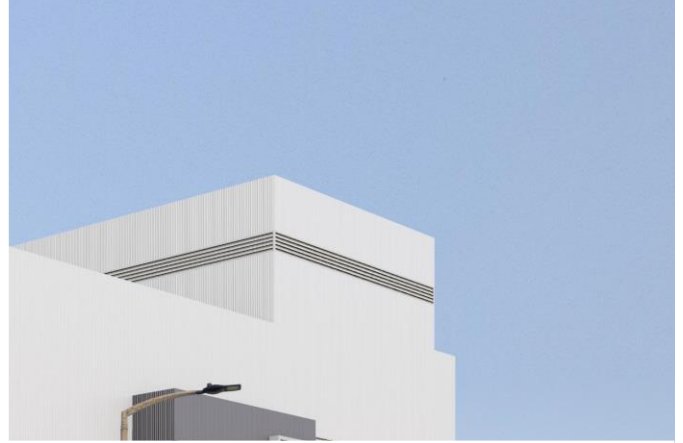


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**BH EnergyGap (Doncaster) Ltd**

BAT Assessment

## Document approval

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# 1 Introduction

In accordance with the requirements of Environment Agency guidance EPR5.01 – Incineration of Waste, this report presents quantitative BAT assessments for acid gas abatement, nitrogen oxides abatement and combustion technology for the Doncaster Energy Recovery Facility (the Facility).

Each assessment follows the structure of Technical Guidance Note EPR-H1 Annex K (Cost and Benefit Analysis associated with the IED) and includes comments on all of the environmental parameters identified in the guidance.

## 1.1 Assumptions

The Facility will be a single stream design, with a design capacity of approximately 37.6 tonnes of waste fuel per hour, with an average net calorific value (NCV) of 11.5 MJ/kg. This equates to a design capacity of approximately 301,000 tonnes per annum, assuming 8,000 hours operation per annum. However, it is acknowledged that the plant is capable of processing more than 301,000 tpa of waste through increased availability and/or the processing of waste with a lower NCV.

The Facility will generate approximately 38.9 MWe of electricity at the design point, and have a parasitic load of approximately 12%, or 4.7 MWe. Therefore, the export capacity of the Facility at the design point will be approximately 34.3 MWe. However, the power exported by the Facility will fluctuate depending on the quality of the waste fuel, the ambient temperature and whether heat is exported.

For the purposes of the EP application it has been assumed that the reagent within the SNCR system will be ammonia solution; however, the selection of the reagent will be subject to detailed design and it is proposed that this is confirmed via pre-operational condition.

For the purposes of this report, a quantitative assessment of the available technologies for the proposed capacity has been undertaken using data obtained by Fichtner from a range of different projects using the technologies identified within this assessment.

In the relevant operating costs sections of this assessment, the following unit costs have been assumed:

- Water ..... £0.85 per m<sup>3</sup>
- Lime Slurry ..... £90 per tonne
- Lime (hydrated) ..... £91 per tonne
- Sodium Bicarbonate..... £155 per tonne
- Activated Carbon ..... £710 per tonne
- Ammonia..... £164 per tonne
- Sand (with defined particle size distribution) ..... £100 per tonne
- Bottom Ash Processing ..... £25 per tonne
- Lime Air Pollution Control residue (APCr) Disposal ..... £136 per tonne
- Sodium bicarbonate APCR Disposal..... £163 per tonne
- Landfill Tax (1 April 2020) ..... £94 per tonne
- Imported power ..... £67 per MWh
- Electricity revenue ..... £50 per MWh

## 2 Combustion techniques

### 2.1 Options considered

The available techniques for waste combustion have been reviewed in a qualitative BAT assessment within Section 2.6.1 of the supporting information. The qualitative 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 fuels with varying net calorific values, such as those proposed to be processed at the Facility. 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, such as a pre-processed refuse-derived fuel (or RDF).

### 2.2 Environmental performance

#### 2.2.1 Emissions to air

The emissions to atmosphere would not be affected by the choice of combustion technology. Although NO<sub>x</sub> concentrations from the furnaces would be different, both options would require further abatement to achieve the relevant emission standards to comply with the BAT-AELs. This means that the actual effect would be to change the amount of reagent required to abate the NO<sub>x</sub>. This is considered in Section 2.2.6.

Table 2-1: Combustion techniques – NO<sub>x</sub> emissions comparison

Option	NO <sub>x</sub> emissions from furnace (mg/Nm <sup>3</sup> ) <sup>(1)</sup>
Moving Grate	320-380
Fluidised Bed	250-300

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

#### 2.2.2 Deposition to land

There would be no change to deposition from atmospheric emissions for either combustion system.

#### 2.2.3 Emissions to water

There are no emissions to water for either combustion system.

#### 2.2.4 Photochemical Ozone Creation Potential

There would be no change to the Photochemical Ozone Creation Potential (POCP) for either combustion system.

#### 2.2.5 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 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 10 mg/Nm<sup>3</sup> (in accordance with data presented within the Waste Incineration BREF).

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 349 kg CO<sub>2</sub> equivalent per MWh has been used, as justified in the greenhouse gas assessment – refer to Appendix E of the Application.

The results are presented in Table 2-2 below. For both combustion technologies the overall global warming potential is less than zero, as there is a net reduction in carbon dioxide equivalents due to the displacement of primarily fossil fuel power generation. Thus, more negative figures are better.

*Table 2-2: Combustion techniques – global warming potential comparison*

		<b>Grate</b>	<b>Fluidised Bed</b>
Power generated	MWh pa	311,000	311,000
Parasitic Load	MWh pa	4.7	5.1
Global warming potential	t CO2 eq pa	-95,600	-94,200

### 2.2.6 Raw materials

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

*Table 2-3: Combustion techniques – raw materials comparison*

	<b>Units</b>	<b>Grate</b>	<b>Fluidised Bed</b>
Ammonia	tpa	1,200	800
Sand	tpa	-	3,960

### 2.2.7 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 and source-segregation of the incoming waste. It is therefore assumed that it will be identical for both options and has not been considered further.
- The fluidised bed will produce more non-hazardous ash than a grate 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.

- The small quantity of boiler ash that is produced within a grate system is included within the value for bottom ash, as in typical moving grate incineration plants these two ash streams are often combined.
- Both options produce APCr. For the purposes of this assessment it is assumed that both options will generate the same quantity of APCr (i.e. it is assumed the emissions of acid gases will be comparable between each combustion system and that similar amounts of acid gas reagent will be required, resulting in similar production rates of APCr).

Estimated figures are shown in Table 2-4 below.

*Table 2-4: Combustion techniques – waste streams comparison*

	<b>Units</b>	<b>Grate</b>	<b>Fluidised Bed</b>
Bottom Ash	tpa	67,100	26,300
Boiler Ash	tpa	-	44,700
APCr	tpa	11,200	11,200

### 2.3 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 Facility in this assessment.

*Table 2-5: Combustion techniques – comparison of annual material costs and revenues*

	<b>Grate</b>	<b>Fluidised Bed</b>
Reagent costs (subtotal)	£200,000	£530,000
Residue disposal costs (subtotal)	£4,260,000	£4,360,000
Annual reagent and residue costs (materials)	£4,460,000	£4,890,000
Annual power revenue	£13,686,300	£13,486,500

As can be seen from Table 2-5, the fluidised bed 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.

If it is assumed that the Facility will only receive pre-processed fuels and the parasitic load associated with fuel preparation is excluded from the assessment, the power generated by the two systems is comparable. However, for a fluidised bed, the fuel may require additional screening and processing to ensure that there are no contaminants which could impact on the operation of the fluidised bed. This potential additional parasitic load has not been allowed for within the assumed power revenues.



## 2.4 Conclusions

The table below compares the two options.

*Table 2-6: Combustion techniques – option comparison summary*

		<b>Grate</b>	<b>Fluidised bed</b>
Global warming potential	t CO <sub>2</sub> eq pa	-95,600	-94,200
Ammonia consumption	tpa	1,200	800
Residues (total ash and sand)	tpa	78,300	82,200
Annual total materials costs (reagents & residues)		£4,460,000	£4,890,000
Annual power revenue		£13,686,300	£13,486,500

The grate has a similar global warming potential to the fluidised bed but would consume more ammonia during 'typical' operations. The use of a fluidised bed would generate a higher quantity of residues than a moving grate and would also have higher operating costs. Although these differences are noticeable, it is acknowledged that they are marginal.

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 these differences are marginal.

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.

The Operator has extensive knowledge of moving grate systems and does not consider that a fluidised bed is a suitable technology for the Facility. Due to the robustness of grate combustion systems, they are considered to represent BAT for the Facility.

## 3 Nitrogen oxides abatement

### 3.1 Options Considered

Three options have been considered for NO<sub>x</sub> abatement and are listed below.

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).

For the purposes of this BAT assessment it is assumed that ammonia will be used as a reagent within the NO<sub>x</sub> abatement system.

### 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: NO<sub>x</sub> abatement – emissions to air comparison*

	Units	SNCR	SCR	FGR + SNCR
Nitrous oxide	mg/m <sup>3</sup>	10	10	10
Ammonia	mg/m <sup>3</sup>	10	10	10
NO <sub>x</sub> , unabated concentration	mg/m <sup>3</sup>	380	380	340
NO <sub>x</sub> , unabated rate	tpa	730	730	660
NO <sub>x</sub> , abated concentration	mg/m <sup>3</sup>	120	80	120
NO <sub>x</sub> released after abatement	tpa	230	150	230
NO <sub>x</sub> removed	tpa	500	580	430

A long term abated emission concentration of 80 mg/Nm<sup>3</sup> (11% reference oxygen content) is used for SCR for the purposes of this BAT assessment, as this is an appropriate level that the technology can achieve on a long-term basis, based on operational data presented in the Waste Incineration BREF (the BREF). The two SNCR systems, with and without Flue Gas Recirculation (FGR), would be required to achieve an emission limit of 120 mg/Nm<sup>3</sup>. The ELV of 120 mg/Nm<sup>3</sup> has been applied to be consistent with the Air Quality Assessment and the upper range of the BREF.

The unabated emission with FGR is assumed to be 10% lower.

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, refer to Appendix E of the Application. Table 3-2 below shows the predicted ground level concentrations for the three options.

Table 3-2: NO<sub>x</sub> abatement – air emissions comparison

Abatement System:		SNCR	SCR	SNCR + FGR
<b>Long Term</b>				
Process Contribution (PC)	µg/m <sup>3</sup>	0.62	0.41	0.62
Background	µg/m <sup>3</sup>	20.70	20.70	20.70
Predicted Environmental Contribution (PEC)	µg/m <sup>3</sup>	21.32	21.11	21.32
Air quality objective (AQO)	µg/m <sup>3</sup>	40	40	40
PC as % of AQO		1.55%	1.03%	1.55%
PEC as % of AQO		53.30%	52.78%	53.30%
<b>Short Term</b>				
Process Contribution (PC)	µg/m <sup>3</sup>	8.21	5.47	8.21
Background	µg/m <sup>3</sup>	41.40	41.40	41.40
Predicted Environmental Contribution (PEC)	µg/m <sup>3</sup>	49.61	46.87	49.61
Air quality objective (AQO)	µg/m <sup>3</sup>	200	200	200
PC as % of AQO		4.11%	2.74%	4.11%
PEC as % of AQO		24.81%	23.44%	24.81%

The short-term PC of process emissions for all systems can be screened out as insignificant in accordance with the criteria as presented in the EA Guidance, titled, 'Air emissions risk assessment for your environmental permit' (referred to as the Air Emissions guidance).

The long-term PC of process emissions for all NO<sub>x</sub> abatement systems cannot be screened out as insignificant in accordance with the Air Emissions guidance. Therefore, the PEC has been calculated. As the PEC values are less than 70% of the long-term environmental standard, they can be described as 'not significant', in accordance with the Air Emissions guidance.

### 3.2.2 Deposition to land

The impact of nitrogen deposition on sensitive habitats has been assessed in the Air Quality Assessment (refer to Appendix E of the Application). As can be seen from the results presented in the Air Quality Assessment, the PC is less than 1% of the relevant Critical Loads at the European Designated Site and less than 100% of the Critical Loads at all local nature sites and can be screened out.

### 3.2.3 Emissions to water

There are no emissions to water from any of the NO<sub>x</sub> abatement systems.

### 3.2.4 Photochemical Ozone Creation Potential

Nitrogen dioxide (NO<sub>2</sub>) has a photochemical ozone creation potential (POCP) of 2.8 and nitrogen oxide (NO) has a POCP of -42.7. Assuming that 10% of NO<sub>x</sub> is released as NO<sub>2</sub> and the rest as NO, the POCP is -8,800 for the SNCR options and -5,700 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, thus 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 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, typically using steam, which will reduce amount of steam available for power generation.

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 349 kg CO<sub>2</sub> equivalent per MWh has been used, as justified in the greenhouse gas assessment - refer to Appendix E of the Application.

*Table 3-3: NO<sub>x</sub> abatement – global warming potential comparison*

	Units	SNCR	SCR	SNCR + FGR
Power consumed	kWe	340	690	450
Power not generated	kWe	-	510	-
Change in exported power	MWh pa	2,700	9,600	3,600
Global warming potential	t CO <sub>2</sub> eq pa	900	3,400	1,300

### 3.2.6 Raw materials

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

*Table 3-4: NO<sub>x</sub> abatement – raw materials comparison*

	Units	SNCR	SCR	SNCR + FGR
Water	tpa	3,000	1,030	2,600
Ammonia	tpa	1,200	400	1,000

### 3.2.7 Waste streams

There will be no additional residues generated from any of the NO<sub>x</sub> 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 the H1 guidance.

Table 3-5: NO<sub>x</sub> abatement – costs

	SNCR	SCR	SNCR + FGR
Capital Cost	£600,000	£9,600,000	£1,300,000
Annualised Capital Cost	£58,000	£934,000	£127,000
Maintenance	£12,000	£192,000	£26,000
Water and reagents	£196,000	£66,000	£169,000
Loss of exported power	£135,000	£480,000	£180,000
<b>Total Annualised Cost</b>	<b>£401,000</b>	<b>£1,672,000</b>	<b>£502,000</b>

### 3.4 Conclusions

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

Table 3-6: NO<sub>x</sub> abatement – comparison table

	Units	SNCR	SCR	SNCR + FGR
NO <sub>x</sub> released after abatement	tpa	230	150	230
NO <sub>x</sub> removed	tpa	500	580	430
Photochemical Ozone Creation Potential (POCP)	t ethylene-eq pa	-8,800	-5,700	-8,800
Global warming potential	t CO <sub>2</sub> eq pa	900	3,400	1,300
Ammonia Used	tpa	1,200	400	1,000
Total annualised cost	£ pa	£401,000	£1,672,000	£502,000
Average cost per tonne NO <sub>x</sub> abated	£ p.t NO <sub>x</sub> .	£800	£2,880	£1,170

As can be seen from the table above, applying SCR for the abatement of NO<sub>x</sub>:

1. increases the annualised costs by approximately £1,271,000;
2. abates approximately an additional 80 tonnes of NO<sub>x</sub> per annum;
3. reduces the benefit of the facility in terms of the global warming potential by approximately 2,500 tonnes of CO<sub>2</sub>;
4. reduces reagent consumption by approximately 800 tonnes per annum; and
5. costs approximately 3.6 times more per tonne of NO<sub>x</sub> abated compared to an SNCR system.

The incremental cost for the additional abatement from SCR compared to SNCR is approximately £15,890 per additional tonne of NO<sub>x</sub> abated.

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 design of the NO<sub>x</sub> abatement system increases the cost per tonne of NO<sub>x</sub> abated by approximately 50% compared to SNCR without FGR. It has no effect on the direct environmental impact of the plant, but it increases the impact on climate change by approximately 400 tonnes of CO<sub>2</sub> per annum while reducing ammonia consumption by approximately 200 tonnes per annum compared to an SNCR system without FGR.

However, this is based on the assumption that FGR reduces the NOx generation within the furnace. This is not necessarily the case for all furnace manufacturers. Some designs can achieve lower levels of NOx without FGR. For these technology providers incorporating FGR into the design might even cause additional problems e.g. reduced availability of the plant.

The proposed designs do not currently include FGR. However, it is requested that a pre-operational condition is included within the permit to allow details of the NOx abatement system to be confirmed during detailed design of the Facility. Therefore, taking the above into consideration, the use of SNCR with or without FGR is considered to represent BAT for the abatement of NOx within the Facility.

## 4 Acid gas abatement

### 4.1 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. 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 and a reduction in the power generating efficiency of the plant. The dry and semi-dry systems have been considered further.

### 4.2 Environmental performance

#### 4.2.1 Emissions to air

The impact of emissions to air is considered in the Air Quality Assessment, refer to Appendix E of the Application. The acid gas emissions were assessed at the daily emission concentrations of 30 mg/m<sup>3</sup> for sulphur dioxide and 6 mg/m<sup>3</sup> for hydrogen chloride which is in accordance with the upper range of the BAT-AELs from the WI BREF applied within the Air Quality Assessment.

Table 4-1 shows the emission concentrations at the stack and the predicted ground level concentrations for each option. For sulphur dioxide, the 99.18<sup>th</sup> percentile of the daily means is shown. For hydrogen chloride, the hourly mean 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 4-1: Acid gas abatement – emissions to air comparison

Abatement System	Units	Dry		Semi-dry	
		SO <sub>2</sub>	HCl	SO <sub>2</sub>	HCl
Unabated emission concentration	mg/m <sup>3</sup>	450	900	450	900
Unabated emission rate	Tpa	870	1,740	870	1,740
Abated emission concentration	mg/m <sup>3</sup>	30	6	30	6
Abated emission rate	Tpa	60	10	60	10
Total emissions abated	Tpa	810	1,730	810	1,730
Process Contribution (PC)	ug/m <sup>3</sup>	4.29	1.31	4.29	1.31
Background	ug/m <sup>3</sup>	22.6	1.42	22.6	1.42
Predicted Environmental Contribution (PEC)	ug/m <sup>3</sup>	26.89	2.73	26.89	2.73
Air quality objective	ug/m <sup>3</sup>	125	750	125	750
PC as % of AQO	-	3.43%	0.17%	3.43%	0.17%
PEC as % of AQO	-	21.51%	0.36%	21.51%	0.36%

In accordance with the criteria presented in the Air Emissions Guidance, for both systems, the PC for emissions of SO<sub>2</sub> and HCl can be screened out as ‘insignificant’ when assessed at the daily emissions concentrations.

#### 4.2.2 Deposition to land

The impact of acid deposition on sensitive habitats has been assessed in the Air Quality Assessment (refer to Appendix E of the Application). As can be seen from the results presented in the Air Quality Assessment, the PC is less than 100% of the Critical Loads for all local nature sites and can be screened out as insignificant. However, the PC is greater than 1% of the relevant Critical Load at the European Designated Site (Hatfield Moor SPA/SAC) and therefore acid deposition cannot be screened out at this site. Further analysis was undertaken to determine the effect of interannual variability – at the Hatfield Moor site, the acid deposition PC exceeds 1% of the Critical Load in three of the five years of meteorological data. The average PC over the five years of meteorological data is 1.08% of the Critical Load. Whilst this is only just over the screening criterion, it is considered necessary to assess the potential significance of an approximately 1% PC at this site. This has been considered in the Ecological Interpretation of the AQA submitted in support of the planning application – refer to Appendix I of the EP application.

#### 4.2.3 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.



#### 4.2.4 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 290 tonnes ethylene equivalent.

#### 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 slightly different, which would change the power exported from the Facility. 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 349 kg CO<sub>2</sub> per MWh has been used, as justified in the greenhouse gas assessment presented in Appendix E of the Application. A comparison between the GWP of a dry and semi-dry system is shown in Table 4-2 below.

Table 4-2: Acid gas abatement – global warming potential comparison

	Units	Dry	Semi-dry
Power consumed	kWh/t	30	28.5
	MWh pa	9,030	8,580
Generation lost (water evaporation)	MWh pa	-	11,900
Power not exported	MWh pa	9,030	20,480
Global warming potential	t CO <sub>2</sub> pa	3,200	7,100

#### 4.2.6 Raw materials

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

Table 4-3: Acid gas abatement – raw materials comparison

	Units	Dry	Semi-dry
Additional water consumption compared to a dry system	tpa	-	28,920
Lime slurry	tpa	-	4,380
Lime	tpa	5,650	-
Powdered Activated Carbon (PAC)	tpa	90	90

#### 4.2.7 Waste streams

The only waste stream associated with the acid gas abatement treatment technologies is APCr. These would be a hazardous waste. The production rate for a semi-dry system would be approximately 10,800 tonnes per annum, with approximately 11,200 tonnes per annum for a dry system.

### 4.3 Costs

The estimated costs associated with each option are presented in Table 4-4. 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 4-4: Acid gas abatement – cost comparison

	Dry	Semi-dry
Capital cost	£14,200,000	£14,900,000
Annualised capital cost	£1,380,000	£1,450,000
Maintenance	£710,000	£745,000
Reagents and residues	£3,167,000	£2,968,000
Loss of exported power	£451,000	£1,023,000
<b>Total annualised cost</b>	<b>£5,708,000</b>	<b>£6,186,000</b>

### 4.4 Conclusions

The table below compares the options.

Table 4-5: Acid gas abatement – comparison table

	Units	Dry	Semi-Dry
SO <sub>2</sub> abated	tpa	810	810
Photochemical Ozone Creation Potential (POCP)	t ethylene-eq pa	290	290
Global warming potential	t CO <sub>2</sub> eq pa	3,200	7,100
Additional water consumption compared to a dry system	tpa	-	28,920
APC residues	tpa	11,200	10,800
Annualised cost	£ pa	£5,708,000	£6,186,000

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 reagents 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.

## 5 Reagent Selection

### 5.1 Options Considered

This section examines what reagent represents BAT for acid gas abatement. Reagents for wet scrubbing have not been considered, since wet scrubbing has been eliminated as a technique in Section 4.1 above. Therefore, the two alternative reagents for a dry or semi-dry system have been considered – lime and sodium bicarbonate.

### 5.2 Environmental Performance

#### 5.2.1 Emissions to Air

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

#### 5.2.2 Deposition to Land

There is no change in deposition to land between the two reagents.

#### 5.2.3 Emissions to water

There are no emissions to water associated with either reagent.

#### 5.2.4 Photochemical Ozone Creation Potential

There would be no change to the POCP for either reagent.

#### 5.2.5 Global warming potential

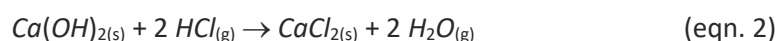
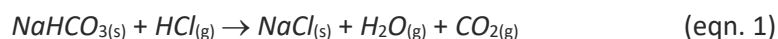
Sodium bicarbonate has a higher optimum reaction temperature than lime, which means that less heat can be recovered in the boiler. However, this can be resolved by recovering additional heat after the acid gas abatement system. Therefore, it has been assumed that there is no impact on global warming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

The reaction of hydrogen chloride and sulphur dioxide with sodium bicarbonate results in an emission of CO<sub>2</sub> whereas the reaction with lime does not.

#### 5.2.6 Raw materials

Sodium bicarbonate (NaHCO<sub>3</sub>) has better solid handling properties and a significantly lower stoichiometric ratio than hydrated lime (Ca(OH)<sub>2</sub>).

NaHCO<sub>3</sub> and Ca(OH)<sub>2</sub> react with the acid gases to produce alkaline salts as the following equations illustrate:



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  $\text{NaHCO}_3$ , the overall consumption of sodium bi-carbonate is actually approximately 60% higher than  $\text{Ca(OH)}_2$  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.

### 5.2.7 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.

## 5.3 Costs

There is little difference in capital cost associated with the two reagents.

The purchase cost of sodium bicarbonate is significantly higher than lime, with bicarbonate costing almost 65% 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 the sodium-based residues is 20% higher than lime residues, giving a disposal cost for sodium bicarbonate of £163 per tonne.

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 5-1: Reagent selection – costs per unit HCl abated

Item	Unit	$\text{NaHCO}_3$	$\text{Ca(OH)}_2$
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	91
Cost of residue disposal <sup>1</sup>	£/tonne	163	136
Overall Cost	£/op. hr/kmol	30.6	17.7
Ratio of costs		1.73	

<sup>1</sup>The figure shown does not include landfill tax.

## 5.4 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 of using sodium bicarbonate compared to lime, explained as follows:

- 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 costs per kmol HCl abated are nearly 75% higher.

Taking into consideration the above, the use of lime as a reagent within the acid gas abatement system is considered to represent BAT for the Facility.

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