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The value of air quality standards

Review and update of cost benefit analysis of National Environmental Standards on air quality

Report to Ministry for the Environment

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Preface

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Executive summary

Introduction

This report updates the cost benefit analysis undertaken in 2004 of the National Environmental Standards (NES) on Air Quality that underpinned the Section 32 assessment for introducing the standards. The NES does this by setting a target level of pollutants in ambient air quality in defined airsheds, prohibiting some activities and setting standards for others that contribute to pollutant emissions, with the aim of reducing the frequency of dangerous levels of air pollution in urban locations.

Introduced in 2005 with the expectation of progressive achievement of the ambient air quality standards across all regions by 2013, the NES is implemented primarily by regional councils. In achieving the ambient air quality targets in their airsheds, regional councils they may adopt measures more stringent than those in the NES.

Limitations of the 2004 cost benefit analysis (CBA)

The 2004 analysis estimated costs and benefits over 17 years (2004-2020), discounted at a real rate of 10% per year, and concluded the NES would achieve a net present value of $318 million and a benefit cost ratio of 3.9.

It had a number of significant shortcomings (and some other less critical limitations) which have been addressed in this update:

- The largest benefit is the avoidance of premature death, hospitalisation and restricted activity days (RADs) and is probably understated because:
  - The benefit calculation attributed only 75% of the value of statistical life (VOSL) per life saved by introducing NES, on an assumption that air quality affects only elderly people and their VOSL is lower than the all-ages average. There is no empirical basis in New Zealand for either of these assumptions.
  - No allowance was made for the qualitative effects of lingering ill-health, the on-going welfare loss from gradual impairment due to prolonged exposure, or the adverse effects of air pollution on infants and those under the age of 30.
  - No explicit allowance was made for medical costs saved by reducing bad air days and their associated adverse effects on health.
- The costs to local, regional and central government in administering and monitoring implementation were probably overstated, because:
  - Recurring costs were attributed to local territorial authorities, but the NES places no direct requirements on them so they are unlikely to face any cost
  - Costs expected to fall on regional councils were substantially higher than those actually observed in the four years of NES implementation
  - Other costs to councils/government in subsidising households to convert to compliant wood burners assumed a subsidy scheme that did not eventuate.

- Costs assumed for industry may have understated the mean compliance cost for emitting sites and restraints on new consents if NES targets are not reached, but there does not appear to be any evidence that this is so.
- The private costs of households converting to compliant wood burners were omitted from the 2004 analysis but should be included, with due allowance for trends in home heating and the effects of recent government subsidies.

Because of data limitations and uncertainties the 2004 analysis tended to be conservative, erring on the high side for costs while being frugal on benefits, to reduce the possibility of a false positive result. The updated analysis does the same.

Updated CBA approach

The updated cost benefit analysis retains the structure of the 2004 analysis but updates it in light of new information. The update is not a retrospective analysis of implementation to date, but rather a forward looking analysis covering the period 2008-2020. If the air quality NES is successful in bringing down the ambient level of pollution, benefits will continue beyond that date.

The comparisons in the updated analysis have changed from a choice between implementing and not implementing the NES, to a choice between:

- Holding the achievements of the NES to date and allowing air quality to progress as business as usual (the counterfactual).
- Continued pursuit of full implementation of the NES and target attainment by 2013.
- Continued pursuit of full implementation of the NES and target attainment by 2020.

Two other scenarios which are more difficult to quantify are examined in a qualitative assessment. These were applying fines to incentivise regional councils that fall behind on their NES ambient air quality target achievement, and requiring non-compliant air plans in regions that fall behind on target achievement.

The updated analysis differs from the 2004 analysis principally in revising the value attached to benefits, updating the costs to industry and local and central government, and estimating a potential cost for households in upgrading their wood burners to comply with the NES. In particular:

- The value attached to lives saved and hospitalisation costs are higher than in the earlier analysis.
- Costs for territorial authorities, schools, hospitals and road authorities in the 2004 analysis have been removed as they are no longer relevant in the update.
- Costs on householders for upgrading wood burners to meet the standards, which were omitted from the 2004 analysis, have been explicitly modelled.
- The discount rate has also changed, from 10% in the 2004 analysis to 8% in the updated analysis, in line with the Treasury's current default discount rate.

Differences in assumptions and inputs used between the 2004 CBA and the 2009 CBA are summarised in the table below.

Summary of coverage of initial and updated analyses

	2004 Analysis	2009 Update
Benefits		
Willingness to pay to avoid		
-Premature loss of life (pain & suffering)	Included	Included
-Lost output/productivity/income		
Direct benefits of avoiding GDP loss	Included	
-Indirect benefits of avoiding GDP loss	Included	Not valued
Avoided costs of medical treatment	Not included	Included
Avoided loss of long term quality of life	Not included	Not valued
Costs associated with ambient air quality		
-Regional councils administration/monitoring	Included	Updated
-Territorial authorities administration	Included	Zero entry
-Government information & administration	Included	Updated
-Industry site adaptation measures	Included	Updated
-Business forgone from consent constraints	Not included	Not valued
Costs associated with prohibition standards		
-Consenting of school & hospital incinerators	Included	Completed
-Alternatives to tar seal burning	Included	Zero entry
-Other activities: landfills, wire burning etc	Zero entry	Zero entry
Costs associated with wood burner standard		
-Householders costs of compliant burners	Zero entry	Included
-Suppliers costs of compliant burners	Zero entry	Zero entry
-Government/council subsidy	Included	Zero entry
Factors taken into account in the analysis		
Infant mortality	Not included	Included
Cost of hospitalisation (medical expenditures)	Not included	Included
Discount rate	10%	8%
Influences on the counter-factual	Not included	Allows for downward trend in wood burners & insulation/ clean heat initiatives

Source: NZIER

Updated CBA results

The results of the updated cost benefit analysis are shown in the table below. They indicate the NES would be worthwhile, with the same benefit cost ratio as in the 2004 CBA but substantially higher net present value. If the standard is achieved by 2013, the early realisation of health benefits would result in a net present value of $955 million and a benefit cost ratio of 3.9. If standard achievement is postponed until 2020, deferral of benefits is greater than the reduction in costs, reducing the NPV to $159 million with a benefit cost ratio of 3.2.

Summary of updated CBA with baseline assumptions

Period to 2020 discounted at 8%

Ambient standards over status quo met by	**2013**	**2020**
Reduction in premature mortality (to 2020)	635	153
Reduction in hospitalisations (to 2020)	565	150
Reduction in Restricted Activity Days (to 2020)	1,034,452	269,367
PV combined benefits $M	1,289	232
PV Costs $M	333	74
NPV (GDP + VoSL - Costs) $M	955	159
B:C Ratio [(Gross Benefits)/(Costs)]	3.9	3.2
C Effectiveness [(Costs)/(Mortality reduction)]	$524,712	$481,807
Distribution of costs		
Regional councils	3.0%	10.9%
Territorial authorities	0.0%	0.0%
Central government	0.3%	1.4%
Industry	4.2%	15.1%
Road controlling authorities	0.0%	0.0%
Households	92.4%	72.6%

Source: NZIER

There are large differences in the distribution of costs across the community. The level and distribution of costs are predominantly driven by the number of households that incur costs in meeting the wood burner standard, which in the analysis is modelled as potentially having rather more impact than the NES prescribes, as more stringent measures may be applied by regional councils in pursuit of their ambient air quality targets under the NES. To the extent that the household impact is less, the cost and distribution of implementing the NES will also change.

Sensitivity analysis

Sensitivity analyses indicate this pattern of results is robust to large changes in some of the inputs used. In particular, the result that deferring target attainment saves implementation costs but incurs larger societal costs due to more deaths and hospitalisation is a recurring result of these analyses.

The size of the net benefit and benefit cost ratios indicate the analysis could accommodate substantially increased costs, and a reduction in benefits, before overturning the positive result.

Uncertainties and caveats

The timing and resources committed to this updated analysis mean that it has relied on existing gathered material, and been undertaken at a national level without delving into detail of local impacts. As it deals with issues over which uncertainty exists, this analysis cannot be viewed as a "precise" depiction of the implementation of the standards, but it can identify the main effects, magnitudes and uncertainties that affect the result.

The calculated net benefit is understated to the extent that benefits from improved air quality beyond 2020 are not counted; and also understated by the wood burner compliance costs assuming greater stringency than is likely under the NES. The main uncertainty on the cost side is the scale of compliance costs for industrial emitters and the possible opportunity cost for industrial emitters if consents are declined because emissions exceed the targeted level – but no such instances have yet been reported.

Key uncertainties that have not been possible to resolve in this analysis include:

- the precise relationship between measures applied, air quality and resulting impacts on health;
- the localised impacts on air quality and activity levels; and
- the likely value of benefits to the young and long term quality of life from improvements in air quality.

There are some omissions in the updated analysis which could be filled with more time and resources committed, for instance further work on localised impacts and consenting restraints. But further analysis is not costless and will only be net beneficial if it yields information that changes the interpretation of the results. The updated analysis has quantified the main effects of the NES and after qualitative assessment concluded that omissions are unlikely to be significant.

This analysis focuses on concentrations of particulate matter (e.g. PM_{10}) which nationally and internationally receives most attention in air quality control because of its adverse health impacts and correlation with other types of pollutant. Pollution levels fluctuate from year to year with the influence of weather and random variation, so recent observations that suggest the current level of PM_{10} pollution is on average lower than it was in 2002 before the NES are not conclusive proof of any actual, permanent improvement in air quality. Even if there has been a permanent reduction, it is not possible yet to determine how much of this can be attributed to the NES.

Conclusions

If the reductions in fatalities, hospitalisations and restricted activity days are as large as indicated in the baseline analysis, and values attached to those reductions are consistent with health and safety benefits elsewhere in the economy, the air quality NES would deliver a substantial benefit which is likely to be well in excess of its costs across New Zealand.

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1. Introduction

This report reviews and updates a cost benefit analysis of the air quality standards introduced in 2004. An ex-ante evaluation carried out for the Ministry for the Environment (MfE) in 2004 ("the 2004 CBA") showed a benefit to cost ratio of 3.9. This report's objective is to update these estimates and also illuminate the distribution of costs and benefits from the implementation of the air quality standards.

This is important because the number of deaths and other pollution related health effects are quite high. The 2004 CBA estimates that the number of deaths in 2003 was 872 and it would gradually reduce in the absence of air quality standards to 678 in 2013. To put this in perspective, Table 1 presents figures on deaths in New Zealand attributable to various causes in recent years.

Table 1 Recent mortality data

Cause of death	Year	Number of deaths
Total	2005	27,141
Cancer	2005	7,971
Ischaemic heart disease	2005	5,807
Cerebrovascular disease	2005	2,587
Lung Cancer	2005	1451
Diabetes	2005	839
Air pollution	2005	804
Breast Cancer	2005	648
Prostate Cancer	2005	564
Road toll	2005	405
Infant deaths	2004	374
Sudden Infant death Syndrome (Cot death)	2004	45

Source: Ministry of Transport for Road toll and Ministry of Health for other data, NZIER

While Cancer, Ischaemic heart disease and Cerebrovascular disease are the major causes of death, pollution effects are high in comparison with other causes of death: for example, the road toll in 2005 was 405, when the estimated number of deaths due to air pollution was 804 (Table 1). Air pollution may also contribute to other types of disease by exacerbating the condition and hastening mortality.

Air pollution is a significant risk to health, accounting for more than 2% of annual recorded deaths and substantial incidence of ill-health and reduced quality of life. Measures to reduce air pollution involve costs, but also result in benefits, the enumeration of which is the subject of this update of the 2004 CBA.

2. Framing the analysis

2.1 The problem to be addressed

The problem addressed by an air quality standard is the risk to health and other social and economic activity caused by air pollution, which at high levels of concentration can trigger restrictions of activities, and aggravation of medical conditions that causes hospitalisation and premature death. This is a localised problem with locally varying contributions from sources such as vehicle emissions, building heating emissions, and various industrial activities. Apart from industrial processes that can be identified and subjected to discharge consents, many of these sources are too numerous and mobile to monitor closely for control purposes (without incurring prohibitive costs), so regulation resorts to broader standards that control the activities that contribute to emissions or the resultant ambient air quality levels.

2.2 The effect of the standards

The basic aim of the National Environmental Standards (NES) is to reduce the frequency with which dangerous concentrations of pollution occur in particular locations where ambient air quality presents most risks to people. They do this by setting a target level of pollutants in ambient air quality, and setting standards for various activities and equipment that contribute to pollutant emissions that affect that ambient quality. The NES for air quality contain 14 standards set for the protection of human health. These include:

- Five standards for outdoor air quality (limits for specific pollutant levels)
- Seven standards that ban activities which discharge toxic substances to the air
- A design and performance standard for new wood burners in urban areas
- A standard on greenhouse gas recovery and destruction from landfills.

Much of the focus of the standards is on lowering the ambient levels of large particulate matter (PM_{10}) in the atmosphere, which is a pollutant implicated in various respiratory ailments and a contributor to premature mortality. There are various sources of PM_{10} in the atmosphere, including a substantial contribution from transport vehicles, but in New Zealand there is also a marked seasonal peak caused by increased fuel combustion for winter heating. Hence another particular focus of the standards is on improving the emission standard on wood burners for home heating, which have been estimated to account for more than 30% of home heating across New Zealand.[1]

2.2.1 What the standards do

The national environmental standard for air quality can be viewed as operating at two levels:

[1] Taylor Baines & Associates (2006) "The impact on housing energy efficiency of market prices, incentives and regulatory requirements"; Report to CHRANZ and Building Research

- Standards applying to individual activities or products, which regional authorities are required to adhere to and enforce
- Performance targets applying to the regional authorities themselves, which set a requirement to meet certain ambient pollution levels by specified dates, and to have the monitoring capability to accurately report on achievement against the required standard and occasions when it is exceeded.

In the case of PM10, the standard is to have a 24 hour average of no more than 50 µg/m3 and no more than one exceedance of this standard per year. The target for all regions is to achieve this standard by calendar year 2013. Regions can be expected to apply a different mix of measures to reach this target, tailored to the different characteristics of their airsheds, and different mix of activities contributing to PM10 concentrations.

A transition path has been determined for each airshed identifying the maximum annual emission levels permitted each year if the region is to meet the 2013 target. If an airshed exceeds the PM10 target on its transition path to 2013, the national standard prohibits further new consents for discharges to air being issued until the airshed is back on track to meet the target level. That provision will restrain only industrial emitters subject to resource consenting, although in most regions industrial emitters are not major contributors to increasing PM10 levels or seasonal peaking.

Domestic heating is a major contributor to the seasonal peaking in PM10 concentrations, which is why the standards tighten up on the design and emissions performance of domestic wood burners. Since September 2005, all new wood burners installed must have particle emissions of less than 1.5 grams per kilogram of dry wood burnt, and also achieve a thermal efficiency of at least 65%. However, the standards only apply to fuel wood burners (i.e. not including open fires, multi-fuel burners, pellet burners etc), and only to those installed in properties of 2 hectares or less (i.e. urban properties).2 The standards are also not retrospective, and hence apply only to the installation of new burners.

The national standard itself will have only a modest impact on changing the stock of wood burners in New Zealand, as new ones are installed or old ones are replaced. Regional councils, however, may introduce more stringent requirements for wood burners (such as retrofitting or upgrading burners to standard when properties are sold) in pursuit of their regional targets for lowering ambient PM10 levels.

2.2.2 How can their effect be measured?

The principal benefit sought from higher air quality standards is a reduction in adverse health effects brought about by poor air quality: i.e. reductions in premature mortality, lost production from days off work from respiratory illness, and restricted activity days (RADs) caused by high levels of atmospheric pollution. Although high

[2] The standard applies only to urban areas, both because this is where air quality problems are concentrated and also to avoid infringing the Trans Tasman Mutual Recognition Agreement with Australia, which makes wood burners to a standard of 4 g/kg wood burnt that cannot be prohibited from sale across all New Zealand under the TTMRA, and are permitted in rural areas.

pollution can directly trigger respiratory illnesses, sometimes so severely they lead on to death, they also have cumulative effects in contributing to the general deterioration in health of the population. Hence pollution contributes to both immediate effects and those with a long latency period which are much more difficult to identify and measure.

So although air quality effects could be measured by identifying associations between "bad air days" and heightened incidence of death, respiratory illness or activity restriction, such short term associations will not capture the full benefit of improved air quality. Longer term observations of the correlations between air quality and adverse outcomes are required. These national environmental standards have been in place for too short a period to provide such long term observations. Moreover, there can be some natural variability in the year-on-year results of ambient measurements, according to such environmental factors as the windiness of a particular location, the length and coldness of winter that affects the use of wood burners, or other extraneous factors that affect the volume and mode of transport used in the region.

The short period of implementation of the air quality NES means that an ex post analysis of implementation based on actual observations is not feasible. Our ex-ante analysis therefore relies on inferences drawn from analogous experience reported elsewhere, on councils' expectations about achievement of targets, and sensitivity testing around those inferences and assumptions.

3. Review of 2004 cost benefit analysis

The 2004 cost benefit analysis contains a health effects model and an economic model relating health outcomes to the wider effects on the economy. The benefits identified in that economic model are:

- Reduction in health (present value of $420 million over 17 years) from
- Reduction in premature deaths brought about by pollution
- Reduction in estimated hospitalisations due to effects of air pollution
- Reduction in estimated restricted activity days (RADs) caused by effects of air pollution
- Reduction in consequential flow on effects on the economy (present value of $9 million over 17 years) of reduced productivity (GDP impacts).

The costs identified in that model, as summarised in its Appendix C, are:

- Costs of implementing ambient air quality monitoring standards for district, councils ($3.5 million per year), regional councils ($3.2 million per year) and government ($0.1 million per year)
- Costs to industry on controlling its emissions ($1 million per year)
- Costs of obtaining consents for schools and hospital incinerators (one-off cost of $1.72 million in first year only)
- Costs of alternatives to road seal burning ($1 million per year)
- Costs to government and councils of implementing the wood burner standard ($5 million per year after the first year).

The analysis is estimated for a period of 17 years (2004-2020) and discounted at a real rate of 10% per year. Other key inputs and assumptions include:

- Growth in real GDP/capita of 1.5% per year
- An average of 1.5 years of life lost per case of premature death
- An average of 6.8 days in hospital and 5 days of recuperation per case of hospitalisation
- 40% of mortality and hospitalisations apply to those in employment, the rest to those retired (assuming health effects are higher among the elderly)
- The proportion of RADs off work is 55% (based on 10% of RADs being serious with 100% days off, 90% minor with 50% days off)
- The GDP impact is calculated by multiplying Full Time Equivalent work days lost, by a factor for inter-sectoral flow on effects with varying assumptions on input substitution.

The 2004 CBA compared the outcome with standards in place against a baseline with no standards, over the period 2004 to 2020. The model underlying the analysis shows no divergence between these two situations until 2009 i.e. if the model is realistic, there would be no observed improvement in outcomes until the end of this year. The results of the central setting of the 2004 CBA model, summarised over the 17 year analysis period, are presented in Table 2.

Table 2 Results of 2004 cost benefit analysis

Estimated over 17 years with discount rate of 10%

Reduction in premature mortality (to 2020)	625
Reduction in hospitalisations (to 2020)	571
Reduction in RADs (to 2020)	1,045,487
PV benefits (GDP)	$9,026,965
PV benefits (VoSL)	$420,152,814
PV combined benefits	$429,179,779
PV Costs	$110,795,472
NPV (GDP + VoSL - Costs)	$318,384,308
B:C Ratio [(GDP + VoSL)/(Costs)]	3.87
C Effectiveness [(Costs)/(Mortality reduction)]	$177,323

Source: Ministry for the Environment spreadsheet

In aggregate over the 17 year period, the air quality standard is expected to result in avoiding 625 cases of premature death (on average 36.7 per year), 571 cases of respiratory-related hospitalisation (33.6 per year), and over 1 million RADs (58,823 locality-specific RADs per year). The slightly unusual result that the reduction in mortality exceeds reduction in hospitalisations may be simply a categorisation effect, caused by those who die after being admitted to hospital being counted as mortality rather than hospitalisation cases.

The largest benefit in the 2004 CBA (PV$420 million) is the value of statistical lives saved (VOSL). This is the social value of avoiding the pain, grief and suffering caused by premature death and hospitalisation, and is derived from the VOSL calculated by the Ministry of Transport for use in transport project appraisals. The VOSL has been adjusted in the air quality CBA to reflect the assumption that most premature deaths will be among the elderly so avoided deaths are worth less than for the average traffic casualty in the prime of life. Whether these adjustments are appropriate will be explored later.

The benefits from avoiding lost productive output (GDP) are relatively small by comparison (PV$9 million). These are calculated from the number of working days lost to restricted activities, hospitalisation or premature death multiplied by one of three distinct factors:

- A pro rata loss of GDP based on the 2002 GDP per capita applied to lost FTE days

- A factor allowing for imperfect substitution of capital, energy and materials for lost labour input, derived from a general equilibrium model of the economy
- A factor allowing for replacement of lost labour by the unemployed with 29% lower productivity, derived from a general equilibrium model of the economy.

The pro rata factor results in a loss of $17 million being avoided, the partial substitution for a loss of $9 million being avoided and the full substitution of labour results in around a loss of $850,000 being avoided. Some substitution is likely to occur, and the greater the substitution the smaller the impact.[3] The initial CBA chose a mid-range estimate, but the differences are insignificant in the overall analysis.

The costs, with a present value of almost PV$111 million, far outweigh the estimated productivity benefit of the air quality standard. But combined with the VOSL benefits, there is a net present value benefit of $318 million and a benefit cost ratio of 3.9. On the basis of the 2004 analysis, the societal cost per premature death avoided would be around $177,000 in present value terms.

3.1 Review of evidence on likely costs

The costs used in the initial cost benefit analysis are summarised in Appendix 3 in the 2004 CBA. This itemises costs under three separate headings: implementation of ambient air quality targets, implementation of prohibitive standards, and implementation of the wood burner standard. The principal items are:

Ambient air quality

- District and city councils: $50,000 per council per year for 70 councils (i.e. $3.5 million per year) for the duration of the analysis
- Regional and unitary authorities: $200,000 per council per year for 16 authorities (i.e. $3.2 million per year) throughout the duration of the analysis
- Central government: $100,000 per year throughout the duration of the analysis
- Industry: 10 sites per year at $100,000 (i.e. $1 million per year) throughout the analysis period.

Prohibitive standards

- Consenting costs for schools and hospital incinerators: a first year, once only cost of $1.724 million
- Alternatives to road seal burning: $1 million per year throughout the analysis period, based on advice received from Transit New Zealand
- Zero costs recorded against applying standards to landfill fires, disposal of hazardous waste to landfill, burning oil and copper wire burning.

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[3] It is well established that computable general equilibrium models which allow for price changes and resulting input substitutions produce smaller flow-on effects and economic multipliers than input output models that make no such allowance. See for instance Dwyer L & Forsyth L (2006) *International Handbook of the Economics of Tourism*, Edward Elgar Publishers, Cheltenham UK

Wood burner standard

- Costs to government and councils of $500 per household per year, amounting to $700,000 in the first year, then $5 million per year for 2005 to 2014, and zero thereafter
- Costs to households: zero costs recorded against this item
- Costs to wood burner suppliers: zero costs recorded against this item.

Whether these costs are reasonable and how much they need to be revised for the updated analysis is examined below.

3.1.1 Ambient air quality standard

A national environmental standard automatically supersedes any existing regional plan so it does not require an explicit plan change, with its associated transaction costs of plan revision and consultation. However, some regions may decide they need to revise or devise air quality action plans in order to achieve the ambient air quality standard and the 2013 target. The standard may also require regions to increase their monitoring of ambient air quality, although only in areas where there is reason to suspect problematic high concentrations may occur. It is reasonable to expect there to be some costs in meeting this standard, although not necessarily widespread or high in all localities.

The council and government costs are similar to those used in analyses of other national environmental standards (on telecommunications facilities, electricity transmission), on the basis of cost per agency. However, although 14 out of 16 regional authorities have airsheds that are unlikely to comply with the NES, not all territorial authorities are in that position. Costs for territorial authorities account for 29% of total costs in the 2004 analysis, so any overstatement would also have an effect on the total costs of meeting the standard.

A report on progress on national environmental standards for air quality identifies locations in 32 separate territorial authorities as likely to exceed the ambient standards (MfE 2008, Table 5), so projecting a standard cost across all 70 territorial authorities may overstate the territorial authority cost in the analysis. In any case, territorial authorities have no direct responsibility for air quality standards, beyond aligning their functions such as building inspections with regional rule requirements. Other information from the Ministry indicates that many regional councils have spent very little on implementing the standard to date, and territorial authorities even less, so the cost assumptions seem on the high side relative to actual experience. However, the existing council costs may be incomplete and insufficient for achieving the air quality targets by their specified dates, so providing for some additional regional council cost over those incurred to date is appropriate.

The 2004 CBA states that a MfE survey of industry indicated costs could range from a few thousand dollars to $100 million, but a “best guess” average of $100,000 per site upgrade was chosen to estimate a rolling upgrade programme of 10 sites per year. There is no way to verify the validity of this assumption without lengthy discourse with the variety of industries concerned, but the preliminary progress report does not identify major problems with compliance among industry (MfE 2008). The effect of the assumption on results can be tested through sensitivity analysis, but as in the central analysis this item accounts for 8% of total costs, it would require a significant increase in average cost to have an appreciable effect on the result.

The 2004 CBA does not include the potential for costs to industry that may occur if a firm is denied discharge consents because the region's emissions exceed its transitional path to the 2013 target level. This is of concern to some regions where inability to consent is equated to loss of business and jobs in the region. The net cost of this is tempered by the uncertainty around whether this constraint of activity will occur, and also, if it does, by the extent to which resources will be available to redeploy on other activities that are not subject to the standard. From a national perspective, inability to obtain consent in one location may simply result in business relocating to regions where it can obtain consents. The business that is relocated is a transfer effect of no consequence to a cost benefit analysis, but there is likely to be some deadweight loss of resource use efficiency in the process of relocation, or if businesses make investments to lower their rate of emissions, and in the possibility that some business may be choked off altogether because it faces prohibitive costs (e.g. expansion at an existing site in a region applying restraints on consenting). The scale of this effect depends on the effectiveness of regions in addressing the NES target and is difficult to estimate. This part of the NES does create an anomaly in that the consented activities subject to this control are not necessarily those that contribute most to excessive emissions that trigger the control.

3.1.2 Prohibitive standards

The 2004 CBA Report notes that costs associated with prohibitive activities reflect that many of these activities are already deemed unacceptable by regional rules, occur at a very low level if at all, and have substitute technologies available, so the cost of many of the proposed bans is zero or minimal. Those that are not relate to the consenting costs for incinerators in schools and hospitals, and for alternatives to tar seal burning on roads. The school and hospital consenting costs amount to 1% of total costs, and road seal costs to 8% of total costs in the initial CBA, so it would require a significant increase in these cost assumptions to have an appreciable effect on the result.

3.1.3 The wood burner standard

There is a discrepancy between the 2004 CBA report's Appendix 3 table (which is drawn from the spreadsheet behind Table 2 above) and its Table 12. Table 12 suggests the wood burner standard involves a cost for households of between $0 and $200 per burner, being the increased expense of low-emission burners.[4] It also

[4] This is an estimate of the modifications that would need to be made to current models in order to achieve the standard, which it is assumed manufacturers would pass on to consumers in prices. It could also represent the difference in price between compliant and non-compliant models that are otherwise equivalent, but there are few non-compliant models sold since the introduction of the NES.

suggests costs to government of between $500 and $2,000 per burner, reflecting subsidy and assistance schemes developed by MfE, EECA and the Climate Change Office to eliminate the worst wood burners and open fires amounting to $5 million per year. The 2004 CBA report's summary of costs and benefits (its Table 13) agrees with the spreadsheet which has zero value for costs to householders and wood burner manufacturers from the wood burner standard. But its description of costs associated with home heating (page 44) does not accord with the standards that are currently in place, and can be treated as superseded for the update of the CBA.

The air quality NES for wood burners only requires new burners installed or replacing old models should meet the standard, and as such does not directly affect the existing stock of old installed burners. However, regional councils may consider applying more stringent and widespread rules for wood burners in pursuit of their ambient air quality targets, as already occurs in Nelson and Canterbury. The spreadsheet's assumption of government/council costs of $500 per household amounting to $50.7 million in total over the 17 year analysis implies 101,400 households covered by this item. The 2006 Census recorded 1.47 million dwellings (occupied and unoccupied), of which 39% (0.574 million) recorded use of wood fuel. Although not all of these will have wood burners subject to the NES, this level of government/ council subsidy, indicates a low level of penetration of the wood burning households that may be far from the total cost of compliance with the standard. There is no subsidy specifically aimed at meeting the air quality NES, so costs on wood burner compliance are likely to fall predominantly as private costs on householders.[5] The private cost to households is an omission that ought to be included in a national cost benefit analysis. Conversely, the subsidy envisaged in the 2004 CBA was never implemented (although other subsidies with similar effect have been applied by EECA for purposes other than NES compliance), so this item does not belong in the updated analysis.

Another potential private cost arises for suppliers of wood burners, who may be faced with costs of redesign and retooling their burners to comply, and those (including importers) who may be left with stocks of non-compliant burners after introduction of the standards. Redesign and retooling are costs to manufacturers/suppliers only to the extent they cannot be passed on to consumers in product prices. Discussion with manufacturers reveals mixed views on their importance: some regarded them as one-off costs of no significance for further implementation of the NES, while others have suggested there can be significant and recurring costs in testing and sometimes retesting each new model to demonstrate it meets the standard. There were suggestions of one-off retooling costs of $10,000-$15,000 to make a NES-compliant version of a non-compliant predecessor, and further costs of $40,000-$50,000 per model to get through the production and compliance-testing process. Such costs vary

[5] EECA's Energywise Clean Heat Programme commenced in fiscal year 2007/08, offering $1.3 million per year for four years to assist in replacement of open fires and old wood-burners in up to 800 houses per year in airsheds with poor air quality. Although related to air quality standards it has other objectives and did not feature in the 2004 CBA. As an existing scheme, it is not an additional cost in the forward looking CBA update: houses converted under the scheme are removed from the total needing conversion to achieve the target ambient standards.

with the size and style of the model, and while manufacturers would seek to recover them through their pricing, in some circumstances they may not do so in full.

Manufacturers report that particularly with in-built burners and those with wetbacks it can be difficult to simultaneously achieve the emission level and the 65% heating efficiency specified by the wood burner standard, as energy used in heating water or the burner surrounds is not counted for the purposes of the performance test,[6] An implication is that manufacturers report releasing fewer in-built and wet-back models onto the market since the standard was introduced, potentially slowing down the rate at which less efficient older models and open fires are replaced.

Where a model incurs repeated testing costs but has low sales, those testing costs may be unrecoverable in full from sales to consumers and borne by suppliers. As the NES only specifies standards for use in urban areas and allows non-compliant burners to still be fitted in situations such as rural areas outside its jurisdiction, suppliers should be able to offload non-compliant burners while gradually adjusting their mix to the new market conditions. So private costs to suppliers from both these sources are likely to be minimal.

a) The possible effect of omitting private costs

In principle, private costs of compliance with the standard for households should be included in the cost benefit analysis – the question is, does its omission make a material difference to the analysis, and what should those costs be? The additional cost imposed on householders by the standard varies with circumstances:

- For a new wood burner being installed into a house, the additional cost imposed by the standard is the difference in cost between a compliant and non-compliant alternative burner that might otherwise be used. Other costs, such as for the flue, hearth and installation, will be the same in either case, so the incremental cost of the standard is only the difference in burner costs.
- For an existing wood burner that is being replaced:
  - If the burner is at the end of its useful life and would need to be replaced in any case, the incremental cost is the same as for a new wood burner, i.e. just the difference between the compliant burner and an equivalent non-compliant one.
  - If the burner has some useful life left in it and can be modified to meet the new standard, the incremental cost is just the cost of modification and hence likely to be small.
  - If the burner has some useful life left in it and has to be replaced (e.g. because of a regional council rule requiring upgrading to compliant burners when a house is sold) the incremental cost of the regulation is the full cost of the item

[6] There may be ways of modifying the testing procedure so that burners that achieve low emission levels are not deemed non-compliant for just failing to meet the energy efficiency level. For instance, a test based on the ratio of emission level and heat output would enable a burner with very low emissions but less than 65% energy efficiency to be sold as compliant, and hence available to replace much smokier and less energy efficient older burners and open fires.

being replaced, i.e. the cost of the new burner plus its installation, if it cannot be used somewhere else.

– If the old burner can be on-sold to a situation where it is not covered by the standard (e.g. in rural properties of greater than 2 hectares), the value realised from that sale can be offset against the cost of the new compliant burner.

The Consumers Institute website lists a range of compliant wood burners whose prices vary from $1,150 to $3,550, depending on size, output and style (free standing or wall insert). The mean price is $2,341 and the median is $2,470, but there is no information on sales across the size range or what a weighted mean might be or how it compares with new non-compliant burners. There are second hand stoves with a reserve price of around $550 listed on Trade Me, but these listings have insufficient information to determine whether stoves are compliant or not and provide a realistic alternative to those currently available in New Zealand.

Discussions with manufacturers suggest the additional cost of compliant models compared with equivalent sized non-compliant models varies with the size of the model: for small to medium models that constitute the bulk of wood burner sales it could be in the range of $125-$250 per burner, but for the largest burners it could be higher - $300-$500 per burner – reflecting greater costs in development, testing and smaller sales volumes. The upper end of the small-medium range would be close to the figure cited in the 2004 CBA, updated to today's prices.

Installation costs are often bundled up with the price of a new burner in a package, and will vary with the conditions of the house, ease of access, and the extent of structural modification required. From enquiries made with retailers, however, the cost of installation alone (i.e. excluding the cost of the burner, flue, and heat-resistant hearth surrounds) is typically in the region of $500-$600 per stove.

On this information, the incremental cost of the NES on new wood burners of $200 as identified in the 2004 CBA report, or a little higher today, is not an unreasonable assumption. The cost of replacing burners that still have useful life left in them would be substantially higher: if the average cost of a burner is $2,400 and cost of installation is $600, the average cost of replacements would be $3,000 per burner, less whatever is recovered from on-selling the old burner, which on average is likely to be less than the value of the subset of burners seen on the second hand market.

b) How critical is the omission of private wood burner costs?

How rectifying this omission would change the initial CBA can be illustrated with reference to Statistics New Zealand's data on building permits and the 2006 Census. The Census shows 0.57 million (39% of the 1.47 million national total) occupied dwellings recorded the use of wood fuel. Not all of these will have wood burners subject to the NES, but a worst case would be to assume that they all do. Annual building permits fluctuate year to year with changes in economic conditions, but the average of new dwellings (excluding apartments which rarely have wood fires) over the years ending March 2004-2009 was 21,909 new dwellings each year (a figure likely to be rather inflated by the housing boom). If these new dwellings install wood stoves at the average rate of the existing housing stock (39%), then there would be at most 8,554 installed in the first year. If the additional cost imposed by NES compliance is $260 per burner excluding GST (derived from the $200 in the initial analysis updated for inflation), the annual cost would be about $2.3 million per year. Projecting that to 2020 the total cost would be around $34 million, with a present value of just under $19.5 million when discounted at the Treasury's current default rate of 8% in real terms.[7] If it is assumed that new dwellings increase at a compound annual growth rate of 1.5% per year (in line with the economic growth assumption in the original 2004 CBA), the cumulative cost to 2020 would rise to about $38 million with a present value of $21 million.

These estimates suggest that even with relatively strong assumptions about the effects of the NES on wood burner costs, this item would have been unlikely to much reduce the net present value in the 2004 CBA ($318 million). The aggregate costs are likely to be much smaller than these estimates suggest because of the range of wood fuelled burners recorded in the census that are outside the scope of the NES controls – including open fires, pellet burners, multi-fuel stoves and all wood stoves outside of urban areas. Also, the number of dwellings that use wood for fuel has been successively declining in recent censuses, which has a bearing on the counter-factual of the situation without the NES, and would lower both the costs and the benefits attributable to the wood burner standard.

c) The potential effect of more stringent regulations

A bigger effect would come if regional councils applied more stringent controls to wood burners in pursuit of meeting their regional airshed targets. For instance, Hawkes Bay Regional Council proposes banning in Napier and Hastings all types of wood fuelled space heating other than those complying with a more stringent emissions standard than that in the NES (particulate emissions of 0.7 gm/kg dry matter rather than 1.5 gm/kg in the NES).[8] This will be implemented through a ban of non-compliant appliances at time of sale of houses, and other inspection measures. The regional council estimates the cost of total wood burner replacement in Hawke's Bay is likely to be at least $40 million.[9]

If all regions across New Zealand took a similar approach to upgrade the 39% of the existing housing stock that uses wood fuel, the societal costs would be very large and could eclipse the net benefits in the initial CBA. Assuming 1/7 of the housing stock changes hands each year (as 7 years is the average length of time that

[7] This has replaced 10% as the Treasury's preferred default social discount rate for cost benefit analysis, following a paper released in July 2008 (Public Sector Discount Rates for Cost Benefit Analysis). If the 10% rate is used as in the initial CBA, all these present values would be smaller.

[8] The more stringent emissions standard being considered in Hawke's Bay and already used in Nelson and Canterbury allow for the fact that in practice burners may not achieve the low levels of emissions that they do in test conditions. A reduced number of models on the MfE's website list of compliant burners have achieved test results at these lower emission levels.

[9] This is the council's rough estimate based on the total number of burners in Napier and Hastings (8,800 + 10,900) times $2,000 (approximate cost of a burner). See Council Agenda Papers – Background Information, Environmental Management Committee 17 September 2008 Agenda Item 11 on www.hbrc.govt.nz/WhatWeDo/CleanHeat

properties turn over in New Zealand), then 81,997 dwellings a year from 2006 to 2012 would need to upgrade their wood burners. If the cost of replacing and installing wood burners is $3,000 per burner, the aggregate cost of complete upgrade over 7 years would be around $1,720 million, with a present value discounted at 8% of $1,280 million.

Such large costs would certainly be a case of overkill, because such an approach would impose costs on properties in locations where the upgrade makes no appreciable difference to achieving the NES target. Moreover, faced with such a substantial incremental cost, some property owners may choose not to upgrade their wood burners but rather replace them with some other heating with lower capital cost. That would aid the achievement of the air quality NES target, but would have contrary economic effects, because these alternatives with lower capital costs commonly have higher operational costs. If electricity is the preferred alternative there could also be a contrary environmental outcome, in that increased demand for electricity would raise the use of high cost generation at the margin: this is often thermal-fired generation with greenhouse gas emissions, which incur a resource cost under the Kyoto Protocol or any other similar international climate change agreement that succeeds it. However, this is unlikely to be significant as it would be only a short term effect, as such increased demand would also bring forward the commissioning of new generation capacity, much of which is for renewable plant.[10]

To the extent that the NES does prompt such substitution to other forms of space heating, the extra running cost relative to a wood burner is an on-going resource cost that ought to be reflected in the cost benefit analysis of the NES. For example, EECA estimates the running costs per kWh of heat of a wood burner to be around 4 cents, compared with 6-7 cents for heat pumps, 11 cents for pellet burners, 16 cents for natural gas or diesel heaters, and around 21-22 cents for plug-in electric and bottled gas heaters.[11] Those options with highest running costs generally have lower capital costs than wood burners, but would still represent an additional cost (albeit less than the cost of replacing a wood burner) attributable to the standard if they resulted in replacement of a still functional burner. There are qualitative differences in the heat provided by these different options (e.g. the mix of radiant and convection, and the distribution throughout the house), and also differences in the labour time involved in running solid fuel and electric or gas heaters. Detailed modelling of the likely extent of such substitution is beyond the scope of this current CBA update, but some allowance needs to be made for the consequences of energy substitution.

The compulsory upgrading of wood burners by retrofitting existing properties is likely to be a high cost exercise. Its effectiveness is also dependent on the regional council's enforcement capability. If regional councils choose such an approach it implies it is the least cost means of reducing ambient levels of PM_{10}, but in many

[10] Much prospective generation capacity is for wind or geothermal plant, neither of which are particularly reliable for responding to short term demand peaks. However, using more of such generation when it is available reduces some use of hydro generation and enables more hydro storage to meet short term peaks, reducing the use of thermal plant for that purpose.

[11] EECA Energywise Action Sheet 5, www.energywise.govt.nz

areas it is unlikely to be the least cost option (as in the Auckland metropolitan area, where changes in national fuel specifications and vehicle emissions are expected to have a significant impact). The corollary is that if retrofitting is used, it would be most effective if targeted on those locations where it will contribute to airshed PM_{10} targets. Apart from Hawke's Bay which is considering retrofitting a wider range of wood fuelled appliances than wood burners, Nelson and parts of Canterbury and Otago already have stringent provisions, with similar effect, but over many regions it is unlikely to be effective or efficient to implement. Costs would also be substantially reduced if regulations can be tailored to only incur the incremental cost of modifying existing burners or installing compliant burners when new, rather than an approach that requires replacement of existing stock. A cost benefit analysis should consider this approach.

Possible complications

The costs and benefits presented in this report are based on replacement of non-complying wood burners. However, if a non-compliant existing wood burner is not allowed to be used and the user is financially unable to install a new wood burner and this results in inadequate heating, there can be adverse consequence on health so that the benefit is actually reduced.[12] The additional cost of enforcement of this nature is also not included in our estimates. An occurrence of such a situation means lower benefit and higher costs. It is not clear to what extent such a possibility exists.

3.2 Review of evidence on the basis of benefits claimed

Benefits are estimated as the avoided societal cost arising from improvement in health impact of pollution. Previous studies (Fisher et al 2002, Wilton 2003, Fisher et al 2007) estimate the impact on mortality, hospitalisation and restricted activity days (RADs), following Künzli et al (2000). These impacts are based on the level of annual average value of 24 hour maximum PM_{10} level. The method used in these studies is explained by Fisher et al (2007) as follows.

Suppose

Pe = crude mortality rate per 1000 in the age group 30 and above

Po = baseline mortality rate per 1000 in the age group 30 and above after deducting the air pollution effects.

[12] It has been estimated that 10-14% of households in New Zealand are in a state of "fuel poverty", in that they need to spend more than 10% of their income on home heating to achieve a healthy home environment, based on meeting WHO guidelines of 21oC in living areas and 18oC in bedrooms (Lloyd 2006). If that were so, removing access to what in many regions is the cheapest form of heating fuel (firewood) could result in a reduction of heating quality in the home. However, the WHO guidelines are substantially warmer (and entail higher energy cost) than actual behaviour observed in New Zealand households, where the average heating in living areas is believed to be around 16-18^{o}C. The WHO guidelines require more energy on space heating in most regions than the average household's total electricity use, so meeting them would inevitably have a major impact on household expenditure and residential energy demand. Based on New Zealanders' current heating behaviour, about 5% of households spend more than 10% of income on household heating.

E = PM_{10} exposure level in the area of interest

B = threshold PM_{10} exposure level for mortality effect

RR = epidemiologically derived relative risk for a 10 $\mu g/m^3$ increment of PM_{10}.

D10 = number of additional deaths per 1000 population in the age group of 30 and above for a 10 $\mu g/m^3$ increment of PM_{10}.

Then

$$Po = \frac{Pe}{1 + [(RR - 1)(E - B)/10]}$$

The increased mortality per 1000 population in the age group of 30 and above is estimated as

D10 = Po (RR-1).

Number of deaths due to $PM_{10}$is estimated as

N = D10*P*(X-B)/10, where P is the population in thousand, and X = PM_{10} exposure level.

The estimates obtained by Wilton (2003) based on this procedure were used in the 2004 CBA.

While the methodology has international provenance and appears appropriately applied to New Zealand data, there remain some ambiguities about the comprehensiveness of their measure of benefits. The estimates include only deaths of people in the age group of 30 years and over. It is not clear to what extent the number of deaths due to chronic diseases are included in this estimate. Künzli et al (2000) note that they have included such cases. However, these are not obvious to identify and hence are likely to be under estimated. Another factor to be considered is loss of life quality of those who suffer from pollution effects, for example, asthma sufferers those who have heart attacks but survive or those who contract cancer which is successfully treatment. In all these cases, those with pollution-exacerbated ailments experience loss of life quality but that has not been included in these estimates. Taking these factors together, the 2004 CBA benefits are likely to be underestimated.

3.2.1 Assumptions made for estimating the PM_{10} level

Wilton (2003) provided the estimates of the annual average value of 24 hour maximum PM_{10} level based on several general assumptions noted below.

- "A 45% decrease in the number of multifuel burners from 2001- 2021 in areas where these are not legislated.

- A 10% decrease in open fires from 2001-2021 in areas where these are not legislated.
- Other solid fuel burners are replaced with new solid fuel burners 15 years from the date of installation.
- A linear relationship between emissions and concentrations for all areas i.e., any reduction in emission would result in a proportional reduction in concentrations.
- No impact of differences in the time of day of different sources relative to meteorological conditions, except in Christchurch. In Christchurch, a box model was developed by the National Institute of Water and Atmospheric Research to describe this relationship (Gimson & Fisher, 1997).
- An aging population, with the proportion of the population over 30 increasing in each area by 20% of the 2001 proportion by 2021. These are estimates based on limited data provided by Statistics New Zealand.
- A 10% increase in industrial emissions in all areas except those with negative population projections.
- A 70% decrease in PM_{10} emissions from motor vehicles from 2001-2021 in areas where area specific modelling and projections have not been carried out. The latter is based on NZTER emission rate projections, allowing for some increased traffic growth and congestion. Assuming NZTER estimates of emission rates are accurate this should be a conservative (underestimate) of the reductions for most areas".

Vehicle emissions, according to Fisher et al (2002), account for about 41% of total PM_{10} emissions. A large proportion of vehicle emissions occur in Auckland (64%), followed by Wellington (14%), Christchurch (10%), with the rest of the country accounting for only 12%.

The estimates of mortality refer to the population of 30 years of age or over. It is assumed in these estimates that the mortality of younger people is not affected. The RADs also refer to the same population (i.e over 30 years old). However, hospitalisation refers to those under 30 years old as well.

As with all such studies there is an issue in determining the counter-factual of emission levels in the absence of the current standard and its track – e.g. allowing for secular trends such as decline of wood-burner numbers and increase in traffic emitting PM_{10}. The linkages between emission levels and health incidence in the 2004 analysis are opaque and drawn from an external model (Wilton 2003) that is now outdated according to its author, but would require substantial resources and time to update.

The discount rate in the 2004 CBA was Treasury's then-default rate of 10%, but this is now 8%.

The differences in assumptions and coverage between the 2004 CBA and the current update are summarised in Table 16 in Appendix A .

4. Updating the analysis and scenario analysis

The section above highlights a number of omissions from and shortcomings of the 2004 CBA. We now outline how we have attempted to address these issues, and present the results of our updated CBA of the NES.

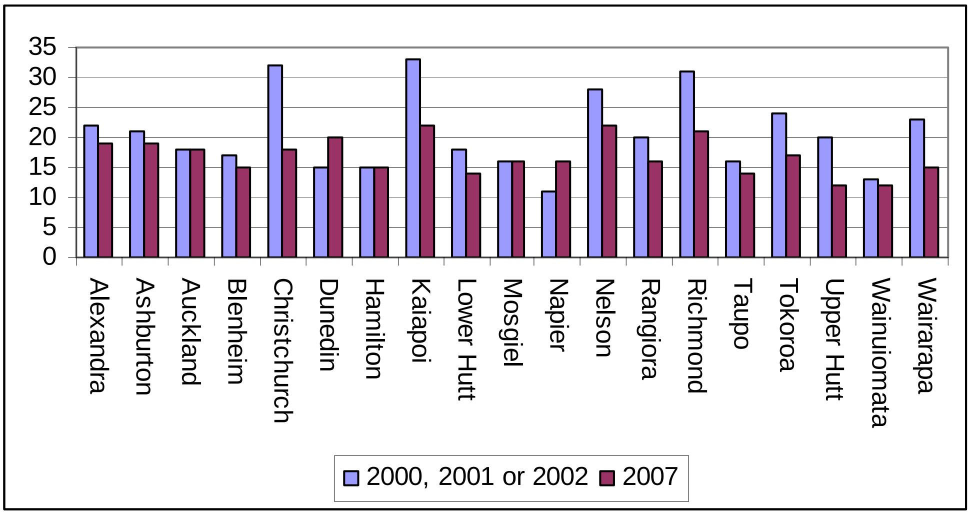
4.1 Context and caveats

The 2004 CBA, and this updated analysis, are limited by the incompleteness and unevenness in the availability of data. Given such gaps in availability, no CBA can claim to be precise, least of all one updated over a tight timeframe on the basis of existing gathered material. Such analysis can however aim to identify the main effects of the proposed policy, define the range of uncertainty and identify where are the main unknowns that could change the result.

The 2004 analysis was based on estimates of maximum and annual (24 hour) average PM_{10} levels. Currently, some airsheds record the 24 hour maximum over a year, or the second highest value. In some cases they record the average of 24 hour maximum values. The estimates of effects in terms of mortality, hospitalisation and RADs are based on the annual average values. The latest annual values are available for 2007. The 2004 estimates of PM_{10} levels which formed the basis for earlier estimates of effects were mostly for year 2001. In some cases these were for the year 2000 or 2002. In one case (Lower Hutt) it was for 1998.

The average maximum value of 24 hour PM_{10} varies from year to year. However, as can be seen in Figure 1, the level has been lower in 2007 in most places. The value is either the same or lower in 2007 in 17 out of 19 areas for which data are available for both periods. This suggests that the current level of PM_{10} pollution is on average lower than it was in 2002 or earlier. It is not clear to what extent this is due to the air quality standard introduced in 2004. On average the 2007 averages are 18% lower than the 2004 averages for these 19 areas for which data are available for 2007 and before the standards were introduced.

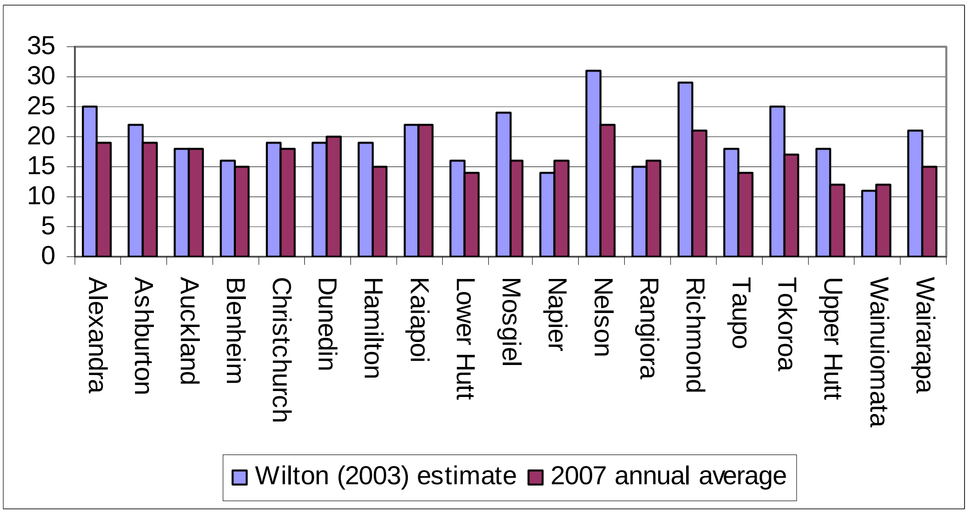
Figure 1 Annual average PM10 in base year and 2007



Source: NZIER

The observed annual averages in 2007 are lower than what was estimated earlier by Wilton (2003) in most cases, as shown in Figure 2. As we understand it, Wilton's estimates were the basis for evaluation of the proposals in the 2004 CBA.

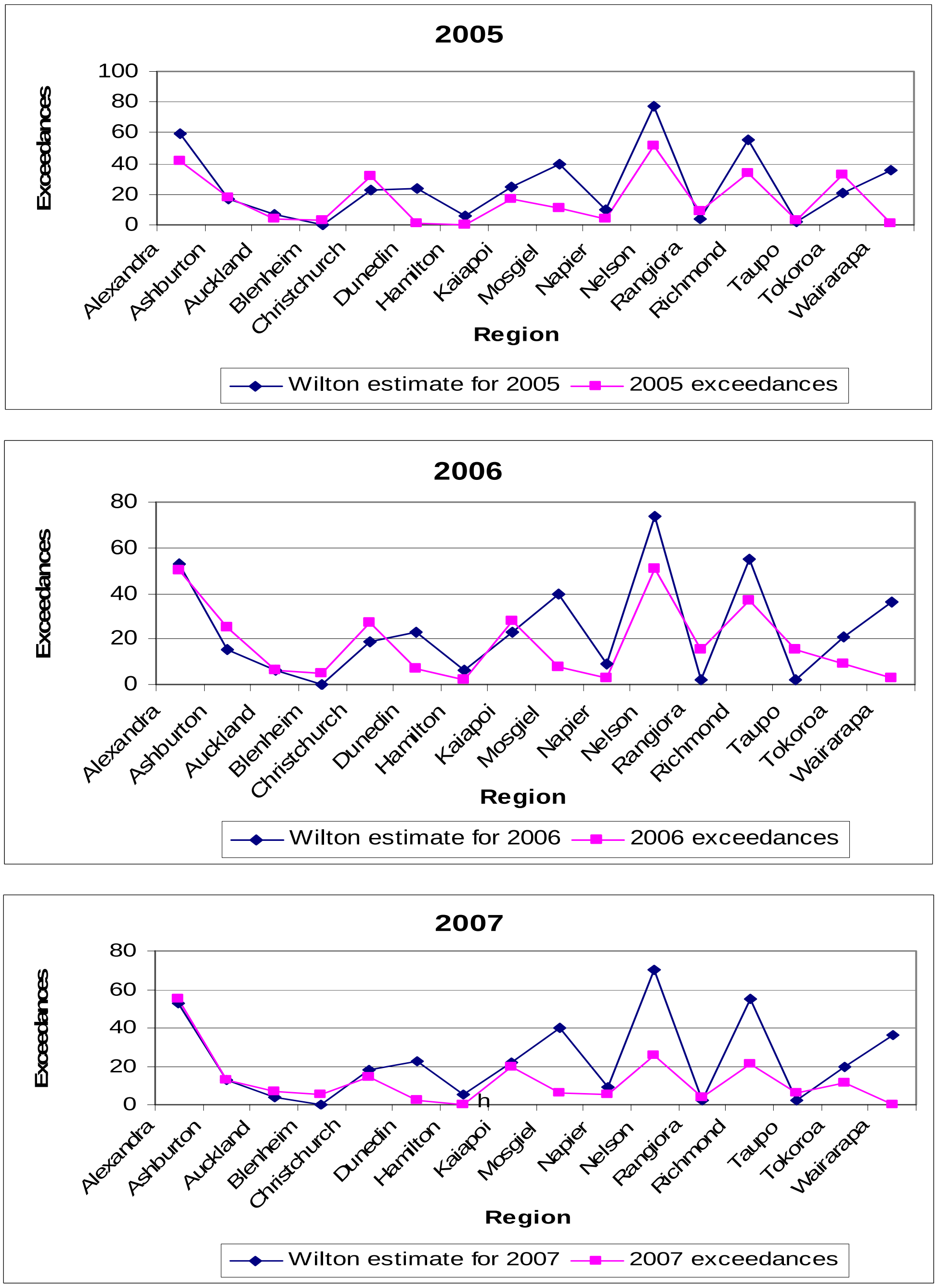
Figure 2 Comparison of annual average PM10 24 hour maximum values



Source: NZIER

The effect of random variation and also variation in weather cannot be ignored. However, overall, for these areas, the 2007 actuals are about 16% lower than the Wilton (2003) estimates for 2007. Unless the weather pattern in 2007 was unusual in favour of PM_{10} exposures, the introduction of standards along with other factors might have contributed to greater than expected improvement in the level of PM_{10} in the environment between 2003 and 2007.

Figure 3 A comparison of actual exceedances with Wilton (2003) estimates



Source: NZIER

In order to see if 2007 was a special case or whether there is a pattern of the actual levels of PM_{10} being lower than those estimated by Wilton (2003), we looked at the number of exceedances for three years 2005-7 and compared them with the Wilton (2003) estimates. The comparison shows (Figure 3) that for a few places the observed number of exceedances is consistently lower than the Wilton estimates, while in other places the differences are small. This suggests that the observed average annual PM_{10} values are are on average lower than the previous estimates.

4.2 The counterfactual

The estimates made in 2003 of the likely impact of PM_{10} pollution on health, which formed the basis for the 2004 CBA, were based on observations prior to 2003. While there are limited available data, these indicate that the average PM_{10} level in 2007 is about 16% lower than the estimates made in 2004.

There can be two explanations for this difference. It could be due to the air quality standard introduced in 2004 or it could be that the status quo estimates were on the high side.

Another factor that is expected to affect the status quo situation in the future is the emphasis and incentives for home insulation, which if implemented would require less heating and hence less emission of PM_{10}.

From the available data it is not possible at this stage to precisely estimate the likely impact of the insulation initiatives. The Künzli (2000) method followed by Wilton (2003) and Fisher et al (2007) indicate that on average the mortality number would be about 8% lower in 2007 if the latest value was used instead of the Wilton (2003) estimates of PM_{10} levels, for areas included in Figure 3 above[13]. Under the standards, the 2004 CBA estimates show no difference in the number of total mortalities in 2007 from that under the status quo situation. It is possible that the introduction of air quality standards effective from 2005 and the guidance to local councils on ensuring that the maximum PM_{10} level is within the straight line path to the 2013 target has produced better than expected results. If that is the case, any relaxation of the standards now could increase the level of PM_{10}. On the other hand, the awareness of health damage might have influenced replacement of wood burners with more efficient burners or other heating systems. In that case, there is a possibility of continuation of this behaviour.

Ideally, analysis of trends in home heating installations over time in terms of heating efficiency and use of less polluting systems would indicate changes in fuel combustion, emissions and pollution. Based on the limited availability of information

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[13] Following Künzli et al (2000), Wilton (2003) appear to have estimated the numbers of mortalities due to pollution effect as $N = \frac{Po(RR-1)Pc(X-B)}{10}$, (as shown in Fisher et al (2007)), where Po = baseline mortality rate per 1000 in the age group 30 and above after deducting the air pollution effects, X = PM10 exposure level in the area of interest, B = threshold PM10 exposure level for mortality effect and RR = epidemiologically derived relative risk for a 10 µg/m3 increment of PM10. From Wilton (2003) estimates we determine the value of $\frac{Po(RR-1)Pc}{10}$ as $\frac{N}{X-B}$, where X is the projected value of annual average maximum PM10 estimated by Wilton (2003). We then use the observed annual average maximum PM10 and estimate the number of mortalities in 2007.

and time constraint, we have only made some rough estimates of the likely impact, discussed later.

We also find that there was a downward trend in the use of wood burners in recent years which, it seems, was not taken into account in Wilton (2003) estimates. The lower PM_{10} levels observed could be partly due to this trend. Also, the effect of the regulation might have already started materialising. Taking these into account and given the uncertainties, we have not changed the status quo estimates based on the recent observations (Table 3). These are the estimated total health effects of levels of pollution reported in Ministry for the Environment (2004), without any allowance for the effects of the ambient air quality standards introduced in 2004, trends in wood burners or changes in insulation policies. We discuss the impact of an 8% reduction in status quo in the sensitivity analysis.

Table 3 Estimated health effects associated with air pollution

Year	Premature death (adults only)	Hospitalisations	Restricted activity days
2007	736	561	2,271,770
2008	711	552	2,229,742
2009	702	550	2,220,958
2010	692	547	2,212,339
2011	688	548	2,212,403
2012	685	549	2,214,841
2013	681	549	2,219,253
2014	677	550	2,222,580
2015	678	553	2,232,999
2016	679	555	2,242,020
2017	680	559	2,252,983
2018	681	561	2,262,999
2019	686	566	2,281,010
2020	689	570	2,296,595

Source: Ministry for the Environment (2004)

4.2.1 Infant mortality

The effects of PM_{10} in the 2004 CBA have been confined to three areas: mortality and hospitalisation due to short time of exposure and restricted activity days for adults 30 years of age and over. Studies (Röösli et al 2005, Glinianaia et al 2004, Woodruff et al 1997, Barnett et al 2005, Canadian Medical Association 2008) indicate mortalities occur also to infants and also that the child could be affected before birth. In addition studies (Jalaludin et al 2004, Barnett et al 2005) show that PM_{10} affects the prevalence of asthma and other respiratory problems in children requiring more visits to the doctor. This is in addition to hospitalisation figures used in the 2004 CBA. Also, some of these may require hospitalisation at a later stage of life and are not included in the number of hospitalisation cases in the 2004 CBA. Many of these cases suffer loss of life quality, which should be included as part of the social cost of PM_{10} health effects.

A South Korean study shows that for infants (babies between 1 month and 1 year old) mortality increased by 102% for each 43 microgram per m^3 increase in PM_{10}. For other age groups it was lower: 6.6% for 2-64 yrs and 6.3% for over 65 (Ha et al 2003). A US study (Woodruff et al 1997) shows that postneonatal mortality increased by 4% for every 10 $\mu g/m^3$ increase in PM_{10}, for normal birth weight the increase was 12% (Air Resources Board 2004).

Glinianaia et al (2004) review a few research papers on the effects of pollution, particularly PM_{10} on infant deaths. These studies show varying results, ranging between no relationship to strong relationships between neo-natal mortality and the PM_{10} level. Stronger relationships were found for post-neonatal mortality. In both cases studies found a stronger association between deaths due to respiratory causes and the PM_{10} level. Considering these and Woodruff et al (1997) study, we use a conservative estimate of 4% of post-neonatal mortality for every 10 $\mu g/m^3$ increase in PM_{10}.

Röösli et al (2005) develop a dynamic model to estimate the loss of life years due to mortality. As in other studies, this study also confines its analysis to the age group of 30 years and above for adults and up to one year for infants. Carrying out a meta analysis of a few studies, they estimate the relative risk of mortality as 1.059 per average exposure of 10 $\mu g/m^3$ increase in PM_{10}. For infants the relative risk is estimated as 1.056. Their estimates indicate similar health effects on adults and infants. While the effects of a PM_{10} increase on these two groups i.e., infants and above 30 years of age are similar, they find relatively low risk for those aged 5 years to 30 years. Finding no other evidence of impact on the 1-30 year age group, we also confine our analysis to infants up to one year old and adults of age 30 years and above.

Röösli et al (2005) estimate the impacts over time of a sudden decrease in average exposure of PM_{10}. Following reduction of 10 $\mu g/m^3$ in the average PM_{10} exposure in a year and returning the level to the before intervention level, Röösli et al (2005) estimate that 39% of the impact occurred in the first year and about 63% in the first two years. This suggests that the estimate of mortality used in the 2004 CBA as the immediate impact of high levels of PM_{10} is likely to be on the lower side. However, we have no data at present to estimate the ongoing effects.

The total of fetal and infant (less than one year old) mortality in 2003 was 697. There were a total of 56,969 births during the year.[14] While the published research indicates

[14] Statistics New Zealand website: http://www.nzhis.govt.nz/moh.nsf/pagesns/394/$File/AF&IStatstables2003_1.1(edit).xls

that PM_{10} exposure adversely affects unborn babies, the quantum of risk increase is not clear. For infants, estimates vary between 4% and 5.6% increase in mortality for every 10 µg increase in PM_{10} level. We have used a conservative estimate of 4% for infant and fetal mortalities.

To estimate the number of fetal and infant deaths due to PM_{10}, we followed two separate approaches. We used the average level (annual average of 24 hour maximum values) of PM_{10} for the country. Secondly in the absence of number of births in each council area, we simply distributed the total number in proportion to population. This is a crude way of estimating the number of births since the age distribution of the population varies between regions. Both approaches provided the estimates of total number of deaths within a narrow range. This indicates that in 2003, the total number of fetal and infant deaths due to PM_{10} exposure would be about 24.[15] The estimate of adult (>30 years of age) mortality due to this exposure was 872 by Wilton (2003). Thus the number of infant deaths in 2003 was about 2.7% of adult deaths. Canadian Medical Association (2008) also notes that children and infants with some health conditions have higher risk of death, however the number is relatively small.

The revised estimates, assuming the number of fetal and infant deaths would be 2.7% of the number of adult deaths, are shown in Table 4. They are simply 1.027 times the deaths reported in the 2004 CBA in Table 3.

Table 4 Estimates including infant mortality

Year	Premature death	Hospitalisations	Restricted activity days
2007	756	561	2,271,770
2008	731	552	2,229,742
2009	721	550	2,220,958
2010	711	547	2,212,339
2011	707	548	2,212,403
2012	703	549	2,214,841
2013	700	549	2,219,253
2014	696	550	2,222,580
2015	697	553	2,232,999
2016	697	555	2,242,020
2017	699	559	2,252,983
2018	700	561	2,262,999
2019	704	566	2,281,010
2020	708	570	2,296,595

Source: NZIER

[15] If PM_{10} increased the number of infant deaths by 5.6% for every 10 µg increase in PM_{10} level, then the total number of deaths would be 33. We have used 24 as a conservative estimate.

We do not adjust the number of hospitalisations from the 2004 CBA as that is expected to have included children and infants as well.

4.2.2 Loss of life quality

The estimates of mortality after taking into account fetal and infant deaths are still likely to be underestimates due to the exclusion of loss of life quality estimates where the effect has been mainly suffering over time and not counted as mortality of the effect of pollution in the above estimates. To give an example, a child or even an adult may suffer from asthma. This does not necessarily shorten the life span or even if it does, it is unlikely to have been included in the mortality estimates. In that case, the loss of quality adjusted life years should be included in the total social cost. The same is also true for cardiopulmonary diseases which did not result in death following an attack, immediately after an exposure, but reduced the life quality of the person. It is not clear to what extent these have been included in the estimates of mortality. The estimates, it seems, are based on Künzli et al (2000). In that study the mortality estimates were based on cohort studies which have taken into account long term effects. However, there are considerable uncertainties on the extent to which all such cases were identified and included in the estimates. A Canadian study finds the shares of cardio vascular and respiratory conditions in the acute premature deaths to be 42% and 11% respectively (Canadian Medical Association 2008).

To get an idea of the magnitude of such cases, we looked at the death statistics for heart disease and cancer. The total number of deaths due to heart disease in New Zealand was 5,912 in 2006 (Ministry of Health 2009). If the WHO estimates indicating PM_{10} accounting for 3% cardiopulmonary disease (Cohen et al 2004), were appropriate for New Zealand then the number deaths in New Zealand due to PM_{10} related heart diseases would be about 177.

The number of lung cancer cases in New Zealand was 1,457 in 2006. If the effect on cancer cases was about 5% as shown in the WHO estimate, then PM_{10}'s share of cancer deaths in New Zealand would be about 73. This makes a total of 250 deaths per year, part of which has perhaps not been included in the estimates. However, the effect of PM_{10} pollution on these two types of chronic diseases in percentage terms could be lower in New Zealand as the level of pollution is lower than in countries covered by the WHO. It is also possible that New Zealand has a lower impact of other factors causing heart disease or lung cancer. Thus it is not obvious that the share in New Zealand would be much lower. We have not included these in our estimates. Therefore, the number of deaths and the corresponding social costs are very likely to be underestimates to the extent that air quality impacts on cardiopulmonary disease and cancers are omitted from the analysis.

Another factor to consider is that death may occur due to complications arising from more than one cause. The cases where PM_{10} is a contributing factor but is not the only cause of death might not have been included in the Künzli et al (2000) estimates and hence in our estimates.

There is another type of effect, which is unlikely to have been included in the estimates. This refers to latent mortality impacts where the impacts are not known for a number of years after the exposure, i.e., no apparent effect on life quality but are manifested at a later date (Rowlatt et al 1998).

We have discussed above the possibility of underestimating the number of deaths due to late effects of pollution that might have not been included in the Künzli et al (2000) study and hence on the mortality estimates based on PM_{10} levels.

In the transport area, hospitalised injuries are considered serious injuries with loss of life quality on average of 10% of normal life. Thus the loss of life quality per hospitalised injury on average is $335,000. If the hospitalisation in the current case has similar severity and hence loss of life quality, then the value to society of loss of life quality due to pollution effects would be $335,000 per hospitalisation.

We have not included this in our update but have used it in the sensitivity analysis.

4.2.3 Downward trend in use of wood burners

A downward trend in the number of wood fuelled heaters being used is evident in recent Census data, suggesting other heating systems are being preferred. According to this data the number of wood burners reduced on average by 0.6% per year from 1996 to 2001 and then by 0.3% per year from 2001 to 2006. A fitted trend line based on this information suggests that the contribution of wood burners reduced by about 0.2% in 2007 and a continuation of the trend would reduce the health impacts by about 1.7% in 2020.

We find from Fisher et al (2007) that domestic emissions of PM_{10} accounts for about 40% of the total number of deaths. Keeping this constant and assuming the same proportionate effect occurs on hospitalisation and RADs we estimate the expected numbers of deaths, hospitalisations and RADs as shown in Table 5.

Table 5 Estimates after accounting for downward trend in wood burner use

Year	Premature death	Hospitalisations	Restricted activity days
2007	755	560	2,270,158
2008	730	551	2,226,706
2009	720	548	2,216,592
2010	709	546	2,206,745
2011	705	546	2,205,645
2012	701	547	2,206,981
2013	697	547	2,210,343
2014	693	547	2,212,678
2015	693	550	2,222,116
2016	694	552	2,230,202
2017	695	555	2,240,252
2018	696	558	2,249,392
2019	700	562	2,266,503
2020	703	566	2,281,225

Source: NZIER

4.2.4 Impact of 2009 budget insulation initiatives

In the 2009 Budget, government announced a scheme to subsidise installation or upgrading of insulation or cleaner heating in existing New Zealand homes. This is a partial subsidy available to homeowners of all income levels for homes built before 2000. The subsidy is available for ceiling and floor insulation, but not for wall insulation or double glazing.

The initiative is expected to enable 180,500 houses (about 12% of the housing stock) to be retrofitted with new or improved insulation over the 4 years from 2010 to 2013. How much of an impact is this likely to have on emissions of particulate matter?

The Ministry of Economic Development's Energy Data File shows total household energy consumption in calendar year 2007 to be 63.8 Petajoules. Of that, 34% (21.7PJ) was used on space heating and of that some 32% would be accounted for by wood fires (6.9 PJ). [16] Across the 574,460 dwellings recording wood fuel in the 2006 Census, this would amount to 12.1 Gigajoules per household per year on space heating.[17] According to Otago University's Household Energy Wastage website,

[16] In contrast to the Census data showing 39% of dwellings use wood fuel, CHRANZ 2004 estimate that wood accounts for 32% of energy used in space heating, a difference attributable to wood fires not always being the sole source of space heating in a property.

[17] In its Household Energy End-use Project, BRANZ has reported average annual use of firewood to be 13.7GJ per household, although this figure includes use for cooking and water heating as well as space heating (BRANZ 2006, Study Report 155, page 26). Table 65 in the same report suggests wood fuel used on space heating to be rather lower and varying with location.

partial insulation (ceilings and floors only) could save between 38% and 49% of the energy to heat a typical home, depending on which part of the country it is located. Across all regions the average saving is 42%. So on the basis of these figures the average house could save 5.1 GJ per year on space heating. Across 180,500 houses taking up the insulation subsidy offer, this would amount to 0.356 PJ per year saved, equivalent to 5.2% of the 6.9 PJ attributed to household wood-fuelled space heating at the end of the 4 year subsidy programme.

If that amounted to a 5.2% reduction in PM_{10} emissions it would contribute to an improvement in air quality and associated health outcomes quite independent of the NES. However, these figures are of necessity approximate. If the subsidised insulation replaces existing substandard insulation, the heat saving will be less than if it is used to insulate totally un-insulated houses. Some householders may choose to heat their houses as much as before, taking the benefit of insulation in the form of extra comfort: to the extent that this happens, the emissions to air are reduced by less, and the value of energy savings forgone can be viewed as an indication of the value placed on extra heating comfort. Some householders may also choose to use the subsidies available for clean heating to change from wood burners to other means of heating, which would further increase the reduction in wood fire emissions.

Assuming a 5.2% efficiency improvement in these houses and negligible improvement in other houses, where the insulation would make it more comfortable for residents but would not significantly affect the pollution level, our estimates indicate that the reduction in annual mortality would be negligible in 2009 and would gradually increase to about four fatalities by 2020. Adjustment taking this into account provides a new set of estimates as shown in Table 6.

Table 6 Estimates after adjusting for insulation initiative

Year	Premature death	Hospitalisations	Restricted activity days
2007	755	560	2,270,158
2008	730	551	2,226,706
2009	719	548	2,215,513
2010	708	545	2,203,013
2011	703	544	2,198,298
2012	697	544	2,195,223
2013	693	544	2,196,190
2014	688	544	2,198,525
2015	689	546	2,207,963
2016	689	549	2,216,049
2017	691	552	2,226,099
2018	691	554	2,235,239
2019	695	559	2,252,350
2020	699	563	2,267,072

Source: NZIER

These are the figures we use in our updated CBA, compared to those in the 2004 CBA presented in Table 3. To recap, they take into account infant deaths, the declining trend of household use of wood burners and the effects of the government's insulation initiative.

Box 1: Summary of assumptions behind the counterfactuals

Starting or base scenario	The status quo estimates in MfE (2004), Table 4 in this report, based on the following assumptions. • a 45% decrease in the number of multi-fuel burners from 2001 to 2021 in areas where these are not legislated • a 10% decrease in open fires from 2001 to 2021 in areas where these are not legislated • other solid-fuel burners are replaced with new solid-fuel burners 15 years from the date of installation • a linear relationship between emissions and concentrations for all areas (i.e. any reduction in emission would result in a proportional reduction in concentrations) • no impact of differences in the time of day of different sources relative to meteorological conditions, except in Christchurch (in Christchurch, a box model was developed by the National Institute of Water and Atmospheric Research to describe this relationship, see Gimson and Fisher, 1997) • an ageing population, with the proportion of the population over 30 increasing in each area by 20% of the 2001 proportion by 2021 (these estimates are based on limited data provided by Statistics New Zealand) • a 10% increase in industrial emissions in all areas except those with negative population projections • a 70% decrease in PM_{10} emissions from motor vehicles from 2001 to 2021 in areas where area-specific modelling and projections have not been carried out. (This is based on New Zealand Transport Emission Rate model (NZTER) emission rate projections, allowing for some increased traffic growth and congestion. Assuming NZTER estimates of emission rates are accurate this should be a conservative (underestimate) of the reductions for most areas).
Adjustments made	• Added estimated health effects on infants • Adjusted for downward trend in wood burner use • Adjusted for recent insulation initiatives

Source: NZIER, drawing on MfE (2004) and Wilton (2003)

4.3 The benefits of improving air quality

The principal benefit sought from air quality standards is a reduction in the social cost of poor air quality. A reduction in emissions of PM_{10} saves lives and reduces loss of life quality for those who suffer from pollution effects. In addition there is loss of output from mortality and also from incapacitation due to health conditions. This loss can be temporary or permanent.

The 2004 CBA considered three types of losses to society: loss of life and output for mortality and loss of output due to hospitalisation and restricted number of days for health conditions, in addition to number of days for recuperation after being released from hospital.

Some affected people may not die soon but may suffer chronic diseases such as cardiopulmonary or cancer. In that case, the life span may be reduced but also they will suffer for a long time between the exposure and death. In the absence of information on the size of this effect we have not included it here. However, we emphasise that this should be looked at and in its absence the estimates of social cost are clearly underestimated.

4.3.1 Premature death

The social cost per premature death is mainly the cost of loss of life. This is measured in New Zealand by the amount people are willing to pay (WTP) to reduce the risk of a fatal injury. The current value is $3.35 million at June 2008 prices (Ministry of Transport 2008). While this value is based on injury risks in the road traffic environment, it is commonly used in other public health policy areas as well. Our understanding is that this is the only official value of statistical life in New Zealand. Considering this we believe it is appropriate to use this value for estimating the social costs of loss of life and life quality due to pollution effects in New Zealand.

In the 2004 study, it was assumed that most fatalities from pollution exposure were elderly with an average of 18 months curtailment of life, and the average value of statistical life for elderly people who suffer this fatal consequence of exposure should be taken as 75% of the average value for the population. It is not obvious from the literature that only the elderly suffer this consequence. Besides, the basis for estimation that the risk of mortality increases by 4.3% for every 10 µg increase in PM_{10} is based on the mortality of people aged 30 years or more. Besides, Künzli et al (2000) note that while death occurs at an elderly age, the exposure occurred at all ages and they considered ages ≥ 30 years. In all cases where the person would suffer for many years, the amount people would be willing to pay to avoid this risk appears to be higher than for the risk of death within a short time after exposure.

Studies also show that the WTP for avoiding cancer can be substantially higher than that for avoiding a pedestrian death in a traffic accident (Rowlatt et al 1998, Clinton et al 2007).

Considering this our view is that in estimating the social cost of air pollution, the average VOSL is the appropriate value, without adjusting it on presumptions about the age of those most at risk[18]. The social cost is still likely to be underestimated due

[18] see Appendix B for further discussion

to the omission of loss of life quality due to the impact of air quality on chronic diseases (as outlined above).

The VOSL of $3.35 million implicitly includes loss of output, as the original New Zealand study (Miller and Guria 1991) on which VOSL is based did not clearly specify in the questionnaire that the respondent's willingness to pay amount should refer to pain and suffering only as was done in a later study (Guria et al 2003). So, the direct loss of output due to permanent disability or death is assumed to be part of the current VOSL. Therefore, we do not add any output loss on top of VOSL in our estimates. Neither do we allow for the indirect effect which would indicate the loss in GDP has not been considered, as that would make only a small difference in the total social cost. So the overall social costs, reduction of which constitute the benefits of the air quality NES, are lower in the analysis than they are likely to be in practice.

4.3.2 Hospitalisation

For hospitalisation cases, only the loss of output due to hospitalisation and recuperation has been included as the loss to society. An important component in the social cost of hospitalisation is the cost of medical treatment. The medical cost per hospitalisation of serious injuries in traffic accidents is estimated as $13,400 per injury. There the average length of hospitalisation is estimated as 12.6 days (Ministry for the Environment 2004). For pollution related hospitalisation, the estimate is 6.8 days per hospitalisation. It is not clear if these cases require similar, more or less expensive treatments. In the absence of that information, we assume that the cost of hospitalisation is proportional to the average days of stay in hospital.

This total medical cost has three components, hospital, emergency or pre-hospital and follow on treatments. Assuming that emergency cost per case would be the same and hospital and follow on costs are proportional to the number of days of hospitalisation, our estimate is $7,700 per hospitalisation.

As far as loss of output per day is concerned, we have followed a simple approach. The average income per employed person from wages and salaries, and self employment is about $548 per week estimated in June 2008 (Statistics New Zealand 2009). This gives a total of $1,804 million. Dividing this by the total population of 4.3 million in June 2008, we estimate the average loss of income as $423 per week. Since days in hospital includes holidays and weekends as well, we estimate the loss of income per day as the loss per week divided by 7 or about $60 per person per day. This includes only the direct effect of hospitalisation on the individual. We have excluded the indirect effect on the economy for simplicity and convenience. Inclusion of indirect effects is unlikely to significantly change the benefit cost ratio as shown later in the sensitivity analysis.

If the loss of life quality due to health effects requiring hospitalisation was similar to that in hospitalised injuries in transport, then the cost per hospitalisation would be higher by $335,000. Including this the social cost per hospitalisation would be

$343,404. We have not included this part in the social cost estimates in the main analysis. However, we have included this in the sensitivity analysis.

4.3.3 Restricted activity days

The 2004 cost benefit analysis assumed 55% of output is lost on restricted activity days. Since we are considering only the loss of work time, we estimate the loss per working day as about $85 (i.e., $423/5). This results in loss per RAD of about $46.50.

4.3.4 Effects on New Zealand's promotional image

It is sometimes suggested that air pollution could damage New Zealand's clean, green image and adversely impact tourism to New Zealand and demand for other New Zealand exports. A study some years back that attempted to value the clean green image (PA Consultants 2004) found some stated preference for New Zealand produce in its export markets relative to other produce from less environmentally favoured sources, but this was not the main factor influencing realised prices for New Zealand produce. International tourism to New Zealand peaks in the summer months when air quality issues are likely to be less noticeable. On balance, the effect of not achieving the standard is indeterminate but likely to be small at present, and reduction in pollution level is unlikely to make a substantial impact.

4.3.5 Estimating the effects of standards

Table 6 shows the estimates of premature deaths, hospitalisations and RADs if there was no regulation. In order to estimate the effects of the standards, we need to estimate these under the options of meeting the ambient air quality standard in 2013 (as is the current situation), in 2020 or any other scenario that we want to investigate.

The 2004 CBA estimates the effects for each of these scenarios and the status quo of not introducing the standards. We consider the effectiveness of the standards for each scenario as the proportionate reduction in number of pre-mature deaths, hospitalisations and RADs under the status quo situation. We then apply the same ratios on our estimates of the status quo (Table 6).

Box 2: Estimating benefits of ambient air quality standards

Benefits defined	Reduction in social costs of death, hospitalisation and RADs
Social costs	Loss of life valued by the Value of Statistical Life ($3.35 million at 2008 prices) Hospitalisation: value included is estimated medical treatment costs RADs valued by loss of productivity – no secondary effect on GDP is included Loss of life quality due to suffering over time – not included in the main analysis – included rough estimates in a sensitivity analysis
Assumptions about effect of the standards	The percentage reduction from the counterfactual is assumed to be the same as in MfE(2004). That is if the estimated reduction in number of fatalities due to the air quality standards was x% of the counterfactual in that analysis, we assume it is x% of the modified counterfactual in the updated analysis.

Source: NZIER

4.4 The costs of achieving improved air quality

Ideally, the costs of implementing the air quality NES would be estimated by identifying emitting activities in particular airsheds and the parts of the NES which restrain those activities to build up a picture of the opportunity costs caused, relative to the counter-factual without the continued NES. Available data linking activities, emissions, ambient pollution levels, populations at risk and resultant health impacts in particular airsheds are incomplete and the time available for this updated CBA is insufficient to rectify these gaps, so the analysis which follows is of necessity at a relatively high level, focusing on nationwide rather than localised effects. Despite the limitations in quantification such an approach can still be informative of whether the standards' implementation is likely to result in costs greater than or less than the benefits.

The basic approach adopted here is to update the items in the 2004 CBA, supplementing with new information where necessary to correct omissions in the original analysis. Costs are estimated exclusive of GST and other indirect taxes, in June 2008 dollars and projected in real terms across the years 2008 to 2020.

4.4.1 Costs to councils and government in administering the standards

Costs to councils and government in the 2004 analysis arose primarily with the monitoring of ambient air quality and with subsidies proposed to assist households to upgrade their wood-burners so that they comply with the standard. There are no subsidies specifically linked to the current air quality standards, so this item drops out of the analysis.

While some consequential monitoring costs (such as purchasing monitoring equipment) will already have occurred and can be considered as sunk costs for a forward looking analysis, there will continue to be recurring costs for councils which need to be taken into account. The estimates in the 2004 analysis are not out of keeping with analyses of other national environmental standards, so as a first approximation they could be retained for this analysis and updated with the Producer Price Index (Inputs). The new values would be:

- $64,987 per year for each of the 72 territorial authorities, rounded to $65,000
- $259,947 per year for each of the 16 regional councils and unitary authorities, rounded to $260,000
- $129,974 per year for central government in disseminating information, rounded to $130,000.

More recent information from the Auckland Regional Council's Stocktake of Council Domestic Fire and Traffic Emission Reduction Programmes at December 2008 suggests the impact on councils could be lighter than those assumed in the 2004 analysis in the updated assumptions above.[19] Territorial authorities may face no additional cost at all, as all the measures are implemented by regional councils. Implementation costs to date vary across regional councils, with major expenditures incurred or budgeted in the airsheds of Auckland, Rotorua, Napier-Hastings, Nelson, Christchurch, Alexandra and Invercargill. With the exception of Auckland (where spending is all on information and awareness), most of this spending is on support for Clean Heat upgrades, which the councils generally recover from recipients in targeted rates or similar charges. That expenditure is a financial cost on councils which would potentially double count the real resource cost of households who ultimately pay for their support through their targeted rates. The real resource costs for councils are confined to such matters as information and awareness campaigns, monitoring and administrative costs incurred.

The Stocktake does not differentiate between real resource costs and subsidy costs. It identifies $186.9 million of spending across all regions on air quality improvements, some already spent, some yet to be spent. Of this total $82.7 million is attributed to councils (as distinct from home-owners or other funding sources such as EECA). Assuming 10% of clean heat programme spending is attributable to administrative and other real resource costs and removing the subsidy component reduces the total to $12.2 million, or an average of $762,125 for each of the 16 regional authorities. This would amount to $127,021 per council per year in meeting the target by 2013, or $58,625 per council per year in meeting the target in 2020.

Central government expenditure on air quality standards has been estimated to be $110,000 per year on average over the period 2004-2008. As the coding picks up general expenses that may not be related to air standards this is more likely to be overstated than understated. Taking the average as occurring in the mid-point of the

[19] A similar conclusion on light territorial authority impacts can be drawn from a letter of 5 November 2008 on *Air Quality NES: Progress by Regional Councils*, from Nelson City Council manager Richard Johnson to the Ministry for the Environment

period (June 2006) and updating the figure by the PPI inputs results in a figure of around $125,000 at the start of the updated CBA. This figure is projected to recur throughout the updated CBA, and is unlikely to understate costs to government.

So in revising the analysis council and government costs are assumed to be:

- zero for territorial authorities
- $762,125 per regional council over the period needed to meet the target (substantially lower than the $260,000 per year from PPI adjusting the 2004 analysis figure)
- $125,000 per year for central government.

These figures would reduce the costs to local government relative to the costs in the original CBA in 2004. Whether this level of cost would be sufficient to achieve the target, or whether councils would need to put more resource into awareness raising and monitoring of compliance with the NES and their own rules to achieve the ambient standard by 2013 or 2020 is a moot point, and the effect of higher levels of spending can be explored through sensitivity analysis.

4.4.2 Costs to industry of complying with the standards

In the initial analysis in 2004, costs to industry were incurred in upgrading a specific number of sites each year to reduce their particulate emissions under the ambient air quality standard. Another industry cost identified was for wood burner manufacturers, but this was entered as zero, as it is part of the market price of wood burners (most of these costs would be passed on to their customers).

The initial study identified a wide potential range of costs to industries under the ambient air quality standards, and chose a representative "average" value of $100,000 per site towards the lower end of that range. Updated to June 2008 dollar terms that value would be $129,974 per site, rounded to $130,000. This is included in the updated analysis as the average cost per site for each year from 2008 to 2020. Alternative values can be tested in sensitivity analysis.

A further cost to industry would arise from the air quality NES if firms were denied resource consents to expand their production because the ambient air quality was below the desired standard. How large these effects might be, and where they might occur, cannot be determined without more detailed modelling than is possible in the scope of this updated analysis.

It is also possible that wood burner manufacturers could incur costs in developing new compliant models that they are unable to recover from consumers. This is unlikely to be large, but is also probably not zero. As with the potential opportunity costs of consenting denied, it is not practical to quantify manufacturers costs in a study of this kind. Rather, these costs will be addressed when considering the sensitivity of analysis results to changes in industry cost assumptions.

4.4.3 Costs for those affected by prohibitive standards

The initial 2004 analysis concluded that many of the activities affected by prohibitive standards were already tightly controlled by local rules, so there would be no additional cost from applying the NES. While this begs the question of whether those parts of the NES are redundant, they are treated the same in the updated analysis.

Two exceptions to this were consenting costs for school and hospital incinerators, and the adoption of alternatives to tar seal burning by roading authorities. On these:

- The air quality NES requires schools and institutions to obtain resource consent for incinerators or remove those unable to get consent by October 2006, so the assumption in this update is that, four years after implementation, the consenting is complete and there are no additional costs from this cause.[20]
- The estimate of alternatives to road seal burning from Transit New Zealand could be updated using the PPI inputs. However, the New Zealand Transport Agency reports that the practice has been phased out altogether for a number of reasons, so this item is now zero cost in the updated analysis.

4.4.4 Costs to householders in complying with the standards

A potentially large omission from the 2004 CBA is the cost imposed on householders in complying with the standard for wood burners. The air quality NES requires that new or replacement burners installed in urban properties should meet its standard for particulate emissions and energy efficiency, but this alone would not significantly affect the existing stock of non-compliant wood burners already installed. However, regional councils may consider (and some already are) applying more far-reaching measures in pursuit of their ambient air quality targets under the standard. It is not feasible in this update to model the individual measures of each regional council. However, it is possible to estimate the effect of converting the entire stock of wood burners in urban areas to compliant status. This is admittedly an extreme outcome of the NES that would only occur if all regional councils applied measures of greater stringency than the NES requires, but it provides a benchmark for the scale of potential effect on householders, and whether it, or some partial, less extreme approach would be large enough to have significant impact on the CBA result.

a) The number of affected wood burners

The number of wood burners that are affected by the NES is critical to the estimation of compliance costs for householders, but there is no reliable information on the stock of wood burners or the extent to which they comply. The 2006 Census reported 574,482 dwellings, 39% of the national total, as having wood fuelled appliances, a proportion that has been dropping in successive censuses but at a declining rate. This total will include open fires, multi-fuel burners, pellet burners and wood-burners in rural areas that are not subject to the air quality NES, but may be subject to more

[20] Ministry for the Environment (2008) *Report on Progress: National Environmental Standards for Air Quality, volume 1 Main Report* indicates that 68 school incinerator consents were issued prior to the target date. Others in schools and hospitals have been decommissioned, and monitoring of operation of consented school incinerators will be covered by councils' general oversight.

local rules and regulations imposed by regional councils in pursuit of ambient air quality targets under the NES.

The Household Economic Survey (formerly the Household Income and Expenditure Survey) tracked changes in the availability of heating appliances up to 2003/04, but this information has not been published with the results of the 2007 survey. In its 2003/04 survey it records the number of houses with a solid fuel burner of some description was virtually the same as it was in the 1985/86 survey, but within that category there had been a significant shift with a reduction in open fires and an increase in slow combustion burners, which in 2003/04 accounted for 67% of solid fuel appliances. If the proportional split among wood burning appliances is the same as that among solid fuel burning appliances, there would have been 384,900 enclosed wood burners across New Zealand in the 2006 Census.

Various sources indicate wood burners have a higher share of heating function in rural areas than in urban areas, so a proportion of those 384,900 are likely to be in areas not subject to the NES. The 2006 Census indicates that of recorded wood fuel appliances, 241,131 (42%) are located in city council areas and 333,327 (58%) are located in district council areas. This distinction cannot be equated as an urban rural split, but most of those in cities will be on properties of less than 2 hectares and subject to the NES, whereas a smaller proportion in districts will be. In the absence of precise details on the subject we have assumed that 90% of appliances in city jurisdictions and 60% of appliances in districts will be subject to the NES, which will therefore cover about 290,000 (73%) of enclosed wood burners across New Zealand.

b) The scenarios being examined

The air quality NES is currently being implemented with target reductions in ambient PM_{10} air pollution levels to be achieved by 2013. An alternative scenario is to consider what would be the effect of deferring the achievement of that target until 2020. Other scenarios look at variations on target setting (the permitted level of exceedances) and changes in the incentives acting on non-performing councils.

A contribution to target achievement will come from natural attrition and turnover of the stock of wood stoves, replacing old and non-compliant stoves with compliant ones. In that case the cost of compliance is limited to the additional cost of buying a compliant stove, compared to a non-compliant one. There is little reliable data on the differences between compliant and non-compliant stoves that are similar in all other respects. Most wood burner models sold on the domestic market now appear to be compliant, with imports and second hand models predominant among non-compliant burners. The initial CBA posited an additional cost of up to $200 for compliant models, which updated to 2008 levels with the PPI would be about $260.

If wood burners are replaced after 15 years use (as suggested by Wilton 2003), 6.67% of the stock will be replaced each year on average, assuming an even age distribution across the stock. This would replace the current stock completely in 15 years, and clear out remaining non-compliant burners. In this case the extra cost of the new standard is simply the additional cost of compliant burners over an equivalent non-compliant one.

To reach this point by 2013, however, requires accelerating the replacement to complete over 6 years, with 16.67% of the stock to be replaced each year. This means that 10% of the stock will be replaced each year before the end of its expected useful life. In the case of this extra 10% the cost imposed by the standard is the full cost of buying and installing the new compliant burner. Based on a selection of retail prices for burner and installation, the average appears around $3,000 per burner, or $2,667 excluding GST.

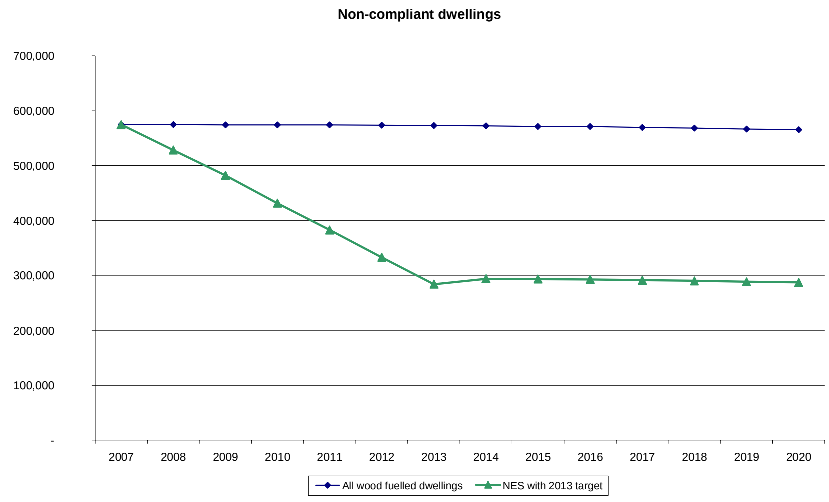
The estimated costs of compliance with the wood burner standard have therefore been based on the stock turnover required to reach the target date for full compliance, with the number of burner conversions split between those that would have occurred with natural turnover, and those which have been “forced” by the imposition of the standard. When the target date is pushed back to 2020, the required rate of turnover is reduced to 7.7% of stock replaced each year – only slightly above the natural renewal rate – with a different mix of low and high incremental cost of compliance.

In addition to this change in the existing stock of wood burners, the stock of dwellings will increase with population growth. Offsetting this slightly is the long term but recently slackening reduction in the proportion of dwellings with wood fuelled appliances, as evident in recent census data. Our analysis allows for this by applying a five year average of wood burners per head of population to population projections drawn from medium forecasts of Statistics New Zealand. For wood burners in new buildings, the incremental cost of the NES is the low compliance cost, i.e. the additional cost of a compliant over equivalent non-compliant burner.

The effects of these projections on the number of dwellings with non-compliant wood burners are illustrated in the following diagrams. Figure 4 shows the accelerated progression to achievement in 2013 of full compliance with the standards among wood burners subject to the standard, and Figure 5 shows the slower transition to compliance by 2020. The residual of around 300,000 non-compliant wood burners after the target dates covers all open fires, pellet burners, multi-fuel burners and wood burners in rural areas not subject to the standard.

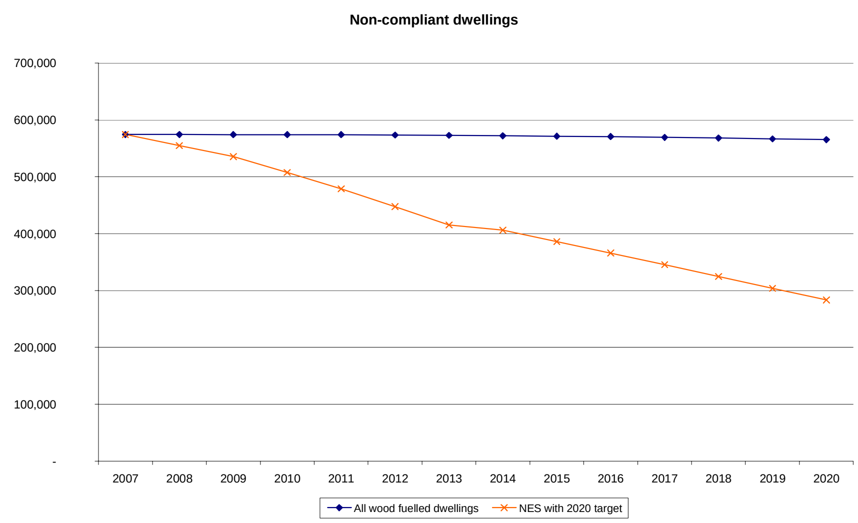
In these figures, the flatter line is the estimated stock of non-compliant wood-fired appliances, based on continuation of recent trends in population growth and the proportion of houses using wood fuelled appliances. Houses being converted under the existing EECA clean heat programme and the Warm Up New Zealand: Heat Smart subsidy programme introduced in the 2009 budget have been removed from the total stock of appliances that needs to be converted by the ambient air quality standard target dates. Almost half of the stock of wood burning appliances would be affected by the air quality NES wood burner standard under these assumptions. These assumptions have a bigger impact than is likely to occur in practice, to illustrate how the benefits compare with even a relatively severe level of costs.

Figure 4 Progression to compliance by 2013



Source: NZIER

Figure 5 Progression to compliance by 2020



Source: NZIER

c) Effect of the clean heating subsidy

In its May 2009 Budget the Government announced a scheme to subsidise insulation and installation of cleaner heating systems in homes built before 2000.[21] This would provide subsidy of up to $1,300 (GST inclusive) per home on ceiling and under floor insulation installed by approved installers, and up to $500 (GST inclusive) on upgrading open fires to cleaner systems such as heat pumps or flued gas heating in

[21] Known as the New Zealand Insulation Fund, or Warm Up New Zealand: Heat Smart.

houses with the new insulation. This scheme will operate over the four years ending June 2013 and build up to support insulation in 180,500 dwellings, and heating upgrades in 90,000 of those.

Insulation has the potential to reduce the energy load required to heat a given area, and reduce the consumption of fuel and associated emissions. However, the potential energy savings are commonly not realised because of the so-called “rebound effect”, whereby owners of the insulated houses consume some of the savings in greater heating and comfort levels. This does not negate the benefit of insulation, rather transforms it to a different form, for the value of comfort gain for householders can be expected to be at least as much as the value of energy saving they forgo. Some energy savings will be realised, but the rebound effect makes it difficult to determine how much.

A more tangible effect on this cost benefit analysis comes from the subsidy of clean heat upgrades, for this may be used to support conversion from non-compliant to compliant wood burners, or substitution to other heating systems such as heat pumps. To the extent that this is so, the subsidised conversion costs would happen in any case, with or without the air quality NES, and cannot be attributed as being caused by the NES. The result for this CBA is that some of the housing stock that is upgrading its wood burners will do so regardless of the NES i.e. the stock to be converted (and associated costs) due to the NES will be lower than without the subsidy scheme. Similarly, some of the benefit observed over the analysis period will be attributable to the subsidy scheme and cannot be credited to the air quality NES.

We have modelled this by removing the number of houses converted each year under the subsidy scheme from the stock of houses with wood burners subject to the scheme. As the subsidy may be used on wood burners not subject to the NES, we assume in the first instance that 39% of homes upgraded under the scheme have wood burners subject to the NES. This assumption can be tested in sensitivity analysis. The effect of the subsidy terminating s apparent in the upward blip that occurs in 2013 in Figure 4 and Figure 5.

d) Choices for those forced to upgrade under the NES

Those required to upgrade their burners because of the NES are faced with replacing their wood burners with a compliant model, or alternatively substituting to an alternative heating system. Heating systems vary in their heat output, their capital costs and their running costs, and choice of a replacement will reflect considerations of the full cost of replacement (capital and running cost) plus other factors such as heat quality, aesthetic appeal and convenience in use.

Based on purely financial considerations wood burners are still a competitive form of home heating, primarily because the fuel has relatively low cost and the householder’s labour in their operation does not appear in such analysis. Table 7 shows a comparison of four heating options for a specified annual heat output of

3800 kWh,[22] comparing capital cost, annualised cost of capital discounted at 8% over 15 years, running cost and the combined total cost per year. On this basis the wood burner has slightly lower running costs but slightly higher capital costs than electric heat pumps that have made big inroads into the space heating market in recent years. Electric plug in and portable gas heaters are less competitive because, despite lower capital costs they have much higher running costs. This pattern is accentuated the higher the heat output, so for heating a large area in a house a wood burner is an attractive option.[23]

Table 7 Comparative cost of 3800 kW heat output

	Capital cost $	Annualised cost $/yr	Running cost $/yr	Total cost $/yr
Wood burner	3,000	350	285	635
Heat pump	2,750	321	293	614
Electric plug in	150	38	741	779
Portable gas	300	45	802	847

Source: NZIER, based on data from EECA and MfE[24]

Should a householder take the opportunity of the NES requirements to upgrade from a wood burner to a heat pump, the economic cost of the next best alternative is still that of a wood burner: any extra cost in running a heat pump can be taken as the value placed by the householder in specific features of the heat pump, such as the convenience of greater control or the versatility of its air cooling functions. On the assumption this is freely chosen by the householder, it is not caused by the NES and the economic cost imposed by the change is that of a replacement wood burner.

However, because some people may baulk at the capital cost of replacing their wood burners (particularly at a time of economic recession), some may opt to substitute to electric plug in or portable gas heaters because of their significantly lower capital costs. This may be economically irrational in terms of the figures above, and commit the household to higher energy costs or reduced heating as a result, but it is an outcome that may arise and for which there is no rationalising benefit.

We have allowed for a proportion of those who upgrade their heating to make such switches, on the assumption that they will most likely switch to the most ubiquitous option, electric plug in heaters. All such houses incur an incremental cost over a wood burner of $102 per year, which is not a one-off cost but recurs every year.

[22] For illustrative purposes, drawing on BRANZ (2006) HEEP Study Report SR155 "Energy Use in New Zealand Households" which states average household energy use per year is 11,410 kWh, of which space heating accounts for 34% or 3,879 kWh.

[23] Note there are also qualitative differences in the heat from different appliances and fuels. Houses heated by LPG or electricity tend to be the coolest, partly because their use is intermittent, whereas those heated by enclosed solid fuel heaters are warmer (BRANZ 2006).

[24] Ministry for the Environment (2005) Warm homes technical report: Detailed study of heating options in New Zealand, Phase 1 Report; and EECA (2009) Energywise Action Sheet 5

e) Overview of costs of the wood burner standard

From the above the costs of the wood burner standard comprise:

- New dwellings are assumed to fit compliant burners, bearing the low incremental cost of $260 on installation, running costs unchanged
- Dwellings with existing burners being replaced on their routine renewal cycle fit compliant burners, bearing the low incremental cost of $260
- Dwellings with existing burners being replaced faster than their routine renewal cycle fit compliant burners, bearing the full cost of installation of a new burner, but with running costs unchanged[25]
- A proportion of dwellings with existing burners replaced faster than routine renewal (initially assumed to be 10%) fit alternative electric heating, bearing additional costs of operation on an on-going basis.

4.4.5 Cost of meeting the standard - preliminary settings

Allowing for the costs to householders in meeting the requirements of the wood burner standard makes a substantial difference to the analysis from that which appeared in the 2004 CBA. It is also an element in the analysis that is very sensitive to the timeframe over which the ambient air quality standard is met.

In meeting the standard by 2013, the wood burner cost alone represents 92% of the total costs incurred over the analysis period to 2020. Total costs over the analysis period in that case amount to $333 million in present value terms, and other significant shares of that cost fall on regional authorities (3.0%), and industry (4.2%).

When introduction of the standard proceeds more slowly to be achieved by 2020, the share of cost borne by households with wood burners drops to 74%. The total cost in that case is $73 million over the analysis period in present value terms, and the other principal shares are borne by regional authorities (10.9%) and industry (15.1%).

Despite these large costs, they are less than the benefits from reduced adverse health effects as estimated by Wilton (2003) and updated for this analysis, as discussed next.

[25] It can reasonably be argued that improved energy efficiency in the specification of compliant wood burners should lower running costs relative to non-compliant ones. However, some wood burner manufacturers have suggested that this efficiency gain is dependent on, and offset by the added costs of, proper maintenance of burners. In the absence of clear evidence of the efficiency gain of burners in use, running costs are unchanged between compliant and non-compliant burners in this analysis.

Box 3: Estimating costs of ambient air standards

Basis for cost estimates in the updated CBA

Regional council administration costs	Updated drawing on recent spending data
Territorial authorities costs	Zero entry
Government information and administration	Updated drawing on recent spending data
Industry site adaptation measures	Updated by PPI, and further examined in sensitivity analysis
Business forgone from consent constraints	Not capable of being valued
Consenting of school and hospital incinerators	Completed, so zero entry
Alternatives to tar seal burning	No longer practised, so zero entry
Other activities, landfills and wire burning	Zero entry
Householder costs of burner compliance	Estimated based on a model allowing for new houses, declining share of houses with wood burners, clean heat/ insulation subsidies, routine replacement of burners, forced replacement and induced substitution to other space heating
Manufacturers' costs in supplying compliant burners	Zero entry
Government/council subsidy	Zero entry, as not solely attributable to the air quality standard

Notes: (1) Items in the left column are the same as those in the 2004 CBA
(2) See Appendix B for differences between 2004 and 2009 CBA

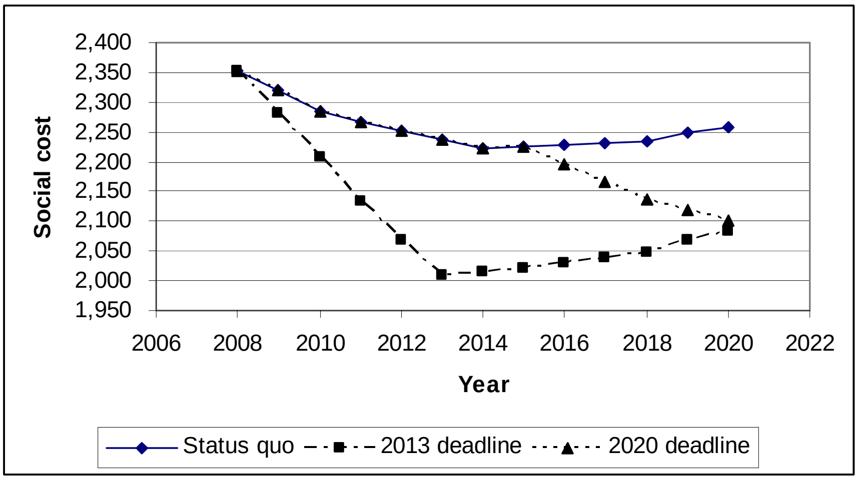
Source: NZIER

4.5 Scenario 1 - prolonging achievement to 2020

We can look at the benefits of establishing the air quality standards effective from 2020 in comparison with no air quality standards at all. In that case, we can compare the benefits as the difference between social costs under the status quo (no standards) and the expected social cost under the option of the ambient air quality standards being met from 2020. The unit social costs are estimated at 2008 prices.

The estimated social costs under the three scenarios: status quo, 2013 deadline and 2020 deadline are shown in Figure 6.

Figure 6 Social costs at 2008 prices



Source: NZIER

We estimate the present value of social costs of health effects with 2008 as the base year. The discount rate used for this purpose is 8%, the current rate advised by the Treasury.

Table 8 Estimated health effects of the ambient air quality standards

Adverse health effects avoided per year by meeting the ambient air quality standard

Year	Meeting the standards from 2013)			Meeting the standards from 2020		
	Deaths	Hospitalisations	RADs	Deaths	Hospitalisations	RADs
2009	12	11	20552	0	0	0
2010	25	22	41093	0	0	0
2011	43	35	69685	0	0	0
2012	58	47	94066	0	0	0
2013	72	59	117523	0	0	0
2014	66	57	108482	0	0	0
2015	64	56	106158	0	0	0
2016	63	56	102089	10	10	17316
2017	61	56	99457	21	20	36395
2018	59	56	95461	31	30	54343
2019	57	55	91579	41	40	72842
2020	55	55	88307	50	50	88470
Total	635	565	1,034,452	153	150	269367

Source: NZIER

This provides the present value of health effects from 2008 to 2020 under status quo, nationwide achievement by 2013 and nationwide achievement by 2020 (Table 9).

Table 9 Estimated Present values of benefits and costs

Option	Estimated total (2008-2020)			
	Deaths	Hospitalisations	RAD	Present value of Social cost ($ million)
Status Quo	9,092	7,143	28,838,238	20,998
2013 AQS	8,457	6,578	27,803,787	19,710
2020 AQS	8,939	6,992	28,568,872	20,766

Difference from status quo						
Option	Deaths	Hospitalisations	RAD	Present value ($ million)		B/C
				Benefit	Cost	
2013 AQS	635	565	1,034,452	1,289	333	3.9
2020 AQS	153	150	269,367	232	74	3.2

Notes: (1) The benefits are estimated as the differences in social costs under status Quo and air quality standards. Any difference observed is due to rounding.

Source: NZIER

This shows that the incremental benefit of achieving standards from 2013 over the status quo situation is $1,289 million. It would be $232 million if the standards were fully achieved only from 2020. This means that delaying the effective date of achieving the standards reduces benefit to society by $1,056 million (or net benefit by $797 million) compared to achieving the standard by 2013.

4.5.1 Cost benefit comparison

If the standard is achieved by 2013, the early realisation of health benefits would result in a net present value of net benefits of $955 million and a benefit cost ratio of 3.9. If standard achievement is postponed until 2020, the deferral of benefits is greater than the reduction in costs, reducing the NPV to $159 million with a benefit cost ratio of 3.2.

The modelling of the wood burner standard is a dominant influence in this result, because of the costs incurred in forcing the turnover of existing stock of burners above the rate at which they would otherwise be replaced. The turnover of existing stock, as modelled in this analysis, goes beyond what the NES requires, and can be regarded as a rather severe interpretation of the outcomes that the standards are likely to induce. Various other factors influencing this result are unknown or are assumed and their effects need to be tested in sensitivity analysis.

Table 10 Summary of updated CBA with baseline assumptions

Ambient standards over status quo met by	2013	2020
Reduction in premature mortality (to 2020)	635	153
Reduction in hospitalisations (to 2020)	565	150
Reduction in Restricted Activity Days (to 2020)	1,034,452	269,367
PV combined benefits $M	1,289	232
PV Costs $M	333	74
NPV (GDP + VoSL - Costs) $M	955	159
B:C Ratio [(Gross Benefits)/(Costs)]	3.9	3.2
C Effectiveness [(Costs)/(Mortality reduction)]	$524,712	$481,807
Distribution of costs		
Regional councils	3.0%	10.9%
Territorial authorities	0.0%	0.0%
Central government	0.3%	1.4%
Industry	4.2%	15.1%
Road controlling authorities	0.0%	0.0%
Households	92.4%	72.6%

Source: NZIER

After updating the CBA and revising it twice in response to feedback received, the results of the CBA do not materially change and suggest net benefits are likely from pursuing the air quality standards.

Compared with the 2004 CBA, NES target achievement in 2013 in the updated analysis has benefits 3 times bigger and costs 3.01 times bigger. The resulting net present value is 3 times greater, but the benefit cost ratio is the same.

When the NES target achievement is pushed back to 2020, the benefit is only 54% as large as in the 2004 study and the costs 33% smaller. The NPV is only 50% as large and the benefit cost ratio is 19% smaller.

The changes from the 2004 analysis have been brought about mainly from an increase in value of premature fatalities avoided and associated ill-health costs. On the cost side, some items have dropped out either because they are likely to be immaterial (e.g. costs for territorial authorities) or because they refer to one off adjustments that are now sunk costs of no relevance to a forward looking analysis.

But those cost reductions are more than offset by allowing for a “worst case” impact on private households from meeting a stringent imposition of wood burner standards. Overall the baseline analysis is likely to be conservative, both because of the over-weighting of wood burner costs and because no account is taken of on-going health benefits from improved air quality beyond the end of the analysis period in 2020.

The analysis results are heavily dependent on the value attached to avoidance of fatalities and other manifestations of ill health. It might be argued that the VOSL value derived from transport studies is not appropriate for use in air quality mortality risk, but that begs the question of what alternative value should be used. The transport VOSL is widely used in New Zealand as a benchmark of value for reducing risks to safety and health, and it is not obvious that it is too high for use in assessment of air quality impacts. International literature on VOSL points towards a higher willingness to pay to reduce risks of fatality that are incurred involuntarily or which involve risk of lingering incapacity, compared with the risk around private transport over which respondents feel they have an element of control, so the VOSL for risks from ambient air quality may be even higher than for transport applications.

4.5.2 Options 2013 vs. 2020

These results suggest there is a significant disadvantage in deferring full implementation of the standards to 2020: more people die and get hospitalised with the greater exposure to bad air quality over the longer time frame. So although there are significant savings in the implementation cost of the standards with a later achievement, there is an even greater increase in the societal cost that the community bears in the meantime, from increased ill health and mortality.

Deferring attainment significantly changes the distribution of costs across the community, because of the effect of forcing more changes on the household wood burner sector. If the ambient air quality targets are to be fully achieved in 2013, under the updated analysis 3% of the costs would be borne by regional councils, 4.2% by industry, 0.3% by central government and just over 92% would fall as private costs on householders. If the same target were to be achieved by 2020, the proportion of costs borne by regional councils would rise to 11%, and industry would bear 15.1% and government 1.4%. But the share borne by householders would drop to 73%, reflecting a reduction in cost imposition due to fewer properties being forced to upgrade or renew burners that still have useful life left in them.

These base results are predicated on a model of household compliance that is more stringent than specified by the air quality NES, and hence they are likely to be more burdensome on households than is likely to eventuate. The less severe the impact on householders, the greater the proportional shares of other affected parties. Another caveat on this analysis is that the linkage between particular measures under the NES and air quality and consequent health impacts is unclear from the 2004 modelling we have to hand. The effect of this and other uncertainties on the results need to be tested in sensitivity analysis.

4.6 Scenario 2 - increasing the number of permitted exceedances

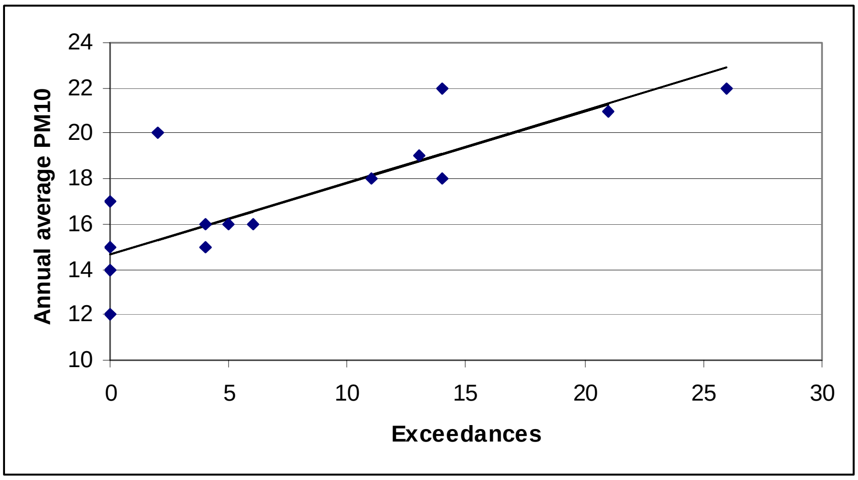
The current regulation requires the 24 hour maximum PM_{10} level not to exceed more than once during a year. One scenario to consider is allowing this level to be exceeded more often, up to five times during a year.

The studies on which the mortality and other effects have been estimated use average annual value of 24 hour maximum PM_{10} levels. We compared the average values and number of exceedances observed in 2007 and estimated a linear regression line. This is shown in Figure 7. This is based on 17 observations. We excluded the observations for Alexandra as its number of exceedances was considerably higher in comparison with other areas.

The estimated relationship suggests that one exceedance during a year corresponds to annual average 24 hour maximum PM_{10} level of about 15. If the number of exceedances per year is five by 2020, that would correspond to an annual average of 16.2. This is about 8% higher than the level required to have only one exceedance per year. However, following the same approach as in section 0, and considering 649 deaths in 2020 with annual average of 24 hour maximum PM10 level of 15.0, the annual average value of 16.2 would result in 759 deaths (about 17% increase) in 2020. The increase in present value of the social cost of bad air quality in 2020 alone would be about $165 million.

This is a very crude estimate. Without putting too much emphasis on the values, we would like to point out that a relaxation of exceedance limit may substantially increase the number of deaths due to PM_{10} pollution.

Figure 7 Annual average of 24 maximum PM_{10} levels



Source: NZIER

4.7 Scenario 3 - introducing fines for non-compliant councils

This scenario suggests that non compliant councils would be fined. This may motivate councils to take actions. However, it is not clear to what extent this would affect the level of PM_{10} emissions.

In the absence of detailed information on how this policy would work and what its likely impact would be, we estimate the effectiveness of any change in the annual average of 24 hour maximum PM_{10} values[26].

From a national perspective, a system of fines to encourage compliance and enforcement is in the nature of a transfer payment from local ratepayers to national taxpayers. But it is not a costless exercise, as there will be deadweight costs in administering, enforcing and collecting the fines. The effectiveness of fines in spurring councils to greater efforts in implementing the standard is also open to question. It will depend on the responsiveness of councils to this particular type of stick incentive, which in turn depends on their circumstances and ability to reallocate resources from other activities to improving air quality.

4.8 Scenario 4 - non-compliance plans for councils not meeting standards

If one council does not take the necessary action to ensure compliance with the air quality standards, it will have a higher level of PM_{10} and consequently higher level of pollution related deaths, as shown in the 2004 case.

This is likely to be confined to the council area and may not affect other areas. In that case, the increase in total number of deaths in the country in percentage terms would depend on the share of this council in the total number of deaths.

For example, share of Auckland in the total number of deaths in 2003 was estimated as 419, when the total for New Zealand was estimated as 872 (Wilton 2003). This suggests Auckland's share in total number of PM_{10} related deaths as about 48%. Our estimates suggest that the total number of deaths in 2020 under the current regulation would be about 644. If the same relativity is held, then the number of deaths in Auckland would be 309.

Our estimates assume the same trend as estimated in the 2004 CBA. We have not estimated the corresponding annual average of 24 hour max PM_{10}. To illustrate the impact of not taking enough action, suppose the regulation would require Auckland to bring down the annual average PM_{10} to 15. If Auckland does not take the necessary

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[26] As we discussed in sections 3.2 and 0, the number of deaths is estimated as $N = \frac{P_0(RR-1)P_C(X-B)}{10}$, where N = number of deaths, P_0 = base line mortality rate per 1000 population, PC = population in thousands, RR-1 is the proportionate increase in number of deaths per 10 µg increase in annual average PM_{10} level. This suggests that

$\frac{\partial N}{\partial X} = \frac{P_0(RR-1)P_C}{10}$. This is a constant at a given level of population. If this is equal to say α, then for every one 1 µg increase in the annual average value of 24 hour maximum PM_{10} increases the number of deaths by α.

action and as a result the annual average is higher to say 17, then the number of deaths in 2020 would be higher by about 76. That would increase the total number of deaths from 644 to 720, an increase of about 12%.

Councils required to implement non-compliance plans would incur additional costs in the development and oversight of these plans. From a national perspective these costs may not be large, but from the viewpoint of the individual councils they could be substantial. The effectiveness of non-compliance plans improving air quality depends on both the quality of the plan and the ability of the councils to find the resources to implement it. The national cost depends on the number of councils required to apply such plans and the cost each council incurs in doing so. The benefit depends on the effect on air quality and resulting changes in adverse health impacts. None of these can be estimated with precision.

4.9 Sensitivity of results to changes in inputs

4.9.1 Higher than expected downward trend

Based on the observed levels of annual average PM_{10} (24 hr. maximum) in 2007 and the 2004 estimate in Ministry for the Environment (2004), we found that the status quo estimates could be lower by 8%. In that case, part of this could be due to the introduction of the air quality standards and some actions already taken by councils. If we excluded the effects, the estimates of present value of social costs of would be as shown in Table 11.

Table 11 Estimated Present values of benefits and costs – with downward adjustment due to observed PM_{10} levels in 2007

Option	Estimated total (2008-2020)			
	Deaths	Hospitalisations	RAD	Present value of Social cost ($ million)
Status Quo	8,380	6,584	26,581,359	19,355
2013 AQS	7,795	6,063	5,627,864	18,167
2020 AQS	8,240	6,445	26,333,073	19,141

Difference from status quo						
Option	Deaths	Hospitalisations	RAD	Present value ($ million)		B/C
				Benefit	Cost	
2013 AQS	585	521	953,495	1,188	333	3.6
2020 AQS	141	138	248,286	214	74	2.9

Notes: (1) The benefits are estimated as the differences in social costs under status Quo and air quality standards. Any difference observed is due to rounding.

Source: NZIER

These estimates are slightly lower (8%) than those shown in Table 9. These suggest that the benefit of standards being met in 2013 is about $1,188 million and it is $214 million if the meeting date is deferred to 2020. The present values of net benefits would be $854 million and $140 million, with benefit/cost ratios of 3.6 and 2.9 respectively.

4.9.2 Variation in discount rates

In this study we have used a discount rate of 8%. For sensitivity analysis we also used 5% and 10% discount rates. The estimated present values of benefits are shown in Table 12.

Table 12 Present value of benefits and costs at 5% and 10% discount rates ($ Million)

Option	Estimated total (2008-2020)				
	Deaths	Hospitalisations	RAD	Present value of Social cost ($ million)	
				at 5% discount rate	at 10% discount rate
Status Quo	153	150	269,367	24,228	19,240
2013 AQS	8,457	6,578	27,803,787	22,675	18,094
2020 AQS	8,939	6,992	28,568,872	23,915	19,048

Difference from status quo									
Option	Deaths	Hospitalisations	RAD	5% discount rate			10% discount rate		
				Present value ($ million)			Present value ($ million)		
				Benefit	Cost	B/C	Benefit	Cost	B/C
2013 AQS	635	565	1,034,452	1,552	355	4.4	1,146	320	3.6
2020 AQS	153	150	269,367	313	85	3.7	192	67	2.8

Source: NZIER

The estimates suggest that the benefits i.e., reduction in social costs would be $1,552 million and $313 million respectively for ambient air quality standards being met by 2013 and 2020, if the discount rate was 5%. The corresponding figures at 10% discount rate would be $1,146 million and $192 million respectively.

At a 5% discount rate, the baseline assumptions in achieving the standard by 2013 would yield NPV of $1,198 million with a BCR of 4.4. With 2020 achievement the NPV would be $228 million and the BCR 3.7. At 10% discount rate meeting the standard would yield NPV of $825 million with BCR of 3.6 if achieved by 2013, or $124 million and BCR of 2.9 if achieved by 2020.

4.9.3 Loss of life quality

As we have noted earlier, the social costs are likely to be underestimated due to the non-inclusion of loss of life quality for those who suffer from illness due to PM_{10} exposure. These include some chronic cases as well as large proportion of latent cases. If those could be estimated and included here, the social costs estimates would be higher than those shown in Table 5.

If the loss of life quality per hospitalisation was similar to that of hospitalised injuries in transport crashes, then the benefit cost ratio of 2013 option would be 4.2 instead of 3.9 and the ratio for 2020 option would be 3.5 instead of 3.2, shown in Table 10.

4.9.4 Including the indirect effects of loss of income

The loss of income due to permanent incapacitation is supposed to be part of the VOSL. We have no estimate of the direct income loss component, as it is not separated out in the VOSL calculations. Without this direct effect, it is not feasible to add the indirect effect.

For hospitalisation and RADs, we have estimated the loss of income to affected individuals which can be treated as the direct effect. If the indirect effect is as much as the direct effect, then the B/C ratio for 2013 option would increase from 3.9 to 4.3 and for the 2020 option, it will increase from 3.2 to 3.5. Thus its effect on the B/C ratio is small.

4.9.5 2020 benefit starts in 2010

The 2004 CBA assumes that the benefits of the ambient air quality standards being met from 2020 start occurring earlier, from 2016, as air quality begins to improve. This might be an appropriate assumption to consider before the introduction of the new standards. Now that the standards have already been introduced, it is expected that the improvement in air quality will have already started. This trend is likely to continue, perhaps at a lower rate, if the 2013 deadline were to be shifted to 2020.

The expected health effects in 2020 should be the same as discussed earlier. However, in the absence of more precise information, we assume that there will be a linear downward trend from now to 2020 in all these three areas of health effects: deaths, hospitalisations and RADs. This would bring the estimates of present value of benefits as shown in Table 13.

Table 13 Estimated Present values of benefits and costs –if PM_{10} is reduced gradually from 2010 to 2020

Option	Estimated total (2008-2020)			
	Deaths	Hospitalisations	RAD	Present value of Social cost ($ million)
Status Quo	9,092	7,143	28,838,238	20,998
2013 AQS	8,457	6,578	27,803,787	19,710
2020 AQS	8,779	6,900	28,670,681	20,421

Difference from status quo						
Option	Deaths	Hospitalisations	RAD	Present value ($ million)		B/C
				Benefit	Cost	
2013 AQS	635	565	1,034,452	1,289	333	3.9
2020 AQS	313	243	167,558	577	74	7.8

Source: NZIER

This suggests that the benefit of the ambient air quality standards being met in 2020 is $577 million in comparison with the status quo, producing a benefit/cost ratio of 7.8 and a net benefit of $503 million on baseline assumptions at 8% discount rate. But even though the benefit/cost ratio of meeting the standard is higher than in 2013, the net present value is less than the $955 million from achieving it by 2013. This also shows substantial improvements in benefits in comparison with the earlier assumed option that the benefits for 2020 AQS would start occurring from 2016 as shown in Table 9. We have assumed the costs would be the same. However, it is likely that there would be some changes in costs as well. We have no information with which to relate costs with benefits for this change in benefit stream, so we would not like to put much emphasis on the net benefit or the benefit cost ratio. However, it is clear that the benefit value would increase substantially if it started occurring early.

The status quo we have considered here is the situation in the absence of air quality standards. However, the standards have now been established. If it is not feasible to achieve compliance of the standards by 2013 as planned in 2004 and it needs to be postponed to 2020, then the above estimates suggest that such an option would produce net benefit. But the net gain would be lower due to higher levels of pollution for a few more years under the 2020 option in comparison with the 2013 option.

Our estimates indicate that under the 2020 option, the present value of benefits (or saving in social costs of health effects) would decrease by $640 million but the present value of NES implementation costs would reduce by only $259 million in comparison with the 2013 option. This indicates there would be net loss in postponing the compliance date of the air quality standards to 2020. There is only merit in delay if it is considered more important to reduce tangible expenditures rather than saving health costs which are uncertain or "hidden".

This suggests that if the 2013 option is achievable, then it is not preferable to postpone the compliance date to 2020. However, if the timeframe is too tight for achieving the standards compliance by 2013, and 2020 is a more appropriate deadline, then the option of making the standards strictly effective from 2020 would still result in net benefit.

4.9.6 Changes in specific costs

Particular uncertainty surrounds two specific items on the cost side of the analysis. One is the cost on industry of converting individual sites to lower pollutant emission levels, and also the possibility of business being forgone because of the consenting restraints brought in due to non-achievement of target ambient levels. The other is the impact on householders of the wood burner standard, given that the NES itself has relatively modest impact on the stock of wood burners but that regional councils may apply more stringent rules in pursuit of the ambient air quality targets.

In the case of industry site costs, the initial CBA (MfE 2004) assumed 10 sites a year would be treated at an average cost of $100,000 per site, towards the lower end of the range of possible costs canvassed from industry consultants. The updated analysis continues this in the base assumption set, with the average cost updated to $130,000. If this average cost is increased 10 fold (or 10 times as many sites affected), holding other things constant, industry's share of total cost of NES implementation would rise to 31%, and the benefit cost ratio of attaining the target in 2013 would fall from 3.9 to 2.8. In pursuing the target achievement by 2020, industry's share of total costs would increase to 64% (because the impact on household wood burners is relatively lighter with the later target) and the benefit cost ratio would fall from 3.2 to 1.3. It would take a 15 fold increase in industry costs relative to the base assumptions to lower industry costs sufficiently to lower the BCR to one in 2020, at which point industry costs would account for 73% of total costs. The same increase in industry costs with 2013 standard achievement would still be net beneficial (BCR=2.4), with industry accounting for 40% of costs.

Without detailed modelling of how ambient pollutant levels are likely to develop in different locations, which is beyond the scope of this updated cost benefit analysis, there is no way to model the forgone business caused by consenting constraints. There is likely to be some real economic cost from this source if existing site expansion is precluded by consenting constraints, or if business has to be relocated to sites without such constraints. In the latter case the costs are limited to those incurred in relocation, as the business itself is recreated in a new location, so the cost to the nation will be less than the costs incurred by particular localities from which such business is diverted. The updated analysis suggests substantial increased industry cost, either on particular sites or borne by wood burner manufacturers, could be incurred before the costs outweighed the benefits. The Ministry for the Environment's progress report has identified no evidence of significant site-based costs, and no instances of consents yet being declined, so at this stage it seems unlikely that industry costs would be large enough to overturn the result.

Regarding the household wood burner costs, the base model is set at a more severe level of imposition than the NES requires, and any relaxation of that setting will reduce the costs incurred. Within the current model, there is also a split by assumption between those who replace wood burners at a relatively high cost of $2,667 per burner, and those who substitute to other heating systems. Changing the setting to no substitution to alternative heating systems would increase the cost for householders and reduce the net benefits somewhat, but not sufficiently to change the net beneficial result of attaining standards in either 2013 or 2020.

The results of these changes are presented in Table 14.

Table 14 Sensitivity to changes in cost items

Standards over status quo achieved by	2013 10 fold industry cost	2020 10 fold industry cost	2020 15 fold industry cost	2013 no fuel substition	2020 no fuel substition
PV combined benefits $ Million	1,289	232	232	1,289	232
PV Costs $ Million	460	174	229	355	75
NPV (GDP + VoSL - Costs) $ Million	829	59	3	934	158
B:C Ratio [(Gross benefits)/(Costs)]	2.8	1.3	1.0	3.4	2.9
C Effectiveness [(Costs)/(Mortality reduction)]	$758,701	$284,960	$373,760	$603,748	$127,068

Source: NZIER

Other cost items are less likely to incur changes of an order large enough to significantly change the result. Because of uncertainties on the precise value of items on both cost and benefit sides of the analysis, it can be informative to look at what the effect would be of unspecified increases in either costs or benefits.

4.9.7 High costs and low benefits

Varying the costs and benefits we find that if cost doubles and benefits do not change, then there remains a net benefit under the 2013 deadline option. Costs could increase by more than 3.8 times before the BCR is driven to 1.0. Holding costs constant a halving of benefit would still yield BCR of 1.9. Benefits could reduce to 26% of their baseline value before BCR is driven to 1.0.

For 2020, if benefits remain the same, costs could be increased by almost 3.2 times before BCR is driven to 1.0. If costs remain the same, the benefits could be reduced to 32% of their baseline value before BCR is driven to 1.

If both change, we find breakeven points with benefits reduced to 52% and costs doubled for 2013, and when benefits reduced to 64% and cost doubled for 2020.

5. Conclusions

This report updates the initial cost benefit analysis of the National Environmental Standards on Air Quality that underpinned the 2004 CBA assessment for introducing the standards. Time and resources mean it relies on existing gathered material and is undertaken at a national level. Given limited information and general uncertainties around the relationships between air pollution and adverse health impacts, it should not be viewed as an accurate prediction of the impact of implementing the standards. It can however identify the main effects of the policy and define the range of uncertainties and put broad magnitudes around them, focusing on nationwide rather than localised effects. In doing so it can be informative of whether the standards' implementation is likely to result in costs greater than or less than the benefits.

This report reviewed the 2004 cost benefit analysis and identified some items that were missing or likely to be under-stated on both the benefits and the costs side. While there are omissions in the 2004 analysis, the size of the net benefit in that report was such that correcting omissions was unlikely to change the net benefit result, although the distribution of costs and benefits would be different.

In updating the analysis, this report compares the cost of nationwide achievement of the ambient air quality standard in 2013 or 2020, against a counterfactual of continuation of the current situation. That current situation includes the effects of some implementation of the NES that is unlikely to be reversed, such as the decommissioning of incinerators in schools and hospitals. Even if the NES were abandoned today, there would be some benefit derived from it over future years.

Many of the items in the original cost benefit analysis have been updated with the producer price index, but some have been amended with supplementary or new information. Major changes from the 2004 analysis arise from:

- Increasing the unit value per avoided air quality related death or hospitalisation, in line with values used in other safety related assessments in New Zealand
- Providing for costs borne by private householders in meeting the wood burner standard, which in the 2004 analysis were assumed covered by a subsidy scheme which in the event did not occur
- Taking account of the effect of clean heat subsidy schemes that have been put in place on the potential costs of compliance for affected households.

The result of the update is that, compared with the 2004 analysis, costs and benefits of achieving the standard by 2013 have each increased about threefold, with the net present value being about 3 times greater but the benefit cost ratio unchanged on the central set of assumptions. The margin between costs and benefits is large enough to accommodate substantial changes in either costs and benefits before the positive result would change to a negative one.

There remain uncertainties around some likely effects, such as the consequences of consenting restrictions in pursuit of ambient air quality, but these missing items would need to be very large to change the result. As it is, the cost imposed on households in the analysis is more extensive than the NES is likely to require, so full implementation of the standards is more likely than otherwise to be net beneficial.

In summary, the updated 2009 CBA differs from the initial cost benefit analysis underpinning the 2004 CBA report of the air quality NES in the following ways.

Table 15 Summary comparison of 2004 and 2009 cost benefit analyses

Item	Treatment in 2004 CBA	Treatment in updated CBA
Largest benefit is from reduction of premature death, hospitalisation and restricted activity days	Probably understated by discounting the Value of Statistical Life (VoSL) on assumption that most who benefit are elderly No allowance made for lingering effects on quality of life No explicit estimate of savings in hospitalisation and medical costs for restricted activity days	No adjustment of VoSL for assumed age of beneficiaries Quality of life omitted but considered in sensitivity analysis
Smaller benefit from reducing GDP losses	Size depends on assumed labour and capital substitution Provides for growth in GDP which should also increase VoSL	Item omitted as immaterial to result, and because of partially covered within VoSL (no double counting)
Largest cost item of NES is in administration and monitoring by local and central government	Probably overstated as estimates are in keeping with other NES CBAs, but regional councils have primary responsibility and have spent less than first assumed in the years since NES introduced	Set costs for territorial authorities to zero Set regional council costs on actual spending to date Update government costs by PPI
Costs to industry per emitting site	Average figure at low end of a potential range	Update with PPI and test variation with sensitivity analysis
Costs of replacing prohibited activities	Now outdated by phase out of road seal burning & institutions' incinerators	Set at zero cost
Cost to wood burner suppliers	Assumed to be negligible, set at zero	Set at zero, but may be some indeterminate costs not recovered in sales
Cost to government/ councils in subsidising wood burner upgrades	Possibly understated – implies low market penetration	Item omitted, as subsidy did not get enacted
Costs to households in upgrading wood burners	Set at zero	Item included, allowing for trends in home heating and substitution to other heating forms

Source: NZIER

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Appendix A Comparison of initial and updated cost benefit analyses

Table 16 summarises the differences between the initial and updated cost benefit analyses.

Table 16 Coverage of initial and updated analyses

	2004 Analysis	2009 Update
Benefits		
Willingness to pay to avoid		
-Premature loss of life (pain & suffering)	Included	Included
-Lost output/productivity/income		
Direct benefits of avoiding GDP loss	Included	
-Indirect benefits of avoiding GDP loss	Included	Not valued
Avoided costs of medical treatment	Not included	Included
Avoided loss of long term quality of life	Not included	Not valued
Costs associated with ambient air quality		
-Regional councils administration/monitoring	Included	Updated
-Territorial authorities administration	Included	Zero entry
-Government information & administration	Included	Updated
-Industry site adaptation measures	Included	Updated
-Business forgone from consent constraints	Not included	Not valued
Costs associated with prohibition standards		
-Consenting of school & hospital incinerators	Included	Completed
-Alternatives to tar seal burning	Included	Zero entry
-Other activities: landfills, wire burning etc	Zero entry	Zero entry
Costs associated with wood burner standard		
-Householders costs of compliant burners	Zero entry	Included
-Suppliers costs of compliant burners	Zero entry	Zero entry
-Government/council subsidy	Included	Zero entry
Factors taken into account in the analysis		
Infant mortality	Not included	Included
Cost of hospitalisation (medical expenditures)	Not included	Included
Discount rate	10%	8%
Influences on the counter-factual	Not included	Allows for downward trend in wood burners & insulation/ clean heat initiatives

Source: NZIER

The initial analysis had one principal benefit, the social willingness to pay value attached to reduction in premature death and hospitalisation (VOSL) and one much smaller benefit of avoided losses in GDP (direct and indirect, arising from deaths, hospitalisation and restricted activity days). These benefits understate the societal value of reductions in deaths and hospitalisations because they omit long term deterioration in quality of life and also omit the cost of medical treatment. The updated analysis includes the willingness to pay value (which includes income loss) and also medical treatments. It acknowledges but does not value long term quality of life and the indirect GDP benefits, which are small relative to VOSL.

The initial analysis had costs associated with standards for ambient air quality, prohibited activities and wood burners. Most of these are updated in the new analysis, except costs to schools and hospitals, which are no longer incurred, and the subsidy cost for wood burners. The new analysis values potential impacts on households for compliance with the wood burner standard. Potential for business forgone by consenting constraints cannot be valued without localised modelling.

Appendix B Responses to peer review comments

We discuss here the major comments and Our response to those comments. The comments refer to estimation of benefits in terms of physical consequences and their valuations and estimation of costs.

B.1 Benefits

Comment: The effects of are not just the effect of short term exposure but the effect of cumulative exposure to PM_{10} of over time. Hence, the benefit of reducing PM_{10} levels should be measured by improvement in life expectancy.

Response: There is merit in the argument that the health effects are the total effect of exposure over time. The standard is expected to keep the PM_{10} emission below a certain level. That should reduce the cumulative effect of the exposure over time. The estimates of health effects used in the report is based on earlier studies which used a methodology developed by Künzli et al (2000), which was based on cohort observations. The health effects were estimated as functions of annual average 24 hour maximum PM_{10} levels. The fatalities, as we understand, were linked to each year's level. It does not necessarily give the cumulative effect. That part needs a separate study.

We have discussed in detail the drawbacks of using the difference in life expectancy in Appendix C .

Comments: The estimate of VOSL used is based on risk reduction in traffic crashes. There are a few differences between this scenario and PM10 effects.

- There is a considerable time lag between exposure and time of death;
- The risk profile may not be the same and that may affect the VOSL for PM_{10} effects.

Response: It is possible that the VOSL used is on the lower side because of the longer time difference between the event and death, as people suffer more during this time (see Appendix C for more details).

True, the risk profile may be different. The current VOSL is an update of the value established in 1991 based on a survey carried out in 189/90. At that time annual road toll was varying between 650 an 750. We have not got the estimate of deaths from pollution during this time period. The estimate for 2003 (872) is higher. If the pollution related death was similar during 1989/91, then the base risk would be lower and that would possibly have some effect but not large enough to substantially affect the cost benefit relativity analysed in the paper.

Comments: There is a likelihood that the number of deaths are under estimated in the 2004 CBA and in this report as uses the same estimates under the status quo scenario, because all urban areas were not covered in the 2004 estimates.

Response: If higher levels of estimated number of deaths were used, the benefit cost ratio and the net benefit would be higher.

Comments: Some tables need better explanation.

Response: We have explained them further and also revised the table presentations for better understanding.

Appendix C Value of loss of life due to pollution

The value of statistical life (VOSL) used in the analysis is based on that used in transport project appraisals in New Zealand, which was derived from a willingness to pay (WTP) survey carried out in 1989/90 (Miller and Guria 1991), and updated since then by indexing it to the ordinary time wage rate (Ministry of Transport 2008).

Before 1991, project appraisals used value of life based on the so called "human capital approach", in which the value of a premature fatality avoided (life saved) is the present value of expected output forgone over the rest of life of the person. The WTP based VOSL is fundamentally different in approach, in being derived from the amount of money people are willing to pay to reduce their risk of death by a certain proportion. The WTP surveys probe in various ways how people change their behaviour in face of risk. It is very unlikely that respondents think about the number of life years left them or adjust their willingness to pay accordingly: not only do they not know when they would otherwise die but also they have to survive today to enjoy any more life beyond then. Surveys carried out to estimate this value lead people to think about the level of risk they face and the scope of reducing the risk and how much they would be willing to pay based on what they can afford, but there is no evidence their answers are framed in terms of a value per life year.

The value of statistical life does not purport to place a value on any identified, discreet individual's life. When subjecting a project to cost benefit analysis, the safety benefit is expressed as a reduction in the risk of accidental injury or death. A reduction in risk of death of 1 in 10,000 is equivalent to saying that one more person will be saved out of a group of 10,000 people. In that sense only is the VOSL used as an expression of the value of a risk reduction that is expected to save one life – the value of a statistical life saved.

Some researchers derive a value of life per year from the WTP based VOSL, in which they assume that the VOSL is the discounted present value of values of life years lost if the person died at that point. In our view it is a wrong approach since the VOSL is based on the value of risk reduction now, not securing years of life ahead. For similar reasons, characterizing the benefit of air quality improvement as an extension of life years before death at some distant point in the future will understate the value of risk reduction now: people do not know in advance when they are going to die with current air pollution and when with pollution reduced, so any expressed willingness to pay is for a generalized reduction in risk that is being incurred now.

The formula of Kunzli et al (2000), which is used to estimate the number of deaths in this study, is based on observed numbers of deaths in the 30 years and over age group. Many of these will not have died immediately after being exposed to high level of pollution, rather they endure respiratory conditions, heart disease and cancer for many years before death. The value of pain and suffering for them would be considerably higher than for those facing risk of instantaneous death. Studies find that people are more willing to pay to save a person from cancer death than a pedestrian death (Rowlatt et al 1998, Clinton et al 2007).

The social cost of pollution is the total loss to society if one person dies as a result of exposure to pollution, so the benefit of a pollution reduction policy is estimated by the expected number of lives saved as a result. Some of these lives would have been lost immediately after exposure (comparable to pedestrian death) but in many cases they would die after years of suffering. Compared to road crashes, in which only deaths which occur within 30 days of a crash are counted as road crash deaths, the loss to society of the average premature death from pollution is likely to be greater than that from the average road fatality.

Taking these factors into account, using the VOSL is more likely to under-state than over-state the cost to society of lives lost as a result of exposure to pollution.

C.1.1 The appropriateness of VOSL to air quality

A reviewer has questioned the relevance of using the value of statistical life (VOSL) derived from transport accident studies for application to mortality changes from air quality improvements, on grounds that such "benefit transfers" (i.e. using benefit estimates from one situation in analogous situations elsewhere) are subject to large discrepancies unless the populations at risk, and the risk profiles they face, are reasonably similar. While issue-specific valuations would be preferable, the survey-based studies to obtain them do not come cheap and no alternative value is currently available. The willingness to pay based value of statistical life is estimated from realistic risk and risk reduction, and respondents' willingness to pay to gain that risk reduction. There is no particular reason what the value in present context should be drastically different, other than the likely higher level of pain and suffering before death. In that case, the VOSL should be higher not lower.

The transport-related VOSL is the only such estimate used in official estimates in New Zealand, and it is widely used with due caution as a practical guide in situations removed from its origins in road transport, such as assessments of aviation and maritime safety. The international literature suggests that willingness to pay to reduce risk is lower in road transport than in situations where respondents feel they have less individual control – for instance in public transport, or when exposed to involuntary risk. For this and other reasons the current transport related VOSL is more likely than not to underestimate the public benefit of mortality reduction in these other settings.

In the case of air quality, the risk from exposure is largely involuntary and out of control of individuals, and may also result in long lingering illness rather than sudden death. Studies have found that people consider deaths from cancer or heart disease worse than pedestrian deaths which occur in most cases within a short time from the crash, and a similar aversion to chronic illnesses brought on by air pollution is likely to increase the public willingness to pay to reduce risk of exposure.

C.1.2 Allowing for delays in death

Only a fraction of total pollution-related deaths occur in the first year after exposure and the rest occur in subsequent years. Some argue that the Value of Statistical Life (VOSL) should be discounted to take into account the delay in death.

This argument is based on only part of the total effect. The VOSL is estimated from the amount people are willing to pay to reduce the risk of death. In transport crashes, the number of deaths is based on those dying within 30 days from the date of accident. There can be more premature deaths due to crashes but they occur after 30 days. In some cases, people potentially involved in them and society in general would be willing to pay more to reduce the risk of such deaths, because of the pain, grief and suffering they cause to the people killed prematurely and those close to them.

In case of pollution, people who die in the second year, third year or a later date suffer more than those who die within a short time from the time of exposure. Some of these are due to being affected by cancer, ischaemic heart disease and cerebro-vascular disease. In these cases, people suffer longer before death than do those involved in a traffic crash. If the value of this extra suffering were added, the VOSL for such cases would be much higher. This may be a reason why the EPA in the USA uses much higher VOSL than that used by the Department of Transport.

One of the peer reviewers suggested that only the change in life expectancy should be considered. This would drastically reduce the value of reducing the risk of pollution effects. It is undoubtedly the wrong approach.

- First the willingness to pay based VOSL does not proportionately vary with remaining life expectancy. The remaining life expectancy reduces with age. The two past studies in New Zealand did not find any relationship with age. Some European studies found an inverted U shaped relationship, i.e., increasing with age up to a certain age and then slowly reducing.
- Calculating the value of a life year by dividing VOSL by the life years forgone by the average premature fatality means the value of a life year is the same for all: this approach would indicate very low value of remaining life for the elderly and very high value for children. Although some people may think such differences in relative values are appropriate, empirical evidence on the amount people are willing to pay to reduce their risk of death does not give any such indication.

Consequently it is inappropriate to conceptualise the benefit of air quality improvements as adding life years at some point in the distant future, or of manipulating the VOSL as if that is what it shows. The VOSL reflects individuals' willingness to pay to reduce risk as a guide to society's willingness to pay to reduce risk through collective action that reduces that risk.

The discussion suggests that the use of VOSL estimated from traffic risk based studies does not over estimate the social cost of air pollution effects. In fact there is a likelihood that it underestimates the social cost.