The Cost-Effectiveness of Reductions in Dioxin Emissions to Air from Selected Sources

Economic Analysis for Section 32 of the Resource Management Act

A report prepared for the
Ministry for the Environment

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Executive summary

1 Introduction

This report is an economic analysis of some technical options for reducing emissions of dioxins to air. Section 32 of the Resource Management Act 1991 requires that regard be taken of (economic) efficiency and effectiveness in the choice of policy instruments. This report provides a basis for a Section 32 analysis.

2 The benefits and costs of reduction of dioxin emissions

Reduction of emissions of dioxins to air can be expected to yield human health benefits because dioxins are toxic, persistent, and bioaccumulative. Because non-tariff barriers to trade are increasingly given an environmental rationale, economic benefits can also be expected from dioxin reduction by strengthening New Zealand's environmental comparative advantage. Any attempt to quantify health and economic benefits in dollars would be extremely speculative, so the benefits from each technical option considered have been expressed in terms of effectiveness in reducing dioxin emissions.

3 Context: Section 32 of the Resource Management Act

The analysis shows the variation in "bang for the buck" obtainable by different dioxin-reduction investments, thereby providing an important input for the design of efficient and effective policy instruments. The types of policy options available are: do nothing; monitoring and analysis; voluntary abatement; command and control; and economic instruments.

4 The methodology of cost-effectiveness analysis

The methodology used for the economic analysis is cost-effectiveness analysis (CEA). The cost-effectiveness of a technical option for dioxin control is expressed as a ratio of cost to effectiveness, measured in units of dollars per milligram of dioxin *not* emitted to air. The lower the ratio, the cheaper it is to reduce dioxin emissions by one milligram, and the more cost-effective the technical option is. Although levels of dioxin control are linked to technical options in order to assess their feasibility, there is no intention to prescribe dioxin control technologies.

5 Emission sources selected for analysis

The largest source of dioxin emissions to air is the uncontrolled burning of waste in landfill fires. This practice is so undesirable that these fires should be banned; an economic analysis of such a ban would be a waste of resources.

One domestic source of dioxin emissions to air – the backyard burning of household waste – was selected for analysis. The cost-effectiveness ratio of a ban on this undesirable activity is relatively low, indicating that such a ban would be relatively economically efficient.

Four significant industrial sources of dioxin emissions to air were selected for detailed analysis: waste incinerators, non-ferrous foundries, wood-fired boilers, and coal-fired boilers. Various engineering assumptions were made in order to model the effectiveness and cost of the technical options.

6 Analysis of waste incinerators

Two main options for reducing dioxin emissions from waste incinerators were analysed. The first involved increasing levels of dioxin control in incinerators. The second was the alternative of replacing incinerators with autoclave and grinding systems. Medical waste and municipal waste incinerators were analysed. The autoclave option is only feasible for medical waste disposal.

7 Analysis of non-ferrous foundries

Two levels of dioxin control were analysed for non-ferrous foundries.

8 Analysis of wood-fired boilers

Two levels of dioxin control were analysed for wood-fired boilers. Dioxin emissions from wood-fired boilers are much higher if the wood has been contaminated or treated. The effectiveness of dioxin controls therefore depends on the type and level of contamination in the wood burned.

9 Analysis of coal-fired boilers

Two main options for reducing dioxin emissions from coal-fired boilers were analysed: two levels of dioxin control, and switching the fuel from coal to gas.

10 Summary of results for industrial sources

Both cost-effectiveness and unit effectiveness (effectiveness per emitter) vary widely across technical options and across sources. Building dioxin-reducing technology into *new* systems appears to be only slightly less cost-effective than retrofitting it into *existing* systems, although there will be exceptions in practice. In most cases, there are strong economies of scale, with the cost-effectiveness of a technical option generally varying significantly with the *size* of the emitter.

11 National results for waste incinerators

The total national effectiveness of dioxin control options for existing waste incinerators has been estimated. This can be done because the number of waste incinerators of different sizes and levels of dioxin control is known. Two "supply curves of reduced dioxin emissions" for medical waste incinerators are presented. The first represents increasing levels of dioxin control in incinerators. The second represents replacement of incinerators with autoclave systems. The latter is as effective as the former, and far more cost-effective.

12 Policy implications

Nine implications for the development of policy instruments follow from an initial examination of the results.

1) Differences in cost-effectiveness between retrofitting abatement technologies and building them into new plants do not appear to be significant.

2) There is a strong case for an "aggressive" approach towards the incineration of medical waste.

3) A large municipal waste incinerator could be a major source of dioxin, and this must be a consideration if an application for such an incinerator is made.

4) More information about non-ferrous foundry sizes and emissions is required before policy instruments can be developed. There is an economic rationale for instruments that are sensitive to foundry size.

5) Economies of scale operate strongly for increasing levels of dioxin control in both wood-fired and coal-fired boilers. The economic case for dioxin reductions in boilers is generally much weaker than for incinerators or foundries.

6) The burning of contaminated wood in boilers is potentially a major concern, but not enough is known about the types of contamination and combustion practices. There is a case for data collection in this area.

7) Switching boilers from coal to gas is a relatively uneconomic environmental intervention if done only for dioxin emission reduction. However, there may be other motivations for such fuel switching such as reduction of carbon dioxide emissions.

8) Although there has been no explicit analysis of landfill fires in this report, the banning of landfill fires should be a high priority. This source of dioxin emissions to air is greater than any other by an order of magnitude, and there is no environmental justification for this practice.

9) Like landfill fires, the backyard burning of waste is a major source of dioxins. Discharges of dioxin to air from this undesirable practice should be prohibited, at least for certain types of waste.

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1 Introduction

Organochlorines are a group of environmental pollutants that are toxic, persistent and bioaccumulative. The dioxins,[1] polychlorinated biphenyls (PCBs), and pesticides such as DDT are organochlorines. In New Zealand, residues of persistent organochlorines can be found in our environment and food. For the dioxins, the margin of safety from the current lifetime level of human exposure and levels known to cause adverse health effects in animals is considered to be small (Smith and Lopipero, 2001).

This environmental health problem is being addressed at a national level in New Zealand through the Organochlorines Programme in the Ministry for the Environment. This programme was initiated in order to reduce industrial emissions of dioxins, clean up sites contaminated with organochlorine residues, and manage the safe disposal of waste stocks of organochlorine chemicals such as the PCBs and DDT. Further information on the Organochlorines Programme is available on the Ministry's web site.[2]

In New Zealand, the use of PCBs and organochlorine pesticides has largely ceased,[3] and so the main management task is one of cleaning up or securing reservoirs of these pollutants. There are also reservoirs of dioxins, but, in contrast with the other organochlorines, dioxins are still being created, and thus released for the first time to the environment. These "new" dioxins are created unintentionally as byproducts of other activities and are emitted to air, land and/or water. A review of the sources and reservoirs of dioxins in New Zealand has recently been published (Buckland *et al.*, 2000). The current report is concerned only with *emissions of dioxins to air*.

Some "new" PCBs can also be emitted from the same industrial processes as the dioxins, but because the pathways by which the dioxins and PCBs are formed in these processes are similar, control of dioxins will also control these new PCBs. Therefore, reduction of new PCBs is an additional unquantified benefit in this economic analysis. With some dioxin control technologies there would also be reductions in other pollutants. For example, the installation of fabric filter equipment on waste incinerators will substantially reduce emissions of airborne particulates.

Under the Organochlorines Programme, the Ministry for the Environment is exploring options for reducing dioxin emissions to air. As part of this process, Section 32 of the Resource Management Act 1991 (RMA) requires evaluation of the benefits and costs of alternative policy instruments, and that regard be taken of (economic) efficiency and effectiveness. This report provides a basis for a Section 32 analysis.

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1 In this report, "dioxins" is shorthand for all polychlorinated dibenzo-p-dioxin (PCDDs) and polychlorinated dibenzofuran (PCDFs). Dioxins are a family of 210 individual, structurally similar chemicals.

2 www.mfe.govt.nz/issues/waste/organo.htm.

3 PCBs are still being used in electrical units, although such use is illegal.

2 The benefits and costs of reduction of dioxin emissions

There are two categories of benefits that would come from the reduction of the rate of flow of dioxins to the environment – *health benefits* and *economic benefits*.[4]

Dioxins have been linked to a number of human health effects, including cancer, immune system suppression, and reproductive and developmental problems (Smith and Lopipero, 2001). Because dioxins do not break down in the environment and accumulate through the food chain, the benefits from reducing flows of "new" dioxins to the environment could, theoretically, continue as long as life persists on Earth. Dose-response relationships are not well understood, and health effects may well rise more than proportionally with exposure to dioxins. If this is so, health benefits would be expected to rise faster over time than population.

Can the health benefits from reducing dioxin emissions be expressed in dollars? It is possible to estimate the number of cases of cancer prevented due to a decrement of dioxin flow using epidemiological studies, although the confidence interval would be extremely wide. Attaching a dollar value to an averted cancer death can be done but there is no "right" answer; values used in the USA range over an order of magnitude. Attempts to monetise other health benefits from non-cancer effects would be even less successful.

Economic benefits from reducing the rate of flow of dioxins to the environment would come from the strengthening (or protection) of the clean green image used for marketing tourism and biologically based products in export markets. Dioxin emissions are low compared with other countries, because New Zealand is not heavily industrialised and population density is low (Buckland *et al*., 2000). The value of such environmental comparative advantages is significant. A loss in revenue from selected markets of up to $569 million for the dairy sector and $938 million for the inbound tourism sector has been estimated if New Zealand's environment was seen to be degraded (MfE, 2001).

Increasingly, non-tariff barriers to trade are environmental. Food containing residues can be blocked from export markets altogether, or sell at lower prices. No residue limits have yet been established for dioxins with respect to overseas trade. However, the CODEX Alimentarius Commission of the WHO and the UN has commenced discussions in this area because maximum levels for these chemicals may eventually be developed.[5] Marketing catastrophes – low probability, huge-damage events – are also a possibility. The recent discovery of dioxins in food exports from Belgium has had a dramatic effect on the economy of a country that does not market itself as "clean and green". The Belgian federal government estimate that the cost to their economy has been between 25 and 40 billion Belgian francs (NZ$1.3 to 2.1 billion) in the first nine months (Ministry of Agriculture and Ministry of Health, 1999).

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[4] Improvements in health do, of course, yield economic benefit by reducing expenditure on health care.

[5] CODEX papers CX/FAC 99/23 and CX/FAC 00/26.

Expression of the economic benefit in dollars of strengthening the country's clean green image on the basis of dioxin reduction would be a formidable task. However, it is prudent to consider reducing flows of dioxins to the environment to levels that are at least consistent with emission limits set in other developed countries.

Because of the difficulty of expressing the benefits of dioxin reduction in dollars, the method of economic evaluation used in this report is cost-effectiveness analysis (CEA). CEA does not require monetisation of benefits. Instead, benefits are expressed in terms of effectiveness, with effectiveness measured in units of reduced dioxin emissions.

3 Context: Section 32 of the Resource Management Act

Under Section 32 of the Resource Management Act, the effectiveness and efficiency of alternative means for reducing dioxin emissions are to be used as a basis for making choices about policy instruments.[6]

One choice is between *national* and *regional* policy instruments. Two points are pertinent from an economic perspective. First, the economic benefit of strengthening the country's clean green image would be undermined by variation across regions. Second, the bioaccumulative nature of dioxins, their persistence in the environment and the means by which people are exposed to these contaminants are such that emissions can have impacts well beyond regional boundaries. For example, people can be exposed to dioxin by drinking milk and eating meat from distant farms where dioxin discharged to air has been deposited on to pasture. Emission of dioxin to air is a national air pollution problem, in contrast with, for example, airborne particulates in Christchurch.[7]

Another choice concerns the *type* of policy instrument. In Europe and North America, the setting of allowable limits on concentrations of dioxins in gaseous emissions from industrial facilities has been chosen as the appropriate policy instrument. Nevertheless, using emission limits is not the only option. Types of policy instruments for reducing emissions of dioxin can be classified into five groups.

1) *Do nothing*. This must always be an option, and should be chosen if the value from reduction of dioxin emissions is considered to be not worth the cost. This choice may also be made if it is expected that dioxin reduction will occur from improved practices taking place for other reasons.

2) *Monitoring and analysis*. If there is great uncertainty about the amount of dioxin being discharged from a source, or little information on the size of the activity nationally, then information collection may be the best choice.

3) *Voluntary abatement*. This may be especially effective for sources where dioxin discharges can be reduced at relatively low cost. Greater progress may be achieved by the use of "carrots" such as green awards and "sticks" such as the threat of a standard.

4) *Command and control*. This may involve bans on certain activities, such as fires in landfills, or setting emission standards that will permit discharges up to a specified level. Standards could be limits on end-of-pipe dioxin emission rates, or limits on concentrations of dioxin in ambient air or other media. While the latter are unlikely to be the most practicable for

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6 See Tonkin and Taylor, 2000.

7 Note, however, that Regional Councils will be involved in the implementation of dioxin reduction. For example, national environmental standards for dioxin emissions to air would be enforced by Regional Councils, since they are responsible for controlling discharges to air under the RMA.

industrial point source dioxin emissions, they can have a role in controlling diffuse emission sources, including those encountered from domestic sources.

5) *Economic instruments*. Pollution (Pigovian) taxes and tradable pollution permits are the two classic economic instruments that could be used to control dioxin emissions. These instruments are, theoretically at least, economically efficient. Another economic instrument that could be considered for dioxin control is a "feebate" scheme. Under a feebate scheme, all those emitting dioxin above a certain level would pay a tax or fee for each unit of dioxin emitted. At the end of the year, the tax revenue would be redistributed as a rebate in proportion to the dioxin reductions achieved during the year.

Finally, there is a set of choices about details – allowable limits, size of taxes, sources to target, and so on. The economic analysis in this report provides a basis for making some of these choices.

For example, suppose that a set of national environmental standards for maximum dioxin concentrations in gaseous emissions from industrial sources is to be adopted. It does not follow that such environmental standards need be economically inefficient, because there is no *ex ante* reason why the same standard should be set for all sources of dioxin emissions to air. The standard could vary with the size of the emission source or the type of process (for example, a waste incinerator or a boiler). Standards will require industrial emitters to invest in reductions of dioxin emission rates. The economic analysis shows the variation in "bang for the buck" obtainable by dioxin reduction investments across different sources, and thus lays a basis for setting standards that will give the *desired effectiveness at the least cost*.

4 The methodology of cost-effectiveness analysis

In cost-effectiveness analysis, the benefit or effectiveness of an intervention is measured in "natural" or physical units. The result of a CEA is a cost-effectiveness ratio, and is analogous to a cost-benefit ratio.

The effectiveness of an abatement intervention in reducing the flow of dioxins to the environment is the decrement in the amount of dioxin emitted in a standard time period. The unit for measuring annual effectiveness has been chosen as thousandths of a gram of dioxins weighted for relative toxicity; that is, mg TEQ of dioxin.[8]

Consequently, the cost-effectiveness ratios calculated in this CEA are expressed in units of dollars per mg TEQ reduced, that is, *not* emitted to air. A low ratio signals a dioxin reduction bargain, since reductions of dioxin emissions can be "purchased" relatively cheaply.

This analysis is concerned solely with emissions of dioxins to air. The first task, therefore, is to select from the dioxin sources that emit to air those that are potential candidates for application of a policy instrument. On the basis of information presented in the New Zealand dioxin inventory (Buckland *et al.*, 2000), CEAs have been performed on a single domestic source: backyard burning; and four industrial sources: waste incinerators, non-ferrous foundries, wood-fired boilers, and coal-fired boilers. This choice is explained in Section 5 of this report.

The second task is to identify potential technology options for reducing dioxin from each source, and to estimate the effectiveness and cost of each option. For the industrial sources, the technology options considered in this report include those most commonly used overseas because they have been shown to be effective in reducing dioxin emissions from waste incinerators, foundries and boilers. This does not mean that technology choices to be made by firms will necessarily be prescribed. However, specific technology scenarios must be selected to estimate the costs of achieving different levels of effectiveness. Note that the costs in the CEAs are engineering costs only. The costs of establishing and operating policy instruments are not estimated in this report, and should be assessed in a separate exercise.

The third task is to divide each source into different kinds of dioxin emitters. All waste incinerators, for instance, are not alike and will vary in ways that will affect both the cost and effectiveness of different technology options.

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8 The use of toxic equivalents (TEQ) is an internationally adopted procedure for assessing the combined toxicity of a mix of different dioxins (see Smith and Lopipero, 2001).

We can expect the *size* of an emitter to matter. Economies of scale will probably make technology options relatively cheaper in large emitters. Whether the technology options are retrofitted to an *existing* system or built into a *new* system might also matter – costs are likely to be lower in the latter case.

The usual economic evaluation problem of the *timing* of effectiveness and costs is present. Effectiveness is measured as an *annual* reduction. Costs have two components – an *initial* capital investment, followed by *annual* operating costs.

There are two ways of dealing with this timing problem. The first is to annualise the capital cost by using a capital recovery factor. The second is to calculate the present value of both streams of effectiveness and cost. The latter has been used because it is more widely used and understood.

A discount rate is required to reflect the effect of timing on the *present value* of both costs and effectiveness. A real discount rate of 10% has been used. Since no monetary value is being placed on a unit of effectiveness and the time profiles of costs are similar across the four sources, the choice of discount rate is relatively unimportant.

Amortisation periods for different technology options are also required. A 10-year amortisation period has been assumed for all technology options. All the technology options would last at least 10 years, and would be likely to be resaleable when remaining lifetimes of plants are shorter than this.

For an economically correct comparison of technology options, the analysis must be *incremental*. In calculating a cost-effectiveness ratio for a particular technology option, both the cost and effectiveness are the increments in cost and effectiveness beyond the preceding technology option.[9]

The data used in the analyses and notes on sources and assumptions are presented in the appendix. Inevitably, there is a great deal of uncertainty in the estimates of costs and effectiveness. For instance, the percentage reduction in the rate of emission of dioxins from a medium-sized incinerator due to good combustion practice and quench cooling of flue gas is estimated to vary between 50% and 95%. Where a range rather than a point estimate has been supplied, the *midpoint* of the range has been used.

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[9] Incremental analysis is the practical expression of marginal analysis. In neoclassical economic theory, the focus is on the marginal benefit from expenditure of the marginal dollar. In this analysis, technology options are "lumpy", and additional costs occur in large increments, not as single dollars on the margin.

4.1 A hypothetical example

A calculation using imaginary numbers is now presented to illustrate the methodology of cost-effectiveness analysis.

Consider an uncontrolled dioxin emitter that currently produces 10 mg TEQ per year. Suppose two technology options (Option A and Option B) for reducing the dioxin emissions exist. Option B is an add-on to Option A. There are thus three choices – the uncontrolled status quo, Option A, or Option A *and* Option B.

In Table 4.1, the costs (capital and operating) and effectiveness of the two technology options are presented in the upper section, and converted to incremental form in the lower section. In Table 4.2, the 10-year time streams of incremental cost and incremental effectiveness are listed for each option in both undiscounted and discounted form. In year t, the discounted form of cost or effectiveness is calculated by dividing the undiscounted form by $(1 + d)^t$, where d is the discount rate.

For example, in year 5, the cost of option A is simply the operating cost of $70,000. The discounted form of this cost is:

$$70{,}000 \;/\; (1 + 0.10)^5 = \$43{,}500$$

The present value of the cost of an option is the sum of the stream of discounted costs. The present value of effectiveness is calculated similarly.

Cost-effectiveness ratio for Option A compared with the uncontrolled status quo:

= 623,000 / 33.8 = $18,400 per mg TEQ reduced

Cost-effectiveness ratio for Option B compared with Option A:

= 1,129,000 / 27.0 = $41,800 per mg TEQ reduced

The results of an incremental analysis should be interpreted carefully.

Option A would reduce emissions from 10 mg TEQ per year to 5 mg TEQ per year. Each mg TEQ reduction would cost $18,400. The addition of Option B would reduce emissions further from 5 mg TEQ per year to 1 mg TEQ per year. Each of these mg TEQ reductions would cost $41,800.

Table 4.1 Illustrative CEA: effectiveness and costs for the two technology options

	Capital cost ($'000s)	**Operating cost** ($'000s/year)	**Emission rate** (mg TEQ/year)
Uncontrolled			10
Option A only	150	70	5
Options A and B	400	200	1
	Incremental capital cost ($'000s)	**Incremental operating cost** ($'000s/year)	**Incremental effectiveness** (mg TEQ/year)
Option A	150	70	5 (10–5)
Option B	250 (400–150)	130 (200–70)	4 (5–1)

Table 4.2 Illustrative CEA: present value calculation

Year	**Cost stream** ($'000s)	**Discounted cost** ($'000s)	**Effectiveness stream** (mg TEQ)	**Discounted effectiveness** (mg TEQ)
Option A				
0	150 + 70 = 220	220	5	5.00
1	70	63	5	4.55
2	70	58	5	4.13
3	70	53	5	3.76
4	70	48	5	3.42
5	70	43	5	3.10
6	70	39	5	2.82
7	70	36	5	2.57
8	70	33	5	2.33
9	70	30	5	2.12
Present value	–	623	–	33.80
Option B				
0	250 + 130 = 380	380	4	4.00
1	130	118	4	3.64
2	130	107	4	3.31
3	130	98	4	3.00
4	130	89	4	2.73
5	130	81	4	2.48
6	130	73	4	2.26
7	130	67	4	2.05
8	130	61	4	1.87
9	130	55	4	1.70
Present value	–	1,129	–	27.04

In some of the analyses in this report, reference is made to "*dominated alternatives*". In cost-effectiveness methodology, an option is described as strongly dominated if it has a higher cost and a lower effectiveness than another option. An option that is not strongly dominated, but still has a higher incremental cost-effectiveness ratio than a more effective option, is described as weakly dominated. Both types of dominated options should be discarded, and the incremental cost-effectiveness ratios of the remaining options recalculated.[10]

In some instances, an alternative to the installation of a sequence of dioxin controlling technology options exists. For example, consider the disposal of medical waste. A sequence of technology options can be used to successively reduce dioxin emissions from medical waste incinerators. However, there is an alternative to using an incinerator to dispose of medical waste: an autoclave system in which the medical waste is ground up finely, sterilised, and the residue deposited in a landfill. This alternative produces virtually no dioxins. Such alternatives can also be analysed in a CEA framework.

[10] See Gold *et al.*, 1996, pp.285–7 for a fuller discussion of the exclusion of dominated options from a CEA.

5 Emission sources selected for analysis

The sources of dioxin emissions to air in New Zealand are estimated in the Ministry for the Environment's dioxin inventory (Buckland *et al.*, 2000). In 1998, a total of between 14,000 and 51,000 mg TEQ of dioxin was emitted to air.

The largest single source by an order of magnitude is landfill fires. While these fires reduce the volume of waste, it is considered the potential environmental damage resulting from the emission of products of incomplete combustion far outweighs the environmental value they contribute. Thus, there is no need to perform an economic analysis on this source of dioxins: these fires should simply be banned.

Some sources emit negligible amounts of dioxins. Other sources are simply not amenable to policy instruments aimed at reducing dioxin emissions; forest fires and cigarette smoking are two examples.

Three significant sources of dioxin emissions are domestic: the burning of coal, wood, and waste. However, their disaggregated nature (many thousands of fires that are individually small though collectively large) make them difficult to target. Nonetheless, all practicable options for reducing dioxin emissions from the domestic sector should be investigated.

Of the three significant domestic sources, dioxin concentrations are highest in combustion gases from the burning of *household* waste. Moreover, the backyard burning of waste generates other air pollutants and is a fire risk, so there are other reasons for discouraging this practice.

Combustion gases from uncontrolled burning of household waste emit roughly the same amount of dioxins for the same quantity of waste as that burned in a landfill fire (UNEP, 2001). The US EPA has reported an emission factor of 300 microgram TEQ per tonne for the burning of mixed domestic waste in barrels (Lemieux, 1997). In New Zealand, the domestic burning of household waste is estimated to emit 5,200 mg TEQ per year (Buckland *et al.*, 2000).

Waste minimisation, recycling and landfilling are all preferable to the backyard burning of household waste. Since the concentration of dioxin in the combustion gases from such uncontrolled burning cannot be reduced, the only option is to ban the discharges of dioxin to air from waste burning; in practical terms, this amounts to a ban on the activity.

Calculation of the cost-effectiveness of a ban on the backyard burning of waste is relatively trivial and is shown in Box 5.1.

Box 5.1. Cost-effectiveness of a ban on the backyard burning of waste

A total of about 58,000 tonnes of domestic waste is burned in backyards each year in New Zealand. Of this 70% is organic waste (green waste and putrescibles), and 30% "household" waste (paper and plastics etc). For assessing the cost-effectiveness of a ban on the backyard burning of waste, the following assumptions are made:

- green waste is not covered by the ban
- there is no increase in recycling and waste reduction initiatives by households as a result of the ban
- the previously burned waste goes to landfills
- standard 60 litre 'bin bags' are used, each holding about 5 kg of waste, and kerbside collection is available
- the cost of a bin bag is $1
- there are no additional transport costs incurred by a household
- 300 microgram TEQ of dioxin is emitted from each tonne of waste burned.

For each additional bag of waste taken to landfill, the amount of dioxin emitted to air is reduced by 1.5 microgram TEQ. This gives a cost-effectiveness ratio of $670 per mg TEQ reduced. This is low compared with most of the cost effectiveness ratios estimated for the selected industrial sources considered in Sections 6 to 9 of this report, indicating that such a ban would be relatively economically efficient.

If the ban led to significant increases in the volume of waste taken to landfills, and the need for new landfills, the cost of disposal would rise. However, the amount of household waste burned in backyards is only about 1% of the total amount of domestic solid waste landfilled in New Zealand each year. If the ban were 100% successful, the amount of dioxin emitted to air each year would fall by about 5,200 mg TEQ.

In rural areas where no kerbside collection is available and householders have to travel some distance to landfills, the cost of landfill disposal would be higher.

In the remainder of this paper, economic analysis is used to assess technology options for four significant industrial sources of dioxin emissions to air. In order of priority, these are:

- waste incinerators
- non-ferrous foundries
- wood-fired boilers
- coal-fired boilers.

The analysis is based on a number of hypothetical case studies. The engineering assumptions used in modelling effectiveness and costs of technology options for these case studies are summarised below. More detailed information is provided in the appendix.

Table 5.1 Ranges of total annual dioxin emissions from the four prioritised industrial sources

Source		Annual emission to air (mg TEQ/year)
Waste incinerators	Clinical, pathological, and quarantine waste incineration	380–3500
Non-ferrous foundries	Non-ferrous metal production	100–1300
	Secondary aluminium production	9.1–1800
Wood-fired boilers	Industrial wood combustion:	
	• wood-processing wastes	280–1200
	• contaminated wood wastes	570–1200
Coal-fired boilers	Industrial, commercial, and agricultural coal combustion:	
	• industrial and commercial appliances	32–3800
	• agricultural appliances	1.7–200
	• power generation	59–110

5.1 Waste incinerators

- Operation: 10 hours per day, six days per week totalling approximately 3000 hours per year.
- Small, medium and large waste incinerators burn medical and/or pathological and quarantine wastes (hereafter referred to as medical waste incinerators). Small waste incinerators burn 50 to 100 kg per hour, medium waste incinerators burn 300 to 500 kg per hour, and large waste incinerators burn 1000 kg per hour.
- Very large waste incinerators burn municipal waste at a rate of greater than 10,000 kg per hour as a mass-burn unit. These units can be modular in design, and a municipal waste incineration facility can consist of more than one module. Currently there are no municipal waste incinerators operating in New Zealand. However, in the last three years, there have been several proposals to establish municipal waste incineration as an alternative to landfills, and therefore a CEA for municipal waste incinerators has been performed.

5.2 Non-ferrous foundries

- Operation: 10 hours per day, six days per week totalling approximately 3000 hours per year.
- Processing capacity is 1 tonne per day for small foundries, 15 to 20 tonnes per day for medium foundries, and 30 to 40 tonnes per day for large foundries.
- Small and medium foundries ventilate one furnace and working area; large foundries ventilate two 1-tonne per hour furnaces and associated working areas.

5.3 Wood-fired boilers

- Operation: 24 hours per day, seven days per week totalling approximately 7000 hours per year.
- Gross heat output is 1 MW for small boilers, 5 MW for medium boilers, and 30 MW for large boilers.
- Units are fitted with multi-cyclone grit arrestors.
- Modern suspension grate burning of untreated wood is a first case, and contaminated wood[11] is a second case.

5.4 Coal-fired boilers

- Operation: 24 hours per day, seven days per week totalling approximately 7000 hours per year.
- Gross heat output is 1 MW for small boilers, 5 MW for medium boilers, and 30 MW for large boilers.
- Units are fitted with multi-cyclone grit arrestors.

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11 This includes processed timber products such as plywood and particle board, and wood that may contain organic compounds or metals as a result of treatment with wood-preservatives or coatings. For CEA, consideration is given only to modern day timber treatment chemicals. It does not consider historical pesticides such as pentachlorophenol, which has been voluntarily withdrawn from use by the timber industry and which was deregistered by the Pesticides Board in 1991.

6 Analysis of waste incinerators

There are two main options for reducing dioxin in the flue gas of waste incinerators.

The first main option involves different levels of control using potential technical changes to waste incinerators, namely:

- good combustion practice: primarily ensuring sufficient temperature, residence time and oxygen levels in the final combustion chamber to destroy dioxins
- quench cooling: rapid cooling of flue gases through the critical temperature zone to prevent *de novo* synthesis
- fabric filter: very high-efficiency particulate removal from the flue gases
- carbon injection: adsorption of dioxins on to powdered activated carbon with subsequent collection in a fabric filter
- reheat and catalyst: destruction of dioxins in a low-temperature monolith catalyst. This requires pre-cleaning of flue gases by fabric filtration followed by reheat to achieve optimal destruction temperatures.

Four types of waste incinerators are considered as candidates for these different levels of control: small, medium, and large medical waste incinerators, and a modular municipal waste incinerator used as an alternative to landfills.

The second main option involves using an alternative technology for the task of disposing of medical waste, namely an autoclave and grinding system. A New Zealand company has already replaced one of its medical waste incinerators with an autoclave unit, and is considering similar replacements for its other waste incinerators. This option is only appropriate for medical waste. The current alternative to municipal waste incineration in New Zealand is to continue landfilling.

Thus, there are two CEAs in this section. The first is concerned with the different levels of control for both medical and municipal waste incinerators. The second is concerned with replacement of medical waste incinerators with autoclave systems.

The data on effectiveness and cost of the technology options are taken from Sections A.1 through A.4 in the appendix.

6.1 Different levels of control

Tables 6.1 through 6.3 contain effectiveness and cost data for small, medium and large medical waste incinerators, and Tables 6.6 and 6.7 contain effectiveness and cost data for municipal waste incinerators. The amount of dioxin emitted annually at each level of control (Tables 6.1 and 6.6) is calculated from the midpoint of the performance data given in Tables A.1 to A.4 (appendix). Cost effectiveness ratios for medical waste incinerators and for new[12] municipal waste incinerators are reported in Tables 6.5 and 6.8 respectively.

Table 6.1 Medical waste incinerators: annual emissions of dioxins at different levels of control

	Dioxin emissions (mg TEQ/year)		
	Small	**Medium**	**Large**
Uncontrolled	200	–	–
Good combustion practice	26	200	400
+ quench cooling	14	55	110
+ fabric filter	5.3	21	42
+ carbon injection	0.19	0.75	1.5
+ reheat and catalyst	0.13	0.5	1.0

Note: Existing medium and large waste incinerators are not uncontrolled.

Table 6.2 Medical waste incinerators: total installed costs for different levels of control

	Total installed costs[13] ($'000s)					
	Small		**Medium**		**Large**	
	Existing (retrofit)	**New**	**Existing** (retrofit)	**New**	**Existing** (retrofit)	**New**
Good combustion practice	70	–	–	–	–	–
+ quench cooling	130	50	100	90	140	120
+ fabric filter	220	120	520	410	570	450
+ carbon injection	Not realistic		610	490	690	550
+ reheat and catalyst	Not realistic		1,300	1,100	1,800	1,400

Table 6.3 Medical waste incinerators: operating costs for different levels of control

	Operating costs ($'000s/year)			
	Small		**Medium**	**Large**
	Existing	**New**		
Good combustion practice	10	–	–	–
+ quench cooling	40	30	60	80
+ fabric filter	120	110	160	190
+ carbon injection	Not realistic		180	220
+ reheat and catalyst	Not realistic		210	260

12 CE ratios are calculated only for *new* municipal waste incinerators because there are no municipal waste incinerators operating in New Zealand.

13 Total installed costs are equipment capital costs plus installation costs.

Table 6.4 Medical waste incinerators: cost-effectiveness ratios for different levels of control[14]

	CE ratios ($/mg TEQ reduced)					
	Small		**Medium**		**Large**	
	Existing (retrofit)	**New**	**Existing** (retrofit)	**New**	**Existing** (retrofit)	**New**
Good combustion practice	117	–	–	–	–	–
+ quench cooling	3,110	2,990	520	505	347	337
+ fabric filter	11,000	10,630	4,770	4,330	2,550	2,340
+ carbon injection	Not realistic		1,650	1,570	1,180	1,110
+ reheat and catalyst	Not realistic		528,000	481,000	426,000	345,000

Table 6.4 shows that, for medium and large incinerators, the incremental CE ratios for the second technical option are larger than those for the third technical option. This indicates that the second option is dominated and should be discarded. In other words, a fabric filter alone is less cost-effective than a fabric filter plus carbon injection. Table 6.5 gives the CE ratios with the dominated alternatives – a fabric filter alone in medium and large incinerators – discarded.

Table 6.5 Medical waste incinerators: cost-effectiveness ratios for different levels of control with dominated alternatives discarded

	CE ratios ($/mg TEQ reduced)					
	Small		**Medium**		**Large**	
	Existing (retrofit)	**New**	**Existing** (retrofit)	**New**	**Existing** (retrofit)	**New**
Good combustion practice	117	–	–	–	–	–
+ quench cooling	3,110	2,990	520	505	347	337
+ fabric filter	11,000	10,630	–	–	–	–
+ fabric filter & carbon injection	Not realistic		3,600	3,300	2,040	1,880
+ reheat and catalyst	Not realistic		528,000	481,000	426,000	345,000

Table 6.6 Municipal waste incinerators: annual emissions of dioxins at different levels of control

	Dioxin emissions (mg TEQ/year)
Good combustion practice (baseline)	4700
Fabric filter	990
+ carbon injection	35
+ reheat and catalyst	23

14 The CE ratios in this section and in Sections 7, 8, and 9 are expressed to three significant figures. This has been done to show the difference between the estimates of the cost-effectiveness of dioxin control options retrofitted to existing plant and the cost-effectiveness of dioxin control options built into new plant. It should not be taken as an indication of accuracy.

Table 6.7 Municipal waste incinerators: total installed costs and operating costs for different levels of control

	Total installed costs cost ($'000s)	**Operating cost** ($'000s/year)
Fabric filter	2,100	480
+ carbon injection	2,600	790
+ reheat and catalyst	7,000	1,300

Table 6.8 Municipal waste incinerators: cost-effectiveness ratios for different levels of control

	CE ratios ($/mg TEQ reduced)
Fabric filter	210
+ carbon injection	400
+ reheat and catalyst	98,000

6.2 Alternative technology: replace with autoclave system

The annual effectiveness is the difference between the quantity of dioxins that would be emitted from a new incinerator and the alternative autoclave system. This has been taken as the annual emissions of dioxins from a new incinerator, since the autoclave system would emit virtually no dioxins to air (ignoring steam boiler discharge), although it would add a smaller quantity to landfill.[15]

Total installed costs for autoclave systems are less than or equal to those for new waste incinerators. Operating costs for autoclave systems are also likely to be similar to those for incinerators. It follows that there is no need to calculate cost-effectiveness ratios for the choice of an autoclave system instead of a *new* incinerator when an incinerator is at the end of its life. Given the assumptions about costs, such a choice would be *cost-saving*.[16]

An *existing* incinerator may be replaced with an autoclave system before it reaches the end of its life.[17] This is termed *premature replacement*. The cost of such a premature replacement is the difference between decommissioning and replacement now, and decommissioning and replacement at the end of the incinerator's useful life. The cost of bringing expenditure forward in time is dealt with by discounting appropriately.

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15 It has been argued that when a full dioxin generation life cycle is considered, the overall dioxin emissions from an autoclave can be higher than a state-of-the-art waste incinerator. A life cycle analysis would include the dioxin emitted during transport of additional solid waste residues from an autoclave to landfill and the additional dioxin in landfill gas. In the New Zealand context, this argument would not hold because none of the small or medium waste incinerators are state-of-the-art, and, of those tested, none achieve the European emission standard for dioxins.

16 One reviewer has suggested that the operating costs of an autoclave system may be 20% higher than the operating costs of an equivalent waste incinerator, but an estimate of this difference in dollars has not been possible. Thus, the economic case for replacement of medical waste incinerators with autoclave systems may be overstated.

17 Not all medical/pathological and quarantine waste can be treated by autoclaving (e.g. ship's dunnage). Consequently, waste incineration, or an alternative technology capable of treating such waste, will always be necessary.

Table 6.8 Medical waste incinerators: data required for calculating analysis of premature replacement of existing medical waste incinerators with autoclave systems

	Small	Medium	Large
Dioxin emitted by autoclave system (mg TEQ/year)	0.017	0.083	0.17
Cost of autoclave system *plus* decommissioning cost of incinerator ($'000s)[18]	400	3,200	6,300
Average remaining incinerator lifetime (years)	5	7	10

The cost-effectiveness of premature replacement by an autoclave system depends on the level of dioxin control in the incinerator replaced. In New Zealand currently:

- the 19 small medical waste incinerators are either uncontrolled or have good combustion practice
- the two medium medical waste incinerators have good combustion practice
- the one large medical waste incinerator has good combustion practice, quench cooling, a fabric filter, and carbon injection.

Table 6.9 Medical waste incinerators: cost-effectiveness ratios for premature replacement of existing waste incinerators by autoclave systems

	CE ratios ($/mg TEQ reduced)		
	Small	**Medium**	**Large**
Early replacement of incinerator with autoclave *compared with*:			
Uncontrolled	110	–	–
Good combustion practice	860	1,200	1,400
Good combustion practice + quench cooling + fabric filter + carbon injection	–	–	280,000

18 The cost for an autoclave system to replace a small incinerator is relatively low because it does not include a boiler. Small autoclaves (less than 100kg/hour) would generally be based on hospital sites where steam is likely to be available. Medium and large units will need a boiler because these are likely to be centralised disposal operations.

7 Analysis of non-ferrous foundries

Two levels of dioxin control can be used for non-ferrous foundries. The first level is achieved by adding a fabric filter (or baghouse). The second is achieved by using carbon injection as well as a fabric filter.

The data on effectiveness and cost of the technology options are taken from Sections A.6 through A.8 in the appendix. The amount of dioxin emitted annually at each level of control (Table 7.1) is calculated from the mid-point of the performance data given in Tables A.5 to A.7 (appendix).

Table 7.1 Non-ferrous foundries: annual emissions of dioxins at different levels of control

	Dioxin emissions (mg TEQ/year)		
	Small	**Medium**	**Large**
No emission control (baseline)	10	200	400
Fabric filter	5.8	120	230
+ carbon injection	1.9	38	75

Table 7.2 Non-ferrous foundries: total installed costs for different levels of control

	Total installed costs ($'000s/year)					
	Small		**Medium**		**Large**	
	Existing (retrofit)	**New**	**Existing** (retrofit)	**New**	**Existing** (retrofit)	**New**
Fabric filter	270	210	500	390	1,000	770
+ carbon injection	330	260	580	460	2,200	1,700

Table 7.3 Non-ferrous foundries: operating costs for different levels of control

	Operating costs ($'000s)		
	Small	**Medium**	**Large**
Fabric filter	99	130	350
+ carbon injection	120	180	580

Table 7.4 Non-ferrous foundries: cost-effectiveness ratios for different levels of control

	CE ratios ($/mg TEQ reduced)					
	Small		**Medium**		**Large**	
	Existing (retrofit)	**New**	**Existing** (retrofit)	**New**	**Existing** (retrofit)	**New**
Fabric filter	32,700	30,600	2,400	2,210	2,930	2,730
+ carbon injection	7,710	7,330	800	780	2,630	2,370
Fabric filter + carbon injection	20,800	19,500	1,640	1,530	2,790	2,560

For all three foundry sizes, the fabric filter option is dominated by carbon injection and should be discarded. A fabric filter alone is less cost-effective than a fabric filter plus carbon injection, especially for small foundries. In the lower section of Table 7.4, the CE ratio for a fabric filter and carbon injection together is given. However, carbon injection is not yet used in foundries, so the effectiveness of this technical option is uncertain.

These results show strong economies of scale for dioxin control in medium and large foundries compared with small foundries. This is because the small foundry is assumed to be only 1/15th the size of the medium foundry, whereas the medium foundry is half the size of the large foundry. It is assumed that the control equipment would be designed to accommodate working space ventilation air. For a small foundry, a greater proportion of the total gas treated in the control equipment is likely to be relatively clean ventilation air compared to medium and large foundries. Thus the size (and cost) of control equipment is not assumed to be proportional to the size of the process.

8 Analysis of wood-fired boilers

Dioxin emissions from wood-fired boilers are much higher if the wood being burned has been contaminated or chemically treated. The literature suggests that dioxin emissions from burning contaminated wood are approximately 10 times greater than from burning virgin wood waste. Unfortunately there is no clear definition of what constitutes "contaminated wood". While there appears to be a correlation of higher dioxin emissions when wood treated with pentachlorophenol (PCP) is burned, the influence of other materials such as resins and chemical additives in processed timber products (such as particle board or plywood), or modern day timber treatment chemicals, are unknown (HMIP, 1995).[19] Overall, about 10% of the wood burned in industrial boilers is thought to be contaminated (Buckland *et al*., 2000).

The first analysis in this section is of wood-fired boilers burning *virgin* wood waste. Two levels of dioxin control are considered. The first level is achieved by adding a fabric filter (or baghouse). The second is achieved using a catalytic fabric filter; this involves replacing the filters in the baghouse with others impregnated with a catalyst capable of destroying dioxins. Data and results are given in Tables 8.1 through 8.4.

The second analysis is of wood-fired boilers burning 100% *contaminated* wood waste. Two levels of dioxin control are considered as in the first analysis. The results are given in Table 8.5.

The data on effectiveness (Table 8.1) and cost of the various technology options (Tables 8.2 and 8.3) are taken from Sections A.10 through A.12 in the appendix.

Since emissions from burning contaminated wood may contain comparatively elevated dioxin levels, stopping the burning of contaminated wood is an intervention that should be considered. However, more information is needed before this intervention can be analysed.

First, information is needed on the reasons why different industries burn contaminated wood. In some cases, contaminated wood may be burned simply as a convenient means of disposal; in others, it may be used as a fuel as well. In both situations, ceasing to burn contaminated wood would incur environmental costs associated with transport to the landfill and by reducing the landfill capacity and lifetime available for other waste streams. If the contaminated wood is being used as a fuel, it may be that the only substitute fuel available is "environmentally unfriendly" coal.

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19 Burning wood contaminated with PCP may result in dioxin emissions up to a thousand times higher than burning virgin wood, wood processing wastes or other chemically treated wood wastes. Since PCP is no longer used as a timber treatment chemical in New Zealand, a CEA of dioxin reductions from the burning of PCP treated timber has not been undertaken. Nevertheless, the potential for highly elevated dioxin emissions from burning PCP treated timber and PCP contaminated wood wastes is such that these wastes should not be burned as an industrial fuel in wood fired boilers.

Second, information is needed on the different types of contamination in wood waste burned by different industries. Contaminated wood waste that does not contain chlorine or other materials closely linked with dioxin emissions should be less of a concern.

Third, information is needed on typical proportions of contaminated wood burned in different industries. Although the total proportion of contaminated wood waste burned nationally is estimated to be about 10% (Buckland *et al*., 2000), wood-fired boilers in some industries may typically burn only contaminated wood waste, and others may typically burn none.

8.1 Different levels of control: virgin wood waste

Table 8.1 Wood-fired boilers: virgin wood waste – annual emissions of dioxins at different levels of control

	Dioxin emissions (mg TEQ/year)		
	Small	**Medium**	**Large**
Multi-cyclone (baseline)	3	15	80
Fabric filter	2.2	11	60
+ catalytic fabric filter	1.2	5.8	31

Table 8.2 Wood-fired boilers: virgin wood waste – total installed costs for different levels of control

	Total installed costs ($'000s)					
	Small		**Medium**		**Large**	
	Existing (retrofit)	**New**	**Existing** (retrofit)	**New**	**Existing** (retrofit)	**New**
Fabric filter	160	120	380	290	1,300	970
+ catalytic fabric filter	240	200	760	670	3,100	2,800

Table 8.3 Wood-fired boilers: virgin wood waste – operating costs for different levels of control

	Operating costs ($'000s/year)		
	Small	**Medium**	**Large**
Fabric filter	170	200	330
+ catalytic fabric filter	210	360	800

Table 8.4 Wood-fired boilers: virgin wood waste – cost-effectiveness ratios for different levels of control

	CE ratios ($/mg TEQ reduced)					
	Small		**Medium**		**Large**	
	Existing (retrofit)	**New**	**Existing** (retrofit)	**New**	**Existing** (retrofit)	**New**
Fabric filter	258,000	250,000	69,200	65,700	26,100	23,700
+ catalytic fabric filter	48,400	48,400	39,300	39,300	25,200	25,400
Fabric filter + catalytic fabric filter	135,000	132,000	51,400	49,900	25,600	–

The incremental CE ratios for the first technical option are larger than those for the second technical option for most cases, indicating that the first option is dominated and should be discarded. A fabric filter alone is less cost-effective than a fabric filter using "bags" impregnated with a catalyst, although the difference is not significant for retrofitting large wood-fired boilers. However, catalytic fabric filters have not been used in New Zealand and their effectiveness in wood-fired boilers is unproven.

8.2 Different levels of control: 100% contaminated wood waste

The CEA for different levels of control on wood-fired boilers burning 100% contaminated wood waste is trivial. It is assumed that dioxin emissions from burning contaminated wood are approximately 10 times greater than from burning virgin wood waste. It follows that the effectiveness of controls will be 10 times greater, and that corresponding CE ratios will be 10 times smaller.

Thus, the CE ratios for different levels of control on the burning of contaminated wood can be estimated by simply scaling the results for virgin wood waste in Table 8.4. This is presented for completeness in Table 8.5.

Table 8.5 Wood-fired boilers: 100% contaminated wood waste – cost-effectiveness ratios for different levels of control

	CE ratios ($/mg TEQ reduced)					
	Small		**Medium**		**Large**	
	Existing (retrofit)	**New**	**Existing** (retrofit)	**New**	**Existing** (retrofit)	**New**
Fabric filter	25,800	25,000	6,920	6,570	2,610	2,370
+ catalytic fabric filter	4,840	4,840	3,930	3,930	2,520	2,540
Fabric filter + catalytic fabric filter	13,500	13,200	5,140	4,990	2,560	–

9 Analysis of coal-fired boilers

There are two analyses in this section. The first is of two levels of dioxin control that can be used for coal-fired boilers. The first level is achieved by adding a fabric filter (or baghouse). The second level is achieved using a catalytic fabric filter; this involves replacing the filters in the baghouse with others impregnated with a catalyst. Data and results are given in Tables 9.1 through 9.4.

The second analysis is of the change that can be achieved by switching the fuel from coal to gas. This could be done through retrofitting gas burners on a coal boiler, but replacement of the boiler is more likely. Data and results are given in Tables 9.5 through 9.8.

The data on effectiveness and cost of the various technology options are taken from Sections A.14 through A.16 in the appendix.

9.1 Different levels of control

Table 9.1 Coal-fired boilers: annual emissions of dioxins at different levels of control

	Dioxin emissions (mg TEQ/year)		
	Small	**Medium**	**Large**
Multi-cyclone (baseline)	2	10	60
Fabric filter	1.4	7	42
+ catalytic fabric filter	0.75	3.8	23

Table 9.2 Coal-fired boilers: total installed costs for different levels of control

	Total installed costs ($'000s)					
	Small		**Medium**		**Large**	
	Existing (retrofit)	**New**	**Existing** (retrofit)	**New**	**Existing** (retrofit)	**New**
Fabric filter	160	120	380	290	1,300	970
+ catalytic fabric filter	240	200	760	670	3,100	2,800

Table 9.3 Coal-fired boilers: operating costs for different levels of control

	Operating costs ($'000s/year)		
	Small	**Medium**	**Large**
Fabric filter	170	200	330
+ catalytic fabric filter	210	360	800

Table 9.4 Coal-fired boilers: cost-effectiveness ratios for different levels of control

	CE ratios ($/mg TEQ reduced)					
	Small		**Medium**		**Large**	
	Existing (retrofit)	**New**	**Existing** (retrofit)	**New**	**Existing** (retrofit)	**New**
Fabric filter	323,000	313,000	85,400	81,000	29,000	26,300
+ catalytic fabric filter	79,700	79,700	66,500	66,500	37,800	38,000
Fabric filter + catalytic fabric filter	196,000	192,000	75,600	73,500	–	–

The incremental CE ratios for the first technical option are larger than those for the second technical option for small and medium coal-fired boilers, indicating that the first option is dominated and should be discarded.

9.2 Alternative fuel: replace coal with gas

There are two ways to substitute gas for coal as the fuel in coal-fired boilers: the *retrofit* of gas burners and the *premature replacement* of coal-fired boilers with gas-fired boilers. The former is less likely than the latter, but both are considered.

Different results are given for the North Island and the South Island, since it is assumed that natural gas would be burned in North Island boilers and LPG would be burned in South Island boilers. The price of LPG is three times that of natural gas, and thus operating costs are much higher in the South Island.

Operating costs are higher for retrofit than replacement because energy efficiency is lower.

The premature replacement option is analysed in the same way as the replacement of a medical waste incinerator with an autoclave system. The cost of premature replacement is thus the cost of bringing forward expenditure in time. This requires an estimate of the average remaining lifetime of coal-fired boilers. Because there is such great variation in the lifetime of coal-fired boilers, results are given for two remaining lifetimes – 10 years and 30 years.

No credit has been taken for the residual value of a replaced coal-fired boiler. The cost of a new gas-fired boiler and a new coal-fired boiler have been taken to be the same, although the former is likely to be cheaper than the latter. Both these assumptions exert upward pressure on the cost-effectiveness ratios (i.e. they make them less cost-effective).

Table 9.5 Coal-fired boilers: switch to gas – annual emissions of dioxins for different fuels

	Dioxin emissions (mg TEQ/year)		
	Small	**Medium**	**Large**
Coal-fired (baseline)	2	10	60
Switch to gas	< 0.2	< 1	< 6

Table 9.6 Coal-fired boilers: switch to gas – total installed costs for retrofit and premature replacement

	Total installed costs ($'000s)		
	Small	Medium	Large
Retrofit gas burners	10	35	200
Premature replacement	70	350	2,100

Table 9.7 Coal-fired boilers: switch to gas – operating costs for retrofit and premature replacement

	Operating costs ($'000s/year)		
	Small	Medium	Large
Retrofit – North Island	180	900	5,400
Retrofit – South Island	414	2,080	12,420
Replace – North Island	150	750	4,500
Replace – South Island	345	1,730	10,350

Table 9.8 Coal-fired boilers: switch to gas – cost-effectiveness ratios

	CE ratios ($/mg TEQ reduced)		
	Small	Medium	Large
Retrofit gas burners:			
North Island	101,000	101,000	101,000
South Island	231,000	231,000	231,000
Premature replacement:			
North Island – 10 years	86,900	86,900	86,900
North Island – 30 years	88,800	88,800	88,800
South Island – 10 years	195,000	195,000	195,000
South Island – 30 years	197,000	197,000	197,000

The CE ratios do not vary with the boiler size, since dioxin emissions, total installed costs, and gas costs are all directly proportional to the quantity of coal burned. The cost-effectiveness of switching to gas is approximately the same for both the retrofit option and the premature replacement option. The CE ratios for the premature replacement option vary only slightly with the remaining lifetime of the coal boiler. This follows from the domination of total installed costs by operating costs.

Since the main determinant of operating costs is the cost of gas compared with coal, CE ratios in practice would be sensitive to gas prices. Widespread adoption of gas could affect the demand for gas and drive up the price.

10 Summary of results for industrial sources

The detail of the preceding four sections makes it difficult to see the overall picture. For this reason, the cost-effectiveness ratios for the four sources are now presented in summary tables.[20]

Although building in dioxin reducing technology in *new* systems is consistently more cost-effective than retrofitting it into *existing* systems, the differences appear to be small for most installations. In general, there is no reason *on grounds of cost-effectiveness* to treat new emitters differently from existing emitters, and the distinction between the two is now dropped. This statement should *not* be taken as an argument against a transition period for existing sources.[21]

With a few exceptions, cost-effectiveness varies very significantly with *size*, and the tables in this section are organised by size as well as by kind of emitter.

The annual *effectiveness* as well as *cost-effectiveness* of the technology options considered in the preceding four sections is also summarised in this section. Note that the effectiveness tables apply to only *one* emitter of each size; that is, *unit* effectiveness not *national* effectiveness is presented.[22]

10.1 Results for waste incinerators

Table 10.1 Cost-effectiveness of technical options for waste incinerators

	CE ratios ($/mg TEQ reduced)			
	Small medical	**Medium medical**	**Large medical**	**Municipal**
Different levels of control				
Good combustion practice	120	–	–	–
+ quench cooling	3,100	510	340	–
+ fabric filter	11,000	–	–	210
+ fabric filter & carbon injection	–	3,500	2,000	400
+ reheat and catalyst	–	500,000	390,000	98,000
Alternative technology				
Autoclave system[23] *compared with*:				
Uncontrolled	110	–	–	–
Good combustion practice	860	1,200	1,400	–
Up to carbon injection	–	–	280,000	–

20 In this section, the CE ratios are averages of those for existing and new systems, and have been rounded to two significant figures.

21 There are a number of reasons for transition periods. For instance, retrofit costs may be very high for some unusual installations.

22 There is much uncertainty in some of the data, and ranges for effectiveness are given for some technical options in the appendix.

23 Note that these CE ratios apply only to the *premature* replacement of an incinerator with an autoclave system.

Table 10.2 Effectiveness of technical options for waste incinerators

	Annual effectiveness per incinerator (mg TEQ reduced/year)			
	Small medical	**Medium medical**	**Large medical**	**Municipal**
Different levels of control				
Good combustion practice	174	–	–	–
+ quench cooling	12	145	290	–
+ fabric filter	9	34	68	3710
+ carbon injection	–	20	41	955
+ reheat and catalyst	–	0.3	0.5	12
Alternative technology				
Autoclave system *compared with*:				
Uncontrolled	200	–	–	–
Good combustion practice	26	200	400	–
Up to carbon injection	–	–	2	–

10.2 Results for non-ferrous foundries

Table 10.3 Cost-effectiveness of technical options for non-ferrous foundries

	CE ratios ($/mg TEQ reduced)		
	Small	**Medium**	**Large**
Fabric filter	32,000	2,300	2,800
+ carbon injection	7,500	790	2,500
Fabric filter + carbon injection	20,000	1,600	2,700

Table 10.4 Effectiveness of technical options for non-ferrous foundries

	Annual effectiveness per foundry (mg TEQ reduced/year)		
	Small	**Medium**	**Large**
Fabric filter	4	80	170
+ carbon injection	4	82	155
Fabric filter + carbon injection	8	162	325

10.3 Results for wood-fired boilers

Table 10.5 Cost-effectiveness of technical options for wood-fired boilers

	CE ratios ($/mg TEQ reduced)		
	Small	**Medium**	**Large**
Virgin wood waste			
Fabric filter	250,000	67,000	25,000
+ catalytic fabric filter	48,000	39,000	25,000
Fabric filter + catalytic fabric filter	130,000	50,000	25,000
100% contaminated wood waste			
Fabric filter	25,000	6,700	2,500
+ catalytic fabric filter	4,800	3,900	2,500
Fabric filter + catalytic fabric filter	13,000	5,000	2,500

Table 10.6 Effectiveness of technical options for wood-fired boilers

	Annual effectiveness per boiler (mg TEQ reduced/year)		
	Small	**Medium**	**Large**
Virgin wood waste			
Fabric filter	0.8	4	20
+ catalytic fabric filter	1	5	29
Fabric filter + catalytic fabric filter	1.8	9	49
100% contaminated wood waste			
Fabric filter	8	40	200
+ catalytic fabric filter	10	50	290
Fabric filter + catalytic fabric filter	18	90	490

10.4 Results for coal-fired boilers

Table 10.7 Cost-effectiveness of technical options for coal-fired boilers

	CE ratios ($/mg TEQ reduced)		
	Small	**Medium**	**Large**
Fabric filter	320,000	83,000	28,000
+ catalytic fabric filter	80,000	67,000	38,000
Fabric filter + catalytic fabric filter	190,000	75,000	–
Switch to gas:[24]			
North Island	94,000	94,000	94,000
South Island	210,000	210,000	210,000

Table 10.8 Effectiveness of technical options for coal-fired boilers

	Annual effectiveness per boiler (mg TEQ reduced/year)		
	Small	**Medium**	**Large**
Fabric filter	0.6	3	18
+ catalytic fabric filter	0.7	3	19
Switch to gas	1.8	9	54

24 Average of retrofit option and premature replacement option.

11 National results for waste incinerators

Section 32 of the Resource Management Act requires assessment of both *efficiency* and *effectiveness* of different policy instruments intended to reduce dioxin emissions. In the previous section, the *efficiency* of different technical options has been presented in terms of cost-effectiveness ratios, and *effectiveness* has been presented in terms of *unit* effectiveness; that is, effectiveness per incinerator, foundry or boiler.

However, estimates of effectiveness should ideally be made at a national level. For this, the numbers of each kind of emitter at each level of control would be required. These numbers are currently available only for waste incinerators. Consequently, this section contains estimates of *national* annual effectiveness of the different technical options for waste incinerators only.

Current numbers of waste incinerators of different sizes and at different levels of dioxin control in New Zealand are:

- 19 small medical waste incinerators, of which five are uncontrolled and the remainder have good combustion practice
- two medium medical waste incinerators, both of which have good combustion practice
- one large medical waste incinerator, which has good combustion practice, quench cooling, a fabric filter, and carbon injection
- zero municipal waste incinerators.

It is important to understand that this is a snapshot of the *current* situation. Increases in the number of medical waste incinerators would change the estimates of national effectiveness. The introduction of municipal waste incineration could have a dramatic effect on dioxin emissions, in which case the effectiveness of dioxin controls would be concomitantly large.

Tables 11.1, 11.2 and 11.3 contain estimates of national effectiveness for the relevant technical options for *existing* small, medium and large medical waste incinerators respectively.[25] These tables also contain the corresponding cost-effectiveness ratios.

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[25] In these tables, the "number of incinerators" column represents the number of incinerators that could add the level of emission control shown.

Table 11.1 Small medical waste incinerators

	Number of incinerators	Unit effectiveness (mg TEQ reduced/year)	National effectiveness (mg TEQ reduced/year)	Cost-effectiveness ($/mg TEQ reduced)
Different levels of control				
Good combustion practice	5	174	870	120
+ quench cooling	19	12	228	3,100
+ fabric filter	19	9	171	11,000
Alternative technology				
Autoclave system	5	200	1000	110
	14	26	364	860

Table 11.2 Medium medical waste incinerators

	Number of incinerators	Unit effectiveness (mg TEQ reduced/year)	National effectiveness (mg TEQ reduced/year)	Cost-effectiveness ($/mg TEQ reduced)
Different levels of control				
+ quench cooling	2	145	290	510
+ fabric filter & carbon injection	2	54	108	3,500
+ reheat and catalyst	2	0.3	0.6	500,000
Alternative technology				
Autoclave system	2	200	400	1,200

Table 11.3 Large medical waste incinerators

	Number of incinerators	Unit effectiveness (mg TEQ reduced/year)	National effectiveness (mg TEQ reduced/year)	Cost-effectiveness ($/mg TEQ reduced)
Different levels of control				
+ reheat and catalyst	1	0.5	0.5	390,000
Alternative Technology				
Autoclave system	1	2	2	280,000

These results can be presented graphically as "supply curves of reduced dioxin emissions". A supply curve is a line showing the relationship between the price of a good and the quantity that can be supplied. Supply curves typically rise as higher prices enable greater quantities of goods to be supplied. In the supply curves presented in this section, a cost-effectiveness ratio is a price and annual effectiveness is a quantity.

Data from Tables 11.1 through 11.3 have been reorganised into a form for plotting such a supply curve in Tables 11.4a and 11.4b. The supply curves for reduced dioxin emissions from existing medical waste incinerators are shown in Figure 11.1a and .Figure 11.1b.[26] Tables 11.4a and 11.4b can be used to interpret the corresponding figures.

There are two supply curves: one for increased levels of dioxin controls in waste incinerators and one for the alternative approach of premature replacement of waste incinerators by autoclave systems. Both supply curves are step functions, where each step represents a technical option for a category of waste incinerator. The technical options are ordered by increasing price; that is, by increasing cost-effectiveness ratio.

The supply curve format is a powerful aid for considering the results of this kind of analysis. Wide, low steps are both effective and cost-effective; narrow, high steps are the opposite.

The cumulative national effectiveness in mg TEQ reduced per year is plotted along the horizontal axis. If, for example, it is considered that an annual reduction of 1 mg TEQ is worth up to $4,000, then about 1.5 grams (1,496 mg) TEQ could be reduced per year using increasing levels of dioxin control in incinerators. However, the autoclave option would yield a reduction of about 1.8 grams (1,764 mg) TEQ per year at the lower cut-off price of $1,200 per mg TEQ reduced.

The first supply curve (increasing levels of control) exhibits typical “diminishing returns” pollution control behaviour: it rises slowly at first, and then increasingly steeply.

[26] Figures 11.1a and 11.1b do not contain all the data in Tables 11.4a and 11.4b. Technical options with cost-effectiveness ratios greater than $12,000 per mg TEQ reduced are not represented in either figures.

Table 11.4a Supply curve 1: increasing levels of dioxin control in medical waste incinerators

Size of incinerator	**Technical option**	**National effectiveness of option** (mg TEQ reduced/year)	**National cumulative effectiveness** (mg TEQ reduced/year)	**CE ratio** ($/mg TEQ reduced)
Small	Good combustion practice	870	870	120
Medium	Quench cooling	290	1,160	510
Small	Quench cooling	228	1,388	3,100
Medium	Fabric filter & carbon injection	108	1,496	3,500
Small	Fabric filter	171	1,667	11,000
Large	Reheat and catalyst	0.5	1,667.5	390,000
Medium	Reheat and catalyst	0.6	1668.1	500,000

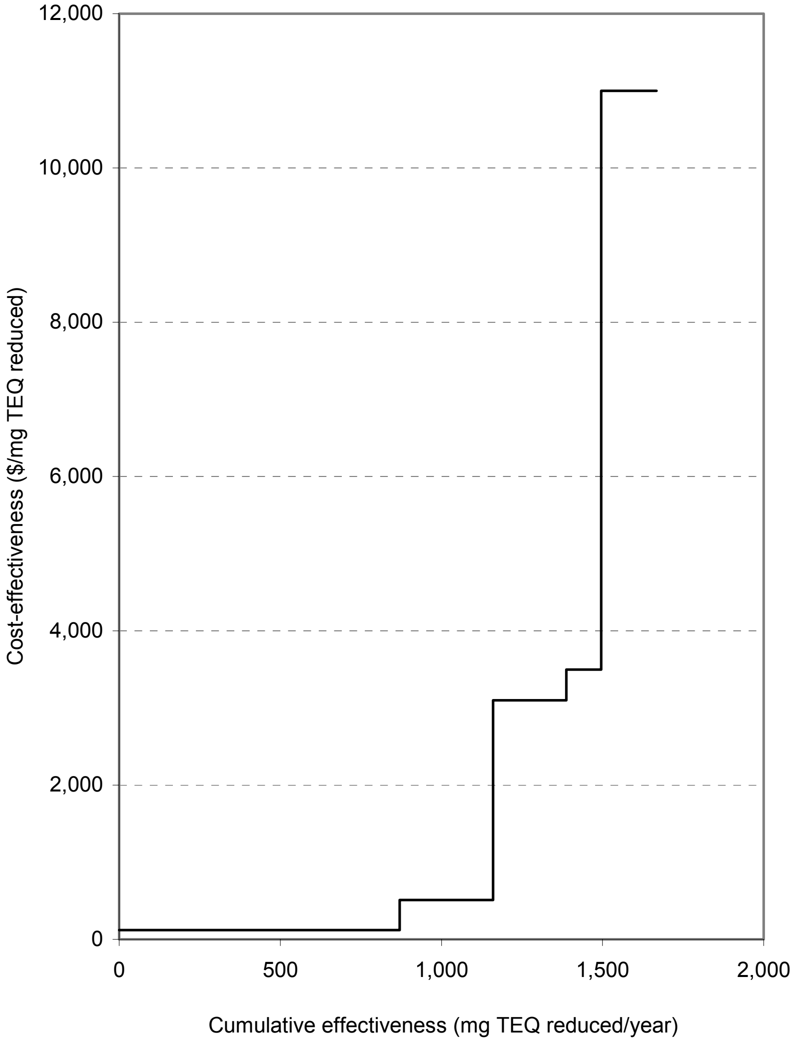

Figure 11.1a Supply curve for increasing levels of dioxin control in existing medical waste incinerators

Table 11.4b Supply curve 2: premature replacement of medical waste incinerator with autoclave system

Size of incinerator	**Level of dioxin control in incinerator**	**National effectiveness of option** (mg TEQ reduced/year)	**National cumulative effectiveness** (mg TEQ reduced/year)	**CE ratio** ($/mg TEQ reduced)
Small	Uncontrolled	1000	1000	110
Small	Good combustion practice	364	1364	860
Medium	Good combustion practice	400	1764	1,200
Large	Up to carbon injection	2	1766	280,000

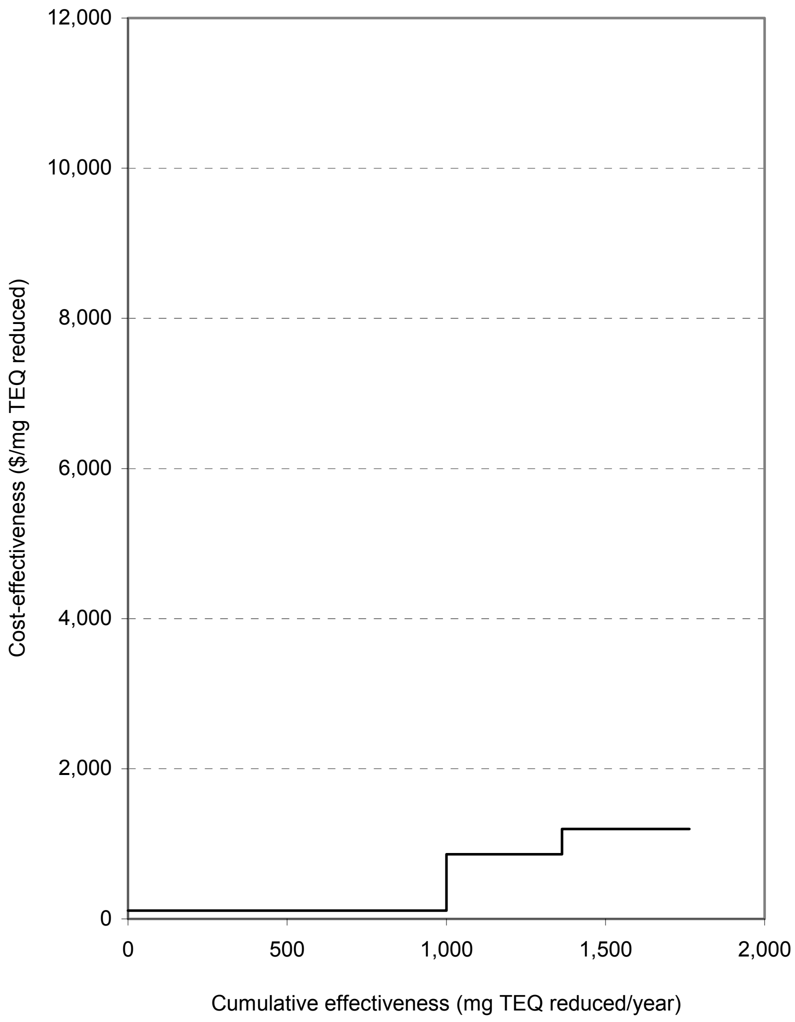

Figure 11.1b Supply curve for premature replacement of existing medical waste incinerators by autoclave systems

12 Policy implications

This analysis gives the cost-effectiveness and (unit) effectiveness of various technical options for selected sources of dioxin emissions to air. Before considering the policy implications following from these results, four caveats are in order.

- The cost-effectiveness ratios should be used *comparatively*. No dollar value has been placed on the benefit of reducing dioxin emissions to air by a milligram. The analysis is a cost-effectiveness analysis, not a cost-benefit analysis.
- The costs are *engineering* costs only. The capital costs of putting policy instruments in place, and the operating costs of monitoring etc. have not been assessed.
- With the exception of medical waste incinerators, the effectiveness estimates are *per emitter*. Without knowing the total number of emitters of each size and control level, the total effectiveness (and total cost) cannot be estimated. However, some rough estimates should be possible using the national dioxin inventory (Buckland *et al*., 2000).
- Reported emissions vary widely as can be seen in the appendix, and consequently the effectiveness of the various technical options is uncertain. In the development of policy instruments, sensitivity analysis could be used to test the relevant results in this report.

Nine implications for the development of policy instruments follow from an initial examination of the results.

1) Differences in cost-effectiveness between retrofitting abatement technologies and building them into new plants do not appear to be significant.[27] It does not necessarily follow that transition periods are undesirable.

2) There is a strong case for an "aggressive" approach towards the incineration of medical waste in order to encourage replacement with alternatives, such as autoclave systems. This would virtually eliminate dioxin emissions into air from the disposal of medical waste. This does not apply to the single large medical waste incinerator, which already operates at a high level of dioxin control.

3) A large municipal waste incinerator could be a major source of dioxin, and this must be a consideration if an application for such an incinerator is made. The counter side is that municipal waste incineration will reduce the volume of waste going to landfills, which have their own adverse environmental effects, including being the largest estimated source of dioxin emissions to air in New Zealand. Therefore, a municipal waste incinerator could result in a net overall reduction in dioxin emissions.

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[27] The replacement of large medical waste incinerators with autoclave systems is an exception. However, for small and medium medical waste incinerators, the economic case for the alternative technology is very strong for both "premature" and "natural" replacement.

The effectiveness of employing fabric filtration in a municipal waste incinerator is many times greater than that of any other option analysed in this study, and, moreover, is extremely cost-effective. However, the use of fabric filtration may not achieve the desired level of emission control required by any future national environmental standard. Certainly, on the basis of emission standards applied on large municipal waste incinerators overseas, a higher level of pollution control equipment would be required.

4) Economies of scale in dioxin control operate strongly for medium and large non-ferrous foundries *vis-à-vis* small foundries. There is thus an economic rationale for instruments that are sensitive to foundry size. More information about foundry size and emissions is required before policy instruments are developed.

5) Economies of scale operate strongly for increasing levels of dioxin control in both wood-fired and coal-fired boilers. The economic case for dioxin reductions in boilers is generally much weaker than for incinerators or foundries.

6) The burning of contaminated wood in boilers is potentially a major concern, but not enough is known about the types of contamination and combustion practices. There is a case for data collection in this area.

7) Switching boilers from coal to gas is a relatively uneconomic environmental intervention if done only for dioxin emission reduction. However, this may occur for other reasons such as responding to incentives for greenhouse gas control.

8) Although there has been no explicit analysis of landfill fires in this report, the banning of landfill fires should be a high priority. This source of dioxin emissions to air is greater than any other by an order of magnitude, and there is no environmental justification for this practice.

9) Like landfill fires, the backyard burning of waste is a major source of dioxins. Discharges of dioxin to air from this undesirable practice should be prohibited, at least for certain types of waste.

Appendix: Control equipment costs and effectiveness

The largest uncertainty in the tables contained in this appendix is the expected performance of the control equipment options. There is also uncertainty in the cost estimates, which are based on hypothetical applications. The control costs provided in this appendix are for both total installed costs and operating costs. Total installed costs include the capital cost of equipment plus installation costs.

There is very little published information on dioxin destruction or removal efficiencies, even for well-known technology. Most of the data that are available are in the form of a discharge concentration or an emission factor (ng TEQ/Sm^3 or μg/tonne), and then only for relatively large sources. It is not a simple task to translate this information into dioxin removal rates. Consequently, the performance estimates presented are based on a degree of judgement being applied to the published emissions data, and the emissions have been presented as a range of likely values.

The control cost information represents likely costs based on hypothetical cases. They may be considered typical, but not necessarily representative of the industry group. If truly representative examples were required it would be necessary to survey the industry groups in question more widely than has been done in the current exercise. This is particularly the case for coal- and wood-fired boilers and metallurgical processes, where the information is much less certain compared with waste incineration costs.

The baseline emissions chosen for each case fall within a range of possible values. For example, emission factors used by Buckland *et al.* (2000) suggest an uncontrolled medium-sized incinerator could have emissions ranging from 5 to 1000 mg/year. The hypothetical example assumes a baseline emission of 200 mg/year.

As far as possible, the technology options chosen have been based on what is either already used in New Zealand or is relatively well known. Some technologies are unknown, however, particularly those where a very high level of removal is required, or for industries where dioxin controls have not traditionally been applied.

The performance of the possible technology options applied to a medium medical waste incinerator is illustrated in Figure A.1.

Specific dioxin control equipment is not usually employed for boilers and foundries, particularly in New Zealand. This may produce a considerable amount of uncertainty in both the cost and reduction figures presented, particularly as controls go beyond standard fabric filtration systems. In the case of boilers, for example, it is assumed that catalytic fabric filters will achieve a high level of dioxin destruction, but this is based on claims from equipment suppliers and no such equipment is used in this country.

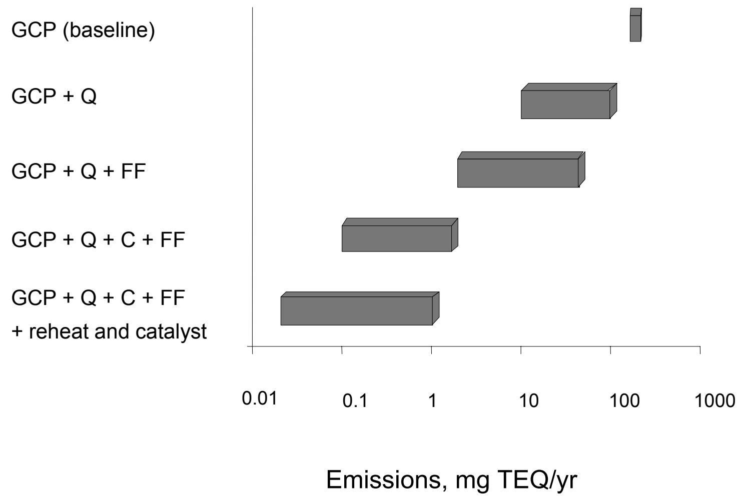

Figure A.1 Likely performance of control options applied to a medium medical waste incinerator
[Based on performance data given in Table A.2. GCP = good combustion practice; Q = rapid cooling or quench cooling; FF = fabric filtration; C = carbon injection.]

The tables include a column that indicates compliance with a potential emission standard. These values do not represent any potential limit that may be applied by the Ministry for the Environment, nor are they recommendations. They reflect an emission concentration that could be met with assurance, bearing in mind that engineers and equipment supplies will need to allow some conservatism when guaranteeing performance. For example, the best performance for control equipment attached to incinerators is conservatively assumed to achieve an emission of 0.05 ng/Sm^3. Equipment suppliers do not generally guarantee performance below 0.1 ng/m^3 since most overseas emission standards are 0.1 ng/Sm^3. However, a growing number of incinerator processes may be able to achieve emissions lower than 0.01 ng/Sm^3. Furthermore, concentration limits (ng/Sm^3) can be very problematic for certain sources, particularly for industries like foundries, where a large proportion of the ventilated air is from the foundry building, which will contain negligible dioxin concentrations.

Where possible, equipment costs have been based on budget estimates provided by suppliers. The US EPA's CO$TAIR model (Vatavuk, 1999) has also been used, particularly for estimating total installed costs for fabric filters over a range of application sizes. The results from this model have proved comparable to data from Damiano and Campbell (1997) and equipment suppliers after allowing for likely exchange rates. The CO$TAIR model uses a factor of 2.17 times the purchased equipment costs to estimate total installed cost for fabric filters, and this follows Damiano and Campbell (1997). For other control equipment, total installed costs are generally assumed to be approximately twice equipment costs. Retrofit costs are assumed to be 30% higher where control equipment for a large plant is required, and 15% for small plant items. Control costs are given in $'000s/year.

More specific comments for each application are provided in the notes identified in the tables.

A.1 Small medical waste incinerators

It is assumed that small medical waste incinerators:

- are capable of burning medical and related waste at a rate of 50–100 kg/hr
- have a baseline emission of 200 mg TEQ/year for 3,000 hours of operation of an uncontrolled unit
- have a discharge equivalent to an exhaust gas concentration of approximately 40 ng TEQ/Sm^3.

Table A.1 Small medical waste incinerator control costs

Selected technology options	Reported emissions (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	**Costs** (NZ$'000s)			
					Existing (retrofit)		**New**	
		% reductions	mg TEQ/year		**Total installed**	**Operating**	**Total installed**	**Operating**
Good combustion practice (GCP)[a]	0.5–10[b]	75–98.8	2.5–50	10	70	10	Zero	Zero
GCP + rapid cooling or quench (Q)[c]	0.5–5[d]	87.5–98.8	2.5–25	5	130	40	50	30
GCP + Q + fabric filter (FF)	0.1–2[e]	95–99.8	0.5–10	2	220	120	120	110
GCP + Q + carbon injection (C) + FF	0.005–0.07[f]	99.8–99.99	0.02–0.35		Not a realistic option			
GCP + Q + C + FF + reheat + catalyst[g]	0.001–0.049[h]	99.88–99.998	0.005–0.25		Not a realistic option			

Replacement with autoclave

There are several alternative treatment options for medical waste, including steam-autoclaving, microwave irradiation, chemical treatment and biological treatment. In New Zealand the most common alternative is to use autoclaving. Replacement of an existing incinerator with an autoclave and grinder would result in a dioxin discharge of less than 0.003–0.03 mg TEQ/year, depending on whether a natural gas or coal-fired boiler is used for steam production and ignoring any potential increase in dioxin emissions from associated activities (e.g. transportation of residual wastes to landfill). This is estimated to cost $400,000 (total installed costs plus 20% decommissioning costs) for treatment of wastes equivalent to that handled by a small medical waste incinerator, with no increase in operating costs over an incinerator.[i]

A.2 Medium medical waste incinerators

It is assumed that medium medical waste incinerators:

- are capable of burning medical and related waste at a rate of 300–500 kg/hr
- have a baseline emission of 200 mg TEQ/year for 3,000 hours of operation of a unit with good combustion practice
- have a discharge equivalent to an exhaust gas concentration of approximately 10 ng TEQ/Sm^3.

Table A.2 Medium medical waste incinerator control costs

Selected technology options	Reported emissions (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	Costs (NZ$'000s)			
					Existing (retrofit)		New	
		% reductions	mg TEQ/year		Total installed	Operating	Total installed	Operating
Good combustion practice (GCP)[a] (baseline)	0.5–10[b]	0	200	10	Zero	Zero	Zero	Zero
GCP + rapid cooling or quench (Q)[c]	0.5–5[d]	50–95	10–100	5	100	60	90	60
GCP + Q + fabric filter (FF)	0.1–2[e]	80–99	2–40	2	520	160	410	160
GCP + Q + carbon injection (C) + FF	0.005–0.07[f]	99.3–99.95	0.1–1.4	0.1	610	180	490	180
GCP + Q + C + FF + reheat + catalyst[g]	0.001–0.049[h]	99.5–99.99	0.02–0.98	0.05	1,300	210	1,100	210

Replacement with autoclave

An autoclave and grinder would result in a dioxin discharge of less than 0.015–0.15 mg TEQ/year. This plus a small boiler is estimated to cost $3,200,000 (total installed costs plus 20% decommissioning costs) for treatment of wastes at a rate equivalent to that handled by a medium medical waste incinerator.[i] Operating cost is taken to be the same as that for an incinerator.[i]

A.3 Large medical waste incinerators

It is assumed that large medical waste incinerators:

- are capable of burning medical and related waste at a rate of 1000 kg/hr
- have a baseline emission of 400 mg TEQ/year for 3000 hours of operation for a unit with good combustion practice
- have a discharge equivalent to an exhaust gas concentration of approximately 10 ng TEQ/Sm^3.

Table A.3 Large medical waste incinerator control costs

Selected technology options	Reported emissions (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	Costs (NZ$'000s)			
					Existing (retrofit)		New	
		% reductions	mg TEQ/year		Total installed	Operating	Total installed	Operating
Good combustion practice (GCP)[a] (baseline)	0.5–10[b]	0	400	–	Zero	Zero	Zero	Zero
GCP + rapid cooling or quench (Q)[c]	0.5–5[d]	50–95	20–200	5	140	80	120	80
GCP + Q + fabric filter (FF)	0.1–2[e]	80–99	4–80	2	570	190	450	190
GCP + Q + carbon injection (C) + FF	0.005–0.07[f]	99.3–99.95	0.2–2.8	0.1	690	220	550	220
GCP + Q + C + FF + reheat + catalyst[g]	0.001–0.049[h]	99.5–99.99	0.04–2	0.05	1,800	260	1,400	260

Replacement with autoclave

An autoclave and grinder would result in a dioxin discharge less than 0.03–0.3 mg TEQ/year. This plus a small boiler is estimated to cost $6,300,000 (total installed costs plus 20% decommissioning costs) for waste treatment equivalent to that handled by a large medical waste incinerator.[i] Operating cost is taken to be the same as that for an incinerator.[i]

A.4 Municipal waste incinerators

It is assumed that municipal waste incinerators are modular in design. A waste incineration facility could typically consist of more than one module. The data given in Table A.4 is for a single module. It is further assumed that each module:

- is capable of burning municipal waste as a mass burn unit at a rate of 10,000 kg/hr
- has the flue gases cooled by a steam generator
- has a baseline emission of 4700 mg TEQ/year for 7,000 hours of operation for a unit with good combustion practice, but without any pollution abatement system
- has a discharge equivalent to an exhaust gas concentration of approximately 5 ng TEQ/Sm^3.

Table A.4 Municipal waste incinerator control costs

Selected technology options	Reported emissions (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	Costs (NZ$'000s)[j]	
		% reductions	mg TEQ/year		Total installed	Operating
Fabric filter (FF)	0.1–2[e]	60–98	94–1,900	2	2,100	480
Carbon injection (C) + FF	0.005–0.07[f]	98.6–99.9	4.7–66	0.1	2,600	790
C + FF + reheat + catalyst[g]	0.001–0.049[h]	99–99.98	0.94–46	0.05	7,000	1,300

A.5 Waste incinerator notes

a. *Good combustion practice* is defined as having a final combustion chamber maintained at a temperature of 1000 ^{o}C with a residence time of 1 second in the presence of at least 6% oxygen, and having continuous monitoring of carbon monoxide and oxygen. It is assumed that only old, small incinerators would not comply with these conditions. The costs for an existing small incinerator achieving good combustion practice are based on a hypothetical starved-air unit in reasonable condition with a final combustion chamber that will have to be upgraded from a 0.5 second residence time to 1 second and fitted with carbon monoxide equipment. This is estimated at $70,000 based on budget information from Crusader Engineering Ltd. Additional operating costs (such as fuel costs) are expected to be minimal. Clearly the potential costs could vary significantly. Some units may require total replacement in order to meet the good combustion criteria. For a municipal waste incinerator, it is realistic to assume that, as a minimum, such a facility will have good combustion practice.

b. Dioxin emissions from good combustion are taken from general information published by Eduljee and Cains (1997).

c. Quench cooling costs assume a wet cooling system is used. An alternative may be a waste heat boiler. Although waste heat boilers are significantly more expensive, they may be preferred because they could allow utilisation of waste energy in the form of steam and may also have operational advantages. Wet systems may give rise to maintenance difficulties caused by corrosion problems. However, it is considered appropriate to base costs on a wet system given that the only large medical waste incinerator in New Zealand uses this form of cooling.

d. The expected emissions from an incinerator employing good combustion and quench cooling are very uncertain. The figures used are based on information on municipal waste incinerators and experimental studies reported by Buekens and Huang (1998).

e. Data published by Eduljee and Cains (1997) suggests dioxin emissions from a low temperature fabric filter (at 150 ^{o}C) with no other controls will range from 0.01 to 0.5 ng TEQ/m^3. The same authors report emissions from an electrostatic precipitator (ESP; an alternative to high-efficiency particulate removal provided by a fabric filter) to range from about 0.2 to 2 ng/m^3. While likely to be better than an ESP, the performance of a fabric filter alone may not be significantly better than good combustion practice with quench. This is because exhaust concentration of particulate, carbon and organic precursors will already be low. With relatively clean exhaust gases, there is less potential for dioxin removal by fabric filtration. It is therefore considered sensible to use less optimistic emission figures, and a range of 0.1 to 2 ng TEQ/m^3 is assumed. Although somewhat arbitrary, these figures lie in the range of values reported for fabric filters and ESPs.

f. Dioxin emissions from carbon injection and fabric filters based on measurements at an Auckland incinerator are 0.04–0.07 ng/Sm^3 as reported by Buckland *et al*. (2000). However, this is from a limited number of measurements compared to the performance data used for the other control options. Overseas information suggests that a well managed carbon injection/fabric filter system can achieve emissions as low as 0.01 ng/m^3 (Eduljiee and

Cains, 1997) or even 0.001 ng/m^3 (Kilgroe, 1996). Thus a lower bound emission limit of 0.005ng/m^3 is assumed.

g. It is possible that carbon injection could achieve considerably better than 0.1 ng/Sm^3, as discussed above, but it is unlikely that this could be achieved with guarantees. It is therefore assumed that it will be necessary to employ further controls, and the technology identified is catalytic destruction following reheat of the gases after leaving the fabric filter. Reheating of the gasses is necessary to raise the temperature from an assumed 100 ^{o}C (exiting the baghouse) to 175 ^{o}C (the optimal temperature for catalytic destruction).

h. Emissions are based on information provided by CRI catalysts on the Shell DeNOx system attached to a municipal solid waste incinerator at Heeren in the Netherlands (CRI publicity material). It should be recognised that CRI will typically only guarantee an emission of 0.1 ng/Sm^3, but they have installed some systems in Japan with a guarantee of meeting 0.01 ng/Sm^3 (H. Tang, CRI Catalysts Asia Pacific, pers comm., August 2000).

i. In background papers to their standards for medical and hospital waste incinerators, the US EPA reports that it may be more cost effective for some facilities to dispose of their waste using alternative methods (US EPA, 1997). This suggests that both capital and/or operating costs of autoclave treatment, a common alternative treatment for medical and hospital wastes, may be less than for incineration. However, for this exercise, the operating costs for an autoclave and grinder are assumed to be the same as those for an incinerator handling an equivalent volume of waste. It is noted that energy costs for raising steam could be more than burning waste containing a portion of combustible material. In addition, landfill disposal costs for autoclaves will likely be higher because they produce larger volumes of residual waste. The operating costs given are taken to include the disposal to landfill of the residual solid waste material. One reviewer has suggested that the operating costs of an autoclave system may be 20% higher than the operating costs of an equivalent waste incinerator.

   The cost of an autoclave and grinding system is based on information provided by AWS Clinical Waste Ltd. Grinding and autoclave costs are based on a unit having the same hourly consumption rate as the nominal incinerator. However, it is possible that smaller autoclave units could replace an incinerator due to the ability to operate for longer periods in a 24-hour day. It is also possible that a single, centralised incinerator could be replaced by several local autoclave units depending on transport costs etc. Incinerator decommissioning costs are assumed to be 20% of replacement costs. For the small units (less than 100 kg/hr) it is assumed that an existing boiler will take up the steam demand, given that a small autoclave will most likely be installed on a hospital site. Costs of a gas-fired boiler are included for the larger units, based on information from Damiano and Campbell (1997).

j. Costs given are for a *new* municipal waste incinerator. Since there are no municipal waste incinerators operating in New Zealand, costs for an existing facility are unnecessary.

A.6 Small non-ferrous foundries

It is assumed that small non-ferrous foundries have a:

- processing capacity of 1 tonne per day
- ventilation system for one furnace and part of the working area
- baseline emission of 10 mg TEQ per year for 3,000 hours of operation for a unit with no emission control[a]
- discharge equivalent to a concentration of 0.2 ng TEQ/Sm^3 in the ventilation exhaust.

Table A.5 Small foundry control costs

Selected technology options	Reported emissions[c,d] (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	Costs (NZ$'000s)			
		% reductions	mg TEQ/year		Existing (retrofit)		New	
					Total installed	Operating	Total installed	Operating
Fabric filter[b]	0.03–0.73	0–85	1.5–10	1	270	99	210	99
Carbon injection + fabric filter	0.005–0.07	65–97.5	0.25–3.5	0.1	330	120	260	120

A.7 Medium non-ferrous foundries

It is assumed that medium non-ferrous foundries have a:

- processing capacity of 15–20 tonnes per day
- ventilation system for one furnace and part of the working area
- baseline emission of 200 mg TEQ per year for 3000 hours of operation for a unit with no emission control[a]
- discharge equivalent to a concentration of 0.2 ng TEQ/Sm^3 in the ventilation exhaust.

Table A.6 Medium foundry control costs

Selected technology options	Reported emissions[c,d] (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	Costs (NZ$'000s)			
		% reductions	mg TEQ/year		Existing (retrofit)		New	
					Total installed	Operating	Total installed	Operating
Fabric filter[b]	0.03–0.73	0–85	30–200	1	500	130	390	130
Carbon injection + fabric filter	0.005–0.07	65–97.5	5–70	0.1	580	180	460	180

A.8 Large non-ferrous foundries

It is assumed that large non-ferrous foundries have a:

- processing capacity of 30–40 tonnes per day
- ventilation system for two 1-tonne per hour furnaces and part of the working area
- baseline emission of 400 mg TEQ per year for 3000 hours of operation for a unit with no emission control[a]
- discharge equivalent to a concentration of 0.2 ng TEQ/Sm^3 in the ventilation exhaust.

Table A.7 Large foundry control costs

Selected technology options	Reported emissions[c,d] (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	Costs (NZ$'000s)			
		% reductions	mg TEQ/year		Existing (retrofit)		New	
					Total installed	Operating	Total installed	Operating
Fabric filter[b]	0.03–0.73	0–85	60–400	1	1,000	350	770	350
Carbon injection + fabric filter	0.005–0.07	65–97.5	10–140	0.1	2,200	580	1,700	580

A.9 Foundry notes

a. Clean scrap can reduce dioxin emissions, but this depends on the extent of cleaning. The most significant potential problem is the processing of PVC material used for wire insulation or in recycled batteries. Three non-ferrous foundry operators were spoken to and they all indicated they do not accept copper unless the scrap had PVC removed by granulation methods, rather than burning. A very small proportion of scrap is expected to be from burnt cable. Therefore the baseline emission is based on the processing of clean scrap only. It is at the high end of the range of emissions reported by Buckland *et al*. (2000) for non-ferrous metallurgical processes in terms of micrograms per tonne of metal produced (35 mg TEQ/tonne). The concentration in the ventilation exhaust is assumed to be low however (roughly 0.2 ng TEQ/Sm^3), because it is expected that the discharge will be diluted by a proportion of relatively clean workspace air.

b. Control costs for small foundries assume a minimum ventilation rate to control some of the workspace air. Thus the required size of a fabric filter is not necessarily proportional to foundry size.

c. There are no dioxin data on foundry emissions in New Zealand. The expected dioxin emissions for foundries with fabric filtration equipment are based on the measured emissions for a New Zealand lead smelter of 0.03–0.73 ng TEQ/Sm^3 (Buckland *et al*., 2000). The high end of this range shows a higher concentration than the assumed baseline emission, but it is considered that a lead smelter discharge will not be diluted with ventilation air to the same extent as a typical non-ferrous foundry. Thus, the performance figures for a fabric filter fitted to a foundry can only be loosely based on the lead smelter data and it is assumed that emissions will range up to 0.2 ng TEQ/Sm^3 (the baseline concentration). There is even less information on the expected performance of powdered carbon injection systems for this industry. The figures used for this exercise are based on the performance of these systems when employed in waste incinerators and should therefore be viewed with caution.

d. A higher level of dioxin control may be achieved by employing further technology, such as catalytic systems, after reheating the fabric filter exhaust (similar to that proposed for waste incinerators). While the performance for waste incinerators suggests this may achieve less than 0.05 ng TEQ/Sm^3, such performance is not proven for foundries. It is also considered an unrealistic option for anything but a very large metallurgical operation. Such catalytic technology is therefore not considered in this analysis.

A.10 Small wood-fired boilers

It is assumed that small wood-fired boilers:

- have a gross heat output of 1 MW
- burn virgin wood waste (see comment below table for contaminated wood)
- are a modern suspension grate unit fitted with multi-cyclone grit arrestors
- have a baseline emission of 3 mg TEQ per year for 7,000 hours of operation[a]
- have a discharge equivalent to a concentration of 0.1 ng TEQ/Sm^3 in the ventilation exhaust.

Table A.8 Small wood boiler control costs

Selected technology options	**Reported emissions** (ng TEQ/Sm^3)	**Likely performance**		**Emission standard achieved** (ng TEQ/Sm^3)	**Costs** (NZ$'000s)[c]			
		% reductions	mg TEQ/year		**Existing** (retrofit)		**New**	
					Total installed	**Operating**	**Total installed**	**Operating**
Fabric filter	0.064–0.086[b]	14–36	1.9–2.6	0.1	160	170	120	170
Catalytic fabric filter (fabric filter with catalytic media)[d]	0.012–0.065	35–88[e]	0.36–2	0.05[f]	240	210	200	210

Contaminated wood

Boilers that burn contaminated wood waste as a fuel will have a considerably higher dioxin emission than when burning virgin wood waste (estimated at least 10 times higher). If the above controls were applied to this activity, it may be assumed that the likely performance discharge figures (in the mg TEQ/year column) could be multiplied by a factor of 10.

In many cases it may be possible to remove the contaminated wood fuel and burn virgin wood waste only. The cost of removing contaminated wood from the fuel is very difficult to assess because it will depend on the proportion of this material in the fuel, local landfill and transport costs, and whether the fuel needs to be substituted with an alternative, such as coal. Some activities could burn a very high proportion of contaminated wood.

A.11 Medium wood-fired boilers

It is assumed that medium wood-fired boilers:

- have a gross heat output of 5 MW
- burn virgin wood waste (see comment below table for contaminated wood waste)
- are a modern suspension grate unit fitted with multi-cyclone grit arrestors
- have a baseline emission of 15 mg TEQ per year for 7000 hours of operation[a]
- have a discharge equivalent to a concentration of 0.1 ng TEQ/Sm^3 in the ventilation exhaust.

Table A.9 Medium wood boiler control costs

Selected technology options	Reported emissions (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	Costs (NZ$'000s)[c]			
		% reductions	mg TEQ/year		Existing (retrofit)		New	
					Total installed	Operating	Total installed	Operating
Fabric filter	0.064–0.086[b]	14–36	9.6–13	0.1	380	200	290	200
Catalytic fabric filter (fabric filter with catalytic media)[d]	0.012–0.065[e]	35–88	1.8–9.8	0.05[f]	760	360	670	360

Contaminated wood

Like the small boiler described in Section A.10, the likely performance discharge figures (in the mg TEQ/year column) could be multiplied by 10 when burning contaminated wood.

A.12 Large wood-fired boilers

It is assumed that large wood-fired boilers:

- have a gross heat output of 30 MW
- burn virgin wood waste (see comment below table for contaminated wood)
- are a modern suspension grate unit fitted with multi-cyclone grit arrestors
- have a baseline emission of 80 mg TEQ per year for 7000 hours of operation[a]
- have a discharge equivalent to a concentration of 0.1 ng TEQ/Sm^3 in the ventilation exhaust.

Table A.10 Large wood boiler control costs

Selected technology options	Reported emissions (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	Costs (NZ$'000s)[c]			
					Existing (retrofit)		New	
		% reductions	mg TEQ/year		Total installed	Operating	Total installed	Operating
Fabric filter	0.064–0.086[b]	14–36	51–69	0.1	1,300	330	970	330
Catalytic fabric filter (fabric filter with catalytic media)[d]	0.012–0.065[e]	35–88	9.6–52	0.05[f]	3,100	800	2,800	800

Contaminated wood

Like the small boiler described in Section A.10, the likely performance discharge figures (in the mg TEQ/year column) could be multiplied by 10 when burning contaminated wood.

A.13 Wood-fired boiler notes

a. Baseline emissions were determined from limited emission measurements undertaken in New Zealand and reported by Buckland *et al*. (2000).

b. Emissions achieved with fabric filters are uncertain. The figures used are based on emissions reported for salt-contaminated wood waste combustion in pulp-mill power boilers (Luthe *et al*., 1997).

c. Fabric filter costs for small and medium sized boilers are based on figures for coal boilers reported by Damiano and Campbell (1997). Costs for the large boiler are based on the CO$T-AIR model (Vatavuk, 1998).

d. The second level of dioxin control is based on catalytic fabric filters. More common technology for this level of dioxin control in New Zealand is to use carbon injection, although this is on waste incineration plants. Little is known about dioxin controls at this level. However, catalytic systems have been used in conjunction with NOx controls on very large boilers overseas. It therefore seems suitable to employ the recently developed catalytic filters in a baghouse (particularly considering there is no need to also remove heavy metals from this activity). The performance of this technology is based on information from WL Gore Ltd on the Remedia D/F filter systems. The costs given for this system are based on the assumption that catalytic filters are about 20 times more expensive than the commonly used Nomex filters.

e. As discussed, the performance of catalytic fabric filters on wood-fired boilers is not known. The expected performance is based on that reported by WL Gore for incinerator dioxin controls. Here the emissions generally range above 0.05 ng/Sm^3, but it is expected that their performance on wood waste boilers will be slightly better.

f. A third level of control for wood-waste combustion is not considered realistic. It may be possible to add a fixed-bed catalyst to the fabric filter exhaust after reheating the flue gases, similar to that suggested for waste incinerators. While this technology has been used in large boiler plant overseas (principally for NOx control), it appears to perform no better than that expected by the less expensive option of employing a catalytic fabric filter. It is not considered further in this analysis.

A.14 Small coal-fired boilers

It is assumed that small coal-fired boilers have:

- a gross heat output of 1 MW
- multi-cyclone grit arrestors
- a baseline emission of 2 mg TEQ per year for 7,000 hours of operation[a]
- a discharge equivalent to a concentration of 0.1 ng TEQ/Sm^3 in the ventilation exhaust.

Table A.11 Small coal boiler control costs

Selected technology options	Reported emissions (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	Costs (NZ$'000s)[c]			
		% reductions	mg TEQ/year		Existing (retrofit)		New	
					Total installed	Operating	Total installed	Operating
Fabric filter	0.06–0.08[b]	20–40	1.2–1.6	0.1	160	170	120	170
Catalytic fabric filter (fabric filter with catalytic media)[d]	0.012–0.065	35–90	0.2–1.3	0.05	240	210	200	210

Replacement with alternative fuel

The greatest dioxin reductions can be achieved by converting to gas (probably over 90%), resulting in a discharge of less than 0.2 mg TEQ/year. While it is possible to retrofit gas burners to any coal-fired boiler, it is considered that in most cases it would be necessary to replace the entire boiler with a package gas-fired boiler. The cost of this would be approximately $70,000 for a small 1 MW boiler. This allows for removal of the old plant, but does not include the remaining value of the coal boiler. A reliable age distribution of coal-fired boilers will require a survey, but 32 case studies reported by Damiano and Campbell (1997) showed a large range in ages, with some boilers manufactured in the 1950s and an average age of 23 years. The annual increase in operating costs would be approximately $150,000 for the North Island, based on a fuel cost difference of $6/GJ for natural gas, or $345,000 based on a cost difference of $14/GJ for LPG in the South Island.

For those boilers where it is practicable to retrofit gas burners, the total installed cost of conversion would be approximately $10,000, with operating costs estimated to be 20% higher than indicated above, since it is necessary to compensate for loss of efficiency due to reduction in radiative heat transfer.

A.15 Medium coal-fired boilers

It is assumed that medium coal-fired boilers have:

- a gross heat output of 5 MW
- multi-cyclone grit arrestors
- a baseline emission of 10 mg TEQ per year for 7,000 hours of operation[a]
- a discharge equivalent to a concentration of 0.1 ng TEQ/Sm^3 in the ventilation exhaust.

Table A.12 Medium coal boiler control costs

Selected technology options	Reported emissions (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	Costs (NZ$'000s)[c]			
		% reductions	mg TEQ/year		Existing (retrofit)		New	
					Total installed	Operating	Total installed	Operating
Fabric filter	06–0.08[b]	20–40	6–8	0.1	380	200	290	200
Catalytic fabric filter (fabric filter with catalytic media)[d]	0.012–0.065[e]	35–90	1–6.5	0.05	760	360	670	360

Replacement with alternative fuel

The cost of replacing a medium (5MW) boiler with a package gas-fired boiler would be approximately $350,000. For a probable dioxin reduction of over 90%, this would result in a discharge of less than 1 mg TEQ/year. The cost allows for removal of the old plant, but does not include the remaining value of the coal boiler. The annual increase in operating costs would be approximately $750,000 for the North Island, based on the cost of natural gas, and $1,730,000 for the South Island, based on the cost of LPG.

For those boilers where it is practicable to retrofit gas burners, the total installed cost of conversion would be approximately $35,000, with operating costs estimated to be 20% higher than indicated above to compensate for loss of efficiency.

A.16 Large coal-fired boilers

It is assumed that large coal-fired boilers have:

- a gross heat output of 30 MW
- multi-cyclone grit arrestors
- a baseline emission of 60 mg TEQ per year for 7,000 hours of operation[a]
- a discharge equivalent to a concentration of 0.1 ng TEQ/Sm^3 in the ventilation exhaust.

Table A.13 Large coal boiler control costs

Selected technology options	Reported emissions (ng TEQ/Sm^3)	Likely performance		Emission standard achieved (ng TEQ/Sm^3)	Costs (NZ$'000s)[c]			
		% reductions	mg TEQ/year		Existing (retrofit)		New	
					Total installed	Operating	Total installed	Operating
Fabric filter	06–0.08[b]	20–40	36–48	0.1	1,300	330	970	330
Catalytic fabric filter (fabric filter with catalytic media)[d]	0.012–0.065[e]	35–90	6–39	0.05	3,100	800	2,800	800

Replacement with alternative fuel

The cost of replacing a large (30 MW) boiler with a package gas-fired boiler would be approximately $2,100,000. For a probable dioxin reduction of over 90%, this would result in a in a discharge of less than 6 mg TEQ/year. The cost allows for removal of the old plant, but does not include the remaining value of the coal boiler. The annual increase in operating costs would be approximately $4,500,000, for the North Island, based on the cost of natural gas, and $10,350,000 for the South Island, based on the cost of LPG.

For those boilers where it is practicable to retrofit gas burners, the total installed cost of conversion would be approximately $200,000, with operating costs estimated to be 20% higher than indicated above to compensate for loss of efficiency.

A.17 Coal-fired boiler notes

a. Baseline emissions were determined from limited emission measurements undertaken in New Zealand and reported by Buckland *et al.* (2000). The figures taken are the largest of the emission factors measured under New Zealand conditions (1.95 µg/tonne of coal).

b. Emissions achieved with fabric filters are uncertain. The figures used are based on the performance figures for fabric filters installed on wood-fired boilers described in Section A.13. Emissions reported for a coal-fired power station fitted with ESPs (Buckland *et al.*, 2000) suggest significantly better performance is possible for coal boilers fitted with high-efficiency particulate collection equipment, but it is considered realistic to expect similar performance to that achieved by fabric filters in wood boilers.

c. Fabric filter costs for small and medium sized boilers are based on figures for coal boilers from Damiano and Campbell (1997).

d. The second level of dioxin control is based on the use of catalytic fabric filters. More common technology for this level of dioxin control in New Zealand is to use carbon injection, although this is on waste incineration plants. Little is known about dioxin controls at this level. However catalytic systems have been used in conjunction with NOx controls on very large boilers overseas. It therefore seems suitable to employ the recently developed catalytic filters in a baghouse. The performance of this technology is based on information from WL Gore Ltd on the Remedia D/F filter systems. The costs given for this system are based on the assumption that catalytic filters are about 20 times more expensive than the commonly used Nomex filters.

e. As discussed, the performance of catalytic fabric filters on coal-fired boilers is not known. The figures used are taken from the performance figures assumed for catalytic fabric filters on wood-fired boilers described in Section A.13.

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