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MedwayOne Energy Hub

Medway Energy Recovery Limited

BAT Assessment

ENGINEERING + CONSULTING

Document approval

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Contents

1 Introduction

Medway Energy Recovery Limited) is applying to the Environment Agency (EA) under the Environmental Permitting Regulations (EPRs) for an Environmental Permit (EP) to operate the MedwayOne Energy Hub (the Facility). The Facility will comprise a twin line waste incineration plant (the Facility) and associated infrastructure including battery storage and hydrogen production facilities, and will be located at Medway One, Kent.

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

1.1 Assumptions

The Facility will use a moving grate as the combustion technology, and will be a two stream design, with a nominal design capacity of approximately 28.5 tonnes of waste per hour, with an average net calorific value (NCV) of 10.5 MJ/kg. This equates to a nominal design capacity of approximately 456,000 tonnes per annum (tpa), assuming 8,000 hours operation per annum. The maximum capacity of the Facility being applied for in the permit is 606,400 tpa of waste.

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

The Facility will generate approximately 49.9 MWe with a parasitic load of 5 MWe.

In addition, it is assumed that ammonia will be used for the Selective Non-Catalytic Reduction (SNCR) nitrous oxides (NO_X) abatement system.

In order to calculate the global warming potential of electricity consumption – assumed to be imported from the grid – the assumption of 371 gCO_2/kWh has been used, as applied in the greenhouse gas assessment presented in Appendix E of the Supporting Information.

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.

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

Water	£0.85 per tonne
Lime Slurry	£90 per tonne
Lime	£192.41 per tonne
Sodium Bicarbonate	£280 per tonne
Activated Carbon	£605 per tonne
Ammonia	£114 per tonne
Sand (with defined particle size distribution)	£100 per tonne
Bottom Ash Processing	£15 per tonne
Lime APCR Disposal	£155 per tonne
Sodium bicarbonate APCR Disposal	£186 per tonne
Landfill Tax (1 April 2023)	£102.10 per tonne

Imported power	£122 per MWh
Electricity revenue	£60 per MWh

2 Combustion Techniques

2.1 Options Considered

The available technologies for the combustion of waste have been reviewed in section 2.6.1 of the supporting information. This 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 a range of non-hazardous wastes, such as that proposed for 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. A fluidised bed requires the waste to be pre-processed to 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 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 Section 2.2.6.

Table 2-1: NO_x Emissions

Option	NOx emissions from furnace (mg/Nm3)
Moving Grate	320-380(1)
Fluidised Bed	250-300(1)

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

2.2.2 Deposition to Land

Deposition from atmospheric emissions would also be unchanged.

2.2.3 Emissions to Water

There are no emissions to water for either system.

2.2.4 Photochemical Ozone Creation Potential

There would be no change to POCP for either 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 (GWP) nearly 300 times that of carbon dioxide. Fluidised beds can be designed to minimise the formation of nitrous oxide. For the purposes of this assessment 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.

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 2-2: Global Warming Potential

		Grate	**Fluidised Bed**
Power generated	MWh pa	399,200	399,200
Parasitic Load	MWh pa	40,000	44,000
GWP	t CO2 eq pa	-133,200	-131,700

2.2.6 Raw Materials

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

Table 2-3: Raw Materials

	Units	**Grate**	**Fluidised Bed**
Ammonia	tpa	1,800	1,400
Sand	tpa	-	5,470

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. In a fluidised bed, it is assumed that the boiler ash is removed via an abatement system 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 will require 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 the table below.

Table 2-4: Waste Streams

	Units	Grate	Fluidised Bed
Bottom Ash	tpa	102,000	39,470
Boiler Ash	tpa	-	68,000
APC Residues	tpa	19,400	19,400
Total ash	tpa	121,400	126,870

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

2.3 Costs

The estimated costs associated with each option are presented below.

Fluidised bed technology is typically 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 2-5: Annual Material Costs and Revenues

	Grate	Fluidised Bed
Reagents	£210,000	£710,000
Residue Disposal	£6,520,000	£6,600,000
Annual Reagent and Residue Costs (Materials)	£6,730,000	£7,310,000
Annual Power Revenues	£21,540,000	£21,300,000

As can be seen from the table above, the fluidised bed option has slightly higher costs associated with the purchase of reagents, and slightly higher costs with 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: Option Comparison Summary

		Grate	Fluidised Bed
Global Warming Potential	t CO2 eq pa	-133,200	-131,700
Ammonia Consumption	tpa	1,800	1,400

		Grate	Fluidised Bed
Residues (Total ash)	tpa	121,400	126,870
Annual Total Materials Costs		£6,730,000	£7,310,000
Annual Power Revenues		£21,540,000	£21,300,000

The grate has a lower global warming potential than the fluidised bed, but will consume approximately 25% more ammonia.

Both combustion technologies will produce similar quantities of ash, although the fluidised bed produces a separate boiler ash residue.

The material costs are approximately 9% higher for the fluidised bed than the grate, whereas the grate system will have a slightly higher power revenue. 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 pre-processing to 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.

3 Nitrogen Oxides (NOx) Abatement

3.1 Options Considered

Three options have been considered for NOx 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).

As stated previously, for the purposes of this assessment, it is assumed that ammonia solution will be the reagent used in the NOx 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: Air Emissions

	Units	SNCR	SCR	FGR + SNCR
Nitrous oxide	mg/m^3	10	10	10
Ammonia	mg/m^3	10	10	10
NO_x, unabated concentration	mg/m^3	350	350	315
NO_x, unabated rate	tpa	950	950	850
NO_x, abated concentration	mg/m^3	120	80	120
NO_x released after abatement	tpa	320	220	320
NO_x removed	tpa	630	730	530

For the purposes of this assessment, a long term abated emission concentration of 80 mg/Nm^3 (11% reference oxygen content) is used for SCR for the purposes of this BAT assessment, since this is the level that the technology can achieve on a long-term basis. The two SNCR systems, with and without Flue Gas Recirculation (FGR), would be required to achieve an emission limit of 120 mg/Nm^3, in accordance with the proposed emission limits for the Facility.

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

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

The impact of emissions to air is considered in detail within the air quality assessment, refer to Appendix E of the Supporting Information. The table below shows the predicted ground level concentrations for the three options.

Table 3-2: Air Emissions

Abatement System:		SNCR	SCR	SNCR + FGR
Long Term				
Process Contribution (PC)	µg/m^3	0.32	0.21	0.32
Background	µg/m^3	20.76	20.76	20.76
Predicted Environmental Contribution (PEC)	µg/m^3	21.08	20.97	21.08
Air Quality Objective	µg/m^3	40	40	40
PC as % of AQO		0.80%	0.53%	0.80%
PEC as % of AQO		52.70%	52.43%	52.70%
Short Term				
Process Contribution (PC)	µg/m^3	5.21	3.47	5.21
Background	µg/m^3	41.52	41.52	41.52
Predicted Environmental Contribution (PEC)	µg/m^3	46.73	44.99	46.73
Air Quality Objective	µg/m^3	200	200	200
PC as % of AQO		2.61%	1.74%	2.61%
PEC as % of AQO		23.37%	22.50%	23.37%

It can be seen that there are no predicted exceedances of air quality objectives for any of the options. Using SCR reduces the long-term PEC by 0.27% of the air quality objective and the short-term PEC by 0.87% 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, refer to Appendix E of the Application Pack, and a Shadow Appropriate Assessment of ecological impacts is provided in Appendix J of the Application Pack.

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 (NO_2) has a photochemical ozone creation potential (POCP) of 2.8 and nitrogen oxide (NO) has a POCP of -42.7. Assuming that 10% of NOx is released as NO2 and the rest as NO, the POCP is -12,200 for the SNCR options and -8,400 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 371 kg CO_2 equivalent per MWh has been used, as applied in the greenhouse gas assessment, refer to Appendix E of the Application Pack.

Table 3-3: Global Warming Potential

	Units	SNCR	SCR	SNCR + FGR
Power consumed	kWe	510	1,050	680
Power not generated	kWe	-	720	-
Change in exported power	MWh pa	4,100	14,200	5,400
GWP	t CO_2 eq pa	1,500	5,300	2,000

3.2.6 Raw Materials

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

Table 3-4: Raw Materials

	Units	SNCR	SCR	SNCR + FGR
Water	tpa	4,900	2,580	4,100
Ammonia	tpa	1,900	1,000	1,600

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 Technical Guidance Note EPR-H1.

Table 3-5: Costs

	SNCR	SCR	SNCR + FGR
Capital Cost	£1,100,000	£17,800,000	£2,500,000
Annualised Capital Cost	£107,000	£1,733,000	£243,000
Maintenance	£22,000	£356,000	£50,000
Water and reagents	£221,000	£116,000	£186,000
Loss of exported power	£246,000	£852,000	£324,000

	SNCR	SCR	SNCR + FGR
Total Annualised Cost	£596,000	£3,057,000	£803,000

3.4 Conclusions

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

Table 3-6: Comparison Table

	Units	SNCR	SCR	SNCR + FGR
NO_x released after abatement	tpa	320	220	320
NO_x removed	tpa	630	730	530
Photochemical Ozone Creation Potential (POCP)	t ethylene-eq pa	-12,200	-8,400	-12,200
Global Warming Potential	t CO_2 eq pa	1,500	5,300	2,000
Ammonia Used	tpa	1,900	1,000	1,600
Total Annualised Cost	£ pa	£596,000	£3,057,000	£803,000
Average cost per tonne NO_x abated	£ p.t NO_x.	£950	£4,190	£1,520

As can be seen from the table above, applying SCR to the Facility:

1. increases the annualised costs by approximately £2.5 million;
2. abates an additional 100 tonnes of NOx per annum;
3. reduces the benefit of the Facility in terms of the global warming potential by approximately 3,300 tonnes of CO_2;
4. reduces reagent consumption by approximately 900 tonnes per annum; and
5. costs more than 440% more per additional tonne of NOx abated, compared to an SNCR system.

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

Including FGR to the SNCR system to abate NOx increases the cost per tonne of NOx abated by 160%. It has no effect on the direct environmental impact of the plant, but it increases the impact on climate change by approximately 500 tonnes of CO_2 per annum while reducing ammonia consumption by approximately 300 tonnes per annum. Allowing for the increase in the costs of NOx abatement for a SCR system compared to the climate change and reagent consumption associated with FGR, an SNCR system with or without FGR is considered to represent BAT.

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 NOx within the Facility.

4 Acid Gas Abatement Technology

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 and generates a visible plume. It is mainly used in the UK for facilities treating hazardous waste where high and varying levels of acid gases in the flue gases require the buffering capacity and additional abatement performance of a wet scrubbing system.
2. Semi-dry, involving the injection of lime as a slurry into the flue gases in the form of a spray of fine droplets. The acid gases are absorbed into the aqueous phase on the surface of the droplets and react with the lime. The fine droplets evaporate as the flue gases pass through the system, cooling the gas. This means that less energy can be extracted from the flue gases in the boiler, making the steam cycle less efficient. The lime and reaction products are collected on a bag filter, where further reaction can take place.
3. Dry, involving the injection of solid lime into the flue gases as a powder. The lime is collected on a bag filter to form a cake and most of the reaction between the acid gases and the lime takes place as the flue gases pass through the filter cake. In its basic form, the dry system consumes more lime than the semi-dry system. However, this can be improved by recirculating the flue gas treatment residues, which contain some unreacted lime and reinjecting this into the flue gases.

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

4.2 Environmental Performance

4.2.1 Emissions to Air

The impact of emissions to air is considered in the air quality assessment, which is presented in Appendix E of the Supporting Information. The impact of the emissions of acid gases was assessed at the daily emission concentrations of 30 mg/m^3 for sulphur dioxide (SO_2) and 6 mg/m^3 for hydrogen chloride (HCl).

The table below presents 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 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: Emissions to Air

Abatement System		**Dry**		**Semi-dry**	
Pollutant	**Units**	**SO_2**	**HCl**	**SO_2**	**HCl**
Unabated emission concentration	mg/m^3	480	900	480	900
Unabated emission rate	tpa	1,300	2,430	1,300	2,430
Abated emission concentration	mg/m^3	30	6	30	6
Abated emission rate	tpa	80	20	80	20
Total emissions abated	tpa	1,220	2,410	1,220	2,410
Process Contribution (PC)	ug/m^3	1.14	2.34	1.14	2.34
Background	ug/m^3	2.00	1.42	2.00	1.42
Predicted Environmental Contribution (PEC)	ug/m^3	3.14	3.76	3.14	3.76
Air Quality Objective	ug/m^3	125	750	125	750
PC as % of AQO		0.91%	0.31%	0.91%	0.31%
PEC as % of AQO		2.51%	0.50%	2.51%	0.50%

The short-term impact of the plant is 0.91% of the daily average air quality objective for sulphur dioxide and 0.31% of the hourly air quality objective for hydrogen chloride. The impact of hydrogen chloride and sulphur dioxide is considered to be insignificant when applying the criteria stated in Environment Agency guidance note H1.

A more detailed assessment of impacts from the release of sulphur dioxide and hydrogen chloride is presented within the air quality assessment within Appendix E of the Application Pack.

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 Pack, and a Shadow Appropriate Assessment of ecological impacts is provided in Appendix J of the Application Pack.

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 380 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 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 371kg CO_2 per MWh has been used, as applied in the greenhouse gas assessment, refer to Appendix E of the Supporting Information. The global warming potential is therefore calculated as follows:

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

The global warming potential associated with each option is shown in the table below.

Table 4-2: Global Warming

	Units	Dry	Semi-Dry
Power consumed	kWh/t	30	28.5
	MWh pa	13,680	13,000
Generation lost (water evaporation)	MWh pa		15,600
Power not exported	MWh pa	13,680	28,600
GWP	t CO_2 pa	5,100	10,600

4.2.6 Raw Materials

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

Table 4-3: Raw Materials

	Units	Dry	Semi-Dry
Additional water consumption compared to a dry system	tpa		40,380
Lime Slurry	tpa		7,900
Lime	tpa	10,200	
Powdered Activated Carbon (PAC)	tpa	150	150

4.2.7 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 19,000 tonnes per annum (19,400 for a dry system and 18,600 for a semi-dry system).

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: Costs

	Dry	Semi-Dry
Capital Cost	£26,400,000	£27,700,000
Annualised Capital Cost	£2,570,000	£2,700,000
Maintenance	£1,320,000	£1,385,000
Reagents and residues	£7,042,000	£5,618,000
Loss of exported power	£821,000	£1,716,000
Total Annualised Cost	**£11,753,000**	**£11,419,000**

4.4 Conclusions

The table below compares the options.

Table 4-5: Comparison Table

	Units	Dry	Semi-Dry
SO_2 abated	tpa	1,220	1,220
Photochemical Ozone Creation Potential (POCP)	t ethylene-eq pa	380	380
Global Warming Potential	t CO_2 eq pa	5,100	10,600
Additional water consumption compared to a dry system	tpa	-	40,380
APC Residues	tpa	19,400	18,600
Total Annualised Cost	£ pa	£11,753,000	£11,419,000

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. In addition, within a semi-dry system recycling of reagent within the process is not proven, but it is proven in a dry system. However, the semi-dry option benefits from medium reaction rates which mean that a shorter residence time is required in comparison with 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 Acid Gas Abatement Reagent

5.1 Options Considered

Within this assessment, reagents for wet scrubbing have not been considered as wet scrubbing systems have been discounted as representing BAT for the abatement of acid gases in Section 4.1 of this assessment. Therefore, only the two alternative reagents for a 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

Again, there is no change between the two reagents.

5.2.3 Emissions to Water

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

5.2.4 Photochemical Ozone Creation Potential

There would be no change to POCP for either system.

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_2 whereas the reaction with lime does not.

5.2.6 Raw Materials

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

Sodium bicarbonate and lime 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 sodium bicarbonate, the overall consumption of sodium bi-carbonate is actually 64% higher than lime on a mass basis.

The reagent required to abate one kmol of hydrogen chloride 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 hydrogen chloride 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 around 77% 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 £186/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 hydrogen chloride:

Table 5-1: 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	280	192.41
Cost of residue disposal[1]	£/tonne	186	155
Overall Cost	£/op. hr/kmol	46.1	26.1

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

Item	Unit	$NaHCO_3$	$Ca(OH)_2$
Ratio of costs		1.77	

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:

- The residue has a higher leaching ability, 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 use of sodium bicarbonate has a slightly higher global warming potential due to the reaction chemistry; and
- The costs per kmol of hydrogen chloride abated are around 77% higher.

Considering 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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