FICHTNER

Consulting Engineers Limited

Thornton Energy Recovery Centre

Sesona Hill House Ltd

BAT Assessment

ENGINEERING -|- CONSULTING

Document approval

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Contents

1 Introduction

Sesona Hill House Ltd (Sesona) is applying to the Environment Agency (EA) under the Environmental Permitting Regulations (EPRs) for an Environmental Permit (EP) to operate the Thornton Energy Recovery Centre (the Facility). The Facility will comprise a twin-line waste incineration plant to incinerate pre-processed refuse derived fuel (RDF). The Facility will be located at the Hillhouse Business Park, Thornton-Cleveleys, Lancashire.

A qualitative assessment of technologies utilised at the Facility is presented within section 2.6 of the Supporting Information.

In accordance with the requirements of the EA's Sector Guidance on Waste Incineration (EPR5.01), this report provides a quantitative Best Available Techniques (BAT) assessment for the technologies proposed at the Facility, specifically for acid gas abatement, nitrogen oxides abatement and combustion technology.

1.1 Assumptions

The combustion technology used at the Facility will be a moving grate. There will be two incineration lines. Assuming a design NCV of 10.11 MJ/kg, the Facility will process approximately 100,000 tonnes of waste per year (at a design capacity of 6.33 tph per line, and assuming 7,900 hours availability). However, the Facility will be capable of processing waste with a range of NCVs. The maximum throughput for the Facility will be up to 120,000 tpa of RDF, assuming 7,900 hours operation and an NCV of 8.43 MJ/kg. Therefore, the maximum capacity being applied for in the permit is 120,000 tpa of waste.

For the purposes of this BAT assessment, the design case is considered to be most reflective of 'normal' operations. It is not expected that the conclusions of the BAT assessment would change with the maximum case.

The Facility will generate approximately 9.284 MWe with a parasitic load of approximately 1.5 MWe.

In order to calculate the global warming potential of electricity consumption – assumed to be imported from the grid – the assumption of 372 gCO$_2$/kWh has been used, as applied in the greenhouse gas assessment presented in Appendix E of the Supporting Information.

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

For the purposes of this assessment, we have undertaken a quantitative assessment of the available technologies for the proposed capacity using 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
- Urea .... £191 per tonne
- Sand (with defined particle size distribution) .... £100 per tonne

- Bottom Ash Processing .... £15 per tonne
- Sodium bicarbonate APCR Disposal... £186 per tonne
- Landfill Tax (1 April 2022) .... £98.60 per tonne
- Imported power[1] ....£122 per MWh
- Electricity revenue[1] ....£57 per MWh

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[1]   https://www.gov.uk/government/publications/updated-energy-and-emissions-projections-2019

2 Reagent selection – acid gas abatement

2.1 Options considered

As explained in section 3.1, wet scrubbing is not considered to be a suitable technique; therefore, reagents for wet scrubbing have not been considered. Therefore, this assessment has only considered the two alternative reagents for a dry system – lime and sodium bicarbonate.

2.2 Environmental performance

2.2.1 Emissions to air

There is no change in emissions to atmosphere between the two reagents. Both would be required to achieve the same level of abatement in accordance with the emission limit values for the Facility.

Sodium bicarbonate does abate acid gases more efficiently at higher temperatures, which is dependent on the conditions of the flue gas passing through the plant. At the Facility, high operational temperatures are expected in the flue gas treatment system (around 220 – 330°C).

2.2.2 Deposition to land

Again, there is no change between the two reagents.

2.2.3 Emissions to water

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

2.2.4 Photochemical Ozone Creation Potential

There would be no change to photochemical ozone creation potential (POCP) for either system.

2.2.5 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.

2.2.6 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:

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

$$Ca(OH)_{2(s)} + 2\ HCl_{(g)} \rightarrow CaCl_{2(s)} + 2\ H_2O_{(g)} \qquad \text{(eqn. 2)}$$
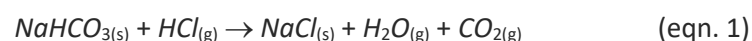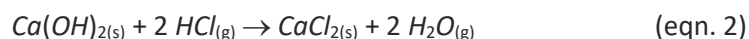
In order to promote the reactions above, excess quantities of sodium bicarbonate or lime will be required. The excess reagent is lost in the residue. The ratio between the quantity of reagent supplied and the minimum required for the reaction is called the "stoichiometric ratio".

For sodium bicarbonate, a stoichiometric ratio of 1.30 is required, whereas for lime, a stoichiometric ratio of 1.80 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 bicarbonate is actually around 70% 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.

2.2.7 Waste streams

Due to the stoichiometric ratio, the amount of residue will be higher with the lime option, although due to the differences in relative molecular weight and the number of acid gas molecules reacting with each absorbent molecule, the lime system does produce 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.

2.3 Costs

There is little difference in capital cost between the two reagents. The raw material cost of sodium bicarbonate is significantly higher than lime, with bicarbonate costing almost 45% more than lime per tonne.

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 2-1, 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 2-1: Costs per unit HCl abated

Item	Unit	NaHCO₃	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[2]	£/tonne	186	155
Overall Cost	£/op. hr/kmol	32.5	20.5
Ratio of costs	-	1.58	

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

2.4 Conclusions

The use of sodium bicarbonate has a number of advantages:

- The quantity of residues produced using sodium bicarbonate is smaller.

- Handling of sodium bicarbonate requires much less health and safety considerations/controls than handling of lime. Lime is a corrosive material and requires strict COSHH controls for handling and transfer. Furthermore, sodium bicarbonate is easier to pump than lime.

- Sodium bicarbonate abates more efficiently than lime at higher temperatures, as is proposed at the Facility.

- The use of a lime system with the proposed design is anticipated to lead to higher risk of corrosion, compared to using a sodium bicarbonate system.

Hence, the use of sodium bicarbonate is considered to represent BAT for the Facility.

The type of acid gas abatement system is assessed in section 3. Sodium bicarbonate is not suitable for a semi-dry system; therefore, a dry system is considered to represent BAT for the Facility. Notwithstanding this, a qualitative assessment has been undertaken and is presented below.

3 Acid gas abatement

3.1 Options considered

There are currently three technologies widely available for acid gas abatement 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.

   In the UK, wet scrubbing facilities are mainly used for treating hazardous waste where high and varying levels of acid gases in the flue gases require buffering capacity and additional abatement performance provided by a wet scrubber.

2. Semi-dry scrubbing, involving the injection of quick 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 scrubbing, involving the injection of solid lime or sodium bicarbonate into the flue gases as a powder. With lime systems, the lime is typically 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. Using sodium bicarbonate, as is proposed for the Facility, results in decomposition of the sodium bicarbonate to carbonate, with the diffusion of carbon dioxide producing a highly porous, high surface area sodium bicarbonate which is very efficient at absorbing acid gases. The efficiency increases with higher flue gas temperatures, as is proposed at the Facility.

Wet scrubbing is not considered to be suitable for the Facility, 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 have been considered further within sections 3.2 to 3.4. For the purposes of this assessment, lime is assumed to be the reagent in the dry system, to allow for a more appropriate comparison with lime slurry used in a semi-dry system. As discussed in section 2.4, the use of sodium bicarbonate as a reagent is considered to represent BAT for the Facility. Sodium bicarbonate is not suitable for use in a semi-dry system. Therefore, a dry system is considered to represent BAT for the Facility. Notwithstanding this, both dry and semi-dry systems have been assessed further in sections 3.2 – 3.4.

3.2 Environmental performance

3.2.1 Emissions to air

The impact of emissions to air is considered in the Air Quality Assessment, presented 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 3-1 shows the emission concentrations at the stack and the predicted ground level concentrations for each option. For sulphur dioxide, the 99.18th 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 3-1: Emissions to air

Parameter	Units	Dry		Semi-dry	
		SO$_2$	HCl	SO$_2$	HCl
Unabated emission concentration	mg/m3	480	900	480	900
Unabated emission rate	tpa	160	300	160	300
Abated emission concentration	mg/m3	30	6	30	6
Abated emission rate	tpa	10	2	10	2
Total emissions abated	tpa	150	298	150	298
Process Contribution (PC)	ug/m3	2.92	4.49	2.92	4.49
Background	ug/m3	13.92	1.42	13.92	1.42
Predicted Environmental Contribution (PEC)	ug/m3	16.84	5.91	16.84	5.91
Air quality objective	ug/m3	125	750	125	750
PC as % of AQO		2.34%	0.60%	2.34%	0.60%
PEC as % of AQO		13.47%	0.79%	13.47%	0.79%

The short-term impact of the Facility is 2.34% of the daily average air quality objective for SO$_2$ and 0.60% of the hourly air quality objective for HCl. Therefore, the impacts can be screened out as insignificant in accordance with the criteria stated in Environment Agency guidance note H1 (i.e., impacts can be screened out when the short-term PC is less than 10% of the short-term environmental standard).

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

3.2.2 Deposition to land

The impact of acid deposition on sensitive habitats has been considered in the Air Quality Assessment presented in Appendix E of the Supporting Information. As can be seen from the assessment, the impact of acid deposition on sensitive receptors is considered to be 'not significant' at all habitat features.

3.2.3 Emissions to water

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

3.2.4 Photochemical Ozone Creation Potential

Sulphur dioxide has a POCP of 4.8. Hence, the POCP for both the dry and semi-dry systems would be 50 tonnes ethylene equivalent.

3.2.5 Global Warming Potential

The direct emissions of greenhouse gases are the same for each option, as the emission concentrations for carbon dioxide and nitrous oxide are unchanged. However, the energy consumption is slightly different, which would change the power exported from the Facility. Therefore, 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 reagent 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. Therefore, more power can be generated if a dry system is used.

In order to calculate the global warming potential of electricity consumption – assumed to be imported from the grid - the assumption of 372 gCO_2/kWh has been used, as applied in the greenhouse gas assessment presented in Appendix E of the Supporting Information. The global warming potential is therefore calculated as follows:

$$Carbon\ intensity\ of\ displaced\ electricity \times net\ power\ generation$$

The global warming potential associated with each option is presented in Table 3-2.

Table 3-2: Global warming potential

	Units	Dry	Semi-Dry
Power consumed	kWh/t	30	28.5
	MWh pa	3,000	2,850
Generation lost (water evaporation)	MWh pa		2,700
Power not exported	MWh pa	3,000	5,550
GWP	t CO2 pa	1,100	2,100

3.2.6 Raw materials

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

Table 3-3: Raw materials

	Units	Dry	Semi-Dry
Additional water consumption compared to a dry system	tpa		5,071
Quick lime use	tpa		786
Lime use[1]	tpa	1,014	
Powdered Activated Carbon (PAC)[2]	tpa	49	49

(1) As described in section 3.1, for the purposes of this assessment, lime is assumed to be the reagent in the dry system, to allow for a more appropriate comparison with lime slurry used in a semi-dry system. Reagent choice is discussed further in section 2. The lime use has been calculated/scaled based on project-specific information for sodium bicarbonate consumption.

(2) PAC usage has been estimated based on Fichtner's previous project experience and scaled for the proposed capacity.

3.2.7 Waste streams

The only waste stream associated with the acid gas abatement treatment technologies is the Air Pollution Control residue (APCr). APCr is classified as hazardous waste. The estimated APCr production rate for both systems is listed within Table 3-5.

3.3 Costs

The estimated costs associated with each option are presented in Table 3-4. In order for a direct comparison to be made between the two acid gas abatement systems, the costs are presented as annualised costs, with the capital investment and financing costs spread over a 30-year lifetime with a rate of return of 9%, using the method recommended in Technical Guidance Note EPR-H1.

Table 3-4: Costs

	Dry	Semi-Dry
Capital cost	£5,800,000	£6,100,000
Annualised capital cost	£560,000	£590,000
Maintenance	£290,000	£305,000
Reagents and residues	£1,410,000	£1,246,000
Loss of exported power	£171,000	£316,000
Total annualised cost	**£2,431,000**	**£2,457,000**

3.4 Conclusions

A comparison of the two options is presented in Table 3-5.

Table 3-5: Comparison table

	Units	Dry	Semi-Dry
SO_2 abated	tpa	150	150
POCP	t ethylene-eq pa	50	50
Global Warming Potential	t CO_2 eq pa	1,100	2,100
Additional water required in a semi-dry system	tpa	-	5,071
APC residues	tpa	5,000	4,500
Annualised cost	£ pa	£2,431,000	£2,457,000

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. 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.

4 Nitrogen oxides abatement

4.1 Options considered

Three options have been considered for NOx abatement as follows:

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).

4.2 Environmental performance

4.2.1 Emissions to air

The emission rates for nitrogen oxides, nitrous oxide and ammonia are shown in Table 4-1 together with the tonnages of nitrogen oxides abated.

Table 4-1: Air emissions

	Units	SNCR	SCR	FGR + SNCR
Nitrous oxide	mg/m^3	20	20	20
Ammonia	mg/m^3	15	15	15
NO$_x$, unabated concentration	mg/m^3	350	350	315
NO$_x$, unabated rate	tpa	119	119	107
NO$_x$, abated concentration	mg/m^3	100	80	100
NO$_x$ released after abatement	tpa	34	27	34
NO$_x$ removed	tpa	85	92	73

A long-term abated emission concentration of 80 mg/Nm^3 (11% reference oxygen content) is used for SCR for the purposes of this BAT assessment, since this is the level that the technology 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 100 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 4-2 shows the predicted ground level concentrations for the two options.

Table 4-2: Air emissions

Abatement system:		SNCR	SCR	SNCR + FGR
Long term				
Process Contribution (PC)	µg/m³	1.09	0.87	1.09
Background	µg/m³	15.70	15.70	15.70
Predicted Environmental Contribution (PEC)	µg/m³	16.79	16.57	16.79
Air Quality Objective	µg/m³	40	40	40
PC as % of AQO		2.72%	2.18%	2.72%
PEC as % of AQO		41.98%	41.43%	41.98%
Short term				
Process Contribution (PC)	µg/m³	7.59	6.07	7.59
Background	µg/m³	31.40	31.40	31.40
Predicted Environmental Contribution (PEC)	µg/m³	38.99	37.47	38.99
Air quality objective	µg/m³	200	200	200
PC as % of AQO		3.80%	3.04%	3.80%
PEC as % of AQO		19.50%	18.74%	19.50%

The short-term air quality impacts associated with all three NOx abatement options can be screened as insignificant. However, the long-term air quality impacts cannot be screened as insignificant. The PEC has been calculated to be less than 70% of the long-term environmental standard. Further analysis has been undertaken within the Air Quality Assessment (refer to Appendix E of the Supporting Information).

It can be seen that using SCR reduces the long-term PEC by 0.55% of the air quality objective and the short-term PEC by 0.76% of the air quality objective when compared to either SNCR or SNCR + FGR.

4.2.2 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, which concludes that "*nitrogen deposition due to emissions from the Proposed Development will not have a significant effect on the identified ecological sites*".

4.2.3 Emissions to water

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

4.2.4 Photochemical Ozone Creation Potential

Nitrogen dioxide has a POCP of 2.8, whilst nitrogen oxide has a POCP of -42.7. Assuming 10% of NO$_x$ is released as NO$_2$ and the rest as NO, the POCP is approximately -1,300 for the SNCR options and approximately -1,000 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.

4.2.5 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 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 Induced Draft (ID) Fan.

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 – assumed to be imported from the grid - the assumption of 372 $kgCO_2$/MWh has been used, as applied in the greenhouse gas assessment presented in Appendix E of the Supporting Information. The global warming potential has been calculated as follows:

Table 4-3: Global Warming Potential

	Units	SNCR	SCR	SNCR + FGR
Power consumed	kWe	120	230	150
Power not generated	kWe		140	
Change in exported power	MWh pa	900	2,900	1,200
GWP	t CO_2 eq pa	300	1,100	400

4.2.6 Raw materials

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

Table 4-4: Raw materials

	Units	SNCR	SCR	SNCR + FGR
Water	tpa	540	260	460
Urea	tpa	380	180	330

4.2.7 Waste streams

There will be no additional residues generated from any of the NO_x abatement options.

4.3 Costs

The estimated costs associated with each option are presented in Table 4-5. 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 lifetime with a rate of return of 9%, using the method recommended in Technical Guidance Note EPR-H1.

Table 4-5: Costs

	SNCR	SCR	SNCR + FGR
Capital cost	£200,000	£3,900,000	£500,000
Annualised capital cost	£19,000	£380,000	£49,000
Maintenance	£4,000	£78,000	£10,000
Water and reagents	£75,000	£35,000	£63,000
Loss of exported power	£51,000	£165,000	£68,000
Total annualised cost	**£149,000**	**£658,000**	**£190,000**

4.4 Conclusions

A summary comparison of the three options is provided in Table 4-6.

Table 4-6: Comparison table

	Units	SNCR	SCR	SNCR + FGR
NO_x released after abatement	tpa	34	27	34
NO_x removed	tpa	85	92	73
POCP	t ethylene-eq pa	-1,300	-1,000	-1,300
Global Warming Potential	t CO_2 eq pa	300	1,100	400
Urea used	tpa	380	180	330
Total annualised cost	£ pa	£149,000	£658,000	£190,000
Average cost per tonne NO_x abated	£ p.t NO_x	£1,750	£7,150	£2,600

As can be seen, incorporating SCR into the design of the Facility to abatement emissions of NOx:

1. increases the annualised costs by approximately £500,000;
2. abates an additional 7 tonnes of NOx per annum;
3. reduces the benefit of the Facility in terms of the global warming potential by approximately 800 tonnes of CO_2;
4. reduces reagent consumption by approximately 200 tonnes per annum; and
5. costs an additional ~£73,000 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 49%. It has no effect on the direct environmental impact of the plant, but it increases the impact on climate change by approximately 100 tonnes of CO_2 per annum. However, it reduces reagent consumption by approximately 50 tonnes per annum. This is based on the assumption that FGR reduces the NOx generation within the furnace.

Therefore, taking the above into consideration, the use of SNCR either with or without FGR is considered to represent BAT for the abatement of NOx within the Facility. The proposed designs currently include FGR, due to the anticipated benefits in reduced NOx generation within the furnace.

5 Combustion techniques

5.1 Options considered

The available techniques for waste combustion have been reviewed in section 2.6.1 of the Supporting Information. The 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 and they have been used widely within the healthcare sector in treating clinical waste. Rotary kilns can operate at higher temperatures than other systems due to the absence of exposed metal surfaces, and can therefore be used to incinerate hazardous, clinical and industrial wastes.

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 options would require further abatement to achieve the necessary 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: NOx emissions

Option	NOx emissions from furnace (mg/Nm^3)
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 between the combustion systems.

5.2.3 Emissions to water

There are no emissions to water for any of the combustion systems.

5.2.4 Photochemical Ozone Creation Potential

There would be no change to POCP for any of the combustion systems.

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 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 other power stations would be different in each case.

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	73,000	73,000	56,000
Parasitic load	MWh pa	12,000	13,000	12,000
GWP	t CO2 eq pa	-22,700	-22,300	-16,000

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**
Urea	tpa	400	400	500
Sand	tpa		1,170	

5.2.7 Waste streams

The three options produce several solid waste streams.

* It is assumed that most metals within the waste will have been removed during any pre-treatment of the incoming waste. Therefore, it is assumed that this will be identical for both options 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 three options produce APC residues. For the purposes of this assessment, it is assumed that the systems will generate comparable quantities of APC residue.

Estimated figures are presented in Table 5-4.

Table 5-4: Waste streams

	Units	Grate	Fluidised bed	Rotary kiln
Bottom ash	tpa	20,000	7,870	20,000
Boiler ash	tpa		13,300	
APC residues	tpa	5,000	5,000	5,000
Total ash	tpa	25,000	26,170	25,000

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

5.3 Costs

The estimated costs associated with each option 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 options. Therefore, it has not been possible to consider the capital costs for the available technologies 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 options, so these were not considered in this assessment.

Table 5-5: Annual material costs and revenues

	Grate	Fluidised bed	Rotary kiln
Reagents	£80,000	£200,000	£100,000
Residue Disposal	£1,570,000	£1,590,000	£1,570,000
Annual Reagent and Residue Costs (Materials)	£1,650,000	£1,790,000	£1,670,000
Annual Power Revenue	£3,477,000	£3,420,000	£2,508,000

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.

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 options.

Table 5-6: Option comparison summary

		Grate	Fluidised bed	Rotary kiln
Global Warming Potential	t CO2 eq pa	-22,700	-22,300	-16,000
Urea consumption	tpa	400	400	500
Total ash	tpa	25,000	26,170	25,000
Annual total materials costs (reagents and residues)		£1,650,000	£1,790,000	£1,670,000
Annual power revenue		£3,477,000	£3,420,000	£2,508,000

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

The material costs are approximately 8% higher for the fluidised bed than the grate, whereas the grate system will have a slightly higher power revenue, but it is acknowledged that it is marginal. The grate system will be able to process a varying waste composition compared to a fluidised bed system which requires a consistent and homogenous fuel and therefore possibly requiring additional treatment of the waste.

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 municipal waste (including 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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