

**SOURCES OF DIOXINS AND FURANS
IN AUSTRALIA:**

AIR EMISSIONS

**REVISED EDITION
MAY 2002**

Preface

Since the first publication in May 1998 of the report, “Sources of Dioxins and Furans in Australia: Air Emissions”, some additional Australian data on emissions from cement and lime production has been provided by the Cement Industry Federation and reviewed by Pacific Air & Environment, the principal authors of the original report. The information from this industry sector has been included in this Revised Edition.

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Executive Summary

Pacific Air & Environment was commissioned by Environment Australia's Environment Protection Group (EPG) to investigate the sources of polychlorinated dioxin and polychlorinated furan emissions in Australia. These two compound groups form two of the twelve persistent organic pollutants (POPs) short-listed in 1995 for an international investigation by the United Nations Environment Programme (UNEP) Governing Council. This report will form part of a larger global study which aims at identifying, quantifying and recommending technologies and strategies for the reduction of risks to human health and the environment arising from the environmental release of the twelve key POPs. Throughout the remainder of the report polychlorinated dibenzodioxin and dibenzofuran will be referred to as CDD and CDF, respectively.

Methodology

The basic methodology used in this study has been to review international dioxin/furan inventory studies, and from these studies collate a list of industries that release CDD/CDF compounds into the atmosphere. Emission factor information for each of the industries were then compiled and compared, and a final emission factor range was then assumed for Australia. Where possible, Australian source test data was also reviewed, however, reliance on international data is significant due to the general lack of this source test data. As a result of this the emission estimates made in this study are subject to uncertainty.

Activity data relating to each industry source was then collected from a number of sources. Where limited data was available best estimates of activity data were made, however, for some sources no activity data was found and so emissions from these industries could not be determined. Using the emission factors and appropriate activity data emissions were quantified for each industry.

Results

The emissions as quantified using the methodology above are presented in the Table E1 below. As discussed in the text, these estimates are subject to considerable uncertainties.

Based on the upper bound of each range, biomass combustion from prescribed burning and wild bushfires are potentially the most significant sources of CDD/CDF compounds in Australia. These two sources contribute approximately 75 percent to the total CDD/CDF estimates.

The second major group of sources are (in decreasing order of contribution):

1. residential wood combustion;
2. coal combustion (utility and industrial);
3. sinter production; and
4. industrial wood combustion.

Based on upper bound estimates, fossil fuel and biomass combustion contribute about 75% of the emissions from the second major group of sources, with sinter production accounting for the remaining 25%. These four sources contribute about 15% to the total estimates (based on upper bound estimates) and, when combined with prescribed burning and bushfires, account for approximately 95% of total dioxin emissions.

The contribution of emissions from motor vehicles is more in line with the US study, as both the UK and Netherlands studies found vehicles to be significant contributors. This study,

however, finds that motor vehicles are relatively minor sources, contributing less than one percent of total emissions.

Table E1 – Emission Estimates for Australian Sources.

Source	Emission (g/year)
Fires – prescribed burning	65-1300
Bushfires	7-400
Residential Wood Combustion	15-98
Coal Combustion	4.5-73
Sinter Production	9-68
Industrial Wood Combustion	10-65
Oil Combustion (industrial and utility)	21
Non-Ferrous Metal Production	1-19
Medical Waste Incineration	0.9-19
Motor Vehicles	0.35-17
Asphalt Mixing	14.5
Iron and Steel Production	0.7-9.4
Landfill Gas	0.8-2.5
Cement and Lime Production	0.31-0.60
Residential Oil Combustion	0.075-0.2
Ceramic	0.02-0.05
Coke Production	0.03
Glass	0.0014-0.0035
Municipal Waste Incineration	0
Sewage Sludge Incineration	ND ^a
Hazardous Waste Incineration	ND
Crematoria	ND
Activated Carbon Regeneration	ND
Halogen Chemicals and VCM	ND
Pesticide manufacture	ND
PCP Wood Treating	ND
Total	150-2100

a ND – no data available

Limitations

It is clear that this study is subject to various shortcomings. The principal sources are discussed below.

- The most significant limitation is the lack of source test data for Australian sources, resulting in a heavy reliance on international data. Emission factors as a tool for estimating emissions are inherently prone to uncertainties as they are typically based on limited testing of a source population. When applying these international emissions data to Australian sources this uncertainty is increased.
- In addition to the above point, some international emission factors were originally derived using assumed conversion factors to supplement data gaps. This adds further to the uncertainties in the original emission factors.
- The emission factors for a large number of sources span several orders of magnitude. This is indicative of the potentially large variations that are observed within a particular source category. With such large ranges it becomes difficult to identify significant

CDD/CDF contributors, particularly if the upper bound indicates that the source may be significant, while the lower bound indicates a minor contribution.

- Some source categories may have such variable process technologies, operational conditions etc. that it may be difficult to reliably predict emissions from these sources using limited data. Better characterisation of industry will enable the identification of these industry types.
- Emission factors for many industries are based on test data taken during very short sample periods. The emission results are likely to be reflective of relatively good combustion and operational practice and therefore may not be indicative of likely emissions during process upsets and/or abnormal operation.
- Several sources have been identified as having emission factors subject to uncertainties above and beyond those discussed in the points above. These are sources for which emissions data is scarce or non-existent, including glass and ceramic production, fires (prescribed and natural), and residential sources.
 1. Emission factors for glass and ceramic production are based purely on data developed for cement production (in the UK study), without any testing of actual facilities.
 2. Emissions from residential ‘wood burning stoves’ are currently assumed to apply to bushfires and the prescribed burning of agricultural waste and grasslands. Due to this lack of data, and considering the highly variable conditions under which these fires can burn emissions estimated from these sources are considered highly uncertain.
 3. In addition to the numerous assumptions used to derive residential emission factors (because of the lack of data), these sources are subject to increased uncertainty due to their uncontrolled, unregulated and widely varying nature of operation. Additionally, the emission estimates are extremely sensitive to assumed quantities of contaminated wood burned, and no information of this nature currently exists.
- Another limitation relates to the lack of activity data for a number of sources. This lack of data prevents the compilation of a comprehensive list of sources with emission estimates. Therefore, a complete picture of the Australian situation cannot be attained. Some of the sources not included are potentially significant contributors (such as waste incineration) and so their inclusion is important.
- An additional source that was identified as a potential CDD/CDF emitter is accidental fires. However, due to the lack of data it was not possible to derive emission estimates. This is a source that requires further consideration.

Considering these limitations, it is stressed that the emission estimates as determined by this study are **INDICATIVE** only of the likely CDD/CDF releases by various sources in Australia. The estimates have been developed based on the best information currently available. These estimates must be further refined as additional data is collected.

Conclusions

As discussed in the limitations of this study it is difficult to draw any definitive conclusions due to the large emission ranges for many industries and that fact that no activity data was available for some potentially large sources. Nonetheless, based on the sources currently included the total quantity of CDD/CDF released into the Australian atmosphere range between 150 and 2100 grams annually.

Potentially, the most significant emission sources appear to be prescribed burning (agricultural and grasslands) and bushfires. The term 'potentially' has been used as this finding is based on the upper bounds of the emission ranges. However, if the emissions actually lie towards the lower bound, while the emissions for other sources lie towards upper bounds then these sources may in fact not be the most significant.

Excluding the prescribed and natural fire emissions, the most significant anthropogenic sources are as follows (ranked from most to least significant):

1. residential wood combustion;
2. coal combustion (utility and industrial);
3. sinter production; and
4. industrial wood combustion.

These four sources contribute about 15% to the total emissions (based on upper bound estimates) and about 75% of anthropogenic sources. Combined with prescribed burning and bushfires, these sources account for approximately 95% of total emissions. However, these findings are limited by the large emission ranges in the source data.

The most significant non-industrial source is residential wood burning, ranked as the second highest anthropogenic source. Other sources such as motor vehicles, however, are not considered significant, contributing less than 1 percent to total emissions. Emissions from residential oil combustion are even less significant.

Recommendations

Based on then findings and limitations of this study, the following recommendations are made:

- Clearly the main limitation relates to the lack of Australian Source test data. It is therefore important that CDD/CDF testing be performed over a broad range of industries. The principal focus should initially be on those industries identified in this study as being 'potentially' significant. It is important that pertinent information such as feed or production rates, process technologies, operating conditions and pollution control equipment utilised be recorded with any test data. Information such as operating temperatures of flue gas control should also be included. Additionally, it is important that all source testing is performed in line with the appropriate CDD/CDF source testing protocols;
- The lack of activity data for a number of CDD/CDF sources is another major limitation of this study, particularly considering that some sources are potentially significant. It is therefore important that activity data on these sources be collected to at least allow a more complete indication of relative source contributions;
- Better characterisation of Australian industries will aid in developing an accurate picture of technologies and operating practices currently employed by particular industries, as well as control technologies used. This will be valuable in allowing better comparison with international data. Additionally, this type of characterisation will identify those industry categories where significant variation in process technologies etc. exist, and thus perhaps the need for more extensive testing within those categories. Even limited source testing combined with better industry characterisation will allow the development of more defined emission ranges;
- Emission sources such as prescribed and natural fires are quantified based on highly uncertain emissions data. The development of emission data for various prescribed

burning operations based on the type of waste burned etc. will allow more definitive estimates to be made in this area. Additionally, attempting to derive more definitive emissions data for bushfires will be advantageous. It is recognised, however, that this may be a difficult task; and

- Better characterisation of the residential combustion of wood will enable more reasonable estimates of clean and treated wood usage. Currently emissions estimates are extremely sensitive to the assumed quantities of treated wood burned. Additionally, international emissions data for residential wood combustion are subject to significant uncertainties, particularly for combustion of treated wood. Testing of residential wood combustion equipment using various wood types will allow the development of more definitive emission factors.

1 INTERNATIONAL BACKDROP

Persistent organic pollutants (POPs) are organic compounds of natural or anthropogenic origin that resist photolytic, chemical, and biological degradation. They are characterised by low water solubility and high lipid solubility, resulting in bioaccumulation in fatty tissues of living organisms. POPs are semi-volatile and, therefore, able to move long distances in the atmosphere, and are also transported in the environment in low concentrations by movement of fresh and marine waters, resulting in widespread distribution across the earth, including in regions where they have never been used or generated. Thus, both humans and environmental organisms are exposed to POPs around the world, in many cases for extended period of time. (Final Report, IFCS Ad Hoc Working Group on POPs, Manila, Philippines, 21-22 June 1996)

These materials are also known to produce a wide range of toxic effects in living organisms, even at low exposure levels. Some of these effects include birth defects, cancers, and dysfunctional immune and reproductive systems.

In May of 1995, as a result of growing international concern, the United Nations Environment Programme (UNEP) Governing Council agreed to adopt a decision to initiate an international assessment of the potential sources of POPs. Initially, the assessment is based on the following list of POPs:

- | | |
|-----------------------|---|
| 1. aldrin; | 7. mirex; |
| 2. DDT; | 8. chlordane; |
| 3. dioxins; | 9. dieldrin; |
| 4. furans; | 10. heptachlor; |
| 5. endrin; | 11. polychlorinated biphenyls (PCBs); and |
| 6. hexachlorobenzene; | 12. toxaphene. |

In October 1995, another international organisation, the Intergovernmental Forum on Chemical Safety (IFCS) established an *ad hoc* working group of representatives from governments, non-governments, industry and community interest groups throughout the world to further understand and discuss the nature and extent of the international presence of POPs. The outcome of this effort, culminated in a meeting in Manila June 1996 was the establishment of a global instrument designed to actualise realistic international actions including reduction and in some cases, elimination programs for POPs.

Since the introduction of this concept, the UNEP Governing Council through the creation of an international negotiating committee (INC) has conducted further work. INC's main goal is to minimise risks to human health and the environment through the negotiation of a global legally-binding instrument. Efforts for the development of this instrument will commence in early 1998. The INC will also establish an expert group to develop science-based criteria and a procedure for the identification of additional POPs as candidates for future international action. UNEP recognises the need to "use separate differentiated approaches to take action on pesticides, industrial chemicals, and unintentionally produced by-products and contaminants".

2 INTRODUCTION

In light of Australia's upcoming commitment to UNEP's international legally-binding instrument, Pacific Air & Environment was commissioned by the Environment Protection Group (EPG) of Environment Australia to investigate the potential sources of emissions of dioxins and furans throughout the country. As indicated in the previous section, dioxins and furans are two of the twelve POPs short-listed by the UNEP Governing Council in 1995.

The results of the investigation will provide EPG with a basis to assist in furthering the understanding of the types and sources potentially identified with the release of dioxin and furan emissions in the Australian environment. This report will form part of a larger global study which aims at identifying, quantifying and reducing the risks to human health arising from the environmental release of the twelve key POPs.

The objective of this report is to:

- a) gather, collate and critically analyse information on the sources of dioxin and furan emissions; and
- b) identify and address the effectiveness of best available technologies suitable for the reduction of dioxin/furan emissions.

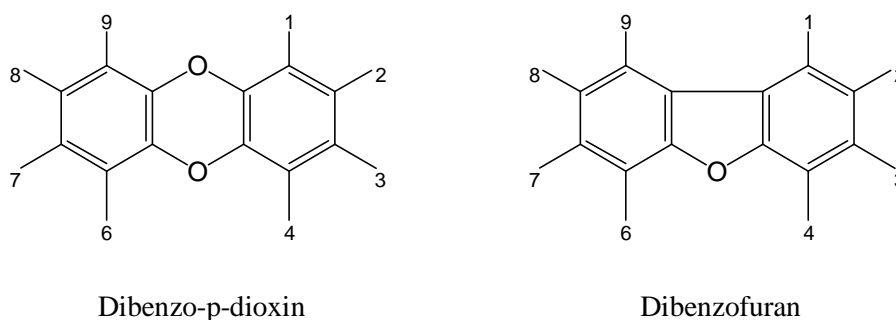
The study will focus on source identification, emission quantification, and discuss the methods available for reducing the release of these compounds into the environment. Although dioxins and furans are released into various environmental media, this report will focus principally on releases into the atmosphere. Additionally, note that throughout this report chlorinated dioxins and furans will be referred to as CDD and CDF, respectively.

3 DIOXIN AND FURAN COMPOUNDS

3.1 General Discussion

A dioxin is any compound containing the dibenzo-p-dioxin nucleus, while a furan is any compound containing the dibenzofuran nucleus. The general formulae for each of these compounds are presented in Figure 1 below.

Figure 1 – General Formulae of Dioxins and Furans.



Each of the positions numbered 1 through 4 and 6 through 9 can be substituted with a chlorine or other halogen atom, an organic radical, or a hydrogen atom.

The principal focus of studies concerning CDD and CDF compounds relate to the chlorinated species, as these are the compounds of most environmental interest and concern. Chlorinated species are those that have a chlorine atom occupying one or more of the eight positions shown in Figure 1 above. In all, there are 210 different chlorinated dioxin and furan isomers. A particular isomer depends on the number of chlorine atoms, and where the chlorine atoms are attached (i.e. which of the eight positions). Thus, a group of isomers have the same chemical composition, but the chlorine atom(s) are attached in differing places.

In this report, the following abbreviations will be used to denote different numbers of chlorine atoms present on dioxins and furans:

- T – tetra (i.e. four chlorine atoms attached);
- Pe – penta (i.e. five chlorine atoms attached);
- Hx – hexa (i.e. six chlorine atoms attached);
- Hp – hepta (i.e. seven chlorine atoms attached); and
- O – octa (i.e. eight chlorine atoms attached).

Physical properties of the various CDD/CDF compounds indicate decreasing volatility with increasing number of chlorine atoms. These compounds have fairly low water solubilities, with furans exhibiting greater solubility than dioxins (particularly for halogenated species).

The most toxic, and consequently the most extensively studied of the dioxins is 2,3,7,8-tetrachlorodibenzo-p-dioxin (i.e. 2,3,7,8-TCDD). This compound is extremely lipophilic, exhibiting a high degree of solubility in fats, oils, and relatively non-polar solvents. Other dioxins and furans tend to be quantified in terms of an 'equivalency factor' (discussed further in Section 3.3). Total emissions from a particular source are then discussed in terms of total equivalent (TEQ) TCDD emitted.

3.2 Health Effects

Health effects arising from CDD/CDF compounds are not the primary focus of this study. Therefore, only a brief overview of the human health effects of CDD/CDF compounds will be given in this report. Furthermore, discussing the toxicological pathways that lead to health effect endpoints is deemed beyond the scope of this study.

As mentioned above, TCDD is the most extensively studied of the CDD/CDF compounds. It is believed that chronic exposure to TCDD can induce a variety of cancers including lymphoma and soft tissue sarcomas, and respiratory system cancer. However, these findings are inconclusive due to other confounding factors such as exposure to other toxicants, ethnic background, and diet etc. (NPI Website). The carcinogenic potential of CDD/CDF compounds is still a source of great discussion (refer below).

In addition to the carcinogenic potential of TCDD, the following health effects have been reported arising from acute exposure to TCDD (NPI Website):

- irritation of the eyes, skin (forming acne-like lesions which may persist for up to year following cessation of exposure) and respiratory tract;
- personality changes;
- loss of energy;
- impairment of vision, taste and muscular coordination;
- nausea;
- vomiting;
- headaches;
- severe muscular aches;
- emotional instability; and
- sleep disturbances.

Note that the Chloracne disease, which results in acne-like skin lesions, has been linked with exposure to most of the dioxin compounds in addition to TCDD (NPI Website).

Recent research on the health effects of dioxin indicates the following (Greenpeace, 1996):

- in fish, birds, mammals and humans, the developing foetus/embryo appears to be very sensitive to the toxic effects of dioxin (referred to as CDD/CDF compounds in this study); and
- animal and/or human studies have shown effects such as cellular changes in the immune system, changes in the levels of male testosterone, and changes in other enzymes and hormones.

The Greenpeace report (1996) also discusses that by blocking the actions of estrogen or altering other hormones, CDD/CDF compounds could disrupt reproduction or sexual development in humans or wildlife. Furthermore, the US Bureau of National Affairs (Chemical Regulation Reporter, 1997) discussed the cancer dose-response analysis with Dr. Christopher Portier, Chief of Laboratory Computational Biology and Risk Analysis at the National Institute of Environmental Health Sciences, a member of the dose-response analysis team of the 'Dioxin Reassessment' study (refer below). Dr. Portier stated that dioxins alter a person's metabolic pathway, which is of particular concern for women as the pathway affected is estrogen-based. After exposure to dioxins the body produces more of a particular enzyme that interacts with the estrogen to create a different, biologically destructive form of estrogen that may lead to the DNA damage that triggers cancer.

There is currently debate as to the health effects of dioxin, particularly with regards to its carcinogenic activity. An extensive US study termed the 'Dioxin Reassessment' has been underway for many years assessing the large body of literature on dioxins and their toxicity in animals and humans. The USEPA Science Advisory Board (SAB) in its review of the dioxin reassessment study stated that in the case of dioxins, animal studies would be categorized as sufficient and the studies of humans as limited, providing for an overall categorization of B1 (USEPA, 1995a). In the categories used to define carcinogens, category B1 would be expressed as '*Probably Carcinogenic* to humans with limited supporting information from human studies'. The SAB indicate the term 'dioxin' includes all 2,3,7,8-substituted dioxins and furans.

3.3 Toxicity Equivalency Concepts

Although this may be classified as a method used for simplifying the quantification of CDD and CDF emissions, it is a fundamental concept that is commonly used when discussing the release of CDD/CDF compounds into the environment. For this reason it will be discussed earlier in this section. Section 11 discusses other methodologies used for estimating CDD/CDF emissions from release sources.

As mentioned above, the toxicity of dioxins and furans are typically related back to a common 'toxicity equivalency' based on 2,3,7,8-TCDD. This is more important when dealing with exposures of complex mixtures of dioxins and furans, as it helps to simplify the analysis. In 1989, as a result of the active involvement of the US EPA in an international effort aimed at adopting a common set of toxicity equivalency factors (i.e. TEFs), International TEFs were implemented and referred to as 'I-TEFs/89' (USEPA, 1995b).

A strong structure-activity relationship exists between the chemical structure of a particular dioxin/furan compound and its ability to elicit a biological/toxic response in various *in vivo* and *in vitro* test systems. Congeners in which the 2,3,7, and 8 lateral positions are occupied with chlorine atoms (additional positions to these four may also be occupied) are much more active than are the other non-2,3,7,8-substituted dioxins/furans.

Available data on short-term *in vitro* toxicity studies for dioxins/furans are used to supplement the lack of long-term *in vivo* results for these compounds. These toxicity estimates, expressed in terms of toxic equivalents (TEQs), or equivalent amounts of 2,3,7,8-TCDD, are generated by using the TEF to convert the concentration of a given dioxin and/or furan into an equivalent concentration of 2,3,7,8-TCDD. The I-TEQs/89 are obtained by applying the I-TEFs/89 to the congener-specific data and summing the results. In assigning TEFs, priority is normally given to the results from long-term studies followed by the results from short-term, whole-animal studies. Among the remaining short-term *in vivo* and *in vitro* data, the results of enzyme induction studies take high priority because a good correlation has generally been observed between enzyme induction activity and short-term, whole-animal results.

The I-TEF/89 approach expresses the TEFs as a rounded order of magnitude because, with the exception of the I-TEF/89 for PeCDF, the I-TEFs/89s are only crude approximations of relative toxicities. A value of 0.5 is assigned to 2,3,4,7,8-PeCDF; 1,2,3,7,8-PeCDF is assigned a value of 0.05. This higher value for the 2,3,4,7,8-PeCDF is supported by data from *in vivo* and *in vitro* studies and is the only instance in which the I-TEFs/89s depart from the guiding principle of simplicity in which TEFs are expressed as rounded orders of magnitude. The I-TEF/89 scheme assigns a value of zero to non-2,3,7,8-substituted homologues. For example, a value of 0.5 is assigned to 1,2,3,7,8-PeCDD, but the 13 other PeCDD (non 2,3,7,8-substituted compounds) are assigned a value of zero (refer to Table 1).

In general, an assessment of the human health risk of a mixture of CDD and CDF using the TEF approach involves the following steps (USEPA, 1995b):

1. Analytical determination of the CDD and CDF in the sample;
2. Multiplication of CDD/CDF compound concentrations in the sample by the appropriate TEFs to express the concentration in terms of 2,3,7,8-TCDD equivalents;
3. Summation of the products in Step 2 to obtain the total 2,3,7,8-TCDD equivalents in the sample;
4. Determination of human exposure to the mixture in question, expressed in terms of 2,3,7,8-TCDD equivalents; and
5. Combination of exposure from Step 4 with toxicity information on 2,3,7,8-TCDD (usually carcinogenicity and/or reproductive effects) to estimate risks associated with the mixture.

In cases where the concentrations of CDD/CDF are known:

$$\begin{aligned}
 \text{2,3,7,8-TCDD Equivalents} = & \Sigma (\text{TEF of each 2,3,7,8-CDD/CDF compound} \\
 & \times \text{the concentration of the respective compound}) \\
 & + \Sigma (\text{TEF of each non-2,3,7,8-CDD/CDF compound} \\
 & \times \text{the concentration of the respective compound})
 \end{aligned}$$

Table 1 – International Toxicity Equivalency Factors (I-TEFs) for CDD/CDF. ^a

Compound	I-TEF
Mono-, Di-, and Tri- CDDs	0
2,3,7,8-TCDD	1.0
Other TCDD's	0
1,2,3,7,8-PeCDD	0.5
Other PeCDD's	0
1,2,3,4,7,8-HxCDD	0.1
1,2,3,7,8,9-HxCDD	0.1
1,2,3,6,7,8-HxCDD	0.1
Other HxCDD's	0
1,2,3,4,6,7,8-HpCDD	0.01
OCDD	0.001
Mono-, Di-, and Tri-CDFs	0
2,3,7,8-TCDF	0.1
Other TCDF's	0
2,3,4,7,8-PeCDF	0.5
1,2,3,7,8-PeCDF	0.05
Other PeCDF's	0
1,2,3,4,7,8-HxCDF	0.1
1,2,3,6,7,8-HxCDF	0.1
1,2,3,7,8,9-HxCDF	0.1
2,3,4,6,7,8-HxCDF	0.1
Other HxCDF's	0
1,2,3,4,6,7,8-HpCDF	0.01
1,2,3,4,7,8,9-HpCDF	0.01
Other HpCDF's	0
OCDF	0.001

^a Source: USEPA (1995b)

The abbreviations for each of the compounds are discussed in Section 3.1. As discussed above, non-2,3,7,8-substituted CDD/CDF compounds are not found to be as active as the 2,3,7,8-substituted compounds, and thus, the I-TEFs assigned equal zero.

4 EXPOSURE TO CDD/CDF COMPOUNDS

Human exposure to CDD/CDF compounds arises through various pathways. The purpose of this section is to briefly discuss the principal exposure pathways that result in the eventual intake by the human body. The three principal exposure pathways are (USEPA, 1994b):

1. ingestion of soil, water, beef, dairy products, fish, fruit and vegetables;
2. dermal contact with soil; and
3. inhalation of particulates and vapours.

It is considered that the spreading of CDD/CDF compounds via the air is the most significant pathway through which humans are exposed to these pollutants. A Netherlands study on dioxin emissions (Bremmer et.al., 1994) states that the air is the most important route for dispersing CDD/CDF compounds into the environment. Additionally, the USEPA Science Advisory Board (SAB) indicate agreement with the 'Dioxin Reassessment' findings that the air-to-plant-to-animal pathway is most probably the primary way in which the food chain is impacted and humans are exposed (USEPA, 1995a). As mentioned in the introduction, air emissions are the principal focus of this study.

Atmospheric levels of CDD/CDF compounds can impact all exposure pathways, either directly through inhalation of airborne vapours and dust, or indirectly through deposition. This results in the presence of these compounds on and in soil and plants, which can then be taken up by animals (i.e. beef cattle). Furthermore, through more complex mechanisms deposition can result in uptake by vegetation (i.e. fruits and vegetables). Deposition can also result in these compounds entering surface waters, either directly or through run-off from contaminated lands, which may subsequently lead to bioaccumulation in fish species (CDD/CDF emissions directly to water can also contribute to this). However, a detailed discussion on the various mechanisms of how the compounds are taken up and absorbed by plants and animals is very complex and will not be covered in this report.

5 SOURCES OF CDD/CDF COMPOUNDS

Before the sources of CDD/CDF compounds are discussed in further detail, it is important that the reader understands the difference between an 'emission source' and an 'exposure source':

1. An *emission source* is identified as a point(s) from which CDD/CDF compounds are generated and subsequently released into the environment. Industrial sources (i.e. from a process stack), and natural sources such as bushfires may be classified as emission sources.
2. An *exposure source*, however, is a point(s) from which CDD/CDF compounds may enter the human body. These sources include foods containing CDD/CDF compounds (such as fish and dairy products), and environmental media such as the air, water and land (i.e. soil).

Therefore, emission sources inevitably lead to the production of exposure sources. A source of exposure to CDD/CDF compounds is the final stage of a particular exposure pathway. Exposure pathways are discussed in greater detail in Section 4. The principal focus of this section is the identification and discussion of emission sources.

Studies have been performed investigating the quantities of CDD/CDF compounds emitted from the known sources of dioxin emissions, and comparing these emission estimates with atmospheric deposition estimates. USEPA (1994a) reviews studies performed in Sweden (Rappe, 1991) and the UK (Harrad et.al., 1992a and 1992b). These two studies concluded that current emission sources account for atmospheric levels well below (between 2 and 20 times) those actually observed. An additional and more recent study was reviewed in this study (Brzuzy and Hites, 1996), which arrived at the same conclusions as for the earlier studies. Reasons for the discrepancies in these 'mass balance' type studies include:

- uncertainty in the emission estimates;
- uncertainty in the deposition estimates;
- long-range transport of CDD/CDF compounds from areas outside the study areas; and
- the existence of unidentified sources.

Thus, a great deal of discussion still exists surrounding the sources of dioxin emissions. No attempt has been made in this study to equate mass emission rates of CDD/CDF compounds with deposition rates.

In this report, CDD/CDF emission sources are divided into the following two broad categories:

1. anthropogenic sources; and
2. natural sources.

Several studies investigating CDD/CDF emissions have indicated that anthropogenic sources are the most significant contributors to currently observed environmental levels (Thomas & Spiro (1996), Brzuzy & Hites (1996)). Additionally, the USEPA Science Advisory Board (SAB) indicate agreement with the 'Dioxin Reassessment' findings that environmental levels of CDD/CDF compounds are derived primarily from anthropogenic sources (USEPA, 1995a). The conclusions by these studies are based on the observation that environmental levels of CDD/CDF compounds have increased significantly since the commencement and widespread development of industrial sources (combustion sources in particular). However,

it is also acknowledged that little is known regarding natural sources such as bushfires, and their contribution to CDD/CDF emissions.

6 ANTHROPOGENIC SOURCES

Anthropogenic sources identified by various international studies as emitters of CDD/CDF emissions are presented in Table 2 below. The emission sources have been ranked in order, descending from the most significant.

Table 2 – CDD/CDF Sources Identified by International Studies.

US ^a	UK ^b	Netherlands ^c
Municipal waste combustion	Municipal waste combustion	Municipal waste combustion
Medical waste incineration	Medical waste combustion	Sintering Processes
Portland cement manufacture	Coal combustion (industrial)	Use of wood preservatives
Industrial wood combustion	Sinter plants	Hazardous waste incineration
Secondary copper smelters	Traffic	Wood Combustion
Residential coal combustion	Iron and steel plants	Traffic
Utility coal combustion	Coal combustion (domestic)	Metal Industry
Open burning/fires	Non-ferrous metal plants	Coal Combustion
On-road mobile vehicles	Crematoria	Various high temp. processes
Residential wood combustion	Wood combustion (domestic)	Hospital waste incineration
PCP in timber processes	Natural fires	Cable and electromotor burning
Sewage sludge incineration	Cement manufacturing	Oil combustion
Iron and steel foundries	Straw combustion	Chemical production processes
Secondary aluminium smelters	Chemical waste combustion	Asphalt mixing installations
Utility residual oil combustion	Sewage sludge combustion	Landfill, biogas and sludge incineration
Hazardous waste incineration	Landfill gas combustion	Crematoria
Secondary lead smelters	Wood combustion (industrial)	
Residential distillate fuel combustion	Waste oil combustion	
Drum and barrel reclamation/incineration	Lime manufacture	
Kraft pulp and paper – black liquor combustion	Coke production	
Waste tire incineration	Tyres combustion	
Carbon regeneration/ reactivation	Asphalt mixing	
Crematoria	PCP in timber processes	
	Pesticide production	
	Ceramic production	
	Halogenated chemicals	
	Glass manufacture	
	Carbon regeneration	

a Source: USEPA, 1995b.

b Source: HMIP, 1995

c Source: Bremmer et.al., 1994.

There may be differences in the estimation methodologies used between the studies, and it is also likely that the confidence in the data used in each study varies. However, as it stands in the table above, it is clear that each study identifies municipal and medical waste incineration as the two principal emitters of CDD/CDF compounds.

The principal aim of this section is to describe each of the principal sources of chlorinated dioxins and furans in Australia. A limited amount of work has been undertaken investigating CDD/CDF sources in Australia (Commonwealth Env. Dept. (1990), Greenpeace (1996)), however, no attempts were made at quantification. Section 8 of this study attempts to make gross estimates of the quantities of CDD/CDF compounds released into the atmosphere in Australia, including a brief comparison of results with international studies.

It is believed that the anthropogenic sources relevant to Australia are as follows:

1. incineration processes;
2. combustion processes;
3. iron and steel manufacturing plants;
4. non-ferrous metal plants;
5. crematorium facilities;
6. cement manufacturing plants;
7. coke production plants;
8. chemical manufacturing (both organic and inorganic);
9. wood treating facilities;
10. carbon regeneration/reactivation;
11. mobile sources (i.e. vehicles);
12. ceramic manufacture;
13. glass manufacture; and
14. open (i.e. prescribed) burning.

Although pulp and paper mills are considered to be sources of CDD/CDF compounds, the principal medium of release is via liquid effluents. Therefore, this release type from these industries is not included in this study, as the principal focus here is atmospheric emissions. However, emissions also arise from this industry through the combustion of wood waste and other waste products (such as black liquor). This is included under 'wood' combustion for this study.

These sources were short-listed based on the findings of the international studies, as presented in Table 2 above. Each of these sources will now be discussed in the following sections. The principal focus of the discussion will be how the emissions are generated, and whether any emission factors have been developed.

As done in various studies (Commonwealth Env. Dept. (1990), HMIP (1995)), anthropogenic emission sources will be divided into the following categories:

- thermal processes; and
- chemical processes.

Thermal processes typically involve very high temperatures (i.e. 800°C and above), but with the potential for some parts of the process train to be maintained at lower temperatures. Essentially all of the processes listed above fall into the 'thermal' category, excepting sources 8, and 9. Due to the significance of combustion sources in particular as CDD/CDF emitters, a brief discussion on the formation of these compounds during combustion is given in Section 6.1.1.

The following conditions have been identified as conducive to the formation of CDD/CDF compounds during thermal processes (HMIP, 1995):

- the presence of chlorine or chlorine-containing compounds;
- temperatures in the system of between 200 - 400°C; and
- pollution control equipment that is operated at temperatures of between 200 - 400°C.

While the above conditions are optimal, CDD/CDF compounds may also be formed (although in much smaller quantities) at combustion temperatures between 800 - 1200°C.

Chemical processes are those used in the manufacture of both organic and inorganic chemicals. CDD/CDF compounds are generated as by-products, and become trace contaminants in the manufacture of a number of chemicals. Sources 8, 9 and 10 in the list above fall into this category. The conditions that are considered to most likely result in the production of CDD/CDF compounds are (HMIP, 1995):

- processes where chlorine is present, either as a reactant, a constituent of a reactant, or an impurity in the feedstock;
- process temperatures greater than 150°C; and
- the presence of alkaline conditions.

Note that a significant fraction of the CDD/CDF compounds formed remain in the chemical product, which is typically either a liquid or solid. Therefore, the principal mode of release is not necessarily via the atmosphere, and CDD/CDF compounds may in fact be discharged via liquid effluent or solid residues.

There are three principal international studies from which emissions information has been attained. These studies were performed in the US (USEPA, 1995b), the UK (HMIP, 1994) and the Netherlands (Bremmer et.al., 1994). Each study contains a compilation of emissions information based on national and international data. Using this data emission estimates are derived for the sources identified as CDD/CDF emitters. Emissions data presented in the following sections are based on these studies, and so unless otherwise indicated any reference to 'US', 'UK' and 'Netherlands' emissions data refers to these reports.

6.1 Thermal Processes

Before thermal processes are discussed any further, it is important to discuss how CDD/CDF compounds are formed during combustion operations. This is briefly discussed first, with the following sections covering each of the emission sources included within the 'thermal processes' category.

Due to the limited characterisation of each industry with respect to the various process technologies employed and operating characteristics, the 'process descriptions' given for each industry are meant to be general overviews only. It must be noted that variations to some processes will exist.

6.1.1 CDD/CDF Formation During Combustion

Many studies have been undertaken investigating the mechanisms by which CDD/CDF compounds are formed in combustion processes. However, the specific mechanisms of formation are considered very complex and are not completely understood nor agreed upon (USEPA (1994a), and HMIP (1995)).

The following three possibilities have been proposed to explain the presence of CDD/CDF compounds in the flue gas from combustion equipment (these should not be regarded as mutually exclusive):

1. The CDD/CDF compounds are already present in the raw feed material and are incompletely destroyed in the combustion process;
2. The presence of chlorinated precursor compounds that structurally resemble the CDD/CDF molecules (such as chlorinated aromatic hydrocarbons – eg. PCB's, chlorinated phenols and chlorinated benzenes) form CDD/CDF compounds via thermal breakdown and molecular rearrangement processes; and
3. The CDD/CDF compounds form through *de novo* synthesis. This means that organic and inorganic compounds which are chemically unrelated to the CDD/CDF molecules combine to produce the emissions.

The first two mechanisms are considered as 'in-furnace' formation mechanisms, while the *de novo* synthesis is the relevant mechanism for 'downstream' formation in the flue gases (USEPA, 1995b).

Extensive studies have been performed investigating each of these mechanisms (USEPA, (1994a)). It was proposed that the first mechanism might account for a very minor proportion of the CDD/CDF compounds in the flue gas, but by no means the main proportion. Studies pertaining to the second and third mechanisms have proposed that fly ash in the flue gas acts as a catalyst in the formation of CDD/CDF compounds. There are, however, many variables that have been identified as affecting the formation of CDD/CDF, and it is still considered a very complex issue. The UK study (HMIP, (1995)) indicates that reactions catalysed on flyash are thought to be the most prominent mechanism for CDD/CDF formation.

6.1.2 Municipal Waste Incineration

6.1.2.1 Process Description

Municipal wastes are combusted for various reasons. Typically, however, the reason is to minimise the waste volume prior to final disposal. In the process of incinerating the waste, heat may be recovered and subsequently used for energy production. The three main types of combustors (USEPA, 1995b) are as follows:

1. mass burn;
2. modular burn; and
3. refuse-derived fuel.

Within these three major combustor categories, there are a number of different designs. The more common designs and their associated processes are briefly described below.

Emissions from municipal waste incinerators can vary significantly depending on the composition of the waste, the combustion practices, the operating practices of control devices used, and other factors (USEPA, 1995b).

Mass Burn Combustors

Mass burn combustors use gravity or mechanical ram systems to feed MSW onto a moving grate where the waste is combusted. Historically, mass burn combustors have been used to combust MSW that has not been pre-processed except to remove items too large to go

through the feed system. Waste that has been processed to remove recyclable materials can also be combusted in these units.

Mass burn combustors can be further divided into the following:

- mass burn/waterwall (MB/WW);
- mass burn rotary waterwall combustors (MB/RC); and
- mass burn/refractory-wall (MB/REF) designs.

Newer units are mainly waterwall designs, which are used to recover heat for production of steam and/or electricity.

Mass Burn Waterwall Combustors: Waste is delivered by an overhead crane to a feed hopper that feeds the waste into the combustion chamber. Most modern MB/WW facilities have reciprocating or roller grates that move the waste through the combustion chamber. The primary purpose of all types of grates is to agitate the waste bed to ensure good mixing of the waste with undergrate air and to move the waste uniformly through the combustor. The combustor walls are constructed of metal tubes that contain pressurized water and recover radiant heat from the combustion chamber.

Mass Burn Rotary Waterwall Combustor: Waste is conveyed to a feed chute and fed to the rotary combustion chamber. The rotary combustion chamber sits at a slight angle and rotates at about 10 revolutions per hour, causing the waste to advance and tumble as it burns. The combustion cylinder consists of alternating watertubes and perforated steel plates. Heat recovery occurs in the rotary chamber water tubes, the boiler waterwall, and various other sections through which the flue gases pass.

Mass Burn Refractory-Wall Combustors: MB/REF combustors have several designs. One design involves a batch-fed upright combustor that may be cylindrical or rectangular in shape. This design does not provide for agitation or mixing of the waste.

A second, more common design consists of rectangular combustion chambers with travelling, rocking, or reciprocating grates. The travelling grate moves on a set of sprockets and provides agitation to the waste bed as it advances through the combustor. Waste burnout is inhibited by fuel-bed thickness and there is considerable potential for unburned waste to be discharged into the bottom ash pit unless fuel feeding, grate speeds, and combustion air flows and distributions are well controlled. Some designs incorporate rocking or reciprocating grates that agitate and aerate the waste bed as it advances through the combustor chamber, thereby improving contact between the waste and combustion air and increasing the burnout of combustibles. A rotary kiln may be added to the end of the grate system to complete combustion.

Modular Combustors

Modular combustors are similar to mass burn combustors in that the waste burned has not been pre-processed, but modular combustors are generally smaller in size [4.5 to 103 Mg/day] and are shop-fabricated. The most common type of modular combustor is the starved-air or controlled-air type (MOD/SA). Another type, which is similar from a combustion standpoint to the larger MB/WW systems, is referred to as an excess-air combustor (MOD/EA).

Modular Starved-Air Combustors: The basic design includes two separate combustion chambers, referred to as the primary and secondary chambers. Waste is batch-fed to the primary chamber by a hydraulically activated ram and is moved through the chamber by

either hydraulic transfer rams or reciprocating grates. Waste retention times in the primary chamber are long, lasting up to 12 hours.

The walls of both the primary and secondary combustion chambers are refractory-lined. Early MOD/SA combustors did not include heat recovery, but a waste heat boiler is common in newer units, with two or more combustion modules sometimes manifolded to a shared boiler.

Modular Excess-Air Combustors: The basic design is similar to that of MOD/SA units and includes refractory-lined primary and secondary combustion chambers and a boiler to recover waste heat. Facilities with multiple combustors may have a tertiary chamber where flue gases from each combustor are mixed prior to entering the heat recovery boiler.

Refuse-Derived Fuel-Fired Combustors

RDF is MSW that has been processed to varying degrees, from simple removal of bulky and non-combustible items accompanied by shredding, to extensive processing to produce a finely divided fuel suitable for co-firing in pulverized coal-fired boilers. Processing MSW to RDF generally raises the heating value of the waste slightly because many of the non-combustible items have been removed.

There are three major types of RDF-fired combustors:

- dedicated RDF combustors;
- coal/RDF co-fired combustors; and
- fluidised-bed combustors (FBCs).

Dedicated Refuse-Derived Fuel-Fired Combustors: Most combustors that are designed to burn RDF as a primary fuel are boilers that use spreader-stokers and fire RDF in a semi-suspension mode. RDF is fed into the combustor through a feed chute using air-swept distributors, which allows a portion of the feed to burn in suspension and the remainder to burn out after falling on a horizontal travelling grate. The travelling grate moves from the rear to the front of the furnace and distributor settings are adjusted so that most of the waste lands on the rear two-thirds of the grate. This allows more time for combustion to be completed on the grate. Underfire air and overfire air are introduced to enhance combustion, and these combustors typically operate at 80 to 100 percent excess air. Waterwall tubes, a superheater, and an economizer are used to recover heat for production of steam and/or electricity.

Co-fired Combustors: RDF can be co-fired in various types of coal-fired boilers, including pulverized coal-fired and cyclone-fired boilers. In a pulverized coal-fired system, coal is pulverized into a powder and injected into the combustor through burners located on the combustor walls. RDF with a particle size of 5 cm or less in diameter is introduced into the combustor by air transport injectors that are located above or even with the coal injectors. A significant portion of the larger, partially burned particles become disengaged from the gas flow and fall onto grates at the bottom of the furnace where combustion is completed. Most RDF/pulverized coal-fired units operate with 50 percent excess air, in contrast to units firing coal alone, which may use as little as 25 percent excess air. Furnace exit temperatures are generally in excess of 1,095°C, which is higher than in other MWCs.

In an RDF/coal-fired, cyclone-fired combustor, crushed coal is injected into one end of a horizontal combustion cylinder. Primary air (about 20 percent of the total combustion air) is introduced tangentially to the burner, which causes the coal to move in a swirling pattern. The RDF is injected into the combustion chamber along with the secondary air in the same tangential direction through ports in the top of the cylinder. The cyclone operates at temperatures exceeding 1,370°C, which melts the coal and RDF ash into a liquid slag. Because of the swirling motion, most of the incoming coal and RDF gets caught in the slag layer on the combustor walls, where it burns rapidly.

Fluidized-Bed Combustors: In an FBC, waste is combusted in a turbulent bed of non-combustible material such as limestone, sand, silica, or aluminium. The RDF may be injected into or above the bed through ports in the combustor wall. Other wastes and supplemental fuel may be blended with the RDF outside the combustor or added through separate openings. The combustion bed is suspended or "fluidized" through the introduction of underfire air at a high flow rate. Overfire air is used to complete the combustion process. Waste-fired FBCs typically operate at 30 to 100 percent excess air levels and at bed temperatures around 815°C.

6.1.2.2 Emission Data

Emissions data for the US, UK and Netherlands are presented in Table 3 below. The emissions information presented in the table is discussed briefly beneath the table.

Table 3 – International Emissions Data for Municipal Waste Incineration.

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne waste burned)
US ^a - Emission range - Average	0.2-600 90
UK ^b - Old plant - New Plant	184-231 0.8
Netherlands ^c - Emission Range - Average	7-277 138

a Source: USEPA, 1995b.

b Source: HMIP, 1995.

c Source: Bremmer et.al., 1994.

US Emission Data

The USEPA assume a ratio of TEQ to total CDD/CDF emissions of 60 to 1, and determined the average presented in Table 3 above using baseline estimates of total CDD/CDF emissions performed in 1991, combined with activity data. Table A1 in Appendix A presents US emission factors by incinerator technology type. The original US emissions data were presented as total CDD/CDF emissions, however, these were subsequently converted to I-TEQ for this study, using the 60 to 1 ratio. The emission ranges shown in Table 3 are based on these incinerator type emissions.

UK Emission Data

The UK study categorised emissions according to the age of the incinerator (i.e. old and new). The emission factors presented in Table 3 were developed based on source test data

collated from UK municipal incinerators. This data was supplied as flue gas concentrations, which were subsequently converted to emission factors assuming a flue gas generation rate of about 5100 Nm³/tonne of waste processed. No additional information was specified which indicated as to what 'new' or 'old' meant. However, only one incinerator was considered 'new', and so it is likely that the plant was constructed in the last 5-10 years.

Netherlands Emission Data

The average emission rate specified in Table 3 was determined as a weighted average based on information regarding each municipal waste incinerator considered in the study (i.e. considering production rate and emissions data from each incinerator). The emission range is simply the lowest and highest emission rates from the facilities included in the study.

The Australian Situation

Note that municipal waste incineration does not currently operate in Australia. As it is considered a potentially significant source internationally, it has been included for informational purposes. In the event that new facilities become operational it is likely that they are designed such that CDD/CDF emissions are low. However, if existing plants become operational again without upgrades, the emissions data presented here may aid in estimation of emissions.

6.1.3 Medical Waste Incineration

6.1.3.1 Process Description

'Medical waste' includes both infectious and non-infectious wastes. The types of materials burned at medical waste incinerators include (USEPA, 1995b):

- medical hospital waste;
- veterinary waste;
- crematorium waste; and
- waste generated at research facilities.

The aforementioned medical wastes are typically incinerated for the following reasons (USEPA, 1995b):

- to render the waste innocuous;
- a reduction in the waste volume; and
- to reduce the mass of the waste.

As for municipal waste incineration the composition of medical waste can vary considerably, resulting in highly variable emissions.

The three main types of medical waste incinerators are:

1. controlled-air incinerators (also known as starved-air);
2. excess-air incinerators; and
3. rotary kiln incinerators.

Controlled-Air Incinerators

Controlled-air incineration has become the most widely used MWI technology in recent years, and it now dominates the market for new systems at hospitals and similar medical

facilities (USEPA, 1995b). This technology is also known as starved-air incineration, two-stage incineration, and modular combustion.

Combustion of waste in controlled-air incinerators occurs in two stages. In the first stage, waste is fed into the primary, or lower, combustion chamber, which is operated at sub-stoichiometric levels of air combustion, hence, the name controlled-air. Combustion air is introduced into the primary chamber beneath the incinerator hearth and below the burning bed of waste. This air is referred to as the primary or underfire air. In the primary chamber, the moisture content of the waste is reduced and the volatile components of the waste are vaporized. Because of the low air addition rates in the primary chamber and the correspondingly low flue gas velocities and turbulence levels, the amount of solids (PM) entrained in the gases leaving the primary chamber is minimized. Temperatures in the primary chamber are relatively low because of the low air-to-fuel ratio, usually ranging from 760 to 985°C.

The hot gases flow to the secondary chamber (second stage), where excess combustion air is added to incinerate the volatile compounds. Temperatures in the secondary chamber may range from 985 to 1,095°C. Optimization of controlled-air incinerators requires thorough mixing of the gases in the secondary chamber and prolonging residence time in order to maximize incineration of the wastes. The primary and secondary chambers may be equipped with auxiliary burners to handle wastes with high moisture content or to assist in burnout during start-up or shut-down.

Excess-Air Incinerators

Excess-air incinerators are typically small modular units and are referred to as batch incinerators, multiple-chamber incinerators, or retort incinerators. Excess-air incinerators typically appear to be a compact cube from the outside and have a series of chambers and baffles on the inside. Although they can be operated continuously, they are usually operated in a batch mode.

As with controlled-air incinerators, incineration of waste in excess-air incinerators occurs in two stages. Waste is fed through a door into the primary combustion chamber. The charging door is then closed and an afterburner is ignited to bring the secondary combustion chamber to a target temperature, typically 870 to 985°C. When the target temperature is reached, the primary chamber burner is ignited. The moisture in the waste is reduced and the waste is incinerated by heat from the primary chamber burner as well as by radiant heat from the chamber walls.

Volatile components in the waste are vaporized, and the hot gases flow out of the primary chamber through a flame port that connects the primary chamber to the secondary, or mixing, chamber. Secondary combustion air is added through the flame port and is mixed with the volatile components in the secondary chamber. Burners are fitted to the secondary chamber to maintain adequate temperatures for combustion of the volatile gases. The gases exiting the secondary chamber are directed to the incinerator stack or to an air pollution control device (APCD).

When the waste is consumed, the primary burner shuts off. Typically, the afterburner shuts off after a set time. After the primary chamber cools down, the ash is removed from the chamber floor and a new charge of waste can be added.

Rotary Kiln Incinerators

Rotary kiln incinerators, like the incinerator types already presented, are designed with a primary chamber where waste is heated and volatilized and a secondary chamber where

combustion is completed. The primary chamber consists of a horizontal, rotating kiln that is slightly inclined to allow the waste material to migrate from the feed end to the ash discharge end as the kiln rotates. The waste feed rate is controlled by regulating the rate of rotation and the incline angle of the kiln.

Combustion air enters the primary chamber through a port. An auxiliary burner is usually used to initiate combustion and to maintain desired combustion temperatures. The rotating motion of the kiln stirs the waste and increases the solids burnout rate; however, it also increases the amount of PM entrained in the flue gases.

Volatiles and combustion gases pass from the primary chamber to the secondary chamber, where combustion is completed. The secondary chamber is operated at below excess-air levels and at temperatures as high as 1,315°C.

6.1.3.2 Emission Data

Emissions information from the US, UK and Netherlands studies are presented in Table 4. The emissions information presented in the table is discussed briefly beneath the table.

Table 4 – International Emissions Data for Medical Waste Incineration.

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne waste burned)
US ^a - Emission Range ^b - Average	1.2-775 1360
UK ^c - Old plant - New Plant	120-480 20-200
Netherlands ^d - Medium Size ^e - Small Size ^e	3000 5000

a Source: USEPA, 1995b.

b This emission range is based upon controlled air and rotary kiln medical waste incinerators only (refer to the discussion below). The average was based on US national estimates, considering all incinerator types.

c Source: HMIP, 1995.

d Source: Bremmer et.al., 1994.

e Medium size was specified as processing between 10 and 500 tonnes/yr of waste, while small incinerators process less than 2.5 tonnes/yr of waste.

US Emission Data

As for municipal waste incineration, the USEPA assumed a ratio of TEQ to total CDD/CDF emissions of 60 to 1, and determined the average presented in Table 4 using 1991 national estimates of total CDD/CDF emissions from medical waste incinerators.

Congener specific emission factors are presented in Table A2 and A3 in Appendix A for controlled air and rotary kiln medical waste incinerators using various emission control combinations. Using the total CDD and CDF estimates presented in those tables, and applying the assumed 60 to 1 ratio, the emission range presented in Table 4 is determined. Note, however, that the emission range is only based on these incinerator types, while the average considers all incinerators operating in the US. It is noted in USEPA (1994a) that emission results from a number of controlled air incinerators yielded a lower range of emission factors than predicted using the 1991 national estimates (as was done by the

USEPA (1995b) report). The report then goes on to say that this 'average' emission factor could actually be considerably lower than the estimate shown in Table 4.

UK Emission Data

The study performed in the UK (HMIP, 1995) gives limited information as to how the emission factors were attained, however, they appear to be based on UK source test data for new and old plants, as well as international data. As for medical waste incinerators, emissions are categorised according to the age of the incinerator (i.e. old and new). No additional information was specified which indicated as to what 'new' or 'old' meant.

It is made clear that these factors are subject to considerable uncertainty, with the likelihood that older poorly operated plants emit significantly greater levels of CDD/CDF compounds.

Netherlands Emission Data

As shown in Table 4, the Netherlands study broke emissions down into two categories depending on the annual quantity of waste processed. These results were determined based on testing at four facilities. The 'medium' facility emissions were taken directly from test results, while 'small' facility emissions were assumed based on the test results from the facilities and the process conditions at those installations, giving them higher emission rates.

The Australian Situation

As for municipal waste incinerators, limited Australian information is available for review, and so comparison with international data is difficult. However, from a Greenpeace (1996) report the following source test information was extracted for two NSW medical waste incinerators:

- Lithgow incinerator - CDD/CDF emission rate (as of 1995): 6.57 ng I-TEQ/m³ of flue gas; and
- Clinical Wastes Australia (CWA) incinerator - CDD/CDF emission rate (as of 1994): 0.45 ng I-TEQ/m³ of flue gas

The UK study presents average flue gas generation rate of between 12000 Nm³/tonne and 20000 Nm³/tonne of waste burned. As for the municipal waste incinerator, it is assumed that the Greenpeace emissions data is presented in terms of 'normal' cubic metres (Nm³ - as this is how source test results are typically presented). Therefore, using this range of flue gas rates, the following range of emission factors are calculated:

- Lithgow – between 80 and 130 µg I-TEQ/tonne waste incinerated; and
- CWA – between 5.4 and 9 µg I-TEQ/tonne waste incinerated.

These emission ranges are comparable with the emission ranges as determined by the UK study, while the US and Netherlands emissions are significantly greater (refer to Table 4). Although there is clearly a lack of Australian information available, based on the estimates made above a range of 20-480 µg I-TEQ/tonne (i.e. the inclusive range of old and new UK facility estimates) will be used for this study.

6.1.4 Sewage Sludge Incineration

6.1.4.1 Process Description

Prior to incineration of the sludge, it is typically dewatered until it is about 15 to 30 percent solids, at which point it will burn without supplemental fuel. Unburned residual ash is removed from the incinerator, usually on a continuous basis, and disposed of. A portion of the non-combustible waste, as well as unburned VOCs, are carried out of the combustor through entrainment in the exhaust gas stream. These gases are then treated by the appropriate control devices.

Several types of incinerators and incineration technologies are used for sewage sludge incineration, including:

1. multiple-hearth furnaces (MHFs);
2. fluidised-bed combustors (FBCs);
3. electric incinerators;
4. co-incineration with refuse;
5. rotary kilns; and
6. high-pressure wet-air oxidation.

Multiple-Hearth Furnaces (MHFs)

The basic MHF is cylindrical in shape and is oriented vertically. The outer shell is constructed of steel and lined with refractory material and surrounds a series of horizontal refractory hearths. A hollow, rotating shaft runs through the centre of the hearths. Attached to the central shaft are the rabble arms, which extend above the hearths. Each rabble arm is equipped with a number of teeth. As the central shaft rotates, the teeth on the rabble arms rake through the sludge and break up the solid material in order to increase the surface area exposed to heat and oxygen. The teeth are arranged on the arms to rake the sludge in a spiral motion, alternating in direction from the outside in and from the inside out between hearths. Burners located in the sidewalls of the hearths provide supplemental heat when necessary.

Partially dewatered sludge is fed onto the perimeter of the top hearth by conveyors or pumps. The motion of the rabble arms rakes the sludge toward the centre shaft, where it drops through holes onto the next hearth below and is raked in the opposite direction. This process is repeated on all of the subsequent hearths. Scum, material that floats on wastewater and is generally composed of vegetable and mineral oils, grease, hair, waxes, fats, and other materials that will float, may also be fed to one or more hearths. Scum may form in many treatment units, including the preparation tanks, the skimming tanks, and the sedimentation tanks. Quantities of scum are generally small compared to other wastewater solids.

Most of the moisture in the sludge is evaporated in the drying zone, which comprises the upper hearths of an MHF. The temperature in the drying zone is typically between 425 and 760°C. Sludge combustion occurs in the middle hearths as the temperature is increased to between 815 and 925°C. The cooling zone comprises the lowermost hearth(s), where the ash is cooled by the incoming combustion air.

Fluidized-Bed Combustors (FBCs)

FBCs are cylindrically shaped and oriented vertically with an outer shell constructed of steel and lined with refractory material. Nozzles designed to deliver blasts of air (called tuyeres) are located at the base of the furnace within a refractory-lined grid. A bed of sand approximately 0.75 meters thick rests on the grid.

Two general configurations can be distinguished on the basis of how the fluidizing air is injected into the furnace. In the hot windbox design, the combustion air is first preheated by passing it through a heat exchanger, where heat is recovered from the hot flue gases. Alternatively, ambient air can be injected directly into the furnace from a cold windbox.

Partially dewatered sludge is fed onto the furnace bed. The bed is maintained at temperatures of 725 to 825°C. Air injected through the tuyeres fluidizes simultaneously the bed of hot sand and the incoming sludge. Fluidization of the bed achieves nearly ideal mixing between the sludge and the combustion air, and the turbulence facilitates the transfer of heat from the hot sand to the sludge. As the temperature of the sludge rapidly increases, evaporation of the moisture and combustion of the organic materials occur almost simultaneously. The remaining combustible gases are burned in the area above the furnace bed (the freeboard area). The freeboard area functions essentially as an afterburner.

Electric Incinerators

An electric incinerator consists of a horizontally oriented, insulated furnace. A belt conveyor extends through the length of the furnace, and infrared heating elements are located in the roof of the furnace above the conveyor. Electric incinerators consist of a number of prefabricated modules that can be linked together to provide the necessary furnace length.

Dewatered sludge is deposited on the conveyor belt at the entrance of the incinerator. A roller mechanism levels the sludge into a continuous layer approximately 1 inch thick across the width of the belt. As the sludge travels through the incinerator and beneath the heating elements, it is dried and then burned. The ash remaining on the belt is discharged into a hopper at the exit end of the incinerator.

Co-incineration with Refuse

Virtually any material that can be burned can be combined with sludge in a co-combustion process. Common materials for co-incineration are coal, MSW, wood waste, and agriculture waste. Rotary kilns and other incinerators with feed and grate systems that will handle sewage sludge are used for co-incineration. When sludge is combined with other combustible materials in a co-combustion scheme, a furnace feed may be created that has both a low water concentration and a heat value high enough to sustain combustion with little or no supplemental fuel.

There are two basic methods for combusting sewage sludge with MSW:

1. by adding dewatered or dried sludge along with MSW to a municipal waste combustor; and
2. by adding processed MSW along with sludge to a sewage sludge incinerator.

With the latter method, MSW is processed by removing non-combustibles, shredding, and screening.

Cyclonic Incinerators

The cyclonic reactor is designed for small capacity applications. It is constructed of a vertical cylindrical chamber that is lined with refractory. Preheated combustion air is introduced into the chamber tangentially at high velocities. The sludge is sprayed radially toward the hot refractory walls. Combustion is rapid: The residence time of the sludge in the chamber is on the order of ten seconds. The ash is removed with the flue gases.

Rotary Kiln Incinerators

These are also generally used for small capacity applications. The kiln is inclined slightly from the horizontal plane, with the upper end receiving both the sludge feed and the combustion air. A burner is located at the lower end of the kiln. The circumference of the kiln rotates at a speed of about 15 centimetres (cm) per second. Ash is deposited into a hopper located below the burner.

Wet Air Oxidation

The wet oxidation process is not strictly one of incineration; it instead utilizes oxidation at elevated temperature and pressure in the presence of water (flameless combustion). Thickened sludge, at about six percent solids, is first ground and mixed with a stoichiometric amount of compressed air. The slurry is then pressurized. The mixture is then circulated through a series of heat exchangers before entering a pressurized reactor. The temperature of the reactor is held between 175 and 315°C. The pressure is normally 7,000 to 12,500 kilopascals. Steam is usually used for auxiliary heat. The water and remaining ash are circulated out the reactor and are finally separated in a tank or lagoon. The liquid phase is recycled to the treatment plant. Offgases must be treated to eliminate odours: wet scrubbing, afterburning, or carbon absorption may be used.

6.1.4.2 Emission Data

Emissions information for sewage sludge incinerators from the US, UK and Netherlands studies are presented in Table 5. The emissions information presented in the table is discussed briefly beneath the table.

Table 5 – International Emissions Data for Sewage Sludge Incineration.

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne dry waste burned)
US - Emissions Range ^a - Average ^b	12-63 28
UK ^c - Old plant (pre-1985) - New Plant (post-1985)	77 9
Netherlands ^d	5
Europe ^e	5-120

a Source: USEPA, 1994a.

b Source: USEPA, 1995b.

c Source: HMIP, 1995.

d Source: Bremmer et.al., 1994.

e Source: The European Environment Agency, 1996.

US Emission Data

It is unclear as to how the average emission factor was developed for sewage sludge incinerators. Based on the two USEPA reports footnoted beneath Table 5, the factor appears to be based on source test data, rather than national emission estimates (average emission factors for both medical and municipal waste incinerators were determined using national estimates). The emission range was determined in the USEPA (1994a) report by assuming that emissions varied by a factor of five between the lower and the higher estimates, with the 'average' emission as the geometric mean (note that the average was 27 µg I-TEQ/tonne in

that report, while 28 µg I-TEQ/tonne is shown in Table 5 above). Tables A4 and A5 in Appendix A present emission factors for multiple hearth and fluidised bed sewage incinerators employing various emission control technologies, with a limited breakdown on specific 2,3,7,8-substituted isomers.

UK Emission Data

The UK study assumes that the quantity of flue gas generated from sludge incineration is approximately 11000 Nm³/tonne of dry solids burned. Based on source test data available in the UK and the assumed flue gas flow, the emission factors presented in Table 5 were determined. It was further indicated that most pre-1985 plants were of the multiple-hearth design, while most post-1985 plants were fluidised bed incinerators.

Netherlands Emission Data

Sewage sludge incineration in the Netherlands occurs principally in fluidised bed furnaces. The emissions data presented in Table 5 were determined based on source testing from two fluidised bed facilities.

European Emissions Data

This was published in the Atmospheric Emission Inventory Guidebook (The European Environment Agency, 1996), which is not actually a CDD/CDF emissions study. Although little information was given, these emission factors are based on results from two incineration plants using particulate control only. Furthermore, it was noted that test results displayed a huge amount of variability (up to an order of magnitude) between different sampling periods, indicating the huge amount of variability between and even within sources.

The Australian Situation

Information is only available (from the Greenpeace (1996) report) for a sewage sludge incinerator, which was decommissioned in 1995. The source test information is as follows:

- Sydney Water incinerator - CDD/CDF emission rate (as of 1995): 0.278 ng I-TEQ/m³ of flue gas.

As mentioned above, the UK study used an assumed flue gas flow of 11000 Nm³/tonne of dry waste burned. Again, it is assumed that the Greenpeace emissions data is presented in terms of 'normal' cubic metres. Therefore, using this assumed flue gas rates the following emission factor is calculated:

- $0.278 \text{ ng/m}^3 \times 11000 \text{ m}^3/\text{tonne} = 3.0 \text{ } \mu\text{g I-TEQ/tonne dry waste incinerated.}$

This emission rate falls at the lower end of the emission ranges presented in Table 5. As the average emissions and emission ranges for each of the international studies are quite comparable, an emission range of between 5 and 120 µg I-TEQ/tonne dry waste burned will be used in this study.

6.1.5 Hazardous Waste Incineration

6.1.5.1 Process Description

Hazardous wastes include any wastes that may be ignitable, corrosive, reactive, and/or toxic.

Hazardous wastes are incinerated to destroy the hazardous constituents in the waste, with the potential for energy recovery (i.e. if used in industrial boilers/furnaces). The various incinerators, boilers, and furnaces typically used to incinerate hazardous waste are discussed below.

Hazardous Waste Incinerators

The five hazardous waste incinerators types discussed in this section are:

1. liquid-injection;
2. fume-injection;
3. fixed-hearth;
4. fluidized-bed; and
5. rotary kiln.

Liquid-injection incinerators are usually single-chamber units and may be either vertically or horizontally oriented. Liquid wastes are transferred from drums or tank trucks into a feed tank, where recirculation systems or mixers are used to mix the tank contents. Before introduction of the waste, a gaseous auxiliary fuel (such as propane) is normally used to preheat the incinerator system to an equilibrium temperature. The waste is then pumped from the tank and sent either directly to the incinerator or to a blending tank to be combined with other wastes before incineration. The waste is atomized by gas-fluid nozzles and injected into the incinerator.

Fume-injection incinerators are very similar to liquid-injection incinerators in design and are used to destroy gaseous or fume wastes.

The fixed-hearth incinerator utilises a stationary combustion chamber into which solids and sludges are introduced and burned. Units of this type may have a single (primary) combustion chamber or may have two chambers (primary and secondary). Fixed-hearth incinerators are usually equipped with oil or gas burners for start-up and for providing auxiliary fuel as needed. Combustion in these units is enhanced by the addition of a grate system, which allows combustion air to flow above and below the waste. Solids and sludges are fed into the primary chamber, where they are burned. Liquid waste may be introduced into either the primary or secondary chamber.

Fluidized-bed combustors (FBCs) were previously described in Section 6.1.4 (i.e. Sewage Sludge Incineration). FBCs used to dispose of hazardous waste are very similar to those used to incinerate sewage sludge except for their additional capability of handling liquid wastes. FBCs are suitable for disposing of combustible solids, liquids, and gaseous wastes.

Rotary kiln incinerators have a combustion chamber that is slightly inclined from the horizontal and rotates. Rotary kilns were described earlier in the Section 6.1.3. They are designed to incinerate many types of waste, hazardous or nonhazardous. Solid, liquid, and containerized wastes are usually fed simultaneously to the kiln, but liquid wastes also may be injected into the afterburner. The rotary kiln incinerator can be used to destroy any form of hazardous waste material that is combustible, excepting those with a high moisture content or that contain significant amounts of toxic metals.

Boilers

In contrast to incinerators, whose main objective is to destroy the hazardous constituents of wastes, boilers are constructed to produce steam for electricity generation (utility boilers) or for on-site process needs (industrial boilers). Also, hazardous wastes compose the primary

feed to incinerators, whereas they are usually a supplementary fuel for boilers. The concept of disposing of hazardous wastes in boilers has centred around industrial boilers because their operation is more flexible than utility boiler operation, and they offer the potential of destroying hazardous wastes generated on site.

The primary fuels used in industrial boilers are gas, oil, coal, and wood. Industrial boilers may be distinguished by their type of fuel-firing mode. The major types of firing modes are single- or opposed-wall, tangential, cyclone, and stoker. The terms single- or opposed-wall and tangential refer to the arrangement of the burners in the combustion chamber. In cyclone-fired units, fuel and air are introduced circumferentially into a water-cooled, cylindrical combustion chamber. Stoker-fired boilers are designed to burn solid fuels on a bed. The bed is either a stationary grate through which ash falls or a moving grate that dumps the ash into a hopper.

Industrial Furnaces

Industrial furnaces are defined as designated devices that are an integral component of a manufacturing process and that use thermal treatment to recover materials or energy. Types of industrial furnaces are cement kilns, lime kilns, aggregate kilns, phosphate kilns, and coke ovens. The types of industrial furnaces are too numerous for process descriptions to be included here. Basically, they are alike in that industrial furnaces are used to liberate heat and transfer the heat directly or indirectly to a solid or fluid material for the purpose of effecting a physical or chemical change. Industrial furnaces usually have a chamber(s) in which the material is processed into a product. Their operation and function can be compared to those of a kitchen oven. Primary fuels for industrial furnaces are normally oil, gas, or coal. Waste fuels include used lube oil, hydraulic fluid, coolant oil, and metal-working oil.

6.1.5.2 Emissions Data

Emissions information from the US, UK and Netherlands studies are presented in Table 6. The emissions information presented in the table is discussed briefly beneath the table.

Table 6 – International Emissions Data for Hazardous Waste Incineration.

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne waste burned)
US - Emission Range ^a - Average ^b	0.18-119 2.66
UK ^c	5-30
Netherlands ^d - Emission range - Average	2.2-310 105

b Source: USEPA, 1994a.

a Source: USEPA, 1995b.

c Source: HMIP, 1995.

d Source: Bremmer et.al., 1994.

US Emission Data

As indicated in the footnote to Table 6 the average emission factor was taken from USEPA (1995b), while the emissions range was extracted from USEPA (1994a). The latter reference gave an average emission factor of 27.2 µg I-TEQ/tonne of waste burned, which is almost exactly a factor of ten greater than the 'average'. No reference or indication was given as to

where or how the average used in the USEPA (1995b) study was attained, and therefore it is not clear whether any errors (i.e. typos) are present in either study. The average emission factor from USEPA (1995b) appears to encompass waste burning in hazardous waste incinerators, boilers and industrial furnaces.

The emission range is based on testing performed at six incinerators. The emission data was expressed in terms of total CDD/CDF, which was subsequently converted to I-TEQ using a conversion factor of 1.75 ng I-TEQ/ng total CDD/CDF developed for the EPA.

It is noted in the USEPA (1995b) study that emissions depend on the constituents in the waste being incinerated, and hazardous waste streams can vary significantly from facility to facility. Therefore, CDD/CDF emission factors for a particular facility would only be specific to that facility (USEPA, 1995b). The emission range presented in Table 6 above is indicative of the potentially large emissions variation.

UK Emission Data

The UK CDD/CDF inventory study (HMIP, 1995) arrives at an emission factor range based on source test data available from a limited number of facilities. The range that was taken to be representative of all licensed facilities in the UK was 0.5-3 ng I-TEQ/m³. It was then further assumed that a single tonne of hazardous waste (referred to as chemical waste) generates 10000 m³ of flue gas, thereby resulting in the emission factor range presented in Table 6. This emission range covered both commercial installations as well as 'in-house' facilities.

Netherlands Emission Data

The Netherlands study determined CDD/CDF emissions based on quite an extensive knowledge of emissions from a large number of installations. More detailed information is presented in Table A6 in Appendix A, which breaks down the emission according to incinerator technology. The emission range presented in Table 6 has been determined based on the lowest and highest emission factors shown in that table. The average emission rate was determined on a weighted basis considering the emissions from each incinerator type and the annual quantity of waste processed. It should be noted, however, that the small 'thermal afterburner' facilities were not considered, as no processing rates were given from which emission factors could be derived.

The Australian Situation

Due to the lack of information regarding emissions from Australian hazardous waste facilities, no comparison with international data is possible. As the emission factors presented in the Netherlands study have been determined using quite extensive test data, the emission range as reported by that study (i.e. 2.2-310 µg I-TEQ/tonne) would be used to represent the possible range of emissions from Australian hazardous waste incinerators. This range gives conservatively high bounds, and essentially includes the ranges as determined by the other international studies.

6.1.6 Coal and Oil Combustion

It is important to note up front that CDD/CDF emissions are not recognised as being generated during the combustion of natural gas. USEPA (1995b) states that detectable levels of CDD/CDF compounds from gas-fired boilers were not identified. Therefore, this section only deals with emissions arising from coal and oil combustion.

6.1.6.1 Process Description

Coal and oil may be combusted for the following reasons:

- to generate steam for electricity production;
- to generate process steam; and
- to provide heating.

These combustion sources generally have extremely low CDD/CDF emissions potential as the fuel used contains only small amounts of chlorinated compounds that can form CDD/CDF (USEPA, 1995b). However, due to the large quantities used, total emissions may be significant.

Various furnace configurations are used to burn coal and oil, which include (USEPA, 1995b):

- tangentially-fired;
- wall-fired;
- cyclone-fired;
- stoker-fired; and
- fluidised bed.

Tangentially-fired boilers are based on the concept of a single flame zone within the furnace. The air-to-fuel mixture in a tangentially-fired boiler projects from the four corners of the furnace along a line tangential to an imaginary cylinder located along the furnace centreline. Tangentially-fired boilers commonly burn coal. However, oil or gas may also be burned.

Wall-fired boilers are characterized by multiple individual burners located on a single wall or on opposing walls of the furnace. In contrast to tangentially-fired boilers that produce a single flame, each of the burners in a wall-fired boiler has a relatively distinct flare zone. Wall-fired boilers may burn coal, oil, or natural gas.

Cyclone-fired boilers burn crushed rather than pulverized coal. Fuel and air are burned in horizontal cylinders, producing a spinning high temperature flame. Cyclone-fired boilers are almost exclusively coal-fired.

Stoker-fired boilers are mostly used at older plants. The most common stoker type is the spreader stoker. Spreader stokers are designed to feed solid fuel onto a grate within the furnace and remove ash residue. Spreader stokers are capable of burning all types of bituminous and lignite coals.

Fluidized bed combustion (FBC) is a newer boiler technology that is not as widely used as the other boiler types. In a typical FBC boiler, crushed coal in combination with inert material (sand, silica, alumina, or ash) and/or sorbent (limestone) are maintained in a highly turbulent suspension by the upward flow of primary air from the windbox located directly below the combustor floor. This fluidization provides a large amount of surface contact between the air and solid particles, which promotes uniform and efficient combustion at low furnace temperatures, between 860 and 900°C compared to 1,370 and 1,540°C for conventional coal-fired boilers. Fluidized bed combustion boilers have been developed to operate at both atmospheric and pressurized conditions.

6.1.6.2 Emissions Data

Coal Combustion

International emission factors for coal combustion are presented in Table 7. The information is then briefly discussed beneath the tables.

Table 7 – International Emissions Data for Coal Combustion.

Emissions Study	CDD/CDF Emissions (ng I-TEQ/tonne coal fired)
US ^a - Average ^b	194
UK ^c - Power plant (i.e. utility) - Industrial/commercial	60-320 40-4800
Netherlands ^d - Power plant (i.e. utility) - Industrial	350 1600

a Source: USEPA, 1995b.

b Note that the USEPA indicate that emissions from industrial combustion equipment should be the close to those from utility combustors.

b Source: HMIP, 1995.

c Source: Bremmer et.al., 1994.

US Emission Data

The average emission factor is based on a composite of emissions test data compiled from various furnace configurations and control devices (USEPA, 1995b). This emission factor appears to be based directly upon the speciated emissions data presented in Table A7, Appendix A. Using the I-TEQ conversions (refer to Section 3.3) an emission factor of 7.6 pg I-TEQ/MJ coal fired is derived. Assuming a heating value of about 25 MJ/kg for US coal arrives at an emission factor of 190 ng I-TEQ/tonne coal, which is essentially the same as the factor presented in Table 7. The US study notes that emissions from industrial installations are likely to be similar to those from utility scale combustors.

UK Emission Data

The UK study categorises emissions from coal combustion into larger (power plants) and smaller facilities (industrial/commercial), as presented in Table 7. These factors are based on test results taken at various facilities around the UK. Source test data for the various installations are presented in Table A8, Appendix A. This data was used to arrive at the emission ranges presented in Table 7.

The UK data seems to contradict the US information, which indicates that emissions from coal fired industrial boilers may be similar to those from utility boilers. The UK emissions data shows that industrial/commercial coal combustion can potentially emit much greater quantities of CDD/CDF compounds. However, they also indicate that emissions may be similar and even lower from industrial sources. Thus, the data is also indicative of the potential variation in emissions from industrial/commercial scale combustion equipment.

Netherlands Emission Data

As for the UK study, the Netherlands data is divided according to the size of the facility. The emission factor for power plants was determined based on testing of a single facility that used an electrostatic precipitator and wet scrubber control technology. The industrial

emission factor was derived from test results from a grass drying plant using cyclone control technology. The study indicated that no CDD/CDF compounds had been detected when testing various coal fired power plants.

As for the UK study, the emissions factors used for the Netherlands study indicates that industrial scale combustion sources potentially emit much greater CDD/CDF levels than utility sources. This finding is in contradiction with the US study, which indicates similar emission potential for utility and industrial scale sources.

Oil Combustion

As shown in Table 8, the US study only includes the combustion of non-polluted oil, while the UK study only considers waste oil. The Netherlands study on the other hand includes emissions from the combustion of both oil types.

Table 8 – International Emissions Data for Oil Combustion.

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne oil fired)
US ^a - Average ^b	0.345
UK ^c - Untreated waste oil - Solvent plus oil - Treated waste oil ^d	2 4.8 6
Netherlands ^e <i>Waste Oil</i> - Emission Range ^f - Average <i>Non-polluted Oil</i> - Small scale ^g - Large scale (inc. refineries)	2-6 4 0.5 <0.001

a Source: USEPA, 1995b.

b This is from the combustion of residual (i.e. heavy) oil utility size combustion equipment. Note, however, that the USEPA indicate that emissions from industrial combustion equipment should be the close to those from utility combustors.

c Source: HMIP, 1995.

d The 'treating' of spent oil only involves the removal of solids, and not the removal of chlorine containing compounds

e Source: Bremmer et.al., 1994.

f This emission range was determined from data for various waste oil types (refer to discussion below).

g 'Small scale' includes small industry installations, those used in construction and for domestic use (domestic use is not covered in this section).

US Emission Data

The average emission factor is based on a composite of emissions test data compiled from various furnace configurations and control devices burning residual oil (USEPA, 1995b). Although it can not be stated for certain, the average emission factor was most likely determined based on the speciated CDD/CDF profile for residual oil combustion presented in Table A9, Appendix A. This is not certain as the emission factor was presented as 1.1×10^{-10} lb/barrel of oil. If the density of the oil is assumed to be 0.9 kg/litre, and using a conversion factor of 160 litres/barrel, the emission factor presented in Table 8 is determined. Using the speciated profile in Appendix A, and assuming a heating value of the oil of 45 MJ/kg, an emission factor of 0.342 µg/tonne is determined, which is essentially the same as the result from the speciated CDD/CDF profile.

The USEPA study does not refer to waste oil combustion, unless it is included under 'residual' oil. Note, however, that the term 'residual oil' is not commonly used to include waste oil, and so for this study it is assumed that the emission factor presented in Table 8 relates to non-polluted oil.

UK Emission Data

The UK study bases its emission factors on data taken from the Netherlands study. It is unclear however as to why non-polluted oil was not included, as it is also covered by the Netherlands report. It is indicated by the UK report that combustion of non-polluted oil generates extremely low levels of CDD/CDF compounds.

Netherlands Emission Data

The polluted oil emission factor range is based on the combustion of 'unprocessed' waste oil, which gives the lower emission rate, while the combustion of 'processed' waste oil results in emissions at the upper end. As mentioned above, the emission data used for the UK study were extracted from the Netherlands study. The emission values given in Table 8 for the UK study are in fact a breakdown of the 'waste oil' emission factor range presented for the Netherlands.

The emission factor for non-polluted oil combustion at small installations was not determined directly from the testing of any facilities. Through comparing CDD/CDF levels in soot from small coal combustion facilities with the levels observed in soot produced at oil fired installations, and emissions from diesel mobile sources, the 'small scale' emission factor presented in Table 8 was determined. Emissions of CDD/CDF compounds from large utility combustion sources were not detected (i.e. $<0.001 \text{ ng/m}^3$) during the testing of two residual oil fired power plants. It was then assumed that this emission level also applies to refineries. Based on these conclusions, the emission factor shown in Table 8 for 'large scale' combustion sources was produced.

Of the three principal studies reviewed, the Netherlands study was the only one to break down emission from oil-fired sources according to the size of the facility (i.e. for non-polluted oil firing). As for coal fired sources, the emission factors presented in the Netherlands study contradict the indication by the US study that CDD/CDF emissions from utility and industrial scale facilities are similar.

The Australian Situation

Very limited information is available regarding CDD/CDF emissions from coal and oil fired sources. From a Greenpeace report (1996) information for a power plant firing both coal and waste oil was attained, as follows:

- emissions detected during 94/95 testing: $0.016\text{-}0.049 \text{ ng I-TEQ/m}^3$.

It was indicated in the Greenpeace report that the waste oil is only used for start-up, and so not likely to be indicative of emissions during normal operation. However, using this data and assuming a flue gas generation rate of $10000 \text{ Nm}^3/\text{tonne}$ of coal fired (this rate is identified in both the UK and Netherlands studies) the following emission factor range is attained:

- emission range: $160\text{-}490 \text{ ng I-TEQ/tonne coal fired}$.

This emission range is in good agreement with the emission levels determined by each of the international studies (refer to Table 7). However, it may be likely that during normal

operation (i.e. without waste oil) emissions are slightly lower. Considering this factor, and the comparability of the international data, an emission range of 60-350 ng I-TEQ/tonne will be used for utility scale coal combustion in this study. Due to the lack of information for industrial scale sources, an emission range of 40-4800 ng I-TEQ/tonne will be used, based on the UK and Netherlands studies.

No emissions data was available pertaining to oil combustion sources in Australia. Therefore, the data used in this study will be based solely on international emission factors. For the combustion of polluted waste oil, the range of 2-6 µg I-TEQ/tonne will be used in this study. The US study gives an average emission value of 0.345 µg/tonne for both industrial and utility scale sources, while the Netherlands study indicates 0.5 µg/tonne for small sources, with undetectable levels for larger sources. For conservative reasons an emission factor of 0.5 µg I-TEQ/tonne is assumed to be representative of all combustion sources fired with non-polluted oil.

6.1.7 Industrial Wood Combustion

The combustion of wood is typically only utilised in industrial/commercial size combustors, rather than for utility scale applications. Furthermore, it is typically confined to those industries where it is available as a by-product, where it is burned both to obtain energy and alleviate waste disposal problems. Wood waste may include large pieces, such as slabs, logs, and bark strips, as well as cuttings, shavings, pellets, and sawdust. Wood wastes generally contain higher levels of chlorine and CDD/CDF precursors than fossil fuels. Also, wood waste has a higher moisture content, which may decrease combustion efficiency. Therefore, the CDD/CDF emission potential of wood combustion is greater than that of fossil fuel combustion (USEPA, 1995b).

Wood that has been treated with PCP may also be combusted in wood-fired units, and studies have indicated that a strong correlation exists between CDD/CDF emissions and the combustion of PCP contaminated wood. Furthermore, many industrial installations burning wood wastes will burn residues from furniture manufacturing, which may include melamine facing and contain additives such as resins used in the production of chipboard. The influence of these chemicals on CDD/CDF emissions is unknown (HMIP, 1995).

Emissions of CDD/CDF from wood-fired boilers are dependent on several principal variables (USEPA, 1995b):

- wood waste composition and variability;
- fossil fuel type and quantity, if any, co-fired with the wood waste;
- combustor type and performance; and
- air pollution control systems.

The composition of wood waste can impact CDD/CDF emissions and this depends largely on the industry from which it originates. Pulping operations, for example, produce great quantities of bark that may contain more than 70 percent by weight moisture, along with sand and other non-combustibles. Because of this, bark boilers in pulp mills may emit considerable amounts of organic compounds to the atmosphere unless they are well controlled. On the other hand, some operations, such as furniture manufacturing, produce a clean, dry wood waste, 5 to 50 percent by weight moisture, with relatively low particulate emissions when properly burned. Still other operations, such as sawmills, burn a varying mixture of bark and wood waste that results in particulate emissions somewhere between those of pulp mills and furniture manufacturing. Additionally, when fossil fuels are co-fired with wood waste, the combustion efficiency is typically improved; therefore, organic emissions may decrease.

6.1.7.1 Process Description

Various boiler-firing configurations are used in burning wood waste. One common type in smaller operations is the dutch oven or extension type of furnace with a flat grate. This unit is widely used because it can burn fuels with very high moisture. Fuel is fed into the oven through apertures in a firebox and is fired in a cone-shaped pile on a flat grate. The burning is done in two stages: (1) drying and gasification, and (2) combustion of gaseous products. The first stage takes place in a cell separated from the boiler section by a bridge wall. The combustion stage takes place in the main boiler section.

In another type of boiler, the fuel-cell oven, fuel is dropped onto suspended fixed grates and is fired in a pile. The fuel cell further uses combustion air preheating and repositioning of the secondary and tertiary air injection ports to improve boiler efficiency.

In many large operations, more conventional boilers have been modified to burn wood waste. These units may include spreader stokers with travelling grates or vibrating grate stokers, as well as tangentially fired or cyclone-fired boilers. The most widely used of these configurations is the spreader stoker, which can burn dry or wet wood. Fuel is dropped in front of an air jet, which casts the fuel over a moving grate. The burning is done in three stages: (1) drying; (2) distillation and burning of volatile matter; and (3) burning of residual carbon.

A recent development in wood-firing is FBC boilers. Refer to Section 6.1.6 for a description of this boiler-type. Because of the large thermal mass represented by the hot inert bed particles, FBCs can handle fuels with high moisture content (up to 70 percent, total basis). Fluidized beds can also handle dirty fuels (up to 30 percent inert material). Wood is pyrolyzed faster in a fluidized bed than on a grate due to its immediate contact with hot bed material. As a result, combustion is rapid and results in nearly complete combustion of organic matter, thereby minimizing emission of unburned organic compounds.

6.1.7.2 Emissions Data

Emissions factors determined by each of the international studies are presented in Table 9 below. These data are then further discussed below the table.

Table 9 – International Emissions Data for Industrial Wood Combustion.

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne wood burned)
US ^a - Average ^b	1.95
UK ^c - Clean wood - Contaminated wood	1-2 9-19
Netherlands ^d - Clean wood - Contaminated wood with PCP without PCP	1-2.2 25-50 5-10

a Source: USEPA, 1995b.

b This average is half the average stated in the USEPA report (i.e. 3.9 µg/tonne), by assuming 50% of the wood content is moisture. This allowed a conversion to µg/'total' wood burned, rather than 'dry' wood burned (refer to discussion below)

c Source: HMIP, 1995.

d Source: Bremmer et.al., 1994. The ranges relate to whether the facility uses pollution control. The lower value is for controlled, while the upper bound is for uncontrolled facilities.

US Emission Data

No information is given in the USEPA (1995b) report indicating how the average emission factor presented in Table 9 was determined. However, the USEPA (1994a) report quotes the same emission factor, which was derived from testing at two facilities. One facility employed fabric filter control while the other used multiple cyclones. It was discussed that because the data from the two facilities varied greatly, the 'average' emission factor was subject to significant uncertainty.

The USEPA (1995b) study indicates that assuming an approximate wood moisture content of 50 percent is reasonable. The emission factor in Table 9 was determined for this study by applying this assumption (i.e. $3.9 \times 0.5 = 1.95 \mu\text{g/tonne}$). This allowed easy comparison with the emissions data from the other studies (i.e. so they are all in terms of 'total' wood burned).

Although no indication as to whether clean or contaminated wood waste was burned, data from the other studies indicates that primarily clean wood was used. However, insufficient information is available to make any definitive conclusions.

Additionally, the US study briefly investigates emissions from wood fired sources burning black liquor (or sludge), which is typically generated in the paper industry. However, the test results indicated that the burning of this sludge has little impact on CDD/CDF emissions from wood combustion (USEPA, 1995b).

UK Emission Data

The UK study breaks down emissions into categories based on whether or not the waste wood is contaminated. The clean wood burning emission factors presented in Table 9 appear to be based on international data, while emissions from the burning of contaminated wood are based on UK specific data. The UK data was attained from a plant using particulate control. A table presenting the data reviewed by the UK study is shown in Table A10, Appendix A.

Netherlands Emission Data

The emissions data presented in Table 9 are based on limited source testing. Source test data was available for the burning of contaminated wood without PCP, and for various wood-fired residential installations. Based on this information and a series of detailed assumptions, the emission factors for industrial wood-fired combustion sources were determined. The reader is referred to the Bremmer et.al. (1994) document for additional information regarding the assumptions used. The ranges relate to whether the installation is fitted with flue gas cleaning, with the upper bound for uncontrolled facilities and the lower for controlled. The controlled emission levels are based primarily upon control using cyclones.

It appears that the UK study based the 'clean wood' emissions data on those predicted by the Netherlands study.

The Australian Situation

No source test data is available for industrial wood-fired combustion sources, and so a comparison with international data is not possible. However, as the international data compare quite well, the following emission factors will be used for this study:

- clean wood: 1-2 $\mu\text{g I-TEQ/tonne wood burned}$; and
- contaminated wood: 5-50 $\mu\text{g I-TEQ/tonne wood burned}$.

These ranges encompass the emissions data from all three international studies.

6.1.8 Residential Combustion

The residential sector includes the following equipment types (USEPA, 1995b):

- furnaces;
- stoves; and
- fireplaces.

These combustion devices may burn coal, oil, gas, and wood to produce heat for individual homes. However, gas-fired furnaces are considered unlikely sources of CDD/CDF (USEPA, 1995b), are thus not discussed any further. Based on 1994 statistics, coal only accounts for approximately 0.2 percent of residential fuel use (NGGIW, 1996a), and so will not be considered any further. Therefore, this section will discuss emission from wood and distillate oil combustion.

The combustion of fossil fuels or wood in residential units (woodstoves, furnaces, fireplaces) is a relatively slow and low-temperature process. Furthermore, the potential to form CDD/CDF compounds during combustion is greater than for the larger units discussed above, due to the following reasons (USEPA, 1995b):

- combustion in the residential sector tends to be less efficient than in other sectors;
- inadequate maintenance of the combustion units may increase potential for CDD/CDF formation on particulate matter; and
- residential combustion units are generally not equipped with gaseous or particulate control devices.

Emission factors are presented for wood-fired furnaces, stoves, and fireplaces.

6.1.8.1 Process Description

In the residential sector, coal is usually combusted in underfeed or hand-stoked furnaces. Stoker fed units are the most common design for warm-air furnaces and for boilers used for steam or hot water production. These units are typically controlled with an automatic thermostat and designed for a specific type of coal. Other coal-fired heating units include hand-fed room heaters, metal stoves, and metal and masonry fireplaces. These units operate at low temperatures and do not efficiently combust fuel. Generally, coal contains small quantities of chlorine and CDD/CDF precursors and so the potential for CDD/CDF formation exists.

Oil-fired residential furnaces are designed with varying burner configurations, each attempting to optimize fuel combustion efficiency. Emissions from fuel oil combustion depend on the grade and composition of the oil (with lighter grade oils exhibiting lower emissions), the design of the furnace, and the level of equipment maintenance.

Woodstoves are used commonly in residences as space heaters to supplement conventional heating systems. They transfer heat by radiation from the hot stove walls to the room. Circulating stoves convert radiant energy to warm convection air. Combustion efficiencies for woodstoves are dependent on stove design and operating characteristics. Consequently, combustion efficiency and emissions can vary significantly. Woodstoves have a greater potential to emit CDD/CDF than fossil fuel-fired units due to the likelihood of chlorine and/or CDD/CDF precursors present in wood.

6.1.8.2 Emissions Data

Emissions data from each of the international studies pertaining to the combustion of wood and distillate oil is presented in Table 10. The emission information is then discussed beneath the table.

Table 10 – International Emissions Data for Residential Fuel Combustion.

Fuel	Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne fuel burned)
Wood	US ^a	
	- Average	0.5 ^b
	UK ^c	
	<i>Clean Wood</i>	
	- Wood stove	1-3
	- Fireplace	1-29
	<i>Treated Wood</i>	
	- Wood stove	10-50
	- Fireplace	10-50 ^d
	Netherlands ^e	
Oil	US ^a	
	- Average	0.19
	UK ^b	ND ^f
	Netherlands ^e	
	- Average ^g	0.5

^a Source: USEPA, 1995b.

^b Source: HMIP, 1995.

^c As for industrial wood combustion, this average is half the value stated in the USEPA report, by assuming 50% of the wood content is moisture. This allowed a conversion to µg/'total' wood burned, rather than 'dry' wood burned (refer to discussion below).

^d It is believed that there is an error in this range, based on a review of where the emissions data appears to have been taken from (further discussed below).

^e Source: Bremmer et.al., 1994.

^f The UK study does not consider emissions from residential oil combustion.

^g Note that this is the same emission factor presented in Table 8 (refer to Section 6.1.6.2) that was determined for 'small scale' oil-fired combustion sources.

US Emission Data

Wood Combustion

The US emission factor for residential wood combustion was determined based on studies performed in Denmark and Switzerland, which measured directly CDD/CDF emissions from woodstoves. Although no additional information is given regarding these studies in USEPA (1995b), they are discussed further in USEPA (1994). The two studies considered a number

of different stove designs as well as various wood fuels (including natural wood, waste wood, and charcoal and household waste). An average emission factor of 1 µg/tonne dry wood burned was determined from the studies (note that all levels observed were close to 1 µg/tonne), which was converted to 0.5 µg/tonne wood burned for this study, by assuming approximately 50 percent moisture of the wood (refer to the discussion on industrial wood combustion - Section 6.1.7).

Oil Combustion

The emission factor for distillate oil combustion was determined from CDD/CDF compositions in soot samples collected in 21 residential furnaces used in central heating, combined with particulate emission factors. This estimation methodology is sound if it is assumed that most of the CDD/CDF compounds exit the chimney with the particulate, and the composition of the soot is in fact representative of the particulates emitted. It is noted by USEPA (1995b) that emissions estimates in this way may represent upper limits, as soot may not be representative of particulates emitted.

The original emission factor was presented as 5.26×10^{-8} lb/1000 barrels burned, which was subsequently converted to the emission factor presented in Table 10 by using the following information:

- 1 barrel = 160 litres;
- 2.2 lb = 1 kg; and
- assuming the density of the distillate oil is approximately 0.8 kg/L.

UK Emission Data

Wood Combustion

As for industrial wood combustion, the study breaks down emissions into clean wood burning and treated wood burning. The emissions data presented in Table 10 are based on the emissions information presented below (presented as an appendix to the UK report):

- wood burning stove, treated wood, no PCP: 10 µg I-TEQ/tonne wood;
- wood burning stove, treated wood, presence of PCP: 50 µg I-TEQ/tonne wood;
- wood burning stove, clean wood: 0.7-3.3 µg I-TEQ/tonne wood;
- fireplace, treated wood, no PCP: 100 µg I-TEQ/tonne wood (estimated);
- fireplace, treated wood, presence of PCP: 500 µg I-TEQ/tonne wood (estimated); and
- fireplace, clean wood: 0.7-28.5 µg I-TEQ/tonne wood.

Note that the ranges presented above are summaries of ranges given by various references (eg. if three ranges were given these were combined into a single range), with the Netherlands study appearing to be a principal source of the emissions information. Where '(estimated)' appears it was indicated that these emissions were estimated by the author of the reference. This emissions data indicates that contaminated wood combustion releases significantly greater quantities of CDD/CDF compounds. A more detailed table showing the information reviewed by the UK study can be found in Table A11, Appendix A.

It appears to be quite straightforward as to how each of the emission ranges shown in Table 10 were determined, with the exception of treated wood burning in fireplaces. An emissions factor range of 100-500 µg I-TEQ/tonne seems to be more appropriate based on the information presented above. However, an assumption is made in the UK report that PCP is

present in only a small fraction of treated wood that is burned. Even if this were the case a lower bound of 100 µg/tonne is appropriate based on the information above.

Oil Combustion

The UK study does not consider domestic oil combustion. It is noted in the UK report that burning of unpolluted oil generates extremely low levels of CDD/CDF compounds, and this may be the reason it is not considered.

Netherlands Emission Data

Wood Combustion

The Netherlands study breaks down emissions in a similar manner to the UK study (i.e. clean and treated wood). It is noted that the wood fireplace emissions are subject to considerable uncertainty due to the methods used to estimate the emissions (except for clean wood). The emission factors for clean wood burning in both wood-stoves and fireplaces are based on source testing. A wood-stove was tested under three loading conditions (minimum, average and maximum), while a fireplace was tested under two operating conditions (burning on a grate and on the floor). Averages were taken and the emission factors for clean wood presented in Table 10 were derived. The emission factors for polluted wood combustion were then determined using a series of assumptions, and comparison with other literature data. A further discussion on the methodology used to estimate the emissions can be found in the Netherlands report (Bremmer et.al., 1994).

Oil Combustion

The oil combustion emission factor is the same as the emission factor presented in Table 8 for 'small scale' sources. It was indicated in the Netherlands study that small scale installations includes those used for domestic purposes. The discussion in Section 6.1.6.2 indicates how the emission factor was determined.

The Australian Situation

There is no information available regarding emissions from domestic wood burning equipment in Australia. Therefore, estimates will be made based on the international emission data presented in Table 10.

Domestic Wood Combustion

The US emission factor is comparable to the lower end of the emission ranges determined by the UK and Netherlands studies. However, for conservative reasons and based on the uncertainties involved, this study will adopt the emission ranges used by the UK report for stoves burning clean and treated wood and fireplaces burning clean wood, as they incorporate the Netherlands data as well as other source test information. However, due to the uncertainties regarding how the UK study determined the emission range for treated wood burning in a fireplace, this study will use the Netherlands emissions information for such burning operations.

Therefore, the following emission factors will be used in this study:

- clean wood – stove: 1-3 µg I-TEQ/tonne wood;
- clean wood – fireplace: 1-29 µg I-TEQ/tonne wood;
- treated wood – stove: 10-50 µg I-TEQ/tonne wood; and
- treated wood – fireplace: 100-500 µg I-TEQ/tonne wood.

Domestic Oil Combustion

The emissions data from the US and Netherlands are in good agreement. Therefore, an emission range of between 0.2-0.5 µg I-TEQ/tonne will be used for this study.

6.1.9 Crematoria

6.1.9.1 Process Description

There are two types of crematory furnaces (Bremmer et.al., 1994):

- ‘cold types’, in which the coffin is placed inside the furnace at a temperature of approximately 300°C; and
- ‘warm types’ where the coffin is placed inside at a temperature of 800°C or higher.

When the crematory reaches the operating temperature the body container is placed on the combustion chamber grate and the ignition burner is fired to attain a target combustion temperature sufficient for the proper reduction of human remains. The afterburner preheats the chamber to the desired temperature, depending on whether the furnace is a ‘cold’ or ‘warm’ type. When the body container is introduced into the combustion chamber, the burner is ignited, and cremation begins at about 870 to 980°C (USEPA, 1995b). The gases produced during cremation are passed through the afterburner. The cremation process lasts about 2-2.5 hours in the ‘cold’ type furnaces, while between 1.2-1.5 hours is required for ‘warm’ type furnaces. Cooldown follows for 45 minutes to 1.5 hours.

6.1.9.2 Emissions Data

Emissions data for each of the international studies are presented in Table 11. This information is then discussed below the table.

Table 11 – International Emissions Data for Crematoria

Emissions Study	CDD/CDF Emissions (ng I-TEQ/body burned)
US ^a - Emission range - Average	0.097-0.776 0.374
UK ^b - Emission Range	2400-80000
Netherlands ^c - Emission Range - Average	2400-4900 (28000) ^d 4000

a Source: USEPA, 1995b.

b These emissions data were determined from a speciated profile of the 2,3,7,8-substituted isomers (further discussed below)

c Source: HMIP, 1995.

d Source: Bremmer et.al., 1994.

e The principal range from which the representative average was attained is 2400-4900, however, the highest value that was presented in the study was 28000 ng/body (quoted from a (former) West Berlin crematory).

US Emission Data

The US emissions factors were determined based on speciated CDD/CDF profiles developed during the testing of two propane fired crematories over a two week period. It is unclear as to whether these crematories were designed to state of the art standards, however, their

emissions levels are orders of magnitudes below those determined by both of the other studies. The US emissions data are questionable due to the enormous difference between them and the UK and Netherlands data, and the similarity in emission data between the latter two studies. It is noted that the US crematories tested used afterburners, however, all crematories considered in the Netherlands study also used afterburners.

The speciated 2,3,7,8-substituted isomers are presented in Table A12, Appendix A. The emission factors were determined using the I-TEQ conversion factors presented in Table 1.

UK Emission Data

The emission factors presented in Table 11 are based on UK source test data, as well as data presented in the Netherlands study. It was noted that emissions might be high from the UK cremation facilities tested, as the plants were not state of the art. This certainly appears to hold true when the emissions are compared to the US data.

Netherlands Emission Data

The Netherlands study was the only study that defined emissions according to the two furnace types. The following emission rates of dioxins were determined from two sources tested:

- cold type: 2.4 µg I-TEQ/body; and
- warm type: 4.9 µg I-TEQ/body.

These emissions from these two furnace types produce the range presented in Table 11. Note however that the higher emissions from the warm type crematory furnace were partly attributed to the fact that it utilises flue gas cooling, which results in the formation of a proportion of the CDD/CDF compounds. The maximum emission rate presented in the study was 28 µg I-TEQ/body for a crematory in (former) West Berlin, however, this does not seem to be included in determining the final average emission rate presented above.

The Australian Situation

Due to the lack of source test data for crematoria in Australia, reliance on international data is required to estimate emissions. As the US data is considered dubious, the UK and Netherlands studies will form the basis of emissions estimates, as they are very comparable. Even if the US results are in fact correct, it is unlikely that Australian crematoria are of such a high level of design and operation. The UK emission range of 2.4-80 µg I-TEQ/tonne will be used to represent Australian sources, as it encompasses the Netherlands information.

6.1.10 Iron and Steel Production

The iron and steel industry can essentially be divided up into the following categories:

- primary iron and steel production; and
- iron and steel foundries.

The two principal sources of emissions in the primary iron and steel industry have been identified as:

- sinter production; and
- coke production.

Each of these process operations is considered separately in Sections 6.1.11 and 6.1.12. Of the remaining operations in the iron and steel industry, the use of scrap materials is of principal concern. Therefore, iron and steel foundries and the use of electric arc furnaces (EAF) in the primary industry (as these use 100% scrap feed) are the focus of this section. The use of EAFs is discussed in the following section in relation to foundries, however the same applies for their use in primary operations.

6.1.10.1 Process Description

The major operations utilised by foundries include (USEPA, 1995b):

1. raw material handling and preparation;
2. metal melting;
3. mold and core production; and
4. casting and finishing.

Iron and steel castings are produced in a foundry by injecting or pouring molten metal into cavities of a mold made of sand, metal, or ceramic material. The raw materials may include (USEPA, 1995c):

- pig iron;
- iron and steel scrap;
- foundry returns; and
- metal turnings

The metal melting process is accomplished in cupola (or blast) furnaces and electric furnaces (i.e. arc or induction furnaces). Iron castings are typically produced using cupolas, while steel foundries rely primarily on EAFs or induction furnaces for melting purposes.

In either type of foundry, when the poured metal has solidified, the molds are separated and the castings removed from the mold flasks on a casting shakeout unit. Abrasive (shotblasting) cleaning, grinding, and heat treating are performed as necessary. The castings are then inspected and shipped to another industry for machining and/or assembly into a final product.

A brief discussion on each of the aforementioned furnace technologies is given below.

Cupola (Blast) Furnaces

This furnace is typically a vertical, cylindrical steel shell with either a refractory lined or water cooled inner wall. Refractory linings usually consist of silica brick, or dolomite or magnesium brick. Cupolas are charged with alternate layers of coke, metallics (pig iron and scrap etc.), and fluxes. Combustion air is introduced into the cupola through tuyeres located at the base.

The heat produced by the burning coke melts the iron, which flows down and is tapped from the bottom of the cupola. Fluxes combine with non-metallic impurities in the charge and form slag, which is removed through tap holes at the bottom of the cupola.

Electric Arc Furnaces (EAF)

Electric arc furnaces (EAF) are used to produce carbon and alloy steels. The input material to an EAF is typically 100 percent scrap. Cylindrical, refractory lined EAFs are equipped with carbon electrodes to be raised or lowered through the furnace roof. With electrodes retracted, the furnace roof can be rotated aside to permit the charge of scrap steel by

overhead crane. Alloying agents and fluxing materials usually are added through the doors on the side of the furnace. Electric current of the opposite polarity generates heat between the electrodes and through the scrap. After melting and refining periods, the slag and steel are poured from the furnace by tilting.

Scrap steel is charged to begin a cycle, and alloying agents and slag materials are added for refining. Stages of each cycle normally are charging and melting operations, refining (which usually includes oxygen blowing), and tapping.

Electric Induction Furnaces

Electric induction furnaces are cylindrical or cup-shaped refractory-lined vessels that are surrounded by electrical coils. When these coils are energized with high frequency alternating current, they produce a fluctuating electromagnetic field that heats the metal charge. The induction furnace is simply a melting furnace to which high-grade scrap is added to make the desired product. Induction furnaces are kept closed except when charging, skimming and tapping. The molten metal is tapped by tilting and pouring through a hole in the side of the vessels.

6.1.10.2 Emissions Data

The CDD/CDF emission factors as determined by the US, UK and Netherlands studies are presented in Table 12 below. Each of the emission factors are discussed beneath the table.

Table 12 – International Emissions Data for Iron and Steel Foundries

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne product)
US ^a	1.84
UK ^b	0.7-10 ^e
Netherlands ^{c,d}	4.3

a Source: USEPA, 1995b.

b Source: HMIP, 1995.

c Source: Bremmer et.al., 1994.

d The units of this emission rate are actually µg I-TEQ/tonne scrap processed, rather than per tonne of product.

e This emission range represented the utilisation of 'no chlorine' and 'high chlorine' operational conditions (i.e. the type of scrap used).

US Emission Data

The US study determined the emission factor based on source testing results from a cupola furnace charged with pig iron, scrap iron and steel, coke and limestone. The furnace employed afterburner and baghouse control technologies.

UK Emission Data

The UK study reviews emissions data from quite a number of principally European literature sources. Most of the emissions data were in units of µg/m³ of flue gas, with the exception of the following emission data for an iron and steel plant using baghouse emission control:

- scrap metal with cutting oils: 1.0 µg/tonne;
- scrap metal with PVC: 20 µg/tonne;
- scrap metal with 'no chlorine': 0.7 µg/tonne; and
- scrap metal with calcium chloride: 0.2 µg/tonne.

It was indicated that emissions were 50 percent higher prior to passing through the baghouse. Based on the above emission factors and the other data reviewed, the emission range presented in Table 12 was assumed to be representative of 'no chlorine' and 'high chlorine' operational conditions.

Netherlands Emission Data

The Netherlands study determined an emission rate based on monitoring performed at a large iron and steel foundry processing scrap materials. This larger company processed approximately 65 percent of the total scrap handled by iron and steel foundries and utilised fabric filter control. The emissions factor shown in Table 12 was then assumed to apply to the remaining facilities. Note, however, that facilities processing 'clean' scrap were assumed to emit no CDD/CDF compounds and so only a fraction (approx. half) of the remaining facilities were assumed to be dioxin sources.

It is important to note that for the large facility tested, stack emissions did not appear to be the principal source. Rather, emissions via the roof caps accounted for over 95 percent of total atmospheric emissions. Emissions via the roof caps were due to the dust that is not collected in the dust arrestor.

The Australian Situation

No source testing data is available for iron and steel foundries in Australia. Therefore, international data will be used to generate CDD/CDF estimates. All of the emission estimates made by the international studies compare well, and so the range specified by the UK report (i.e. 0.7-10 µg/tonne) will be used in this study as it encompasses the data from the other studies.

6.1.11 Sinter Production

As mentioned in the section above sinter production is considered one of the principal sources of CDD/CDF compounds originating from the primary iron and steel industry.

6.1.11.1 Process Description

The sintering process converts fine-sized raw materials, including iron ore, coke breeze, limestone, mill scale, and flue dust, into an agglomerated product, sinter, of suitable size for charging into the blast furnace. The raw materials are sometimes mixed with water to provide a cohesive matrix, and then placed on a continuous, travelling grate called the sinter strand. A burner hood, at the beginning of the sinter strand ignites the coke in the mixture, after which the combustion is self supporting and it provides sufficient heat, 1300 to 1480°C, to cause surface melting and agglomeration of the mix. On the underside of the sinter strand is a series of windboxes that draw combusted air down through the material bed into a common duct, leading to a gas cleaning device. The fused sinter is discharged at the end of the sinter strand, where it is crushed and screened. Undersize sinter is recycled to the mixing mill and back to the strand. The remaining sinter product is cooled in open air or in a circular cooler with water sprays or mechanical fans. The cooled sinter is crushed and screened for a final time, then the fines are recycled, and the product is sent to be charged to the blast furnaces. Generally, 2.3 tonnes of raw materials, including water and fuel, are required to produce 0.9 tonnes of product sinter (USEPA, 1995c).

6.1.11.2 Emissions Data

International emissions data for the production of sinter for iron and steel operations are presented in Table 13. Note that sintering emissions were not considered in the USEPA (1995b) study. They were mentioned in the earlier USEPA (1994) study, however at the time of that study insufficient information was available to develop any US specific emission factors. Additionally, insufficient information was available in the Netherlands study to produce emission factors. However, the data presented in the Netherlands report is reviewed in the UK study.

Table 13 – International Emissions Data for Sintering Operations

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne sinter produced)
UK ^a	1.2-9.0

^a Source: HMIP, 1995.

The emission range presented in the table above was developed based on extensive source test data for the sinter plants operating in the UK. Although the emission factor range was determined, it was not required as emissions were actually calculated using the direct data taken at each facility. Upon comparing source test results with the Netherlands data, the UK study showed that results were in relatively good agreement.

The Australian Situation

Limited source test data is available for sinter plants in Australia. From a Greenpeace (1996) report, the following source test results for two BHP sinter plants was extracted:

- BHP Newcastle: 3.48 ng I-TEQ/m³; and
- BHP Port Kembla: 2.72 ng I-TEQ/m³.

A set of stack gas emission rates was given for each of the sinter plants considered by the UK study. An average of these gives approximately 2100 Nm³/tonne sinter. If it is assumed that the BHP test results are expressed as ng/Nm³, then the following emission factor range is determined:

- emission range: 5.7-7.3 µg/tonne sinter.

These emission estimates are in very good agreement with those given in the UK report. However, due to the uncertainties in these estimates, the range given in the UK study will be used for this study.

6.1.12 Coke Production

In addition to the production of sinter (discussed above), coke production is identified as the second major source of CDD/CDF emissions from primary iron and steel production. Note however that coke production is not limited to the iron and steel industry.

6.1.12.1 Process Description

Destructive distillation ("coking") of coal occurs in coke ovens without contact with air. These ovens must remain airtight under the cyclic stress of expansion and contraction. Each oven has 3 main parts:

1. coking chambers;
2. heating chambers; and
3. regenerative chambers.

All of the chambers are lined with refractory (silica) brick. The coking chamber has ports in the top for charging of the coal.

A coke oven battery is a series of 10 to 100 coke ovens operated together. Offtake flues on either end remove gases produced. Process heat comes from the combustion of gases between the coke chambers. Individual coke ovens operate intermittently, with run times of each oven coordinated to ensure a consistent flow of collectible gas. Approximately 40 percent of cleaned oven gas (after the removal of its by-products) is used to heat the coke ovens. The rest is either used in other production processes related to steel production or sold.

Coal may be added to the ovens in either a dry or wet state. Prepared wet coal is finely crushed before charging to the oven. Flash-dried coal may be transported directly to the ovens by the hot gases used for moisture removal. Wall temperatures should stay above 1100°C during loading operations and actual coking.

The blended coal mass is heated for 12 to 20 hours for metallurgical coke. Thermal energy from the walls of the coke chamber heats the coal mass by conduction from the sides to the middle of the coke chamber. During the coking process, the charge is in direct contact with the heated wall surfaces and develops into an aggregate "plastic zone". As additional thermal energy is absorbed, the plastic zone thickens and merges toward the middle of the charge. Volatile gases escape in front of the developing zone due to heat progression from the side walls. The maximum temperature attained at the centre of the coke mass is usually 1100 to 1150°C. This distils all volatile matter from the coal mass and forms a high-quality metallurgical coke.

After coking is completed (no volatiles remain), the coke in the chamber is ready to be removed. Doors on both sides of the chamber are opened and a ram is inserted into the chamber. The coke is pushed out of the oven in less than 1 minute, through the coke guide and into a quench car. After the coke is pushed from the oven, the doors are cleaned and repositioned. The oven is then ready to receive another charge of coal.

6.1.12.2 Emissions Data

Emissions data for each of the international studies are presented in Table 14. This information is then discussed below the table. Note that the UK study relied solely upon emission data taken from the Netherlands study. Therefore, only the Netherlands information is included in the table below. The US study did not consider emissions from coke production.

Table 14 – International Emissions Data for Coke Production

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne coal)
Netherlands ^a	0.23

^a Source: Bremmer et.al., 1994. Note that this was the same factor used by the UK study.

Netherlands Emission Data

The above emission factor for the Netherlands was based on testing at one large coke production facility. The sources that were investigated were:

- flue gases from coke quenching;
- emissions from charging; and
- emissions from ejection.

It was found that emissions from the latter two sources were negligible compared with emissions from quenching. The UK study adopted the same emission factor as given in Table 14, with the exception that the factor presented in the UK report was based on quantity of coke produced. If this is done then an emission factor of 0.3 µg I-TEQ/tonne coke produced is determined.

The Australian Situation

Due to the lack of source test data for Coke production emissions in Australia, the emission factor presented in Table 14 will be used in this study. However, as the activity data is expressed in terms of coke produced, the factor as presented in the UK report (i.e. 0.3 µg/tonne) will be more appropriate.

6.1.13 Non-Ferrous Metal Production

The principal focus of emissions from the non-ferrous metal industry relates to secondary operations (eg. secondary copper smelting etc.). The Netherlands study does not consider primary operations because of the very low chlorine levels in the raw materials. The US study does not indicate why primary operations are not considered. The UK study uses the quantity of primary and secondary non-ferrous metals produced to determine the annual emissions of CDD/CDF compounds, but relies solely upon emissions data from secondary non-ferrous operations, which may be questionable. This study will focus on secondary operations only.

The secondary non-ferrous operations to be considered include:

- copper;
- aluminium; and
- lead.

6.1.13.1 Process Description

Pre-treatment

There are many different methods used for pre-treating the scrap materials to be smelted. The pre-treatment technique identified as a potential source of CDD/CDF compounds is pyrometallurgical cleaning (USEPA, 1995b). This type of cleaning method may be used on

secondary copper and aluminium operations. Depending on the particular non-ferrous metal, this type of cleaning may involve the following operations:

- sweating;
- burning off insulation (particularly from wire scrap);
- drying (burning off volatiles and oil); and
- roasting, which is similar to drying except it vaporises or carbonises organic contaminants in the scrap material.

This pre-treatment source is only identified in the USEPA (1995b) study, and emissions data was only presented for a secondary aluminium plant. The other two studies essentially discuss emission from the smelting furnaces.

Smelting Operations

There are various furnaces employed for smelting/refining of the pre-treated scrap materials. These include:

- cupola/blast furnace;
- reverberatory furnace;
- rotary kiln;
- crucible furnace; and
- multichamber furnace.

Blast Furnace: Alternative charges of scrap, coke, and limestone are placed on top of a burning bed of coke. As the scrap is burned, the metal melts and is drawn off through a tap hole and spout at the bottom of the furnace, and the combustibles are subsequently burned off. Oxides of copper and heavy metals are chemically reduced. Various impurities, such as iron, combine to form a slag, which collects on top of the molten metal and can be drawn off separately

Reverberatory Furnace: This is a brick lined furnace that operates through reflecting (reverberating) the heat from the ignited fuel back down from the roof into the charge material. A typical furnace also has an enclosed melt area where the flame heat source operates directly above the molten metal.

Rotary Kiln: This furnace type is usually a discontinuously operated device, consisting of a horizontal cylinder that slowly rotates around its axis.

Crucible Furnace: The scrap materials are melted in a crucible of refractory material, through direct heating with flame, or by an induction spiral. In order to tap the furnace, it is turned over manually or by hydraulic means.

Multichamber Furnace: The scrap material is fed into one of the two chambers. In the other chamber the scrap is heated directly using a flame, identical to the reverberatory furnace. The warm metal is transported to the room with the scrap, and by circulating the liquid metal through it, the scrap is melted.

6.1.13.2 Emissions Data

Emissions data for each of the international studies are presented in Table 15. The data is primarily for smelting furnaces, unless otherwise indicated. The information is further discussed below the table.

Table 15 – International Emissions Data for Secondary Non-Ferrous Smelting

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne feed) ^a
US ^b - Aluminium - Copper - Lead ^d rotary furnace blast furnace blast/reverb furnace ^e average	11 ^c 780 0.06-0.71 ^f 0.84-8.8 ^f 0.4-1.34 ^f 4.5
UK ^g - Emission Range	5-35 ^h
Netherlands ⁱ - Aluminium - Copper/bronze - Lead	1.7-35 35 5

a These are the units unless otherwise specified.

b Source: USEPA, 1995b.

c This accounts for emissions from the pyrometallurgical pre-treatment of aluminium scrap only.

d The units of this factor is actually µg/tonne of lead produced.

e This source was a blast furnace and a reverberatory furnace which had their exhausts combined prior to control

f These emission ranges represent emissions controlled with a baghouse (upper limit) and emission controlled with a baghouse and scrubber (lower limit)

g Source: HMIP, 1995.

h The units of these factors are µg/tonne of non-ferrous metal produced.

i Source: Bremmer et.al., 1994.

US Emission Data

Aluminium

Emissions data were only available in the US study relating to the pyrometallurgical cleaning of the aluminium scrap prior to smelting. The smelting operations were identified as potential sources of CDD/CDF compounds, however, it then indicates that no US emission factors were located for these sources. The emission factor presented in Table 15 for aluminium is based on test results taken at two delacquering process stack gases. One process was controlled with a venturi scrubber, while the other was assumed to be uncontrolled.

Copper

The emission factor for copper was derived from test results taken at a cupola-type blast furnace. The unit was controlled by gas-fired afterburners and a fabric filter.

Lead

The emissions factor developed for secondary lead processing appears to be based on a greater level of information than for aluminium and copper. The US NESHAP (National Emission Standards for Hazardous Air Pollutants) Program for Secondary Lead Smelting generated test data and subsequently emission factors for the three principal furnace types used in the US (these furnaces are listed in Table 15). A comprehensive NESHAP database was used to estimate national emissions considering emission control and furnace types. Using this national emission estimate and the activity data supplied, the average emission factor shown in Table 15 was determined.

UK Emission Data

The UK study relied on international data to estimate emissions. The emission range presented in Table 15 was determined based on Netherlands data and assumed to represent the entire non-ferrous metal industry. Additional overseas data was reviewed which indicated flue gas levels similar to those determined in the Netherlands study; however, these had not been converted to emission factors. There are two questionable aspects of how the data was used in the UK study:

- the emission data are actually on a 'per tonne of scrap *feed* processed' basis, but have been applied with tonnes of non-ferrous metals *produced* activity data; and
- these emission factors have been applied to both primary and secondary non-ferrous operations.

Primary non-ferrous operations and even secondary operations using clean scrap are not considered sources of CDD/CDF emissions in the Netherlands study. Additionally, the US study only considers emissions from secondary operations.

Netherlands Emission Data

Emission factors were generated based on the source testing of two facilities processing strongly polluted scrap aluminium (resulting in the lower and upper bounds of 'aluminium' emissions in Table 15). Various assumptions were then made regarding emissions from the other facilities processing scrap with varying levels of contamination, and control technologies. The assumptions used are further discussed in the Netherlands report (Bremmer et.al., 1994). A table presenting the emission levels and typical control technologies used by the various non-ferrous industries is presented in Table A13, Appendix A.

As mentioned above, the Netherlands report does not consider emission from the primary non-ferrous metal industry due to the extremely low chlorine contents of the raw materials. Additionally, it was assumed that smelters exclusively processing clean scrap were not dioxin sources. A large number of foundries were also excluded from the analysis as they were not expected to be CDD/CDF sources due to the quality of the scrap processed (i.e. clean).

The Australian Situation

No data is available pertaining to emissions from the secondary non-ferrous metals industry, and so this study will base CDD/CDF estimates on international data. There is no reason to consider the UK data, as it is essentially a summary of the Netherlands information. Therefore, data developed by the US and Netherlands study will be the principal focus. It will be noted that a large fraction of the emission factors presented in Table 15 are in units of 'per tonne scrap feed'. Data is more readily available in terms of 'per tonne of metal product' and so a 0.9:1 ratio will be assumed for this study (i.e. 900 kg of product per tonne of scrap feed). Therefore, all emission factors in terms of scrap feed are divided by 0.9.

Aluminium

The emission factor range presented by the Netherlands study considers smelting operations, while the US data considers only emissions from pyrometallurgical pre-treatment. For this study a range of 1.7-45 µg I-TEQ/tonne scrap feed will be used. The lower end considers best case smelter emissions without the use of pre-treatment, while the upper end considers worst case emissions with the use of pyrometallurgical pre-treatment. However, as discussed

above this will be converted to units of 'per tonne product' by dividing through by 0.9, resulting in an emission range of 1.9-50 µg I-TEQ/tonne product.

Copper

It is assumed that the emissions information for copper/bronze (Netherlands) can be compared directly with emissions from copper only (US). While the US data is based on source testing of a single source, the Netherlands data was developed based on assumptions combined with source testing of secondary aluminium smelters. Each of these sources therefore has their shortcomings, however based on the limited data available an emission range of 35-780 µg I-TEQ/tonne scrap feed will be used. Upon conversion this equates to 39-870 µg I-TEQ/tonne product.

Lead

The average emissions factor developed for the US study and the Netherlands data compare very well. While the US data is in the correct units, upon conversion the Netherlands factor becomes 5.5 µg I-TEQ/tonne lead product. Therefore, it will be assumed that an average emission factor of approximately 5 µg I-TEQ/tonne product is representative of the Australian industry.

6.1.14 Landfill Gas

6.1.14.1 Process Description

A municipal waste landfill unit receives household waste, as well as other types of wastes such as commercial solid waste, nonhazardous sludge, and industrial solid waste. CDD/CDF emissions from MSW landfills are expected to originate from the non-household sources of MSW (USEPA, 1995b).

Modern landfill design often incorporates liners constructed of soil (e.g., recompacted clay) or synthetics (e.g., high density polyethylene) or both to provide an impermeable barrier to leachate and gas migration from the landfill (USEPA, 1995b).

Landfill gas collection systems can be broken down into two categories:

1. active systems; or
2. passive systems.

Active systems provide a pressure gradient in order to extract landfill gas by use of mechanical blowers or compressors.

Passive systems allow the natural pressure gradient created by the increase in landfill pressure from landfill gas generation to mobilize the gas for collection.

Landfill gas control and treatment options include:

- combustion of the landfill gas; and
- purification of the landfill gas.

Combustion Techniques

Combustion techniques include those that do not recover energy (e.g., flares and thermal incinerators) and techniques that recover energy and generate electricity from the combustion

of the landfill gas (e.g., gas turbines and internal combustion engines). Boilers can also be employed to recover energy from landfill gas in the form of steam. Flares involve an open combustion process that requires oxygen for combustion; the flares can be open or enclosed. Thermal incinerators heat an organic chemical to a high enough temperature in the presence of sufficient oxygen to oxidize the chemical to CO₂ and water.

Purification Techniques

Purification techniques can also be used to process raw landfill gas to pipeline quality natural gas by using adsorption, absorption, and membranes.

6.1.14.2 Emissions Data

Emissions from landfill gas that is untreated or controlled through combustion techniques are the principal sources of CDD/CDF emissions considered in this study. As natural gas combustion is not considered an emission source, purified landfill gas combustion is also not considered a source. Landfill gas emissions data for each of the international studies are presented in Table 16. This information is then discussed below the table.

Table 16 – International Emissions Data for Landfill Gas

Emissions Study	CDD/CDF Emissions (ng I-TEQ/m³ landfill gas)
US ^a - Flared Gas	2.5
UK ^b - Escaping Gas - Flared Gas - Gas Engine (eg turbine)	0.32-0.36 0.022 ^c 0.006-1.2 ^c
Netherlands ^d - Flared Gas - Gas Engine	0.4 0.5

a Source: USEPA, 1995b.

b Source: HMIP, 1995.

c These emission factors have units of ng I-TEQ/m³ of flue gas after combustion. The flared gas emission factor is actually equivalent to the Netherlands data (0.4 ng I-TEQ/m³ landfill gas), and the gas engine factors are equivalent to 0.06-12 ng I-TEQ/m³ landfill gas (further discussed below).

d Source: Bremmer et.al., 1994.

US Emission Data

Limited data was available for US related landfill gas treatment. Landfill gas was not actually considered in the inventory due to the lack of activity data. The emission factor shown in Table 16 was determined based on speciated CDD/CDF emissions from a single gas flare, using the I-TEQ equivalencies shown in Table 1. The original emissions data were presented in units of ng/MJ of gas flared. It was assumed that 50% of the landfill gas is carbon dioxide, with the remaining fraction as methane (USEPA, 1995c) and the heating value of the gas was thus approximately 19 MJ/m³ (i.e. about half the heating value of methane). Table A14 in Appendix A presents the speciated 2,3,7,8-substituted isomers.

UK Emission Data

The emissions data presented in Table 16 are based on Netherlands information, as well as a review of data for the UK. The emission factor presented for flared gas has actually been

extracted directly from the Netherlands study, and based on additional information presented in that study, is equivalent to 0.4 ng I-TEQ/m³ landfill gas. Furthermore, it is assumed that 1 m³ of landfill gas generates 10 m³ of flue gas. Using this assumption, an emission range of 0.06-12 ng/m³ landfill gas is determined for gas engines. This conversion allows an easier comparison of emission factors using the same basis as the other data (i.e. per m³ of landfill gas rather than flue gas).

Netherlands Emission Data

Emissions testing were performed at two landfill installations, one using a gas engine and the other a flare (results in Table 16). Both facilities did not utilise flue gas control technologies. Emissions were then determined based solely on these test results.

The Australian Situation

Due to the lack of source test data regarding landfill gas and its combustion, reliance on international data is required to estimate emissions. The only data considering the CDD/CDF content of raw landfill gas was from the UK study, and the range is so marginal that an emission factor of 0.36 ng I-TEQ/m³ will be used for this study (to be slightly conservative).

Of the emissions captured the following emission factors will be used:

- gas flaring: 0.4-2.5 ng/m³ landfill gas (based on Netherlands and US data); and
- gas engines: 0.5-12 ng/m³ of landfill gas (to be conservative, based on Netherlands and UK data).

6.1.15 Cement (and Lime) Manufacturing

Australia-specific emissions data was provided by the Australian Cement Industry Federation (CIF). These data provide the basis for the assessment of emissions from this sector.

Cement manufacturing is considered a CDD/CDF emission source by the three principal overseas studies that were also considered for emissions data. The UK study also considers lime production as a potential source and, therefore, lime production is considered in this section.

6.1.15.1 Process Description

Cement Manufacturing

Raw materials used in the process can be limestone and aluminium-containing limestone, iron, silicon oxides, shale, clay, sand and various ash materials (such as pyrite ash and fly ash). The feed materials enter the kiln at the elevated end, and the burner is located at the opposite end. The raw materials are then changed into cementitious oxides of metal by a counter current heat exchange process. The materials are continuously and slowly moved to the low end by the rotation of the kiln while being heated to high temperatures (approx 1400-1500°C) by direct firing. In this stage, chemical reactions occur, and a rock-like substance called "clinker" is formed. This clinker is then cooled, crushed, and blended with gypsum to produce the cement product.

Cement may be made via a wet or a dry process. In the wet process, water is added to the mill while the raw materials are being ground. The resulting slurry is fed to the kiln. In the dry process, raw materials are also ground finely in a mill, but no water is added and the feed enters the kiln in a dry state.

Lime Manufacturing

Lime is produced in essentially the same manner as cement. The most significant difference is that the main raw material is limestone only.

6.1.15.2 Emissions Data

Emissions data for each of the international studies and the Australian data are presented in Table 17. This information is then discussed below the table. No Netherlands data is presented, as insufficient information was available from which to determine emission factors. Note also that lime production was only considered in the UK study.

Table 17 – International Emissions Data for Cement/Lime Production

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne clinker)
Australia ^a - standard fuel - with waste derived fuel	0.0043 – 0.25 0.0087 – 0.28
US ^b - standard fuel - with waste derived fuel	0.32 24.45
UK ^c - emission range ^d	0.02-1.08

a Source: PAE, 2002).

b Source: USEPA, 1994a.

c Source: HMIP, 1995.

d This emission range was also assumed to apply to lime production.

Australian Emissions Data

The data presented in Table 17 includes data from the Australian Cement Industry Federation. The data presented are the results of repeated measurements over the period 1991-2001 and were obtained from plants using a range of process conditions, primary fuels and raw materials. Both wet and dry processes are represented, as are plants using gas and coal as the primary fuel as well as some plants using waste-derived fuels.

The Australian data are in good agreement with the data from the UK study and with the 1998 US data for facilities using standard fuels. The Australian data indicates that there is no significant difference in dioxin emissions associated with the use of waste-derived fuels. The waste-derived fuels used by Australian cement producers have included used tyres and used oil.

Based on plant production volumes and the range of measured emissions from these plants, and scaling for the Australian industry total production, the annual dioxin emissions are in the range 0.31-0.60 g/year for the Australian cement industry and 0.089-0.17 g/year for the Australian lime industry.

US Emission Data

The data presented in Table 17 were derived by the USEPA from a study of 12 facilities using waste derived fuel and 11 facilities using standard fuel. It was noted that the emission factors had a low confidence rating as a consequence of the small number of facilities tested and, for the facilities using waste derived fuel, to the uncertainty regarding whether the measured emissions were representative of normal operations due to trial burn procedures.

It was also noted that observed differences in emissions might have been due to differences other than the use of waste-derived fuel, such as primary fuel, type of secondary fuel and procedures for heating the raw material.

UK Emission Data

The UK study reviewed quite a large number of literature sources, both within the UK and overseas. The data reviewed is summarised as follows:

- US: 0.41-6.93 µg I-TEQ/tonne product (standard fuel);
- Germany: 0.12-0.59 µg I-TEQ/tonne product (standard fuel);
- UK: 0.025-1.04 µg I-TEQ/tonne product (standard fuel); and
- UK: 0.025-1.08 µg I-TEQ/tonne product (waste derived fuel)

The US data reviewed by the UK study is in relatively good agreement with the US data presented in Table 17 for facilities using standard fuel. However, the additional data reviewed indicates much lower emissions from facilities using both standard and waste derived fuels.

It is difficult to make direct comparisons with the limited data available, as the emissions are likely to depend upon the fraction of hazardous waste in the fuel. That is to say that a facility burning almost completely hazardous waste derived fuels may emit greater levels than a facility that relies on hazardous materials in a limited capacity. From the data that are available, it is not clear whether or not this is the case.

The final emission range used was determined based on testing performed by the British Cement Association (BCA). Both sets of results shown above for the UK were determined by the BCA. It was found in the UK study that emissions differ marginally between cement works using waste-derived fuel and those that do not (as indicated by test results).

It is indicated, however, that only waste liquids (i.e. solvents) are used to supplement fuel requirements at UK cement kilns, while the USEPA (1995b) indicate that in addition to solvents, materials such as chipped rubber and shredded municipal waste are used in US kilns. Australian cement manufacturers use waste tyres and waste liquids to supplement fuel requirements and, based on the available test data, no significant differences in dioxin emissions have been observed.

The review of a study comparing emissions from the manufacture of lime and cement in Germany indicated that emissions are almost identical. Table A15 in Appendix A presents the emissions data from this German study. One questionable aspect of the data relates to emissions with coal as the fuel. The 'German' emissions data presented above (i.e. 0.12-0.59 µg/tonne) are identical to the data presented in the table in Appendix A, except with a factor of 10 difference. Based on the references in the UK study the data has been taken from the same source, however, it is not clear which is correct or why there is a difference. Nonetheless, the similarity between cement and lime production as shown by the German data is still evident. Based on this data the UK study used the same emission factors for both

cement and lime production. Emission factors for Australian cement production have been used to derive emissions from Australian lime production.

6.1.16 Glass Manufacturing

6.1.16.1 Process Description

This process involves the use of sand, limestone, sodium carbonate and other materials (such as dolomite, clay or feldspar). These materials are crushed prior to being fed into a melting furnace. The furnace melts the raw materials using temperatures up to 1550°C to produce a viscous flowing mass. The molten glass passes through a refiner that heat conditions the glass prior to forming. The glass may then go on to be shaped by blowing, pressing, drawing, rolling or floating to give the desired product.

6.1.16.2 Emissions Data

There appears to be very little emissions data relating to glass manufacturing. The US study does not consider glass manufacturing, while insufficient information is available in the Netherlands study to determine emission factors.

Emission factors are presented in the UK study, however, these are estimates and are not based on source test data. This emission data is presented in Table 18, and is further discussed below the table.

Table 18 – International Emissions Data for Glass Manufacturing

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne product)
UK ^a	0.002-0.005

^a Source: HMIP, 1995.

The emission factors presented above are assumed based on similarities between the cement industry and the glass industry. The principal similarity is that emissions arise from fuel use. These emissions appear to be based on the German emissions factors for cement manufacturing (fired with oil and gas) that are presented in Table A15, Appendix A. A comparison is drawn between cement and glass manufacture, regarding the average energy usage per tonne of product. Based on the same order of magnitude of fuel usage and assuming that oil and/or gas are used in the glass industry, the emission factors presented in Table 18 were used.

The Australian Situation

As there is a general lack of information both in Australia and overseas, the estimates produced for this industry are subject to more uncertainty than for the other industries. However, based on the limited data, the emission range presented in Table 18 will be used for this study.

6.1.17 Ceramic Production

6.1.17.1 Process Description

Raw materials such as clay, shale and fly ash are mixed with water and formed into the required ceramic shape (eg. brick). The formed materials are then dried with waste heat

from the firing kiln. Once almost completely dry the materials are fed into the kiln, where firing typically occurs in six stages (USEPA, 1995c):

- evaporation of free water;
- dehydration;
- oxidation;
- vitrification;
- flashing; and
- cooling.

Temperatures of approximately 400-600°C are maintained in the earlier stages, reaching a maximum of 1000°C during the intermediate stages. Cooling then occurs prior to storage.

6.1.17.2 Emissions Data

As for glass manufacture, ceramic production also seems to have very little emissions data. The US study does not consider it, while again the Netherlands study does not provide sufficient information to determine emission factors. Emissions data from the UK study are presented in Table 19 below.

Table 19 – International Emissions Data for Ceramic Production

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne product)
UK ^a	0.002-0.005

^a Source: HMIP, 1995.

The emission factors presented above are assumed based on similarities between the cement industry and the glass industry. Again the principal similarity is that emissions arise from fuel use. Refer to the previous section for how the emission factors were determined.

The Australian Situation

As there is a general lack of information both in Australia and overseas, the estimates produced for this industry are subject to more uncertainty than for the other industries. However, based on the limited data, the emission range presented in Table 18 will be used for this study.

6.1.18 Asphalt Mixing Processes

6.1.18.1 Process Description

Hot mix asphalt (HMA) paving materials are a mixture of well-graded, high-quality aggregate (which can include reclaimed asphalt pavement [RAP]) and liquid asphalt cement, which is heated and mixed in measured quantities to produce HMA.

Hot mix asphalt paving materials can be manufactured by (USEPA, 1995c):

1. batch mix plants;
2. continuous mix (mix outside drum) plants;
3. parallel flow drum mix plants; and
4. counter flow drum mix plants.

However, there may be other variations within these principal manufacturing types.

Batch Mix Plants

Processing begins as the aggregate is hauled from the storage piles and is placed in the appropriate hoppers of the cold feed unit. The material is metered from the hoppers onto a conveyer belt and is transported into a rotary dryer (typically gas- or oil-fired). Dryers are equipped with flights designed to shower the aggregate inside the drum to promote drying efficiency.

As the hot aggregate leaves the dryer, it drops into a bucket elevator and is transferred to a set of vibrating screens prior to being dropped into individual "hot" bins according to size. To control aggregate size distribution in the final batch mix, the operator opens various hot bins over a weigh hopper until the desired mix and weight are obtained. Reclaimed asphalt pavement may be added at this point, also. Concurrent with the aggregate being weighed, liquid asphalt cement is pumped from a heated storage tank to an asphalt bucket, where it is weighed to achieve the desired aggregate-to-asphalt cement ratio in the final mix.

The aggregate then passes through several additional mixing operations prior to being conveyed to a hot storage silo or dropped directly into a truck and hauled to the job site.

Parallel Flow Drum Mix Plants

This process is a continuous mixing type process, using proportioning cold feed controls for the process materials. Aggregate, which has been proportioned by size gradations, is introduced to the drum at the burner end. As the drum rotates, the aggregates, as well as the combustion products, move toward the other end of the drum in parallel. Liquid asphalt cement flow is controlled by a variable flow pump electronically linked to the new aggregate and RAP weigh scales. The asphalt cement is introduced in the mixing zone midway down the drum in a lower temperature zone, along with any RAP and particulate matter (PM) from collectors.

The mixture is discharged at the end of the drum and is conveyed to either a surge bin or HMA storage silos. The exhaust gases also exit the end of the drum and pass on to the collection system.

Most parallel flow drum mixers are followed only by primary collection equipment (usually a baghouse or venturi scrubber). However, because the mixing of aggregate and liquid asphalt cement occurs in the hot combustion product flow, organic emissions (gaseous and liquid aerosol) may be greater than in other processes.

Counterflow Drum Mix Plants

In this type of plant, the material flow in the drum is opposite or counter flow to the direction of exhaust gases. In addition, the liquid asphalt cement mixing zone is located behind the burner flame zone so as to remove the materials from direct contact with hot exhaust gases.

Liquid asphalt cement flow is controlled by a variable flow pump that is electronically linked to the virgin aggregate and RAP weigh scales. It is injected into the mixing zone along with any RAP and particulate matter from primary and secondary collectors.

Because the liquid asphalt cement, virgin aggregate, and RAP are mixed in a zone removed from the exhaust gas stream, counter flow drum mix plants will likely have organic emissions (gaseous and liquid aerosol) that are lower than parallel flow drum mix plants.

Recycle Processes

In recent years, the use of RAP has been initiated in the HMA industry. Reclaimed asphalt pavement significantly reduces the amount of virgin rock and asphalt cement needed to produce HMA.

In the reclamation process, old asphalt pavement is removed from the road base. This material is then transported to the plant, and is crushed and screened to the appropriate size for further processing. The paving material is then heated and mixed with new aggregate (if applicable), and the proper amount of new asphalt cement is added to produce a high-quality grade of HMA.

6.1.18.2 Emissions Data

Asphalt mixing installations are only considered in the UK and Netherlands studies. However, the UK study relies solely on Netherlands data and so only the latter source is presented in Table 20 below.

Table 20 – International Emissions Data for Asphalt Mixing Plants

Emissions Study	CDD/CDF Emissions (ng I-TEQ/tonne product)
Netherlands ^a	47

^a Source: HMIP, 1995.

The emission factor presented above was based on source test results from a single facility that processes recycled asphalt as well as new asphalt (approx. 50% is recycled). It was noted that the emissions estimated in the Netherlands might be overestimated due to the fact that the plant uses recycled asphalt, which may subsequently result in higher emissions. The installation was fired with natural gas, and utilises cyclone and fabric filter control technologies.

The Australian Situation

It is clear that there are many variations to the mixing process, with limited information pertaining to emissions from asphalt installations in general. Furthermore, no test information is available for the asphalt industry in Australia. Therefore, with the lack of better data, the emission factor presented above will be used in this study. As noted by the Netherlands study this may be conservative, however insufficient information is available to draw any definitive conclusions.

6.1.19 Activated Carbon Regeneration

6.1.19.1 Process Description

In the regeneration process, organics adsorbed on carbon during use are burned off by placing the spent carbon in continuous internally or externally fired rotary retorts or in multiple-hearth furnaces. In the latter furnace type, the charge (carbon) is stirred and moved from one hearth to the next lower hearth by rotating rabble arms. For smaller-scale regeneration operations, fluidized-bed and infrared furnaces can be used.

In a typical regeneration process, spent carbon in a water slurry form is fed from a surge tank to a dewatering screw, which feeds the spent carbon to the top of the furnace. In the furnace, the spent carbon is dried and the organics on the carbon are volatilized and burned as the carbon is regenerated. The regenerated carbon is then typically fed to a quench tank and is stored as a slurry.

A hot gas, such as steam or carbon dioxide, is introduced into the furnace at temperatures of approximately 800 to 1,000°C, although some excess oxygen is typically present throughout the furnace. The regeneration process is exothermic, using the heating value of the volatile carbon plus heat supplied from supplemental fuel (e.g., natural gas).

6.1.19.2 Emissions Data

Emissions data for each of the international studies are presented in Table 21. This information is then discussed below the table. Note that the Netherlands data did not consider regeneration of activated carbon, and so only information from the other two studies are presented in the table.

Table 21 – International Emissions Data for Carbon Regeneration

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne regenerated carbon)
US ^a	1.75
UK ^b	0.75

a Source: USEPA, 1995b.

b Source: HMIP, 1995.

US Emission Data

The emissions factor presented in Table 21 is a weighted factor, based on furnace type and activity data for the US.

UK Emission Data

The UK study bases its emission data on two published journal sources, indicating that the two sources produce emission concentrations in the region of 0.05 ng I-TEQ/m³. It is then assumed that approximately 15000 m³ of flue gas is generated per tonne of carbon regenerated, which results in the emission factor shown in Table 21.

The Australian Situation

No test data is available for carbon regeneration in Australia, and so the international data will be relied upon in this study. The emissions data for the two international studies are in very good agreement, and so an emission range of 0.75-1.75 ng I-TEQ/tonne regenerated carbon will be used for this study.

6.1.20 Motor Vehicles

6.1.20.1 Process Description

The process description is relatively straightforward for motor vehicles. The petrol engine derives its power from the explosion of a mixture of air and petrol, whereas in the diesel engine the fuel burns rather than explodes. The air-fuel mixture, when ignited, expands rapidly in a cylinder, forcing a piston from the top of the cylinder to the bottom. The exhaust

gases from internal combustion engines are potential sources of CDD/CDF emissions. After exhaust is released from a vehicle, it is diluted approximately 1,000-fold in the first few seconds and cools very rapidly.

Most passenger cars, motorcycles and some trucks are petrol-fueled, but large trucks, buses, and farm and heavy equipment are usually diesel-fueled.

6.1.20.2 Emissions Data

The international emissions data for motor vehicles is presented in Table 22 below.

Table 22 – International Emissions Data for Motor Vehicles

Emissions Study	CDD/CDF Emissions (pg I-TEQ/VkmT) ^a
US ^b <ul style="list-style-type: none"> - Leaded Fuel - Unleaded Fuel - Diesel - Average 	1.1-108 0.36 500 25
UK ^c <ul style="list-style-type: none"> - Cars <ul style="list-style-type: none"> - leaded - unleaded (no catalyst) - unleaded (catalyst) - diesel - Motor Cycles <ul style="list-style-type: none"> - unleaded - Large Buses <ul style="list-style-type: none"> - diesel - Light Vans <ul style="list-style-type: none"> - leaded - diesel - HGV <ul style="list-style-type: none"> - petrol - diesel 	1.1-220 0.36-21 0.36-13 0.65-10 0.36-21 26-37 1.1-220 0.65-10 1.1-220 26-37
Netherlands ^d <ul style="list-style-type: none"> - Leaded Fuel - Unleaded Fuel <ul style="list-style-type: none"> - with catalyst - no catalyst - Diesel 	108 5.1 0.7 12

a Note that a 'pg' is 10⁻¹² grams (or ng/1000000).

b Source: USEPA, 1995b.

c Source: HMIP, 1995.

d Source: Bremmer et.al., 1994.

US Emission Data

Emissions were estimated using the USEPA MOBILE5a software, which estimates emissions from mobile sources. It is not clear as to how extensive the test data used is, however, it would most likely be relatively large. Note also that it is not known as to whether the emissions data was determined through testing of US vehicles, or based on international data, or both. There seems to be large similarities between the US values given in Table 22, and the other studies (neither the UK nor the Netherlands studies used US specific test data). Note that the emissions data used to derive the UK emission factors (present in table A16, Appendix A) were originally reported in the USEPA (1994a) report. It is, therefore, likely that this data is also included in the MOBILE5a software.

UK Emission Data

A large body of literature covering many international studies is reviewed by the UK study, however the entirety of this data was extracted from the USEPA (1994a) report. The data produced by these studies are presented in Table A16, Appendix A. Although it is not clear how some of the emission ranges presented in Table 22 were developed, the data seems to be based principally on this international data.

Netherlands Emission Data

The emissions data presented in Table 22 were developed from the following data as presented in the Netherlands report:

- leaded: 1080 pg I-TEQ/L;
- unleaded (without catalyst): 51 pg I-TEQ/L;
- unleaded (catalyst): 7 pg I-TEQ/L; and
- diesel: 24 pg I-TEQ/L.

The UK study presents consumption factors of 0.1 L/km and 0.5 L/km for petrol and diesel vehicles respectively. These factors were then used to derive the emission factors presented in Table 22 for the Netherlands. Note that these emission factors are the same as those presented in Appendix A (from the UK study), as they are from the same source and converted using the same consumption factors. The diesel emission factor is considerably different, however, and it is noted by the UK study that the USEPA miscalculated that particular factor. The UK study then determines 1.5 pg I-TEQ/km, but this also differs from the factor as developed for this study. It is not clear as to how the error has arisen, or whether the UK study used an alternative consumption factor, however, an emission of 12 pg I-TEQ/km is assumed from the Netherlands data.

The Australian Situation

No data was available pertaining to emissions from motor vehicles in Australia. Therefore, this study will rely on international data to develop emission estimates. The emissions data presented by each of the three international studies are quite comparable, with the exception of the US diesel emission factor. Considering the extensive sources considered in the UK study, the emission factors as presented in Table 22 will be used by this study. However, it is not clear as to why 'leaded' motorcycles are not included in the UK data. Emissions from these sources will be included in this study, with emission factors assumed to be similar to those from leaded cars. This assumption was originally used by the UK study to derive emission factors for 'unleaded motorcycles', 'light vans' and 'heavy goods vehicles (i.e. HGV)'.

Although the Oehme tunnel study is reviewed by the UK study and discussed in the USEPA (1995b) study, it is not considered here due to concerns presented by the UK study and the USEPA (1994a) report. The UK study does not use the data from this study due to uncertainties introduced through extrapolating the tunnel data to tail-pipe emission levels. The USEPA (1994a) report discusses this further, stating concerns that tunnel studies are detecting re-suspended particulates which have accumulated over time, leading to the overestimation of emissions. Data as determined by that study are presented in Table A16, Appendix A.

6.1.21 Fires

Natural fires (i.e. bushfires) are considered in Section 7.1. This section will deal with the following sources:

1. accidental fires; and
2. prescribed burning.

6.1.21.1 Process Description

Accidental Fires

Accidental fires are those that occur in structures and involve a wide variety of materials (eg. paper, plastic, wood, textiles etc.). These types of fires may occur at a number of sources, including (HMIP, 1995):

- buildings;
- warehouses;
- houses; and
- factories.

Combustion conditions are poor and variable, and a wide variety of chemicals may be present, including PCP treated wood, PVC and other aromatics, as well as inorganic compounds that may result in CDD/CDF releases. The potential for emissions is, thus, extremely variable, depending on the conditions of the particular fire and the raw materials involved.

Prescribed Burning

Prescribed burning is the application and confinement of fire under specified weather, fuel moisture, and soil moisture conditions to accomplish planned benefits such as fire hazard reduction, control of undesired species, seedbed and site preparation, wildlife habitat improvement, and tree disease control (USEPA, 1995b).

6.1.21.2 Emissions Data

Accidental Fires

No data pertaining to accidental fires is presented in this report. Each of the studies reviewed expressed that emissions from these sources are subject to considerable variability, and emissions factors were not derived due to the limited data available. The UK study (HMIP, 1995) states that the materials burned and the nature of the fires vary widely and representative sampling of air emissions from a fire would be very difficult to achieve.

Two accidental fires are reviewed by the UK study, the first involving 600 tonnes of rigid PVC and the second involving 200 tonnes of PVC and 500 tonnes of carpeting. The emissions from these events were estimated at 13 kg of total CDD/CDF and 3 mg I-TEQ, respectively. This gives an indication of the wide variation in emissions from these types of fires.

Prescribed Burning

Emissions data for prescribed burning is presented in Table 23 below. The Netherlands study did not derive any emission factors for this source, and so it is not included in the table. Prescribed burning was not considered in the UK study.

Table 23 – International Emissions Data for Prescribed Burning.

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne waste burned)
US ^a	0.5
Europe ^b	10

a Source: USEPA, 1995b.

b Source: The European Environment Agency, 1996.

US Emissions Data

Although there is little discussion regarding prescribed burning emissions, the US study bases emissions on those arising from wood stoves (i.e. residential wood burning), which is the same approach that is used for bushfires.

European Emissions Data

This emission data was contained within the Atmospheric Emission Inventory Guidebook, (The European Environment Agency, 1996) and no discussion was given as to how the emission value was attained. However, it is based on the Netherlands study and a study by Thomas and Spiro (1995). The latter study indicates that emissions for agricultural burning are in the order of 4 µg I-TEQ/tonne waste burned. Assuming that the information from the Netherlands study is based on residential wood burning (as no agricultural emissions data is given), an emission range of 13-28.5 µg I-TEQ/tonne is derived. It appears that the value shown in Table 23 is an average of this data.

Australian Situation

No Australian information is currently available for prescribed burning emissions. Additionally, based on the limited data available from international studies, an emission range of 0.5-10 µg I-TEQ/tonne burned will be used for this study.

6.2 Chemical Processes

In addition to the formation of CDD/CDF emissions from combustion and other thermal processes, these compounds are also produced as unwanted by-products during the manufacture of certain chemicals. Particularly the manufacture of chlorophenols and the reaction of chlorophenols with chlorobenzenes yield CDD as by-products. It is important to note that the CDD/CDF compounds formed remain principally in the chemical product (i.e. solid or liquid), and the potential for releases to the atmosphere are typically quite low (HMIP, 1995). The chemical processing sources of CDD/CDF emissions are further discussed in the following sections.

A similar approach to that used in the UK study will be utilised for this study. This study breaks down emissions sources as follows:

- chemical processes involving halogens and vinyl chloride monomer (VCM) manufacture;
- pesticide manufacture; and

- application of PCP in timber treating processes.

The US study did not derive emissions factors or national emission estimates due to insufficient information, and so is not included in this section. The Netherlands study contained limited data.

The actual release points of CDD/CDF compounds from chemical manufacturing are not clearly discussed. It appears, however, that the principal atmospheric release points are reactor and other process vessel vents.

As for combustion related emissions, a brief discussion will be given regarding the formation of CDD/CDF compounds during chemical processes.

6.2.1 CDD/CDF Formation During Chemical Manufacture

A number of factors influence the amount of dioxins and furans that may be formed in a given manufacturing process, including temperature, pH, catalyst, and reaction kinetics (USEPA, 1995b). In particular the following conditions are considered conducive to the formation of CDD/CDF compounds (HMIP, 1995):

- temperatures of greater than 150°C;
- alkaline media (i.e. high pH); and
- the presence of radicals in the reaction mixture.

The likely mechanisms for CDD/CDF formation during the manufacture of particular halogenated products are briefly discussed below. These formation mechanisms are discussed in greater detail in the UK study (HMIP, 1995) and also the USEPA (1994a) report.

Chlorophenols

Chlorophenols are manufactured using two principal processes:

1. direct chlorination of phenol with chlorine gas (the products are subsequently distilled); or
2. using basic hydrolysis of chlorobenzenes, where the corresponding chlorobenzene and aqueous/methanoic sodium hydroxide are heated to about 200°C.

One route of formation of CDD/CDF compounds is via the reaction of precursor compounds. The most likely precursors are compounds with an oxygen atom directly attached to the benzene ring, and with chlorine in the ortho-substituted position relative to the oxygen atom. Chlorophenols fall into this category and are thus considered to exhibit the greatest potential for CDD/CDF formation (HMIP, 1995).

The potential for CD/CDF formation may arise through the following pathways (HMIP, 1995):

1. direct chlorination of phenols, when chlorophenols are purified by distillation;
2. during hydrolysis, CDDs are formed through chlorophenolate condensation;
3. via radical reactions, because chlorine can act as a radical starter; and
4. through pyrolysis of chlorophenols and chlorophenates at temperatures greater than 250°C.

In addition to chlorophenols, CDD precursors such as 2-phenoxyphenols and chlorinated diphenyl ethers have been shown to readily produce CDD under pyrolysis (HMIP, 1995).

Chlorobenzenes

The formation of CDD from chlorobenzenes is only to be expected with chlorophenols as intermediates and in the presence of oxygen donors. Under pyrolytic conditions at 620°C in the presence of air, CDFs are formed from tri-, tetra-, and pentachlorobenzene, with the latter resulting in low levels of HpCDFs and OCDF (HMIP, 1995).

Table 24 below presents the likely CDD compounds formed during the use of chlorobenzenes in the manufacture of chlorophenols.

Table 24 – CDD Contaminants Associated with Chlorobenzenes ^a

Chlorobenzene	Dioxin Contaminant
1,2-Dichlorobenzene	Dibenzo-p-dioxin
1,2,3-Trichlorobenzene	1,6-Dichlorodibenzo-p-dioxin
1,2,4-Trichlorobenzene	2,7-Dichlorodibenzo-p-dioxin
1,2,3,4-Tetrachlorobenzene	1,2,6,7-TCDD
1,2,3,5-Tetrachlorobenzene	1,3,6,8-TCDD
1,2,4,5-Tetrachlorobenzene	2,3,7,8-TCDD

^a Source: USEPA, 1995b.

Chlorinated Aliphatics

It is assumed that CDD/CDF formation mechanisms in aliphatic chlorine chemistry occurs by way of aromatics or chlorinated aromatics that are formed from aliphatic compounds. A chlorine atom may then be substituted with an oxygen atom (from a source such as sodium hydroxide or atmospheric oxygen) to give the necessary CDD/CDF structure. These reactions can occur as a result of high temperatures (above 150°C) that may be employed during gas phase chlorination processes (HMIP, 1995).

Chlorine in Intermediate Chemicals

Chlorinated intermediates may be used in the following types of processes:

- alkaline hydrolysis to transform the chloroaliphatics into aliphatic alcohols; and
- chloroaliphatic by-products are used to produce perchloroethylene and carbon tetrachloride.

Removal of an ortho-halogen from the phenolic ring is capable of producing α -ketocarbene and subsequently CDD/CDFs. Other compounds capable of undergoing similar processes include sulfonic acids and esters, nitro groups and carboxylate esters, however, CDD/CDFs have not been found in these types of compounds (HMIP, 1995).

6.2.2 Chemical Processes Involving Halogens and VCM Manufacture

The UK study (HMIP, 1995) states that the potential for atmospheric releases from these sources is very low. Emissions data is presented based on the vinyl chloride monomer (VCM) manufacturing process, as follows:

- emission rate: 0.053 ng I-TEQ/m³ of flue gas.

The UK study then assumes a flue gas generation rate of approximately 500 m³ per tonne of product. This results in an emission factor of 0.025 µg I-TEQ/tonne of product. This emission factor is then assumed to apply to all processes involved in the manufacture of halogenated compounds.

The USEPA (1994) report reviews a Greenpeace study performed in 1993. The emission factors derived in the Greenpeace study were based on emission factors used by European PVC manufacturers, which was 0.1-5 µg I-TEQ/tonne VCM produced. This emission factor covered emissions to air, water and land. It is typically believed that emissions to the air form a very small fraction of CDD/CDF emissions from chemical manufacturing (i.e. most are retained in the liquid and solid residues). Thus, if ten percent of the European emission factor is assumed to represent air releases then a factor of 0.01-0.5 µg/tonne is derived, which is in good agreement with the UK emission factor. The Greenpeace report however states an emission range of 50-100 µg/tonne, arguing that 'diffuse' emissions constitute a significant fraction of total emissions. The USEPA (1994) study indicates that this appears to be the only rationale used to justify the large increase from the base European emission factors.

Due to the undocumented nature of the derivation of the Greenpeace emission factors and the comparability between the UK emission factor and that used by the European PVC manufacturers, an emission range of 0.025-0.5 µg/tonne product will be used in this study. As for the UK study, this will be assumed to apply to all processes involving halogenated compounds

It is however apparent that the potential for CDD/CDF releases via fugitive sources (such as flanges, pumps, valves etc.) requires further consideration.

6.2.3 Pesticide Manufacture

As for the section above, the UK study (HMIP, 1995) indicates that atmospheric emissions from pesticide manufacture are likely to be very low. Emissions data is based on the Netherlands source testing information taken at a pesticide plant, as follows:

- 0.02-0.05 ng I-TEQ/m³ flue gas.

It was then assumed, as for the above section, that 500 m³ of flue gas is generated per tonne of product, resulting in an emission factor of between 0.01-0.025 µg I-TEQ/tonne product.

However, reviewing the Netherlands study (Bremmer et.al., 1994) shows that an additional but smaller pesticide plant was also tested. The results from this plant indicated emission levels as high as 7 ng I-TEQ/m³ flue gas. When applying the above flue gas conversion, an emission factor of 3.5 µg I-TEQ/tonne is derived.

It is not clear as to why the emissions test data from this smaller plant were not included in the UK study. Nonetheless, for this study an emission range of 0.01-3.5 µg I-TEQ/tonne product will be used, incorporating the test result from the smaller plant. Note that both plants tested utilised gas scrubber pollution control.

6.2.4 Use of PCP in Wood Treating Processes

Pentachlorophenol (PCP) can be dissolved in an organic solvent and applied to wood as a preservative. A water-soluble salt can also be made from PCP (Na-PCP), and is used as a fungicide.

The UK study (HMIP, 1995) uses the following characteristics of PCP and Na-PCP to estimate CDD/CDF emissions:

- PCP and Na-PCP contain 2.5 g I-TEQ/tonne and 0.7 g I-TEQ/tonne respectively; and
- approximately 0.1 percent of the CDD/CDF content will volatilise annually.

Using the above data, and emission range of between 0.7 mg I-TEQ/tonne (for Na-PCP) and 2.5 mg I-TEQ/tonne for PCP.

The Netherlands study (Bremmer et.al., 1994) uses the following information to derive emissions estimates for atmospheric releases of CDD/CDF compounds:

- PCP and Na-PCP contain 3 g I-TEQ/tonne and 0.3 g I-TEQ/tonne respectively; and
- approximately 0.45 percent of CDD/CDF content present in PCP will volatilise annually.

Note that no data was given for the expected evaporative losses from Na-PCP. Nonetheless, using the data for PCP, an emission rate of about 14 mg I-TEQ/tonne is derived.

Using the information from both studies, which produce comparable estimates, an emission range of 0.7-14 mg I-TEQ/tonne will be used to represent PCP and Na-PCP related emissions.

7 NATURAL SOURCES

Chlorinated dioxins and furans are not found to exist in nature under normal conditions. There is no evidence to suggest that CDD/CDF compounds are formed biosynthetically by living organisms (USEPA, 1995b). However, emissions of CDD/CDF compounds can be generated through the natural bushfires. As bushfires are prevalent in Australia the inclusion of this source is important. The following section discusses any pertinent emissions data available for estimating emissions.

7.1 Bushfires

Emissions data pertaining to bushfires is scarce, and any data available is subject to considerable uncertainty. Furthermore, the problems in collecting representative samples are very difficult to overcome (HMIP, 1995).

Emissions data from the US and UK studies are presented in Table 25 below. The Netherlands study did not actually develop any emission factors for use in the estimation of bushfires due to the limited data available, and so it is not included in the table.

Table 25 – International Emissions Data for Bushfires.

Emissions Study	CDD/CDF Emissions (mg I-TEQ/tonne bush burned)
US ^a	0.5
UK ^b	1-28

a Source: USEPA, 1995b.

b Source: HMIP, 1995.

US Emissions Data

The US factor for bushfire emissions is based on uncontrolled residential wood burning using an open door. It was noted that forest fires and wood stove combustion conditions differ significantly and so the emission factor is considered highly uncertain. The actual emission factor for fireplaces is 1 µg I-TEQ/tonne 'dry wood burned', however this was changed to 0.5 µg I-TEQ/tonne 'total wood burned' assuming approximately 50% moisture content of the wood. This was previously discussed in Section 6.1.8.2.

The USEPA (1994) report investigated three possible methods of estimating CDD/CDF emissions from bushfires. Each of these methods and the corresponding emissions estimated were:

- Soot based approach: 54 µg I-TEQ/tonne burned;
- CO approach: 1000 µg/tonne burned; and
- wood stove approach: 1 µg/tonne burned (this is the same factor used to derive the US value shown in Table 25).

It was then stated that the wood stove approach gave the best estimate of likely CDD/CDF emissions.

UK Emissions Data

The emissions range presented in Table 25 was determined based on emission factors developed for residential wood burning in uncontrolled and open fireplaces (i.e. similar to the US approach). Emissions data for residential wood burning from the Netherlands study and an additional European source were used.

Additional data was reviewed relating to emissions based on CDD/CDF content in chimney soot (indicating emission levels between 18-1125 µg/tonne). However, this data was not used due to the possibility that CDD/CDF compounds may accumulate in deposited soot over time, and hence be unrepresentative of actual emissions (with the likelihood of overestimation.)

Australian Situation

No Australian information is currently available for bushfire emissions. The US emission factor is comparable with the lower limit of the UK data. Thus, an emission range of between 0.5-28 µg/tonne burned will be used for this study.

8 EMISSION QUANTIFICATION

Based on the above analysis of international data, it appears that where sufficient reliable data was available averages were taken and assumed to be representative of the source population. If this were not the case then an emission range was used to account for the possible emissions variation within a particular source group. Australian information is very scarce, so averages cannot be determined with enough confidence to state that it would be representative of the source population. Therefore, for this study emission ranges tend to have been used to account for (1) the possible uncertainty when applying the international data to Australia; and (2) to account for the variation with a particular source group. It is also important to note that the original international emission factors are also subject to uncertainty.

8.1 Methodology

The methodology used to quantify emissions is very simple. The emission factors are multiplied by the relevant 'activity data' to determined annual CDD/CDF emission estimates. The activity data is simply related to the units of the emission factor. For example, the waste incineration emission factors are typically in units of 'per tonne waste incinerated', and so the activity data required would be the annual quantity of waste incinerated.

8.2 Australian Emission Estimates

The emissions for each of the CDD/CDF sources covered by this study are quantified in Table 26 below. Emissions (g/year) for all sources except for Cement and Lime Production are calculated by multiplying the emission factor (µg/tonne) by the activity data (ktonnes/year). Emissions for Cement and Lime Production are calculated using production rates at the time of sampling and the total annual production for the facility, and are scaled to the total Australian industry production for 1998.

Note that the estimates presented below are principally for 1994, as this was the most recent year from which collated data for a substantial number of sources was attained. However, processes such as incineration seem to be varying in recent times, and so any data is based on recent operation. For example, municipal incineration may have taken place in 1994, but is no longer operating in Australia.

Table 26 – Annual CDD/CDF Emission Estimates for Australia.

Source	Emission Factor (mg/tonne)	Activity Data (ktonnes/year)	Emission ^a (g/year)
Municipal Waste Incineration ^b	7-277	N/A	0
Medical Waste Incineration ^b	20-480	40	0.9-19
Sewage Sludge Incineration ^b	5-120	ND ^m	ND
Hazardous Waste Incineration ^b	2.2-310	ND	ND
Utility Coal Combustion ^c	0.060-0.35	70000	4.2-24.5
Industrial Coal Combustion ^c	0.040-4.8	10000	0.4-48
Oil Combustion (industrial and utility) ^c			
- clean	0.5	37600	19
- polluted	2-6	400	0.8-2.4
Industrial Wood Combustion ^c			
- clean	1-2	4300	4.3-8.6
- polluted	5-50	1100	5.5-55

Residential Wood Combustion ^c			
- stove			
clean	1-3	2300	2.3-7
treated	10-50	400	4-20
- fireplace			
clean	1-29	1230	1.2-36
treated	100-500	70	7-35
Residential Oil Combustion ^c	0.2-0.5	375	0.075-0.2
Crematoria ^d	2.4-80	ND	ND
Iron and Steel Production ^e	0.7-10	940	0.66-9.4
Sinter Production ^e	1.2-9	7500	9-67.5
Coke Production ^e	0.3	121	0.036
Non-Ferrous Metal Production ^e			
-aluminium	1.9-50	85	0.16-4.2
-copper	39-870	20	0.78-17
-lead	5	26	0.13
Landfill Gas ^{f,g}			
-uncontrolled	0.36	2040	0.73
-flared	0.4-2.5	11	0.004-0.03
-gas engine	0.5-12	150	0.075-1.8
Cement Production	0.0043-0.28 ^e	6232 ^a	0.31-0.60 ^a
Lime Production	0.0043-0.28 ^e	1780 ^a	0.089-0.17 ^a
Glass ^e	0.002-0.005	700	0.0014-0.0035
Ceramic ^e	0.002-0.005	10000	0.02-0.05
Asphalt Mixing ^e	47	310	14.5
Activated Carbon Regeneration ^h	0.75-1.75	ND	ND
Motor Vehicles ^{ij}			
- Cars			
- leaded	1.1-220	61	0.07-13
- unleaded	0.36-21	61	0.02-1.3
- diesel	0.65-10	4.2	0.003-0.042
- Motor Cycles			
- unleaded	0.36-21	0.8	0.0003-0.02
- leaded	1.1-220	0.8	0.0009-0.18
- Large Buses			
- diesel	26-37	1.5	0.04-0.055
- leaded	1.1-220	0.1	0.0001-0.02
- Light Vans			
- leaded	1.1-220	8.5	0.009-1.9
- unleaded	0.36-21	8.5	0.003-0.18
- diesel	0.65-10	9.3	0.006-0.09
- HGV			
- diesel	26-37	7.4	0.2-0.3
Fires – prescribed burning ^k			
- agriculture and forest	0.5-10	6800	3.4-68
- grasslands	0.5-10	124000	62-1240
Halogen Chemicals and VCM ^e	0.025-0.5	ND	ND
Pesticide manufacture ^e	0.01-3.5	ND	ND
PCP Wood Treating ^l	0.7-14	ND	ND
Bushfires ^k	0.5-28	14300	7-400
Total	NA	NA	150-2100

a Emissions (g/year) for all sources except for Cement and Lime Production are calculated by multiplying the emission factor (µg/tonne) by the activity data (ktonnes/year). Emissions for Cement and Lime Production are calculated using production rates at the time of sampling and the total annual production for the facility, and are scaled to the total Australian industry production for 1998.

b The emission factors have units of 'per tonne waste burned'. Sewage sludge incineration, however, is in terms of 'per tonne dry waste burned'.

c	The emission factors have units of 'per tonne fuel combusted'.
d	The emission factors have units of 'per body cremated'.
e	The emission factors have units of 'per tonne of product'.
f	The emission factors have units of 'ng per m ³ of landfill gas'.
g	The activity data are in units of millions of m ³ of landfill gas.
h	The emission factors have units of 'per tonne carbon regenerated'.
i	The emission factors have units of 'pg per vehicle kilometre travelled (VKT)'. Note that a pg = 10 ⁻¹² g.
j	The activity data are in units of billions of kilometres travelled.
k	The emission factors have units of 'per tonne burned'.
l	The emission factors have units of 'per tonne of PCP and Na-PCP used'.
m	ND – no data available

8.3 Data and Assumptions

The source(s) of the activity data presented above, and any associated assumptions are discussed in the following sections.

8.3.1 Incineration Processes

Information pertaining to the quantities of waste incinerated for each of the categories is scarce. The Environmental Departments of each state were contacted, however it was expressed that little data was available relating to annual incineration estimates for each of the categories. This lack of waste characterisation in Australia was further confirmed through discussions held with Mr Charles Jubb (Aquatech Pty Ltd), who is responsible for the waste component of the National Greenhouse Gas Inventory. The incineration element of the waste emissions is currently left empty due to the lack of information.

Information pertaining to medical waste incineration was determined based on discussion held with Chris Clunies-Ross (Unilabs Environmental) who has been involved extensively with the medical waste incineration industry in Australia. Based on this knowledge a crude estimate of the probable quantity of waste burned annually was derived. It was expressed that the value shown in Table 26 may be an upper bound, and is subject to uncertainty.

8.3.2 Fuel Combustion

The National Greenhouse Gas Inventory workbook (NGGIW) for 1994 (NGGIC, 1996a) gives quite an extensive breakdown of fossil fuel and biomass usage in each industry sector, as well as the residential sector. Essentially the total quantities of coal, oil and wood combusted, as presented in Table 26 above, were extracted from this workbook. As the data was presented in energy terms, a conversion was required to estimate mass quantities of each fuel burned. It was assumed that the heating values for each of the fuels are as follows:

- coal: 22 MJ/kg;
- oil: 45 MJ/kg; and
- wood: 20 MJ/kg.

Note, however, that the numbers presented in Table 26 may not correspond directly to these calculations due to rounding. After converting, the data was then broken down into the required categories (i.e. clean and polluted) using various assumptions, as discussed below.

Little data is known regarding the quantities of clean and polluted oil and wood that is combusted annually. The fraction assumed to be polluted is based on international data and is, thus, subject to considerable uncertainty. The following information were used to derive the approximate quantities of polluted oil and wood combusted:

- the Netherlands study (Bremmer et.al., 1994) indicated that approximately 0.6% of oil burned was from waste sources. To be conservative it was, therefore, assumed that waste oil comprises 1% of the total oil combusted in Australia;
- the Netherlands study showed that about 30% of the industrial wood combusted was polluted, while the UK study presented a value closer to 75%. Discussions with Hue Saddler (Economic Energy Analysis), who is involved with the NGGIW, indicated that although little data is available, the quantity burned is likely to be low. Therefore, a fraction of 20% was used for this study, as it was considered comparatively 'low' based on international data;
- two categories are given in the NGGIW for stationary fuel combustion sources (NGGIC, 1996b), indicating that two thirds of the residential wood combusted is done so in wood heaters while the remaining is used for open fires. It was assumed that wood heaters are equivalent with stoves, while open fires may be equated with fireplaces. Within each of these categories the fraction of treated wood burned was then assumed based on UK and Netherlands data. The UK study assumes that about 10% of total wood usage is treated, while the Netherlands data indicates that treated wood comprises 20% and 5% of wood burned in stoves and fireplaces respectively. Additionally, about 60% (i.e. two thirds) of the residential wood burned in the Netherlands was done so in wood stoves, which is similar to the Australian breakdown. For this study it was assumed that treated wood accounts for 15% and 5% of the wood burned in stoves and fireplaces, respectively.

It is important to note that the assumptions discussed above are subject to considerable uncertainty. This is because international data has been used and assumed to be representative of Australian trends, which may not be the case. However, based on the limited data available, the above assumptions help give an indication of the possible contributions of these sources to total CDD/CDF emissions. Due to the large emission factors associated with combustion of polluted oil and wood, the emissions are quite sensitive to these estimates. Thus, when better data becomes available, further refinement of the emission estimates involving treated oil/wood combustion will be required.

8.3.3 Crematoria

No data was found pertaining to the number of cremations in Australia each year.

8.3.4 Metal Industry

The principal source of data pertaining to this industry was attained through personal communication with ABARE. With the exception of coke and sinter operations, the principal focus of emissions from iron and steel production relates to the use of scrap materials. Electric Arc Furnaces use 100% scrap feed, with foundries operating in a minor capacity (personal comm. - Andrew Maurer, ABARE). The value presented in Table 26 is the quantity of steel produced using scrap materials in Australia.

Coke production is based on the NGGIW data for 1994 (NGGIC, 1996a). However, sinter production data was not available. Discussions with ABARE indicated that assuming 100% conversion of iron to steel is reasonable, with the annual production of steel in blast furnaces (i.e. from iron and not scrap) estimated at 7500 kt. USEPA, (1995c) indicates that about 1.4 tonnes of ore and/or sinter is required to produce 1.0 tonne of iron. It is then assumed that 1 tonne of iron is required to produce 1 tonne of steel. Therefore, to attain a reasonable estimate of sinter production, it is assumed that approximately 1 tonne of sinter is required to produce 1 tonne of steel (this accounts for any additional materials that may be used in the iron and steel process). Thus, an estimate of 7500 kt of sinter is generated.

All activity data for the secondary non-ferrous metal production was attained from ABARE (personal comm. - Ian Haine (copper, lead) and Matthew Stubbs (aluminium)).

8.3.5 Landfill Gas

The activity data for the quantity of materials landfilled was attained from the NGGIW for waste (NGGIC, 1996c). An estimate of the gas generated was then determined using the methodology as specified by that workbook. The supporting calculations are presented in Appendix B.

Note however that no indication is given as to the quantities of landfill gas that are either flared or burned in gas engines. Discussions with Mr. Charles Jubb (Aquatech) indicated that it would be reasonable to assume about 7 percent of landfill gas is burned in gas engines. It was further indicated that the quantities of gas flared are extremely low (less than 1 percent). It was, therefore, assumed that 0.5 percent of landfill gas is flared. Calculations involving these assumptions are also presented in Appendix B.

8.3.6 Non-Metallic Minerals Industry

Activity data for glass and ceramic manufacture is limited. ABS publications contain information for some parts of each industry but not for the complete industry. Using the data available, the production data presented in Table 26 were determined. Due to the limited nature of the data used for calculations, these estimates are subject to uncertainty. Refer to Appendix B for supporting calculations and assumptions.

Note, however, that although the ceramic and glass activity data are subject to potentially significant error, this is not of major importance. Using the current emission factors (refer to Table 26), production estimates in excess of 1 billion tonnes would be required to even push the emission estimates up to 5 g/year (i.e. using upper bound of range). This is highly unlikely, and so emissions will fall at the lower end of significance in terms of dioxin potential (with current emission factors). In the event that better emissions data is developed, more accurate activity data may be required.

8.3.7 Asphalt Mixing

This data was extracted directly from the NGGIW for 1994 (NGGIC, 1996a). It was assumed that the quantity of bitumen used for spray sealing was indicative of asphalt mixing operations.

8.3.8 Activated Carbon Regeneration

No data was available pertaining to carbon regeneration operations in Australia.

8.3.9 Motor Vehicles

All motor vehicle data was extracted from the NGGIW for 1994 (NGGIC, 1996a). The only limitation of the data pertained to the breakdown between unleaded and leaded usage for each vehicle type. A breakdown was supplied for total petrol usage (approximately 50% leaded and 50% unleaded), and so this was assumed to apply to each vehicle type, for which petrol usage was indicated. An exception was buses, for which a comparatively small travel quantity was indicated, and so leaded petrol was assumed to give a conservative estimate.

Note that in deriving emission factors, the UK study (HMIP, 1995) assumed that emission factors for motorcycles, light vehicles and heavy vehicles were the same as for cars using the same fuels. However, the UK study does not consider leaded motorcycles, petrol buses, or unleaded light vehicles. Therefore, this study has adopted the same approach as used in the UK study to attain emission factors for these missing vehicles types (i.e. assuming similar emissions to cars using the same fuels).

8.3.10 Fires

All of the prescribed burning and wild bushfire data was taken from the NGGIW for 1994 (NGGIC, 1996a).

8.3.11 Chemical Manufacture

No data was attained regarding the production of these chemicals, however, PACIA is assisting in the gathering of this information.

9 DISCUSSION

The discussion in this section will exclude those sources for which no activity data was found. This discussion is thereby seriously limited, as some potentially significant sources are not included. However, in the event that additional data becomes available, a more comprehensive discussion of emissions should be made.

It is clear based on the results presented in Table 26 that emission ranges are quite large for many sources, due to the uncertainties discussed in Section 8. This discussion will adopt a type of worst-case approach, looking at the upper bound estimates of each of the sources and thus considering those sources that are 'potentially' large emitters.

Based on the upper bound of each range, biomass combustion from prescribed burning and wild bushfires are potentially the most significant sources of CDD/CDF compounds in Australia. These two sources contribute approximately 80 percent to the total CDD/CDF estimates. It is difficult to compare this result with the international studies as the UK study did not consider emissions from prescribed burning, and the Netherlands study did not consider emissions from natural or prescribed burning. The US study, however, considers both natural and prescribed fires, but does not find them to contribute significantly to CDD/CDF releases (about 1 percent). The US estimates were made using the lower bounds for both prescribed and natural fires (i.e. 0.5 µg/tonne burned). If this were the case in this study, the contribution from fires would be significantly less, but would still be considered significant in Australia.

The second major group of sources are (in decreasing order of contribution):

1. residential wood combustion;
2. coal combustion (utility and industrial);
3. sinter production; and
4. industrial wood combustion.

Based on upper bound estimates, fossil fuel and biomass combustion contribute about 75% of the emissions from the second major group of sources, with sinter production accounting for the remaining 25%. These four sources contribute about 15% to the total estimates (based on upper bound estimates) and, when combined with prescribed burning and bushfires, account for approximately 95% of total dioxin emissions.

These results contrast with the findings of the other inventory studies reviewed by this study, which found incineration to be the principal source. Although municipal waste incineration was considered the number one source in the US, UK and Netherlands studies, it is not currently operating in Australia. Medical waste incineration was considered to be the second major source in the US and UK studies, however is not considered to be a significant source in Australia based on the activity data currently available. This finding is similar to that of the Netherlands study. Due to the deficiencies in activity data for the additional incineration processes it is difficult to draw any definitive conclusions as to the importance of the incineration operations to CDD/CDF releases in Australia.

If incineration is excluded from the international studies, various similarities can be identified. These include the fact that sintering operations are ranked highly in the UK and Netherlands studies. Additionally, contributions from coal and wood combustion sources generally lie towards the higher end of the source lists.

The contribution of emissions from motor vehicles is more in line with the US study, as both the UK and Netherlands studies found vehicles to be significant contributors. This study, however, finds that motor vehicles are relatively minor sources, contributing less than one percent of total emissions.

10 CONTROL TECHNOLOGIES

Based on the literature reviewed, the principal means of reducing CDD/CDF emissions is indirectly through the control of other pollutants (eg. particulates or acid gases). There are however, technologies that enhance the control of CDD/CDF emissions (in conjunction with indirect controls) by directly targeting these compounds and their formation. This section will be divided into the following two categories:

1. indirect control; and
2. direct control.

The first will investigate pollution control devices and other practices (such as maintaining adequate combustion conditions) that indirectly reduce CDD/CDF emissions. The second section will deal with any control technologies that address CDD/CDF emissions specifically. Note, however, that both of these control types may be used in conjunction with each other for maximal effectiveness.

10.1 Indirect Control

Technologies that aid in the removal of CDD/CDF compounds from flue gas streams, but are not used for the direct control of these emissions include:

- fabric filters;
- electrostatic precipitators (ESP);
- spray dry absorbers;
- dry sorbent injection; and
- wet scrubbers.

These devices are typically involved with controlling combustion sources, which encompasses a significant fraction of the 'thermal' processes discussed in this report. The most effective control arises through using various combinations of these control technologies. Additionally, ensuring good combustion practice (GCP) is considered important to minimising CDD/CDF emissions, and so this will also be discussed.

Other technologies that appear to be used as control technologies in various industries are cyclones and afterburners. It is not clear as to whether these help to reduce CDD/CDF emissions, and if so how effective they are. If afterburners are operated under adequate conditions (i.e. sufficient temperature and residence time) it is likely that they help reduce emissions through destruction of the CDD/CDF compounds. Cyclones on the other hand are effective at collecting larger particulates in the flue gas, and so their reduction effectiveness is likely to be limited. However, no discussions were found pertaining to these controls in terms of their potential to control CDD/CDF emissions. Thus, they will not be discussed here any further.

The technologies discussed in this section principally relate to thermal sources of CDD/CDF compounds. However, although no information was found pertaining to emission controls for the chemical industry, technologies such as wet scrubbers and afterburners are typically used.

10.1.1 Fabric Filters (FFs)

Fabric filters remove particles from by passing it through woven or felted fabric. This fabric is typically in the form of tubular bags that are closed at one end. The open end is attached to a plate, referred to as a tube-sheet, that separates the dirty and clean areas. Collection may occur on the inside or outside of the bags.

As particles are removed from the gas stream by the fabric filter, a porous layer referred to as the filter cake, develops on the bag. It is the filter cake rather than the actual fabric that results in the high collection efficiencies.

Fabric filters may not be used as a sole control device for particulate removal. The reason for this include the following:

- the potential for fabric filter blinding from sticky or oily emissions or moisture condensation;
- filter burning and damage as a result of entrained sparks; and
- fabric deterioration caused by acid gases or high temperature excursions (i.e. above normal operation).

Fabric filters subject to the above problems typically use alkaline sorbent systems (such as spray dryer absorbers, dry injection and wet scrubbing systems). The combined system then controls emissions of particulates, acid gases and chlorinated organics (Crowder and DePaul, 1989). The disadvantages mentioned above are, thus, essentially eliminated when fabric filters are used in conjunction with sorbent systems.

The principal advantage of fabric filters over ESPs in terms of CDD/CDF control is that they are much more efficient at removing particulate matter in the submicron range. Additionally, fabric filters are able to deal with upset process conditions that may result in flue gas flow and compositional changes more effectively than ESPs.

10.1.2 Electrostatic Precipitators (ESPs)

An ESP operates using discharge electrodes that are placed between ground parallel plates, resulting in simultaneous charging and collection. Applying a high voltage to the discharge electrodes creates a corona discharge, resulting in the production of negatively ionised gas molecules. The electric field between the discharge electrodes and collection plates causes the ions to migrate towards the plates, subsequently intercepting particles present in the space. These ions are deposited on the particles, which then become charged and migrate towards the collection plate through electrostatic forces.

The two types of ESPs are 'dry' and 'wet'. A dry ESP collects particles and removes them from the collection plate by 'rapping'. A wet ESP may remove the collected particles either intermittently or continuously using water that flows down the plates.

The disadvantage of ESPs, is that they are not very effective in removing particles in the sized range of 0.1 to 1.0 μm (as mentioned above when discussing fabric filters). Particular care must be taken to ensure that these particles are collected with maximal efficiency, as chlorinated organics such as CDD/CDFs tend to be enriched in these smaller particles (Crowder and DePaul, 1989).

10.1.3 Spray Dryer Absorbers

This type of technology utilises alkaline reagents to remove SO₂, acid gases (HCl and HF) and chlorinated organics. A complete system consists of a reagent preparation system, a spray dryer absorber, a dust collector and an ash handling system. The dust collector may be either an ESP or fabric filter. However, collection of chlorinated organics and some heavy metals is considerably enhanced using a fabric filter (Crowder and DePaul, 1989).

A common reagent is hydrated lime that is prepared as a slurry to a desired concentration. The slurry is subsequently atomized and intimately mixed with the incoming flue gas stream. Sulfur dioxide and acid gases in the exhaust stream are absorbed by the slurry droplets and react to form calcium salts. Chlorinated organics are also captured, with the most likely mechanisms involving both condensation and sorption. Additionally, the heat dries and solidifies the slurry into a powder form. Some of this is collected in the bottom of the dryer unit, however most are carried out and collected by the dust collector (i.e. ESP or FF). Note that additional sorption may occur at the dust collector.

10.1.4 Dry Injection

Injection of dry alkaline reagents into the exhaust stream has also been successful in removing SO₂, acid gases and chlorinated organics. The principal difference between this operation and that of spray dryer absorption is that preparation and handling of the wet slurry is eliminated.

The reagent is injected into a vessel through which the flue gases pass, where it fluidises and mixes with the gas stream. The stream then passes through a collection device such as an ESP or FF. However, the constraint in choosing a collector is providing adequate residence time for reaction with the dry reagent. ESP's tend to lack this requirement unless they are designed to be very large, limiting their effectiveness. With fabric filters the reactions continue (as for spray dryer absorbers) as the gas passes through the filter cake, thereby greatly increasing the effectiveness of control. Therefore, dry injection installations tend to favour fabric filters.

10.1.5 Wet Scrubbing

Wet scrubbing can be used to control particulate emissions, as well as acid gas and SO₂ emissions. There are various types of wet scrubber designs, including:

- packed columns;
- spray towers;
- venturi scrubbers; and
- tray towers.

To discuss each of these designs is beyond the scope of the report. However, the underlying principles of operation will be discussed here.

When scrubbers are used for particulate control, the collection occurs via three principal mechanisms. The first and most dominant mechanism involves 'impaction' of the particle directly into a droplet. 'Interception' of the particle by a droplet, as the particle comes close to the droplet is the second mechanism. The final mechanism involves 'diffusion' of the particle through the surrounding gas, until the particle is close enough to the droplet to be captured.

If acid gases and SO₂ etc. are of concern, the incoming gases enter the scrubber and are contacted with an alkaline solution that absorbs SO₂ and acid gases, as well as chlorinated organics. These materials react with the solution and form relatively insoluble salts. The spent solution exiting the scrubber may then be clarified, thickened and filtered to remove the salts and other collected solids. The sludge may also be stabilised by combination with flyash and/or lime prior to landfilling.

The major limitations associated with wet scrubbers involve corrosion problems and the waste sludge that is formed requiring subsequent disposal.

10.1.6 Good Combustion Practice (GCP)

The maintenance of GCP appears to be more so important in the waste incineration industry (particularly municipal waste). This may be because waste combustion systems involve highly variable and non-homogeneous feed materials, and so control of combustion conditions is more difficult. For combustion systems such as those using fossil fuels however, the relatively constant and homogeneous feed materials may allow easier control of adequate combustion conditions. Therefore, although this discussion may be more relevant to waste combustion, the general principals are important for all combustion systems.

As discussed in Section 6.1.1 CDD/CDF compounds arise from three principal pathways. The first two formation mechanisms are potentially important for 'in-furnace' formation, while the third mechanism is important for downstream formation of CDD/CDF compounds (i.e. in the flue gas). Good combustion practice can maximise the furnace destruction of organics and minimise the downstream formation of CDD/CDFs by controlling the amounts of particulate carried out of the furnace with the flue gas. Furnace destruction of organics must include destruction of both gas- and condensed-phase organics (Kilgroe, 1996).

Combustion related variables that affect the destruction, formation and furnace emissions include (Kilgroe, 1996):

- combustion temperatures;
- residence times;
- the amount and distribution of combustion air; and
- mixing.

Sufficiently high *combustion temperatures* must be combined with adequate *residence times* to ensure complete destruction of organic constituents present. The *amount of excess air* must be high enough to minimise the existence of fuel rich pockets, and sufficiently low to avoid quenching of combustion reactions. *Poor mixing* within the combustion chamber increases the amount of organic material available for CDD/CDF formation. It may result in local stoichiometry that is insufficient for complete oxidisation of gas- and solid-phase organics.

The entrainment and carryover of particulate matter into cooler regions of the combustor and the flue gas can lead to downstream formation of CDD/CDF compounds. Metal ion or fly ash carbon can catalyse condensation formation reactions and fly ash can serve as a source of organics for the de Novo synthesis of CDD/CDFs (Kilgroe, 1996). Methods of limiting particulate carryover include proper furnace design, controlling appropriate airflow rates, and fuel and/or waste loading. As operating load is increased above design limits, air flows increase proportionately and the potential for particulate entrainment and carryover increases (USEPA, 1995b).

10.1.7 Application and Effectiveness

As mentioned above, the most effective control arises from using various different combinations of the aforementioned technologies. It is important to remember, however, that these combinations typically aim at controlling emissions such as particulates and acid gases, rather than CDD/CDFs.

When considering CDD/CDF reduction using particulate controls, the operating temperature of these devices is very important. Both ESPs and FFs have been found to act as chemical reactors that generate and emit CDD/CDF's (Kilgroe, 1996). The formation of these compounds is dependent on the temperature at which these devices operate. Lowering the temperature to less than 250°C results in significant reductions in CDD/CDF formation rates and alters the partitioning of vapour- and solid-phase CDD/CDF's (Kilgroe, 1996).

At temperatures above 250°C, de Novo synthesis of CDD/CDF compounds becomes significant and partitioning of these compounds into the vapour phase increases with temperature. Thus, lower temperatures inhibits additional formation, as well as increasing the fraction of CDD/CDF compounds present in the solid phase, which are then controlled more effectively via particulate collection. The limitation associated with lower operating temperatures involves corrosion due to acid gases. If acid gases are not involved, then lower operating temperatures may be possible without corrosion concerns.

Dry injection of sorbent can be used to reduce acid gases and thus allow lower operating temperatures (through quenching) of the associated particulate collector. Injection into flue gases at elevated temperatures (about 800°C) significantly reduces CDD/CDF formation. The reduction in CDD/CDF yield appears to result from both a reduction in the HCl content in the flue gas and an inhibitory effect on fly ash surface reactions (Kilgroe, 1996).

Spray dryer absorber systems combine the advantages of high flue gas quench rates, sorbents that probably modify CDD/CDF synthesis reactions, and lower PM control device operating temperatures (Kilgroe, 1996). All of these factors result in greatly improved CDD/CDF control.

As indicated above, quenching can reduce the temperatures of the flue gas and associated control technologies, subsequently enhancing CDD/CDF control. Quenching is discussed in more detail in the following section.

Control efficiencies of various combinations of the control equipment discussed above are presented in Table 27. These efficiencies are associated with the application of these technologies to municipal waste incinerators, which have relatively high CDD/CDF loading in the flue gas. Lower efficiencies may therefore be expected when these controls are applied to situations with lower CDD/CDF loading. Additionally, note that these data are from a 1989 source, and so current efficiencies may be even higher due to technological advances.

Table 27 – Efficiencies of Various Indirect Control Technologies. ^a

Control System	Collection Efficiency (%)
ESP ^b	25-50
SD/ESP ^c	70-80
SD/FF ^d	90-99+
DI/ESP ^e	(60-70)
DI/FF	90-99+
SD/DI/FF	90-99+
ESP/WS(1) ^f	(80-90)
ESP/WS(2) ^g	(90-99+)

a Source: Crowder and DePaul (1989). Values in parentheses are not supported by test data, and so are subject to uncertainty. It is not clear as to whether the above efficiencies are for total CDD, total CDD/CDF, or I-TEQ.

b ESP is an abbreviation for electrostatic precipitator.

c SD is an abbreviation for spray dryer absorber.

d FF is an abbreviation for fabric filter.

e DI is an abbreviation for dry injection.

f WS(1) is an abbreviation for single stage wet scrubbing.

g WS(2) is an abbreviation for two stage wet scrubbing.

Based on the above discussions it appears that particulate controls combined with acid gas control results in high reductions of CDD/CDF emissions. Additionally, various direct control methods (discussed in the following section) can also be applied in conjunction with these indirect controls to increase performance even further. However, it is not clear as to the practicality of applying particulate and/or acid gas control technologies to installations where they are not required. For these situations, more focus on the direct controls may be required. A problem associated with this is that many of the direct controls require the use of an indirect control. For example, the collection of injected activated carbon or catalyst is required using a particulate control device.

As shown by Table 27 fabric filters are more effective at controlling CDD/CDF emissions than ESPs. This is also indicated by Kilgroe (1996) who states that FF are more effective due to their higher collection efficiency. It is further indicated that FF may capture some vapour-phase CDD/CDF compounds via collected fly ash carbon as the flue gas passes through the filter cake.

10.2 Direct Control

There is currently limited information available regarding direct emission control technologies, and so any discussions are quite brief. Direct control includes those methods that reduce CDD/CDF levels already in the flue gas, and/or methods that attempt to inhibit the formation of additional CDD/CDF compounds. The principal methods of direct control are:

- activated carbon injection;
- catalyst injection; and
- quenching.

These control technologies are relevant to combustion emissions. No information was available regarding the specific control of airborne CDD/CDF emissions generated by the

chemical industry. However, if wet scrubbers are used they may be able to recover activated carbon or catalyst injected into the flue gas. Note that the practicality of this is unknown.

10.2.1 Activated Carbon Injection

The methodology associated with this control involves the injection of activated carbon prior to a particulate collection device. The carbon then adsorbs the CDD/CDF compounds onto its surface, and is then subsequently collected in the particulate device. Discussions with Chris Clunies-Ross (Unilabs Environmental) indicate that typical practice is to inject activated carbon with lime (i.e. using spray dryer absorbers or dry sorbent injection).

Efficiency data for this type of control is limited. Testing at a municipal waste incinerator involved the injection of carbon at the entrance to the spray dryer absorber, reduced emissions from 46.8 ng/dscm without carbon usage (i.e. only the SDA and an ESP) to 5.6 ng/dscm (Kilgroe, 1996). This is an additional control effectiveness of close to 90 percent. The USEPA (1995b) study indicates that activated carbon injection can achieve greater than 50 percent additional reduction over the same control system without carbon usage.

10.2.2 Catalyst Injection

This is a technology that is currently being developed and, hence, very limited information is available (personal comm. – Chris Clunies-Ross). The principal difference between catalyst injection and activated carbon injection is that the CDD/CDF compounds are actually destroyed using the catalyst. Activated carbon on the other hand collects the CDD/CDFs onto its surface without the subsequent destruction. An additional advantage of this is that the CDD/CDF compounds are not transferred to another medium (i.e. the solid waste from the particulate collector) that then requires disposal.

10.2.3 Quenching

As the name implies, quenching is the process of rapidly reducing the temperature of hot flue gases. High flue gas quench rates reduce the time that entrained particles spend in the temperature range associated with CDD/CDF formation rates (Kilgroe, 1996). Additionally, this reduction then allows lower operational temperatures of subsequent control technologies, which significantly aids CDD/CDF control. As discussed in Section 10.1.7 particulate control equipment such as ESPs and FFs can generate CDD/CDF emissions if operating at incorrect temperatures.

Spray dryer absorbers are essentially quench units as well, as they act to reduce the flue gas temperature in the process of controlling acid gases. However, dedicated quench units utilising water sprays may also be used.

A quench/scrubber system has been developed for use by Sinter Plants (Gerbert et. al., 1995). After passing through a coarse dust separator, the water quench unit rapidly cools the exhaust to 50°C, which is then passed through to the fine scrubber that is capable of removing extremely fine dust particles using specially developed spray nozzles. Both the condensation of VOCs (containing large amounts of CDD/CDFs) and precipitation of CDD/CDF compounds on fine dust occur in the quench and fine scrubber units. These components are subsequently removed from the flue gas in these units.

A water spray quench chamber was used to reduce municipal waste incinerator flue gases to various temperatures prior to control using an ESP. Reducing the flue gas temperature from about 300°C to 200°C shows an associated emission reduction of about 90% for CDD/CDF levels exiting the ESP (Kilgroe, 1996).

11 MEASURING/ESTIMATING CDD/CDF EMISSIONS

Actual measurement of emissions through source testing is considered the ideal methodology. This can be performed through intermittent stack testing or continuous emissions modeling. However, there is currently no feasible method for continuously measuring CDD/CDF emissions (Kilgroe, 1996). For this reason the USEPA's strategy is to place limits on CO concentrations, steam load, particulate control device operating temperature and opacity. Without the option of direct stack testing, there are four principal methods used to estimate emissions to the air (Patrick, 1994):

1. predictive emissions modeling;
2. mass balance;
3. engineering calculation; and
4. emission factors.

Predictive emissions modeling is based on developing a correlation between pollutant emission rates and process parameters. As the mechanisms of CDD/CDF formation are currently not well understood, and many variables are involved, PEM is not considered a valid option.

Mass balance techniques will not be feasible for CDD/CDF emissions, as these compounds are generated during the process. Furthermore, much debate still surrounds the role of feed materials (i.e. chlorinated substances such as PVC) in the formation of CDD/CDF compounds during combustion. Current it is also believed that the presence of CDD/CDF compounds in the feed play a minor role in levels in exiting flue gases.

The USEPA (1994b) report relies essentially on the use of either source testing, or various preferable forms of source test information, which may fall generally into the 'emission factors' category. Engineering calculation is also briefly mentioned as the least preferable method. These methodologies are relatively general and are not specific to any particular regulatory agencies. The only variation that may exist relates to the specific source testing protocols that may be used by a particular environmental regulatory agency.

According to USEPA (1994b) the following hierarchy is preferred when calculating/estimating CDD/CDF emissions:

1. direct stack measurement using approved testing protocols;
2. congener-specific test data from similar facilities;
3. homologue profile emissions data from similar facilities;
4. published emission factors; and
5. engineering calculation.

These options are discussed in relation to combustion sources. Clearly the first option is only valid for plants that are already operational. The other options may, however, be applied to proposed or existing plants. Each of these options will be briefly discussed below.

11.1 Direct Stack Measurement

Stack monitoring provides concentrations and mass release rates of the pollutant. Care should be taken to ensure that the emissions characterisation reflects a wide range of operating conditions and also accounts for deterioration in emissions over the useful life of the facility. Procedures to convert data expressed in concentrations or mass release rates to an emission factor are as follows:

1. Firstly the concentration data must be converted to standard temperature and pressure on a dry gas basis, and a standard percent carbon dioxide or oxygen within the combustion gas (eg. 12% CO₂);
2. This step involves converting the concentration of the congener to an equivalent emission factor in units of mass of pollutant per unit mass of feed material burned during the period of testing. This is done as follows:

Where:

E_f = emission factor, $\mu\text{g/kg}$;

C_{fg} = concentration in the flue gas, ng/dscm (dry standard cubic metres);

V_{fg} = volume of flue gas per unit of time, dscm/hr;

M_w = mass of materials burned per unit time, kg/hr; and

0.001 = conversion factor.

3. The final step involves deriving an average based on the number of source tests conducted. This is simply performed by summing the emission factors determined as above and dividing through by the number of tests (i.e. the number of data points used). The average should represent an approximation of long-term emissions.

11.2 Congener Specific Data

The following procedure is recommended for facilities that are constructed but not operational, or are in the planning stages (USEPA, 1994b):

1. Collect and review stack test reports which have measured the emissions of CDD/CDF congeners from facilities that are similar in technology, design, operation, capacity, fuel and/or other feeds (particularly for waste feeds) and pollution control employed;
2. Determine if the stack testing has been conducted in line with the appropriate testing protocols, and discard those that have not;
3. When combining test result data from a number of facilities, care should be taken to convert emissions, process feed rates and stack parameters to consistent units of measurement; and
4. Ranges and average values should be developed for the purpose of exposure analysis.

11.3 Homologue Profile Data

A homologue group essentially encompasses a group of isomers. Thus, there are the following homologue groups:

- Tetra-CDD;
- Penta-CDD;
- Hexa-CDD;
- Hepta-CDD; and
- Octa-CDD.

Within each of these groups are a number of isomers, including the 2,3,7,8-substituted isomers. Note that the terms 'congener' and 'isomer' have the same meaning.

If no congener specific data exists for a specific facility or similar facilities, then the next best option is the use of homologue profiles from similar facilities. These emission data are described in units of μg homologue emitted/kg feed material combusted.

When only homologue emission factors are available, then rough estimates of congener specific factors can be made. The basic methodology involved assumes an equal probability

of occurrence of the specific congener based on relative proportionality. For example, 2,3,7,8-TCDD is one of 22 possible congeners in the TCDD homologue. Thus, the probability of occurrence is assumed to be the ration of 1/22 or 0.045. Multiplication of the total TCDD emission factor by 0.045 will, therefore, give an approximation of the emission rate of the 2,3,7,8-TCDD compound (USEPA, 1994b).

Table 28 below lists the number of active CDD/CDF congeners within each homologue group (i.e. the 2,3,7,8-substituted congeners), and the total number of congeners within each group.

Table 28 – The Number of Active and Total Congeners within Homologue Groups. ^a

Homologue Group	No. of Active Congeners	Total no. of Congeners (N)	1/N
Tetra-CDD	1	22	0.022
Penta-CDD	1	14	0.071
Hexa-CDD	3	10	0.1
Hepta-CDD	1	2	0.5
Octa-CDD	1	1	1
Tetra-CDF	1	38	0.026
Penta-CDF	2	28	0.036
Hexa-CDF	4	16	0.063
Hepta-CDF	2	4	0.25
Octa-CDF	1	1	1

^a Source: USEPA, 1994b.

11.4 Published Emission Factors

If no relevant data of the types discussed above can be located for a specific facility, then published emission factors should be used. This refers to the use various sources such as AP-42 (USEPA, 1995c) and the Atmospheric Emission Inventory Guidebook (The European Environment Agency, 1996). These sources typically contain averages of data compiled from various installations and may not be reflective of specific emission control equipment. The individual values used to develop the averages may vary considerably. Additionally, these sources can be quite limited as to the breakdown of various isomers. Therefore, the use of these emission factors to estimate emissions from any one facility should be done with great care.

11.5 Engineering Evaluation

In the absence of all of the data discussed above, clearly documented engineering evaluation may be required. Documentation should include copies of emission test reports used to derive emission estimates, any assumptions made and then rationale for the conclusions that were made.

An example of engineering evaluation was found in the UK study (HMIP, 1995). In that study it was assumed that emissions from glass and ceramic manufacture were the same as those from cement production. This was based on an assumption that emissions are principally related to fuel usage, and that fuel usage (per tonne product) is in the same order of magnitude as for cement production.

12 LIMITATIONS OF THE STUDY

It is clear that this study is subject to various shortcomings. The principal sources are discussed below.

- The most significant limitation is the lack of source test data for Australian sources, resulting in a heavy reliance on international data. Emission factors as a tool for estimating emissions are inherently prone to uncertainties as they are typically based on limited testing of a source population. When applying these international emissions data to Australian sources this uncertainty is increased due to potential differences in process technologies, operating conditions and practices, and pollution control equipment.
- In addition to the above point, some international emission factors (particularly in the UK study) were originally derived using assumed conversion factors to supplement data gaps. Examples of this include assumed ratios of total CDD/CDF to I-TEQ and flue gas conversion rates (i.e. m^3/tonne) to derive emission factors. This adds further to the uncertainties in the original emission factors.
- The emission factors for a large number of sources span several orders of magnitude. This is indicative of the potentially large variations that are observed within a particular source category. With such large ranges it becomes difficult to identify significant CDD/CDF contributors, particularly if the upper bound indicates that the source may be significant, while the lower bound indicates a minor contribution. Discussions in this study have focused on a worst-case approach, thereby identifying sources that are 'potentially' significant. However, as the ranges indicate, a source thought to be significant may in fact be quite minor. Better source characterisation and source test data will enable greater confidence in smaller ranges.
- Some source categories may have such variable process technologies, operational conditions etc. that it may be difficult to reliably predict emissions from these sources using limited data. Better characterisation of industry will enable the identification of these industry types. Industries where little variation is encountered could perhaps base emission factors upon more limited test data. This type of characterisation would initially be important for sources considered potentially large emitters.
- Emission factors for many industries are based on test data taken during very short sample periods. The emission results are likely to be reflective of relatively good combustion and operational practice and therefore may not be indicative of likely emissions during process upsets and/or abnormal operation.
- Several sources have been identified as having emission factors subject to uncertainties above and beyond those discussed in the points above. These are sources for which emissions data is scarce or non-existent, including glass and ceramic production, fires (prescribed and natural), and residential sources.
 1. Emission factors for glass and ceramic production are based purely on data developed for cement production (in the UK study), without any testing of actual facilities. Although these sources appear to have a very low potential for emissions, better data may still be advantageous.
 2. Emissions from residential 'wood burning stoves' are currently assumed to apply to bushfires and the prescribed burning of agricultural waste and grasslands. Due to this lack of data, and considering the highly variable conditions under which these fires can burn emissions estimated from these sources are considered highly uncertain. It

is noted in the UK study (HMIP, 1995) that attaining representative emission samples are very difficult.

3. In addition to the numerous assumptions used to derive residential emission factors (because of the lack of data), these sources are subject to increased uncertainty due to their uncontrolled, unregulated and widely varying nature of operation. Additionally the emission estimates are extremely sensitive to assumed quantities of contaminated wood burned, and no information of this nature currently exists.
- Another limitation relates to the lack of activity data for a number of sources. This lack of data prevents the compilation of a comprehensive list of sources with emission estimates. Therefore, a complete picture of the Australian situation cannot be attained. Some of the sources not included are potentially significant contributors (such as waste incineration) and so their inclusion is important.
 - An additional source that was identified as a potential CDD/CDF emitter is accidental fires. However, due to the lack of data it was not possible to derive emission estimates. This is a source that requires further consideration.

13 CONCLUSIONS

As discussed in the limitations of this study it is difficult to draw any definitive conclusions due to the large emission ranges for many industries and that fact that no activity data was available for some potentially large sources. Nonetheless, based on the sources currently included the total quantity of CDD/CDF released into the Australian atmosphere range between 150 and 2100 grams annually.

Potentially, the most significant emission sources appear to be prescribed burning (agricultural and grasslands) and bushfires. The term 'potentially' has been used as this finding is based on the upper bounds of the emission ranges. However, if the emissions actually lie towards the lower bound, while the emissions for other sources lie towards upper bounds then these sources may in fact not be the most significant.

Excluding the prescribed and natural fire emissions, the most significant anthropogenic sources are as follows (ranked from most to least significant):

1. residential wood combustion;
2. coal combustion (utility and industrial);
3. sinter production; and
4. industrial wood combustion.

These four sources contribute about 15% to the total emissions (based on upper bound estimates) and about 75% of anthropogenic sources. Combined with prescribed burning and bushfires, these sources account for approximately 95% of total emissions. However, these findings are limited by the large emission ranges in the source data.

The most significant non-industrial source is residential wood burning, ranked as the second highest anthropogenic source. Other sources such as motor vehicles however are not considered significant, contributing less than 1 percent to total emissions. Emissions from residential oil combustion are even less significant.

It is clear that the emission estimates made in this report are subject to considerable limitations. There are various reasons for this including the use of international emission factors that are inherently uncertain, combined with the fact that they are being applied to Australian sources. Additionally, the emission factors for several sources are based on limited data without actual source test information. Therefore, the emission estimates made in this report should be viewed as indicative of likely levels of CDD/CDF released from the sources considered, based on the best data currently available.

14 RECOMMENDATIONS

Based on the findings and limitations of this study, the following recommendations are made:

- Clearly the main limitation relates to the lack of Australian Source test data. It is therefore important that CDD/CDF testing be performed over a broad range of industries. The principal focus should initially be on those industries identified in this study as being 'potentially' significant. It is important that pertinent information such as feed or production rates, process technologies, operating conditions and pollution control equipment utilised be recorded with any test data. Information such as operating temperatures of flue gas control should also be included. Additionally, it is important that all source testing is performed in line with the appropriate CDD/CDF source testing protocols.
- The lack of activity data for a number of CDD/CDF sources is another major limitation of this study, particularly considering that some sources are potentially significant. It is, therefore, important that activity data on these sources be collected to at least allow a more complete indication of relative source contributions.
- Better characterisation of Australian industries will aid in developing an accurate picture of technologies and operating practices currently employed by particular industries, as well as control technologies used. This will be valuable in allowing better comparison with international data. Additionally, this type of characterisation will identify those industry categories where significant variation in process technologies etc. exist, and thus perhaps the need for more extensive testing within those categories. Even limited source testing combined with better industry characterisation will allow the development of more defined emission ranges.
- Emission sources such as prescribed and natural fires are quantified based on highly uncertain emissions data. The development of emission data for various prescribed burning operations based on the type of waste burned etc. will allow more definitive estimates to be made in this area. Additionally, attempting to derive more definitive emissions data for bushfires will be advantageous. It is recognised, however, that this may be a difficult task.
- Better characterisation of the residential combustion of wood will enable more reasonable estimates of clean and treated wood usage. Currently, emissions estimates are extremely sensitive to the assumed quantities of treated wood burned. Additionally, international emissions data for residential wood combustion are subject to significant uncertainties, particularly for combustion of treated wood. Testing of residential wood combustion equipment using various wood types will allow the development of more definitive emission factors.

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APPENDIX A ADDITIONAL EMISSIONS DATA

MUNICIPAL WASTE INCINERATORS (US Emission Data)**Table A1 – US Emission Factors for Municipal Waste Incinerators. ^a**

Combustor Type	Control Device	Emission Factor ^b (mg I-TEQ/tonne waste)
Mass Burn Waterwall	Uncontrolled	14
	ESP ^c	9.7
	Spray Dryer/ESP	5.2
	Duct Sorbent Inject./FF ^d	1.3
	Spray Dryer/FF	0.55
Mass Burn/Rotary Waterwall	Duct Sorbent Inject./FF	0.77
	Spray Dryer/FF	0.45
Mass Burn/Refractory Wall	Uncontrolled	125
	ESP	600
	Duct Sorbent Inject./ESP	3.8
Modular/Excess Air	ESP	18.3
	Duct Sorbent Inject./FF	0.5
RDF-fired	Uncontrolled	78
	ESP	142
	Spray Dryer/ESP	0.88
	Spray Dryer/FF	0.2
Modular Starved Air	Uncontrolled	25
	ESP	31.7

a USEPA, 1995b.

b All emission levels shown have been converted from total CDD/CDF emissions to I-TEQ using the assumed 60:1 ratio.

c ESP is an abbreviation for Electrostatic Precipitator

d FF is an abbreviation for Fabric Filter.

MEDICAL WASTE INCINERATORS (US Data)**Table A2 – US Emission Factors for Controlled Air Medical Waste Incinerators. ^a**

Isomer	Uncontrolled	FF ^b	WS ^c	DSI/FF ^d	DSI/CI/FF ^e	DSI/ESP ^f
TCDD 2,3,7,8	27.3	3.36	0.0645	0.281	0.411	0.0865
PeCDD 1,2,3,7,8	ND	ND	0.304	ND	ND	ND
HxCDD 1,2,3,6,7,8	0.189	ND	0.905	ND	ND	ND
1,2,3,7,8,9	0.607	ND	1.14	ND	ND	ND
1,2,3,4,7,8	ND	ND	0.461	ND	ND	ND
HpCDD 1,2,3,4,6,7,8	2.62	ND	3.47	ND	ND	ND
OCDD	11.1	ND	ND	ND	ND	ND
Total CDD	10700	1340	918	172	26.9	ND
TCDF 2,3,7,8	120	19.7	6.3	2.47	0.365	0.866
PeCDF 1,2,3,7,8	0.378	ND	0.522	ND	ND	ND
2,3,4,7,8	1.04	ND	1.53	ND	ND	ND
HxCDF 1,2,3,4,7,8	3.77	ND	4.48	ND	ND	ND
1,2,3,6,7,8	1.26	ND	1.76	ND	ND	ND
2,3,4,6,7,8	3.59	ND	4.8	ND	ND	ND
1,2,3,7,8,9	ND	ND	0.176	ND	ND	ND
HpCDF 1,2,3,4,6,7,8	8.78	ND	8.97	ND	ND	ND
1,2,3,4,7,8,9	1.36	ND	1.75	ND	ND	ND
OCDF	37.1	ND	0.245	ND	ND	ND
Total CDF	35800	4250	2460	737	47.4	ND

a Source: USEPA, 1995b. Units of all emission factors are µg/tonne waste incinerated.

b FF is an abbreviation for Fabric Filter.

c WS is an abbreviation for Wet Scrubber.

d DSI is an abbreviation for Dry Sorbent Injection. Refer to footnote b for FF.

e CI is an abbreviation for carbon injection. Definitions of FF and DSI are as per footnotes b and d.

f ESP is an abbreviation for Electrostatic Precipitator. The definition of DSI is as per footnote d.

Table A3 – US Emission Factors for Rotary Kiln Medical Incinerators. ^a

Isomer	Uncontrolled	SD/FF ^b	SD/CI/FF ^c
2,3,7,8-TCDD	0.33	0.226	0.0321
Total TCDD	3.61	2.08	0.0777
Total CDD	375	29.0	10.1
2,3,7,8-TCDF	8.37	8.42	0.248
Total TCDF	127	95.8	5.74
Total CDF	2600	396	37.8

a Source: USEPA, 1995b. Units of all emission factors are µg/tonne waste incinerated.

b SD is an abbreviation for Spray Dryer. FF is as per footnote b on Table.

c All of these abbreviations are discussed in the footnotes either in Table or footnote b to this table.

SEWAGE SLUDGE INCINERATORS (US Data)**Table A4 – US Emission Factors for Multiple-Hearth Sewage Sludge Incinerators. ^a**

Isomer ^c	Control Technology ^b					
	Uncontrolled	C/V	C/V/I	I	V/I/A	V/I
2,3,7,8-TCDD	ND	ND	0.3	0.5	0.9	2.0
Total TCDD	63	1.4	ND	28	ND	ND
Total PeCDD	2.7	ND	ND	3.7	ND	ND
Total HxCDD	68	ND	4.4	24	60	38
Total HpCDD	340	0.8	14	73	23	15
Total OCDD	370	3.4	31	53	12	19
Total CDD	850	5.6	110	180	310	270
2,3,7,8-TCDF	620	5.6	ND	180	54	46
Total TCDF	1700	50	180	700	350	600
Total PeCDF	980	11	57	360	130	1.3
Total HxCDF	99	3.4	1.8	110	78	57
Total HpCDF	480	0.9	2.9	200	48	41
Total OCDF	490	0.7	1.8	150	7.7	6.3
Total CDF	3800	66	250	1500	460	930

a USEPA, 1995b. All emission factors are expressed in units of µg/tonne of 'dry' sludge burned. ND indicates no data.

b The control technologies are all types of scrubbers (unless otherwise noted), with the letters meaning the following:

C = wet cyclone

V = venturi

I = impingement

A = afterburner (this is not a scrubber technology)

c Abbreviations for the isomers are discussed in Section 3.1.

Table A5 – US Emission Factors for Fluidised Bed Sewage Sludge Incinerators. ^a

Isomer	Uncontrolled	Venturi/Impingement
2,3,7,8-TCDD	ND	0.3
Total TCDD	ND	2.2
Total PeCDD	1.1	ND
Total HxCDD	ND	0.9
Total HpCDD	ND	0.9
Total OCDD	ND	4.3
2,3,7,8-TCDF	ND	0.2
Total TCDF	ND	6.2
Total PeCDF	ND	5.2
Total HxCDF	ND	4.1
Total HpCDF	ND	1.6
Total OCDF	ND	1.3

^a USEPA, 1995b. All emission factors are expressed in units of µg/tonne of 'dry' sludge burned. ND indicates no data.

HAZARDOUS WASTE INCINERATION (Netherlands Data)

Table A6 – Netherlands Emission Data for Hazardous Waste Incineration. ^a

Furnace Type	Emission Control	Emission factor (mg I-TEQ/tonne waste)
Rotary Kiln	quench, gas scrubber	310
Liquid Injection	quench, gas scrubber	2.2
Liquid Injection	steam boiler, quench, gas scrubber	92.7
Liquid Injection	quench, gas scrubber	2.7
Fixed Grate	afterburner, quench, gas scrubber	3.6
Rotary Kiln	ESP, gas scrubber	89
Rotary Kiln	ESP	222
Rotary Kiln	Fabric Filter	4.4
Fluidised Bed	Fabric Filter, gas scrubber	14

^a Source: Bremmer et.al., 1994.

COAL COMBUSTION – INDUSTRIAL/UTILITY (US Data)**Table A7 – US Emission Factors for Coal Combustion. ^a**

Isomer	Median Emission Factor ^b
2,3,7,8-TCDD	0.69
1,2,3,7,8-PeCDD	1.8
1,2,3,4,7,8-HxCDD	4.2
1,2,3,6,7,8-HxCDD	2.5
1,2,3,7,8,9-HxCDD	3.1
1,2,3,4,6,7,8-HpCDD	2.5
Total OCDD	25
2,3,7,8-TCDF	1.7
1,2,3,7,8-PeCDF	1.0
2,3,4,7,8-PeCDF	4.3
1,2,3,4,7,8-HxCDF	5.6
1,2,3,6,7,8-HxCDF	1.7
1,2,3,7,8,9-HxCDF	3.7
2,3,4,6,7,8-HxCDF	6.9
1,2,3,4,6,7,8-HpCDF	8.6
1,2,3,4,7,8,9-HpCDF	73
Total OCDF	7.3

a Source: USEPA, 1995b. These were actually emission factors for utility coal combustion. However, it was stated that the potential for CDD/CDF emissions from industrial boilers are expected to be similar to those from utility boilers.

b Units are pg/MJ of coal combusted (note pg = picogram = µg/1000000).

COAL COMBUSTION – INDUSTRIAL/UTILITY (UK Data)**Table A8 – UK Emissions Data for Coal Combustion.^a**

Plant Type	Plant Capacity (MW)	Emission Factor (mg I-TEQ/tonne)
Industrial/commercial	0.0175	4.8
	0.15	2.6
	0.15	2.24
	0.7	0.74
	0.7	0.87
	1	0.86
	1	0.54
	1.4	1.68
	4.6	0.2
	5	0.15
	5	0.57
	5	0.19
	5.8	1.6
	5.8	0.39
	19	0.05
	36	0.04
	43	0.07
	63	0.7
	90	0.06
Utility (power generation)	189	0.2
	380	0.32
	380	0.25
	500	0.06

^a Source: HMIP, 1995. The data presented in this table were extracted originally from two sources: (1) Sloss and Smith (1993); and (2) CRE (1994)

OIL COMBUSTION – INDUSTRIAL/UTILITY (US Data)**Table A9 – US Emission Factors for Oil-Fired Utility Boilers. ^a**

Isomer	Median Emission Factor ^b
2,3,7,8-TCDD	2.8
1,2,3,7,8-PeCDD	2.5
1,2,3,4,7,8-HxCDD	5.2
1,2,3,6,7,8-HxCDD	2.3
1,2,3,7,8,9-HxCDD	3.6
1,2,3,4,6,7,8-HpCDD	8.6
Total OCDD	9.9
2,3,7,8-TCDF	2.0
1,2,3,7,8-PeCDF	1.8
2,3,4,7,8-PeCDF	2.1
1,2,3,4,7,8-HxCDF	2.6
1,2,3,6,7,8-HxCDF	1.6
1,2,3,7,8,9-HxCDF	2.5
2,3,4,6,7,8-HxCDF	2.1
1,2,3,4,6,7,8-HpCDF	4.0
1,2,3,4,7,8,9-HpCDF	4.3
Total OCDF	4.3

a Source: USEPA, 1995b. These were actually emission factors for utility oil combustion. However, it was stated that the potential for CDD/CDF emissions from industrial boilers are expected to be similar to those from utility boilers.

b Units are pg/MJ of oil combusted (note pg = picogram = µg/1000000).

INDUSTRIAL WOOD COMBUSTION (UK Data)**Table A10 – Industrial Wood Combustion Emissions Data Reviewed for UK Study. ^a**

Wood System	Emission Factor (mg I-TEQ/tonne)	Reference
Industrial, treated wood, no PCP	9-19 5-10	Cains and Dyke (1993) Bremmer et.al. (1994)
Industrial, treated wood, PCP present	27-168 25-50	Schatowitz et.al. (1993) Bremmer et.al. (1994)
Industrial, clean wood	1-2.2 0.8-2.6 0.53-0.94	Bremmer et.al. (1994) Schatowitz et.al. (1993) LIS (1992)

a Source: HMIP, 1995

RESIDENTIAL WOOD COMBUSTION (UK Data)**Table A11 – Residential Wood Combustion Emissions Data Reviewed for UK Study.^a**

Wood System	Emission Factor (mg I-TEQ/tonne)	Reference
Wood burning stove, treated wood, no PCP	10	Bremmer et.al. (1994)
Wood burning stove, treated wood, PCP present	50	Bremmer et.al. (1994)
Wood burning stove, clean wood	1-3.3 0.77-1.25 0.7-1.2	Bremmer et.al. (1994) Schatowitz et.al. (1993) LIS (1992)
Fireplace, treated wood, no PCP	100 (estimated)	Bremmer et.al. (1994)
Fireplace, treated wood, PCP present	500 (estimated)	Bremmer et.al. (1994)
Fireplace, clean wood	13-28.5 0.7-1.2	Bremmer et.al. (1994) LIS (1992)

^a Source: HMIP, 1995**CREMATORIA (US Data)****Table A12 – US Emission Factors for Crematoria.^a**

Isomer	Average Emission Factor (ng/body)	Emission Factor Range (ng/body)
TCDD 2,3,7,8	0.0208	0.00767-0.0433
PeCDD 1,2,3,7,8	0.0653	0.0163-0.127
HxCDD 1,2,3,4,7,8 1,2,3,6,7,8 1,2,3,7,8,9	0.0785 0.113 0.142	0.022-0.18 0.0236-0.273 0.018-0.367
HpCDD 1,2,3,4,6,7,8	1.08	0.163-2.4
OCDD	1.71	0.302-2.93
Total CDD	6.67	1.37-13.9
TCDF 2,3,7,8	0.15	0.041-0.23
PeCDF 1,2,3,7,8 2,3,4,7,8	0.0912 0.261	0.034-0.193 0.05-0.5
HxCDF 1,2,3,4,7,8 1,2,3,6,7,8 1,2,3,7,8,9 2,3,4,6,7,8	0.271 0.244 0.476 0.098	0.0934-0.567 0.08-0.567 0.173-1.07 0.0367-0.213
HpCDF 1,2,3,4,6,7,8 1,2,3,4,7,8,9	1.4 0.0857	0.333-3.3 0.015-0.223
OCDF	0.458	0.173-0.767
Total CDF	10.2	3.13-24.1

^a Source: USEPA, 1995b.

NON-FERROUS METAL OPERATIONS (Netherlands Data)**Table A13 – Netherlands Emissions Data for the Secondary Non-Ferrous Industry.^a**

Type of Metal	Emission Control	Emission Factor (mg I-TEQ/tonne scrap)
Copper/bronze	lime injection/fabric filter	35
Lead	lime injection/fabric filter	5
Aluminium (strongly polluted)	lime and activated carbon injection/fabric filter	1.7
Aluminium (strongly polluted)	lime injection/fabric filter or after burner	35
Aluminium (slightly polluted)	lime injection	5
Aluminium (slightly polluted)	none	10

^a Source: Bremmer et. al., 1994

LANDFILL GAS EMISSIONS (US Data)**Table A14 – US Emission Data for Landfill Gas Flaring.^a**

Isomer	Emission Factor (pg/MJ gas flared)
2,3,7,8-TCDD	0.989
1,2,3,7,8-PeCDD	4.95
1,2,3,4,7,8-HxCDD	3.96
1,2,3,6,7,8-HxCDD	3.96
1,2,3,7,8,9-HxCDD	13.9
1,2,3,4,6,7,8-HpCDD	40.6
Total OCDD	237
2,3,7,8-TCDF	757
1,2,3,7,8-PeCDF	20.7
2,3,4,7,8-PeCDF	61.1
1,2,3,4,7,8-HxCDF	78.3
1,2,3,6,7,8-HxCDF	22.7
1,2,3,7,8,9-HxCDF	5.93
2,3,4,6,7,8-HxCDF	36.6
1,2,3,4,6,7,8-HpCDF	65.4
1,2,3,4,7,8,9-HpCDF	3.95
Total OCDF	34.4

^a Source: USEPA, 1995b. Note that a 'pg' is 10^{-12} g (or $\mu\text{g}/1000000$).

COMPARATIVE CEMENT AND LIME EMISSIONS (UK Study)**Table A15 – Comparative Data for Cement and Lime Emissions. ^a**

	Coal	Lignite	Oil	Gas
Cement	12-59	118	1.9	4.6
Lime	13-63	125	2.0	4.9

^a Source: HMIP, 1995. Data originally presented in Fiedler, 1994. Emissions have units of ng I-TEQ/tonne of product.

MOBILE SOURCES**Table A16 – Emissions Data Reviewed for the UK Study. ^a**

Study	Country	Fuel type ^b	Catalyst	Emission factor ^c (pg I-TEQ/VKT)
Marklund et.al., 1987	Sweden	Unleaded (2) Leaded (4)	Yes No	<13 20-220
Bingham et. al., 1989	New Zealand	Unleaded (1) Leaded (4)	NR NR	1 5-39
Marklund et. al., 1990	Sweden	Unleaded (2) Leaded (2) Unleaded (1) Leaded (2) Diesel (1)	No No Yes No NR	0.36-0.39 ^d 2.6-6.3 ^d 0.36 1.1-2.6 not detected ^d
Hagenmaier et. al., 1990	Germany	Unleaded (1) Unleaded (1) Leaded (1) Diesel (1)	No Yes No NR	5.1 ^e 0.7 ^e 108 ^e 50 ^{e,f}
Oehme et. al., 1991	Norway	NA (- ^g)	NA	520 (Car uphill) ^h 38 (Car downhill) ^h car average = 280 ^h 9500 (truck uphill) ^h 720 (truck downhill) ^h truck average = 5100 ^h

^a Source: HMIP, 1995. Data originally presented in USEPA, 1994.

^b Values in parentheses indicate the number of vehicles tested.

^c Test samples were taken in the tailpipe, unless otherwise indicated.

^d Test samples were taken before the muffler.

^e The original results were presented in Hagenmaier as pg I-TEQ/litre of fuel. These factors were subsequently changed in the USEPA (1994) study using fuel economy factors published by Marklund et. al. (1990). These factors were 10 km/L and 2 km/L for petrol and diesel vehicles respectively.

^f It was noted in the UK study that an error was made in the conversion of this factor in the US study. The UK study then determined a factor of 1.5 pg/km. However, based on the original emission factor and the quoted fuel consumption, an emission factor of 12 pg/km is derived in this study.

^g Tests were conducted over portions of four days with traffic rates of between 8000-14000 vehicles per day. Heavy-duty vehicles comprised approximately 4-15 percent of the total.

^h Emission factors reported in units of pg Nordic TEQ/km.

APPENDIX B SUPPORTING CALCULATIONS

LANDFILL GAS GENERATION

The National Greenhouse Gas Inventory Workbook (NGGIW) for waste (NGGIC, 1996c) states that the following equation should be used for estimating landfill methane gas generation:

$$Q_{ml} = 0.82 \times 10^{-3} (L)(Q_{lf})$$

Where:

Q_{ml} is the quantity of methane emitted (ktonne/yr);
 L is the methane potential of landfilled waste (L/kg) = 79 L/kg for Australian landfills; and
 Q_{lf} is the average quantity of waste landfilled in the 25 year period prior to the inventory year (ktonne/year).

It was estimated that for 1994 approximately 11000 ktonnes of waste was landfilled. Using the above equation, it is estimated that 713 ktonnes of methane was generated. The approximate composition of landfill gas is 50 percent methane and 50 percent carbon dioxide (USEPA, 1995c).

It was assumed that the gas is generated under approximately ambient conditions (i.e. 25°C and 1 atm). Therefore, using the ideal gas law it is estimated that about 1100 million m³ of methane was generated. Using the landfill gas composition approximately 1100 million m³ of CO₂ was also generated, and so the total volume of gas generated is 2200 million m³.

Based on discussions with Mr Charles Jubb (Aquatech Pty Ltd) who is responsible for the NGGIW for waste it was assumed that 7 percent of the gas generated is combusted in gas engines, while about 0.5 percent was flared. These values are based on the mass of gas produced.

Based on the above calculations, and using 44 g/mol as the molecular weight of CO₂, the mass generation rate of CO₂ is about 2000 ktonnes. Therefore, the total quantity of landfill gas generated is approximately 2700 ktonnes. This equates to about 190 ktonnes of gas that is combusted in gas engines and 1.35 ktonnes that is flared.

Using the data above the following volumes are determined:

- uncontrolled raw gas: 2040 m³/yr;
- combusted in gas engine: 150 m³/yr; and
- combusted in flare: 11 m³/yr.

CERAMIC ACTIVITY DATA

ABS publication No. 8365.0 (Principal Commodities Produced 1993/94) presented the following information:

- 1891 million non-refractory bricks (size 230 mm x 110 mm x 76 mm);
- 120 million non-structural bricks;
- 218 million structural bricks; and
- 250000 tonnes of refractory bricks, blocks and tiles.

Using the above dimensions supplied for non-refractory brick and the density of brick at about 1800 kg/m^3 , the weight of a single brick (conservatively assumed to be solid) is 3 kg. Therefore, to be conservative it will be assumed that all bricks weigh about 4 kg, and so the total quantity of bricks (excluding refractory) is about 9000 ktonnes.

Based on the information available an upper bound of 10000 ktonnes was used for this study.

GLASS ACTIVITY DATA

ABS publication No. 8365.0 (Principal Commodities Produced 1993/94) presented the following information:

- thin sheets: 44000 tonnes (\$193 million);
- safety glass (tempered/laminated): 6814000 m^2 (\$234 million); and
- others (\$845 million).

It has been conservatively assumed that the safety glass is approximately $\frac{1}{2}$ inch thick, and so the total volume of this glass produced is about 82000 m^3 . Using the density of glass at 2600 kg/m^3 the total mass of safety glass produced is approximately 213 ktonnes.

Using the cost data it is determined that the cost per unit of glass lies between \$1million and \$4million per ktonne. To be conservative it is assumed that the glass for which no data is given has a unit cost of about \$2million per ktonne. Using this it is estimated that the quantity of glass from this is about 420 ktonnes.

Based on the above calculations and data an upper bound of 700 ktonnes is derived for glass production.