Environmental Performance Assessment of Froth Flotation for Coal Recovery and Sulfur Removal from Ultrafine Coal Waste

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minerals to metals
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SYNOPSIS

The South African coal mining industry generates large volumes of coal ultrafine waste (< 150 microns) each year, with a significant amount being dumped in tailing slurry dams. These slurry dams have been associated with prolonged pollution and loss of valuable resources. In the two stage flotation process developed at the University of Cape Town, froth flotation is used to both recover coal (stage 1) and remove pyritic sulfur (stage 2) from ultrafine coal waste, resulting in three outputs streams: a saleable coal product, a small volume sulfide-rich stream, and a reduced volume sulfide lean tailings stream. Pre-disposal removal of sulfide sulfur and coal recovery by means of froth flotation is aimed at effectively removing the acid rock drainage (ARD) risk associated with sulfide bearing wastes and at recovering valuable resources respectively. Previous studies have demonstrated the technical feasibility of this process for a number of coal waste types on a laboratory-scale, with results indicating that it is possible to recover large quantities of useable coal whilst generating a tailings waste stream with a reduced sulfur content and negligible ARD risk. An order of magnitude financial model for a fictitious plant has also been developed, and applied to demonstrate the economic viability for selected case studies.

To date, however, studies on the environmental viability of the process have only focused on the ARD mitigating potential of the two stage flotation process and little attention has been given to the systemic environmental implications of the process such as the energy and reagent usage. The research study therefore aims to evaluate the environmental burdens and benefits of the two stage flotation process, particularly from a South African context, and to compare the environmental performance to the conventional disposal of untreated coal ultrafines. Furthermore, this project aims to establish which stages along the process contribute the most to the environmental burdens of the process and how the variations of the input parameters affect the overall environmental performance of the proposed process.

To this end, a life cycle inventory of inputs and outputs was compiled on the basis of the empirical results derived from a previous laboratory-scale case study conducted on a sample of an acid generating ultrafine coal waste from the Waterberg region. Experimental results from the case study, which entailed two-stage flotation (using Naflote 9858 as a coal collector and xanthate (SIBX) as a sulfide collector in stages 1 and 2),and detailed characterisation of the feed and desulfurised tailings, was supplemented with literature information and data from mass and energy balance calculations for a fictitious plant. An environmental impact analysis was subsequently conducted using a combination of Life Cycle Impact Assessment and risk-based impact assessment techniques and criteria. The impact categories selected included climate change, terrestrial acidification, fossil fuel depletion, natural land transformation, aquatic water pollution risk, drinking water quality risk, aqueous acidification, salinity and consumptive water footprint. Aquatic water pollution risk, drinking water quality risk and aqueous acidification impact indicators were calculated by summing up risk potential factors for the constituents of the final disposed waste streams. The rest of the impact categories were calculated by multiplying the inventory result with a characterisation factor developed from impact assessment models.

The case study results indicated that the implementation of the two-stage flotation process results in a notable decrease in eco-toxicity, salinity, consumptive water footprint, metal toxicity, aqueous acidification, fossil fuel depletion and natural land transformation impacts. However, the results also indicated an increase in atmospheric related impacts (climate change and terrestrial acidification impacts), which has been attributed to the additional energy consumption associated with the two-stage flotation process and the production processes associated with the flotation reagents. Analyses of the process contributions to the individual impact categories for the two-stage flotation process revealed the climate change and terrestrial acidification impact categories to be dominated by the electricity production process and the flotation reagents production process. The sensitivity analyses revealed a higher dependence of the fossil fuel depletion impact category on the percentage coal yield than the electricity consumption of the foreground process. Furthermore the sensitivity analyses indicated a strong dependence of the climate change and terrestrial acidification impacts on the electricity consumption and the SIBX dosage in the foreground process.

In the South African context, implementation the two-stage flotation process would result in a significant recovery of coal (approximately 1.2 million tonnes for every 4 million tonnes dry coal ultrafines lost per annum) and a sulfide-rich product which can be utilised for electricity production and sulfuric acid production respectively, hence promoting resource efficiency. Although higher than in the case of conventional land disposal, the energy used in the two-stage flotation process is infinitesimal compared to the energy recovered in the process through the generation of additional coal, and results in only a 0.025% increase in the annual greenhouse gas emissions. The implementation of the two-stage flotation would also result in reduced water losses in comparison to conventional land disposal, which is beneficial in the South African context as South Africa is a water scarce region. Lastly whilst the implementation of the two-stage flotation process would result in the reduction of water related impacts associated with acidification, salinization and metal pollution, it might pose a further threat to aquatic life if the xanthate salt reagents are emitted to local water sources.

The limitations of the study were mainly associated with the quality of the input and output data, the impact categories and the system boundary and scenario development. The multiple sources of information and the variations in literature of the energy input estimates were noted as a source of uncertainty. The lack of characterisation factors for some of the substances in the system as well as the exclusion of the possibility of utilization of the sulfide-lean stream were also part of the limitations associated with the study. Recommendations for future work include improving the environmental assessment by incorporating various case studies and by incorporating downstream processing as well as optimizing the two-stage flotation process by using less energy and by using less toxic flotation reagents.

This dissertation is dedicated to my great grandfather, Mr. Fundikwa Chirongoma, whom I have never met but whose relentless work ethic and culture of generosity echoes to this very day.

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GLOSSARY

List of abbreviations and nomenclature used in the dissertation

ABA	Acid base accounting
ANC	Acid neutralising capacity
ANZEEC	Australia and New Zealand Environment and Conservation Council
ARD	Acid rock drainage
CS2	Carbon disulphide
DME	Department of Minerals and Energy
DWAF	Department of Water Affairs and Forestry
EIA	Environmental Impact Assessment
EPA	Environmental Protection Agency
ICP-MS	Inductively coupled plasma-mass spectrometry
ICP-OES	Inductively coupled plasma optical emission spectrometry
ISO	International Standards Organisation
KWh	Kilowatt hour
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LC50	Median lethal concentration
LD50	Median lethal dosage
MJ	Mega joules
MPA	Maximum potential acidity
NAPP	Net acid production potential
NICNAS	National Industrial Chemicals Notification & Assessment Scheme
PAX	Potassium amyl xanthate
Ppm	Parts per million
RA	Risk Assessments
SETAC	Society of environmental toxicity and chemistry
SEX	Sodium ethyl xanthate
SIBX	Sodium isobutyl xanthate

UNEP	United Nations Environmental Program
WHO	World Health Organisation

List of elements

Al	aluminium
As	arsenic
Cd	cadmium
Co	cobalt
Cr	chromium
Cu	copper
Fe	iron
Hg	mercury
K	potassium
Mg	magnesium
Mn	manganese
Mo	molybdenum
Na	sodium
S	sulphur
Sb	antinomy
Si	silicon
Se	selenium
Th	thorium
Ti	titanium
U	uranium
V	vanadium
Zn	zinc

Units of measurement

g	gram-unit of mass (1g = 1000mg = 10^{-3}Kg)
kg	kilogram-unit of mass (1kg = 1000g = 10^{-3}t)
KJ/mol	kilojoule per mole-unit of energy
mg	milligrams-unit of mass (1mg = 10_{-3}g)
MJ	Megajoules - one million (10^{6}) joules
Mt	Megatones - one million tons (10^{6}) joules
pH	measure of acid concentration (-log [H+])
ppb	parts per billion-unit of concentration (1ppm = 10^{-3}ppm)
ppm	parts per million-unit of concentration (1 ppm = 1 g/t = 1mg/kg in solids and 1 mg/l in solutions)
µm	micrometer-unit of length (1µm = 10^{-3}mm = 10^{-6}m)
t	metric ton-unit of mass (1t = 1000kg)
TJ	Terajoules - one trillion (10^{12}) joules

CHAPTER 1

INTRODUCTION

According to (Reddick, 2006), the coal mining industry generates approximately 12 million tonnes of ultrafine slurry each year, most of which is dumped in tailing slurry dams. These slurry dams are associated with various negative environmental impacts. Amongst the impacts is the prolonged pollution of local water resources and loss of valuable resources. Pre-disposal removal of sulfide sulfur by means of froth flotation is aimed at effectively removing the acid rock drainage risk associated with the sulfide bearing wastes. A two-stage flotation process aimed at removing sulfide sulfur (to mitigate the ARD risk associated with wastes) whilst at the same time recovering valuable coal has been developed at the University of Cape Town (UCT). A series of laboratory testwork has been done in previous studies to establish the technical feasibility of the process. Furthermore, based on this testwork, an order of magnitude financial model for a conceptual plant was developed to demonstrate the economic viability of the process. This particular research study focuses on a holistic assessment of the environmental performance of the two-stage flotation process by evaluating the environmental benefits and burdens of the process relative to those associated with conventional coal waste management approaches.

1.1 Background

1.1.1 The South African coal industry

Coal forms the major part of South Africa’s primary energy source, providing approximately 70% of the total country’s energy and contributing approximately 90% towards electricity production (DME, 2008). It is estimated that South Africa has 15 billion tonnes of coal reserves (Hartnady, 2010), of these reserves most are found in the Central Basin which includes the Witbank, Highveld and Ermelo coalfields. The remainder of the reserves are located in the Waterberg, Sasolburg, Free State and Springbok Flats (refer to Figure 1). According to Eberhard (2011), recent exploration efforts have been around the Waterberg coalfield area and it is speculated that it could become a major coal mining centre in the future. South African coalfields are also known to produce both export quality metallurgical coal and thermal grade coal and it is estimated that 80 Mt of coal is produced in South Africa annually (Snyman and Botha, 1993; DME, 2001; Hartnady, 2010). South Africa ranks fifth as a hard coal exporter, behind Australia, Indonesia, Russia and Colombia (Hartnady, 2010).

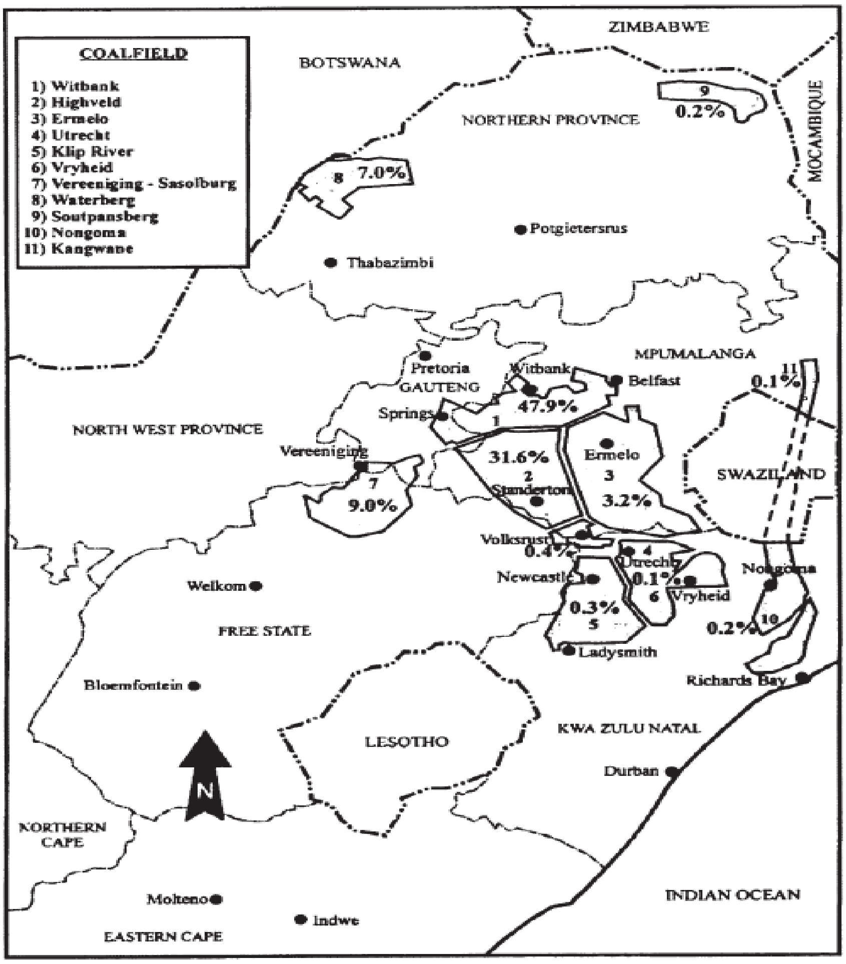


Figure 1 Distribution of coalfields in South Africa (Eberhard, 2011)

1.1.2 Coal preparation

Coal that is received directly from the ground, which is termed ROM (Run-of-Mine) coal, contains a large proportion of impurities such as wood, rock, ash forming minerals and mining fragments. These impurities reduce the heating capacity of the coal and might cause serious damage to the processing machinery. ROM coal therefore needs to be processed further before it can be utilized locally or exported. To separate the impurities from coal, the ROM material goes through two stages of processing. The first stage is screening, where coal is separated into various size fractions and where large foreign objects are eliminated. Depending on the product specifications, coal can be immediately sold after screening but in order to further increase its calorific value it goes through the second stage of separation which is coal washing (Reddick, 2006).

Coal washing (also referred to as coal beneficiation) involves the reduction of the ash and sulfur content of the coal by utilizing density separation techniques. It is the minimum ash and sulfur quantities that are required by the coal buyers that dictates the degree of washing, and therefore the amount of waste that is generated in the process. South African coals have characteristically high ash and inertinite contents and therefore often require beneficiation after screening (Reddick, 2006). The different size fractions coming from the screening stage are treated by different beneficiation techniques. The course coal, which is specified as coal larger than 25 mm, is washed in dense medium drums producing a low ash product coal and generating a high ash discard coal. The intermediate coal, which has a size fraction of about 1-25 mm is washed in dense medium cyclones whilst the fine coal, which has a size fraction of about 0.15-1 mm is washed in spirals, also producing a low ash content coal product and generating a high ash content coal discard. As for the ultrafine coal (size fraction <0.15), in the past, it was only beneficiated in the former Natal province and in the Waterberg coalfield. According to de Korte (2008), this was because the coking coals from these areas were compliant to flotation (froth flotation was usually employed on the minus 0.5 mm size fraction). In the Witbank area, however the ultra-fine coal was not beneficiated and was disposed of by pumping the coal into slurry ponds (de Korte, 2008).

Today there is a move away from this practice due to the environmental regulations and economic considerations with some plants in the Witbank coalfield beneficiating fines and ultrafines via flotation and with other plants, installing filter presses to dry coal ultrafines (de Korte, 2008; Swanepoel, 2012). The coal washing process and the different separation techniques are summarized in Figure 2.

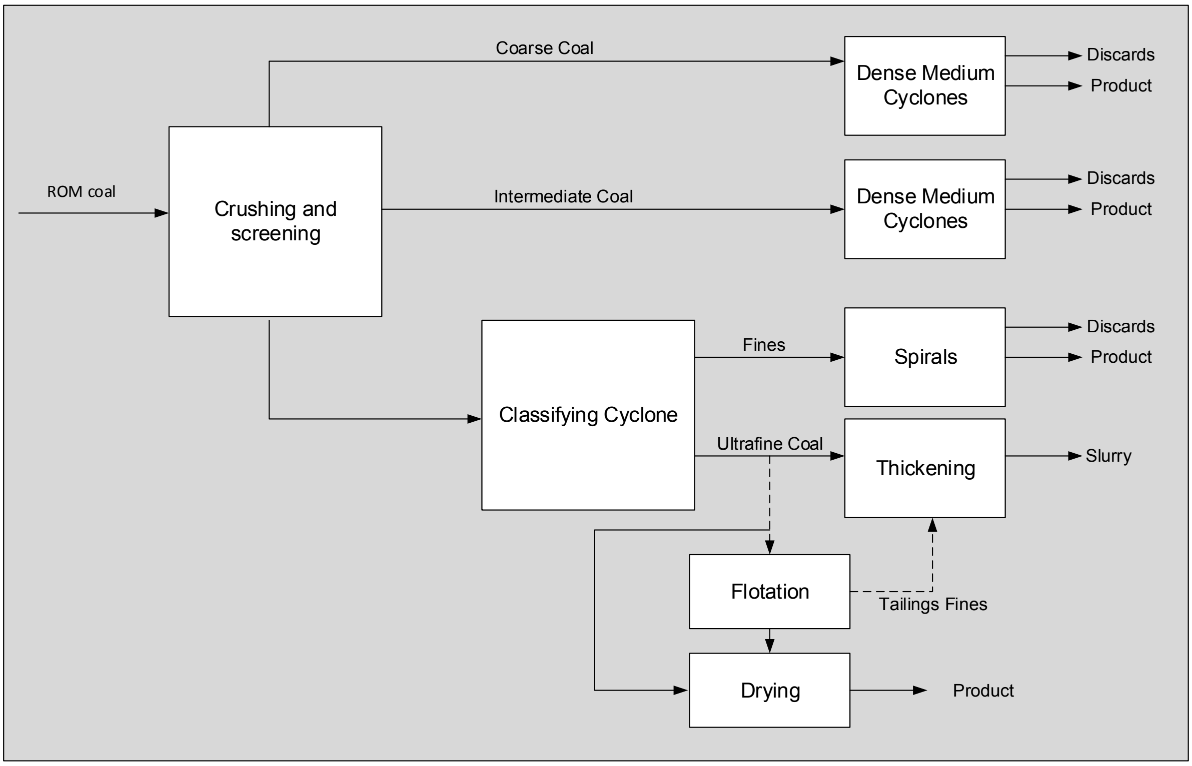


Figure 2 Block flow diagram of the coal washing process (adapted from Reddick, 2006)

1.1.3 Coal wastes disposal practices and associated environmental impacts

Discards, which generally have a high ash content, are disposed of in discard dumps whilst coal ultrafines (which are characterised by a high moisture content) are disposed of either in underground old mine workings, open cast voids or in slurry dams with the discards. The calorific value of the unbeneficiated coal ultrafines is relatively high (approximately 20 MJ/kg) compared to that of coal discards (approximately 11 MJ/kg) and is comparable to that of the run-of-mine; which is typically between 20-27 MJ/kg (Franzidis, 1992; DME, 2001;South African Coal Roadmap, 2013).

The disposal of coal ultrafines and discards into slurry and discard dams has numerous environmental implications some of which include the potential generation of ARD, the leaching of trace elements into freshwater sources, excessive land-use, the loss of resources and the possibility of spontaneous combustion (Bell et al., 2001; Pone et al., 2007; Harrison et al., 2010; South African Coal Roadmap, 2013). According to the Department of Minerals and Energy (2001), an accumulated mass of over 1 billion tons of coal ultrafines and discards are estimated to have been dumped over the years and more than 4,000 hectares of land is occupied by these waste disposal facilities in South Africa. Initiatives are now being taken by researchers to improve this approach to coal waste management. Amongst these initiatives was the development of a two-stage froth flotation process at the University of Cape Town.

1.1 A two-stage froth flotation process for the management of coal wastes

With the aim of mitigating ARD and recovering resources, the two-stage froth flotation process was developed at UCT. In this process, clean coal (coal with a relatively low ash content) is recovered in the first stage whilst sulfidic sulfur is removed in the second stage, achieving a lower volume sulfide lean tailings stream. The different ways in which the sulfide rich stream could be used have been identified and are being explored (Stander, 2013). The same applies for the sulfide-lean tailings stream which can be used as cover material or be safely disposed of in a slurry dam. Figure 3 summarizes the overall process.

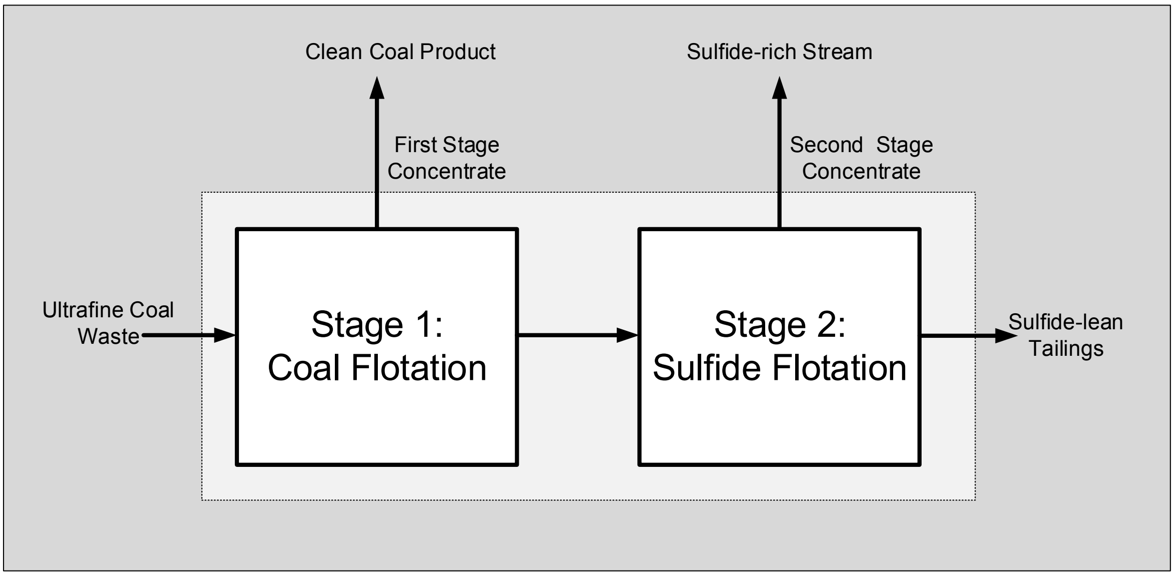


Figure 3 The two-stage flotation process for the recovery of coal and removal of sulfur bearing minerals from ultrafine coal wastes

To date numerous laboratory tests have been done to investigate the technical feasibility of the process for a variety of ultrafine coal wastes and flotation reagents for both flotation stages (Kazadi Mbamba, 2011; Amaral Filho, 2012; Fisher and Toms, 2013; Howlett and Marsden, 2013). Results have indicated that it is possible to recover large quantities of useable coal (up to 89% of coal ultrafines), whilst generating a tailings waste stream with a reduced sulfur content (down to 0.2% of sulfide-lean stream) and negligible ARD risk (Harrison et al., 2013). Based on this laboratory testwork, Jera (2013) conducted an order of magnitude financial model for a conceptual plant in order to demonstrate the economic viability of the process. In the investigation, it was observed that the two-stage flotation fictitious plant was associated with a positive net present value, managing to achieve significant revenues due to the recovered coal product.

1.2 Assessing the environmental impacts of coal waste management methods

To date several approaches and tools have been developed that cater to the assessment of the environmental performance of a process. One of the most common and widely used of these tools is Life Cycle Assessment (LCA). LCA evaluates the environmental burdens associated with a product or process by addressing the impacts across the entire life cycle of the product/process, from the primary resource extraction through to the emissions and stable residues. Due to its approach, LCA allows for a more transparent comparison of the environmental burdens associated with any two processes or products (Hermann et al., 2007). An alternative way of assessing the environmental performance is the use of risk based approaches such as Environmental Risk Assessment. Whilst LCA approaches provide a measure of relative performance, risk based approaches are able to assess the environmental impact of a process in absolute measures. An example of a risk based approach is the scoring and ranking protocol developed by Broadhurst and Petrie (2010) for ranking and scoring solid waste constituents on the basis of their hazard-forming potential and chemical behaviour under disposal conditions.

In the context of solid waste management methods, conventional LCA methods have been criticized for not including aspects such as leachate generation and salinization of local water surfaces due to the exclusion of spatial and temporal emissions in the inventory analysis (Hansen, 2004). Furthermore the lack of equivalency factors for some constituents of solid wastes as well as the considerable uncertainty associated with the metal equivalency factors limits the effectiveness of the existing LCA models in terms of assessing the environmental impacts associated with the solid waste management practices (Notten, 2001; Kunene 2014). Researchers have developed additional impact categories to compensate for these discrepancies whilst other researchers have used risk-based methods and indicators to compliment the LCA method. An example is the development of a salinity impact category developed by Leske and Buckley (2004), which caters for the impact associated with the emissions of salt forming cations and anions.

1.3 Problem statement

In light of the technical and economic assessments carried out on the two-stage flotation process, the process has yet to be demonstrated beyond the laboratory scale and so far the environmental studies done have been limited to the ARD generating potential of the disposed waste. Other environmental implications of the disposal of untreated coal waste such as salinization and contamination of local water sources are yet to be compared against the conventional disposal of sulfide-lean tailings. Little consideration has also been given to the systemic environmental implications of the two-stage flotation process such as energy and reagent usage. Without this knowledge, motivation and justification for the large-scale implementation of the process will be difficult, as will the opportunities for optimisation of environmental performance. It is therefore paramount to evaluate the environmental burdens of this proposed approach to coal waste management and to understand the environmental trade-offs of implementing this method versus the conventional coal waste management method. Furthermore, it is also important to establish which stages along the process contribute the most to the environmental burdens of the process and how the variations of the input parameters affect the environmental impacts so as to gain insight as to how the process can be improved.

1.4 Project objectives and scope

This project is focused on two main case studies; these include the base case, which involves the disposal of untreated coal ultrafines into slurry dams and the two-stage flotation process, which involves the desulfurisation of coal ultrafines and the recovery of clean coal. An environmental performance assessment using a combination of an LCA approach and risk-based tools is to be conducted on both scenarios in order to compare them and to evaluate the environmental burdens and benefits between them. The main research objective for this project is to conduct a holistic assessment of the environmental implications of the implementation of the two stage flotation process. The following are the key questions that are to be addressed in the fulfilment of this research objective.

i. How do the environmental waste burdens of the disposal of untreated coal ultrafines compare to those of the two-stage flotation process?

ii. Which processes associated with both of the coal waste management scenarios contribute the most to the environmental burdens of the scenarios?

iii. How is the environmental performance of the two-stage flotation process affected by the variation of input parameters?

iv. What are the key environmental implications associated with implementation of the two-stage flotation process in the South African context?

This research study is a desktop study with model inputs based on previous in-house laboratory testwork as well as other literature sources

1.5 Dissertation structure

Chapter 1 of the dissertation aims to introduce the background of the study and motivates for the purpose of the dissertation. Chapter 2 entails critical review of the literature, where key related studies are highlighted and linked to the study. Chapter 3 outlines the approach taken, and describes the case study, the tools and techniques used and the key assumptions made. Chapter 4 presents and discusses the case study results whilst Chapter 5 synthesises and further interprets the results in terms of the key research questions outlined in Section 1.4. Finally, the dissertation presents the conclusions of the research study and provides recommendations for future work (Chapter 6). Figure 4 illustrates how the dissertation is structured.

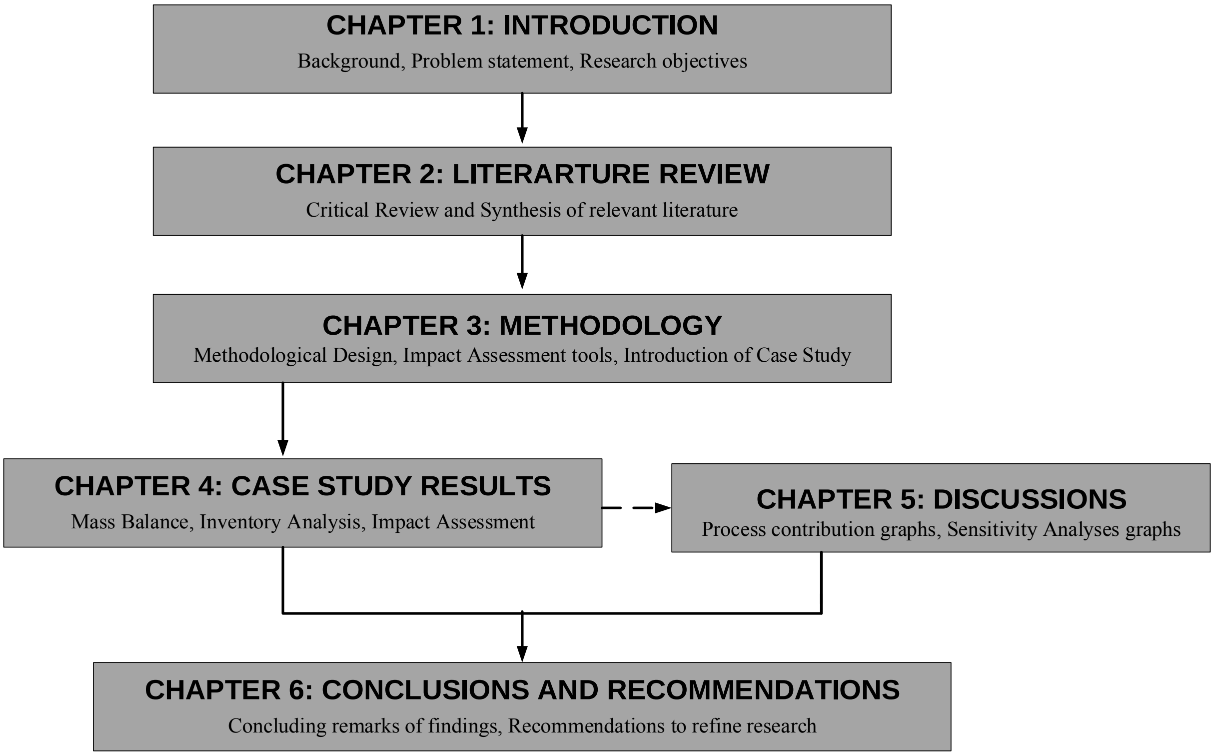


Figure 4 Dissertation structure

CHAPTER 2

LITERATURE REVIEW

This chapter serves to further illustrate the need for the research study as well as establish the relationship between the past work conducted and the current study. The chapter will begin by discussing the environmental implications associated with the direct disposal of coal waste. The focus will then be shifted onto the two-stage froth flotation process, where past in-house laboratory test-work will be summarised and the conceptual plant model that was developed discussed. Key parameters that affect this process will also be discussed. Following from this, the potential environmental implications of the two-stage flotation process will be discussed and the chapter will end by describing the impact assessment tools to be used in this project together with their limitations and shortcomings.

2.1 Environmental implications of the direct disposal of coal waste

Current coal waste management methods in South Africa include the disposal of coal discards in discard dumps and the disposal of coal ultrafines (which are characterised by a high moisture content) in either underground old mine workings, open cast voids or in slurry dams (South African Coal Roadmap, 2013). The main environmental implications associated with these coal waste management practises include the potential generation of ARD, the leaching of trace elements into freshwater sources, excessive land-use, the loss of resources and the possibility of spontaneous combustion (Bell et al., 2001; Pone et al., 2007; Harrison et al., 2010). As with most environmental impacts, some of these negative consequences might not be experienced immediately but might manifest themselves several generations later. The following sub-sections expand on the potential environmental implications of coal processing wastes, with specific emphasis on ultrafine coal slurry waste.

2.1.1 Acid Rock Drainage (ARD)

Nature and formation of ARD

ARD is an acidic, sulfate and metal laden water, comprising of pH values below 2.3, acidity levels around 5000 mg/l and salt concentrations higher than 10000 mg/l (Brown, 1996; Akcil and Koldas, 2006). ARD is formed when sulfide-bearing minerals, particularly pyrite, are exposed to oxygen and water. The water can come into contact with the coal waste through various means such as rainfall, surface run-on, groundwater seepage and liquid effluent used to slurry the waste (Hansen, 2004). Formation of ARD can result in the leaching of toxic metals and salts into local water sources (Harrison et al., 2010) (this will be discussed in detail in the following subsections). Additionally an increase in pH in local water sources presents a threat to aquatic species and vegetation that are sensitive to changes in levels of pH (ANZECC, 2000).

The generation of ARD occurs naturally with factors such as the presence of bacteria, the abundance of oxygen and the particle size of the waste being able to promote its generation (Akcil and Koldas, 2006). The following are the main chemical reactions involved in the formation of ARD through the oxidation of pyrite.

$$2FeS_2 + 7O_2 + 2H_2O \rightarrow 2Fe^{2+} + 4SO_4^{2-} + 4H^+ \qquad (1)$$

$$4Fe^{2+} + O_2 + 4H^+ \rightarrow 4Fe^{3+} + 2H_2O \qquad (2)$$

$$FeS_2 + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16H^+ \qquad (3)$$

Firstly, pyrite is oxidised, releasing ferrous iron (Fe^{2+}), sulfate (SO_4^{2-}) and acid (H^+) into the slurry pore water (Equation 1). Iron and sulfur oxidising bacteria then catalyse the oxidation of ferrous iron to ferric iron (Equation 2), which in turn oxidises pyrite (Equation 3) hence generating acid. The chemical reactions form a continuous cycle, generating more acid (Lottermoster, 2010) which in turn will cause an increase in pH levels of local water sources and the leaching of toxic metals into the water sources.

ARD prediction tests

In order to quantify the amount of acid that can be potentially generated over a long period, with the intention to gauge the possible environmental impact thereof, a set of ARD prediction tests have been developed. The main tests include acid base accounting tests (ABA), the net acid generation (NAG) tests and biokinetic tests. ABA tests measure the difference between the maximum acid forming capacity (MPA) and the acid neutralization capacity (ANC) of a sample based on its sulfur content. The difference is expressed as the Net Acid Producing Potential (NAPP), which is the theoretically determined amount of acid that a sample can produce (Stewart et al., 2009).

NAG tests, which are similar to ABA tests, measure the acid forming potential of a sample by allowing both the acid forming and acid neutralizing reactions to occur simultaneously, using H_2O_2 as the oxidant. The pH of the solution is determined, and then titrated to pH 7, and the resulting acid concentration measured. The measured sulfuric acid concentration is the net acid generated by the sample (Stewart et al., 2009). Biokinetic tests, which are normally used to confirm the results of ABA and NAG tests, investigate the leachability of samples in the presence of microorganisms and provide information on the relative timing of the ARD generation (Kazadi Mbamba et al., 2012). Table 1 presents how the values obtained from the acid base accounting and net acid generation tests are classified as acid forming, potentially acid forming and non-acid forming.

Table 1 ARD classification using ABA and NAG static tests (Stewart et al., 2009)

ARD prediction method	Unit	Result	Classification
Acid Base Accounting	kg H_2SO_4/t	NAPP > 20	Acid forming
		-20 < NAPP < 20	Potentially acid forming
		NAPP > 20	Non-acid forming
Net Acid Generation	kg H_2SO_4/t	NAG pH < 4 & NAG (at pH7) > 10	Acid forming
	kg H_2SO_4/t	NAG pH < 4 & NAG (at pH7) = 5-10	Potentially acid forming
	pH	NAG pH > 4	Non-acid forming
Combined static tests	kg H_2SO_4/t and pH	NAG pH < 4.5 and NAPP > 0	Potentially acid forming
		NAG pH > 4.5 and NAPP < 0	Non-acid forming

From Table 1 it can be seen that a sample with a NAG pH of less than 4 is classified either as acid forming or potentially acid forming depending on the NAG (at pH 7) value. On the other hand a NAG pH greater than 4 is classified non-acid forming. Using a combination of the static tests would mean if a sample has a NAG pH of less than 4.5 and NAPP greater than 0 it would be deemed potentially acid forming. The reverse would deem the sample non-acid forming.

ARD generating potential of ultrafine coal samples

In an investigation conducted by Kotelo (2013), it was demonstrated that pyritic sulfur contributes the highest proportion of the total sulfur (about 50 – 67%) present in the fine coal samples hence making coal ultrafines at a particular risk of ARD generation. Furthermore, Kotelo (2013) found the ARD generating potential in fine coal samples to increase with a reduction in grain size which suggests coal ultrafines to be more prone to ARD generation. ARD prediction tests conducted on ultrafine coal waste have confirmed its potential for acid generation with the NAPP values ranging from 124 kg H_2SO_4/tonne to 3.19 kg H_2SO_4/tonne (Kazadi Mbamba, 2011; Kotelo, 2013; Iroala, 2014) Table 2 summarizes the results of acid base accounting tests and the net acid generation tests conducted on some of the ultrafine coal samples.

Table 2 Acid base accounting and net acid generation test results for ultrafine coal samples (derived from Kotelo (2013) and Iroala (2014))

Sample	ANC tests			NAG tests			
	S grade (%)	NAPP H_2SO_4 (kg/t)	Classification	NAG pH	NAG $_{pH\ 4.5}$ H_2SO_4 (kg/t)	NAG $_{pH\ 7}$ H_2SO_4 (kg/t)	Classification
Waterberg	2.04	52	Acid forming	6.48	-	0.6	Non-acid forming
Witbank	4.18	124	Acid forming	2.29	64.3	10.2	Potentially acid forming
Middleburg	1.08	3.19	Potentially acid forming	3.97	4.51	26.26	Acid forming

2.1.2 Accumulation of major, minor and trace metals in freshwater sources

Major and Minor Metals

As was mentioned previously, the acidic water formed might result in the leaching of metals into groundwater sources (Harrison et al., 2010). Slurry dams containing coal ultrafines and discards are lined with seals or underground leachate collection systems (Keating et al., 2001) which are meant to impede any leachate generated by the fill. However, over time, this leachate can seep into the ground causing an increase in the concentration of dissolved metals in ground water sources (Sheps-pelleg and Cohen, 1998). Some of the major metals occurring in South African coals include Al, K, Mg and Fe and the minor metals include Na and Cr (Mattigod et al., 1990; Eary et al., 1990; Sanyika and Ngcobo, 2014). Table 3 represents the concentrations of the major and minor metals found in coal.

Table 3 Major and minor metal compositions in coal (derived from Mattigod et al., (1990) and Eary et al. (1990))

Element	Concentration Range (wt %)
Aluminium	0.43-3.04
Sodium	0-0.2
Calcium	0.5-2.67
Iron	0.32-4.32
Magnesium	0.1-0.25
Potassium	0.02-0.43
Calcium	0.5-2.67
Barium	≈0.015
Chromium	0-0.016
Zinc	0-0.56
Vanadium	0-0.128
Strontium	≈0.01

Concentrations in coal can be used as an indication of the concentrations in the ultrafine coal waste however variations might occur. Investigations on some ultrafine coal samples have shown concentrations of aluminium and iron in coal to be as high as 40 000 mg/kg and 30 000 mg/kg respectively (Sanyika and Ngcobo, 2014). These metal elements are typically associated with the common mineral phases found in coal such as alunite ($KAl_3(SO4)_2(OH)_6$), pyrite (FeS_2), gypsum ($CaSO_4{\bullet}H_2O$) and kaolinite ($Al_4Si_4O_{10}(OH)_2{\bullet}H_2O$) (Speight, 2005). Table 4 represents the results obtained on investigations on the concentrations of certain metal elements in coal ultrafine samples.

Table 4 Major and minor metal compositions in coal ultrafine samples (derived from Kotelo (2013) and Sanyika and Ngcobo (2014))

Sample	Aluminium (wt %)	Iron (wt %)	Calcium (wt %)	Potassium (wt %)
Witbank	4.29	3.78	0.9	0.16
Waterberg	4.84	3.94	3.66	0.63
Middleburg	3.74	1.4	1.19	0.22

The concentrations of the individual metals in the leachate solution are dependent on the leachabilities of the metals. The leachability of metals is dependent on the form the metals exist as, their liberation and the leach conditions, for example the surrounding pH and, to a lesser extent, redox potential (Jones, 1995). In a recent study, using ultrafine coal samples obtained from the Waterberg and Witbank coalfield region, Sanyika and Ngcobo (2014) conducted an investigation to determine the mobilities of the major elements in coal ultrafines under ARD prediction leach test conditions (described in Section 2.1.1). Solubilisation of these elemental species under NAG test conditions is as a result of the mineral oxidation by the hydrogen peroxide in conjunction with acid-reactions whilst solubilisation under ANC tests is as a result of leaching of the acid-soluble mineral phases at elevated temperatures. Table 5 presents the percent mobilities obtained in the study for the Witbank sample.

Table 5 Percentage mobilities of major metals in coal ultrafines (derived from Sanyika and Ngcobo (2014))

	NAG tests	ANC tests
	% soluble	% soluble
Aluminium	<1	7
Iron	52	32
Calcium	75	85
Potassium	<5	<10

Harrison et al. (2015) notes the differences in solubilisation of Fe in the coal ultrafines between the two test conditions to be as a result of the presence of Fe-bearing mineral phases such as pyrite which do not undergo acid solubilisation without pre-oxidation which is the case for the ANC tests. Sequential Chemical Extraction (SCE) tests were conducted on the same coal ultrafine samples to further investigate the element leachabilities. SCE tests help to provide information on the partitioning of metals between different phases by subjecting sample material to leachants of various strengths (Hansen, 2004). The results obtained confirmed the relatively low leachability of aluminium which was evidenced by its low extraction in all of the stages. Furthermore, the SCE tests indicated the presence of iron in different mineral phases in the coal ultrafines which was evidenced by its extraction at each of the stages as well as the significant amount of iron in the final residue material (Sanyika and Ngcobo, 2014). SCE tests conducted by Hansen (2004) on coal stockpiles show consistent trends in terms of leachability of aluminium. Table 6 shows the different leachabilties obtained from the SCE tests conducted.

Table 6 Major metal leachabilities in coal (derived from Hansen (2004))

Element	Leachability
Aluminium	Strongly bond
Barium	Significant leachability
Chromium	Significant instantaneous release
Strontium	Leachability varied with coal sample
Iron	Strongly bond
Manganese	Significant leachability

Whilst having a relatively low toxicity limit, these metals may accumulate within local water sources to concentrations above the accepted toxicity limits (Broadhurst and Petrie, 2010). High iron and aluminium concentrations in particular have been a concern to fisheries due to their negative impact on aquatic life (Svobodova et al., 1993). Table 7 represents the environmentally significant concentration levels for metals on the basis of their hazard potentials which were calculated using water quality criteria and average crustal abundance data.

Table 7 Environmentally significant concentration levels for metals on the basis of their hazard potentials (adapted from Broadhurst and Petrie (2010))

Group description	Estimated environmentally significant concentration levels (mg/kg)	Elements
Potential for environmental risk if present at moderate available concentration levels	100–1000	Sn, Cr, Ba and V
Potential for environmental risk only if present at relatively high available concentration levels	A: 1000–10,000	Fe, Al, Sr
	B: >10,000	Mg, Na, K and Ca

Trace Metals

Just like major and minor metals, the acidic water formed from ARD generation might also contain trace metals leached from a mineral phase. Trace metals in coal are referred to as metals present in coals or coal waste in concentrations of 100 parts per million (ppm) or less (Bergh, 2013). Most of these trace elements are classified as heavy metals on the periodic table and heavy metals are associated with serious negative health impacts. An example is mercury, which is known to cause neural and cardiovascular diseases or arsenic, which can cause major harm to blood vessels and the human nervous system (Bergh, 2013).

With the motivation of assessing the environmental risk presented by coal mining and utilization activities, several researchers to date have investigated the trace element concentrations present in South African coal samples (Cairncross, 1990; Faure et al., 1996; Wagner and Hlatshwayo 2005; Bergh, 2009; Bergh et al., 2011; Wagner and Tlotleng, 2012). Although there are few reports of trace metal concentrations in coal ultrafine waste, the trace metals found in coal can still give an indication of the concentrations in the coal ultrafines. Table 8 summarizes some of the trace metal concentrations obtained for the different coal samples in an investigation conducted by Bergh (2013) and includes the global concentration averages for trace elements in coal. In another investigation of the trace element concentrations in coal samples, Wagner and Tlotleng (2012) noted that the trace metal concentrations (As, Cd, Pb, Hg, Se, V, U and Th) in some of the run-of-mine coal samples exceeded the global concentration averages. More specifically, the mercury concentrations were found to be as high as 2.43 ppm in the samples, which presents a major environmental concern.

Table 8 Trace element concentrations in some of the South African coal samples (Bergh, 2013)

Trace element	Witbank 4 seam (ppm)	Highveld 4 seam (ppm)	Witbank 2 seam (ppm)	Waterberg upper ecca (ppm)	Waterberg coalfield (ppm)	Global averages (ppm)
Arsenic	5.5	3.1	4.6	11.4	9.3	5
Cadmium	0.3	0.4	0.6	0.2	0.13	0.6
Lead	15	7	10	76	23.4	25
Mercury	0.29	0.2	0.12	0.37	0.9	0.12
Selenium	0.8	1	0.9	1.3	1.19	3
Uranium	2.6	*	4	4.7	3.07	1.2
Thorium	8.9	*	15	9.3	7.5	3.1

*- not analysed

Bergh (2013) also investigated the partitioning of the trace elements within the Coalfield number 4 seam in relation to the organic and inorganic affinity of the trace elements in order to postulate possible trace element reduction methods in coal wastes. In the investigation, through statistical analyses, the development of a correlation matrix and through curve fitting, it was observed that most of the inorganically associated trace elements (Hg, As, Cd, Pb, Mo, Cu, Se and Co) are present as sulfides or are closely associated with sulfide minerals. This is in agreement with other researchers (Babu, 1975; Dai et al., 2006; Diehl et al., 2004; Goodarzi, 2002), who have reported that many trace elements in coal, specifically mercury, have a strong affinity to pyrite. To reduce the associated mineral and maceral content such as pyrite and thereby reduce specific trace element content, Bergh (2013) suggested the use of traditional dense medium, gravity separation and froth flotation processes during coal beneficiation. Whilst some of these metals are readily available (characterised with high leachabilities), some of them are strongly bond and are less likely to leach into solution. Using SCE tests, Hansen (2004) investigated the leachabilities of trace metals in two coal samples from two different South African power stations, Table 9 represents the results of this investigation.

Table 9 Leachabilities of trace metal elements in coal samples from two coal stockpiles (derived from Hansen (2004))

Trace Metal	Leachability
Silver	Strongly Bond
Arsenic	Strongly Bond
Boron	Significant-Instantaneous
Cadmium	Intermediate
Cobalt	Intermediate-Significant
Copper	Strongly Bond- Intermediate
Nickel	Strongly Bond
Lead	Intermediate-Significant
Selenium	Strongly Bond
Stronium	Intermediate- Significant

As can be seen from Table 9, the investigation revealed silver, arsenic and to be strongly bond whilst boron was shown to be the most leachable. Table 10 presents the environmentally significant concentration levels for the trace metals found in coal on the basis of their hazard potentials.

Table 10 Environmentally significant concentration levels for metals on the basis of their hazard potentials (adapted from Broadhurst and Petrie (2010))

Group description	Estimated environmentally significant concentration levels (mg/kg)	Elements
Potential for environmental risk if present at very low (trace) available concentration levels	<10	Hg, Ag, Cd, Se, Sb, As, Au
Potential for environmental risk if present at low (minor) available concentration levels	10–100	Pb and Ni
Potential for environmental risk if present at moderate available concentration levels	100–1000	Mn and B
Potential for environmental risk only if present at relatively high available concentration levels	1000–10,000	Ti

2.1.3 Salinization

Some of the common salt forming anions and cations found in coal such as calcium, magnesium, chlorides, sulfates and bicarbonates may also be leached into freshwater sources in the presence of acidic conditions. Studies have shown sulfate bearing minerals such as jarosite, alunite and gypsum to occur in relatively significant quantities in coal (Gluskoter 1975; Skousen et al. 1997; Naiker et al., 2003; Johnson and Hallberg, 2005; Bryan, 2006; Kotelo, 2013). Coal ultrafines in particular have been noted to contain approximately 2% sulfate-bearing mineral phases with gypsum forming the highest composition (Kotelo, 2013).

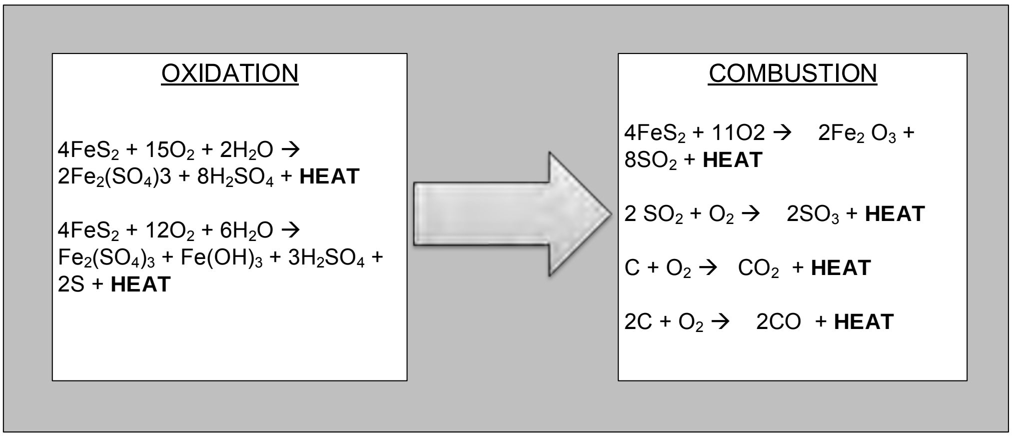
Even though salt forming cations and anions are generally considered non-toxic, their accumulation to high concentrations in local water sources could result in an increase in salinity. A substantial increase in salinity can have an effect on individual aquatic species, domesticated and wildlife animals and on microbial and ecological processes (Dallas et al., 1998). For example, an increase in salinity can result in a decrease in the palatability of water since the main water quality constituents that are associated with palatability effects are chloride, sulfate, magnesium, bicarbonates and calcium (Leske and Buckley, 2004). This in turn might result in the refusal of animals to consume water, or them consuming below the physical requirement, or in some cases, over consuming. Changes in soil structure and osmotic potential can cause acute and chronic toxic effects on animals such as diarrhoea and dehydration (Leske and Buckley, 2004). Furthermore high salt levels in surface water may also cause a reduction in the abundance and diversity of wetland vegetation which in turn, may modify temperatures, sediment inputs, and organic material sources (WRC, 2000). Different species have different tolerances for saline water, and can adapt to a certain degree (DWAF, 1996).

The salinity of water bodies is measured either by TDS (Total Dissolved Salts) or EC (Electrical Conductivity). ANZECC (2000) recommends to keep the salinity of freshwater below 3000 mg/l TDS so as not to present any threat to aquatic species and livestock and notes salinity values above 4000 mg/l TDS to result in loss of production and a decline in animal condition and health .

2.1.4 Spontaneous combustion of coal waste

The spontaneous combustion of coal waste has adverse environmental consequences. Fires started by the spontaneous combustion of coal waste could spread towards nearby settlements and vegetation, posing a serious threat to the ecosystem and the inhabitants of settlements nearby. In areas where there is a tendency for heavy rain and high atmospheric temperature, a minor explosion might even occur (Xiangguo et al., 1994). Spontaneous combustion can also compromise the local air quality since a significant amount of toxic gaseous emissions are emitted as a result. According to Yuping (1994), one tonne of typical coal waste will produce 0.84 kg SO_2, 0.61 kg H_2S, 0.03 kg NO_x, 99.7 kg CO and 0.45 kg smoke in the event of it combusting. Mercury, which is highly toxic, has also been reported to have been emitted from spontaneous combustion of coal waste (Dlamini, 2007). Incidences of spontaneous combustion and burning of coal waste have been reported in some coalfields in Witbank, Sasolburg and the Waterberg coalfield region (Bell et al., 2001; Pone et al., 2007).

Two main stages are involved in the spontaneous combustion of coal waste, these include the pyritic oxidation and the combustion of the coal waste. In the oxidation stage, pyritic sulfur reacts with oxygen, forming sulfuric acid. During the oxidation stage, a significant amount of heat is produced, the heat produced then accumulates in the coal waste until the coal ignition temperature is reached. When the ignition temperature is reached, the carbon present in the coal waste reacts with oxygen to produce carbon dioxide and heat, which is now the combustion stage. Since the combustion chemical reactions are exothermic, the rate of reaction doubles for every 10 °C temperature rise, further perpetuating the reactions (Speight, 1983; Goodarzi and Gentzis, 1991). Figure 5 presents the equations that occur in each of the stages.



OXIDATION

$4FeS_2 + 15O_2 + 2H_2O \rightarrow 2Fe_2(SO_4)3 + 8H_2SO_4$ + **HEAT**

$4FeS_2 + 12O_2 + 6H_2O \rightarrow Fe_2(SO_4)_3 + Fe(OH)_3 + 3H_2SO_4 + 2S$ + **HEAT**

COMBUSTION

$4FeS_2 + 11O2 \rightarrow 2Fe_2O_3 + 8SO_2$ + **HEAT**

$2SO_2 + O_2 \rightarrow 2SO_3$ + **HEAT**

$C + O_2 \rightarrow CO_2$ + **HEAT**

$2C + O_2 \rightarrow 2CO$ + **HEAT**

Figure 5 Equations associated with the two main stages of spontaneous combustion (derived from Liu et al. (1997))

The ignition temperature of coal in general varies with the coal type. Lignite coals ignite from around 250°C - 450°C whilst bituminous coals ignite from 400°C - 500°C and anthracite coals ignite from 700°C - 800°C (Shen, n.d.). Since South Africa's coal quality is mainly bituminous steam coal (Eberhand, 2011), the coal ignition temperatures are likely to range around 400°C - 500°C. The ignition temperatures of the coal waste will mimic that of the parent coal especially in the case unbeneficiated ultrafine coal, which is essentially coal (calorific value close to that of ROM coal).

Factors affecting the likelihood of spontaneous combustion in coal and coal waste

Guney (1968) categorized the factors that affect the spontaneous combustion of coal into intrinsic and extrinsic factors, intrinsic being the factors to do with nature of the coal or the coal waste and extrinsic being the factors to do with external conditions. Table 11 summarizes some of these factors and their corresponding effects on the likelihood of spontaneous combustion.

Table 11 Intrinsic and extrinsic factors affecting the likelihood of spontaneous combustion of coal

Intrinsic Factors	Extrinsic Factors
Presence of pyrite	Temperature, moisture, barometric pressure, oxygen concentration
Inherent moisture	Presence of bacteria
Particle size and surface area	Disposal method
Rank and petrographic constituents	Ventilation and/or air flow rate
Mineral matter and inorganic components	Timbering roadways

With an increase in the pyrite content, the likelihood of spontaneous combustion increases. This is due to its participation in the oxidation reactions. Chandra and Prasad, (1990) however postulated that low pyrite contents (<2%) had negligible effect on the probability of spontaneous combustion of the coal. With regards to the rank and petrographic constituents, it has been noted that an increase in the amount of carbonaceous material present results in an increase on the spontaneous combustibility of the coal waste (Neiburger et al., 1977; Smith and Glasser, 2004). The same relationship applies for an increase in microbial activity and exposure to air. Furthermore, a decrease in particle size would result in an increase in the risk of spontaneous combustion occurring (Sujanti and Zhang, 1999). Ultrafine coal waste is particularly at a high risk of spontaneous combustion due to its high carbonaceous content (approximately 41%-56%) and its fine nature.

Prevention of spontaneous combustion

Due to the complex nature of the combustion rate equations and initial activation energies, no one equation has been established to accurately predict the likelihood of a specific coal sample to spontaneously combust. Prevention methods are therefore centred on the manipulation of the previously mentioned parameters (Liu et al., 1997). Some of the prevention methods include the removal of the pyrite from the coal or coal waste, the recovery of the coal from the coal waste, and the use of an appropriate disposal method (Liu et al., 1997). With regards to waste disposal methods, it has been noted that compacting the pile to restrict the availability of air (which participates in the oxidation of pyrite), addition of soil covers to the coal waste and the effective removal of moisture from the dump is an effective way of preventing ignition of the waste (Fierro et al., 1999; Watkiss, 1999).

2.1.5 Land use

Mining operations are associated with extensive land disruption and can change the topographical features and land use patterns of a specific area. Slurry dams specifically can extend for many kilometres, and often result in land sterilisation (Notten, 2001). Even when the dumps do get eventually rehabilitated, the land areas are rarely returned to their original state (Russell, 1991). The Waterberg area, which is occupied by most of the coal fields in South Africa (refer to Figure 1) has traditionally been used extensively for farming (Wells et al., 1992) therefore land use for coal waste management results in loss of valuable agricultural land which is detrimental to the economy. Some of the mines in the Waterberg region such as the Grootegeluk Mine have slurry dams that can cover an average area of 76.1Ha of land and are about 60m deep (Exxaro, 2006). This is a significant amount of land that could have been utilized for agriculture or settlement.

2.1.6 Dust emissions

Mine waste deposited in open slurry dams also presents a risk of dust emissions (Notten, 2001). The factors which affect dust emissions can be divided into two main categories, these include aerodynamic factors and the physical characteristics of the solid particles (Gillette, 1979). Aerodynamics is the study of the properties of moving air and the interaction between the air and solid bodies moving through it. Physical characteristics which affect the magnitude of dust emissions include solid particle size, segregation and density. Solid particles which are fine in nature, more segregated and have a low density are more prone to be carried away in the wind resulting in more dust emissions compared to those that comprise of larger and denser particles (Gillette, 1979). The aerodynamics of the air in a specific region containing the coal waste would depend on the climatic properties of that area. In general, areas that are windier are more prone to dust emissions emanating from coal waste dumps. The emissions of dust to the air can greatly compromise the local air quality, with inhalation of dust particles likely to result in respiratory illnesses, lung diseases and in some cases heavy metal poisoning, especially at higher levels of exposure (WHO, 1999).

2.1.7 Loss of resources

Since 2003, South Africa's coal reserves have notably reduced and there has been an indication that the present remaining reserves comprise of only about 15 billion tonnes of coal (Hartnady, 2010). Given South Africa's heavy dependence on coal for power generation (described in Section 1.1.1) and an anticipated peak in the economic production in 2020, the shortage of coal will likely have a significant impact on the economy (Hartnady, 2010). This rapid decline in coal reserves in South Africa has drawn attention to the importance of identifying alternative coalfields besides the main Waterberg region. Approximately 240 000 TJ (calculated from the calorific value of ultrafine coal waste and the coal ultrafine slurry lost each year) of energy however is lost annually by the disposal of an estimated 4 million tonnes of dry ultrafine coal each year (DME 2001; Reddick 2006).

Ultrafine coal has the same calorific value as that of ROM coal and can be utilized for power generation. Opportunities have become available to recover the ultrafine coal in South Africa and research is being undertaken to investigate the options available for ultrafine coal utilisation, with specific emphasis on power production and clean coal technology (Reddick, 2006). Amongst these clean coal technologies investigated, fluidised bed combustion (FBC) has been identified as a lucrative option in terms of energy recovery and technical feasibility (North, 1990). Whilst some coal beneficiation plants have already started beneficiating ultrafine coal waste, many plants still dispose it with one of the main reasons being the energy requirements of drying the coal ultrafines to the required moisture content (de Korte, 2008). On the issue of drying the coal ultrafines, van der Scholtz and Trautmann (2007) note the reclamation of the ultrafine coal using solar drying to be more environmentally preferable compared to the reclamation of the coal ultrafines using thermal drying.

Typical thickeners are able to achieve 15-40% solids concentration in the underflow therefore coal ultrafines are disposed of whilst they still contain approximately 60% to 85% of water (Reddick, 2006; Jera, 2013). This results in the loss of water through evaporation (typically 5–40% of tailing water), seepage (typically 5% of tailing water) and entrainment (30–50% of tailing water) (Bleiwas, 2012). This translates into approximately 8 million tonnes of water lost annually. Loss of water resources is detrimental to the South African economic and social development goals as South Africa is a semi-arid country with an average rainfall of 420mm, less than half the global average (WWF, 2011). Furthermore it has been estimated that there will be a gap of about 17% in water demand and supply by 2030 if no significant policy shifts occur (WWF, 2011). Lastly, disposal of coal waste results in the loss of pyrite which can be utilized for different purposes (Stander, 2012), this possibility is explored more in Section 2.2.5.

2.2 The two-stage froth flotation process for the recovery of coal and removal of sulfidic sulfur

Treatment of ARD involves employment of chemical, biological and physical processes, some of which include chemical treatment with mineral precipitation, membrane processes, ion exchange and biological sulfate removal processes. Because of the expense associated with some of these methods, as well as the problems associated with the maintenance of end-of-pipe treatment facilities in perpetuity, prevention of ARD has become the more desirable way of mitigating acid rock drainage (GARD Guide, 2013). The prevention of ARD generation rests on the manipulation of the factors that influence its generation with the some of the prevention methods centred on the minimization of the risk of oxidation. These methods include microbial inhibition, back filling the mine wastes back into the voids created by the mining operations, co-disposal of waste rock with tailings, the use of capping covers and desulfurisation of mineral wastes (Skousen et al., 1998; GARD Guide, 2013).

The two-stage froth flotation process was developed at the University of Cape Town with the aim of removing sulfidic sulfur (for ARD mitigation) whilst simultaneously recovering clean coal from ultrafine coal waste hence promoting resource efficiency. Froth flotation is a separation process that takes advantage of the difference in the surface properties of the valuable minerals and the unwanted gangue minerals (Wills and Napier-Munn, 2006). Froth flotation involves three phases, the solids phase, the water phase, and the froth phase. In the process, valuable minerals are made aerophilic (air-loving) and the gangue minerals, aerophobic (water-loving) by adjusting the conditions in the pulp phase by various methods and reagents (Wills and Napier-Munn, 2006). In the first stage of the two-stage froth flotation process, coal with a low ash content is recovered and in the second stage, sulfur bearing minerals are recovered leaving a reduced volume of a non-acid generating tailings stream.

2.2.1 Laboratory-scale flotation testwork

In order to establish the technical feasibility of the process, on a laboratory scale, laboratory test work has been done on the individual stages of the process as well as on both stages together (Kazadi Mbamba, 2011). The reagents used for the flotation stages include collectors, frothers and depressants. For the coal flotation, the collector reagents that have been used include oily collectors such as kerosene, diesel and oleic acid (Kazadi Mbamba, 2011; Iroala, 2014). Recently however, with the increasing demand on high quality coal, novel collectors such as Nalflote, manufactured by NALCO have been considered (Howlett and Marsden, 2013; Iroala, 2014). For the frother, methyl isobutyl carbinol (MIBC) has been used.

The flotability of coal in a flotation process has been described as being largely dependent on the rank of coal with the degree of hydrophobicity of various coals decreasing with decreasing rank (Gutierrez-Rodriguez et al., 1984). This has been demonstrated by investigations conducted on various coal ultrafine samples which revealed a variation of coal recoveries with different coal samples. Table 12 represents the coal product yields obtained from different coal ultrafine samples using reagent dosages of 2.79 kg/t oleic acid and 0.28 kg/t MIBC.

Table 12 Coal product yields on different coal ultrafine samples using oleic acid as a collector at a reagent dosage of 2.79 kg/t and MIBC as a frother at a dosage 0.28 kg/t (Kazadi Mbamba, 2011; Amaral Filho, 2012; Iroala, 2014)

Sample	Feed Characteristics		Product		
	Ash content (%)	Total sulphur (%)	Yield (%)	Ash content (%)	Sulphur content (%)
Middleburg	33.0	1.08	55.98	18.1	0.50
BHP Billiton ultrafine coal	31.4	0.93	30.75	15.62	0.58
Exxaro ultrafine coal	24.1	0.68	89.01	17.4	0.49
Witbank ultrafine coal	56.4	5.40	41.0	32.1	3.00
Waterberg ultrafine coal	49.1	1.92	15.5	45.2	0.8

As can be seen in Table 12, coal yields can go up to as much as 89% but can also be as low as 16%. Whilst variations of the coal product yields exist, laboratory testwork has shown that it is possible to recover a significant amount of coal product from the ultrafine coal waste. As for the sulfide flotation process, the collectors used include xanthate salts such as potassium amyl xanthate (PAX), sodium isobutyl xanthate (SIBX) and sodium ethyl xanthate (SEX). The same frother used in the first stage flotation is also used in addition to a depressant, namely dextrin that aims to supress the flotation of coal in the second stage. Laboratory testwork investigations have also shown variations in the sulfide-rich stream yields from various coal tailings (from the first stage). Table 13 represents the sulfide-rich stream yields obtained using a PAX dosage of 2.33 kg/t, MIBC dosage of 0.11 kg/t and dextrin dosage of 0.93 kg/t. .

Table 13 Sulfide-rich stream recoveries on different coal ultrafine samples using PAX as a collector at a dosage of 2.33 kg/t, MIBC as a frother at a dosage of 0.11 kg/t and dextrin as a depressant at 0.93 kg/t (Kazadi Mbamba 2011; Amaral Filho 2012; Iroala 2014).

Coal Sample	Flotation Stream	Deportment from feed (%)	Ash (%)	Total Sulfur (%)
Middleburg	Feed (coal tailings)	-	38.8	0.92
	Sulfide-rich concentrate	16.3	28.9	2.68
	Sulfide-lean tailings	83.7	40.8	0.38
BHP Billiton	Feed (coal tailings)	-	43.2	0.8
	Sulfide-rich concentrate	45.5	31.7	1.3
	Sulfide-lean tailings	54.5	46.7	0.5
Witbank	Feed (coal tailings)	-	40.2	2.5
	Sulfide-rich concentrate	14.98	4.00	1.2
	Sulfide-lean tailings	85.02	35.6	1.5
Waterberg	Feed (coal tailings)	-	-	-
	Sulfide-rich concentrate	6.86	77.3	0.4
	Sulfide-lean tailings	93.2	-	-

* Compositions for the coal tailings and sulfide-lean tailings were not conducted in the investigation.

As can be seen from Table 13, the sulfide-rich stream yields can be as high as 50% and the total sulfur content in the sulfide-rich stream can be as high 2.7%. Considering the two stages combined, laboratory testwork has shown that the total sulfur in an ultrafine coal waste sample can be effectively reduced from 1% to 0.5%.

Investigations on the effect of varying the reagent dosages and types on the two-stage flotation process

In order to optimize both flotation stages and to investigate the reagents that promote the technical efficiency of the process, different flotation reagents have been tested for both stages. The investigations conducted were mostly focused on varying the collector dosages and collector types. Studies have shown that increasing the coal collector dosage would result in an increase in the recovery. An example is the investigation conducted by Kazadi Mbamba et al., (2012) on a Middleburg coal sample on the effect of varying the dodecane collector dosages on the final yield. In the study it was shown that increasing the dodecane collector dosage by 100% resulted in an increase in the coal yield by 33%. Figure 6 illustrates the results obtained from this investigation. In another investigation on a BHP Billiton coal sample, it was observed that increasing the oleic acid dosage by 160% increases the coal yield by 46% but also increases the coal ash content by approximately 17% (Magabane and Naidoo 2011).

With regards to the relative performance of the collector reagent types, Kazadi Mbamba (2011) investigated the effect of varying the oily collector types on the coal yield. In the investigation it was noted that oleic acid achieved the highest recoveries compared to dodecane and kerosene (illustrated by Figure 6). As can be seen from Figure 6, oleic acid achieved approximately 90% higher recoveries than kerosene and dodecane at a dosage of 2.79 kg/t. Other investigations conducted demonstrated novel collectors such as Nalflote 9858 to be able to achieve better coal yields at lower dosages compared to oily collectors (Howlett and Marsden, 2013). Specifically, it was shown that using Nalflote 9858 instead of oleic acid results in an increase of the coal yield by 10% as well as a decrease in the clean coal product ash content by approximately 7%.

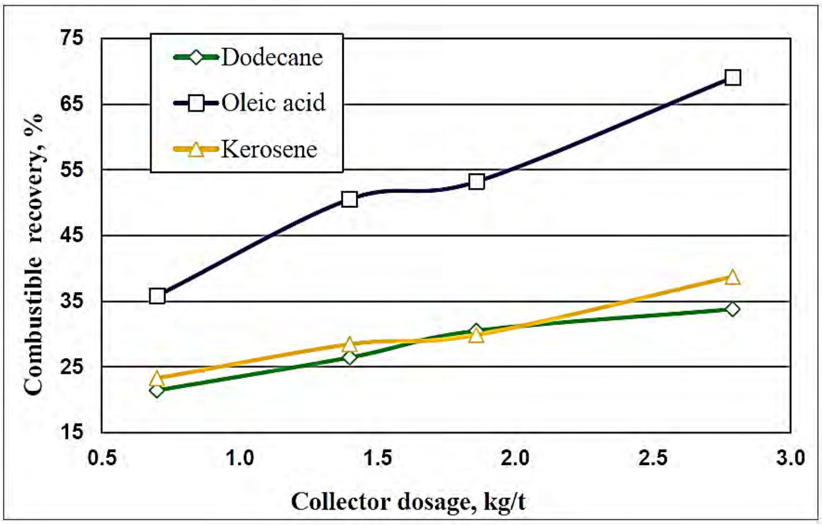


Figure 6 Effect of varying oily collector types on the coal recoveries of a Witbank coal sample (Kazadi Mbamba et al., 2012)

For the sulfide flotation stage, studies have been also conducted to investigate the effect of varying the collector dosage and type. Investigations have shown that increasing the collector dosage increases the sulfide-sulfur recovery. One such investigation demonstrated that increasing the PAX collector dosage on a Middleburg coal sample from 0.93 kg/t to 2.33 kg/t increased the yield from 5% to 27% (Kazadi Mbamba et al., 2012). Investigations on varying collector types have revealed PAX to achieving the highest percentage recoveries compared to sodium isobutyl xanthate (SIBX) and sodium ethyl xanthate (SEX). In one investigation it was demonstrated, on a Waterberg coal waste sample, that up to 85% of the total sulfur can be removed using potassium amyl xanthate as a collector but only about 65% of the total sulfur can be removed using Sodium Isobutyl Xanthate (SIBX) as a collector (Fisher and Toms, 2013). Figure 7 and Figure 8 illustrate the results obtained from the investigations.

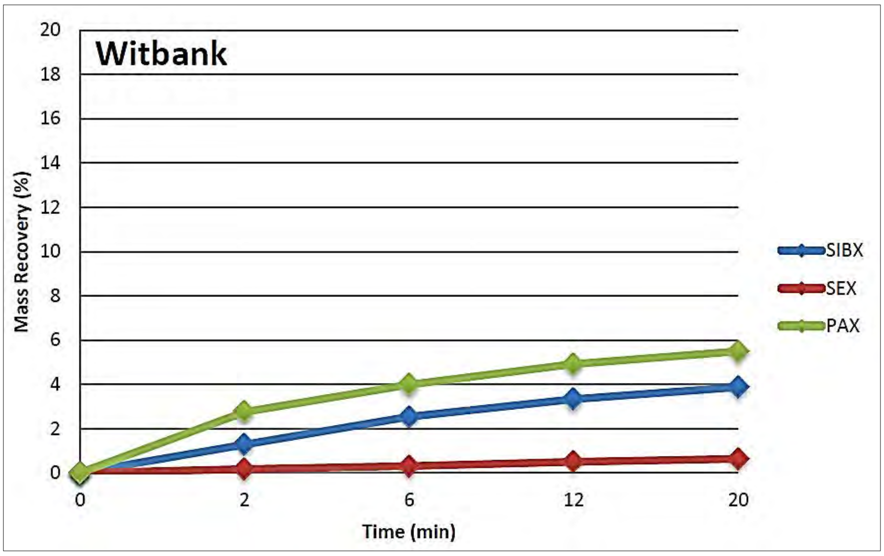


Figure 7 Effect of varying xanthate collector types on the sulfide sulfur recoveries of a Witbank coal sample (Fisher and Toms, 2013)

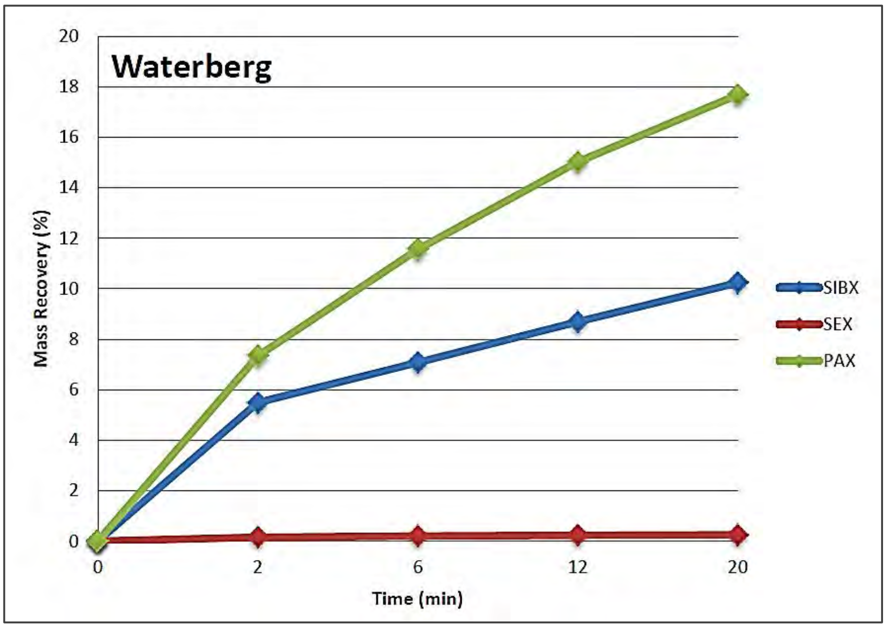


Figure 8 Effect of varying xanthate collector types on the sulfide sulfur recoveries of a Witbank coal sample (Fisher and Toms, 2013)

2.2.2 Acid generating potential

To investigate the acid generating potential of the samples after the two-stage flotation, acid base accounting (ABA) and net acid generation (NAG) tests were conducted on the samples (described in Section 2.1.1). Investigations reported most of the coal waste samples as achieving a non-acid forming classification after the sulfide flotation stage. Table 14 represents the acid generating classifications for the sulfide-lean tailings stream of selected coal waste samples.

Table 14 ARD prediction tests for selected coal samples before and after desulfurisation (Kazadi Mbamba 2011; Amaral Filho 2012; Iroala 2014).

Sample	Sulfide-lean Tailings				
	S (%) grade	NAG $_{pH\ 4.5}$ H_2SO_4 (kg/t)	NAG $_{pH\ 7}$ H_2SO_4 (kg/t)	NAPP (kg H_2SO_4/t)	Classification
Waterberg	0.65	-	0.5	-101	Non-acid forming
Middleburg	0.38	0.00	6.86	-45.17	Non-acid forming
BHP Billiton	0.5	-	-30	-59	Non-acid forming

2.2.3 Conceptual plant model for the two-stage froth flotation process

Based on the laboratory results, literature sources and design principles, Jera (2013) developed fictitious plant models for the two-stage flotation process. Two fictitious plant models were designed to treat nominal coal fines from a dump of an abandoned mine in the Witbank/Middleburg coal field at a basis of 100 ton/hr fine coal (dry) feed rate basis. The first conceptual plant model included a milling circuit, classification circuit, flotation circuit and dewatering circuit whilst the second conceptual plant model excluded the milling and classification circuit. The summarized process diagram of the conceptual plant model is presented by Figure 9.

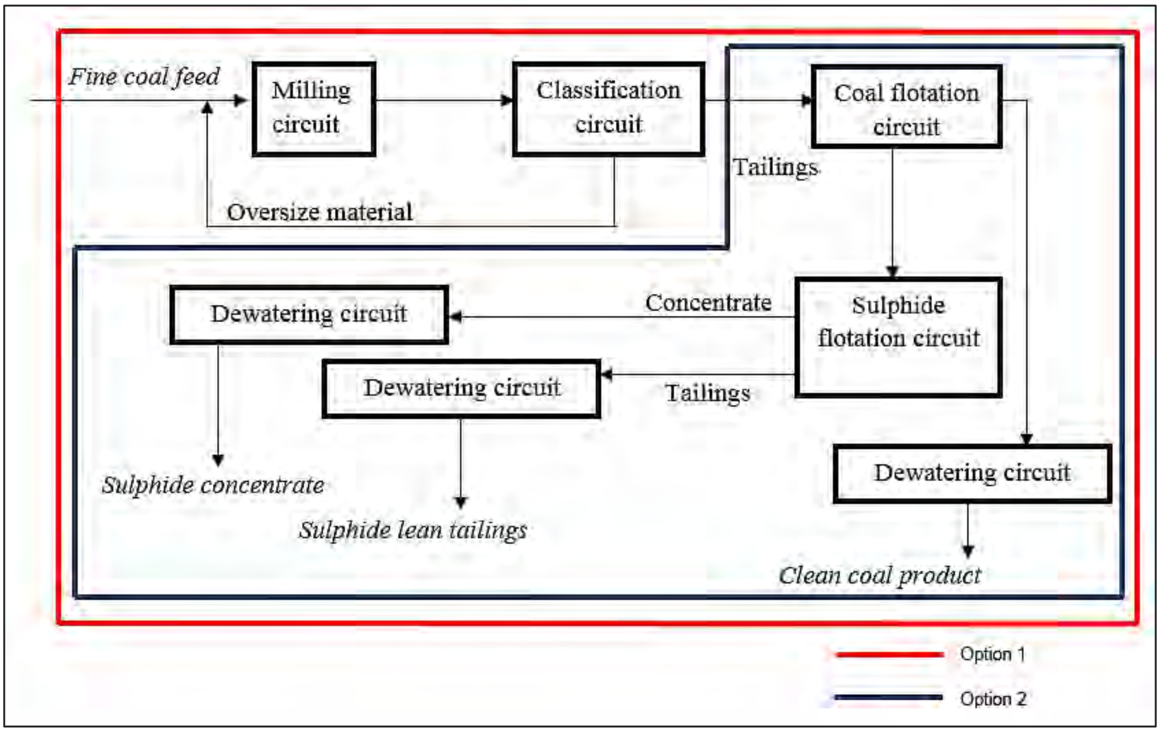


Figure 9 Process diagram for the fictitious plant models (Jera, 2013)

The dewatering circuit included thickeners and filter presses. The thickener, which was calculated to have a diameter of 20m, was assumed to achieve 40% solids in the underflow slurry whilst the filter presses for the clean coal, sulfide-lean tailings and sulfide-rich concentrate, which were modelled to have capacities of 15 m^3, 2.2 m^3 and 1.5 m^3 respectively, were assumed to achieve 10% moisture content in the product. The operating energies of the clean coal, sulfide-lean tailings and sulfide-rich tailings filter presses were calculated to be 110 KW, 90 KW and 30 KW respectively and were calculated to be operating at a rate of 7 cycles/hr. As for the flotation circuit, the coal flotation unit and sulfide flotation unit each comprised of 7 and 11 flotation cell banks respectively with each cell having a volume of 43 m^3. The total energy usage for the first plant model was estimated to be about 2MW whilst that of the second plant model was estimated to be about 1.2 MW.

2.2.4 Economic feasibility

Following from the development of the fictitious plant model, Jera (2013) conducted an economic assessment of the two-stage froth flotation process for the first fictitious plant model (highlighted in Figure 9). In the economic assessment, all three products of the two-stage flotation process were identified to having revenue potential however only the clean coal product was considered when calculating the total revenue. The major expenses of the process came from the flotation reagents (MIBC, oleic acid, PAX and dextrin) and the total electricity usage of the process. For the treatment of 100 t/h coal waste using an oleic acid dosage of 2.79 kg/t to achieve a coal yield of 80% (based on laboratory test work), the model showed a Net Present Value (NPV) of ZAR50 million and an Internal Rate of Return (IRR) of 19%, which Jera (2013) describes as being economically lucrative.

The economic assessment was subjected to a sensitivity analysis in order to show how positive and negative changes in the input variables may affect the project Net Present Value (NPV). From the sensitivity analysis it was realised that reducing the selling price of coal by 20% from the base line resulted in a net loss of the business. Furthermore, it was noted that the input parameters having the most impact on the NPV of the model were the operating costs and the coal selling price. The effect of variation of coal price, coal yield, reagent costs and operating costs is illustrated by Figure 10.

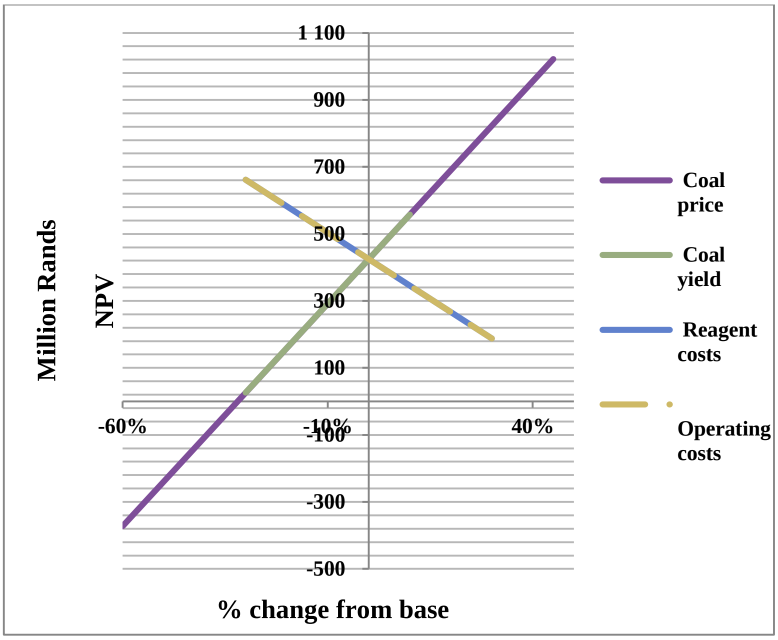


Figure 10 Relationship of NPV of the fictitious plant model with respect to reagent costs, capital costs and operating costs (Jera, 2013)

2.2.5 Downstream utilisation of separated tailings

Sulfide-rich product

Several potential uses of the sulfide-rich product have been identified in industry with many still under investigation. Some of the potential uses that have been identified include its use in the production of sulfuric acid, as a soil ameliorant, in the production of ferric sulfate or ferrous sulfate heptahydrate crystals, in copper smelting, in chrome reduction, in cemented paste backfill, in lead refining, as a raw material in solar panels, for heat generation in bioleaching, in actuators, as a heat sink, to make sulphite cellulose and as cathode material in batteries (Stander, 2012).

Whilst some of these uses are relatively new technologies and largely still under the development and in the feasibility stage (Stander, 2012), some of them are already well established and practised in some countries. An example is the commercial production of sulfuric acid using sulfide-sulfur recovered from coal dumps in China (Harrison et al., 2010). Specifications for the sulfide-rich product for sulfuric acid manufacture range from 5-8% maximum carbon limit and 42-45% minimum sulfur limit. Comparing this specification with the sulfide-rich stream associated with the two-stage flotation process, where the highest concentration of sulfur achieved has been 18%, utilization of this stream for sulfuric acid production will require the product to be first substantially upgraded (Stander, 2012).

Sulfide-lean tailings

The sulfide-lean tailings can be used in backfilling, as cover material, as co-disposal material, to construct wetlands, in road construction, as a cement additive, and as raw material for glass or rock wool and glass ceramics manufacturing (Marabini et al., 1998; Sobolev and Arikan, 2002; Doye and Duchesne, 2005). The potential use of specific sulfide-lean tailings is heavily dependent on the nature of the parent rock, for example, the sulfide-lean tailings which have a high silicate content are more suited to the production of glass ceramics whilst the tailings with a neutralizing capacity are more suited to be used as a co-disposal material with acid generating tailings (Harrison et al., 2010) Examples of mines that have incorporated this utilisation include the Doyon mine, Quebec, Canada, where the sulfide-lean tailings were used as part of backfilling material (Benzaazoua et al., 2008) and at the HZL zinc plant in Chanderia, India, where sulfide-lean tailings were used for backfill operations, in constructing embankments, mine roads, playgrounds, or as landfill liners (Agrawal et al., 2004).

It has been noted through investigations that the utilisation of the sulfide-lean tailings for co-disposal reduces the permeability of the deposit which minimizes the acid forming conditions. Furthermore it has been observed that co-disposal of the waste results in increased cementation and stabilization of the waste. This concept is currently under study at the University of Cape Town.

2.3 Potential environmental implications of the two-stage flotation process

As outlined by Broadhurst et al. (2014), the pre-disposal removal of sulfide sulfur from mine wastes can result in both environmental benefits and burdens. In accordance with a case study conducted on base metal sulfides (Broadhurst et al., 2014; Kunene, 2014), environmental benefits include a decrease in human toxicity and eco-toxicity associated with metal release, urban land occupation and natural land transformation, whilst a detailed inventory analysis indicated additional benefits due to reduced dissipative water losses, enhanced opportunity for recovery of abiotic mineral resources, and reduced aqueous acidification and salinization. However, this study also showed an increase in climate change, fossil fuel depletion and terrestrial acidification impacts, which were attributed largely to the additional consumption of fossil-fuel based electricity in the flotation process, as well as the environmental burdens associated with the production of flotation reagents. The study also indicated that the desulfurised tailings could be expected to contain residual levels of the xanthate flotation reagent used, although the toxicity effects of xanthate relative to that of the base metals, particularly Zn, was found to be negligible for the base metal case study (Broadhurst et al., 2014; Kunene, 2014). Similar impact categories are expected to be of relevance to the two-stage flotation process proposed for the management of coal wastes, although the nature and characteristics, and hence associated impacts, of some of the material inputs and outputs will be different. The following subsections expand on the potential impacts associated with the flotation reagents of specific relevance to the recovery of coal and removal of sulfide sulfur from coal wastes.

2.3.2 Flotation reagent toxicity

To gauge the possible environmental implications of the release of flotation reagents into local water sources it is paramount to establish the properties and toxicities of these reagents. Table 15 represents a summary of the environmental properties and toxicities of the flotation reagents used in the two-stage flotation process. Toxicological information is presented in terms of LC50s and LD50s. LD50 (which can be referred to as lethal dosage) is a standard measurement of acute toxicity that is measured in milligrams (mg) of pesticide per kilogram (kg) of body weight (USEPA, 2015). It is the individual dose required to kill 50 percent of a population of the test species (e.g., rats, fish, mice). The lower the LD50 dose of a substance, the more toxic the substance. LC50 is defined as the median lethal concentration of a substance needed to kill 50% of the organisms within a specified period of time (USEPA, 2015).

As can be seen from Table 15, kerosene is more toxic in terms of its LD50 than oleic acid. In terms of the hazardous nature of the oily collectors, oleic acid is the least hazardous with kerosene and dodecane having very similar hazardous properties. Unfortunately no information is available on the nature of the synthetic collector, Nalflote, hence its toxic nature and hazard properties cannot be compared with the oily collectors. With regards to the xanthate collectors, in terms of the toxicological data, SEX is more toxic than PAX however in terms of their hazardous nature and ecological properties, all three collectors are very similar. Comparing the xanthate collectors to the oily collectors, the xanthate collectors are more toxic in terms of toxicological properties, hazardous nature and ecological properties. Several literature sources report xanthate salts to be highly toxic to aquatic organisms especially when discharged directly to waterways (Hawley, 1977; Xu et al., 1988; Webb et al., 1976; Boening, 1998; Qun et al., 2011). However, since xanthates decompose rapidly in aquatic ecosystems (NICNAS, 1995), the environmental implications of their disposal might be linked to the toxicity of the degradation products and not necessarily the xanthate salts themselves. Carbon disulfide which is one of the degradation products, has been noted to be deadly and to cause nervous system damage and psychosis (Smith and Timmerman, 2000; Newhook et al., 2002). Unfortunately no information of the properties of Nalflote is available as the collector mixture is a patented technology by NALCO Company.

Table 15 Properties and toxicities of the flotation reagents associated with the two-stage flotation process (Derived from Sciencelab (2015) and NICNAS (1995))

Reagent		Toxicological Information	Hazardous Identification	Ecological Information
Collectors	Oleic acid	Acute oral toxicity (LD50): 25000 mg/kg Species: Rat	Potential Acute Health Effects: Slightly hazardous in case of skin contact or if ingested and causes irritation when in contact with eyes	Toxicity of the Products of Biodegradation: The product itself and its products of degradation are not toxic.
	Kerosene	Acute oral toxicity (LD50): 15000 mg/kg Species: Rat, 20000 mg/kg Species: Guinea pig,	Potential Acute Health Effects: Hazardous in case of skin contact (irritant), of eye contact (irritant), of ingestion, of inhalation. Slightly hazardous in case of skin contact.	Products of Biodegradation: Possibly hazardous short term degradation products are not likely. However, long term degradation products may arise.

Reagent		Toxicological Information	Hazardous Identification	Ecological Information
		2835 mg/kg Species: Rabbit	Severe over-exposure can result in death.	
	Dodecane	Acute oral toxicity: LD50: unavailable VAPOR (LC50): Acute: >142 ppm 8 hours Species: Rat	Potential Acute Health Effects: Hazardous in case of inhalation. Slightly hazardous in case of skin contact (irritant), of eye contact (irritant), of ingestion. Severe over-exposure can result in death.	Possibly hazardous short term degradation products are not likely. However, long term degradation products may arise
	Nalflote	unavailable	unavailable	unavailable
	SIBX	unavailable	Potential Acute Health Effects: Harmful in contact with skin and if swallowed. Irritating to eyes and skin.	Toxic to aquatic organisms. Degrade to ethanol, carbon disulphide and caustic soda in tailings impoundments
	SEX	Acute oral: LD50 730 mg/kg Mice (male)	Potential Acute Health Effects: Harmful in contact with skin and if swallowed. Irritating to eyes and skin. Irritating to eyes and skin.	Toxic to aquatic organisms. Degrade to ethanol, carbon disulphide and caustic soda in tailings impoundments
	PAX	Acute oral: 1 000 mg/kg Rats	Potential Acute Health Effects: May be fatal if swallowed. Harmful if inhaled. Causes skin and eye irritation	Toxic to aquatic organisms. Degrade to ethanol, carbon disulphide and caustic soda in tailings impoundments
Frother	MIBC	The acute oral LD50 values for MIBC are 2260 - 2970 mg/kg	Potential Acute Health Effects: Can cause irritation to the eyes, nose and throat	94% is biodegraded within 20 days.

Reagent		Toxicological Information	Hazardous Identification	Ecological Information
Depressant	Dextrin	unavailable	Potential Acute Health Effects: Slightly hazardous in case of ingestion, of inhalation	Possibly hazardous short term degradation products are not likely. However, long term degradation products may arise

As for the frother and depressant reagents, methyl isobutyl carbinol, has a relatively low toxicity, in terms of its toxicological properties compared to the other flotation reagents. With regards to Dextrin, no information is available on its toxicological properties however it is not associated with any serious hazardous properties and does not exhibit any potential harm to aquatic life, which as was mentioned, is the case for the xanthate salts.

2.3.3 Flotation reagent production processes

Xanthate salts are produced by reacting carbon disulfide with an alcohol and a caustic according to the equation 5, represented below, where MOH is a caustic, ROH is an alcohol and $ROCS_2M$ is the xanthate salt. Production of the carbon disulfide process involves reacting carbon and sulfur according to equation 6, where H_2S represents hydrogen disulfide, a bi-product of the reaction. Unreacted sulfur from the carbon disulfide reaction is removed in a sulfur condenser whilst carbon disulfide is separated from hydrogen sulfide by absorption before it is stripped and sent to a distillation column to remove small amounts of impurities. Sulfur is recovered from the H_2S gas stream by combusting the H_2S with air to produce sulfur dioxide and further reacting the H_2S with the SO_2 to produce sulfur (Equation 7 and 8). This process can be associated with the release of a considerable amount of SO_2 emissions to the atmosphere (Kunene, 2014).

$$ROH + CS_2 + MOH \rightarrow ROCS_2M + H_2O \tag{5}$$

$$CH_4 + 4S \rightarrow CS_2 + 2H_2S \tag{6}$$

$$H_2S + 1.5O_2 \rightarrow SO_2 + H_2O \tag{7}$$

$$H_2S + SO_2 \rightarrow 2S + H_2O \tag{8}$$

The production of oleic acid is associated with the production of fatty acids. In the process, stearic acid is dehydrogenated to give the monounsaturated derivative oleic acid (Cornils and Lappe, 2000). The production of kerosene and dodecane on the other hand, is associated with the fractional distillation of petroleum (Collins, 2007). Dextrin, a compound of a group of low molecular weight carbohydrates is produced by the hydrolysis of starch or glycogen by applying dry heat under acidic conditions (pyrolysis or roasting) (Haas and Hill, 1929). Lastly, the production of methyl isobutyl carbinol is associated with the production of organic alcohols through organic synthesis. All of the reagent production processes require a considerable amount of energy with the amount of energy required varying with the nature of the production process. Consumption of this energy will therefore result in the release of emissions and the consumption of resources.

2.4 Assessing the environmental impacts of managing coal waste

The assessment and quantification of potential environmental impacts associated with coal waste disposal methods is paramount and forms a crucial part in the decision making process. To date various analytical tools have been developed to assess environmental performances of processes in general. These tools are supported by various technical elements, modelling tools, and supporting data (Hansen, 2004). The following subsections will explore the life cycle assessment tool and risk-based environmental performance indicators in detail.

2.4.2 Life Cycle Assessment

LCA is a technique used to evaluate the environmental aspects and potential impacts associated with a product or process. It involves collating an inventory of inputs and outputs of a product system, evaluating the potential environmental impacts associated with those inputs and outputs and interpreting the results ((ISO)14040, 2006). Life Cycle Assessment stages include the goal and scope definition stage, the life cycle inventory stage, the life cycle impact assessment stage and the interpretation stage. In the goal definition stage, the purpose of the study, the scope, the boundary of the research and the functional unit are clearly defined before moving on to the life cycle inventory stage where the foreground and background data is collected. Foreground data is data that is required to model a system, specifically data that describes a particular product system whilst background data is the data for materials, energy, transport and waste management systems (Goedkoop et al., 2009). A crucial step in LCA is the impact assessment. Impact assessment stages include the selection of impact categories based on their relevance to the study, assignment of results from the inventory to the selected impact categories (termed classification), calculation of category indicator results using characterisation factors, calculation of the category indicator results relative to reference values (termed normalisation) and finally weighting and data quality analysis.

The last three steps are optional and can be included or excluded depending on the goal and scope of the project. (Hansen, 2004) notes however that since normalisation may be based on a number of factors, including social, economic, global, technical and base line indicators it may result in a clearer representation of the information which in turn allows results to be compared more easily. Weighting on the other hand, which is meant to enhance the interpretation of impact assessment through the weighting of the different impact categories according to their perceived relative importance is associated with very little consensus (Finnveden et al., 1995) and could introduce bias to the study. Figure 11 illustrates the LCA methodology.

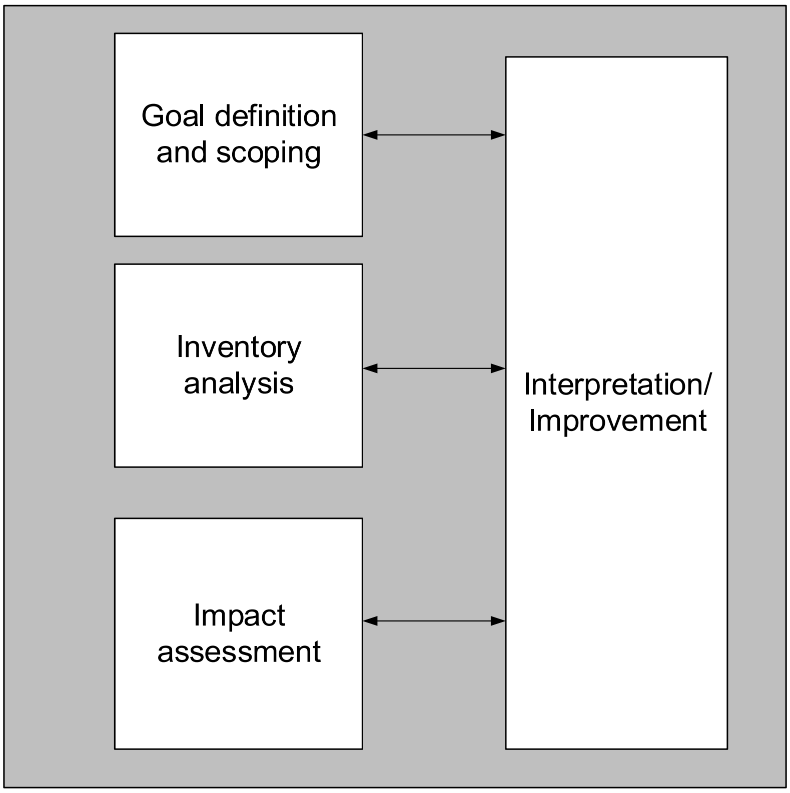


Figure 11 Components of the LCA methodology (ISO, 1997)

Impact Categories and Category Indicators

An impact category, can be defined, according to Notten (2001), as "*relevant classes of environmental concern*" and a category indicator can be defined as a quantifiable representation of an impact category which is obtained by multiplying the inventory result by a characterisation factor. Characterisation factors (or equivalency factors) are developed to quantify the relative severity of each environmental intervention's contribution to an impact category. According to the goal of the study, impact categories can be selected at the midpoint or endpoint of the cause-effect chain however it is generally noted that as one moves down the cause effect chain, whilst the study becomes more relevant, the credibility of the study is reduced (Notten 2001). Figure 12 illustrates the relationship between LCI results and the cause-effect chain.

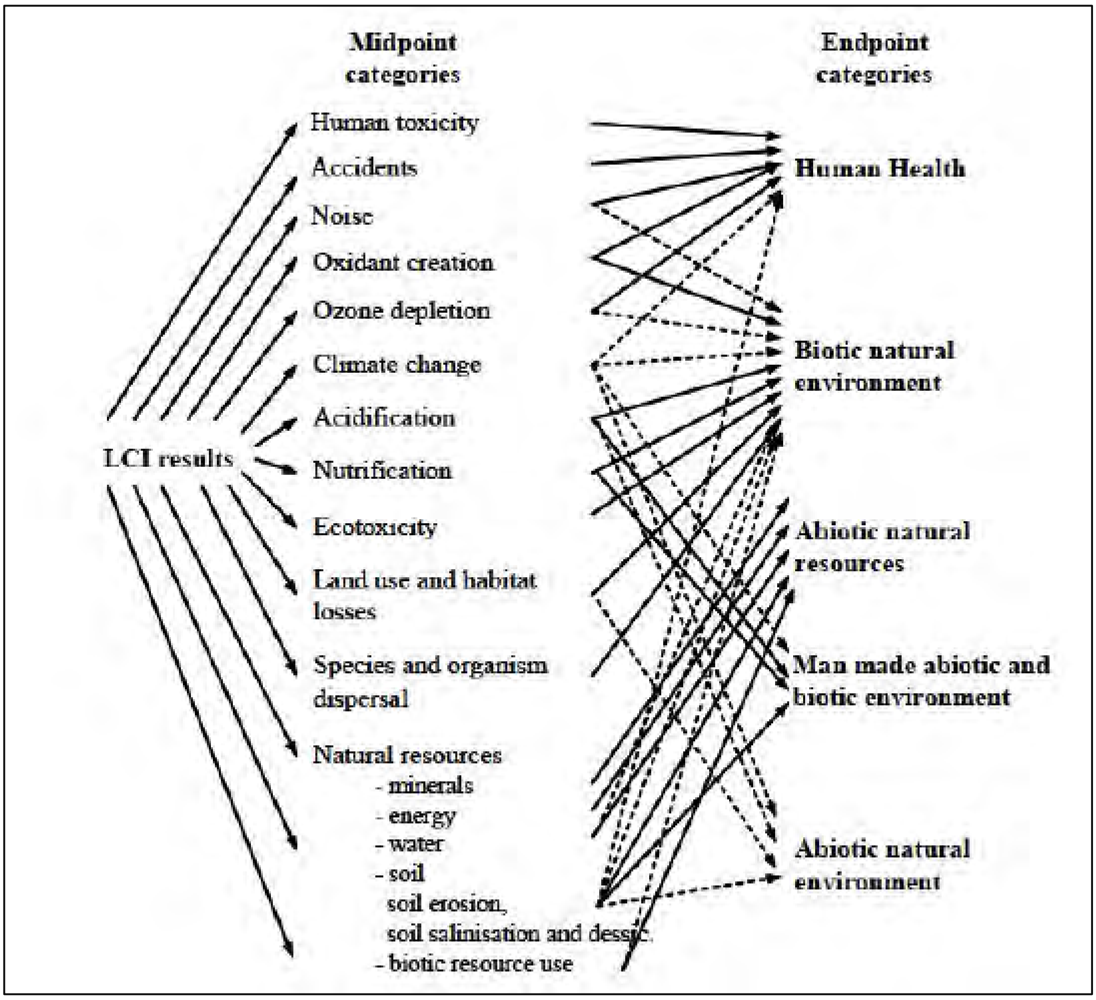


Figure 12 Relationship between LCI results and the cause effect chain (Jolliet et al., 2003)

To date, a number of standard impact categories and characterisation models have been incorporated into the LCA methodology. The impact categories can be categorized according to the ones pertaining to depletion, pollution and disturbance (White et al., 1994). Table 16 represents some of the LCIA midpoint impact categories as well as common impact assessment models which have been developed to calculate the characterisation factors for the different impact categories.

Table 16 Common midpoint LCIA categories and bases of some of the characterisation models (adapted from Guinée et al. (2001))

Type of Impact	Midpoint Impact Categories	Basis of characterisation model	Scale
Depletion	Abiotic resource depletion	Concentration reserves and rate of de-accumulation approach	Global
	Biotic resource depletion	Concentration reserves and rate of de-accumulation approach	Global
	Land-use	Occupation of a certain area of land during a certain time and transformation of a certain area of land.	Global, continental, regional, local
	Water depletion	Freshwater reserves and consumption rate	Global, continental, regional, local
Pollution	Climate Change	Model based on the definition of the global warming potential of different greenhouse gases	Global
	Ozone depletion	Model based on the definition of the ozone depletion potential of different greenhouse gases	Global
	Human toxicity	Based on multimedia fate, exposure and effect models	Global, continental, regional, local
	Eco-toxicity	Based on multimedia fate, exposure and effect models	Global, continental, regional, local
	Acidification	Based on fate and deposition of acidifying substances	Continental, regional, local
	Photo-oxidant formation	Based on trajectory models	Continental, regional, local

As can be seen from Table 16, characterisation models of human toxicity and eco-toxicity are based on multimedia fate, exposure and effect models. These characterisation models take into account the environmental persistence (fate), accumulation in the human food chain (exposure) as well as the toxicity (effect) of a chemical which can be derived from toxicity data on human beings and laboratory animals (Goedkoop et al., 2009). An example of a multimedia fate, exposure and effects model is the Uniform System for the Evaluation of Substances (USES-LCA), which is commonly used in the development of eco-toxicity and human toxicity characterisation factors (Huijbregts et al., 2000). In the solid waste management context, the multimedia fate, exposure and effect models have been criticized for not including certain elements specific to solid wastes and for not including the mobility of the elements (Notten, 2001; Heijungs et al., 2004). Furthermore studies have also revealed the inconsistences of equivalency factors of metals across various impact assessment models (Kunene 2014).This has been noted to possibly lead to the lack of assessment or the overestimation of the potential impact of a particular element on the environment (Notten, 2001; Heijungs et al., 2004). Hansen (2004) further note that the lack of consideration of local physical properties such as local water properties by the models renders the characterisation models limited.

With regards to abiotic and biotic resource usage, whilst some of the LCIA methods are based on concentration reserves and rate of de-accumulation approaches as indicated in Table 16, other methods also take into account the mining cost associated with extractions, the relatively higher energy required for extraction of metals or other substances from low-grade sources and the environmental impacts associated with the mining and processing of mineral resources (Schneider 2014; Klinglmair et al. 2014). Klinglmair et al. (2014) note however that relatively little regard is given to biotic resources and their renewal rates and Kunene (2014) criticizes conventional LCIA resource usage methods for their lack of accounting for the resources losses associated with the non-utilisation of solid waste.

In general, conventional LCIA categories have also been criticized for not encompassing aspects such as leachate generation and salinization of local water surfaces (Hansen, 2004). In a recent study where the LCA tool was used by Kunene (2014) to compare two base metal tailings waste management methods, limitations associated with the tool were found to be its lack of aqueous acidification and salinization impact categories as well as its non-accountability of temporal behaviour of emissions from solid wastes.

Addressing limitations associated with conventional LCA methods

With the motivation of addressing the limitations associated with conventional LCA methods, to date several researchers have sought to develop additional impact categories as well as new impact assessment methods. One example is the development of a water footprint impact assessment method by Pfister et al., (2009). The impact assessment method involves the incorporation of a consumptive water use impact indicator and a degradative water use impact indicator into a water footprint impact category. The consumptive water use indicator caters to assess the consumptive water use impacts of a region in relation to the scarcity and availability of water in that region as well as the scarcity and availability of water globally. Ridoutt and Pfister (2013) describe this approach to be superior to other approaches in that it does not combine water consumption from water scarce and water abundant regions as the former is associated with more potential for harm. As for the degradative water use indicator which describes a quality change in water used and released back to the same watershed, it is calculated by modelling emissions released to water separately using standard LCA methods at the endpoint level (Ridoutt and Pfister, 2013).

Another example is the development of a salinity impact category in LCA by Leske and Buckley 2004). Leske and Buckley (2004) developed an impact assessment model suited to generate equivalency factors to cater for a new salinity impact category for South African environmental life cycle assessments. This model used to generate the equivalency factors was developed by combining the effect factors and fate factors for the salt forming emissions into the various initial release compartments. The model developed also took into account the physical properties of South African waters which makes it applicable in the South African context. Table 17 represents the equivalency factors for the total salinity potential for different release compartments.

Table 17 Equivalency factors for total salinity potential for the different initial release compartments (Leske and Buckley, 2004).

Initial release compartment	Total salinity potential (kg total dissolved salts equivalent/kg)
Atmosphere	0.013
Surface Water	0.165
Natural Surfaces	0.031
Agricultural Surfaces	1.000

As was mentioned in the previous section, conventional LCIA impact indicators are deficient in that they do not cater to some of the environmental issues associated with solid waste management and that they lack some of the equivalency factors for the environmental interventions pertaining to solid waste management. Hermann et al. (2007) note that combining environmental performance indicators into a life cycle approach would address the some of the limitations associated with LCA whilst maintaining the standard procedure for data format and quality and standardisation that comes with LCA.

2.4.3 Risk based environmental performance approaches

Risk assessment has been around since the 1980s as a regulatory tool to assess and quantify the impact of point source releases (Murray and Claassen, 1999). Risk assessment aims to evaluate the likelihood that adverse effects may occur as a result of exposure to one or more stressors and forms part of a formal structured environmental performance process (Murray and Claassen, 1999). Risk based environmental performance indicators which are developed for risk assessment methods are based on the comparison of predicted environmental concentrations with levels believed to cause adverse environmental effects (Hansen 2004). Murray and Claassen (1999) note that since risk based indicators are based on a well-established logical structure they allow for results to be easily and comprehensively understood, documented and communicated. Simmonds et al. (1992) notes however some of the limitations of risk based approaches to include the accuracy of the models relating to the nature and magnitude of emissions in the environment, uncertainties in contaminant fate and transport models and the extrapolation of toxicity data from high-dose conditions to low-dose conditions.

An example of a risk-based performance indicator is the risk potential factor developed by Broadhurst and Petrie (2010) for ranking and scoring solid waste constituents on the basis of their hazard-forming potential and chemical behaviour under disposal conditions. The method calculates risk potential factors of waste constituents in order to rank their environmental significance (calculated according to equation 4). Broadhurst and Petrie (2010) note that this method of risk characterisation can be considered to be a vital and integral part of reliably quantifying the environmental life cycle impacts of solid mineral wastes. This risk potential factor can therefore be used to compliment and supplement life cycle assessment indicators for the potential environmental impact associated with the release of metals into local water sources.

$$RPF_i = \frac{(AC_i)^2}{ARC_i \times BC_i} \tag{4}$$

Where:

AC_i= available concentration (ppm)
ARC_i= environmentally accepted concentration (ppm)
BC_i = natural background concentration (ppm)

For metals, the environmentally acceptable concentration can be taken as the acceptable aquatic ecosystems metal exposure or the acceptable drinking water metal exposure according to established water quality regulations. Differences between the environmentally acceptable concentrations of drinking water and aquatic ecosystems (concentrations presented in Appendix B) exist due to the differences in safe exposures and differences in the aesthetic requirements (DWAF, 1996). The natural background concentration can be taken as the crustal abundance concentration (presented in Appendix B) of a metal whilst the available concentration represents the metal concentration that could leach into solution under certain physical conditions (Broadhurst, 2007). The available metal concentrations can be derived from the ARD and SCE tests described in Section 2.1.2.

2.5 Literature summary and synthesis

Literature has shown that there are various negative environmental implications of directly disposing untreated coal waste into slurry dams. The main implications being the risk of formation of ARD, spontaneous combustion, detrimental land usage, dust emissions, loss of resources (particularly mined coal and water), metal contamination of local groundwater sources and an increase in salinity in groundwater sources. The presence of pyritic sulfur in ultrafine coal waste has been linked to ARD generation, metal contamination of water sources, salinization of water sources and an increase in the risk of spontaneous combustion occurring. It has also been noted that due to the fine nature of ultrafine coal waste in particular, it is more prone to spontaneous combustion and presents an increased risk of dust emissions.

The technical feasibility of the two-stage flotation process has already been established with past laboratory testwork indicating that it is possible to recover as much as 89% of the coal from the coal ultrafines and remove as much as 50% of the sulfide sulfur, producing a non-acid forming waste stream. Previous studies have indicated a dependence of the process' technical efficiency on the reagent types and reagent dosages, with certain reagent types managing to recover substantially more product than others. More specifically, Nalflote 9858, a synthetic coal collector, has demonstrated the ability to achieve significant coal recoveries at lower dosages compared to the oily collectors such as kerosene and oleic acid whilst potassium amyl xanthate, a sulfide collector, demonstrated the ability to achieve higher recoveries compared to other xanthate salts for various ultrafine coal samples.

The economic viability of the two-stage flotation process demonstrated a dependence of process' NPV on the coal revenue and operational costs. Furthermore it has been shown that the fictitious plant model is associated with the consumption of approximately 1.2 MW of electrical energy due to the operation of the flotation cells and dewatering units. The sulfide-lean and the sulfide-rich products have also been shown that they can be utilised in a number of ways with some of the suggested uses still in the early stages of development.

Environmental implications of the two stage flotation process so far have been limited to ARD generating potential, with little consideration having been given to broader environmental burdens. The potential environmental implications of the process have been highlighted as the release of the flotation reagents into local water sources, the consumption of non-renewable resources, the release of emissions into the air due the consumption of electricity and the transformation of land due to the disposal of the sulfide-lean tailings. Literature has shown that the xanthate salts (potassium amyl xanthate, sodium isobutyl xanthate and sodium ethyl xanthate) exhibit the most toxic properties and that the release of xanthate salts into water sources can be harmful to aquatic life. Comparison of the toxicity properties of the coal flotation collectors revealed oleic acid to having less toxic properties than kerosene. Unfortunately, due to lack of information on the toxicity properties of Nalflote, it could not be compared to the other coal flotation collectors. MIBC has been shown to be associated with the least toxic properties amongst all the flotation reagents. Lastly it has been highlighted that the production of the xanthate reagents is associated with the release of a considerable amount of SO_2 to the atmosphere.

The life cycle assessment tool has been described as a tool used to evaluate the environmental aspects and potential impacts associated with a product or process by collating an inventory of inputs and outputs of a product system and evaluating the potential environmental impacts associated with those inputs and outputs. Literature commends it for its approach to the compilation of input and output data over a product/process life cycle and its avoidance of environmental burden shifting however its conventional LCIA impact categories have been criticized for not encompassing some of the crucial aspects associated with solid waste management such as salinization of local water sources and leachate generation. New impact categories and impact assessment methods such as the total salinity potential have recently been developed to curb some of these limitations. Furthermore researchers have suggested combining the LCA approach with environmental performance indicators so as to address some of its limitations whilst maintaining the standard procedure for data format, quality and standardisation that comes with it. Risk based approaches to the environmental performance analyses have been described as being formal and structured which makes it easy to assess some of the environmental burdens associated with solid waste management practises. The drawbacks however associated with risk based approaches have been highlighted as limitations in the models relating to the nature and magnitude of emissions in the environment as well as the extrapolation of toxicity data from high-dose conditions to low-dose conditions.

CHAPTER 3

METHODOLOGY

This chapter presents the methodology used to address the research questions and the study objectives. The main stages of the methodology employed include the inventory analysis stage, the impact assessment stage and the interpretation stage. The inventory analysis stage consisted of scenario development, compilation of case study data and analysis of input and output data. The impact assessment stage involved the classification of impact categories and the calculation of category indicators. The classification of impact categories was characterized by the selection of impact categories relevant to the research study and the assignment of input and outputs to the impact categories. Finally the interpretation stage involved the comparison of the coal waste management scenarios, analysis of contributing processes, sensitivity analyses and evaluation of the study limitations. Figure 13 summarizes the structure of the methodology.

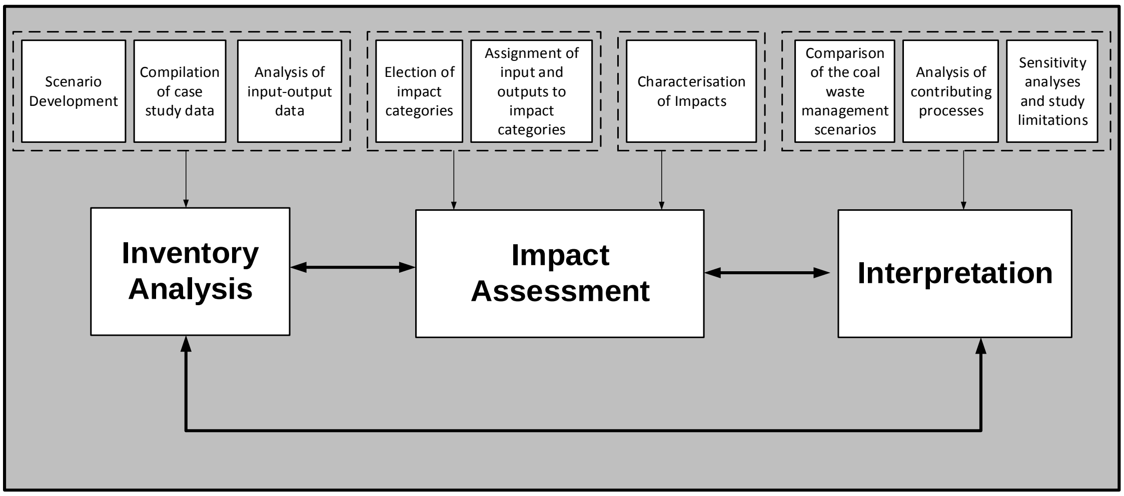


Figure 13 Overview structure of the methodology

3.1 Inventory analysis

3.1.1 Scenario development

Two scenarios were developed in order to evaluate the environmental implications of the two-stage flotation process for the pre-disposal treatment of ultrafine coal waste. The disposal of untreated coal ultrafines into slurry dams (a method employed in some coal beneficiation plants in South Africa) (described in Section 1.1.3) was established as the first scenario (termed the base case). In this method, coal ultrafines generated from the coal beneficiation process are fed into a thickener, where excess water is recovered before the underflow slurry is disposed of in a slurry dam. The overflow water from the thickener is then recycled back to the beneficiation plant whilst the remaining water in the slurry dam is lost either through seepage, evaporation or entrainment. As was mentioned in Chapter 2, the exiting slurry might be disposed of still containing 60-85% water (depending on the thickener's efficiency). There is no set value or approximation of the solids concentration of the feed into thickener as this would vary depending on the classification cyclone however the solids concentration in the thickener feed stream is likely to be less than that in the exit stream.

In the second scenario, which is the two-stage flotation process, the coal ultrafines are first passed through the first froth flotation stage, where a concentrate rich in coal and with a low ash content is recovered and dewatered by means of a filter press. The tailings from the first flotation stage are then fed to the second froth flotation stage, where a rich-sulfide concentrate product is recovered and dewatered also by means of a filter press. Lastly, the tailings from the second froth flotation stage are fed into a thickener and following from that to a filter press before being disposed of. The excess water recovered from all the dewatering processes is then recycled back to the coal beneficiation plant. The functional unit used in this project was 864 kilotons tonnes per annum of dry coal ultrafine waste. This was based on the basis flowrate of the conceptual plant model developed by Jera (2013) which was 100 tons per hour and on the assumption that the plant would be operating 24 hours a day and 360 days in a year, with 5 days for maintenance and shutdown.

The recovered coal product and sulfide-rich stream are to be sold to an electricity production plant and a sulfuric acid producing plant respectively. Furthermore it is assumed that the sulfuric acid producing company the sulfide-rich product will be sold to will have an already existing upgrading unit so as to upgrade the total sulfur concentration in the product before using it. Based on the developed fictitious plant model (described in Section 2.2.3), it is assumed that the filter press manages to achieve the 10% minimum allowed moisture content in product coal. Lastly, it was also assumed that no other coal waste management scenario currently in practise could effectively mitigate the formation of ARD which is why the two-stage flotation process was the only scenario considered as an alternative to the base case. Figure 14 illustrates the system boundary of the environmental performance assessment for both of the waste management scenarios and Figure 15 represents the detailed block flow diagram of the two-stage flotation process on which the mass balance calculations were applied onto.

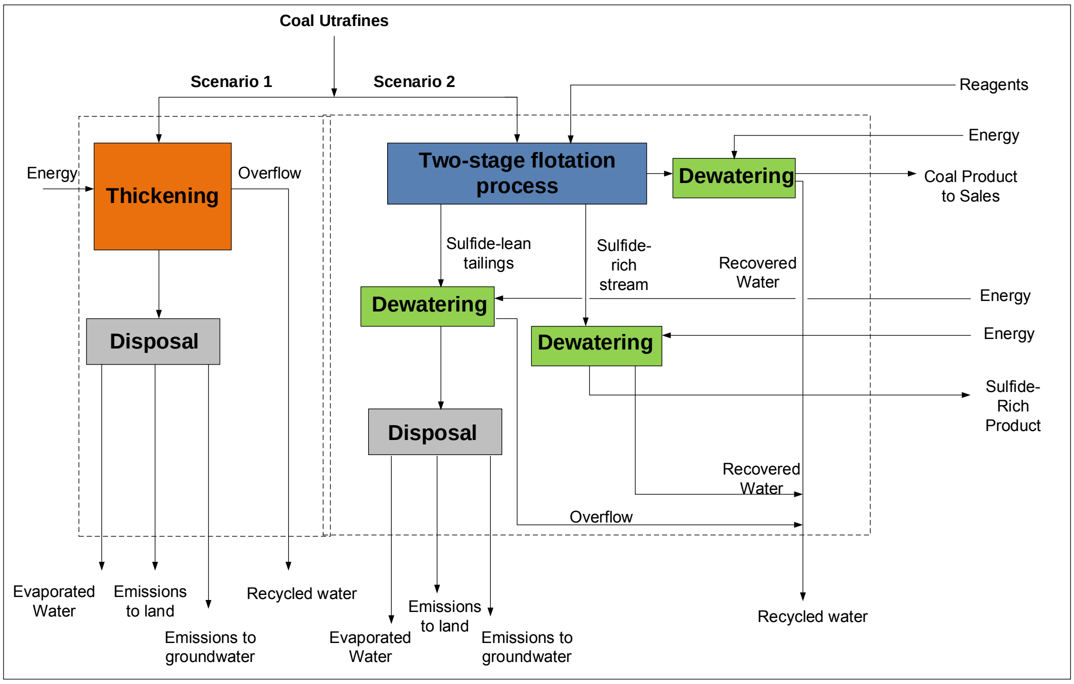


Figure 14 System Boundaries for scenario 1 and scenario 2

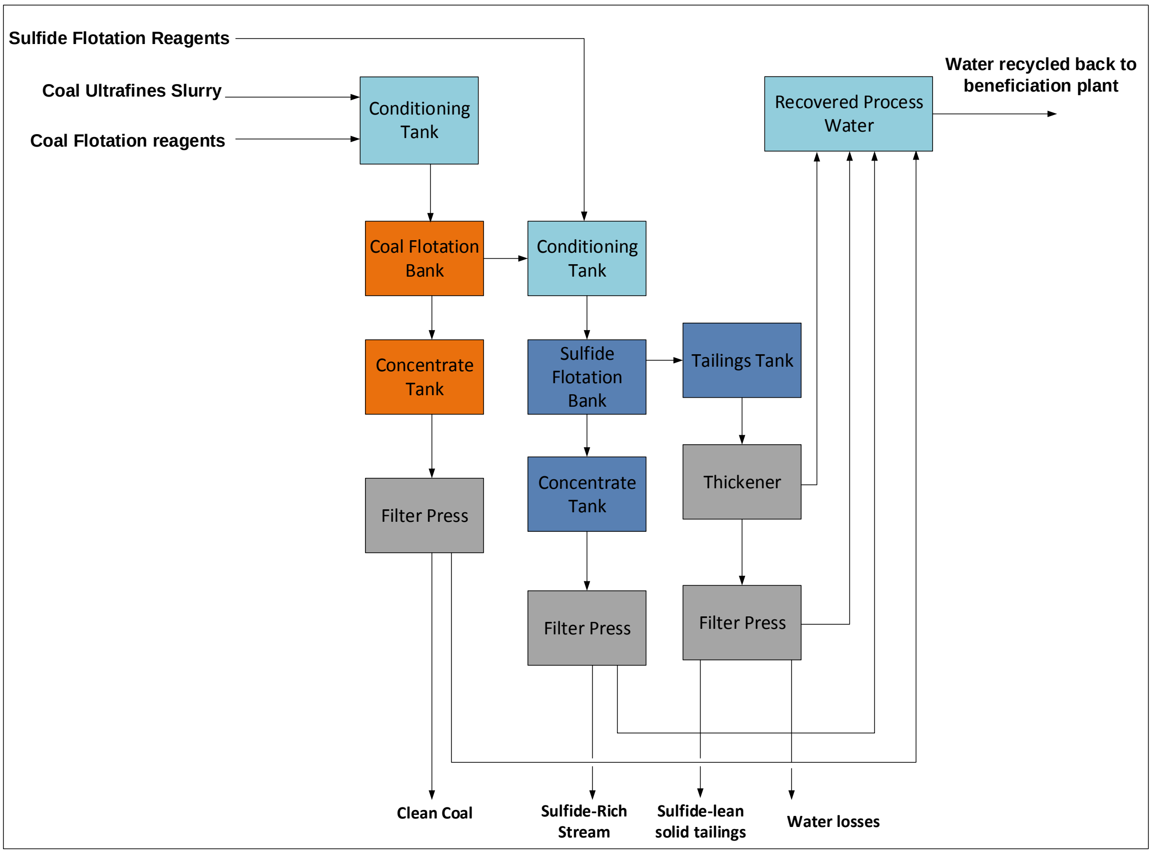


Figure 15 Detailed block flow diagram of the conceptual plant for the two-stage flotation process

The solid, liquid and gaseous emissions as a result of the utilization of the recovered products were not taken into account since it was assumed that the products will be sold to already existing production companies. The associated impacts as a result of the utilization of the products would therefore have occurred regardless, with companies utilizing raw materials from the ground.

3.1.2 Case study description

The selected case study was based on a coal sample from the Waterberg region and was selected based on the availability of comprehensive laboratory test work information. Table 18 represents the mass balance across the two-stage flotation process obtained from the laboratory testwork on the sample and Table 19 represents the reagent dosages used during the experiments. Figure 16 serves to illustrate the deportment of ash and sulfur across the two-stage flotation process. Using ARD prediction tests, the laboratory test work results also showed that the final disposed sulfide-lean stream is non-acid generating. Table 20 tabulates the net acid producing potentials of the major streams obtained from the laboratory testwork. Comparing this case study to other case studies, this case study performs relatively poorly in terms of coal yield and sulfide-sulfur recovery. Whilst previous studies have shown that it is possible to achieve a coal yield of 89% and a sulfide sulfur yield of 50%, this case study only manages to achieve a coal yield of 30.2% and sulfide-sulfur yield of 25%.

Table 18 Mass balance across the two-stage flotation process (derived from Iroala (2014))

Stream name	Feed	Coal Concentrate	First Stage Tailings	Sulfide Rich Stream	Sulfide Lean
Total Solids Deportment from Feed (%)	-	30.2	69.8	2	68
Stream Compositions (%)					
Sulfide Sulfur	1.07	1.14	1.04	12.8	0.69
Sulfates	0.0280	0.03	0.03	0.36	0.02
Organic Sulfur	0.822	1.50	0.53	15.1	0.10
Total Sulfur	1.92	2.66	1.60	28.3	0.81
Ash	49.1	24.3	59.8	49.1	62.29
Fixed Carbon and Volatile matter	49.0	73.0	38.6	22.6	36.90

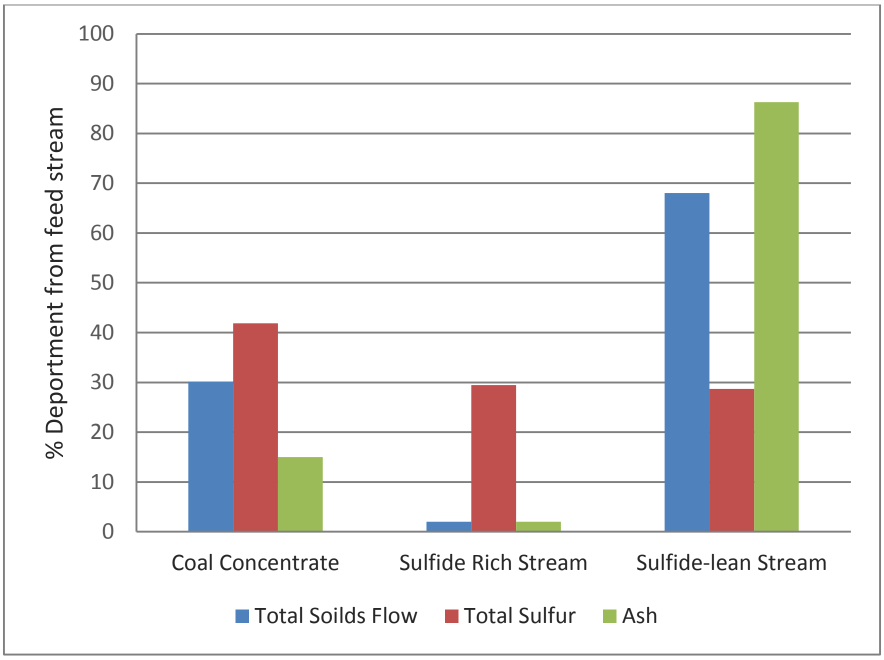


Figure 16 Deportment of ash and sulfur across the two-stage flotation process

Table 19 Reagent dosages for the two-stage flotation process (derived from Iroala (2014))

Reagent		Dosage (kg/t)
1st stage- Coal Flotation	Nalflote 9858	1.4
	MIBC	0.11
2nd stage - Sulfide flotation	Sodium isobutyl xanthate	2.33
	Methyl isobutyl carbinol	0.28
	Dextrin	0.93

Table 20 ARD prediction tests

Stream	NAG pH	NAG $_{pH\ 7}$ (H_2SO_4 kg/t)	NAPP (H_2SO_4 kg/t)	Classification
Feed Stream	2.04	0.6	52	Acid forming
Sulfide-rich Stream	1.40	12.1	532	Acid forming
Sulfide-lean Stream	0.65	0.5	-101	Non-acid forming

The concentrations and percent mobilities of potassium and calcium were derived from Sanyika and Ngcobo (2014) who determined these under NAG test conditions using ICP-OES analysis (inductively coupled plasma atomic emission spectroscopy). Furthermore, additional experimental analyses were conducted on the feed stream and sulfide-lean stream leachate solutions generated under NAG test conditions (described in Chapter 2) using ICP-MS (inductively coupled plasma mass spectrometry) technology to determine the leachable metal concentrations in the disposed streams (refer to Appendix B for detailed calculations). ICP-MS is a type of mass spectrometry which is capable of detecting metals at concentrations as low as one part per quadrillion. The analysed metals include aluminium, iron, titanium, zinc, chromium, manganese, cobalt, nickel, copper, vanadium, arsenic, barium, lead, uranium and were based on the identified trace elements in coal presented in Chapter 2. Mercury, selenium and antimony could not be analysed due to their complex nature. Table 21 represents the available concentrations obtained for the two streams.

Table 21 Available metal concentrations in the feed stream and in the sulfide-lean stream (Harrison et al., 2015)

Element concentration	Unit	Feed concentration	Desulfurized tailings concentration
Aluminium	ppm	182	500
Iron	ppm	1481	1237
Calcium	ppm	27889	472000
Potassium	ppm	240	7622
Titanium	ppb	95438	84563
Zinc	ppb	38600	12363
Chromium	ppb	1466	1581
Manganese	ppb	9681	22488
Cobalt	ppb	291	516
Nickel	ppb	1583	1531
Copper	ppb	2238	2063
Vanadium	ppb	2789	2299
Arsenic	ppb	1451	1554
Barium	ppb	2810	2466
Lead	ppb	1228	1141
Uranium	ppb	117	141

3.1.3 Compilation of input and output data

Input and output datasets were divided into foreground data, which is the input and output flows directly linked to the process, and background data, which is the input and output data associated with the secondary processes. For the foreground data, the flotation reagents included in the inventory were SIBX (sodium isobutyl xanthate) and oleic acid. Nalflote 9858 (which is the coal collector used in the case study) was not used due to lack of information with regards to its chemical compositions and production inputs and outputs. This is why oleic acid, another commonly used coal flotation reagent (Refer to Section 2.2) was used as a proxy for Nalflote 9858. The rest of the reagents such as the frother and the depressant were omitted from the inventory based on the assumption that they would have negligible impact on the system due to their relatively low dosages and toxicities compared to the collector reagents. Table 22 presents a summary of the bases of calculations and assumptions made during the inventory compilation.

Table 22 Summary of the bases of calculations and assumptions made during inventory compilation of the foreground process

Inventory Elements	Comments	Data Source(s)
Energy	• The primary energy source was taken to be electricity, which in South Africa is mostly produced from coal. • The electricity consumptions for individual units were calculated using engineering design principles. (Refer to Appendix A for detailed calculations)	(Walas, 1990; Perry and Green, 1999; Jera, 2013)
Materials	• It was assumed that the tailings stream coming from upstream contains 10% solids based on the conceptual plant model. • SIBX and oleic acid reagent dosages were derived from the laboratory testwork data.	(Jera, 2013; Iroala, 2014)
Emissions to Water	• The reagent affinity to material was derived from literature and applied to the mass balance to obtain the residual xanthate and oleic acid left in the sulfide-lean tailings (Appendix A). • The available concentrations of salt forming anions (potassium and calcium) were based on in-house laboratory testwork. • The available concentrations of major and trace metals were obtained from the ICP-MS analysis of leachate solutions generating under NAG tests conditions. Refer to the Appendix A for the detailed protocol. • The acid emitted to water was based on the NAPP obtained from the laboratory testwork data.	(Gebhardt et al., 1985; Sanyika and Ngcobo, 2014; Iroala, 2014)
Emissions to Land	• The percentage of feed that comprises the final tailings streams was applied to the mass balance for both scenarios	(Iroala, 2014)
Resources	• The water recovered from the dewatering processes was assumed to be recycled back to the beneficiation plant. • The remaining water was assumed to be lost through evaporation, seepage and entrainment. • The unrecovered coal was taken as the remaining carbon and volatile matter left in the final disposed waste.	(Reddick, 2006; Bleiwas, 2012)
Product	• The percentage recovered coal from the laboratory testwork information was applied to the mass balance.	(Iroala, 2014)

Three main processes were considered for the background datasets, these include electricity production, xanthate production and oleic acid production. The datasets for the energy production process (South African electricity mix production) and for oleic acid production (fatty acid production process) were taken from the Ecoinvent (version 2.2) database found in the SimaPro software. The background datasets for xanthate production were taken from an investigation conducted by Kunene (2014) on the Life Cycle Assessment of the production of xanthate salts. The water losses due to the background processes as well as the transport inputs and outputs were not taken into account due to the minimal availability of information in the databases. Figure 17 and Figure 18 provide a graphic representation of the input and output streams for the base case and the two-stage flotation process respectively.

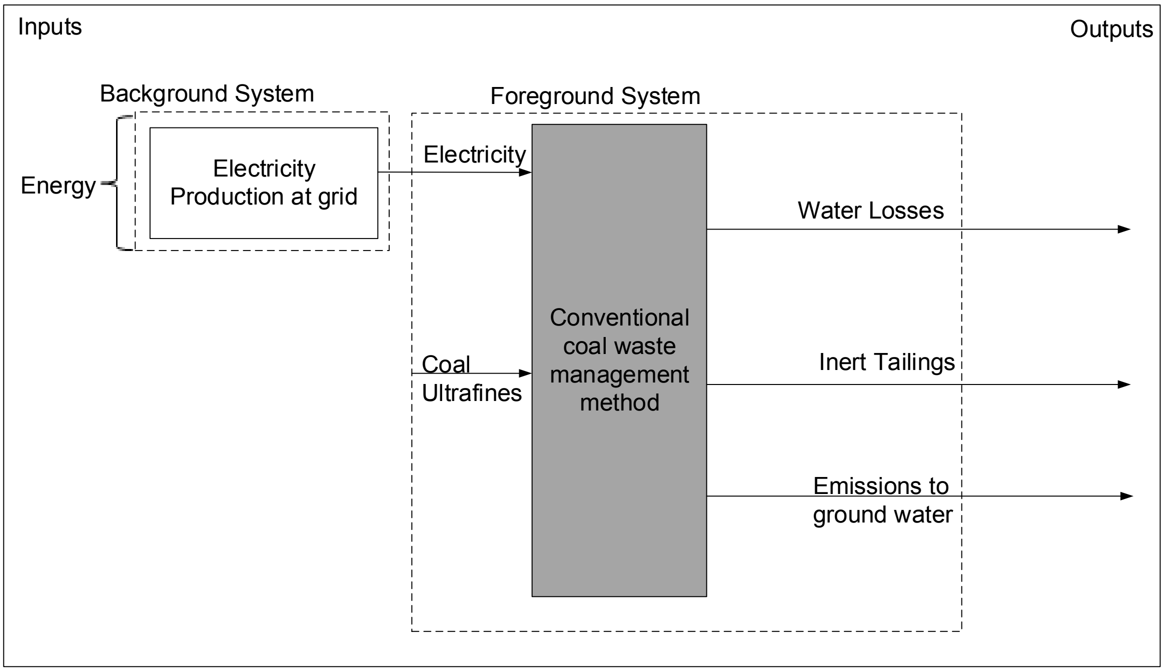


Figure 17 Summary of inventory elements of the base case in terms of inputs and outputs

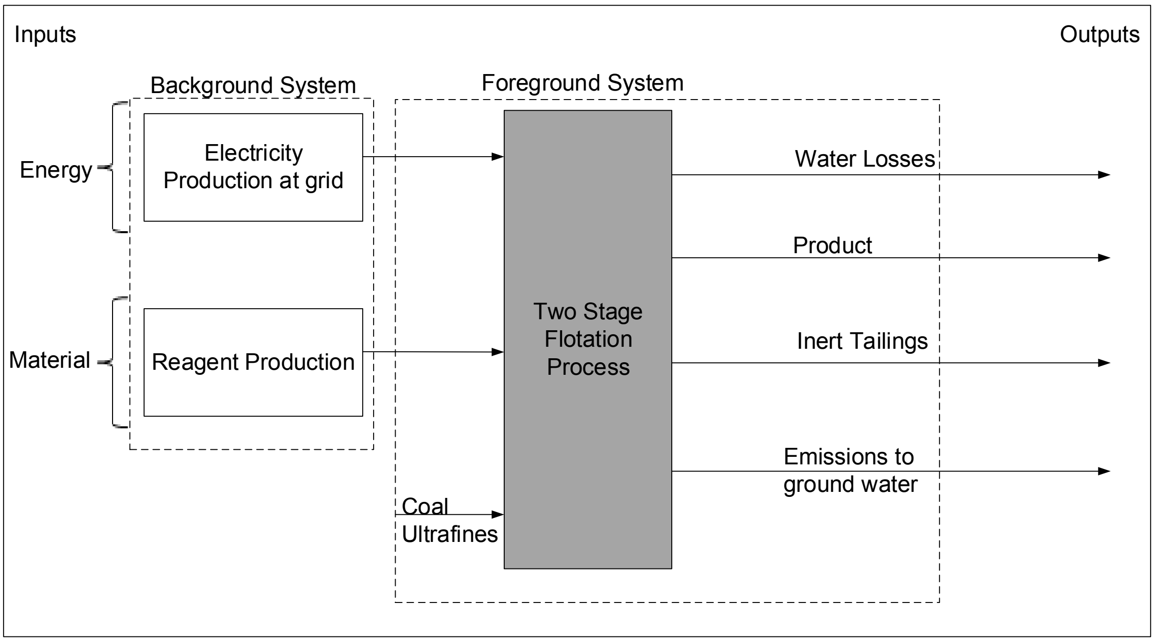


Figure 18 Summary of Inventory Elements of two-stage flotation process in terms of inputs and outputs

3.2 Impact assessment

3.2.2 Selection of impact categories and classification

Following from the inventory analysis stage, the inventory data was translated into meaningful environmental indicators. Relevant LCA impact categories were selected and supplemented by water-related risks to cater for the release of acids and metals into local water resources. The selected LCA impact categories include fossil fuel depletion, climate change, terrestrial acidification, eco-toxicity, salinity and water footprint whilst the water quality risks include aquatic water pollution risk, drinking water quality risk and aqueous acidification. The human toxicity impact category was not evaluated due to the unavailability of characterisation factors for aluminium, iron, sulfuric acid and the flotation reagents.

Selection of the impact categories was based on the potential environmental implications of both coal waste management methods. Table 23 summarizes the reasons for the selection of the impact categories. The selected impact categories were grouped into four main impact categories. These include atmospheric related impacts (as a result of the emissions of gases from background processes), water quality impacts (as a result of the seepage of solid waste constituents and flotation reagents into local water sources) and resource related impacts (as a result of resource usage and recovery).

Table 23 Reasons for the selection of impact categories

	Impact Category	Reason for selection
Atmospheric related impacts	Climate change	To evaluate the impacts associated with greenhouse and acidifying gases.
	Terrestrial acidification	
Resource related impacts	Fossil fuel depletion	To evaluate the impacts associated with the consumption and recovery of non-renewable resources.
	Natural land transformation	To evaluate the impact associated with the disposal of inert solid waste to the land.
	Consumptive Water Footprint	To evaluate the impact associated with the consumption of water by both waste management scenarios.
Water quality impacts	Freshwater ecotoxicity	To assess the impacts associated with the emissions of toxic substances to water sources.
	Salinity	To evaluate the impact associated with the emissions of salt forming cations and anions (chlorides, sulfates, carbonates, potassium and calcium) to local water sources.
	Aquatic water pollution risk	The three impact categories were selected as a form of additional analyses following from the freshwater eco-toxicity impact category in order to evaluate the relative risks associated with the emission of metals and acid to local water sources.
	Drinking water quality risk	
	Aqueous acidification	

Following from the impact category selection, the inventory elements were specifically linked to each of the selected impact categories. Figure 19 is a schematic of how the input and output data of the processes was linked to the inventory element categories and how the inventory element categories were linked to the impact categories.

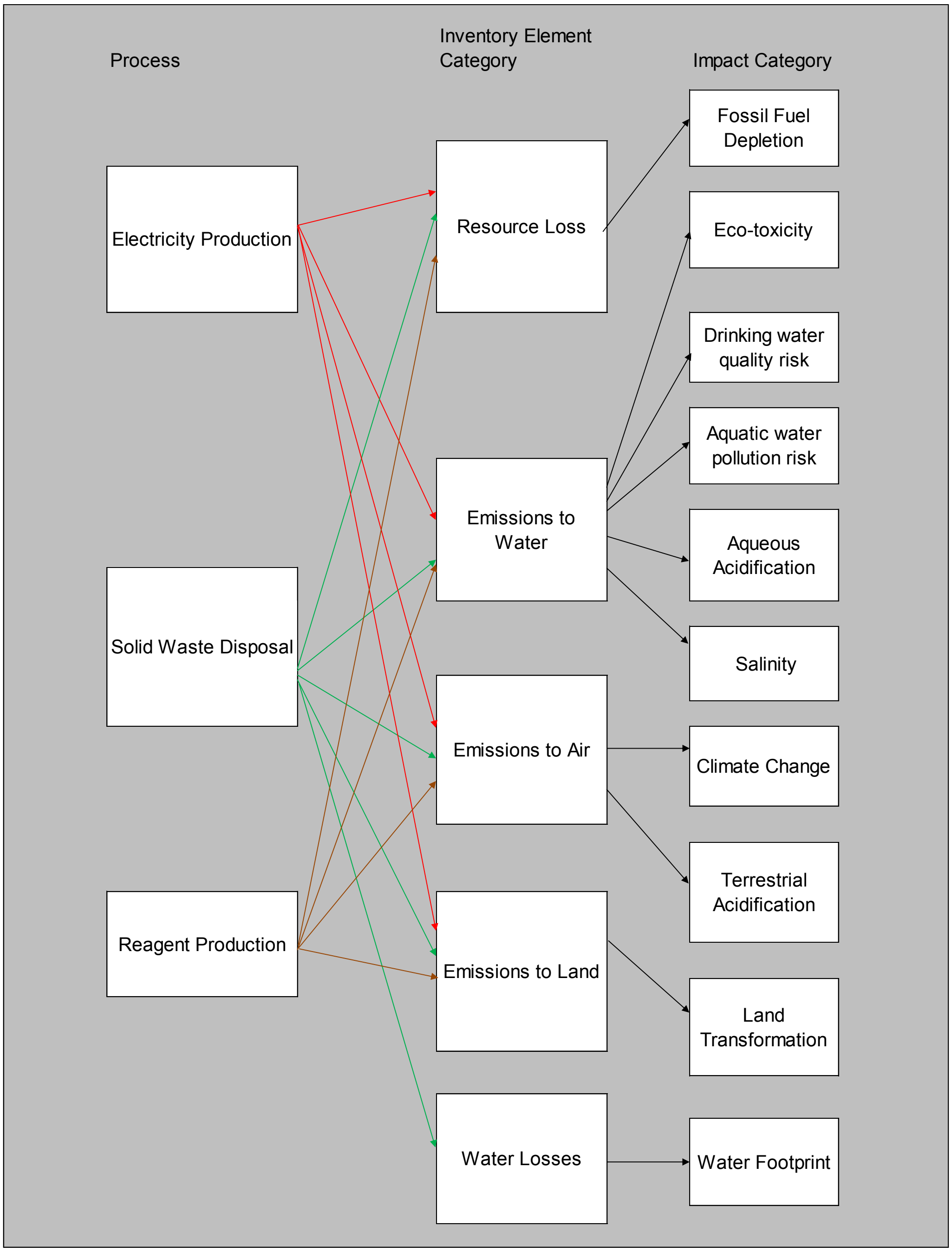


Figure 19 Inventory elements and the associated environmental impact categories

3.2.3 Characterisation factors and impact indicators

Table 24 represents the characterisation factors, impact indicator results and characterisation factor sources used in this research study. Climate change, terrestrial acidification, fossil fuel depletion, ecotoxicity and land transformation characterisation factors were taken from Goedkoop et al., (2009) whilst characterisation factors for water footprint and total salinity potential were taken from Pfister et al., (2009) and Leske and Buckley (2004) respectively. Since fossil fuel depletion, in the strict sense, is based on the extraction of fossil fuels and does not account for the possibility of fossil fuel recovery from waste, the unrecovered coal (the coal contained in the final disposed stream) was accounted for as an “extraction”. It should be noted that the sulfide-sulfur recovered or lost was not included in the fossil fuel depletion category as it is not a fossil fuel. Furthermore it could also not be evaluated under the abiotic depletion category (as a mineral) since the characterisation factors available are only for a considerably pure sulfide-sulfur resource. The aquatic and drinking water quality risk impacts as well as aqueous acidification impact indicators of both of the final disposed streams were assessed using risk potential factors.

Table 24 Summary of the characterisation factors and indicator results

Impact Category	Characterisation Factor	Indicator Result	Source of characterisation factor
Climate Change	Global Warming Potential for each greenhouse gas (kg CO2/ kg emission)	kg CO_2 equivalents/year	Goedkoop et al. (2009)
Terrestrial Acidification	Acidification Potential (AP) for each emission to air (kg SO2/kg emission)	kg SO_2 equivalents/year	Goedkoop et al. (2009)
Fossil Fuel Depletion	Fossil Fuel Depletion Potential for each extraction of fossil fuels	kg of oil equivalents/year	Goedkoop et al. (2009)
Water Footprint	Stress weighted Global consumptive water footprint	m^3/year	Pfister et al. (2009)
Land Use	Characterization factors for land occupation/land transformation	m^2/year	Goedkoop et al. (2009)
Salinity	Total Salinity Potential for a specific sub-category (in this case groundwater)	Total Dissolved Salts kg equivalent/year	Leske and Buckley (2003)
Eco-Toxicity	Eco-toxicity potentials for a specific sub-category (in this case it is freshwater)	kg 1,4 Dichlorobenzene equivalent/year	Goedkoop et al. (2009)
Aquatic water pollution risk	Risk Potential Factor	Dimensionless Risk Potential Factor	Broadhurst and Petrie (2010), DWAF (1996)
Drinking water quality risk	Risk Potential Factor	Dimensionless Risk Potential Factor	Broadhurst and Petrie (2010), DWAF (1996)
Aqueous Acidification	Risk Potential Factor	Dimensionless Risk Potential Factor	Broadhurst and Petrie (2010), ANZEEC (2000), DWAF (1996)

Eco-toxicity characterisation factors were available for aluminium, zinc, chromium, manganese, cobalt, nickel, copper, vanadium, arsenic, barium, lead, oleic acid, sodium isobutyl xanthate and sulfuric acid. Unfortunately the characterisation factor for iron, uranium and titanium were unavailable. Most of the impact indicators were calculated by multiplying the characterisation factor with the inventory result however the risk potential factors were calculated using equation 4 (presented in Section 2.4.2) developed by Broadhurst and Petrie (2010) to obtain a total risk potential score. In calculating the risk potential factors, the environmentally acceptable concentration (refer to equation 4) for aquatic water pollution risk was taken as the acceptable aquatic ecosystems exposure whilst that of drinking water quality risk was taken as the acceptable drinking water exposure according to DWAF (1996). The natural background concentration was taken as the crustal abundance concentration of each metal (Broadhurst, 2007). This same method was applied when evaluating the aqueous acidification impact indicator, a risk potential factor (according to equation 4) based on acid concentrations in the final disposed waste was evaluated. The environmentally acceptable concentration was taken as the acceptable aquatic pH according to ANZECC (2000) and the background concentration was taken as the natural pH of South African waters according to DWAF guidelines (1996). The characterisation factors, water quality guidelines and crustal abundance concentrations used are detailed in Appendix B.

3.3 Interpretation

Interpretation of the results involved summarizing results from both the inventory analysis and the impact assessment stages. Process unit contributions to each impact indicator were demonstrated for each scenario in order to identify the processes which contribute most to the environmental burdens. The major processes considered were the flotation reagents production process, the electricity production process and the solid waste disposal process. The impact indicators were calculated separately for the production of electricity (electricity consumed in the process), production of flotation reagents and for the disposal of the solid waste and their percentage contribution to the overall impact indicator assessed.

Sensitivity analyses were employed at this stage to investigate how the impact indicators are affected by the variation of input parameters and to demonstrate how flexible the environmental performance of the two-stage flotation process is. The parameters that were varied include electricity usage, reagent dosages, coal yield and available iron and aluminium concentrations in the waste streams. Table 25 presents the variation ranges of the parameters selected for the sensitivity analyses.

Table 25 Parameter variation ranges for the sensitivity analyses

Parameter	Variation Range	Deviation from base (%)
Electricity consumption per 100 t/hr coal ultrafines (MWh)	0.5-2.5	-72 - 42
Coal Yield (%)	15-95	-50 - 215
SIBX reagent dosage (kg/t)	1.45-13	-38 - 460
Oleic acid dosage (kg/t)	1-9	-28 - 540
Available iron concentration (ppm)	1000-5000	-20 - 300
Available aluminium concentration (ppm)	100-500	-50 - 160

Valuation, which refers to normalisation and weighting of the impact categories based on economic, global, technical and base line targets was not conducted due to the uncertainties associated with the normalisation of impact categories and lack of consensus on the use of weighting in LCA (Refer to Chapter 2). Furthermore, more than one impact assessment method was used in this research study which makes it difficult to select a normalisation method, which is usually developed for a specific impact assessment method.

CHAPTER 4

CASE STUDY RESULTS

This chapter presents the results obtained in the research study. The chapter begins by presenting the energy inputs of the various process stages of the two coal waste management scenarios. Following from this, the chapter presents the results and analysis of the life cycle inventory stage which includes the foreground and background datasets of the two scenarios. Finally the results of the life cycle impact assessment will be presented.

4.1 Inventory analysis

This section presents the results of the analysis, which involved conducting detailed mass and energy balances of the two coal waste management systems. The compiled input and output datasets are presented in two categories, these include the foreground data and the background data. The foreground data is the input and output information directly related to the coal waste management processes and the background data is the input and output information associated with the production of energy or materials used in the processes. Following from this, the life cycle inventory data is presented to summarize all the inputs and outputs related to the systems. The detailed mass balance calculations for each of the coal waste management scenarios are presented in Appendix A.

4.1.1 Foreground data

As was mentioned in Chapter 3, the energy input at the different process stages for the two coal waste management scenarios was calculated using literature sources and engineering heuristics (detailed calculations shown in Appendix A). Table 26 presents the electricity consumptions of the process units and Figure 20 illustrates how the energy consumption varies along the process.

Table 26 Energy Inputs at different process stages

Process Stage	Energy Inputs (KWh) per 100 tons of ultrafine coal waste	
	Two-stage Flotation Process	Base Case
First Stage Coal Flotation	473	
First Stage Concentrate Filtration	110	
Second Stage Sulfide Flotation unit	473	
Second Stage filtration concentrate	30	
Tailings Thickening	10	10
Tailings filtration	90	
Compressor	461	
Process Water Pumping	110	20.4
Fresh Water Pumping	4	0
Total Electricity Requirement	1760	30.4

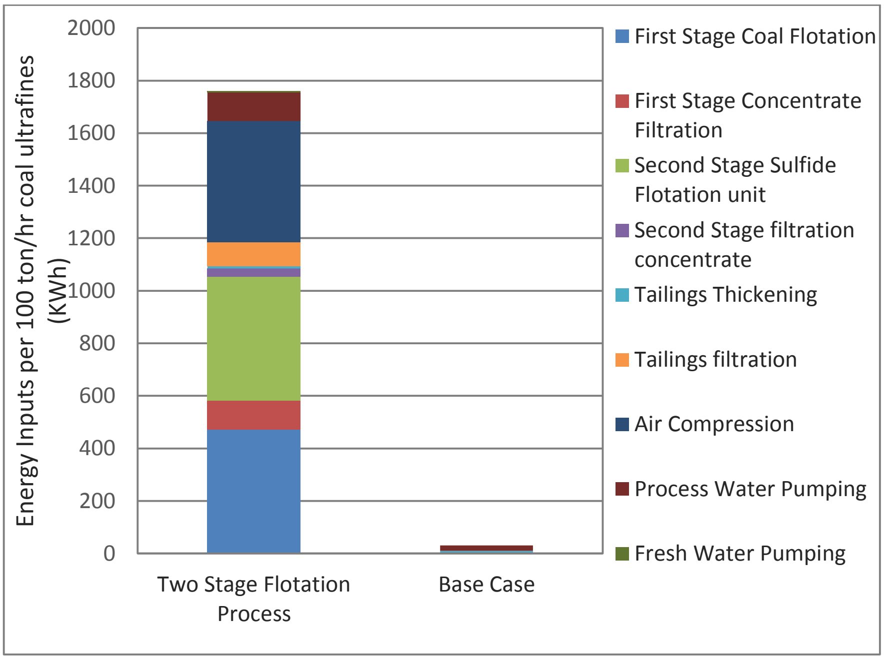


Figure 20 Energy inputs per 100 ton/hr feed coal ultrafines

As can be seen from Figure 20, the major contributors to the energy requirement for the two-stage flotation process are the flotation units as well as the compression unit (responsible for compressing air that is fed into the flotation units), which together contribute about 80% to the energy load. The remaining energy requirement is for the dewatering units and the process pumps. With regards to the base case, the electricity requirement is only divided between the thickener and the process water pumps of which both require infinitesimal energy compared to the two-stage flotation process. Comparing the energy requirement calculated for the flotation units to the Marsden energy model for flotation (modelled for flotation of base metal sulfides), the calculated energy for the flotation units (excluding the regrinding energy) is approximately 35% higher than that of the Marsden energy model (calculations detailed in Appendix A).

The foreground data for the two-stage flotation process and the base case is summarized in Table 27. The foreground data is presented in categories that comprise of energy, materials, emissions to water, emissions to land, loss of resources, recycled resources and recovered product.

As can be seen from Table 27 and as was observed from Figure 20, compared to the two-stage flotation process, the energy input of the base case is infinitesimally small, this is because of the various process stages of the two-stage flotation process that require energy. With regards to the emissions to water and emissions to land, the base case scenario has 94% more emissions to water compared to the two-stage flotation process, with the release of sulfuric acid into local water sources forming the biggest portion of the emissions. It can also be seen from Table 27 that manganese and cobalt are more in the sulfide-lean tailings than the feed stream which defies mass balance laws hence this should be due to experimental anomalies. As for the emissions to land, the two-stage flotation scenario exhibits 32% less emissions to land relative to the base case process due to the reduced amount of solid tailings to disposal.

Table 27 Input and output data of the foreground process

Description			Units	Flows per 100 tonnes coal ultrafines	
				Base case	Two-stage flotation process
Inputs	Energy	Electricity	MWh	0.01	1.76
	Materials	Solids in slurry	tons	100	100
		Water in slurry	tons	900	900
		Oleic Acid	tons	-	0.279
		Xanthate	tons	-	0.163
Outputs	Emissions to Water	Acid(H+)	kg	5200	-
		Oleic Acid	kg	-	8.37
		Xanthate	kg	-	11.4
		Sulfate	kg	500	13.1
		Aluminium	kg	18.5	13.1
		Iron	kg	148.1	83.9
		Calcium	kg	900	373
		Potassium	kg	400	13.6
		Zinc	kg	3.860	0.838
		Chromium	kg	0.147	0.107
		Manganese	kg	0.968	1.525
		Cobalt	kg	0.029	0.035
		Nickel	kg	0.158	0.104
		Copper	kg	0.224	0.140
		Vanadium	kg	0.279	0.156
		Arsenic	kg	0.145	0.105
		Barium	kg	0.281	0.167
		Lead	kg	0.123	0.077
		Uranium	kg	0.012	0.010
	Emissions to land	Total Solid Waste	tons	100.0	67.8
	Loss of Resources	Water losses due to evaporation	tons	60.0	8.3
		Water Losses Seepage	tons	15.0	2.07
		Water losses due to entrainment	tons	75.0	10.37
		Unrecovered Coal	tons	47.0	26.5
		Gangue Material	tons	53.0	41.3
	Recycled Resources	Water	tons	750.0	879
	Recovered Products	Recovered Coal	tons	-	30.2
		Sulfide-Rich Product	tons	-	2.0

4.1.2 Background datasets

As was mentioned in Chapter 2, the background datasets were compiled for the electricity production process, oleic acid production process and electricity production process. Table 28 represents the input and output data for the production of xanthate salts.

Table 28 Input and Output for the production of Xanthate Salts (derived from Kunene (2014))

Description			Units	Flows per tonne xanthate produced	Flows per 100 tons coal ultrafines treated
Inputs	Materials	Coal	tons	1.55	0.25
		Oil	tons	9.23	1.50
		Natural Gas	m^3	178	29
		Nitrogen	tons	0.159	0.026
		Sodium Chloride	tons	0.856	0.139
		Carbon dioxide	tons	6.62	1.08
Outputs	Emissions to Land	Solid Waste	kg	99	16
	Emissions to Air	SO_2	kg	84	14
		CO_2	kg	2627	427
		CO	kg	0.410	0.0666
		N_2O	kg	0.0631	0.0103
		NH_3	kg	0.0857	0.0139
	Emissions to Water	Phosphate	kg	37.8	6.15
		Organic substances, unspecified	kg	5.85	0.951
		Sulfuric acid	kg	75.6	12.3
		Boron	kg	30.2	4.92
		Chloride	kg	2.41	0.39
		Suspended solids, unspecified	kg	44.7	7.27

As can be seen from Table 28, the production of xanthate salts results in the generation of a considerable amount of air emissions, specifically sulfur dioxide and carbon dioxide which together form about 99% of the total air emissions. This is due to the production of carbon disulphide that is subsequently used to manufacture xanthate salts as well as the consumption of coal based energy in the production of xanthate salts (Kunene, 2014). The input and output streams for the associated production processes of xanthate salts are presented individually in Appendix E. The input and output flows for the production of oleic acid (derived from the Ecoinvent database) are presented in Table 29.

Table 29 Input and output data for the production of Oleic acid (derived from the Ecoinvent version 2.2 database)

Description			Units	Flows per tonne oleic acid produced	Flows per 100 tons coal ultrafines treated
Inputs	Materials	Coal, Brown, In the ground	kg	60.5	16.9
		Coal, Hard, Unspecified	kg	108	30.3
		Natural Gas	m^3	96.5	26.9
Outputs	Emissions to Air	CO_2	kg	208	58.1
		CO	kg	26.7	7.44
		NH_3	kg	2.34	0.65
	Emissions to Water	Al	kg	0.000689	0.000192
		As	kg	0.000370	0.000103
		Boron	kg	0.00501	0.00140
		Carbonate	kg	0.00773	0.00216
		Chloride	kg	24.5	6.84
		Calcium	kg	0.399	0.111
		Zinc	kg	0.101	0.0281
		Sulfates	kg	2.70	0.754
	Emissions to Land	Solid Waste	kg	209.55	58.46

As can be seen from Table 29, compared to the production of xanthate salts, production of oleic acid is characterized by lower air and water emissions as well as a reduced consumption of non-renewable resources.

Table 30 represents the major input and output data of the electricity production process. Since South African electricity is largely coal based (discussed in Section 1.1.2), it results in the generation of considerable amounts of sulfur dioxide and carbon dioxide as well as other gaseous emissions. The emissions to water are mostly comprised of salt forming anion and cations (phosphate and calcium) and sulfuric acid.

Table 30 Input and output data for the South African electricity mix production process (Dick, 2012)

Description			Units	Flows per MWh Electricity produced	Flows per 100 tons coal ultrafines treated	
					Base Case	Two-stage Flotation Process
Inputs	Materials/ Resources	Water, process and cooling	tons	1.32	0.01320	2.32
		Coal	tons	0.381	0.00381	0.671
Outputs	Emissions to Air	Carbon dioxide, fossil	kg	1200	12.0	2112
		Sulfur dioxide	kg	7.85	0.0785	13.8
		Nitrogen oxides	kg	3.96	0.0396	6.97
		Dinitrogen monoxide	kg	0.0102	0.00010	0.01795
		Ammonia	kg	0.0000914	0.0000009	0.00016
		Carbon monoxide, fossil	kg	0.114	0.00114	0.200
	Emissions to Land	Inert Material	kg	110	1.100	194
	Emissions to water	Phosphate	kg	0.0419	0.00042	0.0738
		Organic substances, unspecified	kg	0.00648	0.00006	0.0114
		Sulfuric acid	kg	0.0838	0.00084	0.148
		Boron	kg	0.0335	0.00034	0.0590
		Chloride	kg	0.00267	0.00003	0.00470
		Suspended solids, unspecified	kg	0.0495	0.00050	0.0871

Figure 21 illustrates the percentage differences in the gaseous emissions of the electricity, xanthate and oleic acid production processes per 100 tons of ultrafine coal treated by means of the two-stage flotation process.

As can be seen from Figure 21, the carbon dioxide emissions associated with the electricity production are significant compared to those associated with the xanthate and oleic acid production processes. However, the sulfur dioxide emissions associated with xanthate salts production are similar to those associated with the production of electricity. Compared to the xanthate and electricity production process, production of oleic acid results in the least atmospheric emissions. Figure 22 illustrates the percentage differences in the water and land emissions of the electricity, xanthate and oleic acid production processes per 100 tons of ultrafine coal treated.

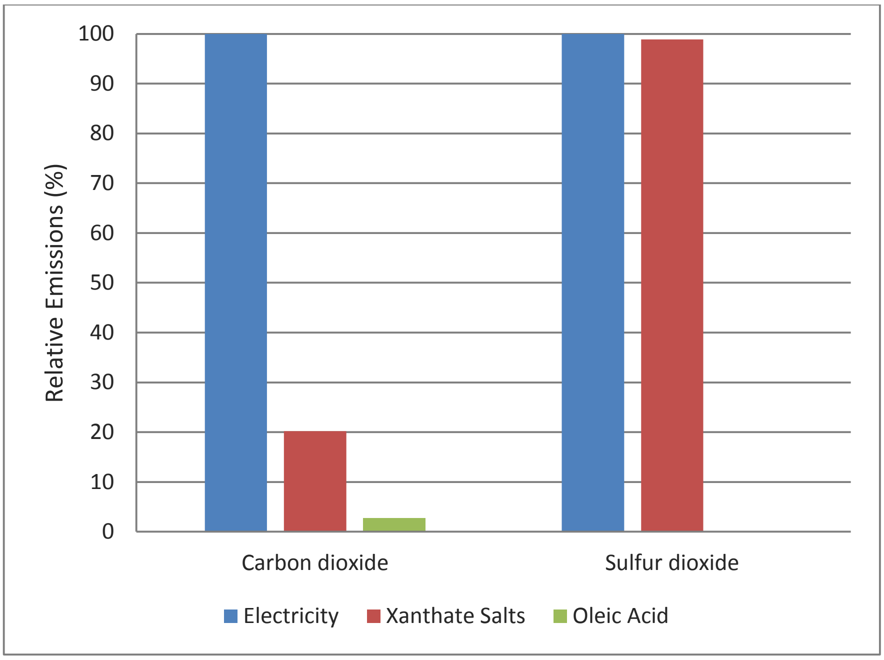


Figure 21 Relative gaseous emissions of the electricity, xanthate and oleic acid production processes per 100 tons of ultrafine coal treated by means of the two-stage flotation process

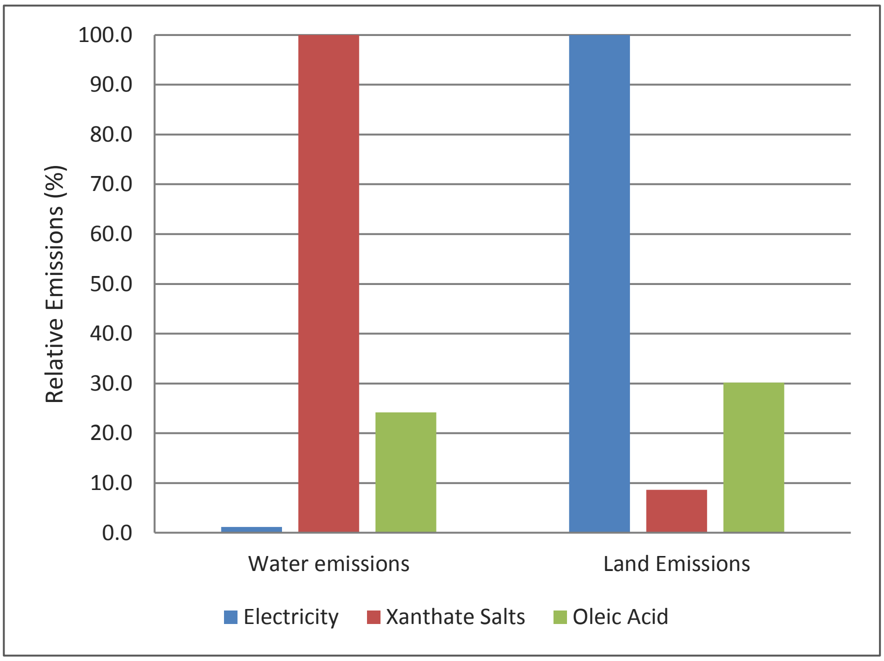


Figure 22 Relative total water and land emissions of the electricity, xanthate and oleic acid production processes per 100 tons of ultrafine coal treated by means of the two-stage flotation process.

As is evident from Figure 22, the xanthate production process exhibits the highest emissions to water compared to electricity and oleic acid production processes whilst the production of electricity exhibits the highest emissions to land compared to the other two background processes (refer to Table 30). Lastly, Figure 23 presents the percentage differences in the resource usages of the electricity, xanthate and oleic acid production processes.

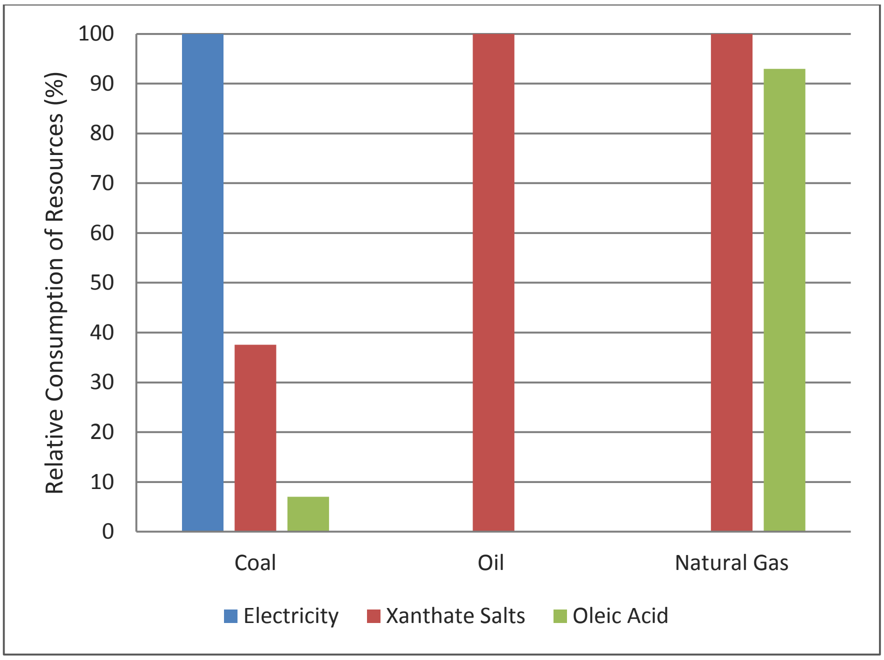


Figure 23 Relative comparisons of the resource usage of the electricity, xanthate and oleic acid production processes

As can be seen from Figure 23, the production of electricity results in relatively high consumption of coal compared to the other background processes whilst the natural gas consumed in the production of oleic acid is very similar to that of the production of xanthate salts.

4.1.3 Life Cycle Inventory

Table 31 represents the life cycle inventory table that combines the input and output datasets of the background and foreground data. Since the impact assessments were carried out per annum, the life cycle inventory was evaluated per annum which is equivalent to 864 kilotons feed coal ultrafines.

From Table 31 it can be seen that the largest contributors to water emissions are from the flotation reagents, sulfuric acid, sulfates, potassium, calcium, aluminium and iron. The rest of the elements are released in trace quantities relative to the feed stream. The biggest contributor to the air emissions is carbon dioxide and sulfur dioxide with the two-stage flotation process having considerably higher air emissions than the base case due to the relatively high energy requirement. To summarize the inventory results, Figure 24 and Figure 25 present the emission and resource loss contributions from the background processes and the foreground process for the two-stage flotation process and the base case respectively.

Table 31 Life cycle inventory for the two-stage flotation process and the base case

Description				Units	Flows Annually	
					Base Case	Two-stage flotation process
Inputs	**Materials**	Coal	Electricity (Foreground Process) Production	tons	32.9	6277
			Xanthate Production	tons	-	2177
			Oleic Acid Production	tons	-	407
		Natural Gas	Xanthate Production	m^3	-	250380
			Oleic Acid Production	m^3	-	232715
		Crude Oil	Xanthate Production	tons	-	12964
		Sodium Chloride	Xanthate Production	tons	-	1203
		Carbon dioxide, fossil	Xanthate Production	tons	-	9314
		Nitrogen	Xanthate Production	tons	-	223
		Water	Water usage by background processes	tons	114.0	34591.9
		Slurry Feed	Solids in slurry	tons	864000	864000
			Water in slurry	tons	7776000	7776000
Outputs	**Emissions to Water**	Acid (H+)	Produced from acid generating tailings	tons	2794	0.0
			Produced from Electricity Production	tons	0.00724	1.38
		Oleic Acid		tons	-	72317
		Xanthate		tons	-	98361
		Sulfate		tons	7972	120
		Aluminium		tons	3256	113
		Boron		tons	0.00289	1
		Iron		tons	15066	725
		Calcium		tons	6798	3223
		Potassium		tons	85.0	117
		Chloride		tons	0.000231	59.2
		Phosphate		tons	0.003620	53.8
		Carbonate		tons	-	0.0186
		Zinc		tons	33.4	7.48
		Chromium		tons	1.27	0.926
		Manganese		tons	8.36	13.2
		Cobalt		tons	0.25	0.30
		Nickel		tons	1.37	0.90
		Copper		tons	1.93	1.21
		Vanadium		tons	2.41	1.35
		Arsenic		tons	1.25	0.91
		Barium		tons	2.43	1.44
		Lead		tons	1.06	0.669
		Uranium		tons	0.101	0.0824
	Emissions to Air	Carbon dioxide, fossil		tons	104	20274
		Sulfur dioxide		tons	0.678	129
		Nitrogen oxides		tons	0.342	65.2
		Dinitrogen monoxide		tons	0.001	0.17
		Ammonia		tons	-	5.64
		Carbon monoxide, fossil		tons	0.010	66.1

Description				Units	Flows Annually	
					Base Case	Two-stage flotation process
	Emissions to Land	Solid Waste	Foreground Process	tons	864000	585792
			Electricity Production	tons	0.00110	1673
			Xanthate Production	tons	-	145
	Loss of Resources	Evaporated Water		tons	518400	71730
		Water lost through seepage		tons	129600	17932
		Water lost through entrainment		tons	648000	89662
		Unrecovered Coal		tons	405734	232152
		Remaining Gangue material		tons	458266	353640
	Recycled Resources	Water		tons	7240780	7631268
	Product	Recovered Coal		tons	-	228971
		Recovered Sulfide-Rich Fraction		tons	-	17280

To summarize the inventory results, Figure 24 and Figure 25 present the emission and resource loss contributions from the background processes and the foreground process for the two-stage flotation process and the base case respectively.

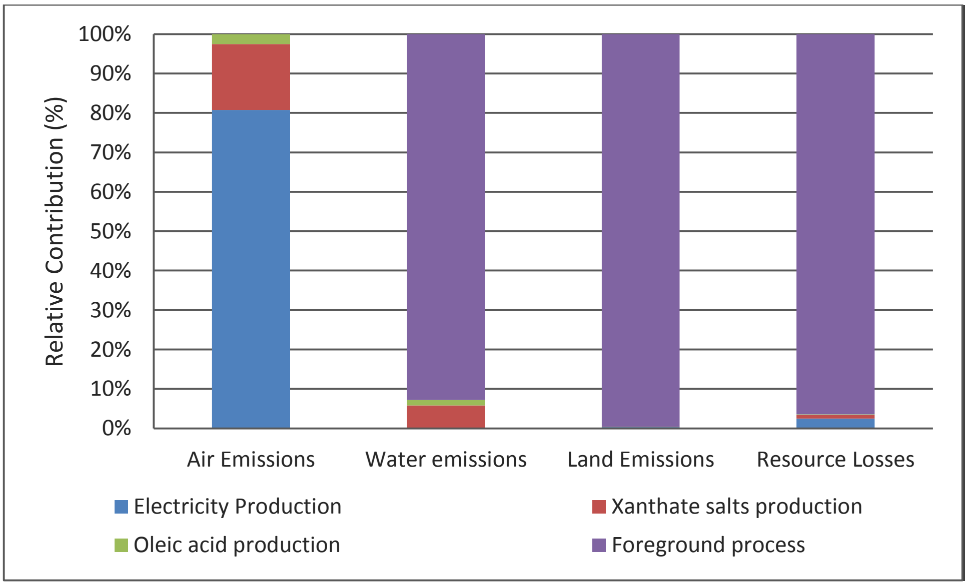


Figure 24 Emission and resource loss contributions from the background processes and the foreground process for the two-stage flotation process

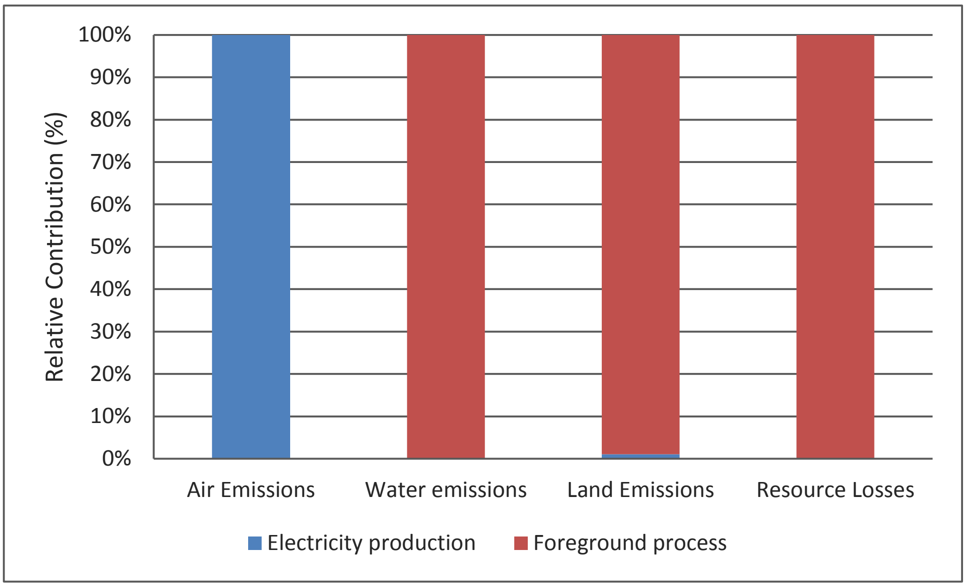


Figure 25 Emission and resource loss contributions from the background processes and the foreground process for the base case

From Figure 24 it can be seen that the water and land emissions and resource losses are dominated by the two-stage flotation foreground process. Air emissions are dominated by the electricity production process, with a smaller contribution (<15%) attributed to xanthate production. The contribution of the two-stage flotation foreground process (or solid waste disposal process) to the total resource losses is due to the significant amount of unrecovered coal remaining in the final disposed stream. Even though the water emissions are dominated by the foreground process, the background processes also make a notable contribution (approximately 8%) to the emissions. As for the base case (represented in Figure 25), most of the water and land emissions and the resource losses are dominated by the foreground process whilst the air emissions are dominated by the electricity production process, since there are no air emissions associated with the foreground process.

4.2 Impact assessment

This section presents the impact assessment results of the study. The impact categories have been divided into impacts related to atmospheric emissions, impacts related to resource usage and recovery and impacts related to emissions to water. The impact categories related to atmospheric emissions include climate change and terrestrial acidification whilst those associated with resource usage and recovery include natural land transformation, fossil fuel depletion and water footprint. Furthermore, the ones related to emissions to freshwater include salinity, freshwater eco-toxicity, aquatic water pollution risk, drinking water quality risk and aqueous acidification. The impact indicators were evaluated per annum.

4.2.1 Resource related impacts

A crucial aspect of the project involved the evaluation of the impacts due to the resource usages and recoveries associated with each waste treatment scenario, as well as land transformation impacts. Table 32 represents the quantified impact indicators for the resource related impact categories (refer to Appendix C for the detailed impact indicators).

Table 32 Resource related quantified impact indicators

Impact Category	Impact Indicator	Unit	Two Stage Flotation Process	Base Case
Fossil Fuel Depletion	Fossil Depletion Potential	tons oil equivalent/year	254810	405767
Natural Land Transformation	Transformed area	m^2/year	41 889	61 714
Consumptive Water Footprint	Consumptive water use	m^3/year	223 410	1 614 618

Figure 26 illustrates the fossil fuel depletion impact category for the two-stage flotation process and for the base case.

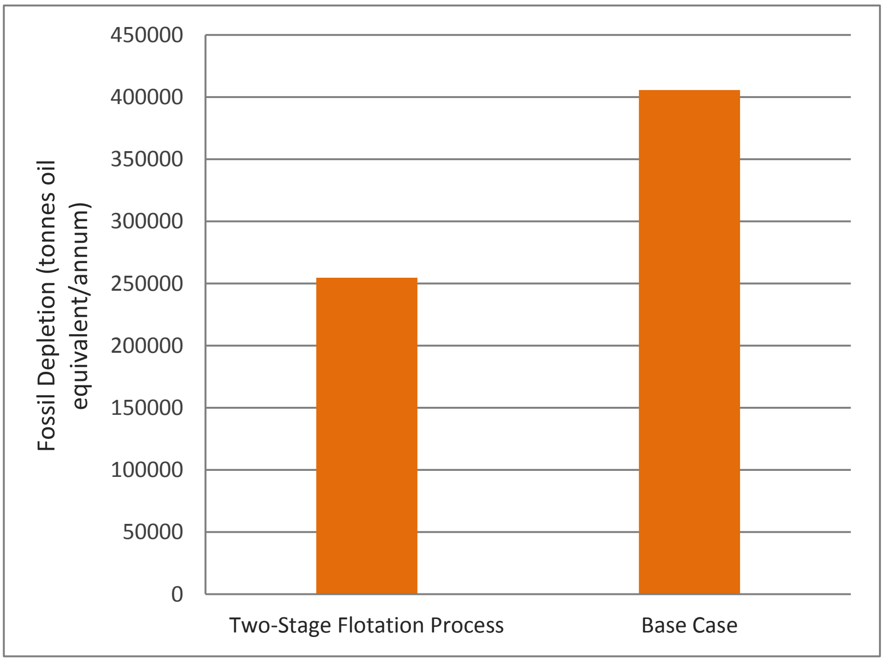


Figure 26 Fossil Fuel Depletion for the two-stage flotation process and the base case for 864 kilotons of feed coal ultrafines/annum

As can be seen from Figure 26, the fossil fuel depletion potential for the two-stage flotation process is significantly less (approximately 37% less) than that of the base case. What is interesting to note is that even though the two-stage flotation process is associated with the consumption of large amounts of non-renewable resources, compared to the base case, the fossil fuel depletion of the two-stage flotation process is still considerably lower. This is so because the coal lost in the waste stream of the base case scenario is notably higher (approximately 170 kilotons higher) than the non-renewable resources consumed as well as the coal lost in the waste stream of the two-stage flotation process (as represented by Table 31). Figure 27 illustrates the water footprint impact category for the coal waste management scenarios.

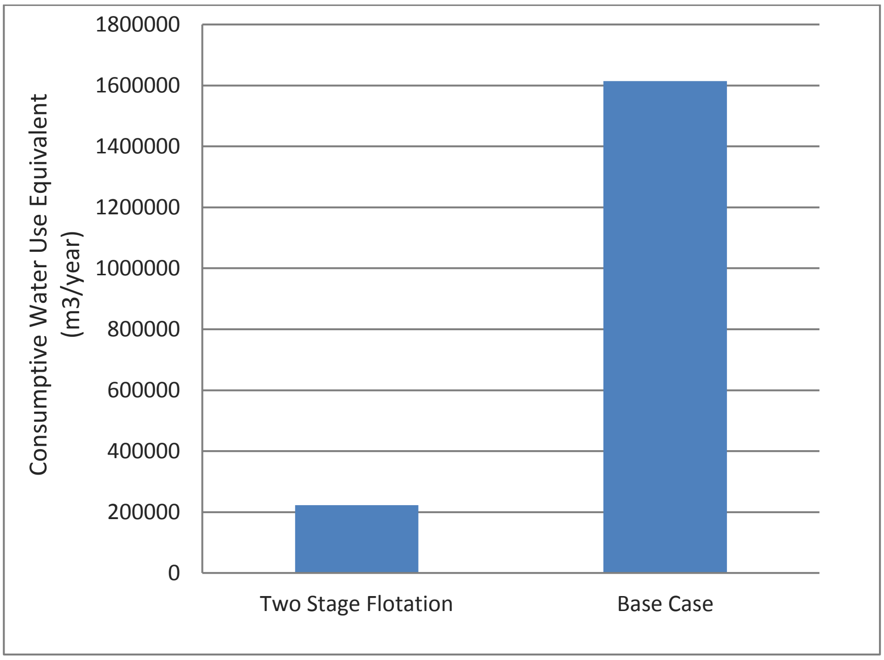


Figure 27 Water Footprint impact indicator for the two-stage flotation process and the base case for 864 kilotons of feed coal ultrafines/annum

As can be seen from Figure 27, the consumptive water use impact indicator for the two-stage flotation process is approximately 87% less than that of the base case. This is largely due to the fact that the two-stage flotation process involves passing the sulfide-lean tailings through a filter press (in addition to the thickener) before disposal. This results in the disposal of a waste stream with a lower moisture content and hence a significantly lower water footprint than the base case. Figure 28 represents the natural land transformation impact category which was calculated from the solid waste emissions to land.

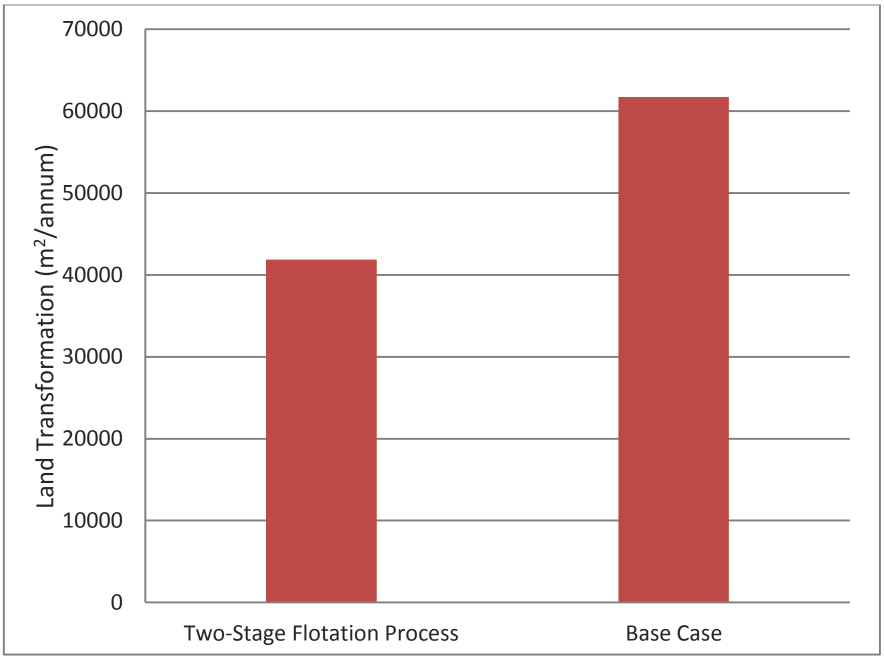


Figure 28 Land transformation impact indicator the two-stage flotation process and the base case for 864 kilotons of feed coal ultrafines/annum

As can be seen from Figure 28, the natural land transformation of the two-stage flotation process was also less than that of the base case due to the recovery of coal and the sulfide rich fraction. However, due to the relatively large solid waste still remaining after the two-stage flotation process, the difference between the two scenarios is small compared to the other impact categories.

4.2.2 Air related impacts

The impacts categories assessed under this category include climate change and terrestrial acidification. Global warming potential, which measures the emissions of greenhouse gases released to air in kilograms CO_2 equivalent, and terrestrial acidification, which measures the amount of acidifying emissions in kilograms SO_2 equivalent, represent the impact indicators. The quantified impact indicators are presented in Table 33 and illustrated in Figure 29 and Figure 30.

Table 33 Quantified air related impact indicators for 864 kilotons of feed coal ultrafines/annum

Impact Category	Impact Indicator	Unit	Two Stage Flotation	Base Case
Climate Change	Global warming potential	kg CO_2 equivalent/year	2.08E+07	1.04E+05
Terrestrial Acidification	Terrestrial acidification potential	kg SO_2 equivalent/year	200222	870

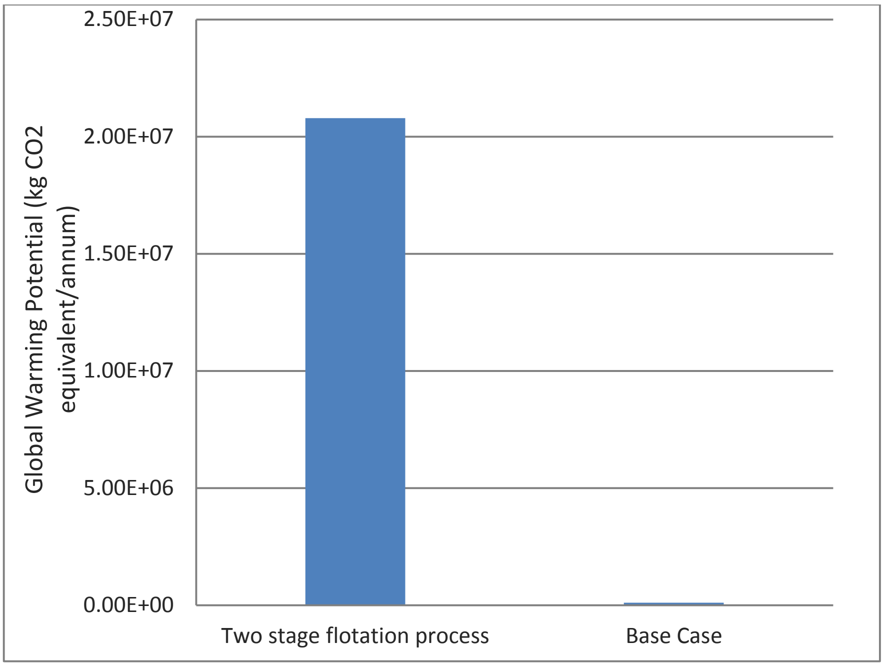


Figure 29 Climate change for the two-stage flotation process and the base case for 864 kilotons of feed coal ultrafines/annum

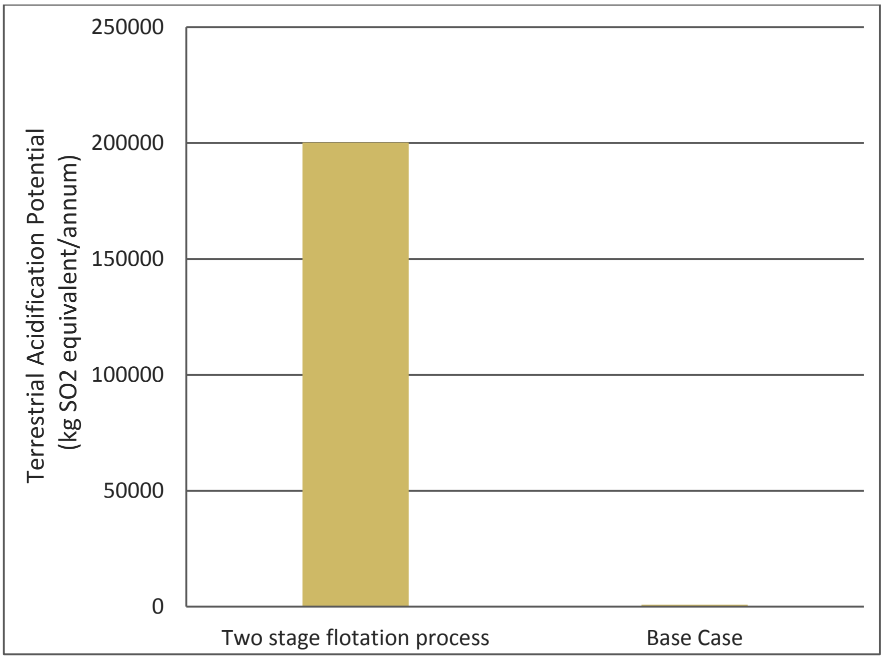


Figure 30 Terrestrial acidification for the two-stage flotation process and the base case for 864 kilotons of feed coal ultrafines/annum

As can be seen from both figures, the impacts associated with the two-stage flotation process are notably higher than those for the base case. This is because the two-stage flotation process has an appreciably higher energy demand compared to the base case and because the reagent production processes release air emissions as highlighted in Section 4.1.

4.2.3 Water related impacts

The water related impacts assessed were to account for the emissions of substances released into local groundwater sources both by the foreground and background processes. To assess this, five impact categories were evaluated, these include, freshwater eco-toxicity, salinity, aquatic water pollution risk, drinking water quality risk and aqueous acidification (acid-formation risk). Table 34 presents the quantified water quality impact indicators.

Table 34 Quantified water related impact indicators for 864 kilotons of feed coal ultrafines/annum

Impact Category	Impact Indicator	Unit	Two Stage Flotation Process	Base Case
Eco-toxicity	Freshwater eco-toxicity potential	ton 1,4 Dichlorobenzene equivalent/year	617	1 157
Salinity	Total salinity potential	ton Total Dissolved Salts (TDS)/year	588	2 558
Drinking water quality risk	Total risk potential	dimensionless	105	149
Aquatic water pollution risk	Total risk potential	dimensionless	3.72	4.99
Aqueous Acidification	Total risk potential	dimensionless	0	101 265

From Table 34, it can be seen that the base case is associated with significant eco-toxicity impacts as well as salinity impacts. The aqueous acidification total risk potential is equal to zero for the two-stage flotation process since the sulfide-lean stream was classified as non-acid forming by the laboratory ARD prediction tests. These results are illustrated diagrammatically and explored more in detail in the following sub-sections.

Freshwater eco-toxicity

The freshwater eco-toxicity impact category, as described in Chapter 3, was determined by multiplying the individual eco-toxicity characterisation factors (developed form fate and exposure models) for various substances with the inventory mass flows. The freshwater eco-toxicity impact assessment however remains limited due to the unavailability of the equivalency factor for iron. Figure 31 represents the freshwater eco-toxicity impacts for the two-stage flotation process and the base case and Figure 32 aims to further illustrate the freshwater eco-toxicity impact for the two scenarios by presenting the contributions of the various substances towards the total freshwater eco-toxicity potential.

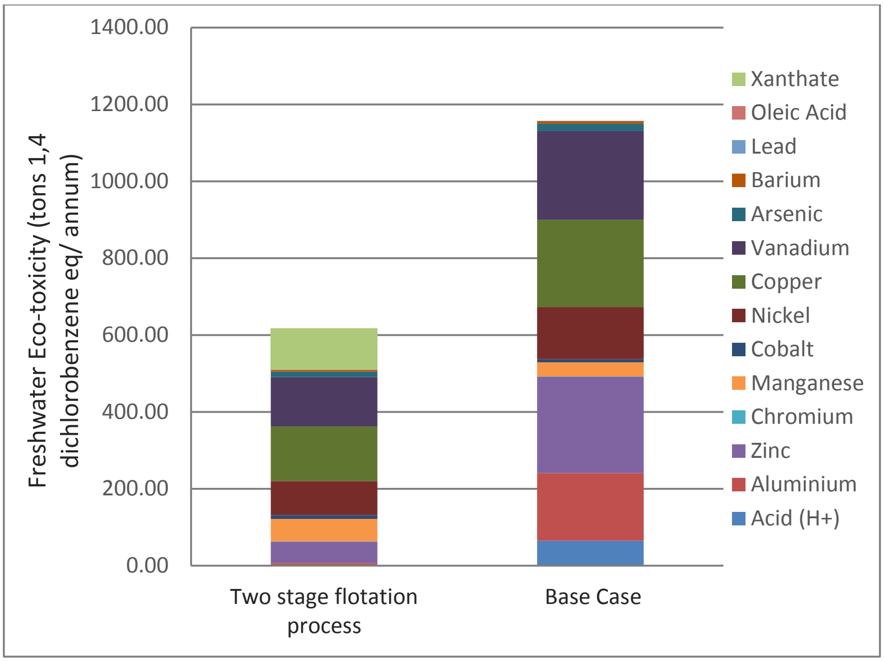


Figure 31 Freshwater eco-toxicity for the two-stage flotation process and the base case for 864 kilotons of feed coal ultrafines/annum

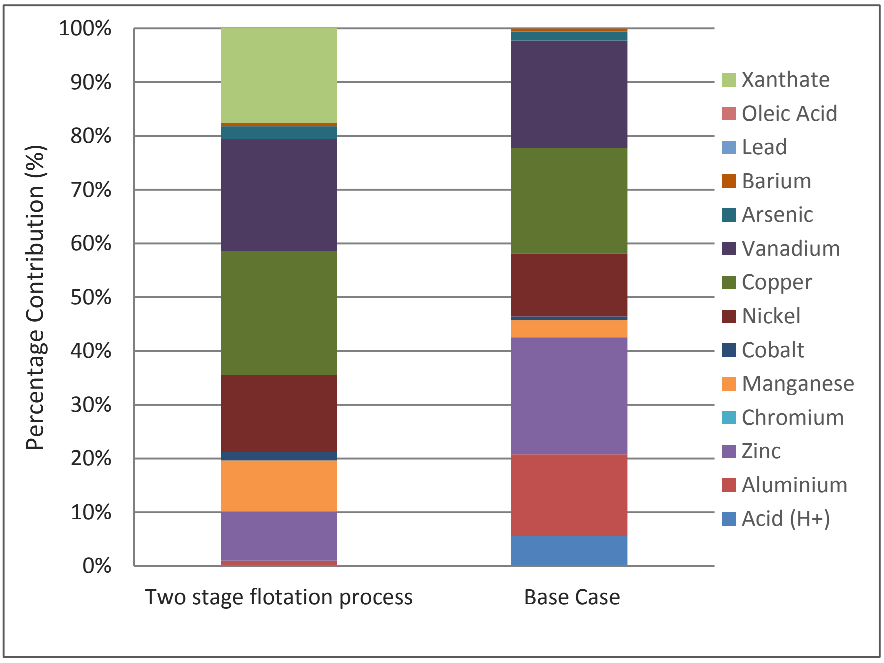


Figure 32 Percentage contributions towards the freshwater eco-toxicity for the two-stage flotation process and the base case for 864 kilotons of feed coal ultrafines/annum

From Figure 31, it can be seen that the freshwater eco-toxicity impact associated with the two-stage flotation process is relatively low compared to that of the base case and in Figure 32 it can be seen that the SIBX reagent contributes a significant percentage towards the eco-toxicity impact (approximately 18%) whilst oleic acid contributes less than 1% towards the eco-toxicity impact. This is consistent with the difference in toxicity properties of the reagents, as discussed in Section 2.3. The metal emissions on the other hand form a significant contribution to the total freshwater eco-toxicity potential even though these metals are available in trace quantities (approximately 82% of the eco-toxicity potential of the two-stage flotation process and 95% for the eco-toxicity potential of the base case). This can be explained by their relatively high equivalency factors compared to the other substances, particularly zinc, manganese, copper and vanadium which have the highest equivalency factors amongst the metals.

Salinity

The salinity impact category, represented by Figure 33, describes the impact associated with the release of salt forming cations and anions into freshwaters. As can be seen from Figure 33, the total salinity potential of the two-stage flotation process is 23% less than that of the base case. This can be explained by the fact that the two-stage flotation process is associated with less salt forming cation and anion emissions such as calcium, cholrides, sulfates, potassium and bicarbonates (refer to Table 31) compared to the base case.

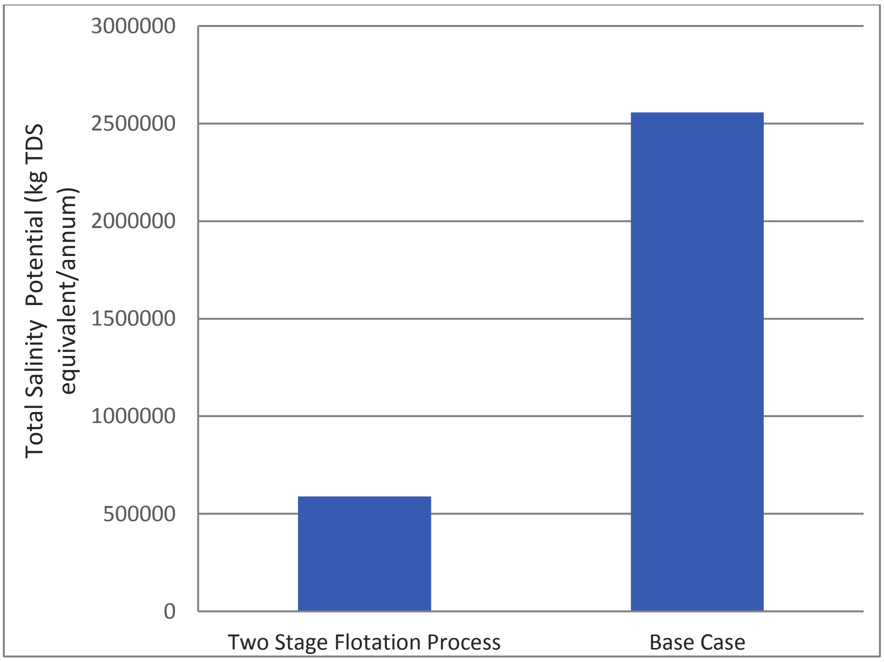


Figure 33 Salinity impact category for the two-stage flotation process and the base case for 864 kilotons of feed coal ultrafines/annum

Aquatic water pollution risk, drinking water quality risk

The aquatic water pollution risk and the drinking water quality risk were evaluated to assess the impact associated with the release of metal emissions to the environment. Figure 34 represents the total aquatic water pollution risk impact of the disposed metals, expressed by the sum of the metal risk potential factors. Figure 35 represents the metals associated with aquatic pollution risk potential factors lower than 0.002.

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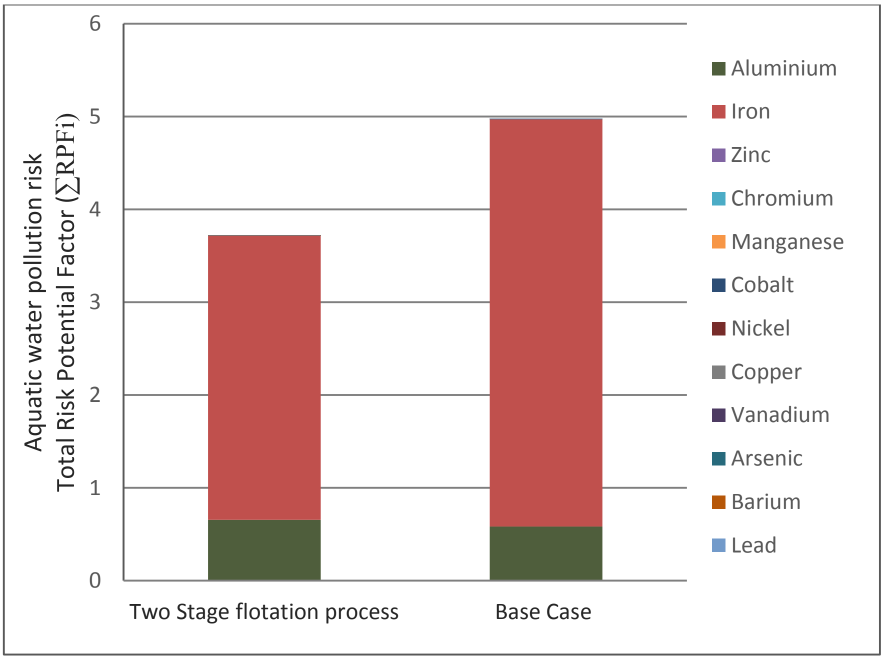

Figure 34 Aquatic water pollution risk for the two-stage flotation process and the base case

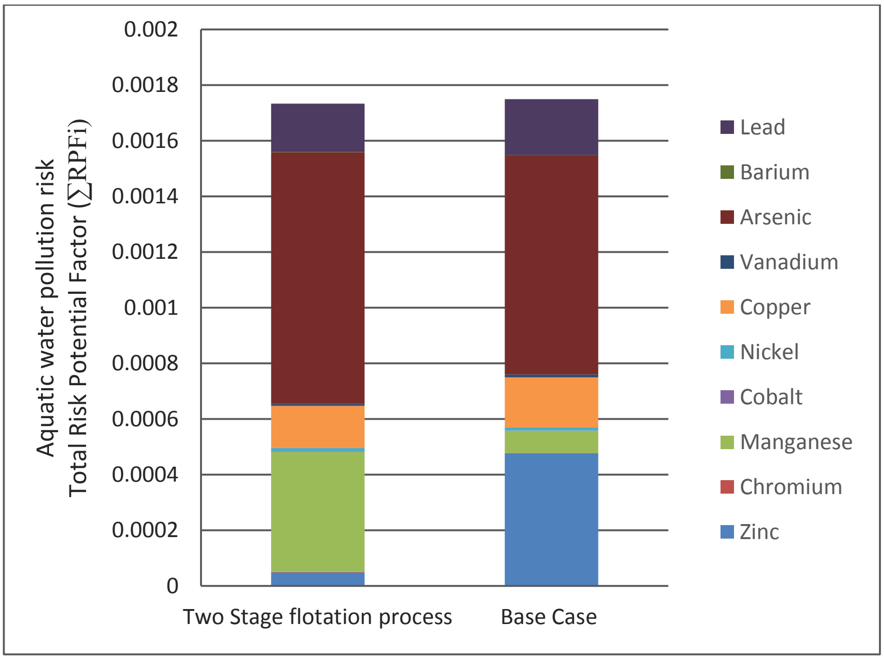

Figure 35 Aquatic water pollution risk for metals with risk potential factors less than 0.002 for the two-stage flotation process and the base case

As can be seen from Figure 34, the metals with the highest risk potential factors for both of the scenarios are iron and aluminium as they have the highest concentrations in both of the final disposed streams and are also associated with low acceptable concentrations in aquatic waters. The small difference in the total risk potential factor of iron between the two scenarios suggests a significant deportment of iron bearing phases to the sulfide-lean tailings stream. This can be explained by the relatively poor sulfide-sulfur separation of the case study compared to the previous studies since the majority of the iron present is associated with sulfide mineral phases as was highlighted in Chapter 2. However, overall, the two-stage flotation process is associated with a lower total risk potential factor than the base case which means that the sulfide-lean tailings present a lower risk of aquatic pollution than the untreated coal ultrafines. The risk potential factors for uranium and titanium were not calculated due to the unavailability of acceptable aquatic concentrations of the metals in the water quality guidelines. From Figure 35, it can be seen that the total aquatic water pollution risk for metals with risk potential factors less than 0.002 for the two scenarios is very similar. It is also evident that the risk potential factors for arsenic and manganese for the two-stage flotation process are higher than those of the base case. This is so because the sulfide-lean tailings stream (from the two-stage flotation process) contains a higher concentration of manganese and arsenic than the feed stream, as was presented in Table 21 in Chapter 3.

Figure 36 represents the total drinking water quality risk of the disposed metals and Figure 37 represents the metals with drinking water quality risk potential factors lower than 0.002. What differentiates the aquatic water pollution risk from the drinking water quality risk is the environmentally acceptable concentration used when calculating the risk potential factors, as was described in Chapter 3. The environmentally acceptable concentrations differ because the safe concentration exposures and aesthetic guidelines differ between drinking and aquatic water.

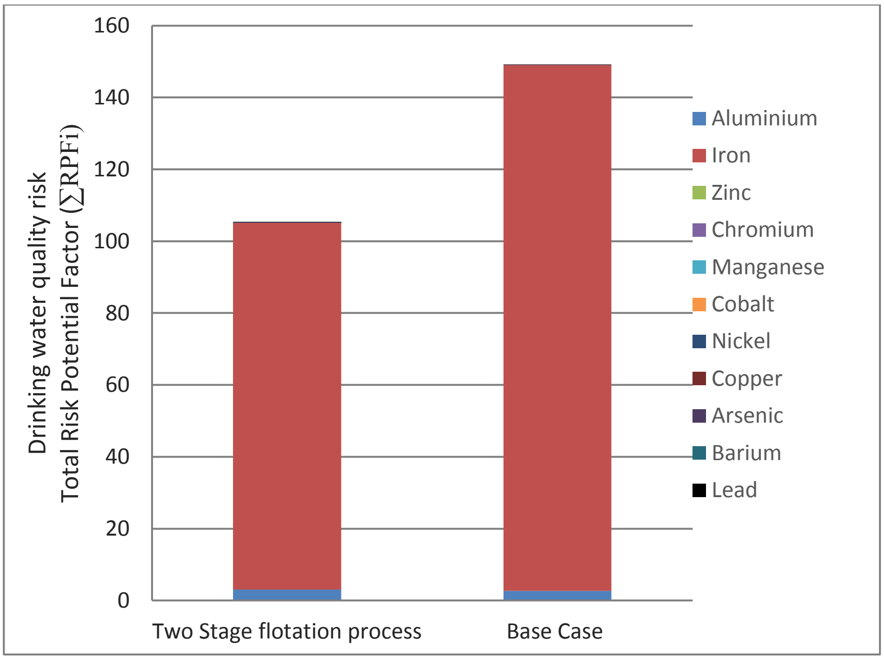

Figure 36 Drinking water quality risk for the two-stage flotation process and the base case

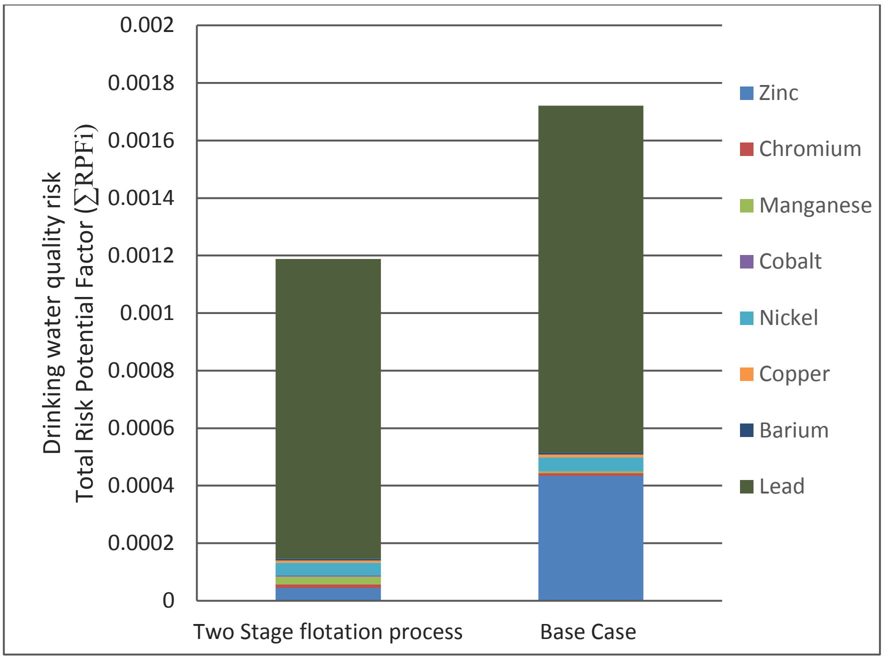

Figure 37 Drinking water quality risk for metals with risk potential factors less than 0.002 for the two-stage flotation process and the base case

Just like the aquatic water pollution risk, it can be seen that the drinking water quality risk is also dominated by iron. Unlike the aquatic water pollution risk however, the aluminium risk potential factor is not as dominant as there are higher acceptable aluminium concentrations in drinking water than aquatic waters. This can be explained by the higher toxicity of aluminium to aquatic life such as fish compared to humans (DWAF, 1996). From Figure 36, it is can be seen that the total risk potential factor of the two-stage flotation process is less than that of the base case which suggests that the disposed sulfide-lean tailings present a lesser risk to drinking water than the untreated coal ultrafines. The risk potential factors for uranium, titanium and vanadium were not calculated due to the unavailability of acceptable drinking water or acceptable domestic water concentrations of the metals in the guidelines. From Figure 37, it can be seen that lead forms a significant contribution to the risk potential factors for both scenarios followed by zinc. The total risk potential factor associated with the two-stage flotation process for these specific metals is lower than that of the base case.

CHAPTER 5

DISCUSSION

The results of the life cycle inventory and impact assessment stage were presented and discussed on an individual basis in Chapter 4. This chapter now aims to interpret and synthesis these results with a view of addressing the key research questions namely:

i. How does the environmental waste burdens of the disposal of untreated coal ultrafines compare to those of the two-stage flotation process and the disposal of desulfurised tailings?

ii. Which processes associated with both of the coal waste management scenarios contribute the most to the environmental burdens of the scenarios?

iii. How is the environmental performance of the two-stage flotation process affected by the variation of input parameters?

iv. What are the key environmental implications associated with implementation of the two-stage flotation process in the South African context?

This chapter begins by summarizing and discussing the relative percentage comparisons of the environmental burdens between the base case and the two-stage flotation process. The chapter then goes on to present the process contribution and sensitivity analyses graphs, which will address the second and third key objectives. Lastly the chapter will end by evaluating the key environmental implications associated with the implementation of the two-stage flotation process in the South African context and highlighting the limitations of the study.

5.1 Environmental burden comparisons

It has been shown in Chapter 4 that the two-stage flotation process is associated with fewer emissions to land and water as well as less resource losses compared to the base case. Figure 38 and Figure 39 represent the relative comparison graphs for the two-stage flotation process and the base case for processes' emissions and resource usages respectively.

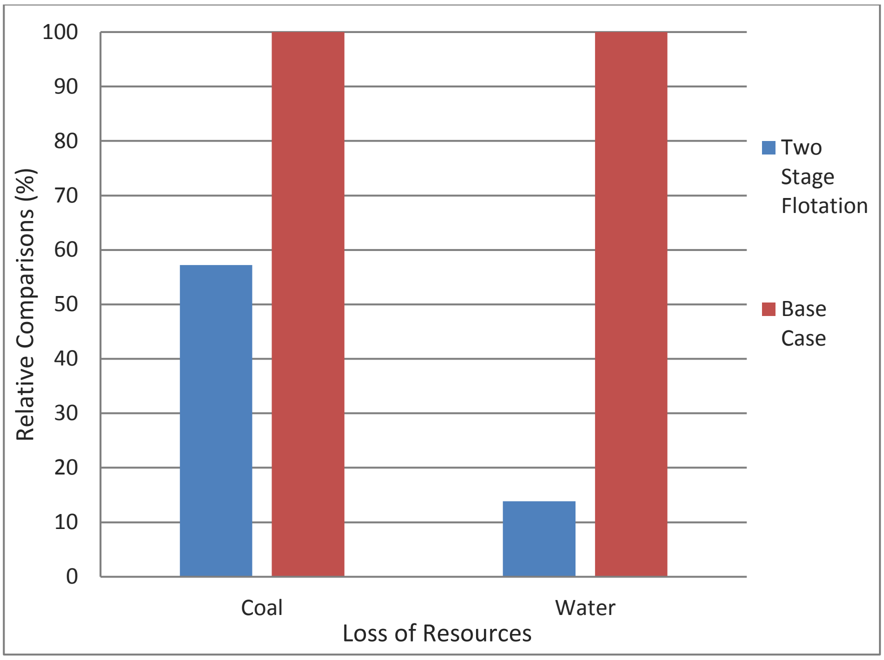

Figure 38 Relative comparisons of the resource losses of the two-stage flotation process and the base case

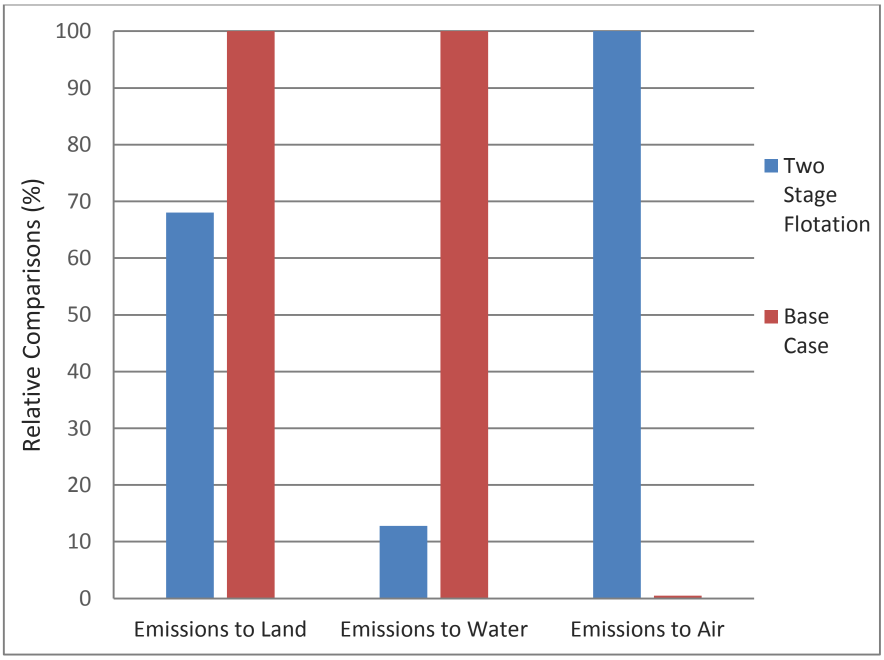

Figure 39 Relative comparisons of the emissions for the two-stage flotation process and the base case

From Figure 38 it can be seen that the coal resource losses associated with the two-stage flotation are 43% less than those associated with the base case and the water resource losses are 86% less than those of the base case. The big difference in the loss of water between the two-stage flotation process and the base case has been attributed to the additional dewatering the sulfide-lean tailings stream goes through before disposal. As for emissions to land and water, it can be seen from Figure 38 that the two-stage flotation process emits 32% less emissions to land than the base case and 13% less emissions to water than the base case. This reduction in emissions to water and land is mainly due to the recovery of the coal and sulfide-rich product. Lastly, for the emissions to air, the base case has 95.5% less emissions than the two-stage flotation process due to the high energy requirement of the latter. Figure 40 represents the summary of the relative comparisons of the base case and the two-stage flotation process for each of the impact categories.

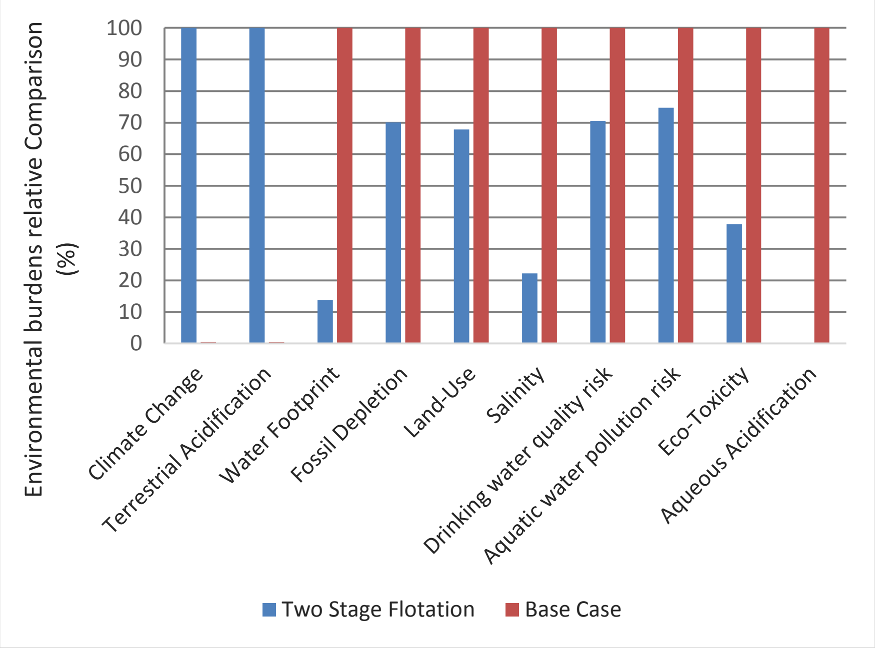

Figure 40 Overall relative comparison graph for two-stage flotation process and the base case

From the graph it can be seen that the impact categories related to atmospheric emissions (climate change and terrestrial acidification) are the only ones in which the base case outperforms the two-stage flotation process (>99.9% difference). For the rest of the impact categories, the two-stage flotation process performs better than the base case. The aqueous acidification impact category has the highest percentage difference (100%) followed by the water footprint and salinity impact categories which have percentage differences of 87% and 78% respectively. Fossil depletion, land-use, aquatic water pollution risk, and drinking water quality risk have similar percentage differences, with the percentage differences between the scenarios being 70%, 68%, 70.6%, 75% respectively. The percentage difference between the scenarios for the eco-toxicity impact category is 53%. The relatively poor performance of the two-stage flotation process in relation to climate change and terrestrial acidification has been attributed to its relatively high greenhouse and acidifying gaseous emissions compared to the base case.

Outside of the evaluated impact categories, one of the major negative environmental implications of the direct disposal of coal ultrafines that has been noted in Chapter 2 is spontaneous combustion. Due to the numerous factors that affect the likelihood of spontaneous combustion, it could not be accurately predicted which waste stream would spontaneously combust therefore the emissions in the event of spontaneous combustion occurring were not incorporated into the inventory. However, since the two-stage flotation process involves the disposal of sulfide-lean tailings which contain less pyrite and carbonaceous material than the untreated coal ultrafines, it can be said that it is at a lesser risk of spontaneous combustion relative to the base case. This is based on literature which has noted that increasing the pyrite contained in the coal waste as well as the carbonaceous material will increase the likelihood of spontaneous combustion occurring (Neiburger et al., 1977; Smith and Glasser, 2004). In the event therefore of spontaneous combustion occurring on the untreated coal ultrafines, the air emissions as a result (refer to Section 2.1.4) will surpass the air emissions associated with the electricity consumption of the two-stage flotation process. This is in the event that all of the waste is combusted.

Dust emissions have also been noted to form part of the negative environmental implications of the direct disposal of ultrafine coal waste. The amount of dust emissions dispersed however was not calculated since the amount of the dust emissions would depend on the specific site of disposal. Taking into account nonetheless the disposed waste streams of the foreground process, assuming the solid wastes of both scenarios are disposed on the same site with the same disposal method, the dust emissions from the base case are likely to be more due to the higher volume of solid waste. Without comparing it to the base case, on its own, the two-stage flotation process is associated with the disposal of a considerable amount of solid waste, of which might contribute a considerable amount of dust emissions.

5.2 Process contributions

Process contributions to impact indicators were evaluated by considering three main processes, these include the solid waste disposal process, the electricity production process and the flotation reagents production process. The risk based impact indicators were excluded from these analyses as they were only selected to cater for the disposed waste streams of the foreground processes and did not encompass the background processes. The process contributions to impacts for the two-stage flotation process are illustrated by Figure 41.

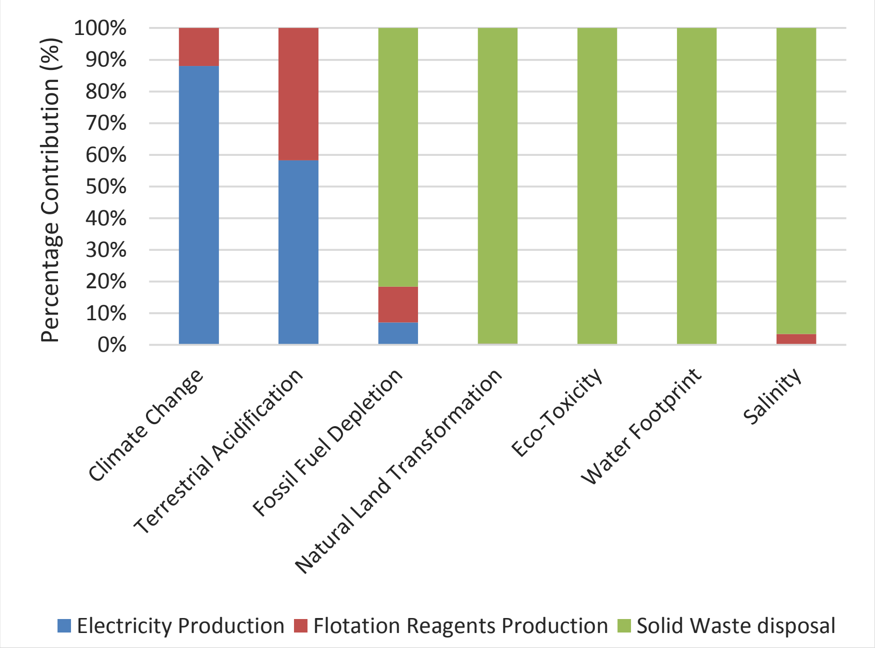

Figure 41 Process contributions to impacts for the two-stage flotation process

It can be seen from Figure 41 that the background processes dominate the climate change and terrestrial acidification impact categories. This is so because the solid waste disposal process does not involve the release of any greenhouse or acidifying gases. It is also interesting to note that, whilst the electricity production process is the process mostly dominant in the climate change impact category (88%), the flotation reagent production process makes a significant contribution towards the terrestrial acidification impact category (42%). This is so because the production of xanthate salts results in the direct release of SO_2 emissions as was presented in the inventory analysis in Chapter 4. Another thing to note is the small contribution of the electricity and the reagent production processes towards the fossil fuel depletion impact category (total of 18%) and the domination of the solid waste disposal process for this impact category. This shows that the non-renewable resources lost in the disposed stream (coal ultrafines) are significantly higher than those consumed in the background processes which was also highlighted in Chapter 4 in Figure 24. The rest of the impact categories (natural land transformation, eco-toxicity, water footprint and salinity) are mostly dominated by the solid waste disposal process, with the flotation reagents production process contributing a small percentage to the total salinity potential (3%). It should be noted however that the presence of flotation reagents themselves in the disposed waste stream forms a significant contribution to the eco-toxicity impact category (highlighted in Section 4.2.3) and this should not be confused with the contribution of their production processes to the impact category.

For the base case scenario (represented by Figure 42), since there is absence of flotation reagents in the process, the only two processes relevant are the electricity production process and the solid waste disposal process. As in the case of the two-stage flotation process, the climate change and the terrestrial acidification impact categories are dominated by the production of electricity since there are no direct atmospheric emissions in the foreground process. The rest of the impact categories are dominated by the solid waste disposal process since the electricity used in the base case is minimal hence unlikely to cause any significant contribution to the impacts.

5.3 Sensitivity analyses

As was described in Chapter 3, sensitivity analyses were conducted to investigate how the impact indicators are affected by the variation of input parameters and to demonstrate how flexible the environmental performance of the two-stage flotation process is. The sensitivity analyses for the variation of electricity consumption, coal yield, reagent dosages and available metal concentrations are presented in the following subsections

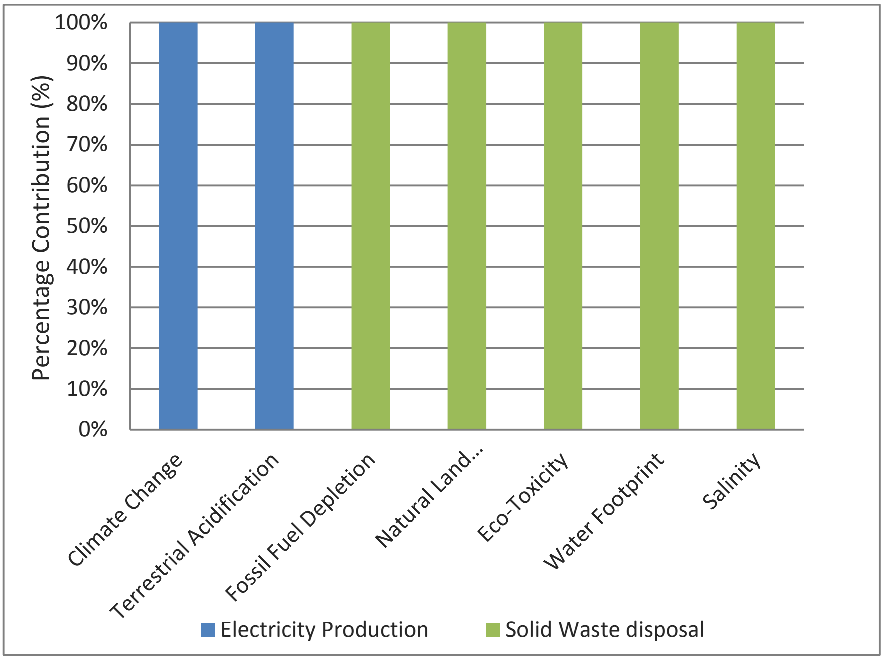

Figure 42 Process contributions to impacts for the base case

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5.3.1 Effect of electricity consumption

Electricity usage was varied because in a real plant situation due to equipment inefficiencies and significant energy losses, the energy required on a processing plant might be higher than the calculated energy. Furthermore there remains a large amount of uncertainty associated with the calculation of energy as was evidenced by the difference between the calculated flotation circuit energy using engineering heuristics and using the Marsden energy model (highlighted in Chapter 4).

The impact categories observed during the variation of this model input were climate change, terrestrial acidification and fossil fuel depletion. Figure 43 represents the effect of varying the electricity consumption of the two-stage flotation process on the climate change, terrestrial acidification and fossil fuel depletion impact categories.

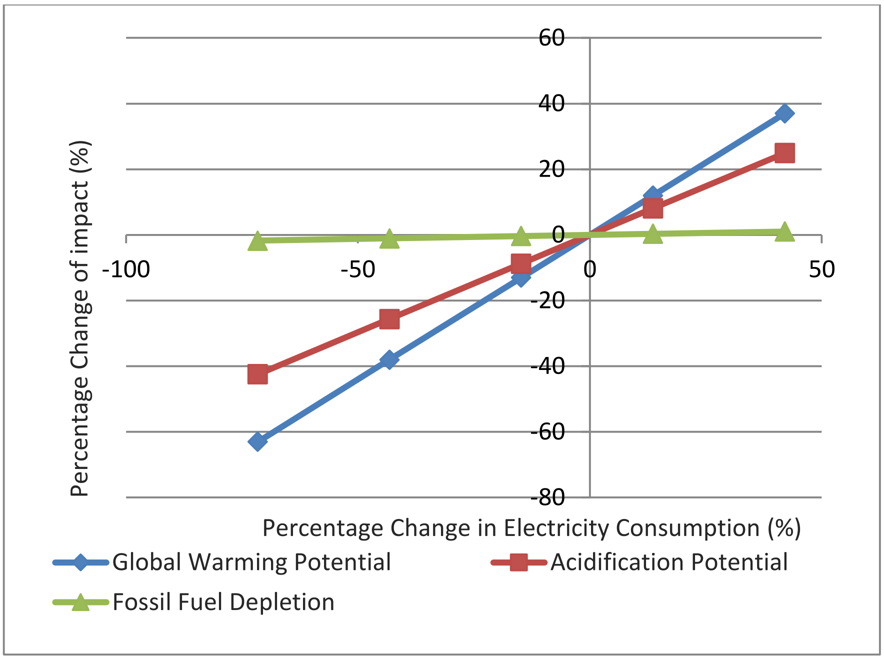

Figure 43 Effect of varying electricity consumption on climate change, terrestrial acidification and fossil fuel depletion

As can be seen from Figure 43, the impact category mostly affected by the variation of electricity consumption is climate change followed by terrestrial acidification and then lastly fossil fuel depletion. An increase in the electricity consumption of 47% results in an increase in the climate change impact category by 37%, an increase in terrestrial acidification impact category by 25% and an increase in fossil fuel depletion impact category by only 1%. This can be explained by the process contribution graph (Figure 41) which shows the electricity production process contributing a significant percentage to the climate and terrestrial acidification impact categories but not so much for the fossil fuel depletion impact category as the coal consumed for the electricity production is notably smaller than the coal recovered.

5.3.2 Effect of coal yield

The coal yield was varied as it forms an integral part of the environmental performance assessment of the two-stage flotation process and since coal yield is likely to be more or less due to the nature of the coal ultrafines. As was described in Chapter 2, from previous laboratory test work studies, coal yields can vary between 15% and 89% (the case study had a coal yield of 30.2%) which suggests the need for a sensitivity analysis on the coal yield. The impact categories that were observed during the variation of the coal yield were fossil fuel depletion and natural land transformation and were selected based on the process contribution graphs which showed the fossil fuel depletion impact and natural land transformation impact categories to be mostly due to the solid waste disposal process which is largely affected by the coal yield.

As can be seen from Figure 44, variation of the coal yield appreciably impacts both categories but impacts the fossil fuel depletion more as evidenced by the steeper slope of the line. An example is that an increase in the coal yield by 83% (equivalent to a coal yield of 55%) results in a 91 % reduction in fossil fuel depletion and a 36% reduction in land transformation. A decrease in the coal yield by 50% (equivalent to a coal yield of 15.1%) on the other hand will result in a 56% increase in fossil fuel depletion and a 22% increase in the natural land transformation. This shows how important the coal yield is to the fossil fuel depletion impact. An interesting thing to note is that lowering the coal yield to as low as 10% still results in a fossil fuel depletion potential lower than that of the base case even though the gap between the two scenarios would have narrowed.

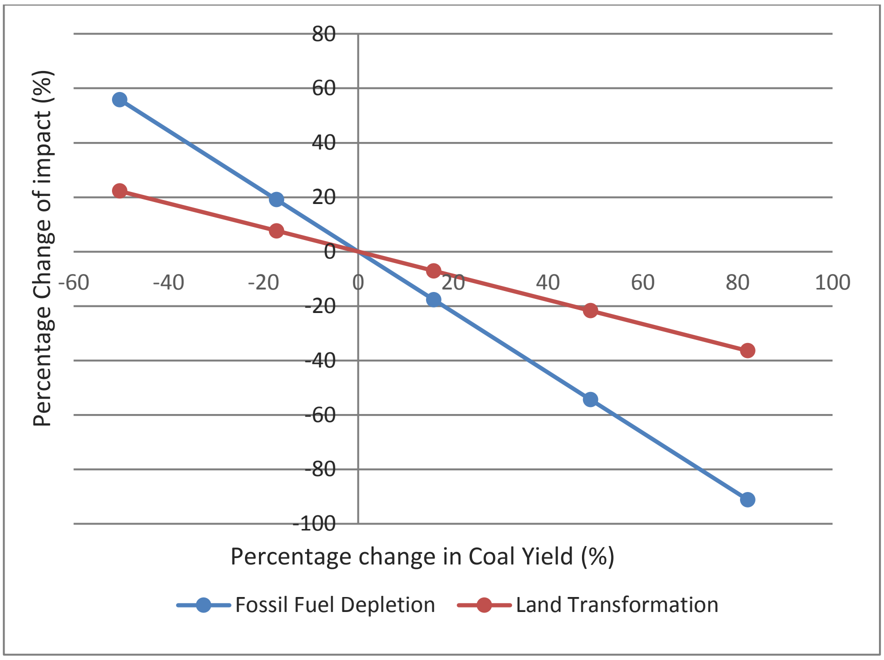

Figure 44 Effect of variation of coal yield on fossil fuel depletion and natural land transformation

5.3.3 Effect of reagent dosages

The reagent dosages were varied since on a real plant, there might be a need to add a higher or lower reagent dosage than intended due to material losses that occur and due to the nature of the coal ultrafines, as was highlighted in Chapter 2, some coal ultrafines achieved higher sulfide and coal recoveries at the same reagent dosages as the other coal ultrafines. Furthermore, since oleic acid is being used as a proxy for Nalflote 9858 and since it was noted in literature that Nalflote 9858 achieved 10% higher recoveries than the oily collectors it is essential to see how the model would react to higher oleic acid dosages that would have matched up with the Nalflote 9858 dosage.

Sodium isobutyl xanthate (SIBX)

The effect of the variation of SIBX reagent dosage on climate change, terrestrial acidification and eco-toxicity is shown in Figure 45. As can be seen from the diagram, varying the dosage of SIBX mostly impacts terrestrial acidification followed by eco-toxicity and then climate change. More specifically, an increase in the SIBX dosage by 100% also results in an increase in the terrestrial acidification impact by 40% whilst giving rise to an increase in the eco-toxicity impact of 20% and an increase in the climate change impact of 15%. The high impact of the terrestrial acidification impact category is because of the relatively high SO_2 emissions associated with the production of xanthate reagents. Even though the variation in the eco-toxicity impact with varying SIBX dosage is lower than that of the terrestrial acidification impact, it is still quite significant. Lastly the climate change is impacted the least due to the relatively low greenhouse gas emissions that come from the production of xanthate salts.

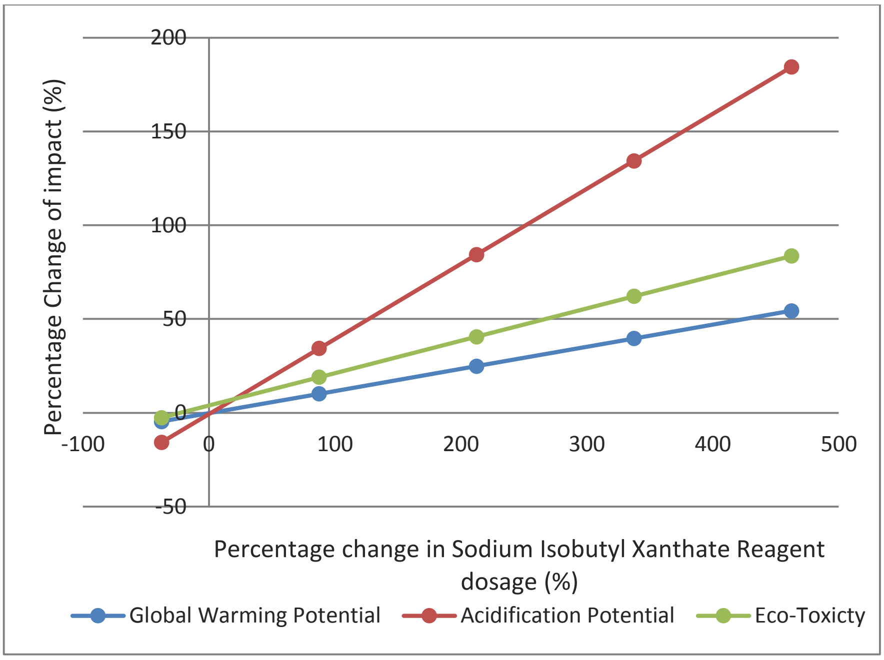

Figure 45 Effect of variation of Sodium Isobutyl Xanthate Reagent dosage on climate change, terrestrial acidification and eco-toxicity

Oleic Acid

The effect of varying oleic acid on climate change, terrestrial acidification and eco-toxicity is represented by Figure 46. As can be seen from Figure 46, climate change and terrestrial acidification vary the most with varying oleic acid reagent dosage followed by eco-toxicity. Increasing the oleic acid dosage by 100% (to a dosage of 0.5 ton/hr), increases the climate change and terrestrial acidification impact by approximately 1.5% and the eco-toxicity impact by approximately 1%. The variation of the impact categories with change in the oleic acid reagent dosage is notably smaller than that of the SIBX. This is so because of the relatively low inputs and output flows of the oleic acid production process as well as the lower eco-toxicity characterisation factor of oleic acid compared to xanthate salts.

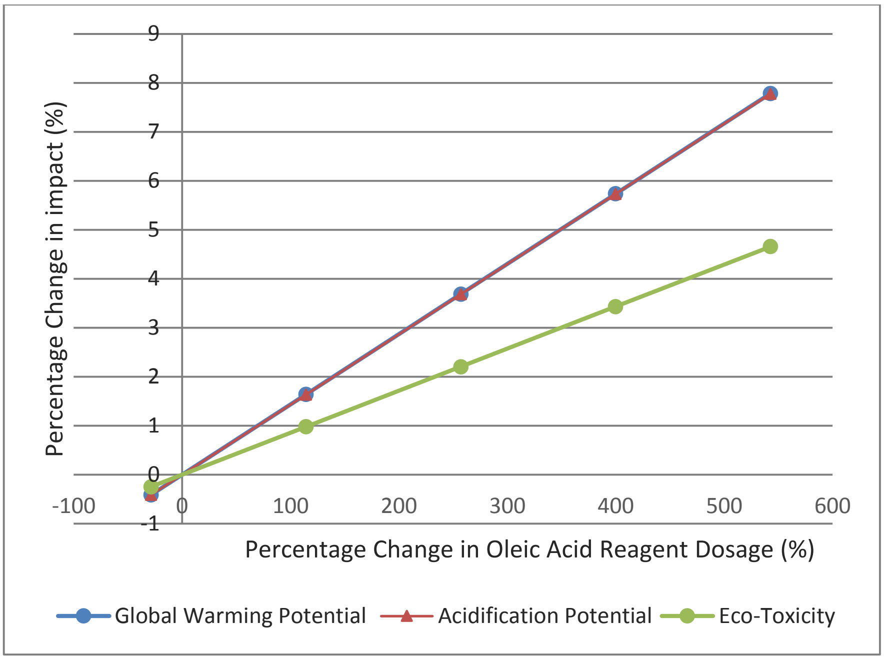

Figure 46 Effect of variation of oleic acid reagent dosage on climate change, terrestrial acidification and eco-toxicity

5.3.4 Effect of metal availability

The available concentrations for the major metal elements were subjected under a sensitivity analysis because depending on the site conditions, specifically the pH in the slurry dam, more or less of the metals might be leached into the water sources. Furthermore, since this case study was associated with relatively low sulfide-sulfur yields (approximately 25%) to the rich-sulfide stream (highlighted in Chapter 3), hence higher deportment of the sulfide associated metals, particularly iron to the sulfide-lean stream, it is important to note how the metal risk indicators will be affected in the event of a good separation which subsequently means less available metal concentration in the final disposed waste stream. Aluminium and iron were selected due to their significant contributions to the total metal risk potentials.

Iron Availability

Figure 47 represents how the aquatic water pollution and drinking water quality risk potential factors vary with varying available iron concentration. The effect of varying available iron concentration on eco-toxicity was not conducted as the characterisation factor for iron was unavailable as was mentioned in Chapter 3. Since the risk potential factor equation is in a quadratic form, the relationship of percentage change in the total of risk potential factors versus available iron concentration follows a curve. The variation of drinking water quality risk with varying available iron concentration is similar to that of aquatic water pollution risk, with an increase in the available iron concentration by 66% resulting in an increase of 132 % in the aquatic water pollution risk impact category and an increase of 156% in drinking water quality risk category. Differences occur due to the variations in the acceptable human exposure concentration and the acceptable aquatic concentration of iron.

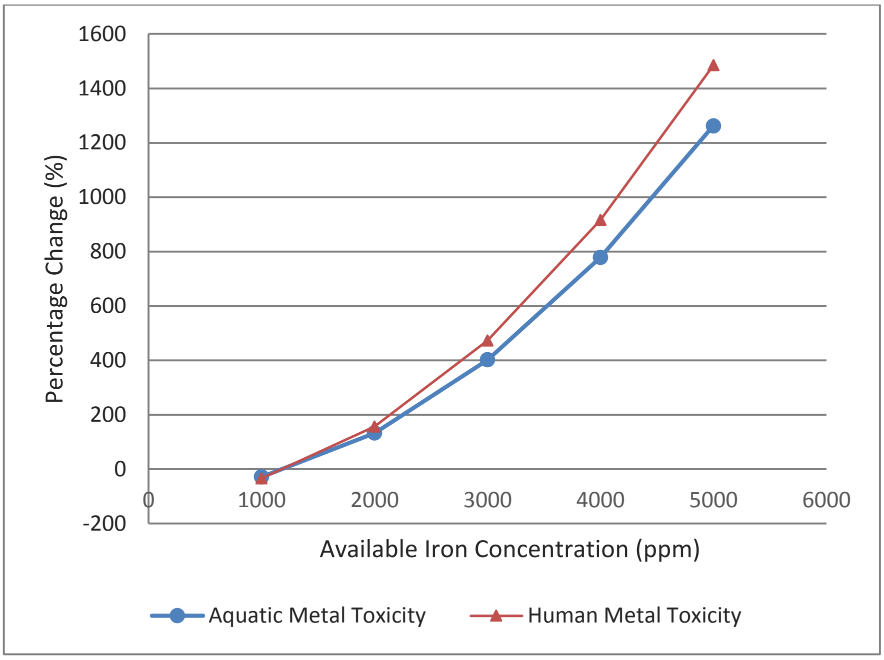

Figure 47 Effect of varying available iron concentration in the disposed waste on the aquatic and human metal toxicities

Aluminium availability

The available iron concentration was varied from 1000 ppm to 5000 ppm, a range that encompasses a deviation from the base of -20% to 300%.The effect of varying the available aluminium concentration in the disposed waste stream on the aquatic water pollution risk , drinking water quality risk and eco-toxicity impacts is represented by Figure 48. As can be seen from the graph, the impact category mostly affected by this variation is the aquatic water pollution risk followed by the drinking water quality risk impact category and then the eco-toxicity impact category. Increasing the available aluminium concentration by 55% in the disposed waste stream increases the aquatic water pollution risk impact by 25%, increases the drinking water quality risk impact category by 4% and increases the eco-toxicity impact category by only 0.5%. This confirms the findings of the impact assessment (presented in Chapter 4) which showed that the aluminium contributes a significant percentage to the aquatic water pollution risk and a very small percentage to the drinking water quality risk and eco-toxicity impact category.

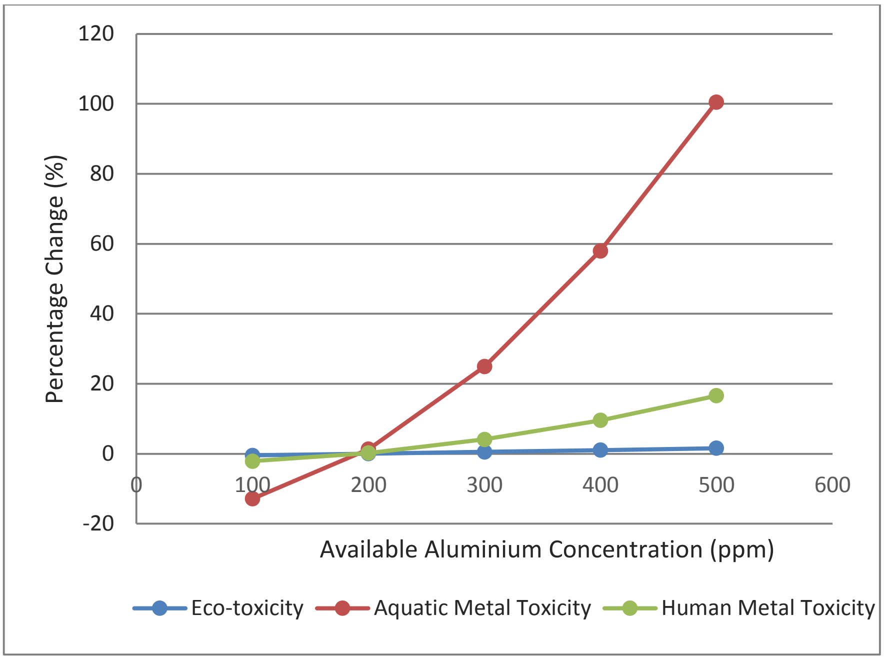

Figure 48 Effect of varying the available aluminium concentration in the disposed waste on the eco-toxicity, aquatic water pollution risk and drinking water quality risk.

5.4 Environmental implications of the implementation of the two-stage flotation process in the South African context

In order to understand the overall environmental performance of the two-stage flotation process, it is paramount to evaluate the environmental implications of the implementation of the two-stage flotation process in the South African context. The results in this study show that the implementation of the two-stage flotation process in all coal beneficiation plants will result in a significant recovery of coal (approximately 1.2 million tonnes for every 4 million tonnes dry coal ultrafines lost per annum) which can be utilised for electricity production. Comparing the total coal that is mined annually in South Africa (approximately 80 Mt) to the recovered coal from the two-stage flotation process, the recovered coal constitutes approximately 1.5% of the coal produced annually (presented in Chapter 1). Whilst this figure is subject to change with varying coal yield, it forms quite a significant percentage of the total coal produced annually. Furthermore, taking into account the fact that the case study was associated with relatively low coal recoveries compared to previous studies, this percentage is likely to be higher than 1.5% in most cases.

The results also show that the implementation of the two-stage flotation will result in less water losses compared to the current coal waste management practices (discussed in Section 2.1.7). As was shown in the previous sections, the two-stage flotation process is associated with a consumptive water footprint 87% less than the base case with the two-stage flotation process managing to recover approximately 96% of the water in the coal ultrafine slurry. More specifically, implementation of the process in all coal beneficiation plants would result in the recovery of an estimated 7 million tonnes of the 8 million tonnes of water lost through the disposal of ultrafine coal slurry annually (refer to Appendix F for detailed calculations). Since it has been highlighted in Chapter 2 that South Africa is a water scarce region, this is beneficial in the South African context.

Still on the local water resources, implementation of the two-stage flotation process would lessen all the water-related impacts associated with the current coal waste management methods which include salinization, metal contamination and aqueous acidification. The implementation of the two-stage flotation could however result in the release of toxic flotation reagents into local water sources. Even though the overall freshwater eco-toxicity of the two-stage flotation process is less than that of the base case, it has been shown that the presence of the residual SIBX in the final disposed waste stream has a significant contribution towards the eco-toxicity impact of the process.

Lastly, the results in Section 4.2.2 also indicate that the implementation of the two-stage flotation process would result in an increase in climate change and terrestrial acidification impacts. As was highlighted in Chapter 4 and previous sections, this is due to the high fossil fuel based electricity consumption of the process. However, comparing the greenhouse gas emissions associated with implementation of the two-stage flotation process with the annual greenhouse gas emissions in South Africa, the two-stage flotation process contributes approximately 0.025% to these emissions (refer to Appendix F for detailed calculations). Whether this percentage is significant or not remains subjective however it still serves to provide another gauge on the environmental performance of two-stage flotation process in the South African context. Furthermore, recovery of the coal ultrafines would translate to the recovery of 4 600 TJ[1] of energy per annum based on the approximated calorific value of coal ultrafines. Comparing this energy to the energy consumed during the process, the energy consumed by the process forms approximately 1% of the energy recovered. Overall, this suggests that the implementation of the two-stage flotation process would promote enhanced efficiency of mined resources in South Africa.

5.5 Limitations of the study

The following are the limitations of the study. The limitations were mainly associated with the data input and output quality, impact categories and the system boundary and scenario development.

5.5.1 Quality of input data

The lack of adequate and reliable experimental data prevented a detailed study of uncertainty and required a number of overarching assumptions to be made on the basis of general literature information and in-house expertise which compromised the quality of the input data. More specifically, the lack of analysis of metals such as Hg, Se and Sb in the disposed waste stream and the experimental anomalies in the concentrations of manganese and cobalt in the disposed waste streams presented notable limitations. Furthermore, only one case study was taken into account, and one that was far from optimum. Previous studies have shown that the compositions of coal wastes and the subsequent coal yields and extent of sulfide sulfur removal during application of the two stage flotation process vary quite considerably which would have a significant impact on the environmental performance of the process.

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[1] Using a basis of 100 ton/hr

5.5.2 Impact categories

The limitations associated with impact categories were due to the lack of characterisation factors for some of the main substances in the system. An example is the exclusion of the human-toxicity impact category due to the lack of human-toxicity characterisation factors for the flotation reagents and some of the metals and reagents (Chapter 3). Another example is the exclusion of iron in the freshwater eco-toxicity impact category due to lack of a characterisation factor in the eco-toxicity impact assessment model used (developed by Goedkoop et al. (2009)). This presented a notable limitation to the study as iron was shown to be one of the major elements in the disposed waste streams of both scenarios. Although the application of risk-based approaches addressed this deficiency to some extent, the lack of documented acceptable concentrations for uranium, titanium and vanadium in aquatic systems and drinking water rendered them limited.

5.5.3 System boundary and scenario development

The exclusion of the possibility of the utilisation of the sulfide-lean tailings stream and the assumption of readily available sulfide-lean market in the two-stage flotation process scenario development presents another limitation to the study. Downstream utilisation will further improve resource utilisation and potentially reduce local water-related impacts. However it may also result in additional consumption of energy and other materials. The implications and trade-offs of the downstream utilization are still not well understood as the processes are largely still in the development phase. Furthermore, to increase the accuracy of the assessment, more detailed background information such as the energy used in the transportation of reagents could have been incorporated into the datasets. However because of lack of available data, this could not be done. Lastly, the assessment did not include other tailings management scenarios (presented in Chapter 2) that could possibly inhibit the formation of acid rock drainage.

CHAPTER 6

CONCLUSIONS AND RECOMMENDATIONS

The research study aimed at conducting a holistic assessment of the environmental implications of the implementation of the two stage flotation process. More specifically, the project aimed at evaluating the environmental burdens associated with the process and identifying key parameters that affect the process. The research study was carried out in order to address the following key questions:

i. How does the environmental waste burdens of the disposal of untreated coal ultrafines compare to those of the two-stage flotation process?

ii. How is the environmental performance of the two-stage flotation process affected by the variation of input parameters?

iii. Which processes associated with both of the coal waste management scenarios contribute the most to the environmental burdens of the scenarios?

iv. What are the key environmental implications associated with implementation of the two-stage flotation process in the South African context?

The research study evaluated these outcomes by applying a life cycle approach to two scenarios, one being the two-stage flotation process and the other being the disposal of coal ultrafines into a slurry dam. The impact categories that were evaluated in the research study included climate change, terrestrial acidification, freshwater ecotoxicity, salinity, aquatic water pollution risk , drinking water quality risk, aqueous acidification, fossil fuel depletion, water footprint and natural land transformation. The following are the key research outcomes that were obtained from the project.

6.1 Case study outcomes

6.1.1 Comparison of the two-stage flotation process to the base case

The case study results have indicated that the implementation of the two-stage flotation process for the recovery of coal and removal of sulfide sulfur from ultrafine coal waste results in a significant decrease in water quality related impacts (eco-toxicity impacts, salinity impacts, consumptive water footprint, aquatic and drinking water quality risks and aqueous acidification) as well as resource related impacts (fossil fuel depletion and natural land transformation impacts). However, the results have indicated an increase in atmospheric related impacts (climate change and terrestrial acidification impacts), which has been attributed to the additional energy consumption associated with the two-stage flotation process and the production processes associated with the flotation reagents. The impact categories exhibiting the highest relative percentage differences between the base case and the two-stage flotation process where aqueous acidification, water footprint, salinity, climate change and terrestrial acidification impact categories.

6.1.2 Process contributions to impacts and sensitivity analyses

Analyses of the process contributions to the individual impact categories for the two-stage flotation process revealed the climate change and terrestrial acidification impact categories to be dominated by the electricity production process and the flotation reagents production process, with the flotation reagents production process showing a more significant contribution in the terrestrial acidification impact category. The rest of the impact categories were mostly dominated by the actual two-stage flotation process solid waste management process, with the electricity production and reagent production processes contributing slightly to the fossil fuel depletion and salinity impact categories. The sensitivity analyses revealed a higher dependence of the fossil fuel depletion impact category on the percentage coal yield than the electricity consumption of the foreground process. Furthermore the sensitivity analyses indicated a strong dependence of the climate change and terrestrial acidification impacts on the electricity consumption of the foreground process and the SIBX dosage in the foreground process. Comparing the effect of varying the oleic acid dosage to the effect of varying the SIBX dosage, variation of the SIBX dosage resulted in significantly higher percentage changes in eco-toxicity, climate change and terrestrial acidification than the variation of the oleic acid dosage. Variation of the available iron and aluminium concentrations in the final disposed sulfide-lean stream demonstrated exponential increases in the aquatic and drinking water quality risk impact categories.

6.1.3 Key environmental implications associated with the two-stage flotation process in the South African context

Implementation the two-stage flotation process would result in a 0.025% increase in the annual greenhouse gas emissions in South Africa however it would result in a significant recovery of coal and a sulfide-rich product which can be utilised for electricity production and sulfuric acid production respectively hence promoting resource efficiency. The energy used in the two-stage flotation process is infinitesimal compared to the energy recovered in the process. The implementation of the two-stage flotation would also result in less water losses which is beneficial in the South African context as South Africa is a water scarce region. Lastly whilst the implementation of the two-stage floatation process would result in a reduction in salinization, aqueous acidification risk and metal pollution it might pose a threat to aquatic life if the xanthate salt reagents are emitted to local water sources.

6.2 Concluding remarks

Using a combination of a life cycle approach and risk based indicators, an analysis and comparison of the environmental benefits and burdens of the two stage flotation process for the recovery of coal and removal of sulfide sulfur from ultrafine coal wastes was successfully conducted. This study is consistent with the principles of environmental sustainability in that it takes into account both the environmental impacts on the earth's natural resources (air, water, land and minerals) and their effective utilisation, and includes those impacts associated with the additional process input requirements such as energy and materials

The study revealed the benefits of using a combination of conventional LCA approaches with risk-based tools, to address some of the shortcomings associated with conventional LCA impact categories and characterisation methodologies, and to derive more detailed information on the environmental risks associated with solid wastes. The scenario study successfully evaluated the significance of these environmental benefits and burdens in the South African context, with particular emphasis on utilisation of mined coal, water quality and utilisation and energy consumption. Furthermore the sensitivity analysis reflected a strong dependence of the fossil fuel depletion impact on the coal yield of the process and a strong dependence on the terrestrial acidification impact on the SIBX dosage used in the process.

The information derived from this study serves as a basis for motivation and justification for the larger-scale testing and implementation of the two-stage flotation process for the management of fine coal waste, whilst providing guidance and a framework for further optimisation studies.

6.3 Recommendations for further work

This study has indicated that further research and development is required to both optimise the environmental performance of the two-stage flotation process, and improve the reliability and accuracy of the environmental performance assessment, as well as its relevance.

6.3.1 Optimisation of the two-stage flotation process

Since the potential increase in climate change and terrestrial acidification has been attributed to the high energy consumption of the two-stage flotation process, future work can be dedicated to identifying areas in which this consumption can be lowered or investigating the use of low greenhouse gas emitting energy sources. An example is the implementation of more efficient flotation cells as the flotation circuit contributed a significant amount to the total energy consumption.

Another recommendation would be to investigate the use of other less toxic sulfide flotation reagents than xanthate salts as well as reagents that are associated with environmentally friendly production processes since this study indicated that the use of xanthate salts contributes significantly to the total eco-toxicity and terrestrial acidification impacts of the two-stage flotation process.

6.3.2 Improvement of the environmental performance assessment

The environmental performance assessment can be improved by conducting more comprehensive studies on a variety of different ultrafine coal samples. The assessment can also be improved by evaluating the environmental performance of the utilization of the coal recovered from the two-stage flotation process for electricity production and comparing it to the utilization of coal mined from the ground. Furthermore the assessment can be extended to include downstream processing of the sulfide-lean and sulfide-rich streams. More investigations can also be conducted on the availability and toxicity of metals and reagents used in the two-stage flotation process. Lastly, a sensitivity analysis could be conducted on different percent leaching of wastes in lieu of different management scenarios

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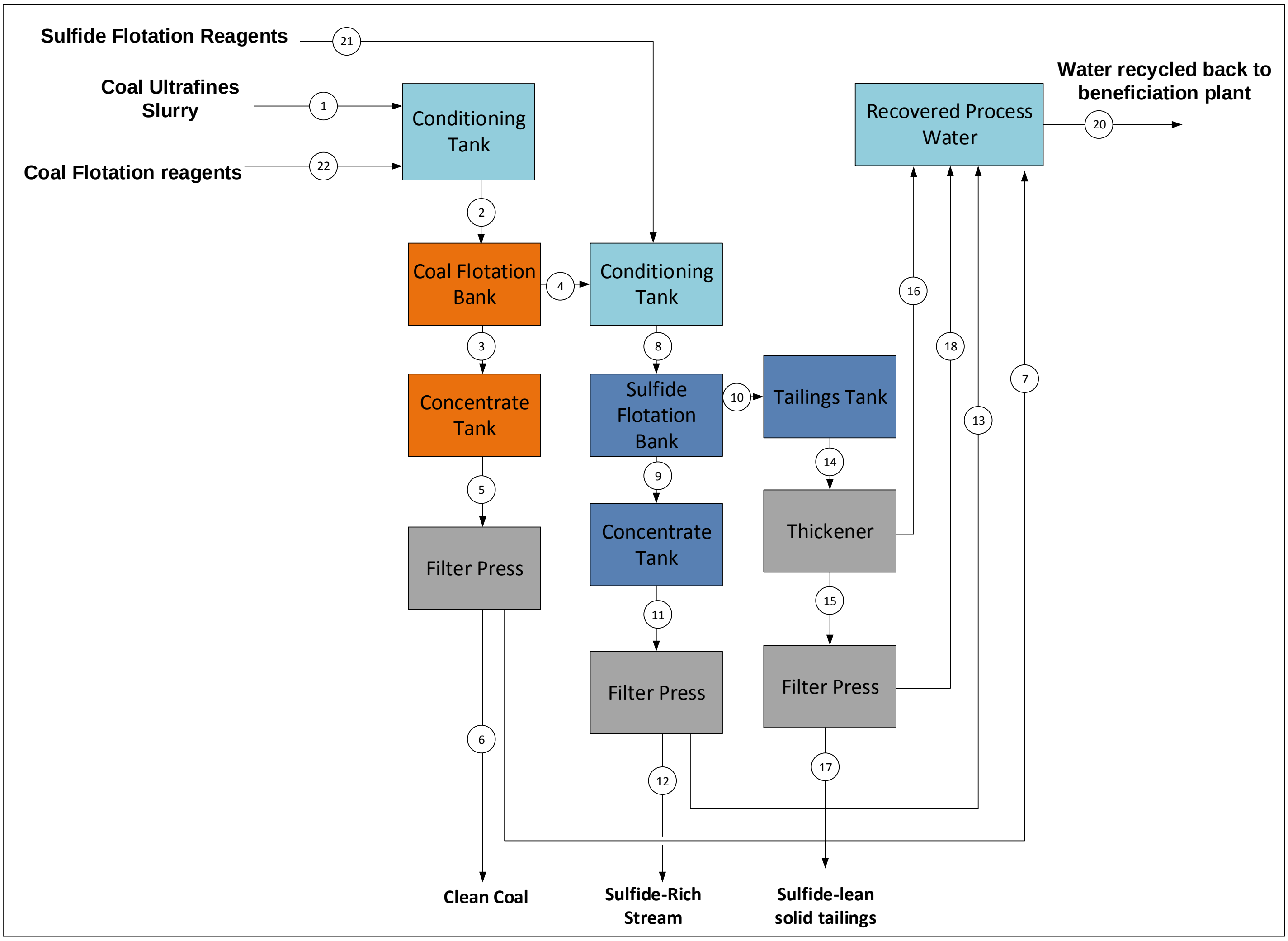
APPENDICES

Appendix A

Mass Balance Calculations

The Two Stage Flotation Process



Block flow diagram of the two stage flotation process

Detailed Mass Balance of the two stage flotation process

Stream	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	21	22	23
Solids [t/h]	100.0	100.0	30.2	69.8	30.2	30.2	0.0	69.8	2.0	67.8	2.0	2.0	0.0	67.8	67.8	0.0	67.8	0.0	0.0	0.163	
Water [t/h]	900.0	900.0	120.8	779.2	120.8	9.2	111.6	779.2	8.0	771.2	8.0	0.6	7.4	771.2	101.7	669.5	20.8	80.9	869.4		
Slurry [t/h]	1000.0	1000.0	151.0	849.0	151.0	39.4	111.6	849.0	10.0	839.0	10.0	2.6	7.4	839.0	169.5	669.5	88.6	80.9			
Slurry [m³/h]	971.4	971.4	142.4	829.1	142.4	30.8	111.6	829.1	9.4	819.6	9.4	2.0	7.4	819.6	150.1	669.5	69.2	80.9			
Slurry density [t/m³]	1.03	1.03	1.06	1.02	1.06	1.28	1.00	1.02	1.06	1.02	1.06	1.28	1.00	1.02	1.13	1.00	1.28	1.00			
Solids volume fraction	0.07	0.07	0.15	0.06	0.15	0.70	0.00	0.06	0.15	0.06	0.15	0.70	0.00	0.06	0.32	0.00	0.70	0.00			
Solids mass fraction	0.10	0.10	0.20	0.08	0.20	0.77	0.00	0.08	0.20	0.08	0.20	0.77	0.00	0.08	0.40	0.00	0.77	0.00			
Ore Density [t/m³]	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40	1.40			
Species balance [t/h]																					
Sulfide Sulfur	1.07	1.07	0.344080	0.73	0.34	0.34		0.73	0.26	0.47	0.26	0.26		0.47	0.47		0.47				
Sulfates	0.03	0.03	0.007758	0.02	0.01	0.01		0.02	0.01	0.01	0.01	0.01		0.01	0.01		0.01				
Oragnic Sulfur	0.82	0.82	0.452060	0.37	0.45	0.45		0.37	0.30	0.07	0.30	0.30		0.07	0.07		0.07				
Ash	49.08	49.08	7.350000	41.73	7.35	7.35		41.73	0.98	40.75	0.98	0.98		40.75	40.75		40.75				
Fixed Carbon and Volatile matter	49.00	49.00	22.046102	26.95	22.05	22.05		26.95	0.45	26.50	0.45	0.45		26.50	26.50		26.50				
Oleic Acid																					0.28
Sodium Isobutyl Xanthate		0.00	0.00	0.00	0.00	0.00		0.16	0.158	0.0049	0.16	0.16		0.0049	0.0049		0.00			0.16	
Total	100.0000	100.0000	30.2000	69.80	30.2000	30.2000		69.80	2.1578	67.8049	2.1578	2.1578		67.80	67.8049		67.8049			0.16	

The two stage flotation laboratory results (Iroala, 2014)

Stream name	Feed	Coal Concentrate	First Stage Tailings	Sulfide Rich Stream	Sulfide Lean
Total Solids Flow (kg/hr)	100	30.2	69.8	2	68
Sulfide Sulfur	1.07	0.344	0.726	0.257	0.469
Sulfates	0.0280	0.00776	0.0202	0.00716	0.0131
Oragnic Sulfur	0.822	0.452	0.370	0.302	0.0680
Total Sulfur	1.92	0.804	1.12	0.566	0.550
Ash	49.1	7.35	41.7	0.982	42.4
Fixed Carbon and Volatile matter	49.0	22.0	27.0	0.453	25.1

Reagent dosages for the two stage flotation process (Iroala, 2014)

Reagent		Dosage (kg/t)
1st stage- Coal Flotation	Nalflote 9858	1.4
	MIBC	0.11
2nd stage - Sulfide flotation	Sodium Isobutyl Xanthate	2.33
	Methyl Isobutyl Carbinol	0.28
	Dextrin	0.93

Major assumptions for the two stage flotation process mass balance

Stream	Calculations	Assumptions/ Basis of calculation
	Water Input = $M_{solids(1)}$*90%/10% $M_{solid\ species}$ = (% contribution of solid species X Total solids flow)	• Basis of 100 t/h solids flow • Feed comes in at 10% solids mass percent • Ore Density= 1.4 t/m3 (Wills and Napier-Munn, 2006) • Amount of ash in coal ultrafines based on table 5 (Iroala, 2014) • Sulfide sulfur, sulfate and organic sulfur is based on the sulfur speciation done on the feed sample

Stream	Calculations	Assumptions/ Basis of calculation
First Stage Flotation Feed	Total Mass= Make up water + Reagents + Feed Coal Ultrafines	- Solids flow into the flotation cells is 10% (Jera, 2013)
1st stage Flotation Concentrate	M concentrate = (% Recovery* Flotation Feed)	- % Recovery based on Iroala (2014) - Solid species compositions based on table 1 (Iroala, 2014)
1st stage tailings	Total Mass= $M_{\text{flotation feed}}$ - $M_{\text{flotation concentrate}}$ M solid species = (% contribution of solid species X Total solids flow)	- No mass is lost during the collection of the concentrate - Solid species compositions based on table 1 (Iroala, 2014)
Filtered water (Product Filtration)	$M_{\text{filtered water}}$ = $M_{\text{filtered feed}}$- $M_{\text{Moisture Content of coal product}}$	- Filtration unit filters product until it contains 20% moisture content (Jera, 2013)
		- The mass of solids present in the filtered water is negligible
2nd Stage flotation concentrate	Total Mass= %Yield X Flotation Feed M solid species = (% contribution of solid species X Total solids flow)	- Based on table 1 (Iroala, 2014) - Solid species compositions based on table 1 (Iroala, 2014)
2nd Stage Tailings	Total Mass= $M_{\text{flotation feed}}$ - $M_{\text{flotation concentrate}}$ M solid species = (% contribution of solid species X Total solids flow)	- No mass is lost during the collection of the concentrate - Solid species compositions based on table 1
Sulfide fraction product filtration	Same calculation as coal filtration	- Same specification as for the coal filter press

Stream	Calculations	Assumptions/ Basis of calculation
Thickener underflow/overflow separation	Overflow flowrate = Thickener feed water – underflow water Underflow flowrate = 60%/40% X solids in underflow	- An assumption was made that the thickener produces a slurry with 40% solids composition based on Reddick (2006) and Jera (2013)
Benign Tailings Filtration	$M_{\text{filtered water}} = M_{\text{filtered feed}} - M_{\text{Moisture Content of filtered product}}$	- The moisture content of the end product was taken to be 30% (de Korte, 2008)
Reagent dosages	Reagent flowrate = [dosage (kg/t) X Solids flowrate into flotation cell (t/h)]/1000	- Based on Iroala (2014)
Seepage Composition: **<u>Major and trace metals</u>**	Mass of solid used in NAG test = 2g Volume of liquid used in NAG test= 125 ml (the detailed NAG test protocol is presented after the table) $\frac{mg}{L} \times \frac{volume\ (L)}{initial\ mass\ of\ solid\ (g)} \times \frac{1000\ g}{1\ kg} = \boldsymbol{available\ concentration\ \frac{mg}{kg}}$ Individual metal flowrate = available concentration X $M_{\text{sulfide-lean Stream}}$	- Concentration in $\frac{mg}{L}$ of metals obtained from the ICP-MS analysis
Seepage Composition: **<u>Salt forming anions and cations</u>**	Concentration of potassium and calcium ($\frac{mg}{kg}$) X Percent mobility (%) Total dissolved sulfates = total sulfates in the final disposed stream (all sulfates dissolve)	Derived from Sanyika and Ngcobo (2014); Iroala (2014); Kotelo (2013)
Seepage Composition: **<u>Sulfuric acid</u>**	H+ flowrate (kg) = Net Acid Producing Potential (NAPP) (kg H_2SO_4/t) X Total stream flowrate (tons)	Based on Iroala (2014)

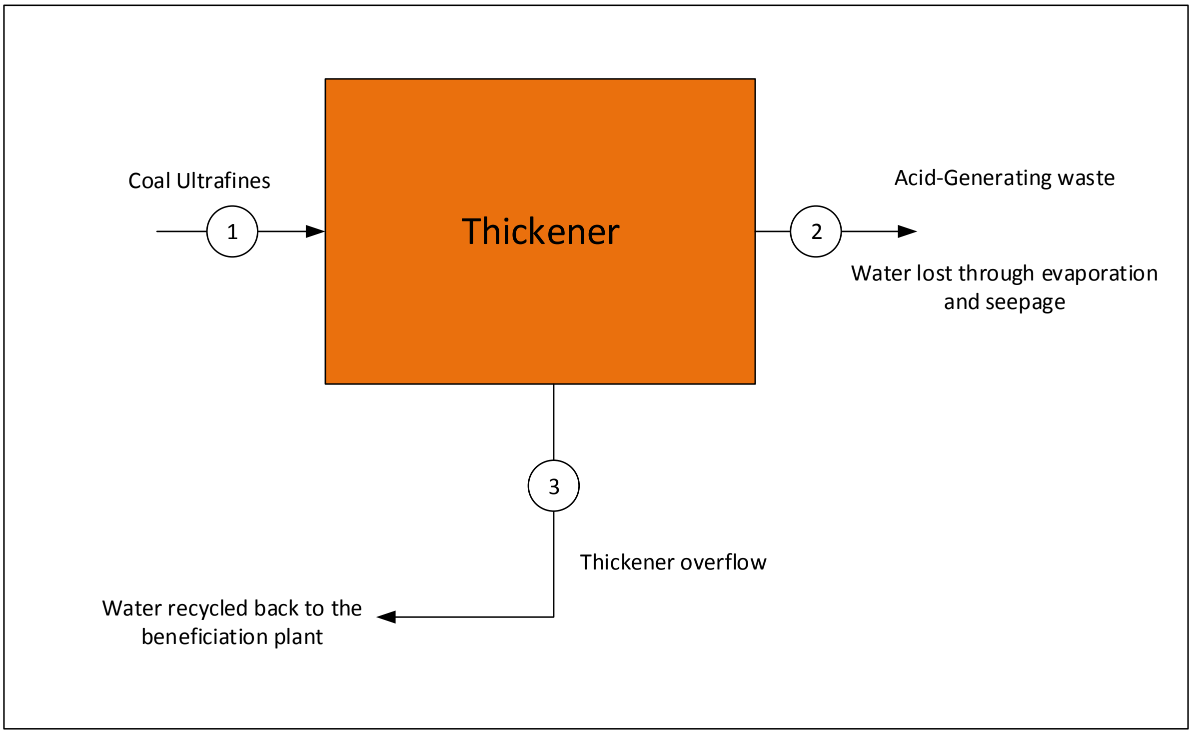
Stream	Calculations	Assumptions/ Basis of calculation
Seepage Composition: **<u>Reagent Present in Seepage</u>**	(100% - Xanthate reagent affinity to material) * Xanthate in Feed = 3% * Xanthate in Feed	Gebhardt et al. (1985) The reagent affinity to material was assumed to be the same for both cases.
	(100% - Oleic acid affinity to material)* Oleic acid in Feed = 3% * Oleic acid in Feed	
Water Losses: **<u>Entrainment</u>**	% Water likely to be entrained * Total Water in Disposed stream= 50% * m_{water}	(Bleiwas, 2012)
Water Losses: **<u>Evaporation</u>**	% Water likely to evaporate * Total water in disposed stream = 40% *m_{water}	
Water Losses: **<u>Seepage</u>**	% Water likely to seep through the ground = 10% * m_{water}	
Unrecovered Coal	The total fixed carbon and Volatile matter in the disposed waste = % Fixed Carbon and Volatile Matter * Total disposed waste.	(Iroala, 2014)
Remaining Gangue Material	Total Solids in Feed Stream – Unrecovered Coal	

NAG test experimental protocol used to generate leachate solutions that were sent in for ICP-MS analysis

b. 1.25 g sample was weighed and placed into a 250ml Erlenmeyer flask. And record mass of flask
c. 125ml of 15% H_2O_2 solution was measured and added to the Erlenmeyer flask
d. The flask was allowed to react for 24 hours in a fume hood.
e. After the reaction, the pre-boil pH was measured to be 5.24 for the feed coal ultrafines and 5.46 for the desulfurised ultrafines.
f. The flask was then placed on a hot plate and gently heated until effervescence stopped.
g. The solution was allowed to cool to room temperature
h. The mass of the flask after the boil was then measured
i. The volume was made up to 125ml by adding de-ionised water.

j. The after-boil pH which is also referred to as NAGpH was measured to be 6.52 and 6.26 for the feed and desulfurised coal ultrafines respectively.
k. Filter, retaining solids residue and liquor.

The Base case



Block flow diagram of the base case scenario

Stream flowrates for the base case

Stream No.	**1**	**2**	**3**
Solids [t/h]	100.0	200.0	0.0
Water [t/h]	900.0	300.0	750.0
Slurry [t/h]	1000.0	500.0	750.0
Slurry [m3/h]	971.4	292.9	750.0
Slurry density [t/m3]	1.03	2.53	1.00
Solids volume fraction	0.07	1.32	0.00
Solids mass fraction	0.10	1.40	0.00
Ore Density [t/m3]	1.40	2.80	1.40
Solid Species balance [t/h]			
Sulfide Sulfur	0.98000	1.96	
Sulfates	0.50000	1.00	
Oragnic Sulfur	0.56000	1.12	
Inert Gangue	51.00000	102.00	
Fixed Carbon and Volatile matter	46.96000	93.92	
Total	100.000	200.00	
Solid Species Mass Fraction		0.0	
Sulfide Sulfur	0.0098	0.0196	
Sulfates	0.0050	0.0100	
Organic Sulfur	0.0056	0.0112	
Inert Gangue	0.5100	1.0200	
Fixed Carbon and Volatile matter	0.4696	0.9392	
Total	1.0000	2.0000	

Mass balance calculations and major assumptions

Stream	Calculations	Assumptions/Basis of Calculations
Feed	Water Input = $M_{solids(1)}$*90%/10% $M_{solid\ species}$ = (% contribution of solid species X Total solids flow)	• Basis of 100 t/h solids flow • Slurry coming from coal washing plant comprises of 10% solids • Ore Density= 1.4 t/m3 (Wills and Napier-Munn, 2006) • Amount of ash in ultrafines based on table 2 (Kazadi-Mbamba, 2012) • Amount of pyrite in Ash is based on the mineralogical composition of ash by Kazadi-Mbamba (2012)

Stream	Calculations	Assumptions/Basis of Calculations
Thickener Underflow	$M_{\text{water in thickener underflow}} = M_{\text{solids(1)}}*60\%/40\%$	- Assuming the thickener removes water until a 40% solids mass fraction is achieved
Recovered Water	$M_{\text{water in overflow}} = M_{\text{water in feed}} - M_{\text{water in thickener underflow}}$	- The solids mass fraction in recovered water was taken to be negligible
Acid Generating Waste	Solids Mass Feed= Solids Waste $M_{\text{water in underflow}}=M_{\text{water in waste}}$	- Mass lost during dewatering was taken to be negligible
Seepage Composition: <u>Major and trace metals</u>	Mass of solid used in NAG test = 2g Volume of liquid used in NAG test= 125 ml $\frac{mg}{L} \times \frac{volume\ (L)}{initial\ mass\ of\ solid\ (g)} \times \frac{1000\ g}{1\ kg} = \boldsymbol{available\ concentration\ \frac{mg}{kg}}$ Individual metal flowrate = available concentration X $M_{\text{sulfide-lean Stream}}$	- Concentration in $\frac{mg}{L}$ of metals obtained from the ICP-MS analysis on the leachate solutions
Water Losses: **<u>Entrainment</u>**	% Water likely to be entrained * Total Water in Disposed stream= 50% * m_{water}	(Bleiwas, 2012)
Water Losses: **<u>Evaporation</u>**	% Water likely to evaporate * Total water in disposed stream = 40% *m_{water}	
Water Losses: **<u>Seepage</u>**	% Water likely to seep through the ground = 10% * m_{water}	
Unrecovered Coal	The total fixed carbon and Volatile matter in the disposed waste = % Fixed Carbon and Volatile Matter * Total disposed waste.	(Iroala, 2014)
Remaining Gangue Material	Total Solids in Feed Stream – Unrecovered Coal	

ICP- MS analysis raw data (leachate volume concentration)

Metal		x	%RSD
27Al	ppm	2.918	0.544
47Ti	ppm	1.527	0.972
51V	ppb	44.63	1.129
52Cr	ppb	23.45	0.584
55Mn	ppb	154.9	0.411
57Fe	ppm	23.70	1.013
59Co	ppb	4.650	2.930
60Ni	ppb	25.33	0.203
65Cu	ppb	35.81	2.017
66Zn	ppb	617.6	1.660
75As	ppb	23.22	14.08
137Ba	ppb	44.96	1.076
208Pb	ppb	19.65	2.174
238U	ppb	1.874	0.154

Energy Consumption Calculations

Process Unit/Stage	Calculation Method	Source(s)
First Stage Concentrate Filter Press	• Filtration Pump Power= (Volumetric Flowrate X Pump Head X Slurry SG)/(1.02 X Pump Efficiency)	Walas (1990); Jera (2013)
Second Stage Tailings Filter press		
Second Stage Concentrate Filter Press		
Thickener	• A generic power rating for most thickener motors was used	
First Stage Coal Flotation unit	• According to Walas (1990), an 8 cell bank with a 4 minute holdup has an energy consumption of 0.6 HP/ft3 of cell. First stage flotation process consists of a 7 cell bank therefore uses 0.525 HP/ft3 • Converting HP/ft3 to KW/m3= 26.334X0.525 HP/ft3	

Process Unit/Stage	Calculation Method	Source(s)
	- Total Energy Requirement = Power Rating/m3 X Total volume of cell bank	
Second Stage Sulfide Flotation unit	The same method was employed for the second stage flotation	
Energy for Air Compression	Energy of air compression (W)= $\frac{wRT}{K} \times ((\frac{P2}{P1})^{K} - 1)$ Where w= R=gas constant T= Temperature k= (µ-1)/(µ-2) µ= Cp/Cv= 1.4 for air at 300K	Perry's Chemical Engineers' Handbook (1999)
Process Water Pump	Filtration Pump Power= (Volumentric Flowrate X Pump Head X Slurry SG)/(1.02 X Pump Efficiency)	Jera (2013)
Fresh Water Pump		
Energy input for the first stage flotation units using Marsden Energy model	Total Power requirement (including regrinder) = (Power requirement) 4.00 KWh/ton X (flotation feed rate) 100 tons= 400 KWh	Marsden (2008)
	Total Power of regrinder= (Power requirement) 0.990 KWh X (flotation feed rate) 100tons = 99 KWh Net energy of flotation unit = 301 KWh % Difference in the calculated energy (using engineering heuristics and using the Marsden energy model = $\frac{472\ KWh - 301\ KWh}{472\ KWh} \times 100 = 36\ \%$	Assuming the power of the flotation regrinder is approximately equal to the secondary crushing power listed in the model
	Total Power of regrinder= (Power requirement) 0.990 KWh X (flotation feed rate) 100tons = 99 KWh Net energy of flotation unit = 180 KWh	

Process Unit/Stage	Calculation Method	Source(s)
	% Difference in the calculated energy (using engineering heuristics and using the Marsden energy model = $\frac{472 - 180.2\ KWh}{472\ KWh} \times 100 = 36\ \%$	

Appendix B

Characterisation Factors

Global Warming Potential Characterisation factors (Goedkoop et al., 2006)

Emission	Characterisation Factor (kg CO2 equivalent/kg)
Carbon dioxide, fossil	1
Dinitrogen monoxide	298
Methane	25
Carbon monoxide, fossil	1.57
Methane, dichloro-, HCC-30	8.7

Terrestrial Acidification Characterisation factors (Goedkoop et al., 2006)

Emission	Characterisation Factor (kg SO2 equivalent/kg)
Sulfur dioxide	1
Nitrogen oxides	0.56
Ammonia	2.45

Fossil Fuel Depletion Potential Characterisation Factors (Goedkoop et al., 2006)

Resource	Characterisation Factor(kg oil equivalent/kg)
Coal, hard from the ground	0.455
Gas, natural, in ground	0.912
Oil, crude, ground	1

Water Footprint Water Stress Indicators (Pfister et al., 2009)

Global Water Stress Indicator	0.602
Regional Water Stress Indicator (South Africa)	0.75

Eco-Toxicity Characterisation Factors

Eco Toxicity	**Characterisation Factor (kg 1,4 Dichlorobenzene equivalent/kg)**
Aluminium	0.054
Iron	-
Zinc	7.52
Chromium	0.902
Manganese	4.41
Cobalt	33
Nickel	98.4
Copper	118

Eco Toxicity	**Characterisation Factor (kg 1,4 Dichlorobenzene equivalent/kg)**
Vanadium	95.6
Arsenic	15.6
Barium	2.69
Lead	0.414
Oleic Acid	2.5E-03
Sodium Isobutyl Xanthate	1.1
Sulfuric Acid	0.0231

Metal background and acceptable concentrations (DWAF, 1996; Broadhurst, 2007)

Metal Toxicity Solid Disposal (Human)	Background Concentration (ppm)	Acceptable Aquatic Concentration (ppm)	Acceptable drinking water/domestic use Concentration (ppm)
Aluminium	81300	0.7	35
Iron	50000	9	0.3
Titanium	0.3	7.31	0.02
Zinc	80	0.83	11
Chromium	100	4.7	0.5
Manganese	1000	0.3	4.9
Cobalt	25	0.97	0.7
Nickel	75	0.75	0.18
Copper	55	0.13	2.3
Vanadium	150	1.3	0
Arsenic	1.8	0.38	0.002
Barium	430	7.8	0.7
Lead	16	0.12	0.02

Environmentally acceptable and background pH in South African waters (DWAF, 1996; Dallas et al., 1994; ANZECC, 2000)

	pH range	Average pH
Environmentally acceptable Aquatic pH	4.5-9	6.75
South African Natural Waters' pH	6.9-9.2	8.05

Salinity characterisation factors (Leske and Buckley, 2002)

Initial Release Compartment	Characterisation Factor (kg Total Dissolved Salts equivalent/kg)
Surface Water	0.165

Land Transformation characterisation factors (Goedkoop et al., 2006)

Natural Land Transformation	**Characterisation Factor**
Transformation from forest	1

Appendix C

Impact Assessment

Process Contributions

Global Warming Potential (kg CO2 equivalent)		Two stage flotation process	Base Case
Electricity Production	Carbon dioxide, fossil (kg)	18249779.27	103680
	Dinitrogen monoxide (kg)	46226.6909	262.62144
	Carbon monoxide, fossil (kg)	2721.954579	15.463872
	Total (kg)	18298727.92	103958.0853
Reagent Production	Carbon dioxide, fossil (kg)	2488253.483	0
	Dinitrogen monoxide (kg)	3854.683877	0
	Carbon monoxide, fossil (kg)	226.9743783	0
	Total (kg)	2492335.141	0

Acidification Potential (kg SO2 equivalent)		Two stage flotation process	Base Case
Electricity Production	Sulfur dioxide (kg)	11257.08312	678.24
	Nitrogen oxides (kg)	60224.27161	191.60064
	Ammonia (kg)	1.390024855	0.019347552
	Total (kg)	71482.74476	869.8599876
Reagent Production	Sulfur dioxide (kg)	118081.9062	0
	Nitrogen oxides (kg)	5021.89372	0
	Ammonia (kg)	5635.764133	0
	Total (kg)	128739.5641	0

Fossil Fuel Depletion	Fossil Depletion 2-stage	Fossil Depletion (Base Case)
Electricity Production (tonnes oil equivalent)	9134.014527	14.977872
Reagent Production (tonnes oil equivalent)	14579.86111	0
Solid Waste Disposal (tonnes oil equivalent)	105628.9475	184609.152
Total (tonnes oil equivalent)	115938.6526	184624.1299

Total Salinity Potential	Two-Stage Flotation Process	Base Case
Solid Waste Disposal	567865.1019	2558304
Electricity Production	111.5025775	0.633464496
Flotation Reagents Production	20256.5649	0
Total	588233.1694	2558304.633

Eco Toxicity (ton 1,4 Dichlorobenzene equivalent/year)	Two stage flotation process	Base Case
Solid Waste Disposal	119113.9761	797318.6324
Electricity Production	0.031826551	0.000167251
Reagents Production	1.838716159	0

Land-Use (m^2 equivalent)	Two-Stage Flotation Process	Base Case
Solid Waste Disposal	41842.28571	61714.28579
Reagent Production	9964.437172	0
Electricity Production	119.4926024	7.85714E-05
Total	51806.72289	61714.28579

Risk Potential Factors

Aquatic water pollution risk (Risk Potential Factor)	Two Stage flotation process	Base Case
Aluminium	0.654524688	0.601388157
Iron	3.0628125	4.388203125
Titanium	-	-
Zinc	4.89061E-05	0.000476787
Chromium	1.36189E-06	1.17001E-06
Manganese	0.00043152	7.998E-05
Cobalt	2.81078E-06	8.91649E-07
Nickel	1.06624E-05	1.14064E-05
Copper	0.0001524	0.000179351
Vanadium	6.93727E-06	1.02145E-05
Arsenic	0.000904264	0.000788258
Barium	4.64014E-07	6.02684E-07
Lead	0.00017366	0.000201105
Uranium	-	-
Total RPFi *1E09	3.71907018	4.9913

Drinking water quality risk (Risk Potential Factor)	Two Stage flotation process	Base Case
Aluminium	2.290836408	2.104858549
Iron	102.09375	146.2734375
Zinc	4.446E-05	0.000433443
Chromium	1.28018E-05	1.09981E-05
Manganese	2.64196E-05	4.89674E-06
Cobalt	3.89494E-06	1.23557E-06
Nickel	4.44267E-05	4.75266E-05
Copper	8.61391E-06	1.01372E-05
Vanadium	-	-
Arsenic	0.149769	0.149769
Barium	5.17044E-06	6.71562E-06
Lead	0.001041961	0.001206633
Uranium	-	-
Total Risk Potential Factor	104.53554	148.52979

Aqueous Acidification (Risk Potential Factor)	Two Stage Flotation Process	Base Case
Sulfuric Acid	0	101264.759

Appendix D

Sensitivity Analyses

Parameter	Percentage Change		
Electricity Consumption (MW)	Global Warming Potential	Acidification Potential	Fossil Fuel Depletion
0.5	-63.01175313	-42.4791158	-1.74061868
1	-38.01110668	-25.62503219	-1.05000796
1.5	-13.01046023	-8.770948583	-0.359397239
2	11.99018622	8.083135025	0.331213481
2.5	36.99083266	24.93721863	1.021824201

Parameter	Percentage Change	
Coal Yield (%)	Fossil Fuel Depletion	Land Transformation
15	55.86917529	22.30669209
25	19.11313667	7.631236767
35	-17.64290196	-7.044218554
45	-54.39894059	-21.71967387
55	-91.15497921	-36.39512919

Parameter	Percentage Change		
Sodium Isobutyl Xanthate dosage (t/hr)	Global Warming Potential	Acidification Potential	Eco-Toxicity
0.1	-4.616817531	-15.66178022	-2.52132289
0.3	10.12499589	34.34842748	19.02131493
0.5	24.86680932	84.35863518	40.56395276
0.7	39.60862274	134.3688429	62.10659058
0.9	54.35043617	184.3790506	83.6492284

Parameter	Percentage Change		
Oleic Acid dosage (t/hr)	Global Warming Potential	Acidification Potential	Eco-Toxicity
0.1	-0.409744778	-0.409279913	-0.245100774
0.3	1.6389791137913	1.637119653	0.980403095
0.5	3.687703006	3.683519219	2.205906963
0.7	5.736426898	5.729918786	3.431410831
0.9	7.785150791	7.776318352	4.656914699

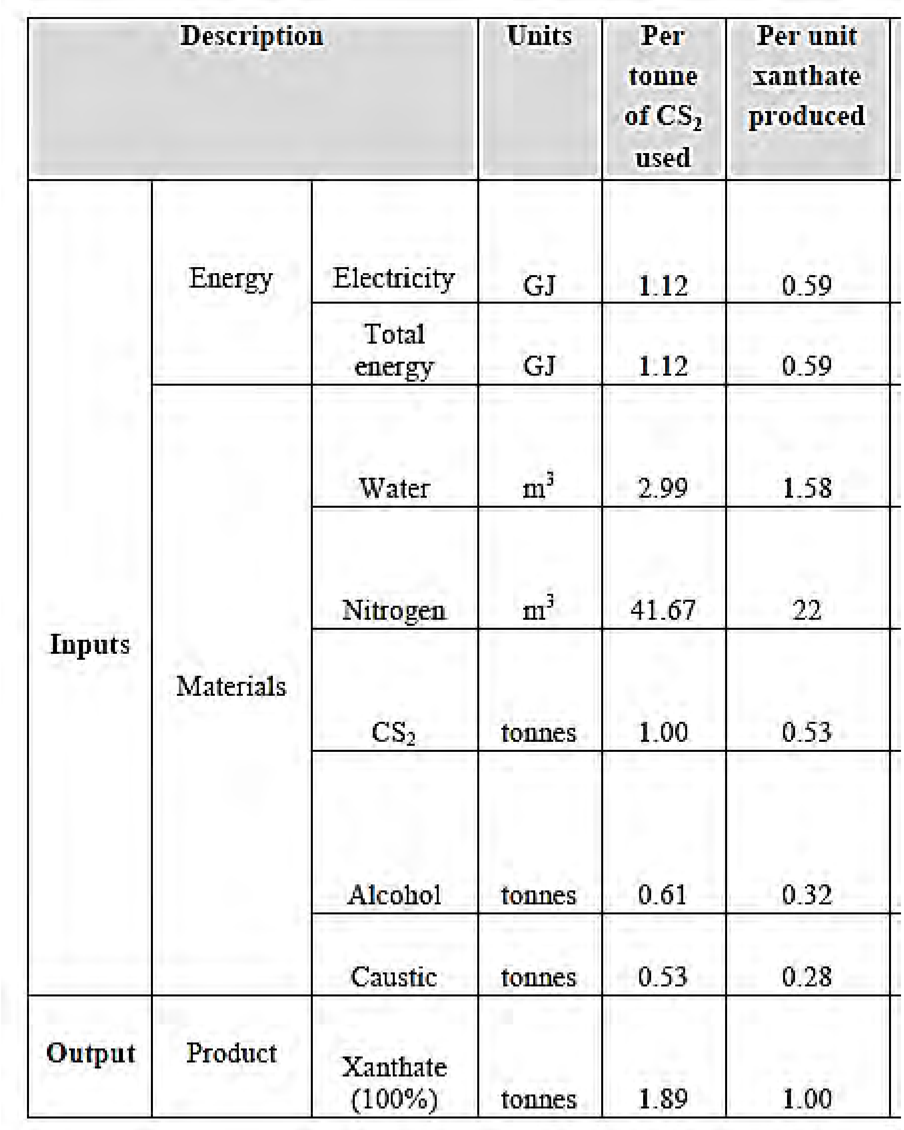
Available Iron Concentration (ppm)	Aquatic water pollution risk (RPF)	Drinking water quality risk (RPF)
1000	-28.57737149	-33.64422367
2000	132.7532761	156.2908232
3000	401.6376889	472.8492347
4000	778.0758667	916.0310109
5000	1262.06781	1485.836152

Available Aluminium Concentration (ppm)	Eco-toxicity (ton 1,4 Dichlorobenzene equivalent/year)	Aquatic water pollution risk (RPF)	Drinking water quality risk (RPF)
100	-0.476293581	-12.87441814	-2.121992658
200	0.035850055	1.29977336	0.214231781
300	0.54799369	24.92342586	4.10793918
400	1.060137326	57.99653936	9.559129538
500	1.572280962	100.5191139	16.56780286

Appendix E

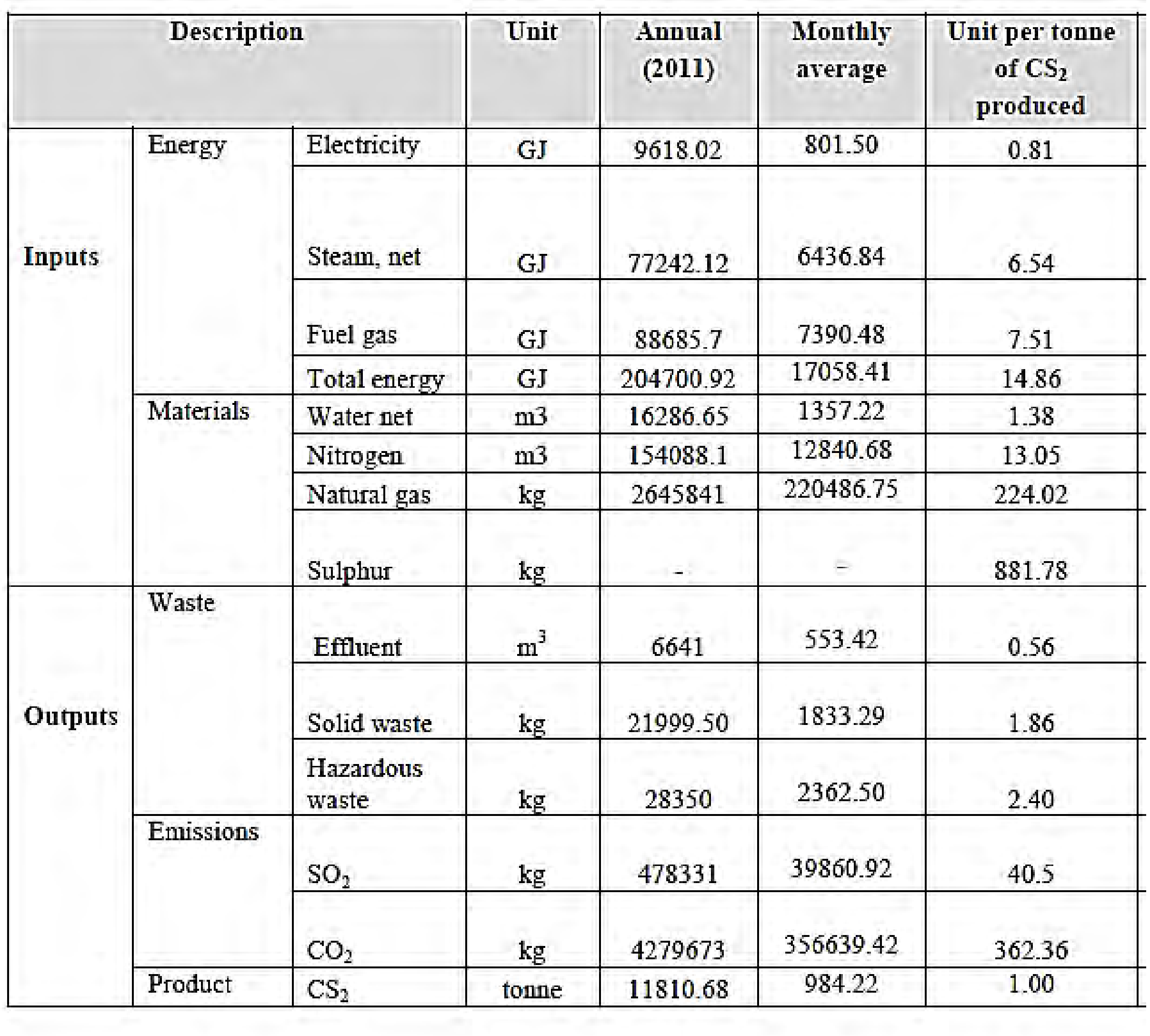
Background Processes

Foreground process of xanthate process (Kunene, 2014)



Description			Units	Per tonne of CS_2 used	Per unit xanthate produced
Inputs	Energy	Electricity	GJ	1.12	0.59
		Total energy	GJ	1.12	0.59
	Materials	Water	m^3	2.99	1.58
		Nitrogen	m^3	41.67	22
		CS_2	tonnes	1.00	0.53
		Alcohol	tonnes	0.61	0.32
		Caustic	tonnes	0.53	0.28
Output	Product	Xanthate (100%)	tonnes	1.89	1.00

Background process of xanthate production: Production of carbon disulphide process (Kunene, 2014)



Description			Unit	Annual (2011)	Monthly average	Unit per tonne of CS_2 produced
Inputs	Energy	Electricity	GJ	9618.02	801.50	0.81
		Steam, net	GJ	77242.12	6436.84	6.54
		Fuel gas	GJ	88685.7	7390.48	7.51
		Total energy	GJ	204700.92	17058.41	14.86
	Materials	Water net	m3	16286.65	1357.22	1.38
		Nitrogen	m3	154088.1	12840.68	13.05
		Natural gas	kg	2645841	220486.75	224.02
		Sulphur	kg	-	-	881.78
Outputs	Waste	Effluent	m^3	6641	553.42	0.56
		Solid waste	kg	21999.50	1833.29	1.86
		Hazardous waste	kg	28350	2362.50	2.40
	Emissions	SO_2	kg	478331	39860.92	40.5
		CO_2	kg	4279673	356639.42	362.36
	Product	CS_2	tonne	11810.68	984.22	1.00

Appendix F

Additional Calculations

a) Comparison of two-stage flotation process to annual climate change in South Africa =

$$\frac{Greenhouse\ gas\ emissions\ associated\ with\ the\ two\ stage\ flotation\ process}{Greenhouse\ gas\ emissions\ in\ South\ Africa\ annually} \times 100$$

Greenhouse gas emissions annually in South Africa = 333 363.70 Gg CO_2 equivalent, according to the Department of Environmental Affairs and Tourism (2009)

Greenhouse gas emissions if the two-stage flotation process was implemented in all coal beneficiation plants = 8.46 X 10^{07} kg $CO_{2\ equivalent}$

$$\% = \frac{8.46\ \times\ 10^{07}}{333\ 363.7\ \times\ 10^{06}}$$

$$= 0.0253\%$$

b) Calculation of total calorific value of the recovered coal ultrafines =

$$Calorific\ value\ of\ coal\ ultrafines\ \left(20\frac{MJ}{kg}\right) \times Total\ recovered\ coal\ ultrafines$$

c) Calculation of the percentage the recovered coal constitutes the coal mined annually in South Africa =

$$Coal\ mined\ annually = 80\ \times 10^{06}\ t$$

$$Dry\ Ultrafine\ Coal\ lost\ annually\ = 12\ million\ tonnes \times 30\%$$

$$Coal\ that\ can\ be\ potentially\ recovered = 12\ million\ tonnes\ \times 30\%\ \times 30.2\%$$

$$\% = \frac{1.208\ \times 10^{6}}{80\ \times 10^{06}} \times 100$$

$$= 1.51\%$$

d) Calculation of water lost due to the disposal of ultrafine coal slurry =

Ultrafine coal slurry disposed annually = 12 million tonnes

Water lost due to disposal of coal ultrafines = 70% *12 million tonnes

8 million tonnes

Water that can be recovered in the event of implementation of the two-stage flotation process = 96% * 8 million tonnes

= 7.7 million tonnes