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BAT ASSESSMENT REPORT

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1 INTRODUCTION

This report presents quantitative BAT assessments for acid gas abatement, nitrogen oxides abatement and Energy from Waste (EfW) technology for the Kelvin Energy Recovery Facility (the Facility).

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

1.1 Assumptions

The Facility will use a moving grate as the combustion technology. The plant will be a single stream, with a nominal design capacity of approximately 45 tonnes of MSW, C&I, and RDF per hour, with a net calorific value (NCV) of 10 MJ/kg. This equates to a nominal design capacity of 360,000 tonnes per annum, assuming 8,000 hours operation per annum.

The facility will generate up to 35 MWe with a parasitic load of 4.0 MWe.

In addition, it has been assumed that the reagent within the SNCR system will be ammonia.

For the purposes of this report 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 .... £1 per tonne
- Quick Lime .... £90 per tonne
- Hydrated Lime .... £94 per tonne
- Sodium Bicarbonate..£155 per tonne
- Activated Carbon...£600 per tonne
- Ammonia..£135 per tonne
- Sand (with defined particle size distribution)................................ £40 per tonne
- Bottom Ash Processing... £10 per tonne
- Lime APCR Disposal ....£125 per tonne
- Sodium bicarbonate APCR Disposal..£150 per tonne
- Landfill Tax (1 April 2017) ....£86.10 per tonne
- Imported power ....£130 per MWh
- Electricity revenue .... £45 per MWh

2   ACID GAS ABATEMENT

2.1   Options Considered

There are currently three technologies widely available for acid gas abatement on EfW 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 EfW's 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 Annex 5. The acid gas emissions were assessed at the daily emission concentrations of 50 mg/m^3 for sulphur dioxide and 10 mg/m^3 for hydrogen chloride.

The table below 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 2.1 – Emissions to Air					
Abatement System		**Dry**		**Semi-dry**	
Pollutant		**SO_2**	**HCl**	**SO_2**	**HCl**
Unabated emission concentration	mg/m^3	480	900	480	900
Unabated emission rate	t/a	1,090	2,040	1,090	2,040
Emission concentration	mg/m^3	50	10	50	10
Emission rate	t/a	110	20	110	20
Emissions abated	t/a	980	2,020	980	2,020
Process Contribution (PC)	ug/m^3	1.90	10	1.90	10
Background	ug/m^3				
Predicted Environmental Contribution (PEC)	ug/m^3	16.30	2.84	16.30	2.84
Air Quality Objective	ug/m^3	125	750	125	750
PC as % of AQO		1.52%	1.4%	1.52%	1.4%
PEC as % of AQO					

The short-term impact of the plant is 1.52% of the daily average air quality objective for SO_2 and 1.4% of the hourly air quality objective for HCl. The impact of HCl and SO_2 is considered to be insignificant when applying the criteria stated in Environment Agency guidance note H1. As the impact is screened as insignificant the PEC has not been calculated for the purposes of this assessment.

A more detailed assessment of impacts from the release of sulphur dioxide and hydrogen chloride is presented within the air quality assessment within Annex 5.

2.2.2 Deposition to Land

The impact of acid deposition on sensitive habitats has been assessed in the Air Quality Assessment presented in Annex 5. As can be seen from this assessment, the impact of acid deposition on sensitive receptors is not considered to be 'insignificant' at all habitat features.

2.2.3 Emissions to Water

There are no emissions to water for either the dry or the semi-dry systems, therefore the impact of these systems is the same.

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 530 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 359 kg CO_2 per MWh has been used, as applied in the greenhouse gas assessment presented in Annex 5.

This is shown in the table below.

Table 2.2 – Global Warming			Dry	Semi-Dry
Power consumed	kWh/t		30	28.5
	MWh p.a.		10,800	10,260
Generation lost	MWh p.a.			-12,200
Power not exported	MWh p.a.		10,800	22,460
GWP	t CO_2 p.a.		3,900	8,100

2.2.6   Raw Materials

The estimated consumption of raw materials for both options is shown below.

Table 2.3 – Raw Materials		Dry	Semi-Dry
Additional water consumption compared to a dry system	t.p.a.		33,890
Quick Lime	t.p.a.		5,000
Hydrated Lime	t.p.a.	18,200	
Powdered Activated Carbon (PAC)	t.p.a.	540	540

2.2.7   Waste Streams

The only waste stream associated with the acid gas abatement treatment technologies is the Air Pollution Control Residues. These would be a hazardous waste. The production rate for both systems would be approximately 20,000 tonnes per annum.

2.3 Costs

The estimated costs associated with each option are presented below. 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 2.4 – Costs		
	Dry	**Semi-Dry**
Capital Cost	£15,500,000	£16,300,000
Annualised Capital Cost	£1,510,000	£1,590,000
Maintenance	£775,000	£815,000
Reagents and residues	£6,355,000	£5,387,000
Loss of exported power	£594,000	£1,235,000
Total Annualised Cost	**£9,234,000**	**£9,027,000**

2.4 Conclusions

The table below compares the options.

Table 2-5 – Comparison Table			
		Dry	**Semi-Dry**
SO$_2$ abated	t p.a.	980	980
Photochemical Ozone Creation Potential (POCP)	t ethylene-eq p.a.	530	530
Global Warming Potential	t CO$_2$ p.a.	3,900	8,100
Additional water consumption compared to a dry system	t.p.a.		12,200
APC Residues	t p.a.	20,000	20,000
Annualised Cost	£ p.a.	£9,234,000	£9,027,000

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 reduced global warming potential and a reduced annualised cost 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 low water consumption and 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 and are listed below.

(1)   Selective Catalytic Reduction (SCR) 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) 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 the table below together with the tonnages of nitrogen oxides abated.

Table 3.1– Air Emissions				
		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	350	350	350
NO$_x$, unabated	t/a	790	790	790
NO$_x$, abated concentration	mg/m^3	200	70	200
NO$_x$ released after abatement	t/a	450	160	450
NO$_x$ removed	t/a	340	630	340

The emission rates for nitrogen oxides and ammonia are shown in the table above.

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, 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), operate to match the emission requirement of 200 mg/Nm^3 as proposed within the Supporting Information.

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 Annex 5. The table below 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/m³	0.60	0.21	0.60
Background	µg/m³	29.50	29.50	29.50
Predicted Environmental Contribution (PEC)	µg/m³	30.10	29.71	30.10
Air Quality Objective	µg/m³	40	40	40
PC as % of AQO		1.50%	0.53%	1.50%
PEC as % of AQO		75.25%	74.28%	75.25%
Short Term				
Process Contribution (PC)	µg/m³	0.18	0.06	0.18
Background	µg/m³	29.50	29.50	29.50
Predicted Environmental Contribution (PEC)	µg/m³	59.18	59.06	59.18
Air Quality Objective	µg/m³	200	200	200
PC as % of AQO		0.09%	0.03%	0.09%
PEC as % of AQO		29.59%	29.53%	29.59%

It can be seen that there are no predicted exceedences of air quality objectives for any of the options. Using SCR reduces the long term PEC by 0.97% of the air quality objective and the short term PEC by 0.06% 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 Annex 5. As can be seen from the results presented in the report, whilst the impact of nitrogen deposition cannot be screened as insignificant at all ecological receptors, it is concluded that there will be 'no likely significant effects' of nitrogen deposition.

3.2.3 Emissions to Water

There are no emissions to water from any of the NOx 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 NOx is released as NO2 and the rest as NO, the POCP is -17,200 for the SNCR options and -6,100 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, 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 plant 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 reduces 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 359 kg CO_2 equivalent per MWh has been used, as applied in the greenhouse gas assessment presented in Annex 5.

Table 3.3 – Global Warming Potential				
		SNCR	**SCR**	**SNCR + FGR**
Power consumed	kWe	400	820	540
Power not generated	kWe	-	590	-
Change in exported power	MWh p.a.	3,200	11,300	4,300
GWP	t CO_2 p.a.	1,100	4,100	1,500

3.2.6 Raw Materials

The estimated consumption of raw materials for each option is shown below.

Table 3.4 – Raw Materials				
		SNCR	**SCR**	**SNCR + FGR**
Water	t.p.a.	5,700	4,640	5,700
Ammonia	t.p.a.	2,200	1,800	2,200

3.2.7 Waste Streams

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

3.3 Costs

The estimated costs associated with each option are presented below. 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 3.5 – Costs			
	SNCR	**SCR**	**SNCR + FGR**
Capital Cost	£700,000	£10,500,000	£1,500,000
Annualised Capital Cost	£68,000	£1,022,000	£146,000
Maintenance	£14,000	£210,000	£30,000
Water and reagents	£452,000	£370,000	£452,000
Loss of exported power	£176,000	£622,000	£237,000
Total Annualised Cost	£710,000	£2,224,000	£865,000

3.4 Conclusions

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

Table 3.6 – Comparison Table				
		SNCR	**SCR**	**SNCR + FGR**
NOx released after abatement	t p.a.	450	160	450
NOx removed	t p.a.	340	630	340
Photochemical Ozone Creation Potential (POCP)	t ethylene-eq p.a.	-17,200	-6,100	-17,200
Global Warming Potential	t CO_2 p.a.	1,100	4,100	1,500
Ammonia	t p.a.	2,200	1,800	2,200
Annualised Cost	£ p.a.	£710,000	£2,224,000	£865,000
Average cost per tonne NOx abated	£ p.t NOx.	£1,870	£3,530	£2,280

As can be seen from the table above, applying SCR to the Installation:
(1)   increases the annualised costs by more than £1.4 million;
(2)   abates an additional 290 tonnes of NOx per annum;
(3)   reduces the benefit of the facility in terms of the global warming potential by more than 2,500 tonnes of CO_2;
(4)   reduces reagent consumption by approximately 1,400 tonnes per annum; and
(5)   costs more than an additional £1,240 per additional tonne of NOx abated.

The additional costs associated with an SCR are not considered to represent BAT for the Installation. Therefore, SNCR is considered to represent BAT for the Installation.

The two SNCR options, with and without FGR, are very similar. FGR results in a reduction of reagent consumption, but requires more power to operate, and therefore it has a higher global warming potential and slightly higher total annualised costs.

The choice of whether to include FGR is supplier dependent. Some furnace suppliers have designed their combustion systems to operate with FGR and these suppliers can gain benefits of reduced NOx generation from the use of FGR. Other suppliers have focused on reducing NOx generation through the control of primary and secondary air supply and the furnace design, and they can gain little if any benefit from the use of FGR. On this basis, and since the differences between the two options are small, the use of FGR for the proposed installation will be considered during the technology procurement process. It is proposed that a pre-operation conditions is included within the EP, which requires that the Operator provides confirmation to the EA of the final designs for the abatement of NOx.

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 above. We have therefore only considered the two alternative reagents for a dry system – hydrated 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.
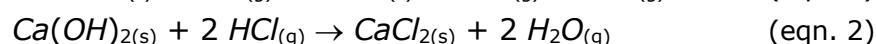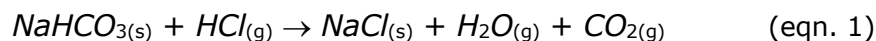
4.2.5   Global Warming Potential

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 around 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 $NaHCO_3$ is significantly higher than $Ca(OH)_2$, with bicarbonate costing almost 65% more than hydrated 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 the sodium based residues is 20% higher than lime residues giving a disposal cost for sodium bicarbonate of £150 /te.

The operating costs for the two options are compared below, 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	94
Cost of residue disposal[1]	£/tonne	150	125
Overall Cost	£/op. hr/kmol	29.5	16.9
Ratio of costs		1.74	1.00

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

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

- The costs per kmol HCl abated are almost 75% 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 reviewed in section 2.6.1 of the supporting information. The assessment has been expanded to provide a cost-benefit analysis of moving grates and fluidised beds.

(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 pre-processed RDF will satisfy the fuel requirements for a fluidised bed combustion system.

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 NOx concentrations from the furnaces would be different, both 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 NOx. This is considered in a later section.

Table 5.1 – NOx Emissions	
Option	**NOx emissions from furnace (mg/Nm^3)**
Moving Grate	320-380[1]
Fluidised Bed	250-300[1]
(1) Presented at 11% oxygen with standard reference conditions	

5.2.2   Deposition to Land

Deposition from atmospheric emissions would also 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 fluidised beds, whilst having lower emissions of nitrogen dioxide can have elevated emissions of nitrous oxide. Nitrous oxide is a greenhouse gas with a global warming potential (GWP) with 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 we have 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 10mg/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.

In order to calculate the global warming potential of electricity consumption, the figure of 359 kg CO_2 equivalent per MWh has been used, as applied in the greenhouse gas assessment presented in Annex 5.

The results are presented in the table below. 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**
Power generated	MWh p.a.	312,000	312,000
Parasitic Load	MWh p.a.	32,000	35,000
GWP	t CO_2 p.a.	-101,000	-99,000

5.2.6 Raw Materials

The estimated consumption of raw materials for each option is shown below.

Table 5.3 – Raw Materials			
		Grate	**Fluidised Bed**
Ammonia	t.p.a.	2,200	2,500
Sand	t.p.a.		4,110

5.2.7 Waste Streams

The two 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. It is therefore assumed that it will be identical for both options and has not been considered further.
- The bottom ash generation is lower for fluidised beds. Assuming a suitably licensed facility can be identified, boiler ash will be recovered as a secondary aggregate.

- Fluidised beds have much greater carry-over of fine particles and so produce an additional fly ash stream, which is removed in a cyclone before the acid gas abatement reagent is added. This separate fly ash stream (Fly Ash) could be usable for building aggregate, but this is not certain and it is possible that it will need to be sent to a hazardous landfill. For the purposes of this assessment it has been assumed that it cannot be used as a building aggregate and requires disposal in a non-hazardous landfill.
- Both options produce APC residues. The fluidised bed option would generate less APC residue because more of the fly ash will have been removed from the gas stream.

Estimated figures are shown in the table below.

Table 5.4 – Waste Streams			
		Grate	**Fluidised Bed**
Bottom Ash	t p.a.	86,000	32,810
Fly Ash	t.p.a.		57,300
APC Residues	t p.a.	20,000	17,200

5.3   Costs

The estimated costs associated with each option are presented below.

Fluidised bed technology is typically up to 5% more expensive, 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 two 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 for the proposed facility in this assessment.

Table 5.5 – Annual Material Costs and Revenues		
	Grate	**Fluidised Bed**
Reagents	£120,000	£490,000
Residue Disposal	£5,080,000	£5,120,000
Total Materials Costs	£5,200,000	£5,610,000
Power Revenues	£15,400,000	£15,235,000

As can be seen from the table above, the fluidised bed option has slightly lower 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 power generated by the two systems is comparable, if we assume that the facility will only receive pre-processed fuels and therefore the parasitic load associated with fuel preparation is excluded from the assessment.

For a fluidised bed there may be costs associated with screening the fuel 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

The table below compares the two options.

Table 5.6 – Option Comparison Summary			
		Grate	**Fluidised Bed**
Global Warming Potential	t CO_2 p.a.	-101,000	-99,000
Ammonia Consumption	t.p.a.	2,200	2,500
Residues		-	Less bottom ash, more fly ash
Total Materials Costs	p.a.	£5,200,000	£5,610,000
Power Revenue	p.a.	£15,400,000	£15,235,000

The grate has a lower global warming potential than the fluidised bed, however it would use over 10% more ammonia to abate emissions of NOx.

Both combustion technologies will produce similar quantities of ash, although the fluidised bed produces more fly ash.

The material costs are approximately 5% higher for the fluidised bed than the grate, whereas the grate system will have a slightly higher power revenue. This is considered to outweigh the higher ammonia consumption. However, it is acknowledged that it is marginal and should be noted that this assessment is based on the assumption that the incoming waste will not require any additional treatment and will be suitable for combustion within a fluidised bed.

As stated within the qualitative BAT assessment, refer to section 2.6.1 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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