per 1000 Capita	Tuberculosis DALYs from $PM_{2.5}$	Tuberculosis DALYs from $PM_{2.5}$ per 1000 Capita
2014	606	0.447	13501	9.963	662	0.488	15204	11.219	90	0.066	5185	3.826
2015	633	0.450	14110	10.027	673	0.478	15466	10.991	90	0.064	5215	3.706
2016	661	0.453	14719	10.085	684	0.469	15728	10.776	91	0.062	5248	3.596
2017	688	0.455	15332	10.141	696	0.460	15990	10.577	92	0.061	5280	3.493
2018	716	0.458	15947	10.195	707	0.452	16253	10.391	92	0.059	5313	3.397
2019	744	0.460	16565	10.248	719	0.445	16515	10.217	93	0.057	5345	3.307
2020	772	0.462	17186	10.298	730	0.437	16778	10.054	93	0.056	5377	3.222
2021	801	0.467	17848	10.402	742	0.432	17051	9.938	93	0.054	5346	3.116
2022	831	0.472	18513	10.502	754	0.428	17324	9.828	92	0.052	5314	3.015
2023	861	0.476	19180	10.599	766	0.423	17597	9.724	92	0.051	5281	2.918
2024	891	0.480	19850	10.692	778	0.419	17870	9.625	91	0.049	5248	2.827

Table H-4 (Part 1). Estimated health burden from $PM_{2.5}$ under Scenario 1

Year	Trachea, Bronchus, Lung Cancers Deaths from $PM_{2.5}$	Trachea, Bronchus, Lung Cancers Deaths from $PM_{2.5}$, per 1000 capita	Trachea, Bronchus, Lung Cancers DALYs from $PM_{2.5}$	Trachea, Bronchus, Lung Cancers DALYs from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$ per 1000 children	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$ per 1000 children	COPD Deaths from $PM_{2.5}$	COPD Deaths from $PM_{2.5}$ per 1000 capita	COPD DALYs from $PM_{2.5}$	COPD DALYs from $PM_{2.5}$ per 1000 capita
2014	62	0.046	1555	1.147	166	0.122	1.119	14214	10.489	95.901	93	0.068	3625	2.675
2015	63	0.044	1564	1.112	170	0.120	1.090	14528	10.324	93.399	97	0.069	3797	2.698
2016	63	0.043	1571	1.077	173	0.118	1.090	14793	10.136	93.370	101	0.069	3955	2.710
2017	63	0.042	1576	1.042	175	0.116	1.086	15012	9.929	93.053	105	0.069	4102	2.713
2018	63	0.040	1578	1.009	177	0.113	1.079	15182	9.706	92.451	108	0.069	4238	2.709
2019	63	0.039	1578	0.976	179	0.110	1.069	15301	9.465	91.566	111	0.069	4361	2.698
2020	63	0.038	1575	0.944	179	0.107	1.055	15367	9.208	90.401	114	0.068	4472	2.680
2021	63	0.037	1572	0.916	180	0.105	1.070	15447	9.003	91.712	118	0.069	4608	2.686
2022	63	0.036	1566	0.888	181	0.102	1.082	15465	8.774	92.680	121	0.069	4728	2.682
2023	62	0.034	1557	0.860	180	0.099	1.089	15421	8.521	93.284	123	0.068	4832	2.670
2024	62	0.033	1544	0.832	179	0.096	1.091	15312	8.247	93.505	126	0.068	4919	2.649

Table H-4 (Part 2). Estimated health burden from $PM_{2.5}$ under Scenario 1

Year	IHD Deaths (Ischemic Heart Disease) from $PM_{2.5}$	IHD Deaths (Ischemic Heart Disease) from $PM_{2.5}$ per 1000 capita	IHD DALYs (Ischemic Heart Disease) from $PM_{2.5}$	IHD DALYs (Ischemic Heart Disease) from $PM_{2.5}$ per 1000 capita	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke Deaths from $PM_{2.5}$	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke Deaths from $PM_{2.5}$ per 1000 capita	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke DALYs from $PM_{2.5}$	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke DALYs from $PM_{2.5}$ per 1000 capita	Tuberculosis Deaths from $PM_{2.5}$	Tuberculosis Deaths from $PM_{2.5}$ per 1000 Capita	Tuberculosis DALYs from $PM_{2.5}$	Tuberculosis DALYs from $PM_{2.5}$ per 1000 Capita
2014	606	0.447	13501	9.963	662	0.488	15204	11.219	90	0.066	5185	3.826
2015	625	0.444	13929	9.898	667	0.474	15329	10.893	89	0.063	5114	3.634
2016	644	0.441	14337	9.823	672	0.460	15442	10.580	87	0.060	5042	3.455
2017	661	0.437	14728	9.741	676	0.447	15543	10.281	86	0.057	4967	3.285
2018	678	0.433	15098	9.653	680	0.435	15630	9.993	85	0.054	4888	3.125
2019	694	0.429	15448	9.556	683	0.423	15701	9.713	83	0.052	4804	2.972
2020	708	0.424	15775	9.453	685	0.411	15754	9.441	82	0.049	4717	2.826
2021	723	0.422	16112	9.391	687	0.401	15796	9.207	79	0.046	4570	2.663
2022	737	0.418	16422	9.317	688	0.390	15815	8.972	77	0.043	4420	2.508
2023	750	0.414	16703	9.230	688	0.380	15807	8.735	74	0.041	4268	2.358
2024	761	0.410	16951	9.130	686	0.370	15769	8.493	71	0.038	4112	2.215

Table H-5 (Part 1). Estimated health burden from $PM_{2.5}$ under Scenario 2

Year	Trachea, Bronchus, Lung Cancers Deaths from $PM_{2.5}$	Trachea, Bronchus, Lung Cancers Deaths from $PM_{2.5}$, per 1000 capita	Trachea, Bronchus, Lung Cancers DALYs from $PM_{2.5}$	Trachea, Bronchus, Lung Cancers DALYs from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections Deaths (0-4) from $PM_{2.5}$ per 1000 children	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$ per 1000 capita	Lower Respiratory Infections DALYs (0-4) from $PM_{2.5}$ per 1000 children	COPD Deaths from $PM_{2.5}$	COPD Deaths from $PM_{2.5}$ per 1000 capita	COPD DALYs from $PM_{2.5}$	COPD DALYs from $PM_{2.5}$ per 1000 capita
2014	62	0.046	1555	1.147	166	0.122	1.119	14214	10.489	95.901	93	0.068	3625	2.675
2015	61	0.043	1526	1.084	165	0.117	1.058	14096	10.017	90.622	95	0.067	3707	2.635
2016	60	0.041	1490	1.021	161	0.111	1.019	13835	9.479	87.321	96	0.066	3758	2.575
2017	58	0.038	1446	0.956	157	0.104	0.971	13423	8.879	83.207	97	0.064	3777	2.498
2018	56	0.036	1394	0.891	150	0.096	0.914	12853	8.217	78.269	96	0.061	3760	2.404
2019	53	0.033	1333	0.825	141	0.087	0.846	12118	7.496	72.519	95	0.059	3704	2.291
2020	51	0.030	1263	0.757	131	0.078	0.770	11221	6.724	66.012	92	0.055	3603	2.159
2021	47	0.028	1185	0.690	119	0.070	0.708	10220	5.957	60.679	89	0.052	3481	2.029
2022	44	0.025	1094	0.621	106	0.060	0.635	9084	5.153	54.436	84	0.048	3296	1.870
2023	40	0.022	992	0.548	92	0.051	0.555	7860	4.343	47.545	78	0.043	3037	1.678
2024	35	0.019	875	0.471	77	0.042	0.472	6619	3.565	40.424	69	0.037	2692	1.450

Table H-5 (Part 2). Estimated health burden from $PM_{2.5}$ under Scenario 2

Year	IHD Deaths (Ischemic Heart Disease) from $PM_{2.5}$	IHD Deaths (Ischemic Heart Disease) from $PM_{2.5}$ per 1000 capita	IHD DALYs (Ischemic Heart Disease) from $PM_{2.5}$	IHD DALYs (Ischemic Heart Disease) from $PM_{2.5}$ per 1000 capita	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke Deaths from $PM_{2.5}$	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke Deaths from $PM_{2.5}$ per 1000 capita	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke DALYs from $PM_{2.5}$	Ischemic Stroke, Hemorrhagic and other non-ischemic stroke DALYs from $PM_{2.5}$ per 1000 capita	Tuberculosis Deaths from $PM_{2.5}$	Tuberculosis Deaths from $PM_{2.5}$ per 1000 Capita	Tuberculosis DALYs from $PM_{2.5}$	Tuberculosis DALYs from $PM_{2.5}$ per 1000 Capita
2014	606	0.447	13501	9.963	662	0.488	15204	11.219	90	0.066	5185	3.826
2015	618	0.439	13755	9.774	661	0.470	15192	10.796	87	0.062	5016	3.565
2016	626	0.429	13949	9.557	658	0.451	15127	10.364	84	0.057	4834	3.312
2017	632	0.418	14076	9.311	652	0.432	14994	9.918	80	0.053	4631	3.063
2018	634	0.405	14124	9.030	643	0.411	14772	9.444	76	0.049	4404	2.816
2019	632	0.391	14076	8.708	628	0.388	14430	8.927	72	0.045	4151	2.568
2020	625	0.374	13911	8.336	606	0.363	13924	8.344	67	0.040	3864	2.316
2021	612	0.357	13629	7.943	574	0.335	13193	7.690	61	0.035	3498	2.039
2022	591	0.335	13156	7.464	528	0.299	12125	6.879	54	0.030	3095	1.756
2023	558	0.308	12432	6.870	460	0.254	10561	5.836	46	0.025	2645	1.462
2024	510	0.275	11364	6.121	360	0.194	8280	4.460	37	0.020	2139	1.152

Works Cited

Allen RW, Adar SD, Avol E, Cohen M, Curl CL, Larson T, et al. 2012. Modeling the residential infiltration of outdoor $PM_{2.5}$ in the Multi-Ethnic Study of Atherosclerosis and Air Pollution (MESA Air). Environ. Health Perspect. 824:824–830; doi:10.1289/ehp.1104447.

Baigalmaa D, Nishimura A, Ito K. 2006. Smoking cessation rate 12 months after a group counseling program in Mongolia. Asian Pac. J. Cancer Prev. 7: 399–402.

Burnett RT, Pope CA, Ezzati M, Olives C, Lim S, Mehta S, et al. 2014. An integrated risk function for estimating the Global Burden of Disease attributable to ambient fine particulate matter exposure. Environ. Health Perspect. 122:397–403; doi:10.1289/ehp.1307049.

Cameron T, Ostro B. 2004. Advisory Council on Clean Air Compliance Analysis response to agency request on cessation lag - letter to the United States Environmental Protection Agency.

Chilkhaasuren B, Baasankhuu B. 2010. Population and economic activities of Ulaanbaatar. 1–14.

Cowlin S, Kaufmann R, Edwards R, Smith K. 2005. Impact of Improved Stoves on Indoor Air Quality in Ulaanbaatar, Mongolia.

Desai MA, Smith KR, Prüss-üstün A, Campbell-lendrum D, Corvalán C, Woodward A. 2004. Indoor smoke from solid fuels: Assessing the environmental burden of disease at national and local levels.

Dockery DW, Spengler JD, Harriman K. 1981. Indoor-outdoor relationships of respirable sulfates and particles. Atmos. Environ. 15:335–343; doi:10.1016/0004-6981(81)90036-6.

Doll R, Peto R, Boreham J, Sutherland I. 2004. Mortality in relation to smoking: 50 years' observations on male British doctors. BMJ 328:1519; doi:10.1136/bmj.38142.554479.AE.

Draxler RR. 1999. HYSPLIT4 user's guide. NOAA Tech. Memo (ERL ARL-230).

Draxler RR, Hess GD. 1998. An overview of the HYSPLIT_4 modeling system of trajectories, dispersion, and deposition. Aust. Meteorol. Mag. 295–308.

Draxler RR, Hess GD. 1997. Description of the HYSPLIT_4 modeling system. NOAA Tech. Memo (ERL ARL-224).

Guttikunda SK, Lodoysamba S, Bulgansaikhan B, Dashdondog B. 2013. Particulate pollution in Ulaanbaatar, Mongolia. Air Qual. Atmos. Heal. 6:589–601; doi:10.1007/s11869-013-0198-7.

Hänninen OO, Lebret E, Ilacqua V, Katsouyanni K, Künzli N, Srám RJ, et al. 2004. Infiltration of ambient $PM_{2.5}$ and levels of indoor generated non-ETS $PM_{2.5}$ in residences of four European cities. Atmos. Environ. 38:6411–6423; doi:10.1016/j.atmosenv.2004.07.015.

Institute for Health Metrics & Evaluation (IHME). 2014. Global Burden of Disease data visualization. Available: http://www.healthmetricsandevaluation.org/tools/data-visualizations [accessed 10 April 2014].

Institute for Health Metrics & Evaluation (IHME). 2013. Mongolia Global Burden of Disease Study 2010 (GBD 2010) results 1990-2010.

Jantunen MJ, Katsouyanni K, Lebret E, Maroni M, Saarela K, Zmirou D. 1998. Final Report: Air pollution exposures in European cities: the EXPOLIS study.

Japan International Cooperation Agency (JICA). 2013. Capacity Development Project for Air Pollution Control in Ulaanbaatar City Mongolia, Final Report.

Kakchapati S, Yotthanoo S, Choonpradup C. 2010. Modeling tuberculosis incidence in Nepal. Asian Biomed. 4: 355–360.

Lakshmi PVM, Virdi NK, Thakur JS, Smith KR, Bates MN, Kumar R. 2012. Biomass fuel and risk of tuberculosis: a case-control study from Northern India. J. Epidemiol. Community Health 66:457–61; doi:10.1136/jech.2010.115840.

Lim SS, Vos T, Flaxman AD, Danaei G, Shibuya K, Adair-Rohani H, et al. 2012. A comparative risk assessment of burden of disease and injury attributable to 67 risk factors and risk factor clusters in 21 regions, 1990-2010: a systematic analysis for the Global Burden of Disease Study 2010. Lancet 380:2224–60; doi:10.1016/S0140-6736(12)61766-8.

Long CM, Suh HH, Catalano PJ, Koutrakis P. 2001. Using time- and size-resolved particulate data to quantify indoor penetration and deposition behavior. Environ. Sci. Technol. 35:2089–2099; doi:10.1021/es001477d.

Lozano R, Naghavi M, Foreman K, Lim S, Shibuya K, Aboyans V, et al. 2012. Global and regional mortality from 235 causes of death for 20 age groups in 1990 and 2010: a systematic analysis for the Global Burden of Disease Study 2010. Lancet 380:2095–128; doi:10.1016/S0140-6736(12)61728-0.

Mendsaikhan S, Gerelt-Od G, Erdenesuren B, Ganbat B, Bajiikhuu K, Oidovdanzan R. 2011. Mongolian Statistical Yearbook 2010.

Murray CJL, Ezzati M, Flaxman AD, Lim S, Lozano R, Michaud C, et al. 2012. The Global Burden of Disease Study 2010: design, definitions, and metrics. Lancet 380:2063–6; doi:10.1016/S0140-6736(12)61899-6.

National Statistics Office of Mongolia (NSOM). 2007. Mongolian Statistical Yearbook 2006.

National Statistics Office of Mongolia (NSOM). 2012. The 2010 population and housing census of Mongolia.

Pillarisetti A, Hanning C, Smith KR. 2013. Presentation to the Global Alliance for Clean Cookstoves, September 25: An overview of HAPIT: Household Air Pollution Intervention Tool. Estimating the health benefits of scaling up clean cooking technologies.

Schmidt M, Lipson H. 2013. Eureqa (Version 0.99.6 beta) [Software].

Sherman MH. 1987. Estimation of infiltration from leakage and climate indicators. Energy Build. 10:81–86; doi:10.1016/0378-7788(87)90008-9.

Smith KR, Bruce N, Balakrishnan K, Adair-rohani H, Balmes J, Dherani M, et al. 2014. Millions dead : how do we know and what does It mean? Methods used in the Comparative Risk Assessment of household air pollution. Am. Rev. Public Heal. 35: 185-206.

Social Impact (SI). 2013a. Impact evaluation final results report - MCA Mongolia Energy and Environment Project Energy Efficient Stove Subsidy Program (not yet finalized).

Social Impact (SI). 2013b. Presentation, September 27: Preliminary impact evaluation results from the MCA Mongolia Energy and Environment Program (EEP) stove subsidy project.

Stapleton J, West R. 2012. A direct method and ICER tables for the estimation of the cost-effectiveness of smoking cessation interventions in general populations: application to a new cytisine trial and other examples. Nicotine Tob. Res. 14:463–71; doi:10.1093/ntr/ntr236.

Statistics Department of Ulaanbaatar. 2013. Population and Household Census 2012.

Ulaanbaatar Clean Air Project (UB CAP). 2012. Ulaanbaatar Clean Air Project: Air quality mitigation in central UB component: Ulaanbaatar district heating supply in imporvement feasibility study, terms of reference.

United Nations Department of Economic and Social Affairs (UN DESA): Population Division. 2013. World Population Prospects The 2012 Revision Volume I : Comprehensive Tables. I.

United States Environmental Protection Agency (US EPA). 1995. User's Guide for the Industrial Source Complex (ISC3) Dispersion Models, Volumes I and II (EPA 454/B-95/003a).

Williams R, Suggs J, Rea A, Sheldon L, Rodes C, Thornburg J. 2003. The Research Triangle Park particulate matter panel study : modeling ambient source contribution to personal and residential PM mass concentrations. Atmos. Environ. 37:5365–5378; doi:10.1016/j.atmosenv.2003.09.010.

World Health Organization (WHO). 2010. Mongolian steps survey on the prevalence of noncommunicable disease and injury risk factors.