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BAT Assessment Report

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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 Kingmoor ERF (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 design, with a design capacity of approximately 31.3 tonnes of waste fuel per hour, with an average net calorific value (NCV) of 10 MJ/kg. This equates to a design capacity of approximately 250,000 tonnes per annum, assuming 8,000 hours operation per annum.

The Facility will generate approximately 29.3 MWe of electricity in full condensing mode and with average UK ambient temperatures. The Facility will have a parasitic load of approximately 1.9 MWe. Therefore, the export capacity of the Facility, with average UK ambient temperatures will be approximately 27.4 MWe. The grid connection is limited to a maximum export capacity of 35 MWe.

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

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

- Water £1 per tonne
- Lime Slurry £90 per tonne
- Lime..... £94 per tonne
- Sodium Bicarbonate..... £155 per tonne
- Activated Carbon £650 per tonne
- Ammonia..... £203 per tonne
- Sand (with defined particle size distribution) £100 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 2019) £91 per tonne
- Imported power £70 per MWh
- Electricity revenue £45 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 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.

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_x 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 NO_x. This is considered in Section 2.2.6.

Table 2-1: Combustion techniques – NO_x emissions comparison

Option	NO _x emissions from furnace (mg/Nm ³) ⁽¹⁾
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 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/Nm³.

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 CO₂ 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

		Grate	Fluidised Bed
Power generated	MWh pa	234,000	234,000
Parasitic Load	MWh pa	15,000	17,000
Global warming potential	t CO ₂ eq pa	-76,000	-76,000

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

	Units	Grate	Fluidised Bed
Ammonia	tpa	1,600	1,100
Sand	tpa	-	2,860

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 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.
- Both options produce APC residues. For the purposes of this assessment it is assumed that both options will generate the same quantity of APC residue.

Estimated figures are shown in Table 2-4 below.

Table 2-4: Combustion techniques – waste streams comparison

	Units	Grate	Fluidised Bed
Bottom Ash	tpa	50,000	19,560
Boiler Ash	tpa	-	33,300
APC residues	tpa	11,000	11,000

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

	Grate	Fluidised Bed
Reagent costs (subtotal)	£320,000	£510,000
Residue disposal costs (subtotal)	£2,880,000	£2,910,000
Annual reagent and residue costs (materials)	£3,200,000	£3,420,000
Annual power revenue	£9,855,000	£9,765,000

As can be seen from the table above, 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.

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 2-5.

2.4 Conclusions

The table below compares the two options.

Table 2-6: Combustion techniques – option comparison summary

		Grate	Fluidised bed
Global warming potential	t CO2 eq pa	-76,000	-76,000
Ammonia consumption	tpa	1,600	1,100
Residues (total ash and sand)	tpa	61,000	63,850

		Grate	Fluidised bed
Annual total materials costs (reagents & residues)		£3,200,000	£3,420,000
Annual power revenue		£9,855,000	£9,765,000

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 6.9% 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 a robust 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_x 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_x 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_x abatement – emissions to air comparison

	Units	SNCR	SCR	FGR + SNCR
Nitrous oxide	mg/m ³	10	10	10
Ammonia	mg/m ³	10	10	10
NO _x , unabated concentration	mg/m ³	380	380	340
NO _x , unabated rate	tpa	490	490	440
NO _x , abated concentration	mg/m ³	120	80	120
NO _x released after abatement	tpa	160	100	160
NO _x removed	tpa	330	390	280

A long term abated emission concentration of 80 mg/Nm³ (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 Final Draft Waste Incineration BREF. The two SNCR systems, with and without Flue Gas Recirculation (FGR), would be required to achieve an emission limit of 120 mg/Nm³. The ELV of 120 mg/Nm³ has been applied to be consistent with the Air Quality Assessment and 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_x abatement – air emissions comparison

Abatement System:		SNCR	SCR	SNCR + FGR
Long Term				
Process Contribution (PC)	µg/m ³	2.22	1.48	2.22
Background	µg/m ³	13.50	13.50	13.50
Predicted Environmental Contribution (PEC)	µg/m ³	15.72	14.98	15.72
Air quality objective (AQO)	µg/m ³	40	40	40
PC as % of AQO		5.6%	3.7%	5.6%
PEC as % of AQO		39.3%	37.5%	39.3%
Short Term				
Process Contribution (PC)	µg/m ³	7.66	5.11	7.66
Background	µg/m ³	27.00	27.00	27.00
Predicted Environmental Contribution (PEC)	µg/m ³	34.66	32.11	34.66
Air quality objective (AQO)	µg/m ³	200	200	200
PC as % of AQO		3.8%	2.6%	3.8%
PEC as % of AQO		17.3%	16.1%	17.3%

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*'.

The long-term PC of process emissions for all systems cannot be screened out as insignificant in accordance the EA Guidance; however, the impact can be described as insignificant in accordance with the EA Guidance. Utilising SCR reduces the long-term PEC by 1.2% of the AQO 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 (refer to Appendix E of the Application). As can be seen from the results presented in this report, at all locally designated sites the process contribution is less than 100% of the relevant Critical Loads, and the impact of the Facility can be screened out as 'insignificant'.

At the identified European and UK statutory designated sites, the process contribution is less than 1% of the relevant Critical Loads and can be screened out as 'insignificant', with the exception of nitrogen (and acid) deposition for fen, marsh and swamp and Broad-leaved, mixed and yew woodland habitats at the River Eden and Tributaries designated site. The significance of effect has been discussed further in Appendix D of the Air Quality Assessment.

3.2.3 Emissions to water

There are no emissions to water from any of the NO_x abatement systems.

3.2.4 Photochemical Ozone Creation Potential

Nitrogen dioxide (NO₂) 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_x is released as NO₂ and the rest as NO, the POCP is -6,100 for the SNCR options and -3,800 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 349 kg CO₂ equivalent per MWh has been used, as justified in the greenhouse gas assessment - refer to Appendix E of the Application.

Table 3-3: NO_x abatement – global warming potential comparison

	Units	SNCR	SCR	SNCR + FGR
Power consumed	kWe	280	570	370
Power not generated	kWe	-	440	-
Change in exported power	MWh pa	2,200	8,100	3,000
Global warming potential	t CO ₂ eq pa	800	2,800	1,000

3.2.6 Raw materials

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

Table 3-4: NO_x abatement – raw materials comparison

	Units	SNCR	SCR	SNCR + FGR
Water	tpa	4,100	1,290	3,500
Ammonia	tpa	1,600	500	1,360

3.2.7 Waste streams

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

3.3 Costs

The estimated costs associated with each option are presented 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_x abatement – costs

	SNCR	SCR	SNCR + FGR
Capital Cost	£600,000	£8,700,000	£1,200,000
Annualised Capital Cost	£58,000	£847,000	£117,000
Maintenance	£12,000	£174,000	£24,000
Water and reagents	£329,000	£103,000	£280,000
Loss of exported power	£99,000	£365,000	£135,000
Total Annualised Cost	£498,000	£1,489,000	£556,000

3.4 Conclusions

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

Table 3-6: NO_x abatement – comparison table

	Units	SNCR	SCR	SNCR + FGR
NO _x released after abatement	tpa	160	100	160
NO _x removed	tpa	330	390	280
Photochemical Ozone Creation Potential (POCP)	t ethylene-eq pa	-6,100	-3,800	-6,100
Global warming potential	t CO ₂ eq pa	800	2,800	1,000
Ammonia Used	tpa	1,600	500	1,360
Total annualised cost	£ pa	£498,000	£1,489,000	£556,000
Average cost per tonne NO _x abated	£ p.t NO _x	£1,510	£3,820	£1,990

As can be seen from the table above, applying SCR for the abatement of NO_x:

1. increases the annualised costs by approximately £991,000;
2. abates approximately an additional 60 tonnes of NO_x per annum;
3. reduces the benefit of the facility in terms of the global warming potential by approximately 2,000 tonnes of CO₂;
4. reduces reagent consumption by approximately 1,100 tonnes per annum; and
5. costs approximately 2.5 times more per tonne of NO_x abated compared to an SNCR system.

The incremental cost for the additional abatement from SCR compared to SNCR is approximately £16,500 per additional tonne of NO_x 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 the SNCR system to abate NO_x increases the cost per tonne of NO_x abated by approximately 32% 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 200 tonnes of

CO₂ per annum while reducing ammonia consumption by approximately 240 tonnes per annum compared to an SNCR system without FGR.

However, this is based on the assumption that FGR reduces the NO_x generation within the furnace. This is not necessarily the case for all furnace manufacturers. Some designs can achieve lower levels of NO_x without FGR and adding of FGR might even cause some additional problems related e.g. for the availability of the plant.

The proposed designs do not include FGR. Therefore, taking the above into consideration, the use of SNCR without FGR is considered to represent BAT for the abatement of NO_x 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 are considered further below.

4.2 Environmental performance

4.2.1 Emissions to air

The impact of emissions to air is considered in the air quality assessment, which can be found in Appendix E of the Application. The acid gas emissions were assessed at the daily emission concentrations of 30 mg/m³ for sulphur dioxide and 6 mg/m³ for hydrogen chloride. The BAT-AEL emission limit in accordance with the Final Draft WI BREF has been applied to be consistent with the Air Quality Assessment.

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 maximum hourly 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 4-1: Acid gas abatement – emissions to air comparison

Abatement System	Units	Dry		Semi-dry	
		SO ₂	HCl	SO ₂	HCl
Unabated emission concentration	mg/m ³	450	900	450	900
Unabated emission rate	tpa	580	1,170	580	1,170
Abated emission concentration	mg/m ³	30	6	30	6
Abated emission rate	tpa	40	10	40	10
Total emissions abated	tpa	540	1,160	540	1,160
Short-term Process Contribution (PC)	ug/m ³	3.18	25.73	3.18	25.73
Background	ug/m ³	5.96	1.42	5.96	1.42
Predicted Environmental Contribution (PEC)	ug/m ³	9.14	27.15	9.14	27.15
Air quality objective	ug/m ³	125	750	125	750
PC as % of AQO	-	2.54%	3.43%	2.54%	3.43%
PEC as % of AQO	-	7.31%	3.62%	7.31%	3.62%

The short-term PC of process emissions for both 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'.

A more detailed assessment of impacts from the release of sulphur dioxide and hydrogen chloride is presented within the AQA - refer to Appendix E of the Application.

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 this report, at all locally designated sites the process contribution is less than 100% of the relevant Critical Loads, and the impact of the Facility can be screened out as 'insignificant'.

At the identified European and UK statutory designated sites, the process contribution is less than 1% of the relevant Critical Loads and can be screened out as 'insignificant', with the exception of nitrogen (and acid) deposition for fen, marsh and swamp and broad-leaved, mixed and yew woodland habitats at the River Eden and Tributaries designated site. The significance of effect has been discussed further in Appendix D of the Air Quality Assessment.

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 190 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 CO₂ 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	7,500	7,130
Generation lost (water evaporation)	MWh pa	-	9,500
Power not exported	MWh pa	7,500	16,630
Global warming potential	t CO ₂ pa	2,600	5,800

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	-	19,000
Lime slurry	tpa	-	2,700
Lime	tpa	3,500	-
Powdered Activated Carbon (PAC)	tpa	100	100

4.2.7 Waste streams

The only waste stream associated with the acid gas abatement treatment technologies is the Air Pollution Control (APC) residues. These would be a hazardous waste. The production rate for the semi-dry system would be approximately 13,900 tonnes per annum, with 14,000 tonnes per annum for dry systems.

4.3 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 4-4: Acid gas abatement – cost comparison

	Dry	Semi-dry
Capital cost	£13,000,000	£13,600,000
Annualised capital cost	£1,270,000	£1,320,000
Maintenance	£650,000	£680,000
Reagents and residues	£2,774,000	£2,665,000
Loss of exported power	£338,000	£748,000
Total annualised cost	£5,032,000	£5,413,000

4.4 Conclusions

The table below compares the options.

Table 4-5: Acid gas abatement – comparison table

	Units	Dry	Semi-Dry
SO ₂ abated	tpa	540	540
Photochemical Ozone Creation Potential (POCP)	t ethylene-eq pa	190	190
Global warming potential	t CO ₂ eq pa	2,600	5,800
Additional water consumption compared to a dry system	tpa	-	19,000
APC residues	tpa	14,000	13,900
Annualised cost	£ pa	£5,032,000	£5,413,000

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 POCP for either reagent.

5.2.5 Global warming potential

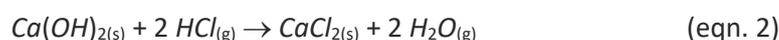
Sodium bicarbonate has a higher optimum reaction temperature than lime, which means that less heat can be recovered in the boiler. However, this can be resolved by recovering additional heat after the acid gas abatement system. Therefore, it has been assumed that there is no impact on global warming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

The reaction of hydrogen chloride and sulphur dioxide with sodium bicarbonate results in an emission of CO₂ whereas the reaction with lime does not.

5.2.6 Raw materials

Sodium bicarbonate (NaHCO₃) has better solid handling properties and a significantly lower stoichiometric ratio than hydrated lime (Ca(OH)₂).

NaHCO₃ and Ca(OH)₂ 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 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.

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 between 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 £150/tn.

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	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	

¹The 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 approximately 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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