

Riverside Energy Park

Environmental Permit Appendices

APPENDIX:

E

BAT ASSESSMENT

December 2018 | Revision 0 |

Contents

1	Introduction	1
1.2	Project Description	1
1.3	The Objective	1
2	Assumptions.....	3
3	Combustion Techniques	4
3.1	Options Considered.....	4
3.2	Environmental Performance.....	4
3.3	Global Warming Potential.....	4
3.4	Raw Materials.....	5
3.5	Waste Streams.....	5
3.6	Costs	6
3.7	Conclusions	7
4	Nitrogen Oxides Abatement.....	8
4.1	Options Considered.....	8
5	Acid Gas Abatement	10
5.1	Options Considered.....	10
5.2	Environmental Performance.....	10
5.3	Global Warming Potential.....	11
5.4	Raw Materials.....	12
5.5	Waste Streams.....	12
5.6	Costs	13
5.7	Conclusions	13
6	Reagent Selection	14
6.1	Options Considered.....	14
6.2	Environmental Performance.....	14
6.3	Global Warming Potential.....	14
6.4	Raw Materials.....	14
6.5	Waste Streams.....	15
6.6	Costs	15
6.7	Conclusions	16

This page is intentionally blank

1 Introduction

1.1.1 Cory Environmental Holdings Limited (trading as Cory Riverside Energy) (Cory or the Applicant) is applying to the Environment Agency (EA) under The Environmental Permitting (England and Wales) Regulations 2016 (Environmental Permitting Regulations) for an Environmental Permit (EP) to operate an integrated Energy Park, to be known as Riverside Energy Park (REP or the Proposed Development). REP would comprise waste treatment facilities together with an associated Electrical Connection.

1.2 Project Description

1.2.1 A detailed description of REP is presented in sections 1.4 to 1.6 of the Supporting Information. REP would be constructed on land immediately adjacent to Cory's existing Riverside Resource Recovery Facility (RRRF), within the London Borough of Bexley and would complement the operation of the existing facility.

1.2.2 The main elements of REP would be as follows:

- Energy Recovery Facility (ERF): to provide thermal treatment of Commercial and Industrial (C&I) residual (non-recyclable) waste with the potential for treatment of (non-recyclable) Municipal Solid Waste (MSW);
- Anaerobic Digestion facility: to process food and green waste. Outputs from the Anaerobic Digestion facility would be transferred off-site for use in the agricultural sector as fertiliser or as an alternative, where appropriate, used as a fuel in the ERF to generate electricity;
- Solar Photovoltaic Installation: to generate electricity. Installed across a wide extent of the roof of the Main REP Building;
- Battery Storage: to store and supply additional power to the local distribution network at times of peak electrical demand. This facility would be integrated into the Main REP building; and
- On Site Combined Heat and Power (CHP) Infrastructure: to provide an opportunity for local district heating for nearby residential developments and businesses. REP would be CHP Enabled with necessary on site infrastructure included within the REP site.

1.3 The Objective

1.3.1 The objective of this report is to demonstrate, in accordance with the requirements of Environment Agency's sector guidance on waste incineration, titled '*Incineration of waste (EPR5.01)*', that the proposed techniques within the ERF for acid gas abatement, nitrogen oxides abatement and combustion technology represent Best Available Technology (BAT) through a cost benefit analysis – referred to as a quantitative BAT assessment.

1.3.2 Each assessment within this report follows the structure of Environment Agency Horizontal Guidance Note H1 (Annex K), titled '*H1 Annex K: cost & benefit analysis*'. It is noted that this guidance note was withdrawn by the UK Government on 1 February 2016. However, in withdrawing the guidance note, the UK Government has stated on the '.gov' website that '*this guidance is no longer being updated*'. In the absence of alternative guidance being published by the UK Government, this is considered to be the only available guidance for developing a cost benefit analysis. Therefore, this report and assessment has been developed in accordance with Environment Agency Horizontal Guidance Note H1 (Annex K).

- 1.3.3 For clarity, this report does not include a cost benefit analysis of the technologies employed in the anaerobic digestion facility as this is not required by the relevant sector guidance.

2 Assumptions

- 2.1.1 The proposed ERF would include a two stream energy recovery process. The ERF has been designed to thermally treat incoming waste with a range of net calorific values (NCV's). The nominal design capacity of the thermal treatment lines is approximately 41 tonnes per hour of waste, with an average NCV of 9 MJ/kg. The ERF would have an assumed availability of approximately 8,000 hours per annum. On this basis, the ERF would have a nominal design capacity of approximately 655,000 tonnes per annum. Allowing for an availability of 8,760 hours per annum and a variability of incoming waste (NCV 7 – 13 MJ/kg), the maximum throughput of the ERF would be approximately 805,920 tonnes per annum. However, for the purposes of this assessment and to be consistent with the Greenhouse Gas Assessment, the nominal design capacity has been applied.
- 2.1.2 The turbine would be designed to generate up to 67.6 MWe and up to 30 MWth of heat. The ERF would have an estimated parasitic load of 6.1 MWe. Therefore, the maximum export capacity of the ERF would be 61.5 MWe.
- 2.1.3 In the operating costs sections within the individual assessments, the following unit costs have been assumed:
- | | |
|---|------------------|
| ▪ Water | £1 per tonne |
| ▪ Quick Lime | £90 per tonne |
| ▪ Hydrated Lime | £94 per tonne |
| ▪ Sodium Bicarbonate | £155 per tonne |
| ▪ Activated Carbon | £650 per tonne |
| ▪ Ammonia | £203 per tonne |
| ▪ Sand (with defined particle size distribution) | £100 per tonne |
| ▪ Bottom Ash Processing | £10 per tonne |
| ▪ Lime Air Pollution Control Residues (APCR) Disposal | £125 per tonne |
| ▪ Sodium bicarbonate APCR Disposal | £150 per tonne |
| ▪ Landfill Tax (1 April 2018) | £88.95 per tonne |
| ▪ Imported power | £70 per MWh |
| ▪ Electricity revenue | £53 per MWh |
- 2.1.4 This report utilises data obtained by Fichtner (Cory's consultant in respect of this EP application) from a range of different projects which use the technologies identified, to compare with the technologies proposed.

3 Combustion Techniques

3.1 Options Considered

- 3.1.1 The available techniques for waste combustion have been reviewed in Section 2.6 of the Supporting Information. The assessment has been expanded to provide a cost-benefit analysis of moving grates and fluidised beds.
- Grates are the leading technology in the UK and Europe for the combustion of biomass and waste fuels. The air-cooled moving grate comprises of inclined fixed and moving bars that would move the fuel from the feed inlet to the residue discharge. The grate movement would turn and mix the fuel along the surface of the grate to ensure that all fuel would be exposed to the combustion process.
 - Fluidised beds are designed for the combustion of relatively homogeneous waste. For the purposes of this assessment it is assumed that the incoming waste would need to be pre-processed to satisfy the fuel requirements for a fluidised bed combustion system.

3.2 Environmental Performance

Emissions to Air

- 3.2.1 The emissions to atmosphere would not be affected by the choice of combustion technology. Although NO_x concentrations from the furnaces would be different (as illustrated in Table 3-1), both options would require further abatement to achieve the relevant emission limits. This means that the actual effect would be to change the amount of reagent required to abate the NO_x.

Table 3-1 - NO_x Emissions

Option	NO _x emissions from furnace (mg/Nm ³)
Moving Grate	320-380
Fluidised Bed	250-300

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

Deposition to Land

- 3.2.2 Deposition from atmospheric emissions would also be unchanged.

Emissions to Water

- 3.2.3 There are no emissions to water for either system.

Photochemical Ozone Creation Potential (POCP)

- 3.2.4 There would be no change to POCP for either system.

3.3 Global Warming Potential

- 3.3.1 The direct emissions of carbon dioxide are the same for each option. However, whilst fluidised beds have lower emissions of nitrogen dioxide, they can have elevated emissions of nitrous oxide. Nitrous oxide is a greenhouse gas with a global warming potential (GWP) nearly 300

times that of carbon dioxide. Fluidised beds can be designed to minimise the formation of nitrous oxide. For the purposes of this assessment we have assumed that the fluidised bed has been well-designed, and the emissions of nitrous oxide are the same as a grate and would be released at a concentration of 10mg/Nm³.

- 3.3.2 A fluidised bed has a higher parasitic load than a moving grate system due to the additional 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 for both options.
- 3.3.3 In order to calculate the global warming potential of electricity consumption, the figure of 357 kg CO₂ equivalent per MWh has been used, as applied in the greenhouse gas assessment presented in Appendix D.
- 3.3.4 The results are presented in Table 3-2 below. In each case the overall GWP is less than zero, as there would be a net reduction due to displacement of, primarily, fossil fuel power generation. Thus, the more negative figure produced by the grate is better.

Table 3-2 - Global Warming Potential

	Unit	Grate	Fluidised Bed
Power generated	MWh p.a.	492,000	487,000
Parasitic Load	MWh p.a.	6.10	6.71
GWP	t CO ₂ p.a.	-176,000	-174,000

3.4 Raw Materials

- 3.4.1 The estimated consumption of raw materials for each option is shown below in Table 3-3.

Table 3-3 - Raw Materials

	Unit	Grate	Fluidised Bed
Ammonia	t.p.a.	1,200	1,000
Sand	t.p.a.		6,740

3.5 Waste Streams

- 3.5.1 The two options produce several solid waste streams:
- i. It is assumed that most metals within the waste would have been removed during any pre-treatment of the incoming waste. Therefore, it is assumed that it would be identical for both options and has not been considered further.
 - ii. The bottom ash generation is lower for fluidised beds. Assuming a suitably licensed facility can be identified, bottom ash would be recovered as a secondary aggregate.
 - iii. Fluidised beds have much greater carry-over of fine particles and so produce an additional fly ash stream, which is removed in a cyclone before the acid gas abatement reagent is added. This separate fly ash stream (Fly Ash) could be usable for building aggregate, but this is not certain, and it is possible that it would need to be sent to a

hazardous landfill. For the purposes of this assessment it has been assumed that it cannot be used as a building aggregate and requires disposal in a non-hazardous landfill.

- iv. Both options produce APCR. The fluidised bed option would generate less APCR because more of the fly ash would have been removed from the gas stream.

3.5.2 Estimated figures are shown in Table 3-4 below.

Table 3-4 - Waste Streams

	Unit	Grate	Fluidised Bed
Bottom Ash	t p.a.	157,000	59,340
Fly Ash	t.p.a.		105,100
APC Residues	t p.a.	24,000	20,700

3.6 Costs

3.6.1 The estimated costs associated with each option are presented below in Table 3-5.

3.6.2 Fluidised bed technology is typically up to 5% more expensive than a grate, due to the additional waste screening equipment, sand dosing and recycling equipment, and fly ash separation. At the time of writing this assessment, capital costs are not readily available for the different options. Therefore, it has not been possible to consider the capital costs for the two technologies within this assessment.

3.6.3 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 ERF in this assessment.

Table 3-5 - Annual Material Costs and Revenues

	Grate	Fluidised Bed
Reagents	£240,000	£870,000
Residue Disposal	£6,710,000	£6,780,000
Total Materials Costs	£6,950,000	£7,650,000
Power Revenues	£26,076,000	£25,811,000

3.6.4 As can be seen from Table 3-5 above, the fluidised bed option has 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.

3.6.5 The power generated by the two systems is comparable, if we assume that the ERF would only receive pre-processed fuels and therefore the parasitic load associated with fuel preparation is excluded from the assessment.

3.6.6 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 3-5.

3.7 Conclusions

3.7.1 Table 3-6 below compares the two options.

Table 3-6 - Option Comparison Summary

		Grate	Fluidised Bed
Global Warming Potential	t CO ₂ p.a.	-176,000	-174,000
Ammonia Consumption	t.p.a.	1,200	1,000
Residues	t.p.a.	£6,710,000	£6,780,000
Total Materials Costs	p.a.	£6,950,000	£7,650,000
Power Revenue	p.a.	£26,076,000	£25,811,000

- 3.7.2 The grate has a lower global warming potential than the fluidised bed, and it would use slightly more ammonia to abate emissions of NO_x. Both combustion technologies would produce similar quantities of ash, although the fluidised bed produces more fly ash.
- 3.7.3 The material costs are approximately 10% higher for the fluidised bed than the grate, whereas the grate system would result in a slightly higher power revenue due to a lower parasitic load.
- 3.7.4 As stated within the qualitative BAT assessment (refer to Section 2.6 of the Supporting Information) grate combustion systems are designed for large quantities of heterogeneous waste, whereas fluidised bed systems are more sensitive to inconsistencies within the fuel. Due to the robustness of grate combustion systems and Cory's experience of these systems, a grate is considered to represent BAT for the combustion of waste within the ERF.

4 Nitrogen Oxides Abatement

4.1 Options Considered

4.1.1 Two options have been considered for NO_x abatement and are listed as follows:

- i. Selective Catalytic Reduction with Flue Gas Recirculation (FGR+SCR) involves the injection of ammonia solution or urea into the flue gases immediately upstream of a reactor vessel containing layers of catalyst.
- ii. Selective Non-Catalytic Reduction with Flue Gas Recirculation (FGR+SNCR) involves the injection of ammonia solution or urea into the combustion chamber.

4.1.2 As stated in the Supporting Information, FGR would be intrinsic to the design of the ERF. Therefore, for the purposes of this EP application, the use of SNCR and SCR (both with the addition of FGR) have been considered.

Selective Non-Catalytic Reduction

4.1.3 SNCR involves distributing a spray containing an aqueous ammonia or aqueous urea solution (the de-NO_x reagent) into the flue gas flow path at an appropriate location (typically the secondary combustion chamber), at a gas temperature of 850 to 1,050°C. The reagent reacts with the NO_x formed in the combustion process to produce a combination of nitrogen, water and carbon dioxide (when urea is used as the reagent).

4.1.4 Extensive dosing of reagent or low reaction temperatures can lead to ammonia slip, resulting in the formation of ammonia salts downstream in the flue gas path and discharge to atmosphere of unreacted ammonia. Ammonia may be controlled under the plant's permit and can lead to secondary problems, so should be kept to a minimum.

4.1.5 SNCR is widely deployed across waste, biomass and coal power plants, including at the adjacent RRRF.

Selective Catalytic Reduction

4.1.6 SCR is a means of converting NO_x, with the aid of a catalyst, into nitrogen, water and carbon dioxide. SCR is a leading technology in the abatement of NO_x from combustion systems.

4.1.7 Aqueous ammonia or urea is injected into the flue gas stream and flows across a catalytic surface, typically Titanium Dioxide. The catalyst is installed downstream of the bag filter component of the flue gas cleaning system in order to extend the lifetime of the catalyst.

4.1.8 The reaction takes place at a lower temperature than SNCR, typically 250 to 300°C. However, since the flue gases should be cleaned in a bag filter before the catalyst, which is done at a lower temperature, the flue gas must be heated before the SCR system, by means of steam extraction from the turbine (thereby reducing electrical generating capacity) or auxiliary fuelled burners. This would be subject to detailed design of the SCR system.

4.1.9 The additional components in the flue gas path require a larger induced draft fan to be installed, which also increases the electricity consumption and so reduces the electricity exported from the plant. Ammonia slip would also be a limiting factor of NO_x abatement efficacy in SCR systems in terms of environmental compliance.

4.1.10 An SCR system is considerably more complicated and is more capital intensive than a SNCR system. Whereas the SNCR system consists of a number of injection nozzles, along with

pipework, tanks and pumps, the SCR system includes a large catalyst bed and gas-gas heat exchanger. Despite lower reagent consumption (due to better stoichiometry), the operational costs of an SCR system are higher due in large part to regeneration and replacement of the catalyst.

Selective Catalytic Reduction v Selective Non-Catalytic Reduction

- 4.1.11 The use of SNCR has not been considered, as an SNCR system would not be able to achieve the NO_x emission concentrations being proposed by Cory for the ERF. Due to the very low emission concentrations proposed (75 mg/m³), the use of SCR is considered to represent BAT for the ERF.

5 Acid Gas Abatement

5.1 Options Considered

- 5.1.1 There are currently three technologies widely available for acid gas abatement on waste incineration plants in the UK:
- i. 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.
 - ii. 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.
 - iii. 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.
- 5.1.2 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 in Sections 5.2 to 5.7 below.

5.2 Environmental Performance

Emissions to Air

- 5.2.1 The impact of emissions to air is considered in the Dispersion Modelling Report within Appendix D. As stated in the Dispersion Modelling Report, the acid gas emissions have been assessed at the emission limits (BAT AEL's) stated within the Draft Waste Incineration BREF. The daily emission concentrations which were considered are 30 mg/m³ for sulphur dioxide and 6 mg/m³ for hydrogen chloride.
- 5.2.2 Table 5-1 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 max 1-hour 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 5-1 - Emissions to Air

Abatement System	Pollutant	Dry		Semi-dry	
		SO ₂	HCl	SO ₂	HCl
Unabated emission concentration	mg/m ³	400	800	400	800
Unabated emission rate	t/a	620	1,230	620	1,230
Abated emission concentration	mg/m ³	30	6	30	6
Abated emission rate	t/a	50	10	50	10
Emissions abated	t/a	570	1,220	570	1,220
Short-term process Contribution (PC)	ug/m ³	1.85	9.94	1.85	9.94
Background	ug/m ³	17.3	1.4	17.3	1.4
Predicted Environmental Contribution (PEC)	ug/m ³	36.45	12.74	36.45	12.74
Air Quality Objective	ug/m ³	125	750	125	750
PC as % of AQO		1.48	1.33	1.48	1.33
PEC as % of AQO		29.16	1.70	29.16	1.70

5.2.3 The short-term impact of the plant is 1.48% of the daily average air quality objective for SO₂ and 1.33% of the hourly air quality objective for HCl. The impact of HCl and SO₂ is considered to be insignificant when applying the criteria stated in Environment Agency guidance note H1. As the impact is screened as insignificant, the PEC (predicted environmental contribution) calculated for each pollutant is not considered further.

5.2.4 A more detailed assessment of impacts from the release of sulphur dioxide and hydrogen chloride is presented within the Dispersion Modelling Report within Appendix D.

Deposition to Land

5.2.5 The impact of acid deposition on sensitive habitats has been assessed in the Dispersion Modelling Report within Appendix D.

Emissions to Water

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

Photochemical Ozone Creation Potential

5.2.7 Sulphur dioxide has a POCP of 4.8. Hence, the POCP for both the dry and semi-dry systems would be 380 tonnes ethylene equivalent.

5.3 Global Warming Potential

5.3.1 The direct emission of greenhouse gases would be a lot higher for a semi-dry system as compared to a dry system. Although the carbon dioxide and nitrous oxide emission concentrations are unchanged between the two systems, the power not exported would be

greater for a semi-dry system. The energy consumption would be slightly different, which would change the power exported from the ERF. 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.

- 5.3.2 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.
- 5.3.3 In order to calculate the global warming potential of electricity consumption, the figure of 357kg CO₂ per MWh has been used, as applied in the greenhouse gas assessment presented in Appendix D.
- 5.3.4 The calculated data confirming the above points is shown in Table 5-2 below.

Table 5-2 - Global Warming Potential

	Unit	Dry	Semi-Dry
Power consumed	kWh/t	30	28.5
	MWh p.a.	19,650	18,670
Generation lost	MWh p.a.		26,500
Power not exported	MWh p.a.	19,650	45,170
GWP	t CO ₂ p.a.	7,000	16,100

5.4 Raw Materials

- 5.4.1 The estimated consumption of raw materials for both options is shown in Table 5-3 below.

Table 5-3 - Raw Materials

	Unit	Dry	Semi-Dry
Additional water consumption compared to a dry system	t.p.a.		29,241
Quick Lime	t.p.a.		9,100
Hydrated Lime	t.p.a.	9,460	
Powdered Activated Carbon (PAC)	t.p.a.	240	240

5.5 Waste Streams

- 5.5.1 The only waste stream associated with the acid gas abatement treatment technologies is the APCR. APCR is typically classified as hazardous (due to its elevated pH) in accordance with Technical Guidance WM3: 'Waste Classification - Guidance on the classification and assessment of waste'. The maximum APCR production rate for dry and semi-dry systems would be approximately 24,000 and 23,500 tonnes per annum respectively.

5.6 Costs

- 5.6.1 The estimated costs associated with each option are presented in Table 5-4 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 5-4 - Costs

	Dry	Semi-Dry
Capital Cost	£29,700,000	£31,100,000
Annualised Capital Cost	£2,890,000	£3,030,000
Maintenance	£1,485,000	£1,555,000
Reagents and residues	£6,228,000	£6,080,000
Loss of exported power	£1,041,000	£2,394,000
Total Annualised Cost	£11,644,000	£13,059,000

5.7 Conclusions

- 5.7.1 Table 5-5 below compares the options.

Table 5-5 - Comparison Table

	Unit	Dry	Semi-Dry
SO ₂ abated	t p.a.	570	570
Photochemical Ozone Creation Potential (POCP)	t ethylene-eq p.a.	380	380
Global Warming Potential	t CO ₂ p.a.	7,000	16,100
Additional water consumption compared to a dry system	t.p.a.		29,241
APCR	t p.a.	24,000	23,500
Total Annualised Cost	£ p.a.	£11,644,000	£13,059,000

- 5.7.2 The performance of the options would be very similar.
- 5.7.3 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.
- 5.7.4 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 would be required in comparison with a dry system. In addition, within a semi-dry system recycling of reagent within the process is not proven, but it is proven in a dry system.
- 5.7.5 Due to the low water consumption and proven capability for recycling of reagents, the dry system is considered to represent BAT for the abatement of acid gases within the ERF.

6 Reagent Selection

6.1 Options Considered

- 6.1.1 We have not considered reagents for wet scrubbing, since this has been eliminated as a technique in paragraph 5.1.2. We have therefore only considered the two alternative reagents for a dry system – hydrated lime and sodium bicarbonate.

6.2 Environmental Performance

Emissions to Air

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

Deposition to Land

- 6.2.2 Again, there would be no change between the two reagents.

Emissions to Water

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

Photochemical Ozone Creation Potential

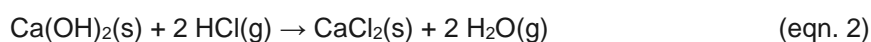
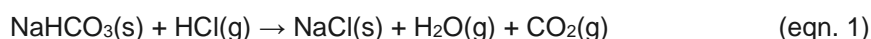
- 6.2.4 There would be no change to POCP for either system.

6.3 Global Warming Potential

- 6.3.1 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.
- 6.3.2 The reaction of hydrogen chloride and sulphur dioxide with sodium bicarbonate results in an emission of carbon dioxide whereas the reaction with lime does not.

6.4 Raw Materials

- 6.4.1 The ratio between the quantity of reagent supplied and the minimum required for the reaction is called the “stoichiometric ratio”. Sodium bicarbonate (NaHCO_3) has better solid handling properties and a significantly lower stoichiometric ratio than hydrated lime ($\text{Ca}(\text{OH})_2$).
- 6.4.2 NaHCO_3 and $\text{Ca}(\text{OH})_2$ react with the acid gases to produce alkaline salts, as equations 1 and 2 below illustrate:



- 6.4.3 In order to promote the reactions above, excess quantities of sodium bicarbonate or lime would be required. The excess reagent is lost in the residue.

- 6.4.4 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 bicarbonate is actually 64% higher than Ca(OH)_2 on a mass basis.
- 6.4.5 The reagent required to abate one kmol of HCl was calculated as approximately 109 kg of sodium bicarbonate and approximately 67 kg of lime.

6.5 Waste Streams

- 6.5.1 The stoichiometric ratio indicates that the amount of residue would 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.
- 6.5.2 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.
- 6.5.3 Furthermore, there are limited waste disposal options for sodium bicarbonate based APCR.

6.6 Costs

- 6.6.1 There is little difference in capital cost between the two reagents.
- 6.6.2 The purchase cost of NaHCO_3 is significantly higher than Ca(OH)_2 , with bicarbonate costing almost 65% more than hydrated lime per tonne. This makes sodium bicarbonate an uneconomic option in comparison to lime.
- 6.6.3 The cost of disposing of the residue must also be considered due to the differences in quantity. Sodium based residues are more difficult to stabilise than lime residues. It has been assumed that the cost per tonne to landfill of the sodium based residues is 20% higher than lime residues giving a disposal cost for sodium bicarbonate of £150 /tn.
- 6.6.4 The operating costs for the two options are compared in Table 6-1 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 6-1 - Costs per Unit HCl Abated

Item	Unit	NaHCO_3	Ca(OH)_2
Mass of reagent required	kg/kmol	109.0	67.0
Mass of residue generated	kg/kmol	84.0	85.0
Cost of reagent	£/tonne	155	94
Cost of residue disposal ¹	£/tonne	150	125
Overall Cost	£/op. hr/kmol	29.5	16.9
Ratio of costs		1.74	1.00

¹ The figure shown does not include landfill tax.

6.7 Conclusions

- 6.7.1 There is a small environmental benefit for using sodium bicarbonate, in that the mass of residues produced is smaller. However, there are a number of significant disadvantages:
- The residue has a higher leaching ability than lime-based residue, which would limit the disposal options;
 - The reaction temperature does not 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 74% higher.
- 6.7.2 Therefore, the use of lime is considered to represent BAT for the abatement of acid gases within the ERF.