FICHTNER

Consulting Engineers Limited

Killoch Energy Recovery Facility

Barr Environmental Ltd

BAT Assessment Report

ENGINEERING ┼ CONSULTING

Document approval

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Contents

1 Introduction

Barr Environmental Limited (Barr) is developing the Killoch Energy Recovery Park (the 'Facility') to thermally treat incoming non-hazardous residual waste fuel. The Facility will be located on land at Barr's existing operational site in Killoch.

In accordance with the requirements of IPPC Sector Guidance Note S5.01, titled "Guidance for the Incineration of Waste and Fuel Manufactured from or including Waste" (herein referred to as S5.01), this report presents quantitative BAT assessments for acid gas abatement, nitrogen oxides abatement and Energy from Waste (EfW) technology for the Facility.

Each assessment follows the structure of Horizontal Guidance Note IPPC-H1 and includes comments on all of the environmental parameters mentioned in IPPC-H1.

1.1 Assumptions

The Facility will use a moving grate as the combustion technology. The Facility will comprise of one stream, with a nominal design capacity of approximately 19 tonnes per hour of residual non-hazardous waste / Refuse Derived Fuel (herein referred to as incoming waste). The Facility is designed to operate at 100% Maximum Continuous Rating ("MCR") between 8 MJ/kg and 14 MJ/kg, with a design point at 10.5 MJ/kg. This equates to a nominal design capacity of approximately 150,000 tonnes per annum, assuming 8,000 hours operation per annum. The maximum annual throughput is 166,000 tonnes per annum assuming 8760 hours operation per annum at the design point. The maximum hourly throughput will be 25 tonnes per hour.

The Facility will generate up to 17.2 MWe (design maximum) with a parasitic load of 1.7 MW$_e$.

In addition, it is assumed that ammonia will be used for the Selective Non-Catalytic Reduction (SNCR) NO$_X$ abatement system.

The quantitative BAT assessments use data obtained by Fichtner from a range of different projects using the technologies identified within this assessment.

In the operating costs sections, the following unit costs have been assumed:

- Water .... £0.85 per tonne
- Lime Slurry .... £90 per tonne
- Lime.. £110 per tonne
- Sodium Bicarbonate.. £155 per tonne
- Activated Carbon .... £605 per tonne
- Ammonia.. £114 per tonne
- Sand (with defined particle size distribution) .... £100 per tonne
- Bottom Ash Processing .... £15 per tonne
- Lime Air Pollution Control residue (APCr) Disposal .... £155 per tonne
- Sodium bicarbonate APCr Disposal .... £186 per tonne
- Landfill Tax (1 April 2020) .... £94 per tonne
- Imported power... £70 per MWh
- Electricity revenue .... £50 per MWh

2 Acid gas abatement

2.1 Options considered

There are currently three technologies widely available for the abatement of acid gases on waste incineration plants in the UK:

1. Wet scrubbing, involving the mixing of the flue gases with an alkaline solution of sodium hydroxide or hydrated lime. This has a good abatement performance, but it consumes large quantities of water, produces large quantities of liquid effluent which require treatment, has high capital and operating costs and generates a visible plume. It is mainly used in the UK for facilities treating hazardous waste where high and varying levels of acid gases in the flue gases require the buffering capacity and additional abatement performance of a wet scrubbing system.

2. Semi-dry, involving the injection of lime as a slurry into the flue gases in the form of a spray of fine droplets. The acid gases are absorbed into the aqueous phase on the surface of the droplets and react with the lime. The fine droplets evaporate as the flue gases pass through the system, cooling the gas. This means that less energy can be extracted from the flue gases in the boiler, making the steam cycle less efficient. The lime and reaction products are collected on a bag filter, where further reaction can take place.

3. Dry, involving the injection of solid lime into the flue gases as a powder. The lime is collected on a bag filter to form a cake and most of the reaction between the acid gases and the lime takes place as the flue gases pass through the filter cake. In its basic form, the dry system consumes more lime than the semi-dry system. However, this can be improved by recirculating the flue gas treatment residues, which contain some unreacted lime and reinjecting this into the flue gases.

Wet scrubbing is not considered to be suitable, due to the production of a large volume of hazardous liquid effluent, a reduction in the power generating efficiency of the plant and the generation of a visible plume. The dry and semi-dry systems are considered further below.

2.2 Environmental performance

2.2.1 Emissions to air

The impact of emissions to air is considered in the Air Quality Assessment, which can be found in Appendix E of the Supporting Information. The acid gas emissions were assessed at the daily emission concentrations of 30 mg/m^3 for sulphur dioxide and 6 mg/m^3 for hydrogen chloride. These emission limits are in accordance with the requirements of the BAT AELs.

Table 2-1 shows the emission concentrations at the stack and the predicted ground level concentrations for each option. For sulphur dioxide, the 99.18[th] percentile of the daily averages is shown. For hydrogen chloride, the annual average is shown. The emission concentrations for a semi-dry system are expected to be the same as for a dry system so the ground level impacts are also the same.

Table 2-1: Emissions to air

Abatement system		Dry		Semi-dry	
Pollutant	**Units**	**SO$_2$**	**HCl**	**SO$_2$**	**HCl**
Unabated emission concentration	mg/m^3	250	1,535	250	1,535
Unabated emission rate	tpa	220	1,350	220	1,350
Abated emission concentration	mg/m^3	30	6	30	6
Abated emission rate	tpa	30	10	30	10
Total emissions abated	tpa	190	1,340	190	1,340
Process Contribution (PC)	ug/m^3	1.54	2.21	1.54	2.21
Background	ug/m^3	6.76	1.42	6.76	1.42
Predicted Environmental Contribution (PEC)	ug/m^3	8.30	3.63	8.30	3.63
Air quality objective	ug/m^3	125	750	125	750
PC as % of AQO		1.23	0.29	1.23	0.29
PEC as % of AQO		6.64	0.48	6.64	0.48

The short-term impact of the Facility is 1.23% of the daily average air quality objective for SO$_2$ and 0.29% of the hourly air quality objective for HCl. Whilst the impact of HCl can be screened as insignificant at the point of maximum impact, SO$_2$ cannot be screened as insignificant, when applying the criteria stated in SEPA's H1 guidance note. Therefore, the PEC has been calculated. The PEC of the Facility for SO$_2$ is 6.64% of the daily average air quality objective, which can be described as a 'not significant' impact.

A more detailed assessment of impacts from the release of sulphur dioxide and hydrogen chloride is presented within the Air Quality Assessment within Appendix E of the Supporting Information.

2.2.2 Deposition to land

The impact of acid deposition on sensitive habitats has been assessed in the Air Quality Assessment presented in Appendix D of the Supporting Information. As can be seen from this assessment, the impact of acid deposition on sensitive receptors is considered to be 'insignificant' at all habitat features with the exception of acid deposition on grassland habitats at Barlosh Moss, woodland habitats at Burnock Water, and woodland habitats at Ancient Woodlands 2, 3 and 4.

The maximum predicted acid deposition process contribution (PC) at these habitats is 6.39% of the Critical Load (CL), based on the maximum predicted PC from all five years of weather data used in the modelling. However, the average PC over the five years of weather data is less than 1% of the CL for all habitats. The significance of effect has been assessed further in the ecology chapter of the Environmental Impact Assessment Report (EIAR) submitted within the planning application documentation. This is provided in Appendix D of the Supporting Information. The Ecological Interpretation concludes that there will be "no significant impact" on these habitats resulting from emissions from the Facility.

2.2.3 Emissions to water

There are no emissions to water for either the dry or the semi-dry systems.

2.2.4 Photochemical Ozone Creation Potential

Sulphur dioxide has a photochemical ozone creation potential (POCP) of 4.8. Hence, the POCP for both the dry and semi-dry systems would be 140 tonnes ethylene equivalent.

2.2.5 Global Warming Potential

The direct emissions of greenhouse gases are the same for each option, since the carbon dioxide and nitrous oxide emission concentrations are unchanged. However, the energy consumption is slightly different, which would change the power exported from the plant. This means that the reduction in greenhouse gas emissions due to the displacement of power generated by other power stations would be different in each case.

The semi-dry system involves the evaporation of water. Since the reaction temperature of the lime and hence the outlet temperature should be the same, this means that the flue gas temperature at the inlet to the abatement system is higher for the semi-dry system than the dry system and hence more power can be generated if a dry system is used.

In order to calculate the global warming potential of electricity consumption, the figure of 371 kgCO$_2$/MWh has been used, as applied in the greenhouse gas assessment presented in Appendix E of the Supporting Information.

The global warming potential associated with each option is presented in Table 2-2.

Table 2-2: Global warming potential

	Units	Dry	Semi-Dry
Power consumed	kWh/t	30	28.5
	MWh pa	4,500	4,280
Generation lost (water evaporation)	MWh pa		5,400
Power not exported	MWh pa	4,500	9,680
GWP	t CO$_2$ pa	1,700	3,600

2.2.6 Raw materials

The estimated consumption of raw materials for both options is shown in Table 2-3.

Table 2-3: Raw materials

	Units	Dry	Semi-Dry
Additional water consumption compared to a dry system	tpa		13,000
Lime slurry	tpa		2,623
Lime	tpa	3,386	
Powdered Activated Carbon (PAC)	tpa	45	45

2.2.7 Waste streams

The only waste stream associated with the acid gas abatement treatment technologies is the APCr. APCr is classified as hazardous waste. The estimated APCr production rate for both systems is listed within Table 2-4.

Table 2-4: APCr production

	Units	Dry	Semi-Dry
APC residues	tpa	6,490	6,500

The quantity of APCr produced is very similar for both of the options, with the dry system having a slightly lower APCr production rate.

2.3 Costs

The estimated costs associated with each option are presented in Table 2-5. In order for direct comparisons to be made, the costs are presented as annualised costs, with the capital investment and financing costs spread over a 30-year plant lifetime with a rate of return of 9%, in accordance with the rates of return stated in SEPA's IPPC H1 Horizontal Guidance Note..

Table 2-5: Costs

	Dry	Semi-Dry
Capital cost	£8,700,000	£9,100,000
Annualised capital cost	£850,000	£890,000
Maintenance	£435,000	£455,000
Reagents and residues	£2,015,000	£1,893,000
Loss of exported power	£225,000	£484,000
Total annualised cost	**£3,525,000**	**£3,722,000**

2.4 Conclusions

Table 2-6 compares the two options.

Table 2-6: Comparison table

	Units	Dry	Semi-Dry
SO_2 abated	tpa	190	190
Photochemical Ozone Creation Potential (POCP)	t ethylene-eq pa	140	140
Global Warming Potential	t CO_2 eq pa	1,700	3,600
Additional water required in a semi-dry system	tpa	-	13,000
APC residues	tpa	6,490	6,500
Annualised cost	£ pa	£3,525,000	£3,722,000

The performance of the options is very similar.

The dry system only requires a small quantity of water for conditioning of the lime so that it is suitable for injection into the reaction chamber, whereas the semi-dry system requires the lime to be held in solution (quick lime). This requires significantly more water than a dry system.

The dry system has a lower global warming potential and annualised cost compared to the semi-dry system. However, the semi-dry option benefits from medium reaction rates that mean that a shorter residence time is required in comparison with a dry system. In addition, within a semi-dry system recycling of reagent within the process is not proven, but it is proven in a dry system.

Due to the lower water consumption and global warming potential, and the proven capability for recycling of reagents, the dry system is considered to represent BAT for the Facility.

3 Nitrogen oxides abatement

3.1 Options considered

Three options have been considered for NOx abatement:

1. Selective Catalytic Reduction (SCR), which involves the injection of ammonia solution or urea into the flue gases immediately upstream of a reactor vessel containing layers of catalyst.
2. Selective Non-Catalytic Reduction (SNCR), which involves the injection of ammonia solution or urea into the combustion chamber.
3. SNCR in combination with flue gas recirculation (SNCR+FGR).

3.2 Environmental performance

3.2.1 Emissions to air

The emission rates for nitrogen oxides, nitrous oxide and ammonia are shown in Table 3-1 together with the tonnages of nitrogen oxides abated.

Table 3-1: Air emissions

	Units	SNCR	SCR	FGR + SNCR
Nitrous oxide	mg/m^3	10	10	10
Ammonia	mg/m^3	10	10	10
NO$_x$, unabated concentration	mg/m^3	380	380	340
NO$_x$, unabated rate	tpa	330	330	300
NO$_x$, abated concentration	mg/m^3	120	70	120
NO$_x$ released after abatement	tpa	110	60	110
NO$_x$ removed	tpa	220	270	190

The emission rates for nitrogen oxides and ammonia are shown in Table 3-1.

A long-term abated emission concentration of 70 mg/Nm^3 (11% reference oxygen content) is used for SCR for the purposes of this BAT assessment, as this is the level that an SCR system can achieve on a long-term basis. The two SNCR systems, with and without Flue Gas Recirculation (FGR), would be required to achieve an emission limit of 120 mg/Nm^3, in accordance with the proposed emission limits for the Facility.

The unabated emission with FGR is assumed to be 10% lower than the other two cases.

The tonnages of nitrogen oxides removed by the abatement options are also shown.

The impact of emissions to air is considered in detail within the Air Quality Assessment, presented in Appendix E of the Supporting Information. Table 3-2 shows the predicted ground level concentrations for the two options.

Table 3-2: Air emissions

Abatement system:			SNCR	SCR	SNCR + FGR
Long term					
Process Contribution (PC)	µg/m^3		0.65	0.38	0.65
Background	µg/m^3		4.79	4.79	4.79
Predicted Environmental Contribution (PEC)	µg/m^3		5.44	5.17	5.44
Air Quality Objective	µg/m^3		40	40	40
PC as % of AQO			1.63	0.95	1.63
PEC as % of AQO			13.60	12.92	13.60
Short term					
Process Contribution (PC)	µg/m^3		4.59	2.68	4.59
Background	µg/m^3		9.58	9.58	9.58
Predicted Environmental Contribution (PEC)	µg/m^3		14.17	12.26	14.17
Air quality objective	µg/m^3		200	200	200
PC as % of AQO			2.30	1.34	2.30
PEC as % of AQO			7.09	6.13	7.09

The air quality impacts associated with all three NOx abatement options cannot be screened as insignificant. It can be seen that using SCR reduces the long-term PEC by 0.68% of the air quality objective and the short-term PEC by 0.96% of the air quality objective when compared to either SNCR or SNCR + FGR.

3.2.2   Deposition to land

The impact of nitrogen deposition on sensitive habitats has been assessed in the Air Quality Assessment presented in Appendix E of the Supporting Information. As can be seen from the results presented in the report, the impact of nitrogen deposition can be screened as insignificant at all ecological receptors.

3.2.3   Emissions to water

There are no emissions to water from any of the NO$_x$ abatement systems.

3.2.4   Photochemical Ozone Creation Potential

Nitrogen dioxide has a photochemical ozone creation potential (POCP) of 2.8 and nitrogen oxide has a POCP of - 42.7. Assuming that 10% of NO$_x$ is released as NO$_2$ and the rest as NO, the POCP is - 4,200 for the SNCR options and - 2,300 for the SCR option, meaning that SCR is less favourable. This is because nitrogen oxide converts to nitrogen dioxide in the atmosphere by reacting with ozone, this removing ozone from the atmosphere. Hence, the abatement of NO actually has a negative impact on POCP.

3.2.5 Global Warming Potential

The direct emissions of greenhouse gases are the same for each option, as the carbon dioxide and nitrous oxide emission concentrations are unchanged. However, the energy consumption is different in each option, which would change the power exported from the Facility in each case. In particular, SCR imposes an additional pressure drop on the flue gases, leading to an increase in power consumption on the ID Fan. In addition, SCR requires the flue gases to be reheated which is typically done by steam which would otherwise be used in the turbine to generate power; therefore, it will reduce the power generated by the turbine.

This means that the reduction in greenhouse gas emissions due to the displacement of power generated by other power stations would be different in each case.

In order to calculate the global warming potential of electricity consumption, the figure of 371 kgCO$_{2e}$/MWh has been used, as applied in the greenhouse gas assessment presented in Appendix E of the Supporting Information.

Table 3-3: Global Warming Potential

	Units	SNCR	SCR	SNCR + FGR
Power consumed	kWe	170	340	220
Power not generated	kWe		250	
Change in exported power	MWh pa	1,400	4,700	1,800
GWP	t CO$_2$ eq pa	500	1,700	700

3.2.6 Raw materials

The estimated consumption of raw materials for each option is presented in Table 3-4.

Table 3-4: Raw materials

	Units	SNCR	SCR	SNCR + FGR
Water	tpa	3,400	2,320	2,900
Ammonia	tpa	1,320	900	1,140

3.2.7 Waste streams

There will be no additional residues generated from any of the NO$_x$ abatement options.

3.3 Costs

The estimated costs associated with each option are presented in Table 3-5. In order for direct comparisons to be made, the costs are presented as annualised costs, with the capital investment and financing costs spread over a 30-year plant lifetime with a rate of return of 9%, in accordance with the rates of return stated in SEPA's IPPC H1 Horizontal Guidance Note..

Table 3-5: Costs

	SNCR	SCR	SNCR + FGR
Capital cost	£400,000	£5,900,000	£800,000
Annualised capital cost	£39,000	£574,000	£78,000
Maintenance	£8,000	£118,000	£16,000
Water and reagents	£153,000	£105,000	£132,000
Loss of exported power	£70,000	£235,000	£90,000
Total annualised cost	**£270,000**	**£1,032,000**	**£316,000**

3.4 Conclusions

The table below provides a summary comparison of the three options.

Table 3-6: Comparison table

	Units	SNCR	SCR	SNCR + FGR
NO_x released after abatement	tpa	110	60	110
NO_x removed	tpa	220	270	190
Photochemical Ozone Creation Potential (POCP)	t ethylene-eq pa	-4,200	-2,300	-4,200
Global Warming Potential	t CO_2 eq pa	500	1,700	700
Ammonia used	tpa	1,320	900	1,140
Total annualised cost	£ pa	£270,000	£1,032,000	£316,000
Average cost per tonne NO_x abated	£ p.t NO_x	£1,320	£3,820	£1,660

As can be seen from Table 3-6, applying SCR to the Installation:

1. increases the annualised costs by approximately £760,000;
2. abates an additional 50 tonnes of NOx per annum;
3. reduces the benefit of the facility in terms of the global warming potential by approximately 1,200 tonnes of CO_2;
4. reduces reagent consumption by approximately 420 tonnes per annum; and
5. costs an additional £2,160 per additional tonne of NOx abated compared to SNCR + FGR and £2,500 per additional tonne of NOx abated compared to SNCR.

The additional costs associated with SCR are not considered to represent BAT for the Facility. On this basis, SNCR is considered to represent BAT.

Including FGR to the SNCR system to abate NOx increases the cost per tonne of NOx abated by approximately 26%. It has no effect on the direct environmental impact of the plant, but it increases the impact on climate change by approximately 200 tonnes of CO_2 per annum while reducing ammonia consumption by approximately 180 tonnes per annum.

However, this is based on the assumption that FGR reduces the NOx generation within the furnace. As explained in Section 2.6.2 of the Supporting Information submitted with this application, this is not necessarily the case for all furnace manufacturers. Some designs can achieve lower levels of NOx without FGR. The proposed designs do not currently include FGR. Therefore, taking the above

into consideration, the use of SNCR without FGR is considered to represent BAT for the abatement of NOx within the Facility. However, this will be subject to detailed design of the Facility.

4 Reagent selection

4.1 Options considered

We have not considered reagents for wet scrubbing, since this has been eliminated as a technique in Section 2.1. We have therefore only considered the two alternative reagents for a dry system – lime and sodium bicarbonate.

4.2 Environmental performance

4.2.1 Emissions to air

There is no change in emissions to atmosphere between the two reagents. Both would achieve the same level of abatement.

4.2.2 Deposition to land

Again, there is no change between the two reagents.

4.2.3 Emissions to water

There are no emissions to water associated with either of the two reagents.

4.2.4 Photochemical Ozone Creation Potential

There would be no change to POCP for either system.
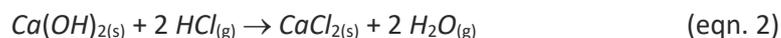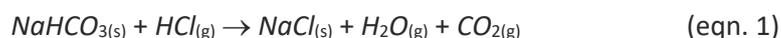
4.2.5 Global Warming Potential

Sodium bicarbonate has a higher optimum reaction temperature than lime, which means that less heat can be recovered in the boiler. However, this can be resolved by recovering additional heat after the acid gas abatement system. Therefore, it has been assumed that there is no impact on global warming potential from this operational difference.

The reaction of hydrogen chloride and sulphur dioxide with sodium bicarbonate results in an emission of CO_2 whereas the reaction with lime does not.

4.2.6 Raw materials

Sodium bicarbonate ($NaHCO_3$) has better solid handling properties and a significantly lower stoichiometric ratio than hydrated lime ($Ca(OH)_2$).

$NaHCO_3$ and $Ca(OH)_2$ react with the acid gases to produce alkaline salts as the following equations illustrate:

$$NaHCO_{3(s)} + HCl_{(g)} \rightarrow NaCl_{(s)} + H_2O_{(g)} + CO_{2(g)} \qquad \text{(eqn. 1)}$$

$$Ca(OH)_{2(s)} + 2\ HCl_{(g)} \rightarrow CaCl_{2(s)} + 2\ H_2O_{(g)} \qquad \text{(eqn. 2)}$$

In order to promote the reactions above, excess quantities of sodium bicarbonate or lime will be required. The excess reagent is lost in the residue. The ratio between the quantity of reagent supplied and the minimum required for the reaction is called the "stoichiometric ratio".

For sodium bicarbonate, a stoichiometric ratio of 1.30 is required, whereas for lime, a stoichiometric ratio of approximately 1.8 is required. This initially appears to be economically advantageous for sodium bicarbonate in comparison to lime. However, due to the higher relative molecular weight, and the fewer molecules of acid gas reacting per molecule of $NaHCO_3$, the overall consumption of sodium bi-carbonate is actually 64% higher than $Ca(OH)_2$ on a mass basis.

The reagent required to abate one kmol of HCl was calculated as approximately 109 kg of sodium bicarbonate and approximately 67 kg of lime.

4.2.7 Waste streams

The stoichiometric ratio indicates that the amount of residue will be higher with the lime option. However, due to the differences in relative molecular weight and the number of acid gas molecules reacting with each absorbent molecule, the lime system produces a similar amount of residue to the sodium bicarbonate option.

The residue production rate for abatement of one kmol of HCl was calculated as approximately 84 kg for sodium bicarbonate and approximately 85 kg for lime.

Furthermore, there are limited waste disposal options for sodium bicarbonate based APCr.

4.3 Costs

There is little difference in capital cost between the two reagents.

The purchase cost of sodium bicarbonate is significantly higher than lime, with bicarbonate costing almost 63% more than lime per tonne. This makes sodium bicarbonate an uneconomic option in comparison to lime.

The cost of disposing of the residue must also be considered due to the differences in quantity. Sodium based residues are more difficult to stabilise than lime residues; it has been assumed that the cost per tonne to landfill of sodium-based residues is 20% higher than lime residues, giving a disposal cost for sodium bicarbonate of £186/tn.

The operating costs for the two options are compared in Table 4-1, for a stoichiometric ratio of 1.8 for lime and 1.3 for sodium bicarbonate, on the basis of the abatement of one kmol of HCl:

Table 4-1: Costs per unit HCl abated

Item	Unit	NaHCO$_3$	Ca(OH)$_2$
Mass of reagent required	kg/kmol	109.0	67.0
Mass of residue generated	kg/kmol	84.0	85.0
Cost of reagent	£/tonne	155	110
Cost of residue disposal[1]	£/tonne	186	155
Overall Cost	£/op. hr/kmol	32.5	20.5
Ratio of costs	-	1.58	-

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[1] The figure shown does not include landfill tax.

4.4 Conclusions

There is a small environmental benefit for using sodium bicarbonate, in that the mass of residues produced is smaller. However, there are a number of significant disadvantages:

- The residue has a higher leaching ability than lime-based residue, which will limit the disposal options;
- The reaction temperature doesn't match as well with the optimum adsorption temperature for carbon, which is dosed at the same time;
- The sodium bicarbonate system has a slightly higher global warming potential due to the reaction chemistry; and
- The overall cost per kmol of reagent required to abate HCl is almost 58% higher.

Hence, the use of lime is considered to be BAT for the Facility.

5 Combustion techniques

5.1 Options considered

The available techniques for waste combustion have been qualitative assessed in section 2.6.1 of the Supporting Information submitted with this application. The qualitative assessment has been expanded to provide a cost-benefit analysis of moving grates, fluidised beds and rotary kilns.

1. Moving grates are an established technology in the UK and Europe for the combustion of high net calorific value fuels, such as those proposed to be processed. The grate turns and mixes the waste along its surface to ensure that all waste is exposed to the combustion process.

2. Fluidised beds are designed for the combustion of relatively homogeneous waste. The feed of incoming waste would be required to be pre-treated to produce a fuel for processing in a fluidised bed combustion system.

3. Rotary kilns function best with a consistent fuel feedstock. Rotary kilns can operate at higher temperatures than other systems due to the absence of exposed metal surfaces and therefore have been regularly applied in the UK for the incineration of hazardous, clinical and industrial wastes. Rotay kilns have had limited application in the UK for the incineration of residual non-hazardous waste / Refuse Derived Fuel as proposed to be processed at the Facility.

5.2 Environmental performance

5.2.1 Emissions to air

The emissions to atmosphere would not be affected by the choice of combustion technology. Although NO_x concentrations from the furnaces would be different, all three technologies would require further abatement to achieve the proposed emission limits. This means that the actual effect would be to change the amount of reagent required to abate the NO_x. This is considered in section 5.2.6.

Table 5-1: NO_x emissions

Option	NO_x emissions from furnace [mg/Nm³]
Moving grate	320-380[1]
Fluidised bed	250-300[1]
Rotary kiln	300-350[1]

(1) Presented at 11% oxygen with standard reference conditions

5.2.2 Deposition to land

Deposition from atmospheric emissions would be unchanged.

5.2.3 Emissions to water

There are no emissions to water for either system.

5.2.4 Photochemical Ozone Creation Potential

There would be no change to POCP for either system.

5.2.5 Global Warming Potential

The direct emissions of carbon dioxide are the same for each option. However, whilst fluidised beds have lower emissions of nitrogen dioxide they can have elevated emissions of nitrous oxide. Nitrous oxide is a greenhouse gas with a global warming potential (GWP) nearly 300 times that of carbon dioxide. Fluidised beds can be designed to minimise the formation of nitrous oxide. For the purposes of this assessment, it has been assumed that the fluidised bed has been well-designed and the emissions of nitrous oxide are slightly elevated and are released at a concentration of 10 mg/Nm^3.

A fluidised bed plant has a higher parasitic load than a moving grate or rotary kiln system due to the sand system and fly ash separation system. The additional parasitic load in the case of the fluidised bed option has been estimated at 10%.

This means that the reduction in greenhouse gas emissions due to the displacement of power generated by a conventional power station would be different for each technology.

In order to calculate the global warming potential of electricity consumption, the figure of 371 kgCO$_2$e/MWh has been used, as applied in the greenhouse gas assessment presented in Appendix D.

The results are presented in Table 5-2. In each case the overall GWP is less than zero, as there is a net reduction due to displacement of primarily fossil fuel power generation. Thus, the more negative figure produced by the grate is better.

Table 5-2: Global Warming Potential

		Grate	Fluidised bed	Rotary kiln
Power generated	MWh pa	138,000	138,000	88,000
Parasitic load	MWh pa	14,000	15,000	14,000
GWP	t CO2 eq pa	-46,000	-45,600	-27,000

5.2.6 Raw materials

The estimated consumption of raw materials for each option is shown in Table 5-3.

Table 5-3: Raw materials

	Units	Grate	Fluidised bed	Rotary kiln
Ammonia	tpa	1,300	900	1,700
Sand	tpa	-	1,800	-

5.2.7 Waste streams

The three options produce several solid waste streams.

- It is assumed that most metals within the incoming waste will have been removed during any pre-treatment processes. Therefore, it is assumed that this will be identical for all three technologies and has not been considered further.

- The fluidised bed will produce more non-hazardous ash than a grate or rotary kiln due to the addition of sand within the boiler which is used as a fluidising medium. The non-hazardous ash will include bottom ash and boiler ash. The boiler ash is removed in a cyclone before the acid gas abatement reagent is added. The non-hazardous ash could be usable for building aggregate,

but this is not certain. For the purposes of this assessment, it has been assumed that the non-hazardous ash cannot be used as a building aggregate and requires disposal in a non-hazardous landfill.

- All options produce APCr. For the purposes of this assessment, it is assumed that all three technologies will generate comparable quantities of APCr.

Estimated figures are presented in Table 5-4.

Table 5-4: Residue streams

	Units	**Grate**	**Fluidised bed**	**Rotary kiln**
Bottom ash	tpa	33,850	13,050	33,850
Boiler ash	tpa	-	22,600	-
APCr	tpa	6,490	6,490	6,490

The Fluidised Bed will produce a slightly higher quantity of residues compared to the grate and the rotary kiln due to the losses of sand which are used within the furnace for the fluidised bed.

5.3 Costs

The estimated costs associated with each technology are presented in Table 5-5.

Fluidised bed technology is typically up to around 6% more expensive than a grate, due to the additional waste screening equipment, sand dosing and recycling equipment, and fly ash separation. At the time of writing this assessment, capital costs are not readily available for the different technologies. Therefore, it has not been possible to consider the capital costs within this assessment.

Similarly, although fluidised beds typically have significantly higher maintenance costs than grate systems, maintenance costs are not readily available for the different technologies, so these were not considered within this assessment.

Table 5-5: Annual material costs and revenues

	Grate	**Fluidised bed**	**Rotary kiln**
Reagents	£150,000	£280,000	£190,000
Residue Disposal	£2,120,000	£2,150,000	£2,120,000
Annual Reagent and Residue Costs (Materials)	£2,270,000	£2,430,000	£2,310,000
Annual Power Revenue	£6,200,000	£6,150,000	£3,700,000

As presented in Table 5-5, the fluidised bed option has slightly higher costs associated with the purchase of reagents and the disposal of residues, assuming that the costs for treatment and re-use of fly ash are similar to those for bottom ash.

The parasitic load associated with fuel preparation has been excluded from the assessment, as it is assumed to be comparable across all three combustion technologies.

For a fluidised bed there may be costs associated with screening the incoming waste to ensure that there are no contaminants which could affect the operation of the fluidised bed. These costs have not been accounted for within Table 5-5.

5.4 Conclusions

Table 5-6 provides a summary comparison of the three technologies.

Table 5-6: Option comparison summary

		Grate	Fluidised bed	Rotary kiln
Global Warming Potential	t CO2 eq pa	-46,000	-45,600	-27,000
Ammonia consumption	tpa	1,300	900	1,700
Residues (total ash)	tpa	40,340	42,140	40,340
Annual total materials costs		£2,270,000	£2,430,000	£2,310,000
Annual power revenue		£6,200,000	£6,150,000	£3,700,000

All combustion technologies will produce similar quantities of residue, although the fluidised bed produces more residue due to the losses of sand from the furnace.

The material costs are approximately 10% higher for the fluidised bed than the grate, whereas the grate system will have a slightly higher power revenue. It is acknowledged that it is marginal. The grate system will be able to process the varying waste composition compared to a fluidised bed system which requires a consistent and homogenous fuel and therefore additional treatment of the waste.

The rotary kiln is less efficient, which has a significant impact on the global warming potential and the associated operating costs. In addition, the capital cost associated with a rotary kiln is likely to be significantly higher as additional streams will be required to achieve the proposed processing capacity.

As stated within the qualitative BAT assessment (refer to section 2.6 of the Supporting Information) grate combustion systems are designed for large quantities of heterogenous waste, whereas fluidised bed systems are more sensitive to inconsistencies within the fuel. Due to the robustness of grate combustion systems, they are considered to represent BAT for the Facility.

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