

4 AIR EMISSIONS CONTROLS

When using any WTE technology to treat MSW, some emissions to air are produced. In conventional combustion, the emissions to air are the result of the actual combustion of MSW. In gasification or pyrolysis, the emissions to air are associated with the combustion of the syngas or pyrolysis products to produce usable energy.

Over the years, vast technological improvements have been made which have assisted in greatly reducing the quantity and toxicity of emissions being released into the atmosphere. Generally speaking, these emissions controls can be grouped into two main categories:

- Operational controls, which act to increase the efficiency of the WTE process leading to lesser production of harmful emissions
- Air Pollution Control (APC) systems, which are usually placed on the back end of a WTE facility and act to capture/treat emissions before they are released.

The following two subsections discuss these operational controls and air pollution control systems. Both of these controls are primarily discussed as they relate to mass burn incineration (conventional combustion) facilities as this is the most common form of WTE technology being used worldwide to treat MSW. Some information regarding operational and APC systems for gasification is also provided, however, much less information is available in comparison to that available for mass burn incinerators as there are very few gasification facilities in operation worldwide that treat MSW in comparison to hundreds of mass burn incinerators. As mentioned previously in this report, gasification is less commercially proven than mass burn incineration in the treatment of MSW.

Little information is available regarding the emissions controls applicable for other WTE technologies.

4.1 Operational Controls

There are a number of operational controls^[83] used in modern WTE facilities that act to increase system performance and efficiency and by doing so, assist in reducing the formation of unwanted byproducts and pollutants. Operational controls act to reduce emissions (to air and water) and also assist in improving the quality of ash produced by a WTE facility. These operational controls are in addition to conventional “back end” air pollution controls that will be discussed further in Section 4.2.

Many of these operational controls have been developed over time as the understanding of WTE processes has increased. This understanding has allowed engineers to fine-tune the waste treatment process to prevent or reduce the creation of unwanted byproducts during waste treatment rather than having to remove these byproducts at the back end of a facility using air pollution control equipment. As mentioned earlier, these operational controls have also helped to increase the performance and efficiency of waste treatment technologies. Better operational controls allow for

⁸³ Much of this material adapted from A.J. Chandler and Associates Ltd. 2006. Review of Dioxins and Furans from Incineration in Support of a Canada-wide Standard Review

more environmentally and economically friendly operation of WTE facilities, and are one of the reasons why such WTE approaches are more broadly accepted in jurisdictions such as the EU.

The operational controls currently being used in modern mass burn incinerators (conventional combustion) and gasification facilities are discussed below.

4.1.1 Operational Controls for Mass Burn Incineration (Conventional Combustion)

In mass burn incinerators, operational controls have been developed to reduce the formation and release of unwanted byproducts (such as NO_x, dioxins/furans, and CO) during the incineration of MSW. Modern mass burn incinerators are designed with highly complex operational controls that ensure the safe and efficient combustion of waste with the accompanying capture of energy.

The operational/combustion controls used in mass burn incinerators compensate for the compositional variability of MSW and act to control the rate of combustion reactions.

The composition of MSW is highly variable and depends on a number of uncontrollable factors such as the general behavior of residents, use of available waste diversion programs and the demographics of the community the WTE facility serves.

The variable composition of MSW affects operational efficiency because each component of the waste stream has its own particular energy content which must be matched with a particular amount of oxygen to ensure proper and efficient combustion of the waste stream. For example, if a large amount of paper is being placed in the refuse stream, this will increase the overall energy content of the material and affect its behavior as a fuel source. In order to ensure that proper combustion conditions are met, the MSW stream must be made as homogenous as possible before and during incineration.

One way to increase the homogeneity of MSW is to ensure that the waste material is well mixed prior to being combusted. This can be accomplished by mixing waste with the grapple crane prior to placing the waste material into the hopper. Even after proper mixing, however, MSW heat values are still quite variable.

This variability is accounted for within the furnace by operational controls. Mass burn incinerators monitor the heat being released from the waste at all times and are able to adjust air flow (oxygen) to compensate for changes in waste composition. Modern facilities also compensate by adjusting the waste fuel feed rate. For example, if too little heat is being produced, more waste can be fed to the incinerator to ensure enough energy is present in the combustion zone. Conversely, if waste with higher energy content enters the furnace, the feed rate can be reduced.

Combustion control is very important to reduce the creation of harmful byproducts (such as CO, TOC and NO_x) as much as possible. Many intermediate steps are involved in the oxidation of long chain hydrocarbons in the combustion gas to products of complete combustion (carbon dioxide and water). By ensuring complete combustion, the creation of unwanted byproducts is minimized and the amount of energy captured from the waste is maximized.

Generally speaking, proper combustion conditions that discourage the generation of unwanted byproducts are those that:

- Ensure that there is complete mixing of the fuel and the air
- Maintain high temperatures in the presence of an adequate amount of oxygen
- Have proper mixing or agitation to prevent the formation of quench zones or low temperature pathways that would allow partially-reacted solids or gases to exit from the combustion chamber.

It is particularly important to prevent the generation of soot in the system because carbon present in the fly ash will lead to increased formation of dioxins and furans. The formation of soot is reduced by following the proper operational controls as discussed above.

The furnace of a typical modern mass burn incineration facility used in the North American market is designed to provide at least a one second retention time at a temperature of approximately 1,000°C in the combustion zone (after the last point of air injection) while processing waste. This has generally been accepted in North American regulations/guidelines as an appropriate requirement. Maintaining 1,000°C for one second in the combustion zone has been recognized by the EU as a condition that can result in internal corrosion, in part as it may cause the fly ash present in the flue gas to melt. The requirements established in the EU are for a minimum two second retention time at 850°C. Both of these temperatures, in combination with the respective retention time, are high enough to ensure the complete destruction of organic substances present in the waste. Even during waste feeding and non-emergency shutdowns, the temperature in the combustion zone is not allowed to fall below 850 – 1,000°C.^[84] Auxiliary burners are used to maintain temperature and residence time in the furnace.

There is merit in considering application of the approach applied in the EU within the BC guideline. At issue is the combustion 'zone' in which the flue gas must be held at or above the required temperature. Generally, this is defined as the last point of air injection (i.e., the over-fire air provided to ensure complete combustion). Depending on the design of the WTE facility, maintaining 1,000°C for one second after this point of air injection may have undesirable consequences. Molten particles within the flue gas can cause fouling and/or corrosion of the heat transfer surfaces for the boiler. Design of the combustion chamber and boiler must address the need to cool the flue gas to approximately 650°C before it reaches the heat transfer surfaces of the boiler. Therefore, some flexibility in specifying the combination of temperature and residence time is necessary to take into account incinerator-specific operational factors.

Several new technologies have been developed to reduce the production of NO_x during combustion by re-circulating part of the flue gas (FGR). These technologies are often applied in Europe. One such technology is Covanta's very low NO_x (VLN™) system. This technology was developed by Martin GmbH in cooperation with partner companies such as Covanta and is described in more detail

⁸⁴ Durham/York Residual Waste Study Environmental Assessment, November 27, 2009, Stantec Consulting Ltd

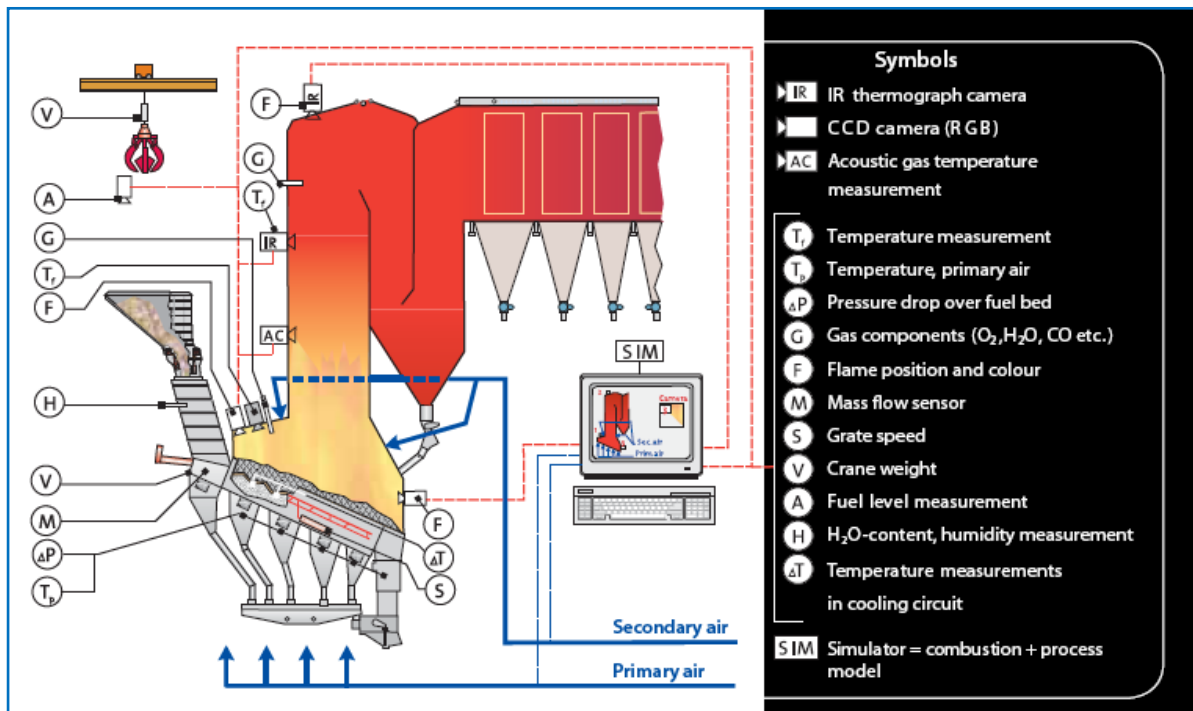
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below.^[85] Another NO_x reduction system has been developed by VonRoll/Wheelabrator, called the VLNR (very low NO_x reduction) system. The system is based on injection of ammonia/urea at various levels. The injection of ammonia/urea is strictly controlled in order to ensure reaction at the most optimal time. Other vendors are using the same principle where it is possible to inject ammonia/urea at different levels depending on the optimum temperature but have not promoted their systems under specific trade names.

Figure 4-1 provides a schematic overview of the furnace operational controls typical for a modern mass burn WTE facility.^[86]

Figure 4-1: Control Components of a Modern Furnace Control System



Source: Babcock and Wilcox Volund. 2009. 21st Century Advanced Concept for Waste-Fired Power Plants: A Solution to Asia's Mounting Waste Problems

The following list identifies a number of the advantages associated with the use of proper operational controls during the waste incineration process.^[87]

- Better bottom ash quality (due to sufficient primary air distribution and a better positioning of the incineration process on the grate)
- Less fly ash production (due to less variation in the amount of primary incineration air)

⁸⁵ Martin GmbH für Umwelt- und Energietechnik: http://www.martingmbh.de/index_en.php?level=2&CatID=6.79&inhalt_id=66, 2010

⁸⁶ Babcock and Wilcox Volund. 2009. 21st Century Advanced Concept for Waste-Fired Power Plants: A Solution to Asia's Mounting Waste Problems.

⁸⁷ Ministry of Housing, Spatial Planning and the Environment. 2002. Dutch Notes on BAT for the Incineration of Waste.

- Better fly ash quality (less unburned material, due to more stable process conditions in the furnace)
- Less CO and hydrocarbon formation (due to more stable process conditions in the furnace, i.e., no cold spots)
- Less (risk of) formation of dioxin (-precursors) (due to a more stable process in the furnace)
- Better utilization of the plant capacity (because the loss of thermal capacity by variations is reduced)
- Better energy efficiency (because the average amount of incineration air is reduced)
- Better boiler operation (because the temperature is more stable, there are less temperature 'peaks' and thus less risk of corrosion and clogging fly ash formations)
- Better operation of the flue gas treatment system (because the amount and the composition of the flue gas is more stable)
- Less maintenance and better plant availability.

The following subsection provides further details for one example of operational NO_x control that can be applied in North America.

Operational NO_x Control: Example Covanta VLN™

The Covanta VLN™ process utilizes a unique combustion air system design, combined with an advanced combustion monitoring and control system, to achieve substantial reduction in NO_x formation. The VLN™ process, in addition to the conventional primary and secondary air systems, features an internal recirculation gas (IRG) injection system located in the upper furnace. IRG is an internal stream drawn from the rear of the combustor, above the burnout zone of the grate. The distribution of flows between the primary air, secondary air and IRG gas streams is controlled to yield the optimal combustion gas composition and temperature profile to minimize NO_x and control combustion. The control methodology takes into account the heating value of the waste and the fouling condition of the furnace.

Figure 4-2 presents a schematic overview of the Covanta VLN™ Process.

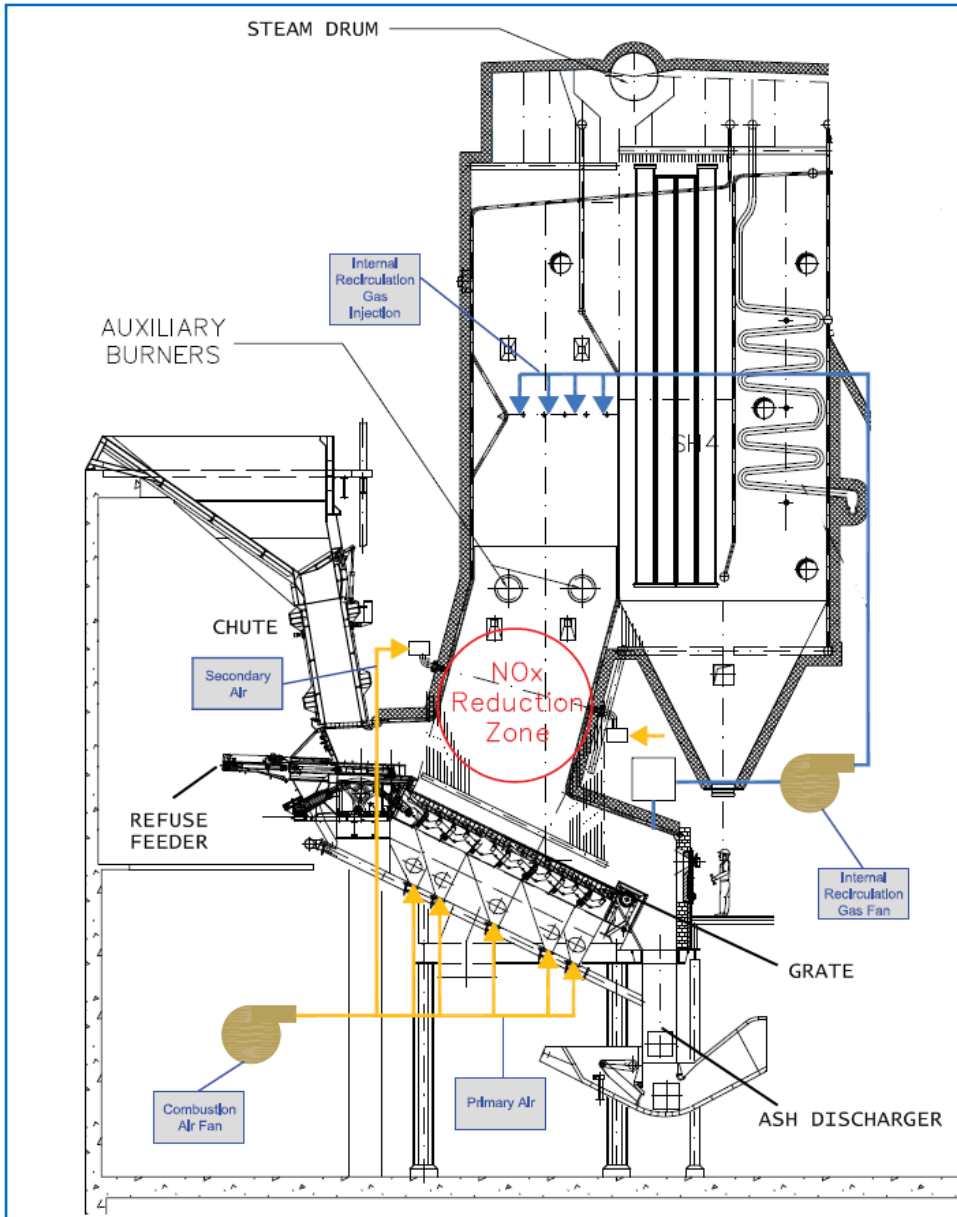
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Figure 4-2: Conceptual Schematic Diagram of Covanta VLN™ Process



Source: Durham/York Residual Waste Study Environmental Assessment, November 27, 2009, Stantec Consulting Ltd

4.1.2 Operational Controls for Gasification Systems

As mentioned previously in this report, technologies that gasify MSW are much less proven than conventional combustion technologies. For that reason, information describing the operational controls used by gasification technologies is quite sparse compared to the operational controls used by mass burn incinerators. Further, the operational controls used by a gasification facility will depend

on the specific gasification technology being considered. For instance, the operational controls used in the Nippon Steel gasification process discussed below are different from those used in the Thermoselect process because there are some fundamental differences between the technologies.

The following paragraphs describe the operational controls used by gasification facilities utilizing the Nippon Steel “Direct Melting System” technology. The Japanese Nippon Steel technology is discussed here as it is one of the more commercially proven MSW gasification technologies, as noted in Section 2.2.1.2. As of 2009, Nippon Steel had 28 operational plants in Japan and one in Korea, which together process more than 1.9 million tonnes of MSW, sewage sludge and other residues per year.^[88]

The Nippon Steel “Direct Melting System” operates as follows.^[89] MSW is fed into the top of the furnace (by a crane) with the required amounts of coke and limestone. The waste is charged into the melting furnace when the signal from the burden level meter (installed in the furnace) indicates that the burden level has dropped to the specified level. At the base of the melting furnace, molten materials are discharged into a water granulator and are then separated into slag and metal. The syngas produced is transferred to a combustion chamber. The heat is recovered from the gas via a hot-water generator and then the flue gas is treated by APC equipment before it is released from the stack.

The following list illustrates the digital control systems utilized by the Nippon Steel technology:

- The waste, coke and limestone feed rates and the molten residue generation rate are all measured and recorded to ensure proper feed rates.
- The pressure and temperature in the melting furnace and combustion chamber and the flow rate of air supplied to the melting furnace and combustion chamber are all continually monitored to ensure efficient operation.
- The composition of syngas leaving the melting furnace (CO, CO₂, O₂, CH₄, H₂) and supplied to the combustion chamber, and the composition of the waste gas leaving the combustion chamber (CO₂, O₂, CO, NO_x) are also continuously monitored.

All this data is sent into a distributed control computer and used for real-time analysis of material balance and to ensure the plant is operating at optimal efficiency. Figure 4-3 illustrates the instrumentation system used in one of Nippon Steel’s demonstration plants.^[90] It should be noted that the APC train depicted in the figure is from one of Nippon Steel’s older facilities. Their newer facilities tend to include a bag filter and NO_x reduction system.

⁸⁸ University of California, Riverside. 2009. Evaluation of Emissions from Thermal Conversion Technologies Processing Municipal Solid Waste and Biomass

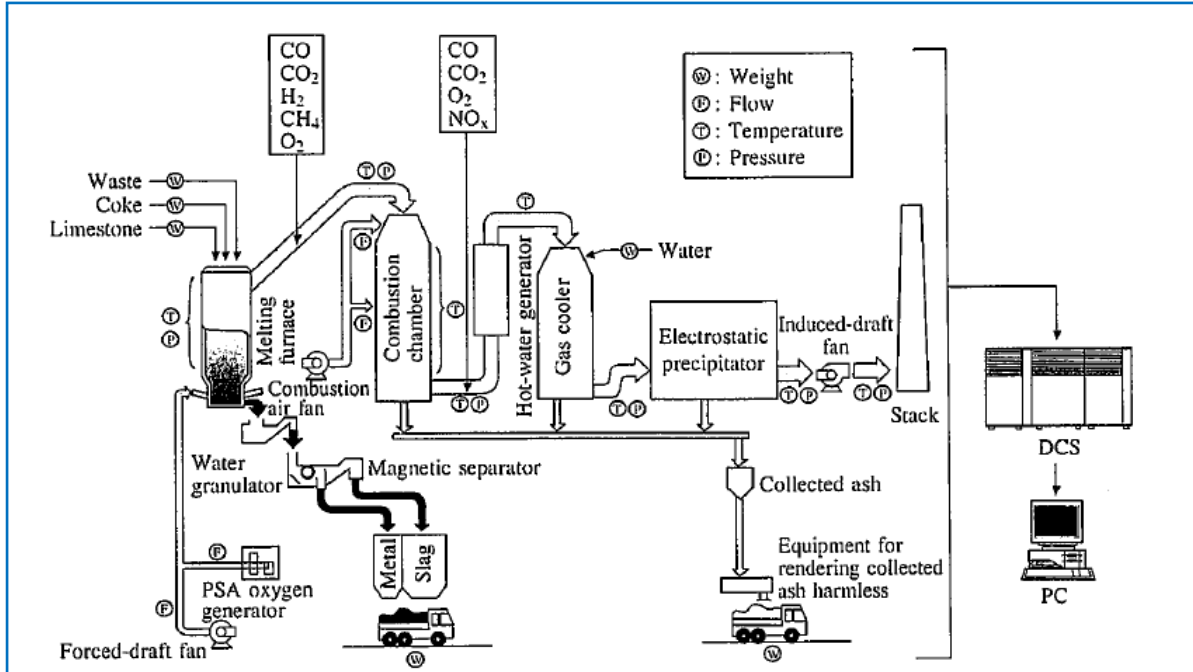
⁸⁹ It should be noted that all Nippon Steel facilities utilize the DMS technology.

⁹⁰ Nippon Steel Technical Report No. 70. July 1996. Research and Development of Direct Melting Process for Municipal Solid Waste

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Figure 4-3: Conceptual Diagram of Operational Controls Used by Nippon Steel



Source: Nippon Steel Technical Report No. 70. July 1996. Research and Development of Direct Melting Process for Municipal Solid Waste

4.2 Air Pollution Control Systems

WTE facilities convert municipal solid waste into gaseous, liquid and solid conversion products with a simultaneous or subsequent release of heat energy which is captured. Air emissions released from WTE facilities generally arise from the compounds present in the waste stream, and are formed as a normal part of the combustion process.

In order to reduce the environmental impacts associated with WTE facilities air pollution control (APC) systems have been developed. In general, APC systems are used to cool flue gases, scrub acidic gases and capture particulate matter and various contaminants such as heavy metals and trace organics.

Significant improvements have been made in APC systems of WTE incinerators over the past few decades and advancements continue to be made to the types of APC systems used for both MSW and Hazardous Waste incinerators.^[91]

Up to the mid-1960s, waste incineration flue gas treatment was relatively simple. A common method was to cool the flue gas down to a temperature of 250 – 300°C by injecting water (evaporative cooling) and the flue gas was passed through a cyclone to remove fly ash. In the late 1970s and

⁹¹ A.J. Chandler & Associates Ltd.

1980s, semi-dry and wet flue gas treatment systems were developed, followed by systems to address NO_x and dioxins (mainly based on activated carbon) in the late 1980s and 1990s. These systems included the introduction of bag filters for dust removal.^[92]

There are a large number of air pollution control technologies that are currently used by WTE facilities worldwide to control the release of harmful pollutants to the atmosphere. Most of these controls are post-combustion controls, or controls added to the back-end of an incinerator to remove the unwanted byproducts of incineration. The sub-sections below provide an overview of the most common air pollution control technologies and how they act to limit the release of pollutants.

These sub-sections generally describe the primary elements of a conventional APC system, followed by identification of some of the more common APC trains.

4.2.1 Primary Air Pollution Control System Components

This section provides an overview of the primary components that would be included in the APC train for a WTE facility. Further discussion in Section 4.2.2 describes factors and aspects considered to select and combine these various components together within APC trains.

4.2.1.1 Activated Carbon Adsorption (Mercury, Dioxin/Furan Control)

Activated Carbon is used in an APC system to control the release of trace organics (including dioxins/furans) and mercury into the atmosphere. Activated carbon achieves this by adsorbing these chemicals onto its surface.

There are two main types of activated carbon adsorption systems, namely powdered activated carbon (PAC) injection and carbon bed filters (known commercially as activated char reactors (ACR) or absorbers). By far, the most common type used in WTE facilities is PAC injection (six of the seven facilities currently operating in Canada use this form of carbon adsorption).^[93] The following paragraphs present an overview of the two types of systems.

PAC injection systems are used at the back end of WTE facilities as the first step in flue gas treatment following incineration. This is the method that is being considered in the proposed Durham/York incinerator project and is currently used at the Algonquin Power incinerator in the Peel Region.^{[94],[95]} PAC injection systems operate in the following way. Powdered activated carbon is injected into the flue gas prior to a fabric filter baghouse (this will be discussed later) and dioxins/furans and volatilized mercury are adsorbed onto the carbon particles. The particles of activated carbon with adsorbed organic molecules are then captured in the fabric filter baghouse where it forms a cake on the filter's surface allowing for additional adsorption as well as filtering. While PAC injection systems have lower removal efficiency as compared with fixed activated carbon

⁹² Ministry of Housing, Spatial Planning and the Environment. 2002. Dutch Notes on BAT For the Incineration of Waste

⁹³ GENIVAR Ontario Inc. in association with Ramboll Danmark A/S, 2007. *Municipal Solid Waste Thermal Treatment in Canada*

⁹⁴ <http://www.peelregion.ca/pw/waste/facilities/algonquin-power.htm#apc>

⁹⁵ Durham/York Residual Waste Study Environmental Assessment, November 27, 2009, Stantec Consulting Ltd

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bed filter systems, the injection of powdered activated carbon into the gas stream is a far less expensive method of removing dioxin/furans and mercury from the flue gas.

Fixed activated carbon bed filters can be installed at the end of the APC system to clean gases and remove trace organics and mercury before the flue gas is released into the atmosphere. The carbon bed filter consists of a vertical chamber with a bed depth of typically between 0.5 to 1 m. Carbon bed filters allow for a maximum flue gas velocity of 0.2 m/s and require an operating temperature of between 120 and 150°C to avoid condensation.

Carbon bed systems have the highest known removal efficiency for dioxins and furans (>99.9%) as well as for many other pollutants and are commonly used in Europe, Asia and Australia. The major disadvantage of the carbon filter system is the capital investment and operating costs associated with these systems, as well as the need for proper disposal of spent carbon adsorbent. In Europe, the spent carbon adsorbent is usually combusted in the incinerator. Both PAC injection systems and carbon bed systems can be used to achieve regulatory compliance in the jurisdictions studied within this report, and can achieve compliance with the most stringent of the regulatory limits. As a result, the decision to use either system may often be based on cost, as part of a reasonable BACT assessment process.

4.2.1.2 Fabric Filter Baghouses (Particulate Matter Control)

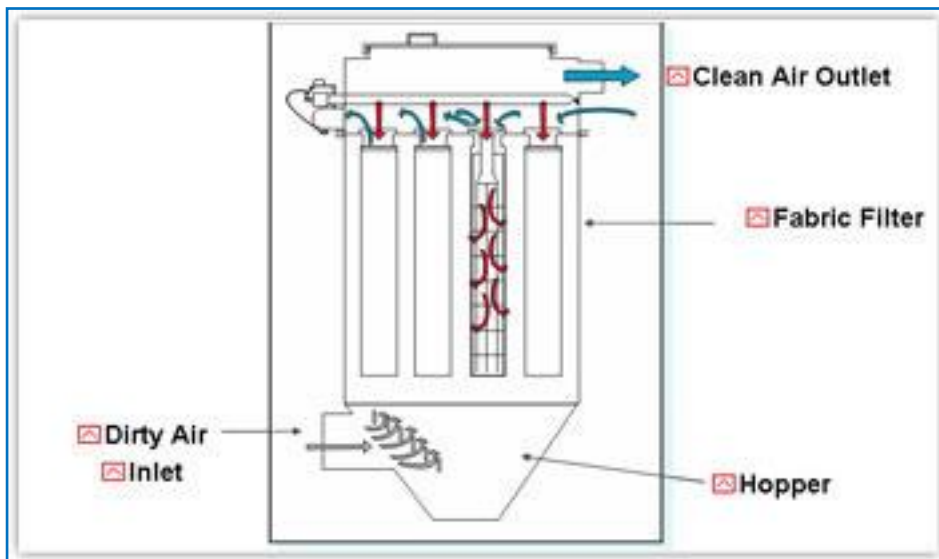
Fabric filter baghouses are used to remove particulate matter from the WTE flue gas before it is released into the atmosphere. Their operation is fairly simple: as flue gases pass through a tightly woven fabric, particulate matter collects on the fabric, preventing it from being released into the atmosphere. The “dust cake” which forms on the surface of the filter due to the collection of particulate matter also helps (up to a point) to increase the filtering efficiency by creating an increased barrier to air movement.

Baghouses are classified based on the method used to clean them. There are two main types of baghouse systems: reverse air baghouses and pulse-jet fabric filters. In a reverse-air baghouse, the flue gas flows upward through the insides of vertical bags which open downward. Fly ash from the flue gas collects on the insides of the bags, and the flow of gas keeps the bags inflated. To clean the bags, a compartment of the baghouse is taken off-line, and the gas flow in this compartment reversed. This causes the bags to collapse, and collected dust to fall from the bags into hoppers. The cleaning cycle in a reverse-air baghouse typically lasts about three minutes per compartment. Because reverse-air cleaning is gentle, reverse-air baghouses typically require a low air-to-cloth ratio.

In a pulse-jet fabric filter, the dirty flue gas air flows from the outside of the bags inward, and the bags are mounted on cages to keep them from collapsing. Dust which collects on the outsides of the bags is removed by a reverse pulse of high-pressure air. This cleaning does not require isolation of the bags from the flue gas flow, and thus may be done on-line. Because pulse-jet cleaning is harsh, the bags remain relatively clean, so that a higher air-to-cloth ratio (i.e., a smaller baghouse as

compared to reverse-air) may be used. Figure 4-4 illustrates a schematic overview of a pulse-jet type fabric filter baghouse.^[96]

Figure 4-4: Schematic Overview of a Pulse-Jet Fabric Filter Baghouse



Source: http://www.neundorfer.com/FileUploads/RichTextboxImages/Image/pulse_jet_baghause.jpg

Baghouse performance is determined by a variety of factors including the fabric chosen, the cleaning frequency and methods, and the particulate characteristics. A number of different fabrics can be used in baghouses. Fabrics can be chosen which will intercept a greater fraction of the particulate. Some fabrics are coated with a membrane with very fine openings for enhanced removal of submicron particulate. However, often these highly efficient fabrics are much more expensive than more conventional materials.

It is important to realize that the particles are not only caught by interception. Electrostatic forces and Brownian movements also play a role, especially for particles that seem to be too small to be caught by the fabric (or the accumulated dust cake). Consequently, baghouse filters have their lowest collection efficiency at a particle size around 0.3 μm . Both smaller (i.e., nanoparticles) and larger particles are more effectively removed.

Baghouses are often capable of 99.9% removal efficiencies. Removal efficiency is relatively level across the particle size range (except at around 0.3 μm), so that excellent control of PM_{10} and $\text{PM}_{2.5}$ can be obtained.^[97]

⁹⁶ http://www.neundorfer.com/FileUploads/RichTextboxImages/Image/pulse_jet_baghause.jpg

⁹⁷ <http://www.icac.com/i4a/pages/index.cfm?pageid=3398>

4.2.1.3 Electrostatic Precipitators (ESP) (Particulate Matter Control)

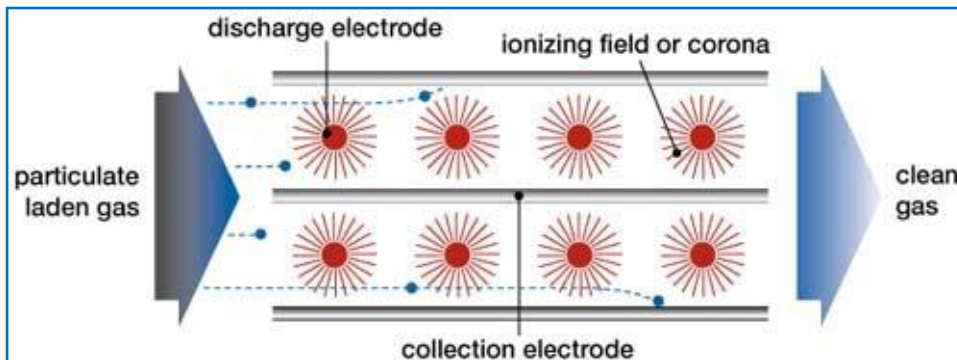
Electrostatic precipitators (ESPs) use electrical fields to remove particulate matter from flue gas. ESPs have been in common use for a long time. Typically, ESPs have low energy requirements and operating costs.

Basically, there are three steps that an ESP uses to remove particulate matter from the flue gas.^[98]

- Particle charging
- Particle collection
- Removal of collected particulate.

In an ESP, an intense electric field is maintained between high-voltage discharge electrodes and grounded collecting electrodes or plates. A corona discharge^[99] from the discharge electrodes ionizes the gas passing through the precipitator, and gas ions subsequently ionize other particles. The electric field drives the negatively charged particles to the collecting electrodes. Periodically, the collecting electrodes are rapped mechanically to dislodge collected particulate, which falls into hoppers for removal. Figure 4-5 provides a conceptual overview of an ESP.^[100]

Figure 4-5: Conceptual Overview of an ESP



Source: <http://web.njit.edu/~avs9/Procedure%20Draft%20Final.htm>

Most precipitators have three to five independent electrical sections in series (referred to as sectionalization). Each independent section removes a fraction of the particulate in the gas stream; this arrangement allows the use of higher voltages in the first sections of the precipitator, where there is more particulate to be removed. Lower voltages must be used in the final, cleaner precipitator sections to avoid excessive sparking between the discharge and collecting electrodes.

⁹⁸ <http://hamon-researchcottrell.com/HRCTechnicalLibrary/Reviving%20an%20Electrostatic%20Precipitator.pdf>

⁹⁹ A corona discharge is an electrical discharge brought on by the ionization of a fluid surrounding a conductor, which occurs when the potential gradient (the strength of the electric field) exceeds a certain value, but conditions are insufficient to cause complete electrical breakdown or arcing.

¹⁰⁰ <http://web.njit.edu/~avs9/Procedure%20Draft%20Final.htm>

Precipitator sectionalization also allows for the collection of particles re-entrained (due to rapping) in the flue gas stream to be collected in downstream sections of the precipitator.

There are several factors which contribute to ESP performance and particle removal efficiency including precipitator size, flow uniformity, re-entrainment, and particle resistivity.

- **Precipitator Size** – determines particle treatment time. The larger (longer) the precipitator the higher chance a particle will be charged and collected.
- **Flow Uniformity** – flow non-uniformity and re-entrainment are factors that decrease precipitator performance. Uniform gas flow ensures that there is no high gas velocity resulting in short treatment time paths through the precipitator. Attaining flow uniformity also minimizes "short circuiting," or gas flows bypassing the electrical fields.
- **Re-entrainment** – re-entrainment of collected particles can occur during rapping (particle collection). The amount of re-entrainment can be reduced through proper rapper design and timing and maintenance of hopper ash levels and flow uniformity.
- **Resistivity** – resistivity is the resistance of particles to the flow of electric current. Particles with resistivity in the range of $10^7 - 10^{10}$ ohm-cm lend themselves to collection by ESPs as they are easy to charge and only lose their charge once they are deposited on the collection electrode. Particles with low resistivity (less than 10^7 ohm-cm) are more difficult to collect using ESP. These particles lose their charge to a collecting electrode so rapidly that they tend not to adhere to the electrode. This results in a high rate of particle loss. High-resistivity particles form ash layers which adhere very strongly to the collecting electrodes which again may lead to injection of positively charged ions into the space between the discharge and collecting electrodes ("back corona"), thus reducing the charge on particles in this space and lowering collection efficiency.

ESPs are capable of removal efficiencies of up to 99.9% with common efficiencies of 99.5%. Precipitators with high overall collection efficiencies will have high collection efficiencies for particles of all sizes, so that excellent control of PM_{10} and $PM_{2.5}$ will be achieved with well designed and operated electrostatic precipitators. In practice, 97 – 98% of all particulate matter under 5 μm in diameter are removed by ESPs.^[101] That said, similar to baghouse filters, ESPs also have their lowest collection efficiency at a particle size of around 0.3 μm due to electrostatic forces and Brownian movements.

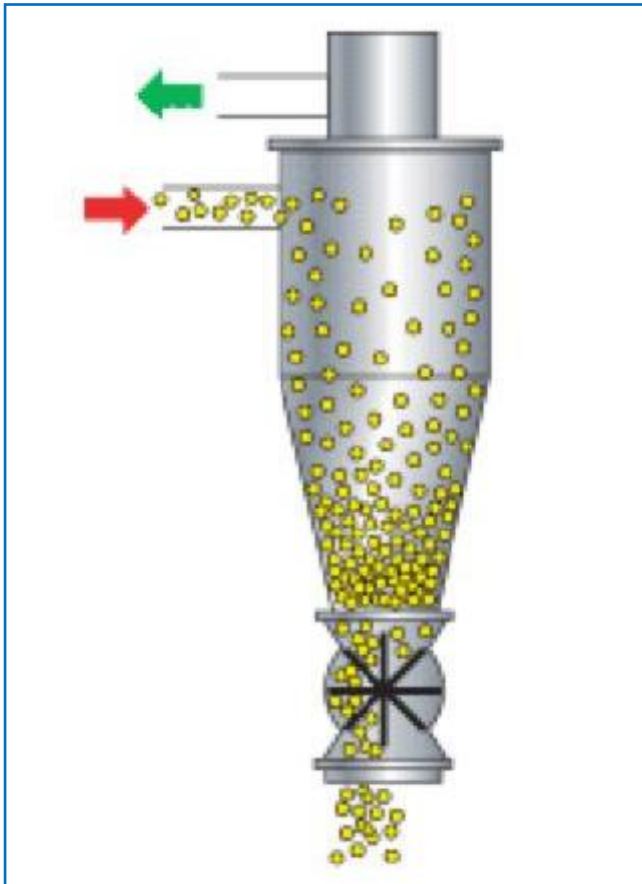
4.2.1.4 Mechanical Collectors (Particulate Matter Control)

Mechanical collectors use mechanical means to remove particulate matter from the flue gas. One of the most common forms of mechanical collection is the cyclone separator. A cyclone separator is a vertical tank with the bottom end tapered into a pipeline and a section of the top open. Using centrifugal force, the cyclone separates larger particles from smaller ones. The efficiency of cyclone separators

¹⁰¹ <http://www.icac.com/i4a/pages/index.cfm?pageid=3398>

depends largely on flue gas flow vis-à-vis the cyclone diameter: as velocity of the flue gas increases, so does the collection efficiency. Figure 4-6 illustrates the operation of a cyclone separator.^[102]

Figure 4-6: Schematic Overview of Cyclone Separator



Source: http://www.fmdaircontrol.com/cyclone_separators.jpg

Often cyclone separators are combined to increase removal efficiency and are referred to as multiple cyclones. A multiple cyclone consists of an array of cyclones in parallel. Overall, multiple cyclones have removal efficiencies of 70 – 90%. Removal efficiency is largely dependent on particle size; as particle size decreases, removal efficiencies met by the cyclone drop off quickly. The removal of finer particulates such as $PM_{2.5}$ is quite limited. Typically, cyclone removal efficiencies are approximately 90% for particles greater than PM_{10} , 70% for $PM_{2.5}$ and 50% for PM_1 .^[103]

Although multiple cyclones have no moving parts, they require regular cleaning and preventative maintenance to ensure that collection efficiency is maintained.

¹⁰² http://www.fmdaircontrol.com/cyclone_separators.jpg

¹⁰³ <http://www.icac.com/i4a/pages/index.cfm?pageid=3398>

In general, cyclones on their own cannot achieve the emission levels now applied to modern waste incinerators. They can, however, have an important role to play where applied as a pre-duster before other flue gas treatment stages.^[104]

4.2.1.5 Acid Gas Scrubbers (Multi-Pollutant Control)

“Scrubber” is a general term that describes APC devices that use both physical and chemical absorption to remove pollutants from the flue gas stream. Scrubbers, which are generally classified as either “wet” or “dry/semi-dry”, rely on a chemical reaction with a sorbent to remove acidic gases including sulfur dioxide (SO₂), hydrochloric acid (HCl) and hydrofluoric acid (HF) from the flue gas stream. In addition to acidic gases, scrubbers are also capable of removing particulate matter and heavy metals such as mercury.

According to the EPA and others, both wet and dry scrubbers have been shown to reduce HCl emissions by 95% and more, and wet scrubbers have been shown to reduce HF emissions by more than one-third. Others have reported ranges of 87 – 94% removal of HCl and 43 – 97% removal of HF by both wet and dry scrubbers.^[105] The following subsections discuss wet and dry scrubbers separately in more detail.

Wet Scrubbers

In a wet scrubber, the flue gas stream is brought into contact with a scrubbing liquid or sorbent. This is accomplished by various methods including spraying the flue gas with sorbent, forcing it through a pool of liquid or by some other method. The gaseous or particulate pollutants present in the flue gas stream come into direct contact with sorbent and are dissolved or diffused (scrubbed) into the liquid.

The sorbent is typically some kind of alkaline slurry of limestone which reacts with the acidic gases to form neutralized byproducts (i.e., SO₂ reacts to form calcium sulfite or calcium sulfate). The wet solid byproduct which is formed typically requires further treatment (dewatering, precipitation of heavy metals) before it is released from the facility. The dewatered, purified sludge can then be disposed of via other conventional methods and the treated wastewater can leave the site.

Alternatively, instead of using an alkaline sorbent, water can be used as the sorbent in the wet scrubber. When water is used, it mixes with the acidic compounds and increases the PH (reduces the acidity). Water is equally as effective as an alkaline sorbent at capturing particulate matter. A wet scrubber utilizing water requires an additional step which takes the watered solution and treats it with alkaline substances (i.e., limestone) to lower its acidity. This system also results in wastewater which must be treated to remove heavy metals resulting in sludge and a wastewater stream which leaves the site.

Generally, wet scrubbers have relatively small space requirements and require relatively little capital investment (although they tend to be more expensive than dry or semi-dry systems). Wet scrubbers are able to process high temperature, high acidity, and high humidity flue gas streams. Scrubber

¹⁰⁴ European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration

¹⁰⁵ <http://www.icac.com/i4a/pages/index.cfm?pageid=3398>

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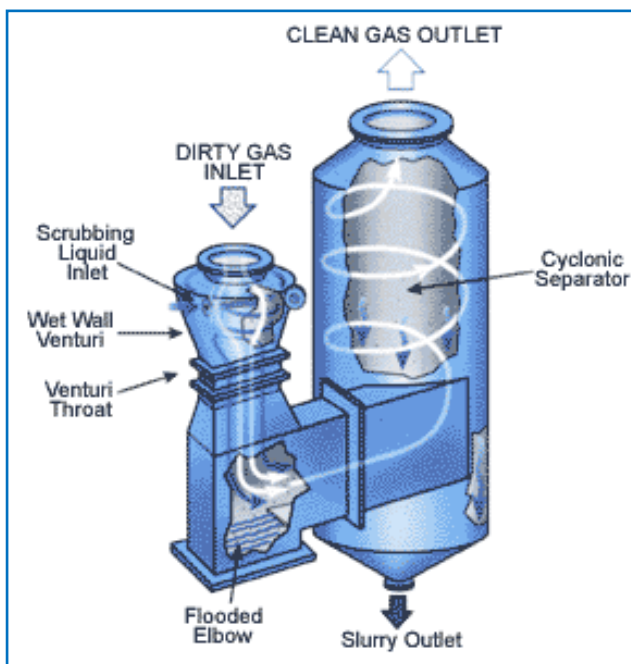
energy requirements have also continued to decrease, helping to lower operating costs. Design and operating parameters relating to the operating efficiency of wet scrubbers include the shape of the scrubber, liquid spray or injection locations, gas residence time, gas velocities, gas and liquid temperatures, gas and liquid pressure drop, and, the liquid/gas flow rate ratio.

Wet scrubbers can achieve high levels of multi-pollutant control, including the control of acidic gases, SO_2 , fine particulates and heavy metals (e.g., cadmium). New wet scrubbers achieve SO_2 removal efficiencies of 95%, with some scrubbers achieving removal efficiencies of up to 99%. In addition, wet scrubbers also provide for significant removal of arsenic, beryllium, cadmium, chromium, lead, manganese, and mercury parameters from flue gas.

As mentioned previously, wet scrubbers are capable of removing a large proportion of particulate matter from the flue gas stream. Venturi scrubbers (a kind of wet scrubber) are commonly used to remove particulate matter from flue gas.

In a venturi scrubber, the “scrubbing” liquid and flue gas are accelerated through a duct which narrows to a small opening and then opens back up. As they reach the small opening, the flue gas and scrubbing liquid are moving at very high velocities which cause the scrubbing liquid to break apart into very fine droplets. These very fine droplets each provide a surface on which particulate matter can be absorbed. Venturi scrubbers are often combined with cyclone separators to remove the water droplets from the flue gas stream. Venturi scrubbers are often capable of removing greater than 90% of particles with diameters above 10 microns. The efficiency of removal for smaller particles is much lower. Figure 4-7 presents a schematic overview of a venturi scrubber.

Figure 4-7: Schematic Overview of a Venturi Scrubber



Source: Mikropul. 2009. Wet Scrubbers

Dry/Semi-Dry Scrubbers

In a dry or semi-dry scrubber, particles of an alkaline sorbent are injected into the flue gas, producing a dry solid by-product. In some cases moisture is added to the sorbent prior to injection (semi-dry scrubber) and in some cases no moisture is added to the sorbent (dry scrubber). If no water is added to the sorbent (dry scrubber), a wet spray humidifier is often placed prior to the dry scrubber in the APC train to humidify the flue gas and to aid in the dry scrubber's operation. Due to the lower levels of moisture used in the dry scrubber, there is no wastewater produced in the dry or semi-dry systems.

Dry/semi-dry scrubber systems can be grouped into three categories: spray dryers (semi-dry), circulating spray dryers (semi-dry), and dry injection systems (dry). All three of these systems offer multi-pollutant control opportunities by combining acid gas, SO₂, particulate, and heavy metal control.

In a spray dryer, alkaline reagent slurry (typically lime based) is atomized into the hot flue gas to absorb pollutants. The resulting dry material, including fly ash, is collected in a downstream particulate control device such as a fabric filter baghouse. Spray dryers commonly are designed for SO₂ removal efficiencies of 70 – 95%.^[106]

A circulating dry scrubber uses an entrained fluidized bed reactor for contacting the reagent, usually hydrated lime, with acid gas and particulate laden flue gas. The fluidized bed promotes an intensive gas-solid mixing that encourages the reaction of acidic gases in the flue gas with the dry lime particles. Similar to spray dryers the mixture of reaction products, unreacted lime, and fly ash is carried to a downstream particulate collector such as a fabric filter baghouse. In a circulating dry scrubber, water spray is introduced into the fluidized bed separately. This enhances the performance of the system by optimizing the surface moisture content of the lime which allows for lesser amounts of lime to be used by the system. Circulating dry scrubbers can provide removal efficiencies of more than 90%.^[107]

¹⁰⁶ <http://www.icac.com/i4a/pages/index.cfm?pageid=3398>

¹⁰⁷ <http://www.icac.com/i4a/pages/index.cfm?pageid=3398>

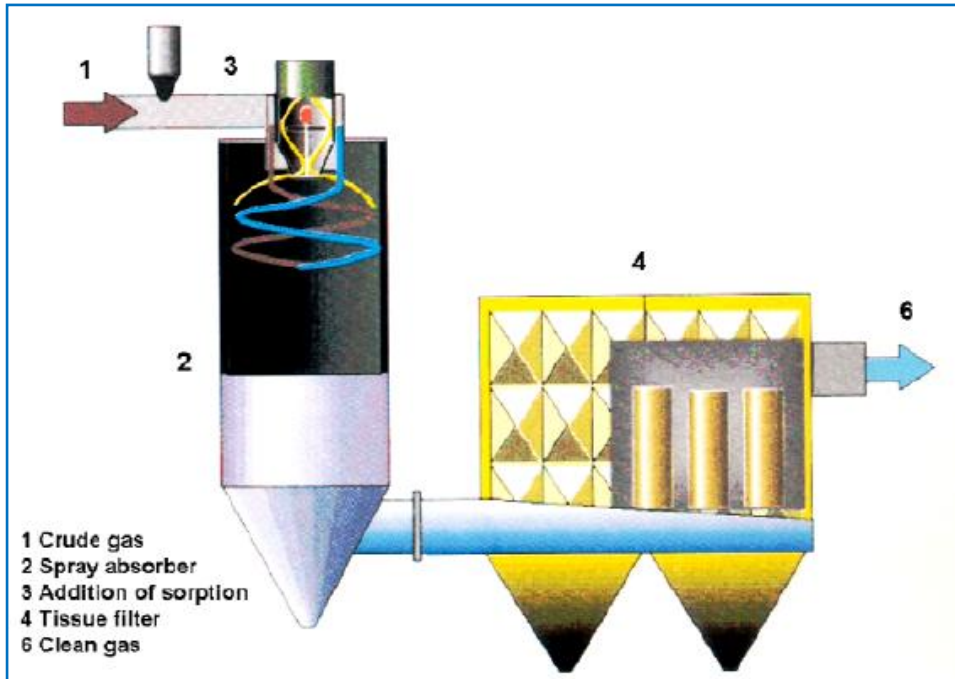
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Figure 4-8: Overview of a Circulating Dry Scrubber



Source: European Commission. 2006. *Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration*

Generally, dry/semi-dry scrubbers are simple and have low capital and maintenance costs associated with them. Dry scrubber energy requirements, while less than wet scrubber systems, continue to decrease which helps to lower operating costs.

4.2.1.6 Nitrogen Oxide Control

Selective Catalytic Reduction (SCR) (Nitrogen Oxide Control)

The basic principle of SCR is the reduction of NO_x to N_2 and H_2O . This is accomplished by injecting ammonia (NH_3) into the flue gas stream which then reacts with NO_x gases within a catalyst bed.

The basic operation of an SCR system is quite simple. It consists of a reactor chamber with a catalyst bed, and an ammonia handling and injection system, with the ammonia injected into the flue gas upstream of the catalyst. The system involves no moving parts and other than spent catalyst, the SCR process produces no waste products.

Several different catalysts are available for use at different exhaust gas temperatures. In use the longest and most common are base metal catalysts, which typically contain titanium and vanadium oxides, and which also may contain molybdenum, tungsten, and other elements. Due to the catalyst the reaction can take place at a lower temperature normally around 250°C , however, references to a temperature interval between $180 - 350^\circ\text{C}$ are available.

In principle, SCR can provide reductions in NO_x emissions approaching 100%. In practice, commercial SCR systems have met control targets of over 90% in many cases.

SCR (as opposed to SNCR) is typically used if there is a strict regulatory limit or if a financial incentive to reduce the NO_x emission is introduced. Normally SCR processes achieve emission levels of between 20 – 70 mg/Nm₃.

The SCR process is typically located at the downstream (tail-end) portion of the APC plant where SO₂ and SO₃ levels in the flue gas are reduced to prevent precipitation of ammonia hydrogen sulphate. It also prolongs the lifetime of the catalyst when Hg, HCl and dust are removed.

A disadvantage with the tail-end SCR process is that the flue gas temperature within this portion of the APC train is lower than required, normally around 140 – 150°C, and reheating of the flue gas is necessary. Reheating is normally done through a combination of a heat exchanger where the ingoing flue gas to the SCR-process is preheated by means of the flue gas leaving the SCR and additional heating by approximately 25°C which can be done by the usage of steam from the boiler or the turbine or by means of natural gas/oil.

For the tail-end SCR process, where the concentration of other pollutants in the flue has already been reduced, reactions can be carried out at a lower temperature without incurring too high a risk for precipitation. Some plants have tested SCR at temperatures from 180 – 220°C but the experience is so far not sufficient and the majority of the suppliers still recommend operating temperatures of around 250°C.

Alternatively the SCR process can be placed before the flue gas treatment plant. To have a sufficiently high temperature without needing reheating of the flue gas, the most advantageous placement is at the outlet of the boiler and before the economizer. Due to the high sulphur content the temperature in the SCR process has to be approximately 280°C to prevent precipitation. The experience with high dust catalysts is very limited, and the few plants with high dust SCR system have experienced problems.

If considerations are taken during the design of the catalyst, SCR can absorb dioxin as well. The adsorption of dioxin is dependent on the chemical composition of the catalyst as well as the size of the catalyst.

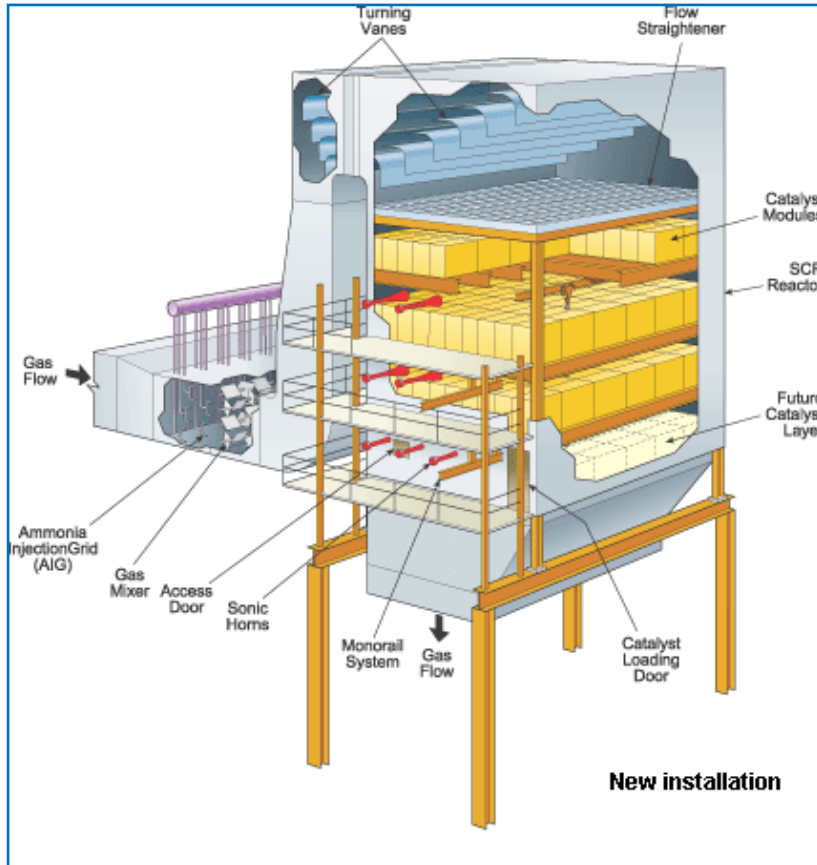
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Figure 4-9: Diagram of a Selective Catalytic Reducer



Source: The Babcock and Wilcox Company. 2010. Environmental Equipment: Selective Catalytic Reduction (SCR) System. http://www.babcock.com/products/environmental_equipment/scr.html

Selective Non-Catalytic Reduction (SNCR) (Nitrogen Oxide Control)

Similar to SCR, SNCR is a chemical process that converts NO_x into N_2 and H_2O using ammonia (NH_3). At suitably high temperatures ($870 - 1,150^\circ\text{C}$), the desired chemical reactions occur.

The operation of an SNCR system is quite simple. Ammonia (or urea) is injected/sprayed into and mixed with the hot flue gas. The ammonia or urea then reacts with the NO_x in the flue gas stream, converting it into nitrogen and water vapour. The main difference from SCR is that SNCR does not utilize a catalyst. SNCR is "selective" in that the reagent reacts primarily with NO_x , and not with oxygen or other major components of the flue gas.

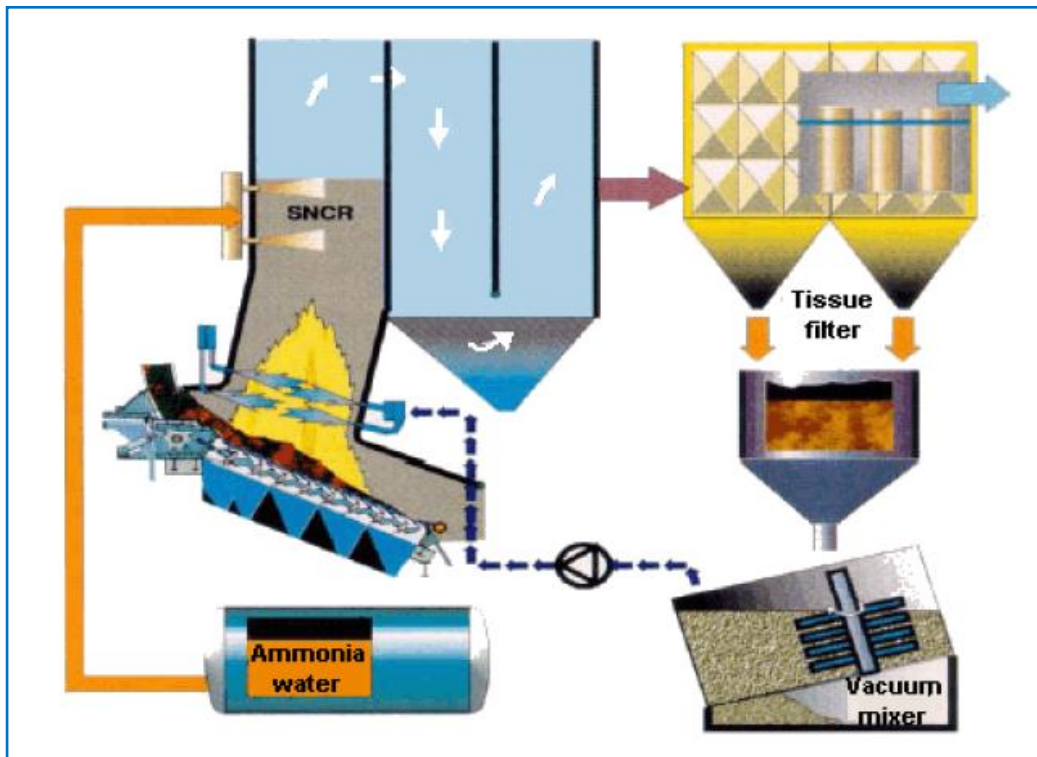
The principal components of an SNCR system are the reagent storage and injection systems, which includes tanks, pumps, injectors, and associated controls, and often NO_x continuous emissions monitors (CEMs). Given the simplicity of these components, installation of SNCR is easy relative to the installation of other NO_x control technologies. SNCR retrofits typically do not require extended source shutdowns.

While SNCR performance is specific to each unique application, NO_x reduction levels ranging from 30% to over 75% have been reported. Temperature, residence time, reagent injection rate, reagent-flue gas mixing, and uncontrolled NO_x levels are important in determining the effectiveness of SNCR.

Emission values around 150 mg/Nm^3 are common for the SNCR process. Lower values – to around 100 mg/Nm^3 – are possible with the SNCR process but the consumption of ammonia is relative high and the risk for ammonia slip will increase.

The ammonia slip is normally limited to $5 - 10 \text{ mg/Nm}^3$ as ammonia may result in a light odour of the flue gas residues.

Figure 4-10: Overview of SNCR System



Source: European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on Best Available Techniques for Waste Incineration

4.2.1.7 Conditioning Towers or Wet Spray Humidifiers

Some WTE facilities utilize a conditioning tower or wet spray humidifier as part of their APC equipment. A conditioning tower consists of a vertical tower where water is sprayed into the gas stream, humidifying the gas stream while decreasing the temperature to about $160 - 185^\circ\text{C}$.

With current APC design, conditioning towers are often used to cool the flue gases prior to the inlet of the baghouse filter at the end of the APC train, in order to protect the baghouse filters and to ensure the optimal temperature range for chemical reactions with lime.

Conditioning towers/humidifiers can be used to reduce gas temperature and elevate humidity to allow for a more effective operation of other downstream APC equipment such as dry acid gas scrubbers. Conditioning towers can also decrease the potential for dioxin and furan formation by dropping flue gas temperatures rapidly below the temperature range for de novo synthesis.

4.2.2 APC System Design and Operation

The individual components of an APC system are combined into APC trains to provide an effective overall system for the treatment of pollutants that are found in the flue gases. There are several common APC trains currently used at operating WTE facilities, to control the release of unwanted pollutants into the atmosphere. The selection of an air pollution control train for a WTE facility depends on a number of factors, such as the desired emissions reductions necessary to meet applicable regulations, the ability of various APC components to function with one another (not all APC equipment is compatible) and the cost of the equipment (capital and operating).

Generally speaking, when choosing an APC train for a WTE facility the first thing considered is how to control the release of acid gases such as SO₂, HCl and HF. After an appropriate control for acid gases is chosen, compatible and appropriate components can be selected for the control of particulate matter, dioxins, mercury and NO_x. In other words, the selection of the APC component to treat acid gases forms the backbone of the APC train and affects the type and placement of other APC controls that manage the release of other chemicals of concern.

There are three main types of treatment systems that treat acidic compounds, and thus three main types of APC trains that are built around the acid gas control measures:

- Dry/semi-dry systems
- Wet systems
- Semi-wet systems (combination of dry/semi-dry and wet systems).

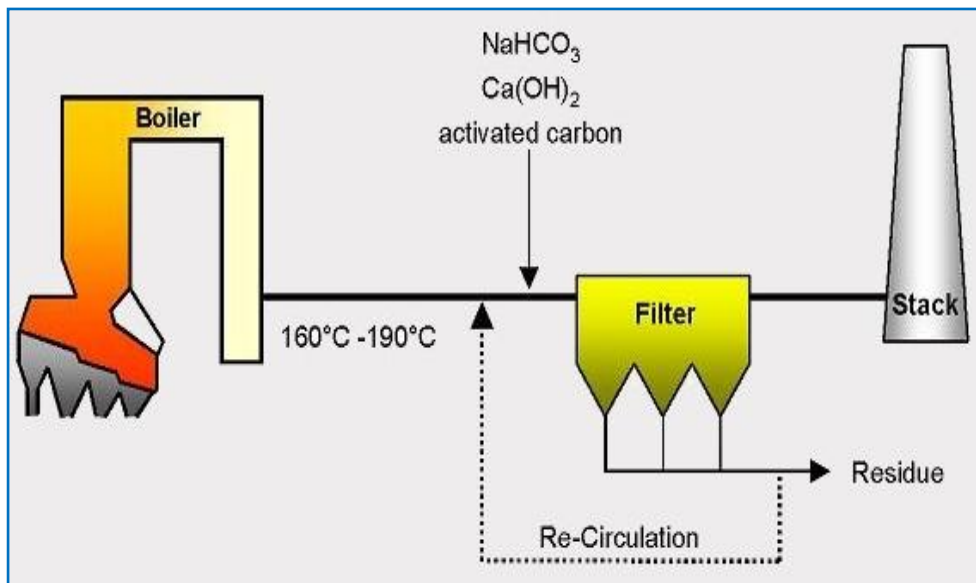
The most common form of APC system currently used by WTE facilities in Canada is the dry/semi-dry system.^[108] The following sections provide an overview of each of these systems.

¹⁰⁸ GENIVAR Ontario Inc. in association with Ramboll Danmark A/S, 2007. Municipal Solid Waste Thermal Treatment in Canada

4.2.2.1 Dry/Semi-Dry Systems

As discussed previously, dry/semi-dry systems for acid gas control^[109] can be grouped into three categories: spray dryers (semi-dry)^[110] circulating spray dryers (semi-dry), and dry injection systems (dry), but the basic operation of each system is similar. In each system, the acidic compounds in the flue gas react in a vessel with a sorption agent (normally calcium hydroxide ($\text{Ca}(\text{OH})_2$) for the dry system and lime milk (a suspension of calcium hydroxide) in the semi-dry system. Alternatively dry sodium bicarbonate (NaHCO_3) can be used as the sorption agent. In dry systems, wet spray humidifiers are often added to the front of the APC train to assist in the operation. Figure 4-11 presents a simple schematic overview of a Dry/Semi-Dry APC system.

Figure 4-11: Schematic Overview of a Dry/Semi-Dry APC System



Source: Fisica Babcock Environment GmbH. 2007. Wet Scrubbing. Accessed March 15, 2010 from <http://www.fisia-babcock.com/index.php?id=183>

The injected sorption agent reacts with the acidic compounds converting them into solid compounds ($\text{HCl} \rightarrow \text{CaCl}_2$, $\text{HF} \rightarrow \text{CaF}_2$, $\text{SO}_2 \rightarrow \text{CaSO}_3$ or CaSO_4). The solid by-products formed are removed later on in the APC train in a fabric filter baghouse or other dust collecting device such as an ESP. By this process, the majority of the acidic compounds present in the flue gas are neutralized and prevented from being released into the atmosphere.

In addition to the adsorption of acidic compounds the dry/semi-dry system also assists in the reduction of other harmful pollutants including particulate matter and heavy metals.

¹⁰⁹ Ramboll

¹¹⁰ Spray dryers followed by fabric filters have become the norm for WTE facilities in the United States (Air Pollution Control For Waste to Energy Plants – What Do We Do Now?, 1997)

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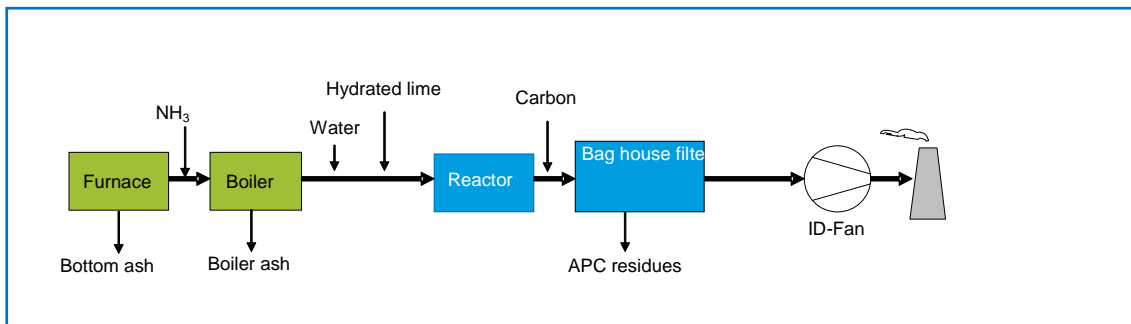
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In the dry/semi-dry system, other APC components can then be added to the APC train to assist in the reduction of dioxin/furans, mercury and NO_x emissions. Normally, an activated carbon injection system is added after the acidic gas reactor to adsorb both mercury and dioxins which are then captured in the fabric filter baghouse preventing them from being released into the atmosphere. The last step would be adding a SCR or SNCR APC component, respectively to reduce NO_x emissions.

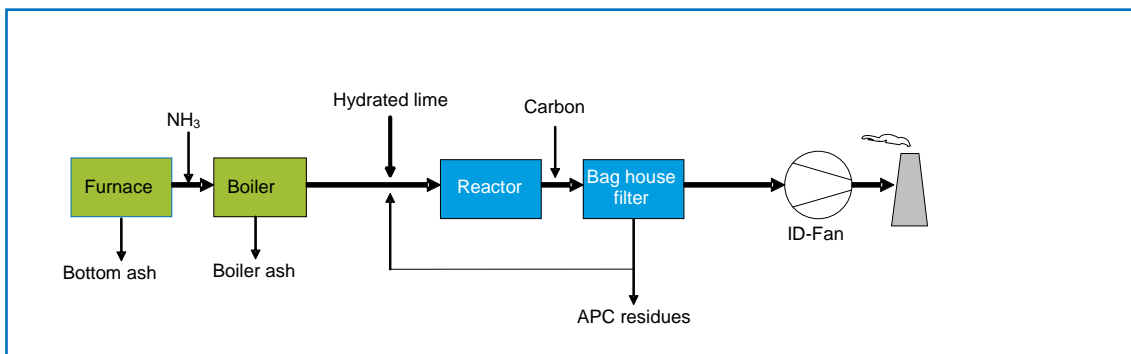
Figures 4-12 to 4-14, below provide an overview of three types of common dry/semi dry APC trains and the combination of key APC components compatible with dry/semi dry acid gas control.

Figure 4-12: Dry APC System



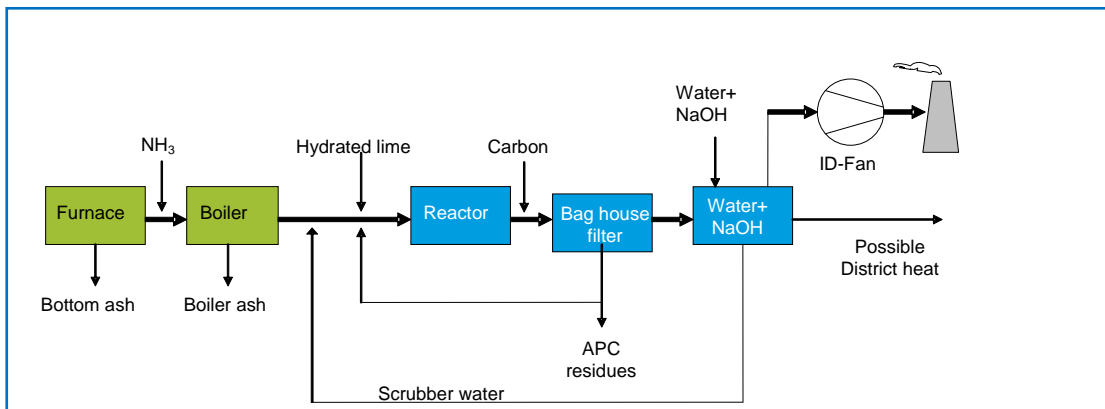
This system includes SNCR for NO_x control, a dry scrubber, use of activated carbon injection to control dioxins/furans and mercury, and a bag house to control particulate and the majority of heavy metals.

Figure 4-13: Semi-Dry System, Example 1



This system includes SNCR for NO_x control, a dry scrubber with recirculation of recovered water from APC residue treatment for humidification, use of activated carbon injection to control dioxins/furans and mercury, and a bag house to control particulate and the majority of heavy metals.

Figure 4-14: Semi-Dry System, Example 2

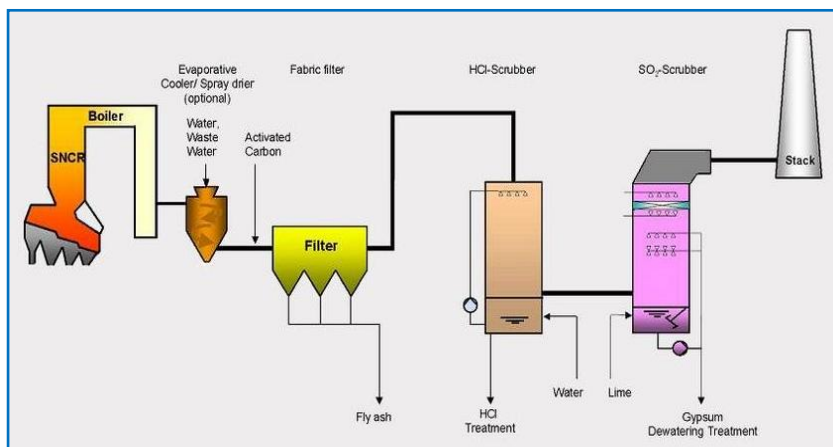


4.2.2.2 Wet Systems

Of all three flue gas treatment methods, the wet system is the only one which generates wastewater to be discharged and based upon our research is the least common type of APC train used in WTE facilities in North America. That said, the wet system is often used in Europe where additional incentives are in place to reduce emissions to air and as up until recently, wet systems were typically able to reduce emissions to a greater degree than dry/semi-dry systems.

Wet systems can be grouped into numerous different categories based upon their geometric shape and method for gas-liquid interaction including packed-bed, counter-flow, cross-flow, bubble-plate, open spray (single and double loop) tower, dual-flow tray, cyclonic, etc. Generally speaking, however, they all function in a similar manner. Figure 4-15 provides a general schematic overview of a wet APC system.

Figure 4-15: Schematic Overview of a Wet APC System



Source: Fisia Babcock Environment GmbH. 2007. Wet Scrubbing. Accessed March 15, 2010 from <http://www.fisia-babcock.com/index.php?id=183>

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The first stage in a wet system is normally the removal of dust and particulate matter from the flue gas with either an ESP or fabric filter baghouse prior to the wet scrubber. This filtration helps to remove some particulate matter and reduce the concentration of heavy metals in the flue gas. In the next treatment stage in the wet system, the acidic compounds present in the flue gas are washed with water in an “acid scrubber” which produces a wastewater stream. Washing the flue gas with water removes the majority of HCl as it becomes a diluted hydrochloric acid solution. The liquid effluent from the water washing is then passed on to a wastewater treatment stage (to neutralize the acid and to remove heavy metals which may still be present in high concentrations).

The flue gas moves on to an “alkaline” scrubber, in which it is washed with a solution of either sodium hydroxide or a suspension of limestone which removes the majority of SO₂ from the flue gas. The waste liquid remaining after the alkaline scrubber is also sent to wastewater treatment prior to being released from the facility.

After both scrubbing stages, the flue gases are then treated with activated carbon injection to remove the remaining dioxins/furans and mercury. The activated carbon with adsorbed material is then captured in a downstream fabric filter baghouse.

The wastewater from the acid and alkaline scrubbers is normally neutralized to approximately pH 9 by CaCO₃ and NaOH. The heavy metals and other solids present in the wastewater are then precipitated out by the addition of chemicals such as CaCl₂, NaOH, FeCl₃ and TMT 15. The precipitates are dewatered in a filter press before proper disposal while the treated wastewater is discharged from the facility. Similar to dry/semi-dry systems, wet systems also assists in the reduction of other harmful pollutants including particulate matter and heavy metals. Figures 4-16 to 4-18, below, provide examples of typical wet APC systems.

Figure 4-16: Wet APC System, (a)

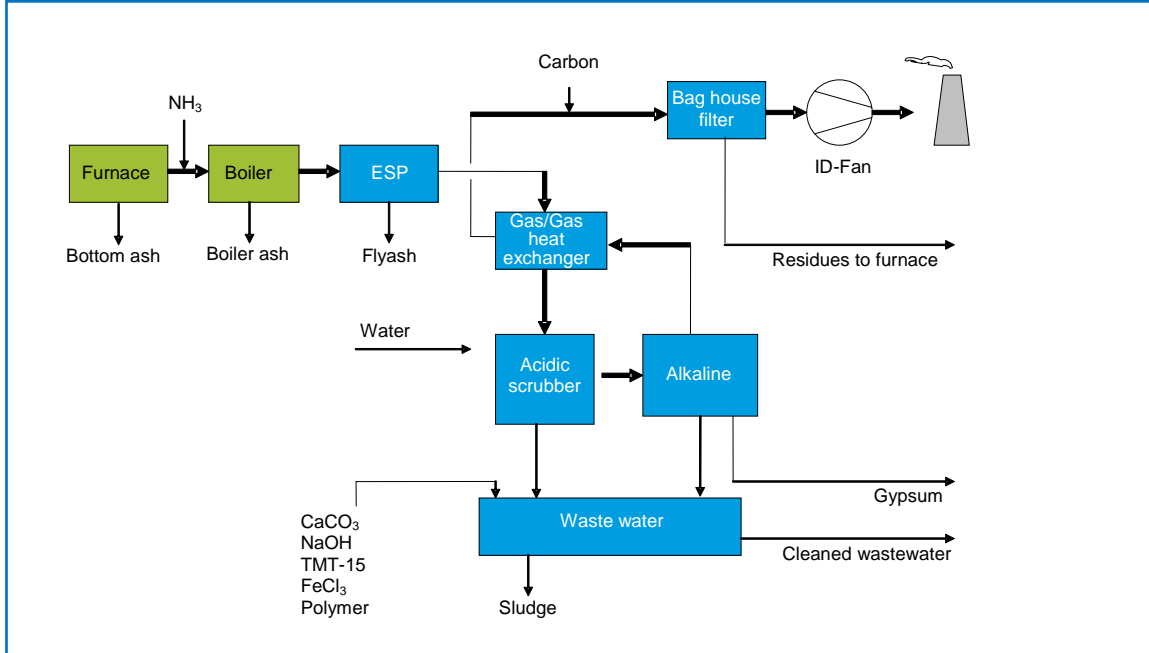
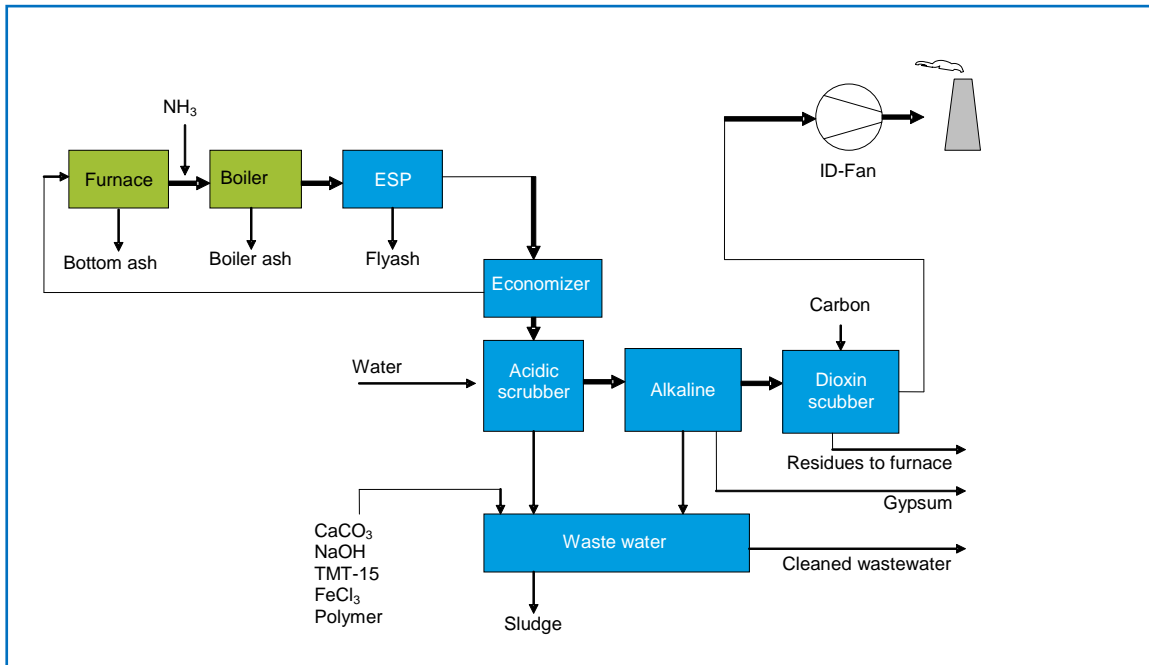


Figure 4-17: Wet APC System, (b)



The diagram illustrates the process flow of a waste-to-energy plant. The process begins with waste entering a **Furnace**, which produces **Bottom ash**. The waste then moves to a **Boiler**, which produces **Boiler ash** and feeds into a **Bag house filter** to produce **Flyash**. The gas stream then passes through an **Economizer**, which preheats the feedwater. The feedwater is then heated in an **Acidic scrubber** (which also receives **Water** and **CaCO₃**, **NaOH**, **TMT-15**, **FeCl₃**, and **Polymer**) before entering an **Alkaline** scrubber. The **Alkaline** scrubber produces **Gypsum** and feeds into a **Waste water** treatment unit. The **Waste water** unit produces **Sludge** and **Cleaned wastewater**. The gas stream then passes through a **Gas/Gas heat exchanger** (preheated by **ID-Fan**), a **Condensing scrubber** (which produces **Water to scrubber**), another **Gas/Gas heat exchanger**, and finally a **Steam heat exchanger**. The **Steam heat exchanger** produces **Drum steam** and feeds into a **SCR (catalyst)** unit. The **SCR (catalyst)** unit produces **Condensate to feedwater** and feeds into the **Acidic scrubber**. The final gas stream is released into the atmosphere through a stack.

Semi-wet systems are basically a combination of semi-dry and wet systems. The semi-wet system combines the semi-dry system with a polishing wet stage in such a way that the water from the wet stage can be used in the preparation of the lime suspension for the semi-dry treatment. Because of this, the semi-wet system is wastewater-free. By adding NaOH to the water in the wet stage the removal efficiency is increased and the production of solid residue decreased correspondingly.

Table 4-1 illustrates the relative advantages and disadvantages of the dry/semi-dry, wet, and semi-wet Systems. As mentioned previously, based upon our research the majority of current WTE facilities in Canada utilize a dry/semi dry APC system while wet systems are more common in the EU.

Table 4-1: Advantages and Disadvantages Associated with Dry/Semi Dry, Wet, and Semi-Wet Flue Gas Treatment Systems

	Dry/Semi Dry System	Wet System	Semi-Wet System
Advantages	<ul style="list-style-type: none"> Simple technology No wastewater Relatively low capital costs Requires less space than a wet system 	<ul style="list-style-type: none"> High efficiency Small amount of solid residue Possible destruction of dioxins in the furnace Generally large margin to limit values Little sensitivity to HCl and SO₂ peaks in the flue gas Relatively low operational costs 	<ul style="list-style-type: none"> Generally large margin to limit values Less sensitive of HCl and SO₂ peaks in flue gas than Dry/Semi Dry System Lower capital costs than wet system No wastewater Less space requirements than Wet System
Disadvantages	<ul style="list-style-type: none"> Uses large quantities of lime and thereby has high operational costs Large amount of solid residue Dioxins in the solid residue Often little margin to the limit values Consumption of lime and amount of solid residues are sensitive to high content of HCl and SO₂ in the flue gas 	<ul style="list-style-type: none"> Many process stage Production of wastewater Relatively high capital costs Requires more space than a dry/semi-dry system 	<ul style="list-style-type: none"> More expensive than dry/semi-dry system Medium amount of solid residue

4.2.2.4 NO_x Control System Components

After the acidic gas control system has been selected, the type of NO_x control is determined. As discussed previously, there are two types of NO_x control systems normally used in WTE facility APC trains. Namely, these are Selective Catalytic Reduction (SCR) and Selective Non-Catalytic Reduction (SNCR). Both NO_x control systems are currently in use in Canada, for example the Greater Vancouver Regional District Waste to Energy Facility utilizes SNCR while the Algonquin Power Peel Energy-From-Waste Facility utilizes SCR.^[111]

In state-of-the-art WTE facilities, sophisticated control systems have been developed that greatly reduce the production of NO_x during regular combustion. However, these control systems are usually not able to reduce NO_x emissions to below applicable regulatory limits and thus additional NO_x controls must be put in place.

¹¹¹ GENIVAR Ontario Inc. in association with Ramboll Danmark A/S, 2007. Municipal Solid Waste Thermal Treatment in Canada

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In SNCR, ammonia is injected into the flue gas stream directly in the furnace at the location where the temperature is around 850°C.

In SCR, the reaction between NO_x and ammonia takes place in a catalytic bed at temperatures normally between 200 and 250°C. In SCR, the catalytic bed is often the last treatment step in the APC train (final treatment device) as dust and SO₂ greatly decrease the lifespan of the catalytic surface. Because of this, the flue gas is often at too low a temperature for the catalytic reaction to take place, requiring the flue gas to be preheated prior to the SCR. Often the incoming flue gas into the SCR system is preheated by the flue gas leaving the SCR which reduces the need for additional heating (which can be done with high pressured steam or natural gas). The consumption of ammonia for an SCR system is normally 1.5 kg 25% NH₃ per kg of NO_x.

The types and choice of Denox currently being used in Europe include both SNCR and SCR, with the choice of system being based both on regulatory requirements and economics. For example:

- In Denmark all Denox systems are based on the SNCR technology as the emission limit of 200 mg/Nm³ can be met with such systems. A NO_x tax has recently been implemented but given the current low level of the tax there is no incentive for further reductions in NO_x emissions.
- In Sweden a high NO_x tax has increased the feasibility of SCR such that most of the new WTE plants are equipped with SCR systems which operate with very low emission levels – often below 20 mg/Nm³.
- In Norway (not member of EU) the regulation can be fulfilled with SNCR but a tax on NO_x based on the size of the WTE facility make the choice of SNCR or SCR comparable.
- Austria has implemented a NO_x emission limit at 70 mg/Nm³ compared to the 200 mg/Nm³ in EU WID and thus in order to meet this limit, SCR systems have been used for many years. The plant in Vienna, Spittelau, has had SCR for close to 20 years. The experience with the catalyst itself is good, however, the design of the preheat-system as well as the possibility for manual inspection and cleaning of the catalyst is not optimal. For new SCR-systems these problems have been addressed and new installations operate satisfactory.
- In Germany the 200 mg/Nm³ emission limit for NO_x was introduced by the national regulation before the EU-regulation was implemented. Many of the German plants are equipped with SCR and have significant operational experience. Some of the older plants have experienced clogging problems. Clogging problems refer to the SCR catalyst being blocked by the chemical reaction products which is mainly due to the design of the catalyst itself because awareness concerning the SO₂ content of the flue gas was not known when initially designing these facilities. For new facilities the reliability of the SCR is high.
- In Italy most WTE plants use SNCR processes. ASM Brescia has experienced good operation and very low emission levels with SNCR. However, the Italian regulation is becoming more stringent especially in the northern part of Italy and ASM Brescia is testing a catalyst system at present.

- Switzerland (not a member of EU) has, like Austria, introduced a NO_x emission limit of 80 mg/Nm³. Most of the WTE facilities in Switzerland are equipped with SCR and have experienced good operation. The SCR is commonly a tail-end solution. One of the Swiss suppliers has good experience from operation of high temperature-low dust SCR solutions.
- In France and Belgium both SNCR and SCR processes are installed.
- In the Netherlands the emission limit is 70 mg/Nm³ and due to that most of the WTE facilities, and all new facilities, are equipped with SCR.

Summary of NO_x Control Systems

The following table (Table 4-2) illustrates the advantages and disadvantages of SNCR and SCR.

Table 4-2: Advantages and Disadvantages Associated with SNCR and SCR

	SNCR	SCR
Advantages	<ul style="list-style-type: none"> ▪ Simple technology ▪ Low capital costs 	<ul style="list-style-type: none"> ▪ Lower consumption of ammonia ▪ Lower emissions possible (10 mg NO_x/m³ can be obtained if enough NH₃ is added)
Disadvantages	<ul style="list-style-type: none"> ▪ Consumes about 30% more ammonia than SCR ▪ Small quantities of ammonia can slip through and pollute the solid residue in dry/semi-dry systems or the wastewater of the wet systems ▪ Typically, vendors may guarantee limits between 100 to 150 mg NO_x/m³ 	<ul style="list-style-type: none"> ▪ High capital costs

4.2.2.5 Mercury and Dioxin/Furan Control System Components

The release of mercury and dioxins/furans from WTE facilities is normally reduced via an activated carbon injection system. Basically, the gaseous mercury and dioxin/furan compounds are adsorbed onto the surface of the activated carbon particles which are later collected in a fabric filter baghouse. This type of control system is capable of removing mercury and dioxin/furans from the flue gas to below regulatory concentration limits. The dioxin filter can either be wet or dry. The dry system is the most commonly used.

4.2.2.6 Trace Heavy Metal Control System Components

The concentration of heavy metals released from WTE facilities is reduced by more than one component of the APC train. In other words, heavy metal control is not specifically associated with any one APC component.

For example, acid gas scrubbers are typically quite efficient in removing large quantities of heavy metals from the flue gas even though this is not their primary purpose. Specifically, wet scrubbers can provide for the significant removal of arsenic, beryllium, cadmium, chromium, lead, manganese and mercury from the flue gas.

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ESPs and fabric filter baghouses also play an important role in the reduction of heavy metals in the flue gas. They accomplish this because volatilized heavy metals often bind to fly ash particles in the flue gas and large quantities of this particulate matter are captured in an ESP or a fabric filter baghouse. In this way, by removing the particulate matter, large quantities of heavy metals are also captured.

Activated carbon is reported to be also used for reducing heavy metals emissions.^[112]

The control of specific heavy metals depends on their distinctive physical and chemical characteristics. For example, mercury is a unique heavy metal in that it vapourizes at a fairly low temperature (357°C) in comparison to other heavy metals. Mercury remains in a gaseous state after passing through the furnace and boiler and its removal from the flue gas depends largely on the speciation of mercury in the flue gas. The speciation of mercury depends on a number of factors such as the amount of mercury present in the waste and the chlorine content of the waste.

At higher chlorine contents (MSW usually contains a sufficient quantity) mercury will be primarily in an ionic form which can be removed by acid gas scrubbers. Metallic mercury (on the other hand) is much harder to control because it is very insoluble in water. Metallic mercury is normally controlled by being transformed into ionic mercury (by adding oxidants) so that it can then be captured by the wet scrubber; or by direct deposition on activated carbon and captured in a downstream ESP or fabric filter baghouse. A small amount of mercury is released into the atmosphere in a vaporous state during the combustion process, while the majority ends up in the APC residue after treatment. Very little mercury ends up in the bottom ash.

Other heavy metals (e.g., arsenic, beryllium, cadmium, lead, manganese etc.) are converted mainly into non-volatile oxides during the incineration process and bind to particulate matter in the flue gas and are then captured by ESPs and fabric filters (some are also captured by activated carbon). The majority of these heavy metals end up in the APC residue after treatment. Typically, a lesser amount of these heavy metals remain in the bottom ash (although for some there is a fairly even distribution between the bottom ash and APC residue).^[113]

4.2.2.7 Particulate Matter Control System Components

As discussed in detail in Section 4.1, particulate matter control is achieved using an electrostatic precipitator or a fabric filter baghouse.

4.2.2.8 Other APC Systems

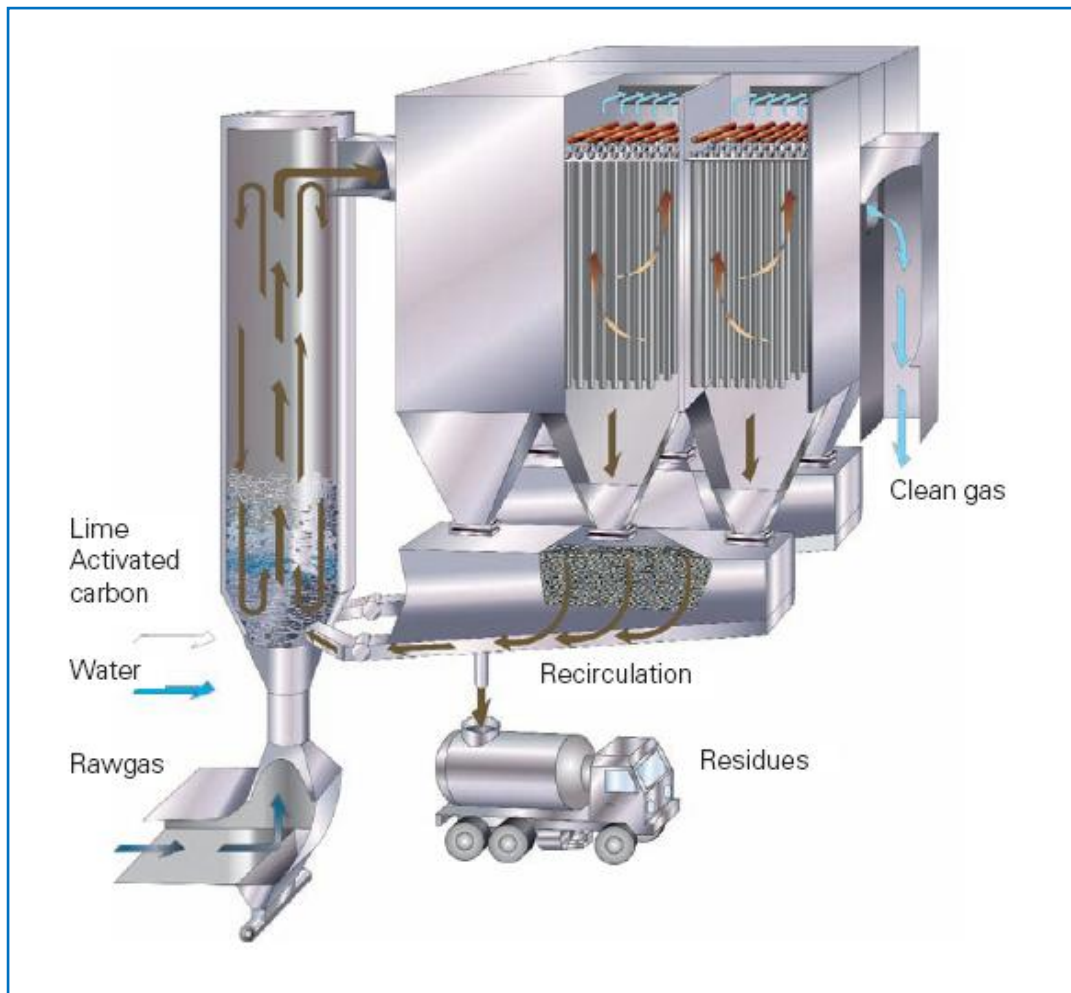
There are several other fairly new APC systems currently being used in Europe. Recently some of the European technologies have been proposed in US. An overview of two such technologies is provided below.

¹¹² European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration

¹¹³ European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration

The Turbosorp solution is promoted by Von Roll Inova. The Turbosorp® process employs a turboreactor with fluidized lime activated carbon and a downstream bag filter. Briefly, the Turbosorp® process works this way: Downstream of the combustion section and steam generator, flue gases are channelled directly into the turbosorber without pre-treatment. Reagents for separation (hydrated lime or calcined lime and activated carbon) are metered into the stream here and water is injected at the same time. The temperature drops below 160°C as a result, improving separation while activating the lime. Pollutants react with the additives in the turbosorber forming products that can be trapped by the downstream fabric filter.^[114] Figure 4-19 provides a schematic overview of the Turbosorp process.

Figure 4-19: Schematic Diagram of the Turbosorp® Turboreactor



Source: Von Roll Inova. 2007. Accessed March 15, 2010 from http://www.aee-vonrollinova.ch/aee_vonroll_inova/products_services/abgasreinigung/turbosorp_r

¹¹⁴ Von Roll Inova. Turbosorp Flue Gas Purification

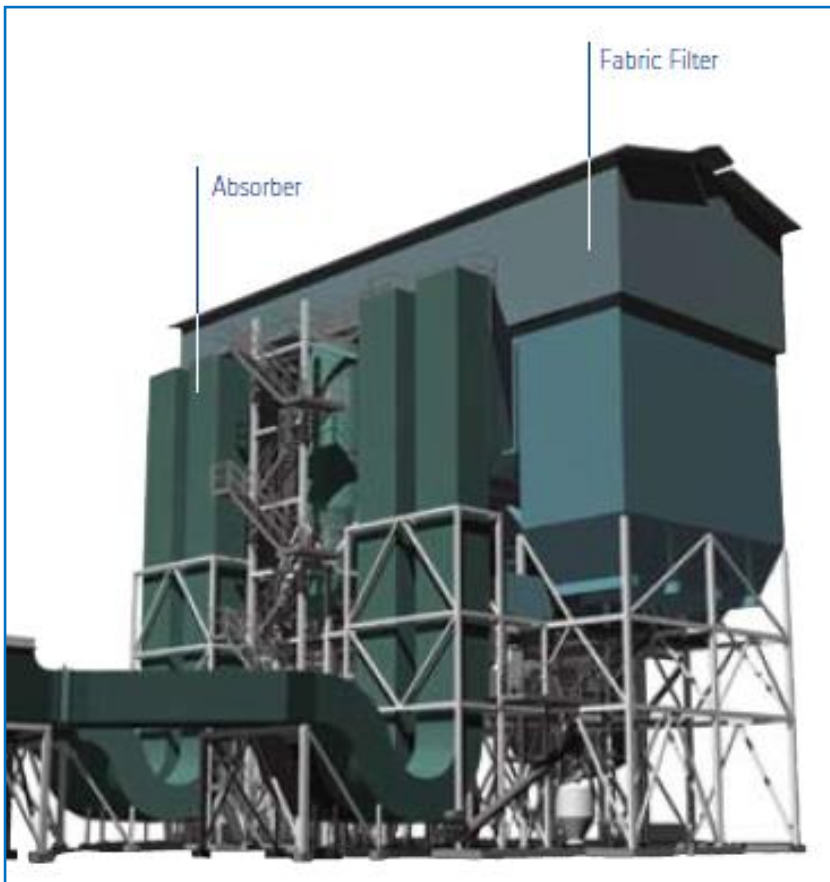
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The NID™ system is a Dry Flue Gas Desulphurization (DFGD) process that is based on the reaction between SO_2 and $\text{Ca}(\text{OH})_2$ in humid conditions. The humidified mixture of hydrated lime and reaction product is injected into the NID system absorber and cools the inlet flue gas by evaporation. The cooled flue gas then flows to the dust collector, preferably a Fabric Filter (FF) or an Electrostatic Precipitator (ESP), where the particles in the flue gas are removed and recycled back through the NID FGD system. In addition to desulphurization, the cooled, humid flue gas combined with a fabric filter provide excellent filtration and reaction conditions, resulting in very low particulate emissions and additional gas absorption (SO_2 , HCl , SO_3 , HF , Hg) in the dust cake.

Figure 4-20 presents a schematic overview of the NID System.

Figure 4-20: Schematic Diagram of the NID System



Source: NID™ Flue Gas Desulphurization System for the Power Industry. Alstom. Brochure

4.2.3 APC for Gasification Facilities

The requirement for an APC system for a gasification facility and the type of system it would use, depends primarily on whether or not the syngas being produced is being utilized onsite for energy generation (in which case some type of APC system would be required) or if the syngas is exported

for use off-site.^[115] If the syngas is exported offsite and used for an alternative purpose (i.e., production of hydrogen or methanol) there may be no emissions to air associated with the facility's operation.

The APC system associated with the Nippon Steel "Direct Melting System" and the APC system associated with the Thermoselect technology are discussed below as both are representative of facilities where the syngas is used on-site.

In the Nippon Steel "Direct Melting System", the syngas produced by the melting furnace is combusted immediately within the facility for energy generation. From limited but various sources, Stantec determined that the typical APC train used at these facilities is as follows. After the combustion chamber, the gas is cooled in a conditioning tower (wet spray type). The cooled gas is then passed through a bag filter (to remove particulate matter) and finally, NO_x is reduced via Selective Catalytic Reduction before the flue gas is released via a stack into the atmosphere. At one of their demonstration plants, Nippon Steel utilized an electrostatic precipitator rather than a bag filter. As can be observed, the APC system utilized by Nippon Steel is very similar to that used by mass burn facilities, although some common treatment steps are not present (i.e., activated carbon injection).^{[116],[117]} Based upon the limited data available, it appears that the Nippon Steel technology is capable of meeting European emissions standards.^[118]

Whether or not a gasification facility utilizing Thermoselect technology requires an APC system depends on how the syngas produced by the facility is to be used. A Thermoselect facility is capable of utilizing the syngas onsite to produce energy (via gas engines for electrical power generation or via boilers for heat or power generation) or export offsite to be used to produce energy or as a reagent in the production of various useful products such as methanol or ammonia. If the syngas is to be utilized onsite for energy generation, some type of APC system would be required.

At Thermoselect facilities, high efficiency gas engines are often used on site to produce electricity by combusting the syngas. In this case, the exhaust gas from the engine would be treated by SCR to reduce NO_x emissions and a catalytic converter would be used to reduce CO emissions (convert it to CO₂). Alternatively, the syngas could be used onsite to produce energy via a steam boiler in which case flue gas produced during the process would be treated prior to being released into the atmosphere. NO_x would generally be reduced via SNCR and a dry adsorption unit could be added to the facility to primarily reduce SO₂ and mercury emissions (sodium bicarbonate injection followed by fabric filter).

One way in which the Thermoselect technology assists in reducing the potential emissions to air associated with the combustion of the syngas it produces is via thorough syngas cleaning. Other gasification technologies also often utilize some form of syngas cleaning. Besides the main

¹¹⁵ If the syngas is exported and combusted offsite, the emissions to air associated with the combustion would truly be associated with the gasification facility itself

¹¹⁶ Nippon Steel Technical Report No. 70. July 1996. Research and Development of Direct Melting Process for Municipal Solid Waste.

¹¹⁷ Nippon Steel Technical Report No. 92. July 2005. Development of High-performance Direct Melting Process for Municipal Solid Waste.

¹¹⁸ University of California, Riverside. 2009. Evaluation of Emissions from Thermal Conversion Technologies Processing Municipal Solid Waste and Biomass.

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components of syngas (CO , CO_2 , H_2 , and H_2O), raw syngas also contains HCl , HF , H_2S , dust and metal compounds. The Thermoselect technology cleans syngas in several steps as follows:

- The hot syngas from the high temperature reactor is quenched rapidly preventing the de novo synthesis of dioxins/furans. The quench consists of a graphite cylinder with spraying nozzles.
- The syngas is then “pre-cleaned” by acidic scrubbing. HCl and HF are dissolved in the quench. This lowers the pH value of the quench to approximately a pH of 2 which assists in dissolving heavy metals as chlorides and/or fluorides and also binds trace amounts of ammonia as ammonia chloride.
- Following acidic scrubbing, dust is removed from the syngas. Dust is removed via a de-dusting scrubber (a water jet pump device) which removes dust and carbon particles from the syngas.
- After dust removal, the syngas undergoes desulphurization. This takes place through the adsorption of H_2S and the partial oxidation to elementary sulphur. Iron chelate is sprayed into the syngas flow causing the reaction.
- Fine dust is then removed from the syngas by a wet electrostatic precipitator if the downstream syngas utilization requires very low levels of dust.
- Finally, the syngas is reheated if a wet electrostatic precipitator is used. By reheating, the temperature of the syngas is raised slightly to avoid water condensation in downstream equipment.

As the list illustrates, the syngas cleaning process utilized by Thermoselect is quite thorough and greatly reduces the contaminants present in the syngas, thereby preventing the potential release of these substances into the air if the syngas is combusted. It should be noted that the Thermoselect process does not produce any wastewater. The water condensed during the different phases of the gas treatment is fed into the process water treatment. The process water undergoes a multiple stage treatment and is then reused for cooling purposes.^[119]

4.3 BACT for APC Systems

In both the Netherlands and Austria, for large waste incineration plants, wet flue gas treatment is considered as BACT. These two countries are considered leaders in the use of WTE and have some of the lowest emissions limits in the world, and information regarding the consideration of BACT in these jurisdictions was considered in the development of the European Commission BREF document on BAT for waste incineration.

The EU waste incineration BREF does not suggest the best method for air pollution control as the decision depends on a number of different factors depending on the particular circumstances

¹¹⁹ Thermoselect. 2005. Thermoselect Plant and Process Description

surrounding a facility. The selection of an APC system should be based on the optimal reduction of air emissions, but should also consider other aspects such as ^[120] ^[121]:

- Type of waste, its composition and variation
- Type of combustion process, and its size
- Flue gas flow and temperature
- Flue gas content and fluctuations in flue gas composition
- Land and space availability
- Availability and cost of outlets for residues accumulated/recovered
- Availability and cost of water and other reagents
- Energy supply possibilities
- Availability of subsidiaries for exported energy
- Tolerable disposal charge for the incoming waste
- Reduction of emissions by primary methods (operational controls)
- Generation of noise
- Minimization of effluent discharge
- The additional overall system compatibility issues that arise when retrofitting existing installations
- Consumption of chemicals and energy
- Maximum energy recovery.

Those factors aside, the waste incineration BREF states that an APC system should be selected that can provide for the emissions levels listed in the following table (Table 4-3) for releases to air.

The BREF also provides a comparative matrix to use when selecting between wet, semi-dry and dry APC systems. Although the comparison is not exhaustive, it does provide a helpful overview of the advantages and disadvantages associated with each of the systems. Table 4-4 presents the comparative matrix as given in the BREF document.

In order to ensure that a WTE facility will meet current stringent emissions limits, vendors of WTE technology are often willing to guarantee that their facility will meet certain emission figures lower than the approved limit criteria. Normally, the contract between the client wishing to have the facility and the vendor building the facility will explicitly state the concentration range for each pollutant that would be guaranteed by the vendor. Further, vendors normally specify the raw gas values that they will assume when designing their air pollution control system and would guarantee the amount of substances that their air pollution control system will consume during treatment (i.e., ammonia, lime etc.).

¹²⁰ European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration

¹²¹ Ministry of Housing, Spatial Planning and the Environment. 2002. Dutch Notes on BAT

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Table 4-3: EU BREF: Operational ELV Ranges Associated with the Use of BAT

Substance(s) (in mg/Nm ³ or as Stated)	Non-Continuous Samples	½ Hour Average	24 Hour Average	Comments
Total dust		1 – 20	1 – 5	In general the use of fabric filters gives the lower levels within these emission ranges. Effective maintenance of dust control systems is very important. Energy use can increase as lower emission averages are sought. Controlling dust levels generally reduces metal emissions.
Hydrogen chloride (HCl)		1 – 50	1 – 8	Waste control, blending and mixing can reduce fluctuations in raw gas concentrations that can lead to elevated short-term emissions. Wet FGT systems generally have the highest absorption capacity and deliver the lowest emission levels for these substances, but are generally more expensive.
Hydrogen fluoride (HF)		<2	<1	
Sulphur dioxide (SO ₂)		1 – 150	1 – 40	
Nitrogen monoxide (NO) and nitrogen dioxide (NO ₂), expressed as NO ₂ for installations using SCR		40 – 300	40 – 100	Waste and combustion control techniques coupled with SCR generally result in operation within these emission ranges. The use of SCR imposes an additional energy demand and costs. In general at larger installations the use of SCR results in less significant additional cost per tonne of waste treated. High N waste may result in increased raw gas NO _x concentrations.
Nitrogen monoxide (NO) and nitrogen dioxide (NO ₂) expressed as NO ₂ for installations not using SCR		30 – 350	120 – 180	Waste and combustion control techniques with SNCR generally result in operation within these emission ranges. 24 hour averages below this range generally require SCR although levels below 70mg/Nm ³ have been achieved using SNCR e.g., where raw NO _x is low and/or at high reagent dose rates) Where high SNCR reagent dosing rates are used, the resulting NH ₃ slip can be controlled using wet FGT with appropriate measures to deal with the resultant ammoniacal wastewater. High N waste may result in increased raw gas NO _x concentrations.
Total Organic Carbon		1 – 20	1 – 10	Techniques that improve combustion conditions reduce emissions of these substances. Emission concentrations are generally not influenced greatly by FGT. CO levels may be higher during start-up and shut down, and with new boilers that have not yet established their normal operational fouling level.
Carbon monoxide (CO)		5 – 100	5 – 30	
Mercury and its compounds (as Hg)	<0.05	0.001 – 0.03	0.001 – 0.02	Adsorption using carbon based reagents is generally required to achieve these emission levels with many wastes – as metallic Hg is more difficult to control than ionic Hg. The precise abatement performance and technique required will depend on the levels and distribution of Hg in the waste. Some waste streams have very highly variable Hg concentrations. Continuous monitoring of Hg is not required by Directive 2000/76/EC but has been carried out in some MSs.
Total cadmium and thallium (and their compounds)	0.005 – 0.05			See comments for Hg. The lower volatility of these metals than Hg means that dust and other metal control methods are more effective at controlling these substances than Hg.
Σ Other metals	0.005 – 0.5			Techniques that control dust levels generally also control these metals.
Dioxins and furans (ng TEQ/Nm ³)	0.01 – 0.1			Combustion techniques destroy PCDD/F in the waste. Specific design and temperature controls reduce de-novo synthesis. In addition to such measures, abatement techniques using carbon based absorbents reduce final emissions to within this emission range. Increased dosing rates for carbon absorbent may give emissions to air as low as 0.001 but result in increased consumption and residues.
Ammonia (NH ₃)	<10	1 – 10	<10	Effective control of NO _x abatement systems, including reagent dosing contributes to reducing NH ₃ emissions. Wet scrubbers absorb NH ₃ and transfer it to the wastewater stream.
Benz(a)pyrene	For these substances there was insufficient data to draw a firm BAT conclusion on emission levels. However, the data indicates that their emission levels are generally low. PCBs, PAHs and benz(a)pyrene can be controlled using the techniques applied for PCDD/F. N ₂ O levels are determined by combustion technique and optimisation, and SNCR optimisation where urea is used.			Techniques that control PCDD/F also control Benz(a)pyrene, PCBs and PAHs
PCBs				
PAHs				
Nitrous oxide (N ₂ O)				Effective oxidative combustion and control of NO _x abatement systems contribute to reducing N ₂ O emissions. The higher levels may be seen with fluidized beds operated at lower temperatures e.g., below ~900°C

- NOTES:**
- The ranges given in this table are the levels of operational performance that may generally be expected as a result of the application of BAT – they are not legally binding emission limit values (ELVs)
 - Σother metals = sum of Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V and their compounds expressed as the metals
 - Non-continuous measurements are averaged over a sampling period of between 30 minutes and 8 hours. Sampling periods are generally in the order of 4 – 8 hours for such measurements.
 - Data is standardized at 11 % Oxygen, dry gas, 273K and 101.3 kPa
 - When comparing performance against these ranges, in all cases the following should be taken into account: the confidence value associated with determinations carried out; that the relative error of such determinations increases as measured concentrations decrease towards lower detection levels
 - The operational data supporting the above-mentioned BAT ranges were obtained according to the currently accepted codes of good monitoring practice requiring measurement equipment with instrumental scales of 0 – 3 times the WID ELV. For parameters with an emission profile of a very low baseline combined with short period peak emissions, specific attention has to be paid to the instrumental scale. For example changing the instrumental scale for the measurement of CO from 3-times the WID ELV to a 10-times higher value, has been reported in some cases, to increase the reported values of the measurement by a factor of 2 – 3. This should be taken into account when interpreting this table.
 - One MS reported that technical difficulties have been experienced in some cases when retrofitting SNCR abatement systems to existing small MSW incineration installations, and that the cost effectiveness (i.e., NO_x reduction per unit cost) of NO_x abatement (e.g., SNCR) is lower at small MSWIs (i.e., those MSWIs of capacity <6 tonnes of waste/hour).

SPLIT VIEWS:

1. BAT 35: Based upon their knowledge of the performance of existing installations a few Member States and the Environmental NGO expressed the split view that the 24 hour NH₃ emission range associated with the use of BAT should be <5 mg/Nm³ (in the place of <10 mg/ Nm³)
2. BAT 35: One Member State and the Environmental NGO expressed split views regarding the BAT ranges). These split views were based upon their knowledge of the performance of a number of existing installations, and their interpretation of data provided by the TWG and also of that included in the BREF document. The final outcome of the TWG meeting was the ranges shown in the table, but with the following split views recorded: total dust 1/2hr average 1 – 10 mg/Nm³; NO_x (as NO₂) using SCR 1/2hr average 30 – 200 and 24hr average 30 – 100 mg/Nm³; Hg and its compounds (as Hg) non-continuous 0.001 – 0.03 mg/Nm³; Total Cd + Tl non-continuous 0.005 – 0.03mg/Nm³; Dioxins and furans non-continuous 0.01 – 0.05 ng TEQ/Nm³. Based on the same rationale, the Environmental NGO also registered the following split views: HF 1/2hr average <1 mg/Nm³; SO₂ 1/2hr average 1 – 50 mg/Nm³ and 24hr average 1 – 25 mg/Nm³

Table 4-4: Example of Some IPPC Relevant Criteria for Selection of APC Systems

Criteria	Wet FGT (W)	Semi-wet FGT (SW)	Dry Lime FGT (DL)	Dry Sodium Bicarbonate FGT (DS)	Comments
Air emissions performance	+	0	–	0	In respect of HCl, HF, NH ₃ and SO ₂ wet systems generally give the lowest emission levels to air. Each of the systems is usually combined with additional dust and PCDD/F control equipment. DL systems may reach similar emission levels as DS and SW but only with increased reagent dosing rates and associated increased residue production.
Residue production	+	0	–	0	Residue production per tonne waste is generally higher with DL systems and lower with W systems with greater concentration of pollutants in residues from W systems. Material recovery from residues is possible with W systems following treatment of scrubber effluent, and with DS systems.
Water consumption	–	0	+	+	Water consumption is generally higher with W systems. Dry systems use little or no water.
Effluent production	–	+	+	+	The effluents produced (if not evaporated) by W systems require treatment and usually discharge – where a suitable receptor for the salty treated effluent can be found (e.g., marine environments) the discharge itself may not be a significant disadvantage. Ammonia removal from effluent may be complex.
Energy consumption	–	0	0	0	Energy consumption is higher with W systems due to pump demand – and is further increased where (as is common) combined with other FGT components e.g., for dust removal.
Reagent consumption	+	0	–	0	Generally lowest reagent consumption with W systems. Generally highest reagent consumption with DL – but may be reduced with reagent re-circulation. SW, and DL and DS systems can benefit from use of raw gas acid monitoring.
Ability to cope with inlet variations of pollutant	+	0	–	0	W systems are most capable of dealing with wide ranging and fast changing inlet concentrations of HCl, HF and SO ₂ . DL systems generally offer less flexibility – although this may be improved with the use of raw gas acid monitoring.
Plume visibility	–	0	+	+	Plume visibility is generally higher with wet systems (unless special measures used). Dry systems generally have the lowest plume visibility.
Process complexity	– (highest)	0 (medium)	+	+	Wet systems themselves are quite simple but other process components are required to provide an all round FGT system, including a wastewater treatment plant etc.
Costs –capital	Generally higher	Medium	Generally lower	Generally lower	Additional cost for wet system arises from the additional costs for complementary FGT and auxiliary components – most significant at smaller plants.
Costs – operational	Medium	Generally lower	Medium	Generally lower	There is an additional operational cost of ETP for W systems – most significant at smaller plants. Higher residue disposal costs where more residues are produced, and more reagent consumed. W systems generally produce lowest amounts of reagents and therefore may have lower reagent disposal costs. Op. costs include consumables, disposal and maintenance costs. Op. costs depend very much on local prices for consumables and residue disposal.

NOTES:

- + means that the use of the technique generally offers an advantage in respect of the assessment criteria considered
- 0 means that the use of the technique generally offers no significant advantage or disadvantage in respect of the assessment criteria considered
- means that the use of the technique generally offers a disadvantage in respect of the assessment criteria considered

5 EXPECTED EMISSION RATES FROM COMBUSTION AND CONTROL SYSTEMS

WTE facilities must be well operated and well maintained to ensure that emissions resulting from their operation are as low as possible. Good combustion practices (i.e., operational controls) can reduce emissions by ensuring that the temperature in the combustion chamber and the retention time for the waste in the combustion chamber are kept at optimal levels. The emissions that are produced during combustion are then reduced further via APC equipment.

5.1 Typical Emissions from Mass Burn Facilities

Table 5-1 illustrates the typical concentration of pollutants in untreated flue gas from a modern conventional mass burn incinerator that treats 15 tonnes of waste per hour for 8,000 hours per year (120,000 tonnes per year). The table also includes the European Union emissions requirements (for comparison purposes) and the typical flue gas quality from a 120,000 tonne per year facility utilizing a semi-dry, wet, or semi-wet APC system.^[122]

As presented in this table and as discussed further within this section of the report, modern WTE facilities with modern APC systems in a variety of configurations are capable of high removal efficiencies for various parameters and can typically achieve emissions that are well within regulated limits. It should be noted that this table presents typical average values for new APC systems, in comparison to the EU emissions requirements. Information presented in Section 5.2, regarding the range of emissions performance for existing WTE plants, includes older facilities that may or may not have recent APC upgrades and thus provide an overview of the range of emissions associated with existing facilities. Care should also be taken in comparing the typical daily average values as presented in Table 5-1 with those that represent average data from either CEM's or Stack Tests (particularly in regards to the averaging periods) as they may not be directly comparable.

Table 5-1: Comparison of Emissions in Raw Flue Gas, EU Emissions Requirements, and Emissions Expected from Semi-Dry, Wet and Semi-Wet APC Systems

Component	Unit	Flue Gas Quality (typical, daily average values)					% Range in Reduction
		Raw Flue Gas	EU Emissions Requirements	Semi-Dry System	Wet System	Semi-Wet System	
Total Particulate Matter	mg/Rm ³	2,000 – 4,500	9.2	3	0 – 2	3	99.90 to 99.95 %
SO ₂	mg/Rm ³	180 – 550	46	<30	1 – 10	15	83.3 to 98.18%
NO _x (with SNCR)	mg/Rm ³	200 – 450	183	<120	<120	<120	40 to 73.3%
HCl	mg/Rm ³	450 – 2,000	9.2	<7	1	2	98.4 to 99.95%
HF	mg/Rm ³	5 – 10	0.92	0.01	0.05	0.05	99.0 to 99.90%
Hg	mg/Rm ³	0.1 – 1	0.046	0.01	0.002	0.002	90 to 99.8%

¹²² Ramboll. 2007. The Regions of Durham and York EfW Facility: Comparison of Flue Gas Treatment Systems

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Component	Unit	Flue Gas Quality (typical, daily average values)					% Range in Reduction
		Raw Flue Gas	EU Emissions Requirements	Semi-Dry System	Wet System	Semi-Wet System	
Cd	mg/Rm ³	1.09 – 2.0	N.A.	0.01	0.002	0.002	99.1 to 99.5%
Cd+Tl	mg/Rm ³	1.0 – 2.0	0.046	0.015	0.005	0.01	98.5 to 99.75%
Pb	mg/Rm ³	25 – 35	N.A.	0.005	0.005	0.005	99.98%
Sum of As, Ni, Co, Pb, Cr, Cu, V, Mn, Sb	mg/Rm ³	5 – 50	0.46	0.05	0.04	0.04	99.0 to 99.92%
Dioxins/Furans I-TEQ	ng/Rm ³	0.009 – 14	0.092	0.08	0.05	0.06	99.6%

NOTE:

Rm³ refers to 25°C

11% O₂, and dry fluegas

NA = Not applicable

5.2 Comparison of Emission Rates from Existing Facilities

The actual air emissions performance of several operating WTE facilities has been reviewed to provide a broad understanding of the emissions from current operating facilities.

Table 5-2 below summarizes the reported emissions from various existing and proposed WTE facilities globally. The emission components are only those actually reported for these facilities; not all facilities are required to report the same components. In regards to the reported values, in many cases the emissions reported are daily averages obtained from CEMs and/or average data reported from stack tests.

Examination of the reported emissions data indicates that in general:

- Modern WTE facilities in North American and EU jurisdictions emit many parameters within the same order of magnitude.
- The range of reported emissions values widens as older facilities are included in the reported range of values.
- The majority of North American and EU jurisdictions require monitoring and reporting of similar emissions, although there are some distinct differences in regards to the monitoring and reporting of trace metals and trace organic parameters.

Table 5-2: Comparison of Emissions from Various Existing WTE Facilities

Component	Unit	Metro Vancouver WTE Facility (2007) ^[1]	Modern Italian Waste Incinerator ^[2]	Emissions from retrofitted UK plant (2001) ^[3]	Sheffield Energy Recovery Facility (UK) (March 2010) ^[4]	SELCHP (UK) (December 2009) ^[5]	Range from European MSW Incinerators (2006) ^[6]			Average Values from 87 US WTE Facilities ^[7]	Average of Three High Performing US Facilities ^[7]	Average of 10 Finalists in WTERT 2006 Award ^[7]	SEMASS (US) (July 2006) ^[8]	Wels (Austria) (2000) ^[11]	Spittelau (Austria) (2000) ^[11]	Flotzersteig (Austria) (2000) ^[9]	Average Values from Dutch Incinerators (2002) ^[10]
				Daily Average	Daily Average	Daily Average	Annual Average	Daily Average	Half Hour Average		Daily Average	Daily Average		Half Hour Average	Half Hour Average	Half Hour Average	Daily Average
Total Particulate Matter	mg/m ³	3.8	1.9	0.84	0.75	3.63	0.093–3.73	0.093–9.32	<0.047–13.98	2.8	1.0	2.9	1.7	<0.47	0.75	1.96	0.47–2.8
CO	mg/m ³	23	9.3	4.7	1.86	5.59	1.86–42	0.93–93	0.93–140		11.5	22.4	37.7	18.6	24.5	14.2	4.66–47
SO ₂	mg/m ³	85	7.5		8.39	1.86	0.19–19	0.47–47	0.093–233	11.2	5.3	2.8	71.7	<1.86	1.96	9.69	1.86–28
NO _x	mg/m ³	265	130.4	255.3	85.53	144.42	18.6–168	28–186	18.6–419	227.7	46.3	104	204	50.3	21.4	28.4	37.2–65
HCl	mg/m ³	23.6	6.5	18.6	5.40	4.85	0.093–5.6	0.093–9.3	0.093–75	10.6	1.5	7.9	8.4	<0.093	0.75	0.093	0.47–4.7
HF	mg/m ³	0.1	0.65	<0.093			0.009–0.09	0.09–0.093	<0.019–0.9					<0.047	<0.019	0.13	0.093–0.47
VOCs	mg/m ³		3.07														
TOC	mg/m ³				0.93	0.19	0.093–4.7	0.093–9.3	0.093–23		0.65	0.95			0.47		0–9.32
Methane	mg/m ³	4.3															
Hg	mg/m ³	0.002	0.009				0.00019–0.047	0.0047–0.047	0.0013–0.034	0.007	0.0028	0.0093	0.0009	<0.0019	0.065	0.0335	0.0047–0.019
Cd	mg/m ³	0.0006	0.009	<0.00093				0.0003–0.003		0.0007			0.0001	<0.0019	0.00093	0.00186	
Cd,Tl	mg/m ³						0.0002–0.028										0.0009–0.0093
Pb	mg/m ³	0.0059	0.093					<0.002–0.041		0.014			0.0127	<0.0019	0.0112	0.041	
Sum of As, Ni, Co, Pb, Cr, Cu, V, Mn, Sb	mg/m ³			<0.932			0.0002–0.047										
PAH	µg/m ³	0.13					<9.324										
PCB	µg/m ³	0					<4.66										
Dioxins/Furans I-TEQ	ng/m ³	0.002	0.047	0.0056			0.00019–0.075			0.04	0.0019	0.0186	0.024	0.0028	0.0186	0.0168	0.009–0.047

NOTES:

Reference conditions: 101.3 kPa, 20°C, dry gas, 11% O₂

¹ AECOM Canada Ltd. 2009. Management of Municipal Solid Waste in Metro Vancouver – A Comparative Analysis of Options for Management of Waste After Recycling

² M. Guigliano, *et al.* 2008. Energy Recovery from Municipal Waste: A Case Study for a middle-sized Italian District. In Waste Management 28 (2008) 39 – 50 (representative of modern WTE plants equipped with a dry flue gas cleaning system (dry scrubbing + activated carbon) followed by a fabric filter. Nitrogen oxides are controlled by selective non-catalytic reduction activated by urea.)

³ Porteous. 2001. Energy from waste incineration - a state of the art emissions review with an emphasis on public acceptability.

⁴ Sheffield Energy Recovery Website (<http://www.veoliaenvironmentalservices.co.uk/sheffield/pages/emissions.asp>) (All based on continuous measurements).

⁵ SELCHP Website (<http://www.selchp.com/emissions.asp?year=2009&emissionId=48>) (All based on continuous measurements). APC system is comprised of SNCR, semi-dry lime and activated carbon injection.

⁶ European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration. (All based on continuous measurements except for heavy metals and dioxins/furans which are based on sampling periods generally in the order of 4 – 8 hours).

⁷ C.S. Psomopoulos, *et al.* 2009 Waste-to-energy: A review of the status and benefits in USA. (All based on continuous measurements except for heavy metals and dioxins/furans which are based on spot samples).

⁸ SEMASS Boiler NO. 3 Test Results.

⁹ European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration. APC system includes ESP, three wet scrubbers, and SCR. TPM, HCl, SO₂, TOC, CO, NO_x are based on CEMS, rest are based on discontinuous measurements.

¹⁰ Ministry of Housing, Spatial Planning and the Environment. 2002. Dutch Notes on BAT for the Incineration of Waste. (All based on continuous measurements except for heavy metals and dioxins/furans which are based on spot samples). Almost all Dutch incinerators employ wet scrubbers and SCR.

¹¹ European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration. APC system includes ESP, two wet scrubbers, and activated coke filter, and SCR, TPM, HCl, HF, SO₂, TOC, CO, NO_x are based on CEMS, rest are based on discontinuous measurements.

¹² European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration. APC system includes SNCR, two fabric filters and wet scrubbing. TPM, HCl, SO₂, TOC, CO, NO_x are based on CEMS, rest are based on discontinuous periodic measurements.

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It should be noted that the basis on which emission rates are calculated vary by jurisdiction. Emissions from combustion facilities must be adjusted, or 'corrected', to pre-determined standard conditions. Most emissions are reported on a mass per volume basis, such as milligrams per standard or reference cubic metre (mg/Rm³). The correction to standard conditions is necessary because, as a gas, the volumetric rate of discharge will vary as a result of temperature and pressure (gauge and absolute). The rate also varies with the composition of the gaseous constituents, such as percent O₂ and CO₂. BC Standard Conditions are 20°C, 101.325 kPa, dry gas (0% moisture) and include site specific standard conditions for %O₂ or %CO₂.

5.3 Air Emissions Quality Trends

Air emissions from modern state-of-the-art WTE facilities are greatly reduced in comparison to older facilities that have less stringent operational controls and less effective air pollution equipment and monitoring systems. WTE tends to be highly regulated in consideration of the potential effects of emissions on human health and the environment, and public perception. For this reason, developed countries have very strict emissions standards. Contemporary air pollution control technologies have been developed to stay well within these limits.^[123]

In the United States, there are currently 89 operating WTE facilities that treat MSW. The emissions from WTE facilities have decreased substantially over the past number of decades due to improvements made to waste combustion technologies. A memorandum released by the United States Environmental Protection Agency in 2007 presented the overall emissions reductions achieved by large and small municipal waste combustion (MWC) units which were retrofitted with Maximum Achievable Control Technology from 2000 to 2005. The table presents the emissions from these large and small municipal waste combustion (MWC) Units in 1990 (prior to retrofits) and in 2005 (after retrofits). As the table illustrates, the reduction of emissions was quite significant.^[124]

Table 5-3: Emissions from Large and Small MWC Units at MACT Compliance (US EPA)

Pollutant	1990 Emissions (tpy)	2005 Emissions (tpy)	Percent Reduction
Dioxins/Furans, TEQ basis	4,400	15	99+%
Mercury	57	2.3	96%
Cadmium	9.6	0.4	96%
Lead	170	5.5	97%
Particulate Matter	18,600	780	96%
HCl	57,400	3,200	94%
SO ₂	38,300	4,600	88%
NO _x	64,900	49,500	24%

¹²³ European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration

¹²⁴ United States EPA. 2007. Memorandum: Emissions from Large and Small MWC Units at MACT Compliance

5.4 Factors Affecting Emission Rates

A number of factors affect the emissions rates from the thermal treatment of MSW. Generally, the following factors are the main contributors to emission performance:

- Waste composition and content (depends on jurisdiction and diversion practices in place)
- Thermal treatment technology (the design and operation of the thermal treatment facility)
- Design and operation of the APC equipment.

The following subsections will discuss each of the factors that affect emissions in greater detail.

5.4.1 Waste Composition and Content

Several of the substances of concern that are emitted to air from WTE facilities originate from the MSW being treated. For example, the release of heavy metals such as mercury and cadmium as well as acidic gases such as SO₂ and HCl from WTE facilities is driven largely by the presence of these substances in the post-diversion waste stream.

As an illustration, if the post-diversion waste stream contains a lot of mercury laden waste (compact fluorescent light bulbs, mercury thermometers), more mercury is likely to be released into the atmosphere, even after the flue gas is treated with state-of-the-art APC equipment. Therefore, it is desirable in jurisdictions which utilize thermal treatment to try and minimize contaminants present in the residual waste stream. The composition of MSW depends on the types and quantities of materials being placed in the garbage stream by residents. This depends on the types of diversion programs available in a given jurisdiction, the public's participation in these programs, as well as the types of materials being used and disposed of in a given jurisdiction.

BC has been particularly active at removing these contaminants from the waste stream – further information is available on the web at: <http://rcbc.bc.ca/education/retailer-take-back>.

The following table illustrates how the removal of various materials from the residual waste stream will affect the thermal treatment of the remaining waste stream.^[125]

Table 5-4: Impact of Material Removal and Pre-treatment on Residual Waste

Materials Removed	Impact on the Remaining Waste
Electronics	<ul style="list-style-type: none">▪ Increase in calorific value▪ Decrease in hazardous metal loading, may reduce chlorine loads
Glass and Metals	<ul style="list-style-type: none">▪ Increase in calorific value▪ Decrease in quantity of recoverable metals in slag (or bottom ash)
Paper, Cardboard and Plastic	<ul style="list-style-type: none">▪ Decrease in calorific value▪ Possible reduction in chlorine loads if PVC plastic is common

¹²⁵ European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration

Materials Removed	Impact on the Remaining Waste
Organic Wastes (food and garden materials)	<ul style="list-style-type: none"> Reduction in the moisture loads (particularly during peak loads) Increase in calorific value
Bulky Wastes	<ul style="list-style-type: none"> Reduced need for removal/shredding of such wastes
Hazardous Wastes	<ul style="list-style-type: none"> Reduction in hazardous metal loading Reduction in some other substances (e.g., Cl, Br)
Construction and Demolition Waste	<ul style="list-style-type: none"> Reduction in sulphur content (gypsum from drywall).

Having a diversion program in place does not necessarily mean that it will capture the targeted materials unless residents participate regularly in the program. For example, if a resident discards compact fluorescent light bulbs but chooses not to participate in his/her community's hazardous waste diversion program, this will lead to increased levels of mercury in the waste stream and thus increase the potential for mercury release during thermal treatment. Most jurisdictions try to increase public participation in their diversion programs through aggressive promotion and education campaigns.

Finally, even if a jurisdiction has a mature waste management system and regular participation by residents in the diversion programs, this does not definitively mean that potential hazardous materials will be removed from the garbage stream. For example, if manufacturers increase the use of non-recyclable PVC plastic within their products, the overall chlorine content of the waste will increase leading to a potential increase in HCl production during the thermal treatment of the waste material. The removal of potentially hazardous materials from the residual waste stream is difficult as policies which govern materials such as packaging and product formulation are usually out of the local jurisdiction's control.

5.4.2 Selection of Thermal Technology

The thermal treatment technology being used to treat MSW also has a significant impact on the emissions released. Differences will be observed from technology to technology and within each technology grouping.

The proper operation of a thermal treatment facility plays a significant role in emissions performance. If appropriate operational controls are maintained over the combustion process (proper temperature and residence time, adequate overfire air) less emissions of organic compounds and products of incomplete combustion will be realized (e.g., dioxins/furans, CO). Additionally, the waste stream can be pretreated to ensure proper homogenization and removal of undesirable materials. The above examples are by no means an exhaustive list of potential operational considerations but are meant for illustrative purposes only.

5.4.3 Design and Operation of APC Equipment

The design and operation of a WTE facility's APC equipment will have a significant impact on the type and rate of emissions arising from its operation. As discussed in previous sections, different types of APC trains (i.e., wet, semi-dry) are capable of reducing emissions to varying levels. Wet

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systems tend to provide more flexibility and are typically able to reduce emissions to a greater degree than dry systems.

In addition to the type of APC system, the operation of a given system will also have a great effect on emissions reduction performance. If a system is well maintained and operated under optimal conditions, the rate of emissions will be reduced. For example, in a fabric filter baghouse, the filter cake should be kept at a particular thickness so as to capture the majority of particulate matter without reducing air flow too significantly.

As another example, SNCR systems are capable of reducing NO_x emissions well below emissions requirements depending on the quantity of reagent (NH₃) added to the flue gas stream. The amount of reagent added depends on the desired emissions levels as well as the costs associated with reagent supply.

6 EMISSIONS FROM USE OF REFUSE DERIVED FUEL

6.1 RDF Overview

The composition of Refuse Derived Fuel (RDF) produced from MSW varies according to the origin of the waste material and the sorting/separation process used to produce the RDF. The following table (Table 6-1) presents an overview of the typical composition of RDF produced through the processing of MSW.^[126]

RDF, which is also often called Solid Recovered Fuel (SRF), is typically produced by processing municipal solid waste through: shredding, selective materials recovery (metals), dehydrating and packaging for transport into bale, brick or pellet form. RDF can be comprised of more homogenous residue streams generated by industry such as off-cuts from production of packages, or inorganic (plastic) residues removed from finished compost. RDF can also be generated through source separation of specific material streams such as separation of clean or contaminated wood waste materials from construction and demolition wastes.

Other waste materials can also be processed into waste derived fuels. Waste tires have been used as a fuel supplement as tire derived fuel (TDF) in cement kilns and pulp mill power boilers.

Table 6-1: Typical Composition of RDF Derived from MSW

Waste Fraction	Flemish Region		Italy	UK
	Resulting from Sorting Process (%)	Resulting from Mechanical/Biological Treatment (%)	%	%
Plastic	31	9	23	11
Paper/Cardboard	13	64 ⁽¹⁾	44	84
Wood	12	25 ⁽²⁾	4.5	5 ⁽⁴⁾
Textiles	14		12	
Others	30		14 ⁽³⁾	
Undesirable material (glass, stone, metal)		2	2.5	
Dry-solid content	66%	85%	–	–

NOTES:

⁽¹⁾ Includes, paper, textile, wood

⁽²⁾ Includes rubber, synthetic material

⁽³⁾ Includes organic degradable waste

⁽⁴⁾ Includes glass, wood, textiles and metals

In all cases, the application of this supplemental fuel in industrial or other applications, involves waste materials that have been processed in some way to make them more suitable for introduction

¹²⁶ European Commission – Directorate General Environment. 2003. Refuse Derived Fuel, Current Practice and Perspectives (B4-3040/2000/306517/MAR/E3) Final Report

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into the fuel feed system and to optimize thermal and emissions performance. Unprocessed, raw MSW is not used as a supplemental fuel supply for industrial applications as it would generally not be considered suitable from an operational standpoint given that it is highly heterogeneous.

Beyond the practical advantages of blending the fuel supply, the biogenic portion of RDF may have an environmental and/or monetary value in terms of GHG offsets from fuel substitution if GHG emissions are reduced compared to a business-as-usual scenario and the fuel substitution meets applicable criteria.

Refuse derived fuel (RDF) can be produced from municipal solid waste (MSW) through a number of different processes including the following:

- Separation at source
- Sorting or mechanical separation
- Size reduction (shredding, chipping and milling)
- Separation and screening
- Blending
- Drying and pelletizing
- Packaging
- Storage.

Processing includes removal of any components that could pose quality and environmental concerns. The purpose of the processing of MSW is to generate a fuel source that is relatively homogenous and free of any undesired components.

There are two primary approaches which can produce a high calorific fraction from domestic MSW, which can be used as RDF:

- Mechanical Biological Treatment
- Dry Stabilisation Process.

In a mechanical biological treatment facility (MBT), mixed solid wastes are separated into the following:

- Metals (recovered for recycling)
- Inert materials
- Organic materials (often stabilized using composting processes or anaerobic digestion)
- A residual fraction that has a high-calorific value as it is composed mainly of dry residues of paper, plastics and textiles that can be used as an RDF.

RDF can also be produced through a 'dry stabilization' process, in which residual waste (following removal of the inert portion of the waste and metals) are effectively dried (and stabilized) through a composting process, leaving the residual mass with higher calorific value and suitable for combustion.

The quantity of RDF produced per tonne of processed MSW varies depending on the type of collection, treatment process and quality requirements. The rate of RDF production from MSW can vary between 25 and 85% by weight of waste processed depending on the treatment process used.

The final form and characteristics of RDF produced through processing facilities is usually tailored to the intended industrial application of the material, as the specifications in regards to fuel quality, composition, particle size and density etc. can vary significantly from application to application. The following sections provide discussion on two specific applications of RDF within BC industry, followed by general discussion on how the use of RDF in general should be regulated within the province.

6.2 RDF Use in Wood Fired/Pulp Mill Boilers

6.2.1 General Discussion

Typically, pulp mill boilers are designed to combust relatively clean wood waste in the form of bark, sawdust and small dimension chunks of woody debris, commonly called hog fuel. Contaminants in the hog fuel will vary depending on the location of the mill and source of hog fuel. For example, coastal mills burning wood residuals from timber boomed in salt water will have elevated concentrations of chloride. Timber boomed in a river will have a higher concentration of silt and sand mixed in, potentially forming a nuisance slag in the furnace. There are few other contaminants in the fuel supply for wood fired boilers. Metal, plastic and chlorinated organic compounds are, for the most part, absent from the fuel supply.

Pulp mill boiler APC equipment typically consists of cyclones, baghouses and ESPs, used singly or in combination. Systems to control acid gas or to capture toxic organic compounds are not normally installed on these types of boilers, as these contaminants of concern are not normally produced. Particulate emissions, opacity of the discharge and gaseous components including NO_x, SO_x, CO and unburned hydrocarbons are typically the emissions of concern with wood fired boiler systems. If salt laden wood is burned dioxins and furans are also released (for these situations Ministry permits contain appropriate emission limits). The BC MOE previously commissioned a report on emissions from wood fired combustion equipment in BC which discusses facility and APC design and costs, current performance and achievable emissions limits for various wood fired combustion approaches.^[127]

There is interest in BC to use wood fired boilers for treatment of construction and demolition wastes that have been processed to remove undesirable constituents, such as gypsum, plastic and metals. The option is attractive given the potential to supplement fuel in areas where fibre and fuel supply is constrained. It also eliminates the need for landfilling these wastes while providing the opportunity to convert the waste to energy in the form of electricity, process steam or potentially district heat.

There are a number of constraints to the use of wood fired combustion boilers for treatment of MSW, RDF or construction and demolition debris, including:

¹²⁷ Envirochem, 2008. Emissions from Wood-Fired Combustion Equipment
http://www.env.gov.bc.ca/epd/industrial/pulp_paper_lumber/pdf/emissions_report_08.pdf

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- The waste type needs to be of similar type to the design fuel source intended for the boiler. Issues around calorific value, moisture content and the presence of contaminants of concern can be minimized if the fuel supply is limited to predominantly wood. Raw MSW and most types of RDF will not be suitable for this application as a result of elevated plastic and metal in the fuel supply. Unsorted demolition waste is also not likely to be compatible with the combustion and APC systems as a result of contamination by plastic, gypsum, textile wastes and metals.
- The facility has to have the ability to feed the wastes into the boiler in a manner that maintains operational control and performance without adversely affecting emission quality. It would be necessary to shred (hog) woody debris to make it suitable for feeding into the boiler.
- Given that even processed RDF or construction and demolition waste may include contaminants not present in hog fuel from a sawmill, controlling and monitoring emission quality relative to the ELVs in the facility permit and/or other emission criteria or standards is critical. For example, the current emission limit values for total particulate from wood fired power boilers is typically higher than the value for WTE facilities. Particulate ELVs in BC for wood fired boilers in a non-urban setting range between 120 mg/m^3 to 230 mg/m^3 , in contrast to the current WTE facility particulate ELV of 20 mg/m^3 . The current ELVs for wood fired boilers typically do not specify concentrations of trace metals or toxic organic compounds whereas these are important criteria for a WTE facility.
- In many cases it is reasonable to anticipate that it will be uneconomic to retro-fit APC systems to treat the host of other emissions (in addition to particulate for instance) not normally produced by firing wood waste. Therefore, the emission quality has to be essentially unchanged from the design emission produced by the facility when operating solely on wood waste.

The following sub-sections discuss proposed approaches for the application of two RDF streams in wood fired boilers being wood waste and tire-derived fuel, as these are the potential RDF streams in which the most interest has been demonstrated to-date for such applications.

6.2.2 Use of Wood Waste in Pulp Mill/Wood Fired Boilers

Construction and demolition wastes includes discarded materials generally considered to be not water soluble and non-hazardous in nature, including but not limited to steel, glass, brick, concrete, asphalt material, pipe, gypsum wallboard, and wood waste, from the construction or destruction of a structure or from the renovation of a structure. Wood wastes arising from construction include off cuts from structural timbers, timber packaging, scaffolding, wooden hoardings, whereas wood wastes arising from demolition include used structural timbers, e.g., floorboards, joists, beams staircases and doors.

For the purpose of distinguishing between wood waste sources that could be used as alternative fuels for wood fired boilers, the following defines the two broad categories of wood waste based fuels that may be suitable when recovered from the construction and demolition waste stream.

1. “Clean” wood waste means uncontaminated wood or wood products, from which hardware, fittings and attachments, unless they are predominantly wood or cellulose, have been removed (e.g., clean wooden shakes and shingles, lumber, wooden siding, posts, beams or logs from log home construction, fence posts and rails, wooden decking, millwork and cabinetry), and **excludes**:
 - Any engineered or chemically treated wood products, such as products with added glues or those treated for insect or rot control (oriented strand board, plywood, medium density fibre board, wood laminates or wood treated with chromated copper arsenate, ammoniacal copper arsenate, pentachlorophenol or creosote)
 - Upholstered articles
 - Painted or varnished wood articles or wood with physical contaminants, such as plaster, metal, or plastic
 - Any wood articles to which a rigid surface treatment is affixed or adhered.

Clean wood waste also excludes other materials found in the construction and demolition waste stream such as gypsum or drywall, fibreglass, asphalt or fibreglass roofing shingles, metals or plastics.

2. “Contaminated” wood waste is primarily composed of wood or wood products, but may include of engineered wood products, painted or treated wood, gypsum or drywall, fibreglass, asphalt or fibreglass roofing shingles, metals or plastics.

Land clearing waste is not considered as part of the construction and demolition waste stream for the purpose of this discussion. The sources of land clearing waste can range from land clearing by individual property owners on acreages to developers clearing areas for entire subdivisions.

Generally entire trees are removed, including the root systems which contain soil. In many cases this debris is not left to season before it is disposed of, which results in less than optimal fuel because of the high moisture content and the existence of large quantities of soil.

The chemical composition of clean wood waste and its fuel characteristics are essentially the same as the current permitted fuel stream for existing wood fired boilers. Combustion of clean wood waste as defined above, within existing wood waste boilers, can be accommodated by existing facilities within the currently permitted emissions limits and would be regarded as a minor modification to current operations. Fuel testing would be necessary both initially (to support minor permit changes) and during regular operations to ensure that the wood waste fuel accepted for combustion, continues to meet regulated specifications for ‘clean’ wood waste.

Combustion of wood waste contaminated with organic and inorganic wood protection and wood preservation chemicals has been conducted in BC power boilers over the past two decades. This includes wood contaminated with creosote (railway ties and some structural timber), and pentachlorophenol treated wood (utility pole and some structural timber). It should be noted that chlorophenol use as a wood protection (anti-sapstain) chemical was discontinued in the early-1990s and chlorophenols are now only found in limited wood preservation applications. Therefore, the presence of chlorophenols in refuse derived fuel is now considered to be unlikely. In the past, these

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waste streams have been included on a limited fuel substitution basis in trial burns. While these tests have generally resulted in acceptable emissions from the facility, other constraints including public concern and waste material handling have prevented adoption of larger programs of fuel substitution with these materials. Other applications of 'contaminated' wood waste have included the use of wood waste contaminated by other construction and demolition materials.

Substitution and supplementing fuel supply with 'contaminated' wood waste should be acceptable under specific conditions and would require amendment of current facility permits as follows:

- Use of 'contaminated' wood waste as fuel would likely be considered a major modification to the operations for a given facility and would require permit amendments to address operational changes and revised ELVs, which for a number of parameters would be consistent with those proposed for WTE facilities.
- Testing of the proposed fuels including mass balance analysis to determine the potential shift in emissions concentrations at various substitution rates would be required. This should be accompanied by fuel trials undertaken to demonstrate the actual shift in emissions concentrations associated with use of the proposed fuels.
- As part of the permit amendments, revised ELVs would be necessary in order to limit the potential for effects from air emissions. Revised ELVs could reflect the following:
 - Revised particulate limits to reflect new performance expectations in accordance with those identified in the Envirochem report "Emissions from Wood-Fired Combustion Equipment" which suggests that achievable particulate emission limits for wood fired boilers are in the order of 35 mg/m³ for facilities ranging in size from 3 to 39 MWh or 20 mg/m³ for facilities of 40 MWh and larger.^[128]
 - Retention of the existing limits for CO and NO_x given that emissions performance for these parameters is based on general facility design and operations.
 - Application of the limits proposed for other parameters (heavy metals, persistent organic pollutants) based on those proposed for municipal solid waste incinerators (Section 9.3).
- Fuel quality testing should be undertaken initially to ensure the proposed source and type of material is suitable for consideration, during fuel testing to demonstrate the potential fate of various parameters in the fuel during the combustion process and on a regular basis during operations to ensure that fuel quality specifications (both regulated and unregulated) are being met. During normal operations, it would be reasonable in the first few years for the facility to test its contaminated wood waste fuel supply at least quarterly through random samples to ensure compliance with permits and to ensure that the fuel suppliers meet the requirements set out by the operator.

¹²⁸ Envirochem, 2008. Emissions from Wood-Fired Combustion Equipment
http://www.env.gov.bc.ca/epd/industrial/pulp_paper_lumber/pdf/emissions_report_08.pdf

- Proponents that intend to use a 'contaminated' wood waste as a portion of their fuel stream, would need to identify the proposed rate of fuel substitution and would have to demonstrate their ability to meet the revised ELV's as discussed above, at the proposed maximum substitution rate.

6.2.3 Use of Tire Derived Fuel in Pulp Mill/Wood Fired Boilers

In North America, the use of supplementary fuels in the pulp industry has generally been limited to TDF. About 26 million tires per year are consumed as fuel in US pulp and paper mill power boilers. These facilities typically use wood waste as the primary fuel supply, but the operators have found that the use of TDF increases the stability of the boiler performance. TDF is used in many plants as a supplement to wood because of its high heat value and low moisture content. TDF produces 100 – 200% more energy than wood on a mass basis, according to the US Environmental Protection Agency. The main problem in using TDF in the pulp industry is the need to use de-wired tires. Pulp mills use TDF instead of whole tires because metal wires clog the feed systems. De-wired TDF can cost up to 50% more than regular TDF.^[129]

Within BC, one coastal paper mill supplements the wood waste fuel supply with TDF in one of its three boilers. The boilers were redesigned in the late 1990s to accommodate the use of TDF, believed to be a necessary addition resulting from shortages in fuel supply and an apparent downward trend in the quality of fuel. TDF was selected as a supplementary fuel partly due to the proximity of a local tire recycling facility.

Potential environmental issues relating to the use of TDF at this facility included the risk of:

- Increase in particulate emissions
- Increase in zinc content of the fly ash
- Increase in sulphur content potentially resulting in acid gas generation
- Increase in other trace toxic organic emissions (such as dioxins and furans) that may affect emissions and ambient air quality.

After receiving approval to allow 2 – 5% TDF, performance monitoring results revealed stabilization of the boiler operation when burning lower quality hog fuel, increased fluidized bed temperature, and approximately 5% increase in hog fuel burn rate. Emission monitoring revealed that there was no impact of TDF addition on the total particulate emissions, SO₂ emissions, and no increase in any of the metals in the stack emissions compared with the baseline measurements. Zinc and iron content in fly ash and bottom ash increased. There was no increase in the trace levels of dioxins and furans in the fly ash from TDF addition to the boiler.^[130]

¹²⁹ United States Environmental Protection Agency (September 2008), *Tire-Derived Fuel*, Retrieved February 23, 2010, from <http://www.epa.gov/osw/conservation/materials/tires/tdf.htm>

¹³⁰ L. Cross and B. Ericksen, *Use of Tire Derived Fuel (TDF) in a Fluidized Bed Hog Fuel Paper Boiler at Pacifica Papers Inc.*, Retrieved February 23, 2010, from http://www.portaec.net/local/tireburning/use_of_tire_derived_fuel.html

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Proper equipment or modifications to reduce emission levels are required to burn TDF in these boilers. Several emission control devices and techniques are known, and these have decreased emission levels to within standards. Only a small percentage of industrial boilers have the required combination of system design and fuel type conducive to appropriate TDF substitution and controlling SO_x and particulate emissions is required. SO_x can be controlled by scrubbers present in some systems, especially if the scrubbers operate at a neutral or basic pH. An efficient particulate control device (electrostatic precipitator) is required to prevent increased particulate emissions when burning TDF.^[131] A proper feed system to provide a consistent and well controlled TDF feed rate is recommended. Proper combustion air control on the boiler is required to ensure efficient combustion of the TDF.^[132]

Existing boilers can be modified to meet the requirement for such high temperatures; however these modifications, in addition to TDF processing, can be expensive depending on the model. Until the cost of processing and equipment are lowered the use of TDF will be limited.^[133]

6.3 Use of RDF by Cement Kilns

Cement is a fine grey powder that is mixed with gravel, sand, and water to form concrete, the most widely used construction material in the world. In 2008, the Canadian cement industry produced 14 million tonnes of cement, worth more than \$1.8 billion. Currently, there are 16 operating cement plants in Canada, with three of these located in BC.^[134]

The production of cement consumes a significant amount of raw materials and energy. For example, a dry process cement plant needs roughly 1,600,000 tonnes of raw materials and 150,000 tonnes of fuel (high quality coal) to produce 1,000,000 tonnes of Portland cement clinker per year.^[135] Due to the high consumption of natural resources used in cement production, the cement industry has for many years been investigating the use of alternative raw materials and fuels to help offset the consumption of natural resources without compromising the quality of the cement produced or increasing the environmental impact of cement manufacture.

The European cement industry has been increasingly substituting the use of natural resources for raw materials and fuels with alternative waste-derived materials in order to decrease the environmental impact of their operations. Often these alternative materials are selected industrial by-products and waste streams which have been found to be suitable for cement production due to their physical and chemical properties.

¹³¹ T.A.G. Resource Recovery (November 1997), *Tire Derived Fuel: Environmental Characteristics and Performance*, Retrieved February 23, 2010, from <http://www.p2pays.org/ref/24/23765.pdf>

¹³² L. Cross and B. Ericksen, *Use of Tire Derived Fuel (TDF) in a Fluidized Bed Hog Fuel Paper Boiler at Pacifica Papers Inc.*, Retrieved February 23, 2010, from http://www.portaec.net/local/tireburning/use_of_tire_derived_fuel.html

¹³³ Unknown Author, *Recycling Options*, Retrieved February 23, 2010, from <http://www.p2pays.org/ref/11/10504/html/biblio/htmls2/cgh4.html>

¹³⁴ The Cement Association of Canada. 2010. The Cement Association of Canada – Economic Contribution

¹³⁵ CEMBUREAU. 2004. The Sustainable Use of Alternative Resources in the European Cement Industry

Common alternative waste-derived raw materials used in cement manufacturing in Europe include fly ash, blast furnace slag, silica fume, iron slag, paper sludge, pyrite ash, spent foundry sand, soil containing oil and artificial gypsum (gypsum produced from industrial processes such as acid neutralization). These waste materials are suitable as they are chemically appropriate and provide the constituents required for the production of clinker.^[136]

Alternative waste-derived fuels are also commonly used in cement manufacture. The suitability of an RDF for use in a cement kiln as a fuel is contingent upon the material having the appropriate consistency, heat value and composition as follows:

- The particle size of the fuel is an important factor in determining the suitability of a fuel for use in a cement kiln. Fuels with a particle size of less than 12 mm are acceptable to be introduced directly into the kiln. Fuels with a particle size of less than 50 mm are acceptable to be injected into the precalciner for those facilities that include a precalciner in their design.
- Fuels with a calorific value ranging from 15 to 18 MJ/kg are more suitable to be introduced into the precalciner and fuel with a higher calorific value ranging from 20 to 25 MJ/kg are more suitable to be injected into the kiln.
- The composition of the fuels must be in the appropriate range in regards to moisture content, ash content, sulphur and chlorides as well as trace heavy metals.

In many jurisdictions where the use of alternative fuels has been well established, there are regulations/guidelines in place to regulate their use. The regulatory requirements/guidelines for the maximum levels of contaminants in alternative fuels from some of these jurisdictions are presented in Table 6-2, below. The focus is on regulating contaminants that could contribute to the emissions of chlorinated organic pollutants and heavy metals. It should be noted that generally the mass of chlorine and trace heavy metals within a cement kiln will be dominated by the contribution of these parameters from the raw materials used in cement manufacture. The contribution to the discharge of these contaminants from any fuel source is comparatively small.

Common alternative waste based fuels used in cement manufacturing industry^[137] in Europe are listed in Table 6-2.

¹³⁶ CEMBUREAU. 2006. Air emissions and alternative fuels in the European cement industry

¹³⁷ European Commission. 2009. Integrated Pollution Prevention and Control Draft Reference Document on Best Available Techniques in the Cement, Lime and Magnesium Manufacturing Industries

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Table 6-2: Alternative Fuels Regulatory Requirements/Guidelines for Cement Kilns

		Austria		Switzerland	Germany	Finland			Sweden		Lebanon	United States
		MSW (25 MJ/kg)	Plastic, paper, textile, wood waste	MSW - 25 MJ/kg	Plastic, paper, textile, wood waste	RDF Class I	RDF Class II	RDF Class III	Specialbränsle A	Lattbränsle		
Chlorine	%	1	2	–	1.5	0.15	0.50	1.5	1.0	1.0	–	–
Antimony (Sb)	mg/kg	5	20	5	120	–	–	–	–	–	–	50
Arsenic (As)	mg/kg	15	15	15	13	–	–	–	–	–	10	50
Beryllium (Be)	mg/kg	5	–	5	2	–	–	–	–	–	2	–
Cadmium (Cd)	mg/kg	2	27	2	9	1.0	4.0	5.0	10	5	5	40
Chromium (Cr)	mg/kg	100	300	100	250	–	–	–	300	30	–	200
Copper (Cu)	mg/kg	100	500	100	700	–	–	–	–	–	150	600
Lead (Pb)	mg/kg	200	500	200	400	–	–	–	350	100	100	500
Mercury (Hg)	mg/kg	0.5	2	0.5	1.2	0.1	0.2	0.5	–	5	1	20
Nickel (Ni)	mg/kg	100	200	100	160	–	–	–	–	10	50	50
Thallium (Tl)	mg/kg	3	10	3	2	–	–	–	–	–	2	40
Tin (Sn)	mg/kg	10	70	10	70	–	–	–	–	–	70	100
Vanadium (V)	mg/kg	100	–	100	25	–	–	–	–	50	20	50
Zinc (Zn)	mg/kg	400	–	400	–	–	–	–	2000	–	–	1000

NOTE:

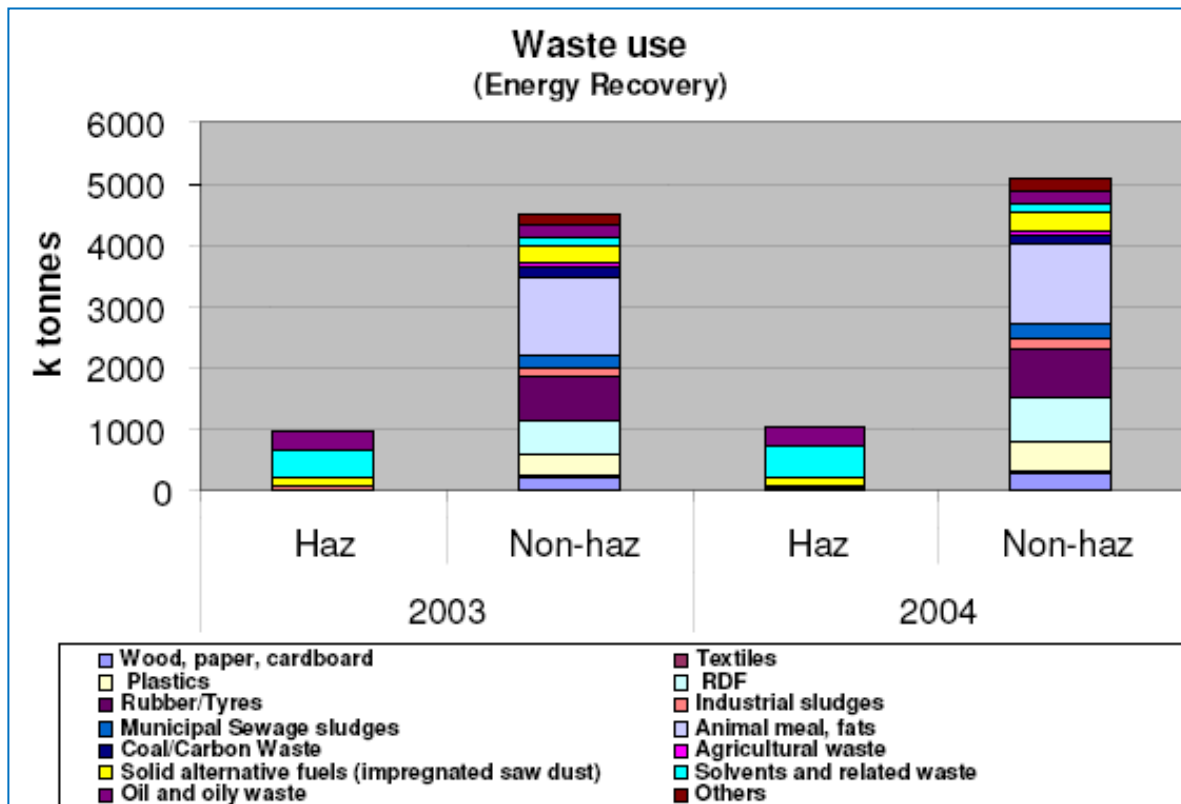
“–” indicates that no regulated value has been set for that parameter by that jurisdiction.

Table 6-3: Types of Alternative Fuels Used in the European Cement Industry

Types of Waste Fuels (Hazardous and Non-Hazardous)	
Wood, paper, cardboard	Municipal sewage sludge
Textiles	Animal meal, fats
Plastics	Coal/carbon waste
Processed MSW fractions (e.g., RDF)	Agricultural waste
Rubber/tires	Solid waste (impregnated sawdust)
Industrial Sludge	Solvents and related waste
Oil and oily waste	

Figure 6-1 illustrates the consumption of different types of hazardous and non-hazardous waste used as fuel in cement kilns in the EU-27 in 2003 and 2004.

Figure 6-1: Consumption of Different Types of Hazardous and Non-hazardous Waste Used as Fuels in Cement Kilns in the EU-27



Source: European Commission. 2009. Integrated Pollution Prevention and Control Draft Reference Document on Best Available Techniques in the Cement, Lime and Magnesium Manufacturing Industries

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Characteristics of the cement production process lend itself to beneficial waste-to-energy and material recycling applications. The following is a list of characteristics of cement production which lend it to the beneficial use of waste materials as fuel:

- Maximum temperatures of approximately 2,000°C (main firing system, flame temperature) in rotary kilns
- Gas retention times of about 8 seconds at temperatures above 1,200°C in rotary kilns
- Material temperatures of about 1,450°C in the sintering zone of the rotary kiln
- Oxidising gas atmosphere in the rotary kiln
- Gas retention time in the secondary firing system of more than two seconds at temperatures of above 850°C; in the precalciner, the retention times are correspondingly longer and temperatures are higher
- Solids temperatures of 850°C in the secondary firing system and/or the calciner
- Uniform burnout conditions for load fluctuations due to the high temperatures at sufficiently long retention times
- Destruction of organic pollutants due to achievement of high temperatures at sufficiently long retention times
- Sorption of gaseous components like HF, HCl, SO₂ on alkaline reactants
- High retention capacity for particle-bound heavy metals
- Short retention times of exhaust gases in the temperature range known to lead to 'de novo-synthesis' of dioxins and furans
- Complete utilization of fuel ashes as clinker components and hence, simultaneous material recycling (e.g., also as a component of the raw material) and energy recovery
- Product specific wastes are not generated due to a complete material utilization into the clinker matrix; however, some cement plants in Europe dispose of bypass dust
- Chemical-mineralogical incorporation of non-volatile heavy metals into the clinker matrix.^[138]

Emissions control in cement kilns is largely based on the use of bag houses to capture particulate matter from the flue gas (which also controls emissions of most heavy metals as discussed below). More modern facilities or retrofitted plants may be equipped with NO_x control, specifically SNCR. Emissions of other parameters such as POPs or acid gases are generally controlled through the operating characteristics of cement facilities as noted above. Monitoring of cement plant emissions generally includes CEMs (for parameters such as NO_x, SO_x, CO, TOC etc.) which serve a dual purpose in both monitoring emissions and determining if the facility is operating appropriately within the parameters required to manufacture quality cement product. Periodic stack testing is usually also required both to ensure effective calibration of the CEMs and to establish performance against regulated ELVs for a broader range of parameters.

¹³⁸ European Commission. 2009. Integrated Pollution Prevention and Control Draft Reference Document on Best Available Techniques in the Cement, Lime and Magnesium Manufacturing Industries

The impact on emissions from cement manufacturing due to the use of waste materials as alternative fuels or alternative raw materials is relatively minor. The following bullet list summarizes the assumed impacts as outlined by the European Commission.^[139]

- Dust emissions remain unaffected by using wastes.
- The use of suitable waste has only a minor influence on metal emissions due to the high retention of metals in the finished product. Non-volatile metals tend to be bound almost entirely in the clinker matrix. Semi-volatile metals such as lead or cadmium tend to be captured in the clinker stream or in dust. Highly volatile metals such as mercury and thallium tend to be of greater concern as they tend to vapourize and leave the kiln system. For this reason, it is important to limit the amount of highly volatile metals in the waste being used.
- NO_x, HCl, HF, SO₂, CO, and TOC are largely unaffected.
- The combustion conditions in rotary kiln systems ensure low emissions concentrations of dioxins and furans. The biggest factor impacting these emissions is what location waste materials are fed into the system (i.e., wastes that are fed into the main firing system tend to reach high enough temperatures and retention times to limit dioxin/furan emissions while wastes fed into the secondary firing zone may not reach high enough temperatures or long enough retention times).

Table 6-4 provides an example of the impact that utilizing waste as a fuel source could have on the emission profile from a typical cement kiln. Note: while the report cited does not specify the original sources of the waste in each application, RDF generation in Germany is generally derived from processing MSW materials (not including specialized waste streams such as construction/demolition material). Also it should be noted that while the monitoring approach for each parameter is not noted, cement kilns in the EU and North America typically use CEMs for parameters such as SO_x and NO_x and periodic stack testing for other parameters (PAHs, metals). As the table illustrates, utilizing waste as a fuel has a minimal impact on the emissions released from the plant, with some parameters decreasing and others increasing within the same order of magnitude.^[140]

Table 6-4: Emission Profile from a Cement Kiln Using RDF

Parameter	Measure	Individual Measurements	
		No Utilization of Wastes	Utilization of Wastes
Total Particulate	mg/m ³	2.8 – 12.9	12.0 – 15.9
HCl	mg/m ³	0.88 – 5.93	0.87 – 1.32
SO _x	mg/m ³	714 – 878	311 – 328
HF	mg/m ³	0.13 – 0.23	0.02 – 0.04
NO _x	mg/m ³	789 – 835	406 – 560

¹³⁹ European Commission. 2009. Integrated Pollution Prevention and Control Draft Reference Document on Best Available Techniques in the Cement, Lime and Magnesium Manufacturing Industries

¹⁴⁰ UBA. 2001. Draft of a German Report with basic information for a BREF-Documents "Waste Incineration". Umweltbundesamt

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Parameter	Measure	Individual Measurements	
		No Utilization of Wastes	Utilization of Wastes
Total C	mg/m ³	11.7 – 23.2	5.7 – 7.1
PAHs	mg/m ³	–	0.0026
Benzene	mg/m ³	0.27 – 0.54	0.45 – 0.55
Cd	mg/m ³	<0.005	<0.007
Tl	mg/m ³	<0.005	<0.005
Hg	mg/m ³	0.014 – 0.044	0.003 – 0.006
Sum of Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V, Sn	mg/m ³	<0.3	<0.5
PCDD/PCDF, I-TEQ	mg/m ³	0.001 – 0.002	0.005 – 0.0065

The following sections provide an overview of the regulatory framework governing the use of waste as a raw material or alternative fuel in cement kilns in Ontario and the European Union.

6.3.1 Regulatory Approach in Ontario

Guideline A-7 (October 2010) applies to all thermal treatment facilities processing municipal waste including manufacturing facilities such as cement and lime kilns, if they use municipal waste as an alternative fuel. The Guideline sets out specific in-stack emission limits for cement and lime kilns which take into account operational differences for these facilities as compared to other “dedicated” thermal treatment facilities (see the following table).

Table 6-5: Emission Limits for Existing Cement and Lime Kilns Burning Municipal Waste (Guideline A-7)

Parameter	In-Stack Emission Limit	Verification of Compliance ⁷
Particulate Matter (PM)	50 mg/Rm ³ or a site specific emission limit where a more stringent stack concentration limit is already in place for existing raw materials and conventional fuels ¹	Results from compliance source testing or calculated as the rolling arithmetic average of four (4) hours of data measured by a continuous emission monitoring system that provides data least once every fifteen minutes
Cadmium (Cd)	7 µg/Rm ³ unless existing raw materials and conventional fuels result in higher concentration ²	Results from compliance source testing (periodic stack testing)
Lead (Pb)	60 µg/Rm ³ unless existing raw materials and conventional fuels result in higher concentration ²	Results from compliance source testing (periodic stack testing)
Mercury (Hg)	20 µg/Rm ³ unless existing raw materials and conventional fuels result in higher concentration ²	Results from compliance source testing (periodic stack testing) or calculated as the rolling arithmetic average of 24 hours of data measured by a continuous emission monitoring system that provides data at least once every 15 minutes

Parameter	In-Stack Emission Limit	Verification of Compliance ⁷
Dioxins and Furans	80 pg/Rm ³	Results from compliance source testing (periodic stack testing); results expressed as I-TEQ
Hydrochloric Acid (HCl)	18 ppm _{dv} (27 mg/Rm ³) unless existing raw materials and conventional fuels result in higher concentration ³	Calculated as the rolling arithmetic average of 24 hours of data measured by a continuous emission monitoring system that provides data at least once every 15 minutes
Sulphur Dioxide (SO ₂)	Site specific limit not to exceed the in-stack SO ₂ concentration resulting from existing raw materials and conventional fuels. ^{4,6}	Calculated as the rolling arithmetic average of 24 hours of data measured by a continuous emission monitoring system that provides data at least once every 15 minutes
Nitrogen Oxides (NO _x)	Site specific limit not to exceed the in-stack NO _x concentration resulting from existing raw materials and fossil fuels ^{5,6}	Calculated as the rolling arithmetic average of 24 hours of data measured by a continuous emission monitoring system that provides data at least once every 15 minutes
Organic Matter	Section 50 of Ontario Regulation 419/05	Calculated as the rolling arithmetic average of 10 minutes of data measured by a continuous emission monitoring system that provides data at least once every minute
Opacity	Section 46 of Ontario Regulation 419/05	Calculated as the rolling arithmetic average of six (6) minutes of data measured by a continuous opacity monitor that provides data at least once every minute

NOTES:

- 1) If there is no limit for particulate matter in an existing Certificate of Approval issued to the facility, the limit of 50 mg/Rm³ can be expected to be included in the Certificate of Approval that will allow burning of municipal waste as an alternative fuel. Where a more stringent site-specific limit for particulate matter is already incorporated into an existing Certificate of Approval for manufacturing of cement or lime using existing raw materials and conventional fuels, the existing limit will be retained if it is more stringent than 50 mg/Rm³.
- 2) Limits for cadmium, lead and mercury can be expected to be included in a Certificate of Approval that will allow burning of municipal waste as an alternative fuel, unless the proponent can demonstrate that one or more of the specified metals are present in the existing raw materials and conventional fuels in such a quantity that the relevant limit(s) would be exceeded without the use of municipal waste as a fuel. In such a case, site-specific limits for one or more of the above metals may be established and incorporated into a Certificate of Approval. The site specific limits can be expected to be developed based on a review of relevant facility specific data that includes information on the discharge of cadmium, lead, and/or mercury from the facility (e.g., source testing data, analytical data for raw materials, mass balance calculations). Such site specific limits will take into account the variability of the raw material composition.
- 3) It is expected that cement and lime kilns can comply with the hydrogen chloride (HCl) limit. A site-specific emission limit for HCl may, however, be incorporated into a Certificate of Approval based on HCl concentrations when using existing raw materials and conventional fuels. This will prevent any increase in HCl emissions resulting from use of municipal waste as fuel for the kiln.
- 4) A site-specific emission limit for sulphur dioxide (SO₂) can be expected to be incorporated into a Certificate of Approval based on SO₂ concentrations when burning conventional fuels. This will prevent any increase in SO₂ emissions resulting from use of municipal waste as fuel for the kiln. For kilns required to use continuous emission monitoring (or a method that will provide estimates of emissions that are at least as accurate as the estimates that would be provided by a continuous emission monitoring system) for SO₂ under Ontario Regulation 194/05 (Industry Emissions – Nitrogen Oxides and Sulphur Dioxide), the limit will be determined based on a review of a minimum of 6-months of Continuous Emission Monitoring System (CEMS) data (or data obtained using another method) for the kiln (1-hour, 24-hour and 30-day SO₂ averages in ppm_{dv} or mg/Rm³). The Ministry will continue to monitor the development of SO₂ control technology worldwide. As new proven technology is developed suitable for this industry sector, the Ministry will review this guideline to determine if limits can be adjusted.

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- 5) A site-specific emission limit for oxides of nitrogen (NO_x) can be expected to be incorporated into a Certificate of Approval based on NO_x concentrations when burning conventional fuels. This will prevent any increase in NO_x emissions resulting from use of municipal waste as fuel for the kiln. For kilns required to use continuous emission monitoring (or a method that will provide estimates of emission that are at least as accurate as the estimates that would be provided by a continuous emission monitoring system) for NO_x under Ontario Regulation 194/05, the limit will be determined based on a review of a minimum of 6-months of CEMS data (or data obtained using another method) for the kiln (1-hour, 24-hour and 30-day NO_x averages in ppm_{dv} or mg/Rm³). The Ministry will continue to monitor the development of NO_x control technology worldwide. As new proven technology is developed suitable for this industry sector, the Ministry will review this guideline to determine if limits can be adjusted.
- 6) Lime kilns that do not currently have CEMS for SO_2 , and NO_x , can be expected to carry out a monitoring program to determine the normal ranges for the parameters when burning conventional fuels. The proponent of an alternate fuel should consult staff of the Ministry when planning such a program. The results of the monitoring program are expected to be included with an application for a Certificate of Approval to burn municipal waste as an alternate fuel.
- 7) Compliance source testing as set out in the facility's Certificate of Approval. Owners and operators of cement and lime kilns can expect to be required, by conditions in Certificates of Approval, to maintain CEMS for SO_2 , NO_x , THC, HCl and opacity.

The approach used in Ontario clearly acknowledges that it is not reasonable to apply exactly the same ELVs to cement or lime kilns that use a waste derived fuel. Rather the approach that is taken applies the same stack limits applied to WTE facilities, for parameters that are directly associated with fuel quality (e.g., heavy metals, POPs) but not for emission parameters that are driven largely by the primary purpose and design of the facility (SO_x , NO_x , PM). For some heavy metals (mercury, cadmium and lead) it is also recognized that the contribution from the raw material stream for some of these trace metals can be more significant than from the fuels, and in those cases site specific ELVs are set.

In order to use RDF as a fuel in Ontario, industrial facilities have to apply for or amend their operating permits (certificates of approval) issued under the *Environmental Protection Act* (EPA). The permitting/application process generally involves the following:

- Fuel testing and comparison of the RDF fuel quality against the conventional fuels. Mass balance analyses are generally used to establish any potential shift in emissions concentrations that could result from the use of the fuels.
- Determination of the appropriate RDF feed rate, based on the outcome of the analysis above and based on review of the impact of various fuel characteristics (e.g., heat value).

The approach used for proposed RDF applications has been to encourage and permit the use of RDF for a fuels test/trial run, the results of which are used to demonstrate that RDF can be used within the current ELVs established for the facility and/or to determine site specific ELVs for various parameters that would apply during regular use of the RDF.

6.3.2 European Union

As noted previously, the use of waste fuels in the manufacture of cement is commonly practiced in Europe. On average, alternative fuels were substituted for 17% of conventional fuels in the

manufacture of cement in EU-23 countries (in 2007). This rate of substitution is equivalent to saving about 4 million tonnes of coal.^[141] For some facilities, the rate of substitution can be as high as 100%.

Two directives apply to the use of waste in cement manufacturing in the EU, namely the Integrated Pollution Prevention and Control Directive (Directive 2008/1/EC) and the Waste Incineration Directive (Directive 2000/76/EC).

The IPPC Directive applies to installations for the production of cement clinker in rotary kiln with a production capacity exceeding 500 tonnes per day.^[142] As discussed previously, the IPPC is aimed at minimizing the emissions of pollutants from large industrial installations through the use of an environmental permit. Permits contain emission limit values (ELVs) and set conditions based on the application of best available technology (BAT). The permits also address energy efficiency, waste minimization, prevention of accidental emissions, and site restoration.^[143] If a cement manufacturing operation uses waste derived fuel or raw materials derived from waste, the facility would still be required to emission limit values (ELVs) set out in its permit.

In May, 2009, the European Commission released a draft reference document on the best available techniques in the cement, lime, and magnesium oxide manufacturing industries. The document goes into considerable detail concerning the use of waste as alternative raw material and fuel in cement manufacturing. The following table (Table 6-6) provides a summary of the best available techniques for the cement industry relating to the use of wastes.^[144]

Table 6-6: Summary of BAT for the Cement Industry Relating to the Use of Wastes

Safety management for the use of hazardous waste materials	<ul style="list-style-type: none"> Apply safety management for the handling, e.g., storage, and/or feeding of hazardous waste materials, such as using a risk based approach according to the source and type of waste, for the labelling, checking, sampling and testing of waste to be handled
Waste Quality Control	<ul style="list-style-type: none"> Apply quality assurance systems to guarantee the characteristics of wastes and to analyse any waste that is to be used as raw material and/or fuel in a cement kiln for parameters/criteria (constant quality, physical criteria, chemical criteria). Control the amount of relevant parameters for any waste that is to be used as raw material and/or fuel in a cement kiln, such as chlorine, relevant metals (e.g., cadmium, mercury, thallium), sulphur, total halogen content Apply quality assurance systems for each waste load

¹⁴¹ CEMBUREAU. 2006. 2004 and 2005 statistics on the use of alternative fuels and materials in the clinker production in the European cement industry

¹⁴² EEF: Integrated Pollution Prevention and Control (IPPC). 2009. [http://www.eef.org.uk/policy-media/policy-briefs/briefings/Integrated-Pollution-Prevention-Control-\(IPPC\).htm](http://www.eef.org.uk/policy-media/policy-briefs/briefings/Integrated-Pollution-Prevention-Control-(IPPC).htm)

¹⁴³ EEF: Integrated Pollution Prevention and Control (IPPC). 2009. [http://www.eef.org.uk/policy-media/policy-briefs/briefings/Integrated-Pollution-Prevention-Control-\(IPPC\).htm](http://www.eef.org.uk/policy-media/policy-briefs/briefings/Integrated-Pollution-Prevention-Control-(IPPC).htm)

¹⁴⁴ European Commission. 2009. Integrated Pollution Prevention and Control Draft Reference Document on Best Available Techniques in the Cement, Lime and Magnesium Manufacturing Industries

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Waste feeding into the kiln	<ul style="list-style-type: none"> Use the appropriate feed points to the kiln in terms of temperature and residence time depending on kiln design and kiln operation Feed waste materials containing organic components that can be volatilised before the calcining zone into the adequately high temperature zones of the kiln system Operate in such a way that the gas resulting from the co-incineration of waste is raised in a controlled and homogeneous fashion, even under the most unfavourable conditions, to a temperature of 850°C for two seconds Raise the temperature to 1,100°C, if hazardous waste with a content of more than 1% of halogenated organic substances, expressed as chlorine, is co-incinerated Feed wastes continuously and constantly Stop co-incinerating waste for operations such as start-ups and/or shutdowns when appropriate temperatures and residence times cannot be reached
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The IPPC Directive also provides BAT for emissions limits from cement manufacturing. The following table provides the emissions limit values as laid out in the document.

Table 6-7: BAT Emissions Limits for Cement Manufacturing in the IPPC Directive

Contaminant	Concentration Units	Integrated Pollution Prevention and Control Directive (2008/1/EC)
Total Particulate Matter (TPM) ¹	mg/Nm ³	<10 – 20
Hydrogen Chloride (HCl)	mg/Nm ³	10
Sulphur Dioxide (SO ₂)	mg/Nm ³	<50 – <400 ⁴
Hydrogen Fluoride (HF)	mg/Nm ³	1
Nitrogen Oxides (NO _x) (pre-heater kilns)	mg/Nm ³	<200 – 4,502 ³
Nitrogen Oxides (NO _x) (lepol and long rotary kilns)		400 – 800
Mercury (Hg) ⁶	ug/Nm ³	<0.05
Cd + Tl ⁶	ug/Nm ³	<0.05
Sum (Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V) ⁶	ug/Nm ³	<0.5
PCDD/F TEQ (I) (Dioxins and Furans) ⁵	ng/Nm ³	<0.05 – 0.1

NOTES:

Under the following conditions: 273 K, 101.3 kPa, 10% Oxygen, Dry Gas. Daily average values unless otherwise noted.

¹ Dust emissions from kiln firing processes – when applying a fabric filter or new or upgraded ESP, the lower level is achieved.

² BAT-AEL is 500 mg/Nm³, where after primary measures/techniques the initial NO_x level is >1000 mg/Nm³

³ Existing kiln system design, fuel mix properties including waste, raw material burnability can influence the ability to be in the range. Levels below 350 mg/Nm³ are achieved at kilns with favourable conditions. The lower value of 200 mg/Nm³ has only been reported as monthly average for three plants (easy burning mix used)

⁴ Range takes into account the sulphur content in the raw materials

⁵ Average over the sampling period (6 – 8 hours)

⁶ Average over the sampling period spot measurement, for at least half an hour.

The Waste Incineration Directive also applies to cement manufacturing facilities that utilize waste as a feedstock. The WID defines cement facilities that utilize waste as “co-incineration” plants. A “co-

incineration plant” is defined in the Directive as any stationary or mobile plant whose main purpose is the generation of energy or production of material products and:

- Which uses waste as a regular or additional fuel, or
- In which waste is thermally treated for the purpose of disposal.

The Directive states that no “co-incineration plant” shall operate without a permit from the appropriate governing agency. The permit must outline a number of specific parameters including ensuring that cement facility is properly designed and is using the appropriate equipment. Further, the permit must list the categories of waste to be treated and the quantities of waste to be treated, include the total waste co-incinerating capacity of the plant, and specify the sampling and measurement procedures to satisfy the obligations imposed for periodic measurements of each air and water pollutants.

If the cement facility is to treat hazardous materials, the permit has to also outline the quantities of different categories of hazardous waste that may be treated and the minimum and maximum mass flows of those hazardous wastes, their lowest and maximum calorific values and their maximum concentration of pollutants (e.g., PCB, chlorine, heavy metals).

The Directive also provides guidance concerning the reception and delivery of waste at the facility so as to limit the effects on the environment and direct risks to human health. It states that the facility operator shall determine the mass of each category of waste prior to accepting the material on site. For hazardous waste, the facility should obtain the physical and as far as practicable chemical composition of the waste as well as the hazardous characteristics of the waste.

The Directive goes on to state that co-incineration plants need to be designed and operated in such as way that waste is treated at a temperature of 850°C for two seconds, (or 1,100°C if the waste has more than 1% of halogenated organic substances) which is the same requirement for a regular waste incineration plant.

The air emissions limit values set out in the Directive for co-incineration plants are slightly different than those set out for incineration plants. The co-incineration plant must be designed, equipped, built and operated in such as way that the emission limit values set out in the following table are not exceeded in the exhaust gas. The primary difference in the WID in regards to emissions from co-incineration plants is that the ELV for NO_x is set significantly higher than that for WTE facilities.

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Table 6-8: Emissions Limit Values for Cement Kilns in the Waste Incineration Directive

Contaminant	Concentration Units	Waste Incineration Directive (2000/76/EC)
Total Particulate Matter (TPM)	mg/m ³	30
Hydrogen Chloride (HCl)	mg/m ³	10
Sulphur Dioxide (SO ₂) ¹	mg/m ³	50
Hydrogen Fluoride (HF)	mg/m ³	1
Nitrogen Oxides (NO _x) (existing plants)	mg/m ³	800
Nitrogen Oxides (NO _x) (new plants)		500
TOC ¹	mg/m ³	10
Mercury (Hg)	µg/m ³	0.05
Cd + Tl	µg/m ³	0.05
Sum (Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V)	µg/m ³	0.5
PCDD/F TEQ (I) (Dioxins and Furans)	ng/m ³	0.1

NOTES:

Under the following conditions: 273 K, 101.3 kPa, 10% Oxygen, Dry Gas

¹ Exemptions may be authorized by a competent authority in cases where these emissions do not result from the incineration of waste

6.4 Proposed Regulatory Approach for RDF

The preceding sections discuss use of RDF by two industrial sectors, for which there is current and general interest in the use of alternative solid fuel materials. Pulp mill/wood fired boilers and cement kilns are not the only industrial sectors where there could be future interest in the use of RDF for co-firing or co-incineration. A consistent regulatory approach that addresses use of RDF by any industry sector is required.

Reviewing the regulatory approach applied in various jurisdictions to the use of RDF as a fuel for co-firing or co-incineration along with current experience with RDF applications in BC, indicates that a reasonable approach to mitigating the risk associated with the use of waste derived fuels would consist of the following:

- Generally when looking across the spectrum of RDF use in co-combustion (some examples of which are discussed above) the RDF usually has the same general characteristics as the conventional fuels used by the facilities. For example, wood fired boilers generally use RDF that is similar in composition (e.g., primarily cellulosic) to conventional wood waste. Cement kilns use a wide range of RDF fuels including waste plastics, given that the conventional fuels used by these facilities are fossil fuel based.

- It would be reasonable to define which waste materials are considered 'waste derived' fuels which would require major modifications and permit amendments, and those that would be considered equivalent to current fuels. For example, as discussed above, it would be reasonable to set a definition for 'clean' wood waste that could be separated from construction and demolition waste for use in wood fired boilers as part of their regular fuel stream and 'contaminated' wood waste that would require major modifications and permit amendments. The BC MOE should develop definitions and potentially RDF fuel specifications similar to those used in other jurisdictions relative to RDF for cement applications. These definitions/specifications and/or proponent driven specifications would be set out in the amended air emission permits.
- Testing of RDF will be required generally either to demonstrate compliance with a regulatory limit for fuel quality and/or to ensure that the fuel falls within the range of specifications required to ensure that the material can be used without compromising the operations of the facility proposing to use RDF as a full or partial fuel substitute. The results of fuel tests would be reported in the application process for regulatory approval, and compared against the quality of the conventional fuels used at the facility. These results could be used to determine through a mass balance analysis if the contribution of parameters in the RDF would result in a shift in emissions concentrations if the RDF was used (e.g., presence of chlorine shifting the emissions concentration of HCl).
- Fuel trials should be undertaken to demonstrate that the proposed RDF can be effectively used as fuel, and to establish site/facility specific ELVs where applicable. Fuel trials will also allow for the facility operator to review standard operations and to determine the appropriate adjustments needed to use RDF effectively as a fuel. Fuel trials should reflect the proposed RDF substitution rates, so that the proponent can demonstrate how at the maximum proposed fuel substitution rate the facility will comply with current and/or proposed ELVs.
- Generally within the air emission permits, the same stack limits (ELVs) would be applied to industrial facilities that use RDF as would be applied to WTE facilities (as set out in Section 8.3), for parameters that are directly associated with fuel quality (e.g., heavy metals, POPs) but not for emission parameters that are driven largely by the primary purpose and design of the facility. For wood fired boilers, design parameters would include parameters such as NO_x and CO, while for cement kilns this would include a broader spectrum of parameters (SO_x, NO_x, CO, TOC, particulates) that are driven by raw material quality and standard facility design.
- Once permitted, facilities would have to implement quality assurance systems to guarantee the characteristics of the RDF and to analyze the RDF for key parameters/criteria including consistency, physical criteria (related to suitability for use at the facility) and chemical criteria (related to ELV compliance). Generally, RDF would have to be tested at random at least quarterly within the first few years of operation. Results from the quality assurance systems would be included with in annual compliance reporting.

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7 ASSOCIATED COSTS AND ENERGY EFFICIENCY

This section investigates the capital and operating costs for WTE and discusses the energy efficiency associated with WTE facilities and potential revenues associated with energy recovery.

7.1 Capital Expenditure and Operating Costs

This subsection provides a summary of current capital and operating costs for the majority of thermal treatment technologies. These are expressed as capital cost per annual design tonne (commonly used for capital cost comparison) and operating costs per annual design tonne. The data presented is based on financial information from jurisdictions in which thermal treatment approaches have been implemented and financial information made available directly from technology vendors.

The range of capital and operating costs reported by individual vendors are influenced by the unique circumstances associated with siting a facility, such as jurisdictional constraints, size of facility, and the form in which the energy is recovered and used. This summary therefore includes:

- i. The potential range of order of magnitude costs, identifying the key factors for both the low and high end of the range and the median values for both capital and operating costs for various technologies.
- ii. Where available, the cost differentials between these technologies and the factors which contribute to these differences.
- iii. Costs specifically associated with the applicable emissions control and/or thermal process control options.

Identification of costs in a North American context can be quite difficult. Few new facilities have reached the stage of development in either Canada and the USA and for proposed facilities, either the financial information is proprietary (particularly if the proposed facility is intended to be owned/operated by a private sector entity) or may not be based on guaranteed pricing through formal procurement processes.

Implementation of projects in North America can be based on a variety of contractual arrangements, each of which has the potential to affect the potential costs and allocation of risk between the technology vendor and the owner/operator of the plant. Some of the typical contractual arrangements for such facilities include:

- **Design/Build:** the intended owner/operator (e.g., municipality) seeks pricing for design and construction of the facility. In such a context the majority of the risk is borne by the owner/operator.
- **Design/Build/Operate:** the intended owner seeks a contract from a technology vendor (usually consortium representing proprietary technology vendors, construction firms and an operating entity) to design and build the facility and to operate the plant for a fixed period of time. Often the owner passes on some of the risk associated with the facility through performance guarantees that have to be met by the preferred vendor.

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- **Design/Build/Finance/Operate (P3):** the intended owner seeks a contract similar to that noted above, wherein the vendor also holds a financing role, seeking return on the investment in the capital cost for the facility over a longer contractual period. Generally, there is increased sharing of risk and concomitant increases in overall unit costs.
- **Design/Build/Own/Operate:** the party requiring capacity for WTE seeks pricing for the use of WTE capacity that is entirely owned/operated/financed by the vendor. These arrangements can be coupled with the provision of some assistance in the form of siting, provision of infrastructure etc. between the parties. Generally long-term fixed “put or pay” contracts are necessary to guarantee revenues to the vendor. Such contracts guarantee that the vendor will receive a set minimum revenue value, associated with a set minimum waste supply. Should the generator not have sufficient waste supply, it is still required to pay the vendor the set minimum fee. Also, generally the unit cost for use of the WTE capacity would be higher given that the risk is almost entirely borne by the vendor.

The potential capital and operating costs and net costs can vary significantly for all WTE technologies as noted in the range of order of magnitude costs as discussed below. Factors that affect the range of costs for conventional combustion as noted below could also be considered to affect the costs for the other technologies as the same considerations would apply.

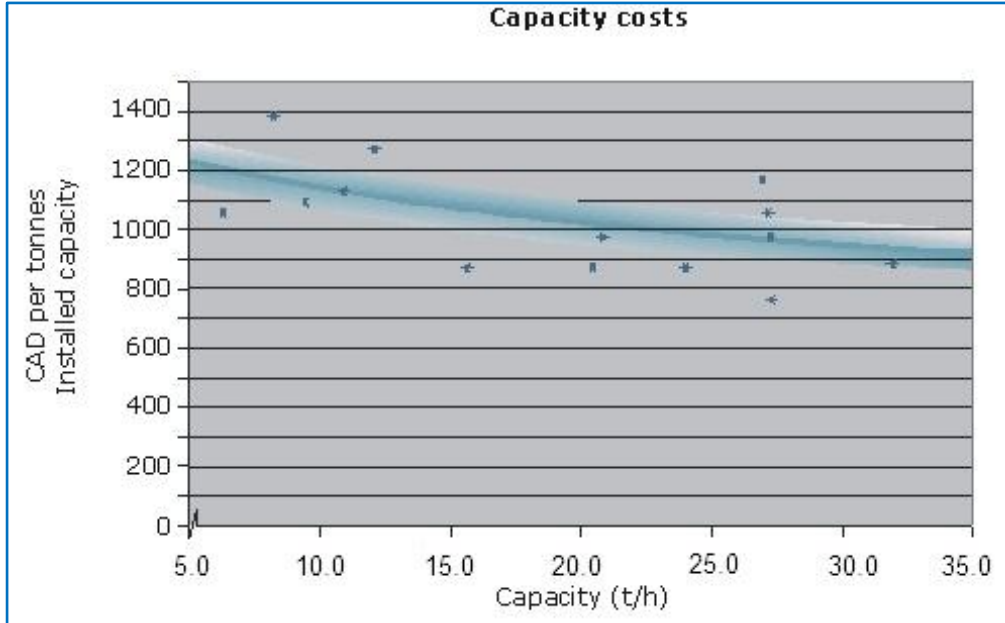
7.1.1 Range of Order of Magnitude Costs

In Figure 7-1, the effect of the size of the WTE plant on the capital costs per tonne of waste are illustrated. The curve shown is based on known capital costs for a wide range of new European Energy from Waste lines, in which Ramboll has been involved during the last 10 years. The background data from 14 European Energy from Waste plants is shown as dots (stars) on the Figure. The background data are actual capital costs adjusted to 2006 price level.

As seen from Figure 7-1, the capital costs per tonne of waste based on European price level are generally \$900 – \$1,200 per tonne of installed capacity. The capital costs between a small (5 tph) and a large (30 tph) incineration plant differs by about 25% (on a cost per throughput tonne basis).

The background data indicated on Figure 7-1 shows that the capital costs differs significantly even for plants of similar size and erected in the same country. This variation indicates that when looking at a preliminary overall level, the capital costs for WTE plants can only be roughly estimated.

Figure 7-1: Comparison of Capital Costs for WTE Facilities per Installed Capacity



Source: Ramboll. 2007. Memo to MacViro during the Durham/York Environmental Assessment

It should be noted that the capital costs noted exclude the purchase of a site and exclude external infrastructure like roads, water, electricity/grid connections, etc. outside the premises of the site.

The capital costs can be split into different components. In Table 7-1 the total capital costs are split into five main components or parts. For each main component, the percentage of the total capital costs related to the specific component is shown. The proposed distribution of capital costs between the different components is based on the general experience with the European market. Of course large variations within the distribution of capital costs between the different main components are foreseen. Furthermore, there might be some differences between the North American market and the European market which will influence the distribution of the total capital costs between the different components/parts. However, the shown distribution can be generally assumed.

Table 7-1: General Distribution of WTE Total Capital Costs

Component	Percentage of Capital Costs
Thermal processing equipment (incinerator/boiler)	40%
Energy production equipment (turbines and generators)	10%
APC system (flue gas treatment)	15%
Building (civil works)	25%
Miscellaneous (approvals, general site works, ash processing, electrical transmission and interconnect etc.)	10%

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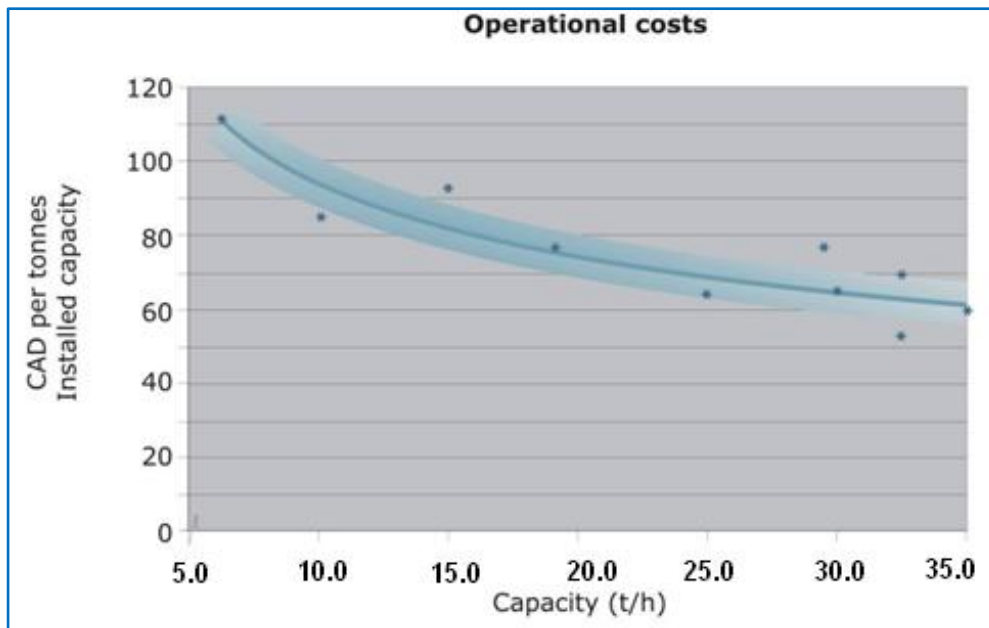
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When estimating the operational costs of WTE facilities, the size of the plant will influence the total costs in the same way as indicated for the capital costs. Furthermore, the total operational costs are to a large extent affected by local conditions such as local labour costs and the cost of consumables.

In Figure 7-2, the effect of the size of the plant on the operational costs per tonne of waste is illustrated. The curve shown is based on estimated operational costs for a range of new or planned European Energy from Waste lines. It is important to be aware that the background data are mainly estimates based on local conditions related to each plant. This includes e.g., type of APC-system, transport-expenses for the residues and the degree of automation of the plant (cranes, gate-control and weighting of incoming waste and outgoing residues etc.). It should also be noted that Figure 7-2 provides information related to gross operating costs, and thus does not take into account income from the sale of energy.

Figure 7-2: Range of Operational Costs for WTE Facilities in the EU



Source: Ramboll. 2007. Memo to MacViro during the Durham/York Environmental Assessment

As seen from Figure 7-2, the operational costs per tonne of waste based on European price level are generally \$60 – \$90 per tonne of installed capacity. The operational costs between a small (6 tph) and a large (35 tph) incineration plant differs by almost 50% (on a cost per throughput tonne basis). The operational costs can be split into different components as indicated:

- Labour and administration 25 – 30%
- Maintenance 35 – 40%
- Utilities and supplies – 20%
- Residues (management and disposal) – 20%.

The following sections discuss the potential range in capital and operating costs for the various WTE technologies presented in Section 2, based on publicly reported data.

7.1.1.1 Conventional Combustion

Capital costs were estimated based on five project estimates found during the literature review.^{[145],[146],[147],[148],[149]} The highest reported cost was \$1,684/annual design tonne (assuming a 200,000 tonne per year capacity where applicable, 2009 CDN\$). The lowest reported cost was \$640/annual design tonne. The median capital cost was \$771/annual design tonne with a standard deviation of 50.27%. For new WTE facilities a certain, and often high capital cost is incurred to improve the architectural appearance of the plant and makes investment costs difficult to compare. In most European countries the APC plant is placed in a building, whereas in North America this has not been the common practice.

Operating costs were also calculated. The maximum reported operating cost/tonne was \$105 and the minimum was \$38.90. The median operating cost/tonne was \$64.09 with a standard deviation of 36%.

These reported values are within the range of true capital and operating costs incurred in many jurisdictions that have been guaranteed through contractual arrangements.

The differences in capital and operating costs between conventional combustion facilities often reflect the following:

- **Size:** economies of scale indicate that larger facilities tend to have lower capital and operating costs per annual design tonne. For example, an Austrian study found that based on recently incurred costs in other EU nations that as the throughput of a WTE facility tripled (from 100,000 to 300,000 tpy) the cost per tonne (operating and capital) for the front end systems to manage MSW and the back end systems to manage ash decreased by 28%.^[150] In regards to the firing system and boiler for such facilities, as the throughput doubled (from 75,000 to 150,000 tpy) the cost per tonne (operating and capital) decreased by 6%.
- **Configuration:** for example, a facility that includes a single 100,000 tpy mass burn combustion unit (boiler) would generally incur capital costs in the order of 25% (or more) less than a facility consisting of two 50,000 tpy combustion units.

¹⁴⁵ European Commission, Integrated Prevention and Control. 2006. Reference Document on Best Available Technology for Waste Incineration

¹⁴⁶ Confidential

¹⁴⁷ Jacques Whitford Stantec Limited. 2009. Durham/York Residual Waste Study Environmental Assessment

¹⁴⁸ MacViro. 2007. City of Ottawa REOI Report

¹⁴⁹ MacViro. 2007. County of Dufferin RFQ Process

¹⁵⁰ Austrian Federal Ministry of Agriculture and Forestry, Environment and Water Management –State of the Art for Waste Incineration Plants, Vienna November 2002

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- **Architecture/Design:** a facility in a high profile location that would warrant special architectural treatments, additional landscaping and other design elements to reduce visual impacts, would generally incur capital costs in the order of 25 to 50% more than a facility located in an industrial zone with minimal requirements to improve the appearance of the facility.
- **APC Train:** as discussed further in this section, there are differences in both capital and operating costs associated with the APC trains suitable for such facilities.
- **Availability of Local Infrastructure:** WTE facilities generally require access to good transportation networks, electricity supply and access to the grid, supply of natural gas, supply of potable water and wastewater services.
- **Potential for Energy Utilization:** the net cost per tonne for WTE facilities declines based on the increased ability of the facility to sell energy. Generally, facilities that are able to sell heat directly to the market incur the lowest range of net costs per tonne, followed by facilities that are able to market both electricity and heat, with the facilities that incur the highest net cost per tonne being those that are only able to sell electricity. Furthermore, costs are affected by infrastructure required to access those markets, and the market price for electricity and heat. In a Canadian context, the ability to market heat at the present time is limited given that there are few policies and little initiative as yet to support district heating schemes. The ability of a WTE to market heat improves with access to industrial users of heat (steam) and/or new commercial/industrial areas where infrastructure for district heating could be considered.
- **Market price for Energy:** the market for energy from WTE varies significantly across North America and the globe, and is affected by energy policy and other legislative initiatives. For example, in jurisdictions such as many Scandinavian nations that have an energy policy that discourages dependence on fossil fuels, higher prices for electrical and heat energy are the norm. Energy pricing in jurisdictions such as Ontario, does not recognize the value of the energy from WTE in the same fashion as that from renewable sources such as biomass, although 50% or more of the energy from such facilities is usually derived from the biomass portion of the waste stream.

Table 7-2 compares two potential WTE facilities under consideration in Ontario, one of which represents the proposed facility for Durham/York Regions and the other representing a plant under consideration elsewhere in the Province. There is a significant difference in the capital costs that have been identified to-date for the two facilities, and some difference in the potential operating costs, based on some of the key factors noted above.

Table 7-2: Comparison of Capital Costs for Two Mid-Size WTE Facilities

	Durham/York 140,000 tpy, Mass Burn	Potential 100,000 to 200,000 tpy Mass Burn
Identified Capital Cost (2009\$)	\$1,500 to 1,700 per annual design tonne	\$900 to \$1,000 per annual design tonne
Identified Annual Operating Cost (2009\$)	\$100 to \$110 per annual design tonne	\$80 to \$90 per annual design tonne
Major Differences in Design	Two-unit facility (two 70,000 tpy mass burn combustion units, potential for two additional 70,000 tpy units) APC designed to achieve BAT EU and Ontario A-7 specifications High-profile location, Clarington Energy Park. Extensive architectural and landscaping treatment. Option for sale of electricity and heat through district heating/cooling.	One-unit facility (one 100,000 tpy mass burn combustion unit, potential for twinning in future) APC designed to meet Ontario A-7 guidelines. Low profile location in existing heavy industrial zone. Minimal architectural treatment or landscaping required. No option for district heating/cooling.
Similarities in Design	Reasonable access to local infrastructure (site servicing). Primary focus on sale of electricity, potential price 8 cents per kwh.	Site is fully serviced. Focus on sale of electricity, potential price 8 cents per kwh.

7.1.1.2 Gasification of MSW

Capital costs were estimated based on ten project estimates found during the literature review.^{[151],[152],[153],[154],[155]} A summary of the reported capital and operating costs for gasification facilities is presented in Table 7-3.

Table 7-3: Summary of Reported Capital and Operating Costs for Gasification Facilities (2009\$ CDN)

	Capital Costs (Annual Design Tonne)	Operational Costs
Lowest Reported Cost	\$134	\$37.22
Highest Reported Cost	\$1,410 (200,000 tpy capacity)	\$117.67
Median Reported Cost	\$803 +/- 42%	\$61.08 +/- 46%

¹⁵¹ MOSA, 2009. Summary Report on Estimated Costs to Develop and Operate new Regional Waste Disposal and Processing Facilities

¹⁵² NYC, 2006. Focused Verification and Validation of Advanced Solid Waste Management Conversion Technologies

¹⁵³ URS, 2005. Los Angeles County, Conversion Technology Evaluation Report

¹⁵⁴ MacViro, 2007. City of Ottawa REOI Report

¹⁵⁵ MacViro, 2007. County of Dufferin, RFP Process

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The reported operating and capital costs are not based on those incurred by actual operating facilities or guaranteed via any procurement process. In most cases this information was provided by technology vendors through REOI or RFQ processes, in which the vendors were not required to guarantee a price. It is not clear that reported capital costs address all capital and construction cost elements. Nor is it clear that reported operating costs address all real costs associated with such facilities.

7.1.1.3 Plasma Arc Gasification

Capital costs were estimated based on four project estimates found during the literature review.^{[156],[157],[158]} A summary of the reported capital and operating costs for plasma arc gasification facilities is presented in Table 7-4.

Table 7-4: Summary of Reported Capital and Operating Costs for Plasma Arc Gasification Facilities (2009\$ CDN)

	Capital Costs (Annual Design Tonne)	Operational Costs
Lowest Reported Cost	\$859	\$87.37
Highest Reported Cost	\$2,027 (200,000 tpy capacity)	\$213.97
Median Reported Cost	\$1,225 +/- 44%	\$119.69 +/- 55%

The reported operating and capital costs are not based on those incurred by actual operating facilities or guaranteed via any procurement process. In most cases this information was provided by technology vendors through REOI or RFQ processes, in which the vendors were not required to guarantee a price. It is not clear that reported capital costs address all capital and construction cost elements. Nor is it clear that reported operating costs address all real costs associated with such facilities.

7.1.1.4 Pyrolysis

Cost range information for pyrolysis was made available through data gathering processes without any requirement for financial guarantees, and thus should be considered to be less reliable. Capital costs and operating costs were estimated based on six project estimates found during the literature review.^{[159],[160],[161]} A summary of the reported capital and operating costs for pyrolysis facilities is presented in Table 7-5.

¹⁵⁶ MOSA, 2009. Summary Report on Estimated Costs to Develop and Operate new Regional Waste Disposal and Processing Facilities

¹⁵⁷ NYC, 2006. Focused Verification and Validation of Advanced Solid Waste Management Conversion Technologies

¹⁵⁸ URS, 2005. Los Angeles County, Conversion Technology Evaluation Report

¹⁵⁹ MOSA, 2009. Summary Report on Estimated Costs to Develop and Operate new Regional Waste Disposal and Processing Facilities

¹⁶⁰ NYC, 2006. Focused Verification and Validation of Advanced Solid Waste Management Conversion Technologies

¹⁶¹ URS, 2005. Los Angeles County, Conversion Technology Evaluation Report

Table 7-5: Summary of Reported Capital and Operating Costs for Pyrolysis Facilities (2009\$ CDN)

	Capital Costs (Annual Design Tonne)	Operational Costs
Lowest Reported Cost	\$161	\$29.76
Highest Reported Cost	\$926 (200,000 tpy capacity)	\$104.58
Median Reported Cost	\$539 +/- 43%	\$50.87 +/- 52%

The reported operating and capital costs are not based on those incurred by actual operating facilities or guaranteed via any procurement process. In most cases this information was provided by technology vendors through REOI or RFQ processes, in which the vendors were not required to guarantee a price. It is not clear that reported capital costs address all capital and construction cost elements. Nor is it clear that reported operating costs address all real costs associated with such facilities. There is some indication that the capital cost per annual design tonne for such facilities is more likely over \$1,500/annual design tonne.

7.1.1.5 Summary of Capital and Operating Costs

Table 7-6 summarizes the information that has been obtained regarding capital and operating cost ranges for the four more common WTE technologies. Generally as the complexity of the technology increases so too does the capital and operating costs.

Table 7-6: Summary of Reported Capital and Operating Costs for Common WTE Facilities (2009\$ CDN)

Conventional Combustion	
Median Capital Cost	▪ \$775/annual design tonne +/- 50%
Median Operating Cost	▪ \$65/tonne +/- 30%
Gasification	
Median Capital Cost	▪ \$800/annual design tonne +/- 40%
Median Operating Cost	▪ \$60/tonne +/- 45%
Plasma Arc	
Median Capital Cost	▪ \$1,300/annual design tonne +/- 45%
Median Operating Cost	▪ \$120/tonne +/- 55%
Pyrolysis	
Median Capital Cost	▪ \$161 to \$926/annual design tonne – data is not as reliable
Median Operating Cost	▪ \$50 to \$105/annual design tonne – data is not as reliable

7.1.2 Cost Differentials between Technologies

As indicated above, generally the reported median capital and operating costs per design tonne reflect the increased complexity of the technologies and the sensitivity of the systems to factors such as the variable nature of MSW. The primary features that result in increased capital and operating costs for these technologies include:

- **Pre-processing of MSW:** generally gasification and pyrolysis processes require a more homogenous waste stream, necessitating a front-end pre-processing system to remove inert materials, blend the MSW and shred/process the combustible fraction of the MSW to a more consistent particle size. This increases both the capital and operating costs for such facilities.
- **Energy Recovery:** generally gasification and pyrolysis processes have a higher parasitic plant load, consuming more electrical energy and/or fossil fuels to operate the facilities. This increases operational costs and decreases the proportion of energy recovered for sale.
- **Emissions Control:** generally gasification and pyrolysis processes include both controls to improve the quality of the intermediate energy product (e.g., syngas) involving various treatment systems, and at least some portion of the APC systems that would normally be used for conventional combustion facilities to control flue gas emissions when the intermediate energy product is combusted.
- **Reliability of the Technology:** as noted in Section 2.1.3, the complexity of gasification and pyrolysis systems is associated with decreased reliability. Scheduled and unscheduled downtime for high temperature gasification is reported as approximately 20%, higher than that for conventional combustion. While actual information on reliability for plasma gasification and pyrolysis is not readily available, it would be anticipated to be similar or worse than high temperature gasification. Some information indicates for example that the refractory for plasma gasifiers requires very frequent replacement, increasing downtime and costs. As a result, generally higher capital replacement costs and operating costs would be incurred in comparison with conventional WTE approaches.

7.1.3 Costs Associated with Emissions Control

Semi-dry APC systems with SNCR generally are the lowest cost emissions control systems for conventional WTE facilities (mass burn). However, some technical limitations associated with the semi-dry system with SNCR may make it less acceptable in some jurisdictions. Limitations include possible exceedances of emission limits in short periods with high HCl or SO₂ concentrations, the potential to produce large amounts of residue that has to be handled, and odours associated with the ammonia in the dry flue gas treatment (FGT) residue.

A report by Ramboll completed in July 2007^[162] analyzed the costs associated with various emissions control technologies (see Table 7-7). The report came to the following conclusions:

- The base-case (semi-dry FGT) has the lowest capital costs.

¹⁶² Ramboll, 2007. The Regions of Durham and York EfW Facility, Comparison of Flue Gas Treatment Systems.

- The wet FGT system with higher capital cost has the lowest operational costs, partly due to high efficiency of the used chemicals and partly due to the lack of dry FGT residues.
- The capital cost of the SCR process is around eight times more expensive than SNCR.
- The SCR uses around 30% less ammonia when reducing the NO_x content just below the emission limit, but the operational costs of SCR is nevertheless higher than SCNR as high pressure steam must be used to heat the flue gas an additional 25°C and spent catalyst must be disposed and replaced.

Table 7-7: Operational and Capital Costs for Different Emissions Control Systems

	Type of FGT System		Yearly Operational Costs (in 1,000 CAD 2007\$)		Capital Costs (in 1,000 CAD 2007\$)		Overall Costs (in 1,000 CAD 2007\$)	
	Acid Gas Treatment	De-NO _x Process	Acid Gas Treatment	De-NO _x Process	Acid Gas Treatment	De-NO _x Process	Total Cost Over 20 Years	Difference from Base Case
Base Case	Semi-dry	SNCR	2,516	90	11,000	1,000	56,928	–
S1	Semi-dry	SNCR + Stripper	2,156	98	11,000	2,200	58,281	1,353
S2	Semi-dry	SCR	2,156	180	11,000	7,000	64,728	7,800
S3	Wet	SNCR	1,887	90	21,000	2,200	61,546	4,618
S4	Wet	SNCR + Stripper	1,887	98	21,000	2,200	62,899	5,971
S5	Wet	SCR	1,887	180	21,000	7,000	69,346	12,418
S6	Semi-Wet	SNCR	2,198	90	14,000	1,000	60,763	3,835
S7	Semi-wet	SNCR + Stripper	2,198	98	14,000	2,200	62,116	5,188
S8	Semi-wet	SCR	2,198	180	14,000	7,000	68,563	11,635

As indicated in Table 7-4, the operational costs over 20 years are lower for wet emissions control systems, however there are significantly higher capital costs associated with this type of system.

7.2 Thermal Efficiency and Energy Recovery

Each of the WTE technologies discussed thus far has relative advantages and disadvantages associated with their operation.

This section of the report will discuss the thermal efficiency and energy recovery typical of mass burn incineration facilities (conventional combustion) and gasification facilities. There is insufficient information currently available to discuss the efficiency and energy recovery rates associated with pyrolysis and plasma arc gasification facilities.

7.2.1 Energy Recovery from Mass Burn Facilities

The combustion of waste is a heat generating process. Most of the energy produced during combustion is transferred to the flue gases which are cooled as they pass through the plant allowing for the capture of energy via a heat recovery boiler (which transfers the heat energy to water causing the production of steam or hot water).

Energy produced by such facilities can be used in the:

- Production and supply of heat (as steam or hot water)
- Production and supply of electricity (i.e., via a steam turbine), or,
- Production of heat and electricity (i.e., combined heat and power, CHP).

The energy produced can be used on-site and/or off-site. Heat and steam are commonly used for industrial processes or district heating systems while electricity is often supplied directly to an energy grid or used within the system.

Several factors influence the energy efficiency associated with mass burn incineration facilities.

These factors include:

- Characteristics of the waste being treated (chemical and physical characteristics – MJ/kg). Typical values of waste net calorific values are between 8 and 12.6 MJ/kg
- Plant design (increased steam parameters – boilers and heat transfer)
- Energy sale possibilities (heat and electricity or just electricity), and
- Local conditions (e.g., meteorological conditions – if the plant is located in a warm environment the use of district heating would not be practical).

The highest levels of waste energy utilization are normally obtained when the heat recovered can be supplied continuously as district heat (or process steam) or in combination with electricity generation. The use of district heat (or process steam), however, is highly dependent on the availability of a user for the energy (as well as local meteorological conditions).

The production of electricity alone is a common method that WTE facilities use to recover energy from the incineration process. Electricity only operations are less efficient than those that recover and use district heat (or process steam) but are less dependent on local conditions and therefore are widely employed.

Modern mass burn facilities that produce only electricity regularly recover and sell electricity in the range of 550 kWh/tonne of waste. Facilities that recover both heat and electricity can generate considerably more energy per tonne of waste treated. The WTE facility located in Brescia, Italy produces/markets 650 kWh and 500 kWh of electricity and heat respectively per tonne of waste treated. The WTE facility located in Malmo, Sweden (a much colder climate therefore increasing the beneficial uses of district heating) produces/markets 280 kWh and 2,580 kWh of electricity and heat respectively per tonne of waste treated. The Metro Vancouver WTE facility produces about 470 kWh

of electricity and 760 kWh of steam per tonne of waste (it should be noted that the Metro Vancouver facility was built in 1988, and higher efficiencies are now possible with BAT).^[163]

The following table (Table 7-8) provides ranges of potential efficiencies at incineration plants in a variety of situations. The actual figures at an individual plant will be site-specific. The purpose of the table, therefore, is to provide a means to compare what might be achievable under favourable circumstances. It should be noted that the reported efficiencies do not take into account boiler efficiencies (which exhibit typical losses in the order of 20%).^[164]

It is important to realize that direct comparison of WTE facilities with other power stations should be avoided. This is due to the fact that the conversion of steam into electricity at WTE facilities is limited by the composition of the waste (e.g., high chlorine content may cause corrosion in the boiler or economizer) and that when flue gas in is the range of approximately 250 – 400°C it cannot generally be used for generation of steam as this is considered to be the range in which de novo synthesis of dioxins/furans take place,^[165] discussed earlier in Section 3.1.1.

Table 7-8: Energy Potential Conversion Efficiencies for Different Types of Waste Incineration Plants^[166]

Plant Type	Reported Potential Thermal Efficiency %
Electricity Generation Only	17 – 30
Combined Heat and Power (CHP)	70 – 85
Heating Stations with Sales of Steam and/or Hot Water	80 – 90
Steam Sales to Large Chemical Plants	90 – 100
CHP and Heating Plants with Condensation of Humidity in Flue gas	85 – 95
CHP and Heating Plants with Condensation and Heat Pumps	90 – 100

NOTE:

The figures quoted in the above table are derived from addition of MWh of heat and MWh of electricity produced, divided by the energy output from the boiler. No detailed account is taken of other important factors such as: process energy demand (support fuels, electrical inputs) or displacement of electricity and heat generation.

A number of factors can be considered when attempting to increase the thermal efficiency of the waste incineration process. These include:

- Waste pre-treatment (homogenization and/or separation of non-suitable materials)
- Design of boilers for increased heat transfer

¹⁶³ AECOM. 2009. Management of Municipal Solid Waste in Metro Vancouver – A Comparative Analysis of Options for Management of Waste After Recycling

¹⁶⁴ European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration

¹⁶⁵ TWG. 2001. Draft of a German Report with basic information for a BREF-Documents “Waste Incineration”

¹⁶⁶ European Commission. 2006. Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration

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- Combustion air pre-heating (can have a positive influence on overall energy efficiency in the case of electricity production)
- Use of water cooled grates
- Flue gas condensation
- Use of heat pumps
- Flue gas re-circulation
- Steam-water cycle improvements.

7.2.2 Energy Recovery from Gasification Facilities

All existing gasification technologies examined, have lower energy recovery efficiencies than those currently being achieved by modern mass burn incinerators.^[167] This is due to the fact that a mass burn process generally results in more complete combustion of the fuel compared to gasification and/or as the support fuel/electrical inputs for gasification tend to be higher.

The gasification process results in the production of syngas which can be used similarly to natural gas. Syngas can be used to fuel a conventional boiler (similar to a mass burn system) to produce steam and drive a turbine which results in the production of electricity, but it can also be used in reciprocating engines to produce electricity and heat, combined cycle gas turbine plants to produce electricity and heat, or fuel cells, or it can be converted into ethanol.

The efficiencies of the gasification process depend on how the syngas is used. When used to produce electricity using a steam boiler and turbine, efficiencies are in the range of 10% to 20%. When burned in reciprocating engines, efficiencies increase slightly to in the range of 13% to 28%, and in combined cycle gas turbines, they can be as high as 30%. It should be noted, that there are no known commercial scale applications of combined cycle gas turbines using syngas produced from MSW, therefore this number should be considered theoretical in nature. When used for district heating (CHP) over 90% efficiencies can be achieved.^[168]

Interstate Waste Technologies (who market the Thermoselect gasification technology in North America) report that the Thermoselect technology can produce 641 kWh of net electricity per tonne of waste treated.^[169] When the Thermoselect technology is combined with reciprocating engines, overall net efficiency is approximately 13% (exported power divided by thermal input).^[170]

7.3 European Union Energy Efficiency Equation Experience

In December 2008, the European Union's (EU) Waste Framework Directive (2008/98/EC) came into force. The Waste Framework Directive (WFD) provides an umbrella for all other European waste

¹⁶⁷ Fichtner Consulting Engineers Limited. 2004. The Viability of Advanced Thermal Treatment of MSW in the UK

¹⁶⁸ AECOM. 2009. Management of Municipal Solid Waste in Metro Vancouver – A Comparative Analysis of Options for Management of Waste After Recycling

¹⁶⁹ Alternative Resources, Inc. 2008. Evaluation of Municipal Solid Waste Conversion Technologies

¹⁷⁰ Fichtner Consulting Engineers Limited. 2004. The Viability of Advanced Thermal Treatment of MSW in the UK

legislation. The WFD includes an energy efficiency equation which will be adopted into legislation in the individual member states by December 31, 2010. The WFD lays down measures to protect the environment and human health by preventing or reducing the adverse impacts of the generation and management of waste and by reducing overall impacts of resource use and improving the efficiency of such use.

The WFD presents a five-step hierarchy of waste management options which must be applied by Member States when developing their national waste policies. The waste hierarchy given is as follows:

1. Waste prevention
2. Re-use
3. Recycling
4. Recovery (including energy recovery)
5. Safe landfill disposal, as a last resort.

The WFD considers energy-efficient waste incineration a recovery operation, provided that it complies with certain energy-efficiency criteria.^[171] In order to determine whether or not a WTE facility is deemed a recovery operation, the WFD presents an energy efficiency formula which calculates a facility's energy efficiency.

The energy efficiency formula is as follows:^[172]

$$\text{Energy efficiency} = (Ep - (Ef + Ei)) / (0.97 \times (Ew + Ef))$$

- *Ep* means annual energy produced as heat or electricity (GJ/year). It is calculated by applying an equivalence factor of 1MWh electricity produced being equivalent to 2.6 MWh of electricity imported from other sources onto the grid and by applying an equivalency factor of 1MWh of fuel replaced by heat produced being equivalent to 1.1 MW of imported fuel.
- *Ef* means annual energy input to the system from fuels contributing to the production of steam (GJ/year)
- *Ew* means annual energy contained in the treated waste calculated using the net calorific value of the waste (GJ/year)
- *Ei* means annual energy imported excluding *Ew* and *Ef* (GJ/year)
- 0.97 is a factor accounting for energy losses due to bottom ash and radiation.

Using this formula, an incineration facility is considered a recovery operation if it reaches an energy efficiency of 0.60 for installations in operation and permitted before January 1, 2009 and 0.65 for

¹⁷¹ European Parliament (November 11, 2008), *The Legislative Observatory Final Legislative Act*, Retrieved February 19, 2010, from <http://www.europarl.europa.eu/oeil/file.jsp?id=5303132>

¹⁷² Official Journal of the European Union (November 22, 2008), Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on Waste and Repealing Certain Directives, Retrieved February 19, 2010, from http://www.wastexchange.co.uk/documenti/europeanorm/DIR2008_98_EC.pdf

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installations permitted after December 31, 2008.^[173] Those WTE facilities that reach these criteria are considered R1 recovery operations.

The drivers behind the WFD and the R1 formula were many and to a certain degree contradicting, some are mentioned below:

- In the EU, when waste is co-incinerated in cement kilns, the process is defined as recovery, whereas incineration of MSW in dedicated WTE facilities is defined as disposal. The WTE industry found this definition unreasonable.
- Recovery of energy from waste is an important component in a European waste management business model. Energy is a precious resource and the WTE industry felt it should be credited this benefit. WTE also allows for material recovery, however material recovery is not accounted for by the energy efficiency equation.
- According to the EU transport regulation, trans-boundary transport of waste for recovery is allowed without any particular control, whereas trans-boundary transport of waste for disposal is subject to multiple restrictions and controls.

As a first step the produced energy is determined by considering produced electricity and thermal energy for commercial use. Two equivalency factors are applied: 2.6 as a factor if electricity is produced in lieu of electricity imported from other energy generating sources onto the grid and 1.1 if thermal energy is produced in lieu of imported fuel. The factor takes into account the efficiency of the energy production which is replaced by WTE production. In a second step the energy input from fuels and sources other than waste is subtracted ("Energy from fuels", "Other imported energy"). Energy input from fuels (e.g., gas firing for start-up operations; electricity supply from the grid) is deducted. The remaining figure is the energy produced only by waste input. In a third step the energy produced only by the waste input is divided by the energy content of the waste (the potential of energy contained in the waste, calculated from the lower calorific value) plus the energy input from fuels. Note: generally the energy content of the waste is determined through published values for specific material streams and/or fuel testing, but there are no specific requirements for fuels/material testing that must be met in application of the energy efficiency formula. In addition the denominator is multiplied by 0.97. This factor accounts for energy losses via bottom ash and radiation.

If a WTE facility does not meet the R1 criteria it is deemed a disposal facility and falls to the lowest level of the hierarchy.^[174]

As indicated in the Figure 7-3 below, WTE facilities generating a mix of both heat and power generally easily fulfill the efficiency formula having an R1 of between 0.6 and 0.8, and are defined as recovery. WTE facilities with optimized power production of over 700 kWh/tonne of waste will as well be able to fulfill the requirement for recovery whereas several facilities, especially older ones, might not be able to fulfill the requirement and will not succeed in being defined as recovery.

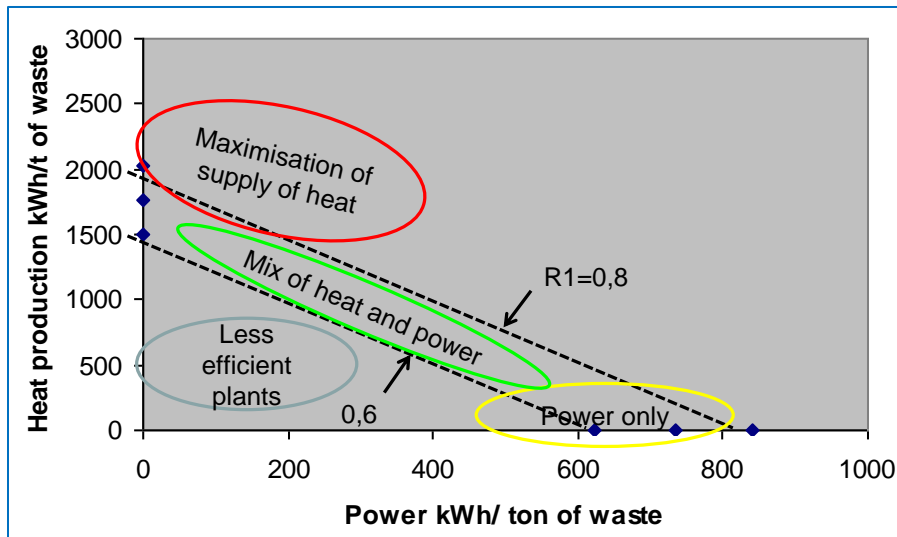
¹⁷³ The formula only applies to incineration facilities dedicated to the processing of municipal solid waste (reference 139)

¹⁷⁴ Institut für Ökologie und Politik GmbH (June 2006), *The Energy Efficiency Formula of Annex II of the Waste Framework Directive, A Critical Review*. Retrieved February 19, 2010, from http://www.eeb.org/activities/waste/waste_strategy/20060630-Okopol-Brief-on-MSWI-efficiency-formula-v5-final.pdf

It should be noted that the equation is not entirely clear and may be interpreted differently from one country to another. In addition, the impact of a facility's internal energy consumption is often discussed (e.g., if pre-treatment is required for the process it should then be calculated independently if pre-treatment is carried out at another location). This is of relevance for some mass burn facilities but even more so for fluidized bed incinerators and for the emerging technologies where the internal consumption of energy for waste pre-treatment is relatively high.

Figure 7-3: Relationship of Heat to Power Production for WTE Facilities

NOTE:



The dashed lines above represent an R1 of 0.6 and 0.8 respectively.

The EU Commission is in the process of further defining the use of the formula, as practical use of the formula showed that a transparent and harmonized way of calculating energy efficiency was necessary among the member states. The commission has engaged consultants, CEWEP, and other interest groups to evaluate and further define the use and the interpretation of the formula.

The Waste Framework Directive has to be implemented in all member states no later than December 31, 2010. For this purpose, the EU Commission will by the end of October 2010 publish European guidance for the use of the R1 energy efficiency formula for incineration facilities dedicated to the processing of MSW. The draft guidance is defining among others:

- The scope of the Energy Efficiency Formula
- The system boundaries
- The qualification procedure and monitoring of compliance.

Some countries in the EU have already adopted and implemented use of the formula. For example, the Netherlands has implemented the formula but takes the internal energy consumption of the facility into account. Five plants, representing approximately 70% of the country's capacity, are

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defined as recovery whereas the remaining facilities did not succeed in fulfilling the required efficiency and are therefore defined as waste disposal.

In Denmark the WFD has been adopted but without the formula. All WTE facilities in Denmark generate both heat and power and have an energy efficiency value of greater than 0.65. All plants will easily be defined as recovery according to the definition in the WFD. It is possible this value may be increased by government to drive continuous improvement in energy efficiency. The definition of recovery versus disposal and use of the equation is further complicated by the potential future imports of MSW, which are currently prohibited, into Denmark.

In Italy it is most likely that the input energy to the WTE facilities will be taken into account. Only energy that is actually sold (as heat and/or power) is allowed to be considered. The application of the formula is complicated by seasonal variations in consumption of energy where district heating is applied. Further, there is uncertainty in how to address facility consumptive use of power in the calculation. Principally this means that a WTE facility that is considered a recovery facility one year may be considered as a disposal facility in subsequent years should some or all of the energy not be sold.

In France a waste incineration tax is charged to plants defined as disposal facilities but not to plants determined to meet the recovery criteria. France recently started using the equation but is awaiting the published guidance later in 2010 for consistent application.

In the UK and in Scotland new WTE facilities have to prove they are able to achieve energy efficiency above 0.65 in order to obtain an operating permit. Similar to France, the UK and Scotland recently started using the formula, pending release of the EU guidance on application.

In summary, there is inconsistent application of the energy equation in the EU. The situation should be clarified somewhat with the release of additional guidance by the EU in the fall of 2010.

7.4 WTE Energy Recovery and Revenue Streams in BC

Direct revenue streams for WTE facilities include those from the sale of energy (including any combination of district heat generation and generation of electricity), from the sale of recovered materials (e.g., metals) and from tipping fees.

For every tonne of MSW consumed in a WTE facility, it is typically possible to generate up to 2 MWh of heat energy (as hot water or steam) and in the order of 0.5 to 0.8 MWh of electrical energy or any combination thereof depending on the design of the plant. The total amount of energy generated and marketed depends on the total available energy associated with the mass of MSW processed, and the ability to find a market for the energy.

Table 7-9 provides an overview of the potential energy generation and energy sales for a 100,000 tpy conventional (mass burn) WTE developed in a BC market, combusting post-diversion residual waste, if the sale of heat energy were to be limited by local market conditions. The composition of the post-diversion residual material assumes that a source separated organics diversion program is in place, diverting in the order of 60% or more of the 'wet' food materials from the waste stream.

Table 7-9: Potential Energy Generation and Energy Sales for a 100,000 tpy Conventional WTE Facility in a BC Market

Electricity Generation	Based on post source separated organics (SSO) waste composition and characteristics: <ul style="list-style-type: none"> ▪ Average Net Energy Production: 770 kWh/tonne ▪ Waste Energy Content: 13 MJ/kg ▪ Plant Heat Rate: 16.9 MJ/kWh
Combined Heat and Power (CHP)	CHP contingent upon development of proximate users of heat energy, that could be limited given local conditions
Auxiliary Fuel	Requires Natural Gas, for start up and temperature control
Bottom Ash Handling	Bottom ash quenched, quench water recycled Bottom ash screened and magnetically separated to remove ferrous and non-ferrous metals with 55% recovery rate
Power Island	One single casing steam turbine generator, mechanical draft cooling tower

Revenue streams for such a WTE Plant could generally include the following:

- Electricity Sales
- Sales of Ferrous and Non-Ferrous Metals, recovered from the bottom ash
- Tipping Fee revenue from commercializing plant capacity.

The value of these revenue streams is entirely contingent upon the market for the commodities noted, and in some cases it is difficult to determine with any degree of relative certainty at this time.

With regard to electricity sales, market prices are contingent upon the jurisdiction. For example, market prices for energy from waste have recently been established in Ontario of 8.5 cents per kWh. At that rate, electricity sales from a 100,000 tpy WTE plant could be in the order of \$6.5 million annually. However, it is likely that lower energy prices would prevail in BC based on the prevalence of renewable energy sources in the market. For BC residential customers, a two-step Conservation Rate is applied on an interim basis.^[175] As of April 1, 2010, the current cost of electricity in BC is 6.27 (Step 1) and 8.78 (Step 2) cents per kWh.

Should a proximate market for heat be developed (e.g., development of greenhouses), the potential for heat recovery for a 100,000 tpy conceptual WTE plant would vary between 46 million kWh (conservative based on high pressure steam, electricity production reduced to 88%) and 136 million kWh (hot water recovery based on BAT EU practice, electricity production reduced to 80% with 2 units of heat produced for each unit of electricity). For heating of greenhouses, the best option would be recovery of hot water that could be supplied and used in radiant heating systems. Heat recovery would decrease net electricity production and revenues, between 12.5 and 20%. The market would be contingent upon the energy requirements for greenhouses which vary, based on design (materials, construction method) and climate. Assuming that the heat sold replaces that which would

¹⁷⁵ BC Hydro. April 1, 2010, Electricity Rates. Website: http://www.bchydro.com/youraccount/content/electricity_rates.jsp.

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be otherwise generated by burning natural gas, and considering potential energy markets, the heat could be sold at approximately \$0.04/kWh. For a 100,000 tpy facility, annual revenues from the sale of heat could vary between \$1.8 and \$5.4 million.

Revenues earned from the sale of recovered materials, could include revenues from the sale of recovered metals (ferrous and non-ferrous), recovered reagents from the APC train (e.g., gypsum) and recovery of aggregate from bottom ash. Considering the current state of the industry in North America, it is reasonable to assume markets for recovered metals, but not necessarily for any other recovered materials. In regards to revenues from the sale of ferrous and non-ferrous metals recovered from the bottom ash of the WTE plant (assuming a 100,000 tpy capacity), approximately 9,000 tpy of metals could potentially be recovered (pending confirmation of the characteristics of the MSW stream that would be managed at the plant). Based on current North American metals markets, which are somewhat depressed compared to previous years, a conservative estimate for this material stream would be \$200/tonne or approximately \$1.8 million annually.

It is difficult to determine if or how much revenue would be generated through tipping fees for a WTE plant in BC. Current Metro Vancouver tipping fees at waste disposal sites are in the order of 82 to 86 \$/tonne.^[176] For a new WTE facility the ownership model (public or private) is anticipated to have a role in setting tipping rates.

As discussed above, the overall energy efficiency (and revenues from sale of energy) are potentially limited by the available markets for sale of heat energy, and other limitations including electricity pricing.

The Environmental Protection Division has an operational policy that addresses the review of SWMPs which include MSW as a feedstock for WTE facilities. This policy states that the ministry prefers WTE facilities that incorporate resource recovery (as part of a waste management hierarchy) and expects that energy recovery facilities would meet at least 60% efficiency based on a calculation similar to the EU energy efficiency equation. However, any new WTE facilities in BC may not be able to achieve an energy efficiency of 60% without further development of infrastructure such as district heating that would facilitate the use of heat generated by a WTE facility, recognizing that a high efficiency is difficult to reach through the production of electricity alone. The lessons learned in Europe as EU member states implement the energy efficiency equation during the last half of 2010 may provide guidance to the ministry about interpretation of the equation and how it may be further applied in a BC context.

¹⁷⁶ Metro Vancouver Disposal Facilities. Website:
<http://www.metrovancouver.org/services/solidwaste/disposal/Pages/disposalfacilities.aspx>

7.5 Summary – BAT for Energy Recovery

The following list outlines the BAT for energy recovery from WTE facilities^{[177] [178]}:

- Overall optimization of energy efficiency and energy recovery taking into account techno-economic feasibility and the availability of users for the energy to be recovered.
- Reduction of energy loss via the flue gases (i.e., reduce flue gas flow to recover more heat energy).
- The use of a boiler to transfer energy with a thermal conversion efficiency of at least 80%.
- Securing where possible, long-term heat/steam supply contracts to large heat/steam users to maximize the heat/steam usage.
- Locate in an area where heat and/or steam use can be maximized through any combination of:
 - Electricity generation with heat or steam supply (combined heat and power – CHP)
 - District heating
 - Process steam to industrial or other facilities
 - Heat/steam supply for use in cooling/air conditioning systems (through the use of absorption chillers, which use steam or hot water to drive a phase change in a medium to create a cooling effect).
- Where electricity is generated, optimization of steam parameters including consideration of the use of higher steam parameters to increase electricity generation.
- The selection of a turbine suited to the electricity and heat supply regime and high electrical efficiency.
- Where electricity generation is a priority over heat supply, the minimization of condenser pressure.
- General minimization of overall facility energy demand including consideration of the following:
 - Selecting techniques with lower energy demand over those with higher energy demand
 - Ordering APC components to avoid the requirement for flue gas reheating
 - If flue gas reheating is necessary, the use of heat exchanger systems to minimize energy demand.
- The location of a new facility so that the use of CHP and/or heat and/or steam can be maximized so as to generally exceed an overall total energy export level of 1.9 MWh/tonne of MSW based on an average net calorific value (NCV) of 2.9 MWh/tonne.
- Reduce the average installation electrical demand to be generally below 0.15 MWh/tonne of MSW processed based on an average NCV of 2.9 MWh/tonne.

¹⁷⁷ European Commission. 2006. Integrated Pollution Prevention and Control: Reference Document on the Best Available Techniques for Waste Incineration.

¹⁷⁸ Federal Environment Agency – Austria. 2002. State of the Art for Waste Incineration Plants.

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