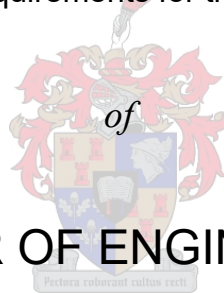# Life cycle assessment of process strategies for value recovery and environmental mitigation of mining waste

by

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Thesis presented in partial fulfilment
of the requirements for the Degree

of

MASTER OF ENGINEERING
(CHEMICAL ENGINEERING)

in the Faculty of Engineering
at Stellenbosch University

Supervisor
Dr Margreth Tadie

December 2020

Declaration

By submitting this thesis electronically, I declare that the entirety of the work contained therein is my own, original work, that I am the sole author thereof (save to the extent explicitly otherwise stated), that reproduction and publication thereof by Stellenbosch University will not infringe any third party rights and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

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i

Abstract

By 2031, South African primary ore gold grades are forecasted to decline to the gold grades expected in mine tailings resources. Furthermore, the reprocessing of mine tailings does not require the costly excavation and size reduction unit operations necessary for primary ore resources. Mine tailings can therefore be viewed as a secondary gold resource. Hazardous pollutants and acid mine drainage (AMD) emanating from Witwatersrand stockpiled tailings dams affect human and ecosystem health. Potential exists to valorise mine tailings to the circular economy as construction raw materials and mine backfill. Sequestration of toxic compounds from mine tailings is however necessary to avert the promotion of environmental impacts up the consumer value chain. Process flowsheets proposed in literature for gold recovery from mine tailings have neglected the evaluation of life cycle impacts of technologies that need to be evaluated before they can be presumed to be "green" alternatives.

Life cycle assessment (LCA) was identified as an environmental impact assessment tool for assessment of ecological impacts of gold recovery process flowsheets. Outotec® HSC chemistry and Thinkstep's GaBi® was identified as software solutions to conduct LCA with the ReCiPe® 2016 life cycle impact assessment (LCIA) methodology. LCA for gold recovery from mine tailings successfully identified environmental hotspots in process flowsheets. The LCA model predicted that the thiosulphate flowsheet reduced environmental impacts for 18 out of 19 impact categories compared to the cyanide flowsheet, apart from freshwater consumption impact category. Electricity consumption during cyanide destruction and emissions from the conventional cyanide flowsheet were recognized as reasons for the increased environmental impacts compared to the thiosulphate flowsheet. Suggestions to further reduce environmental impact based on LCA results were made.

Cyanide and thiosulphate leaching process flowsheet solutions were identified for recovery of gold and ammonium diuranate (yellow-cake uranium) from a hypothetical scenario of West Rand, Witwatersrand mine tailings resource. Environmental impact of mine tailings was reduced. Sulphides and uranium in mine tailings were reduced from 0.18% to 0.03% (82% reduction) and from 54.9 g/tonne to 13.0 g/tonne respectively (75% reduction), thereby decreasing environmental impacts related to acid mine drainage and uranium radionuclide emissions. The processes proposed reduce heavy metal emissions and are not compliant with National Environmental Management: Waste act 2008 (Act No. 59 of 2008) (NEMA) for the protection of water resources. The exceptions are manganese and copper emissions for the thiosulphate flowsheet and manganese, copper and lead emissions for the cyanide flowsheet. Subsequent mine tailings remediation strategies are necessary to render mine tailings inert according to NEMA. LCA provides a basis for environmental impact assessment, but social- and economic- assessments are necessary to determine the viability of process flowsheets proposed and ensure sustainable development in the mineral processing industry.

Opsomming

In Suid-Afrika word die goudgraad van primêre goudertshulpbronne voorspel om te verminder na die goudgraad in mynuitskothulpbronne teen 2031. Verder vereis die herprosessering van mynuitskot nie die duur uitgrawings en grootte reduksie eenheidsbedrywighede wat nodig is vir primêre ertshulpbronne nie. Mynuitskot kan daarom gesien word as 'n sekondêre goudhulpbron. Gevaarlike besoedeling en suurmyndreinering (AMD) wat van Witwatersrand se opgegaarde uitskotdamme vloei affekteer mens- en ekosisteemgesondheid. Potensiaal bestaan om mynuitskot te valoriseer na die sirkulêre ekonomie as roumateriaal vir konstruksie en mynterugvulsel. Sekwestrasie van toksiese samestellings van mynuitskot is egter noodsaaklik om die bevordering van die omgewingsimpak by die verbruikerswaardeketting op, te verhoed. Prosesvloeikaarte voorgestel in literatuur vir goudherwinning van mynuitskot het die evaluasie van lewensiklusimpakte van tegnologieë nagelaat wat geëvalueer moet word voor hulle aangeneem kan word as "groen" alternatiewe.

Lewensiklus assessering (LCA) is geïdentifiseer as 'n omgewingsimpak assessering instrument om goudherwinningsprosesvloeidiagramme se ekologiese impakte te assesseer. Outotec® HSC chemie en Thinkstep se GaBi® is geïdentifiseer as oplossings vir sagteware om LCA met die ReCiPe® 2016 lewensiklus impak assessering (LCIA) metodologie uit te voer. LCA vir goudherwinning van mynuitskot het omgewingsbrandpunte in prosesvloeidiagramme suksesvol geïdentifiseer. Die LCA-model het voorspel dat die tiosulfaatvloeidiagram omgewings impakte vir 18 uit 19 impak kategorieë verminder het, in vergelyking met die sianiedvloeidiagram, afgesien van die impak kategorie van varswaterverbruik. Elektrisiteitverbruik gedurende sianiedverwoesting en uitlaat van die konvensionele sianiedvloeidiagram is herken as redes vir die verhoging van omgewings impakte in vergelyking met die tiosulfaatvloeidiagram. Voorstelle om omgewingsimpak verder te verminder gebaseer op LCA-resultate is gemaak.

Sianied- en tiosulfaatlogingprosesvloeidiagram oplossings is geïdentifiseer vir die herwinning van goud- en ammoniumdiuranaat (geelkoekuraan) van 'n hipotetiese scenario van Wes-Rand, Witwatersrand se mynuitskothulpbronne. Die omgewingsimpak van mynuitskot is verminder. Sulfiede en uranium in mynuitskot is verminder van 0.18% tot 0.03% (82% vermindering) en van 54.9 g/ton tot 13.0 g/ton onderskeidelik (75% vermindering), en daarby is die omgewingsrisiko's verwant aan suurmyndreinering en uraan radionuklied-emissies, verlaag. Die prosesse voorgestel verminder swaar metaal emissies en voldoen gedeeltelik aan die Nasionale Omgewingsbestuur: Afval akte 2008 (Akte no. 59 van 2008) (NEMA) vir die beskerming van waterhulpbronne. Die uitsonderings is mangaan- en koper emissies vir die tiosulfaatvloeidiagram en mangaan-, koper- en lood emissies vir die sianiedvloeidiagram. Daaropvolgende mynuitskotremediëringstrategieë is nodig om mynuitskot inert te maak volgens NEMA. LCA verskaf 'n basis vir omgewingsimpak assessering, maar sosiale- en ekonomiese assesserings is nodig om lewensvatbaarheid van

voorgestelde prosesvloeidiagramme te bepaal, en volhoubare ontwikkeling in die mineraalprosesseringindustrie te verseker.

Acknowledgements

I wish to extend gratitude towards the following people and organisations

Table of contents

List of figures

List of tables

List of abbreviations

Abbreviation	Name
AARL	Anglo American Research Laboratory
AMD	Acid mine drainage
BEGe	Broad energy germanium detector
BV	Bed volume
CFB	Circulating fluid bed roasting
CIL	Carbon in Leach
CIP	Carbon in Pulp
DTP	Dithiophosphate
EF	Ecological footprint
EIA	Environmental impact assessment
ELCA	Exergetic life cycle assessment
ENA	Ecological network analysis
ERGO	East Rand Gold Company
EU	European Union
GDP	Gross Domestic Product
GHS	Globally Harmonized System of Classification and Labelling of Chemicals
HERA	Human and environmental risk assessment
ICP-MS	Inductively coupled plasma - mass spectrometry
ISO	International standards organisation
LCA	Life cycle assessment
LSFO	Limestone forced oxidation
MFA	Material flow analysis
MOI	Mineral of interest
NaMBT	Sodium Mercaptobenzothiazole
NEMA	National Environmental Management: Waste act 2008 (Act No. 59 of 2008)
P_{80}	Particle size/Screen size through which 80% of particles will pass
PGE	Platinum group elements
pH	Potential of hydrogen
PIOT	Physical input – output table
POX	Pressure oxidation
PSD	Particle size distribution
QEMSCAN	Quantitative Evaluation of Materials by Scanning Electron Microscopy
RIL	Resin in Leach
RIP	Resin in Pulp
RO	Reverse Osmosis
RSA	Republic of South Africa (South Africa)
SDG	Sustainable Development Goal
SFA	Substance flow analysis
SGC	Synergistic Glycine Cyanide

SHE	Standard Hydrogen Electrode
UN	United Nations
WAD	Weakly Associated Cyanide Complexes
Wits	Witwatersrand, South Africa
XRD	X-ray diffraction spectrometry
XRF	X-ray fluorescence spectrometry

Nomenclature

Symbol	Property	Unit
N	Agitator speed	$\dfrac{rev}{s}$
E_t	Bond work index	kWh/t
$[…]$	Concentration (Molarity)	M (mol.dm^{-3})
E	Energy required	kWh/t
X_P	Final particle size (80% passing size of product)	µm
ρ	Fluid density	$\dfrac{kg}{m^3}$
μ	Fluid dynamic viscosity	$\dfrac{kg}{m.s}$
D_a	Impeller diameter	m
X_F	Initial particle size (80% passing size of feed)	µm
m	Mass	tonne or kg
pH	Potential of hydrogen	Dimensionless
Re	Reynold's number	Dimensionless
N_p	Power number	Dimensionless
Q	Pumping capacity	$\dfrac{m^2}{s}$
N_Q	Pumping number	Dimensionless
E$_h$	Redox Potential	V
P	Shaft Power	W
P	Shaft Power	W
g_c	Shaft power conversion factor	$\dfrac{m}{s^2}$
β	Stability constant	Dimensionless
T	Temperature	°C or K
V	Volume	L or dm^3

1 INTRODUCTION

1.1. Background

South Africa's supply of mineral resources to the world has led to the accumulation of mine waste, referred to as mine tailings, since the inception of the mineral processing industry in South Africa. Mine tailings are the solid waste effluent from extractive metallurgical operations that are deposited in geotechnical structures called tailings dams (Lyu et al., 2019). Gold mining waste accounts for 47% of the mineral waste in South Africa and is therefore the largest source of solid waste pollution in South Africa (Adler et al., 2007, cited Department of Water Affairs and Forestry, 2001). Valuable minerals may be present in mine tailings owing to shortcomings of technology used when run of mine ore was first processed. Alternatively, precious metals were occluded within refractory minerals and therefore at the time of run of mine ore processing it was uneconomical to recover. Mine tailings may be reprocessed to recover valuable minerals when economically feasible. Otherwise it is stockpiled while mine houses are accountable for managing ecological and structural risks posed by tailings dams. The historical decline in gold grades of primary ore resources in South Africa has increased the rate of tailings emissions per functional unit of gold manufactured. The management of risks associated with tailings dams is an externalised cost that influences the profitability of mine house operations within a competitive market. Therefore, there is a requirement for solutions to minimise risks associated with mine tailings.

The decline in gold grade of primary ores and the accompanying increase in tailings emissions, increases harmful emissions to the environment as well. The ecological impacts associated with mine tailings include the leaching of toxic substances to the environment through a phenomenon called acid mine drainage (AMD) along with the aeolian transmission of pollutants by wind erosion of tailings dams. The pollutants emanating from tailings dams affect fauna and flora while posing a hazard to human health and safety in nearby settlements. An example of such an instance is in Wonderfonteinspruit region in Randfontein, South Africa where gold and uranium mining operations have polluted of surface and ground water resources. This Wonderfonteinspruit contamination has had adverse effects on ecosystems and people dependant on these resources (Winde, 2010; Winde and Sandham, 2004). The aforementioned ecological challenges establish a requirement for minimisation and sequestration of pollutants from mine tailings.

The mitigation of the hazards mentioned, relies on the implementation of approaches that reduce the quantity of mining waste that reports to tailings dams. To reduce the quantity of mine tailings, opportunities for industrial symbiosis have been identified for the use of mine tailings as feedstock for the manufacture of other products. There is consensus between authors that mine tailings can be used to produce products for the construction industry such as bricks, stone-paper, cement additives and aggregates in road construction along with ceramics for consumer use and backfill for decommissioned mines. Backfill is a cemented paste made from mine tailings used to provide

ground support to prevent rock falls and rock burst in mines while providing a safe mode for mine tailings disposal thereby inhibiting the migration of heavy metals and acid mine drainage to above ground ecosystems (Malatse and Ndlovu, 2015; Matinde, 2018; Qi and Fourie, 2019; Sibanda and Broadhurst, 2018). Unfortunately, toxic and radioactive substances first need to be removed from mine tailings to render the solid waste inert before any products are manufactured from tailings or reintroducing tailings to the ecosphere (Sibanda and Broadhurst, 2018). This is imperative to prevent promoting the environmental impacts of mine tailings into the consumer value chain where it can affect other ecosystems and the general population.

In South Africa, the gold grade of primary ore resources is approaching the nominal gold grades in tailings dams in the Witwatersrand region (discussed in section 2.1). There is an opportunity for the gold industry to pursue recovering value from tailings as a secondary gold resource. In light of the legacy of pollution by the gold mining industry, the South African government has implemented stringent regulations on the metallurgical industry concerning social and environmental impacts that these businesses may have on the environment and surrounding communities (Dale, 1997; Department of Minerals and Energy, 1997). This therefore introduces an opportunity to identify extractive metallurgical technologies that reduce overall environmental impacts compared to conventional technologies.

Quantitative information related to the environmental impacts of process technologies are required to develop economically and environmentally sustainable solutions. Life cycle assessment is an environmental impacts assessment tool for evaluating ecological consequences of technology from raw material manufacture through to product use and product disposal. LCA has seen limited implementation in the metallurgical industry and this work will endeavour to contribute to the academic body of knowledge in the field of gold recovery from the secondary gold resource of mine tailings. LCA has been implemented as an environmental impact assessment (EIA) tool in the ISO 14 000 series for reducing environmental burdens of products and services in industry. Therefore, more studies in this field can allow stakeholders in the metallurgical industry to make informed decisions about mineral processing technologies.

1.2. Research questions

1. What existing technologies are available for gold recovery from mine tailings?
2. What processes can be developed to recover minerals of interest and reduce environmental impacts posed by toxic compounds in mine tailings in the West Rand, Witwatersrand region of South Africa?
3. How does the life cycle environmental impacts of technologies in processes proposed affect the overall process environmental sustainability?

1.3. Problem statement

The amassed reservoirs of mine tailings are the largest single source of pollution in South Africa. The transfer of toxic substances from tailings dams into the environment may cause irreparable harm to ecosystems and their ability to support life. Containment strategies are the predominant approach to mitigate the environmental impacts associated with mine tailings. This challenge is not constrained to South African mining activities but to all anthropogenic mining activities worldwide. Unfortunately, the rehabilitation of mine tailings to render them inert is a costly endeavour for mining houses especially if no economic benefit can be gained from such an initiative.

In South Africa, the average head grade (concentration of gold in ore) of primary gold ores has been declining leading to reduction in production of gold in South Africa. This trend has stimulated a trend in mineral processing research to recover gold from secondary sources such as mine tailings. Minerals of interest (MOI) are embedded within tailings because inefficient, low-recovery process have been used in the past to recover MOI's. But new technological advances have made it possible to overcome low-recovery challenges that were previously faced (Syed, 2012). Therefore, the mining industry requires process solutions for high recovery of gold from low grade and complex ores that conventional process flowsheets from earlier plants in history could not recover. These new processes, however, need to ensure that the waste streams produced reduce the environmental burdens associated with mining operations when compared to conventional processes. The performance objectives of the new process need to be aligned with rendering mining waste streams inert so that the waste streams, particularly mine tailings, can be used in other industries thereby reducing the quantities of mine tailings that need to be managed.

1.4. Research aims and objectives

Using the West Rand, Witwatersrand region as a case study, the aim of this study is to identify opportunities to maximise the recovery of valuable minerals from mine tailings. In parallel, approaches to reduce the quantity of toxic environmental contaminants and precursors in mine tailings will be identified. Therefore, a supporting aim of this project is to reduce environmental impacts of mine tailings.

The following objectives need to be realised to aid in achieving the aim:

1. Collate mineralogical and geochemical data on mine tailings from literature
2. Develop process flowsheets to recover value and mitigate environmental impacts of mine tailings.
3. Quantify environmental impact of identified processes using Life Cycle Assessment (LCA)
4. Comparison of environmental impacts of identified processes that yields the lowest environmental impact

5. Identify opportunities to reduce short- and long-term effects of recommended processes with the purpose of developing sustainable processes.

1.5. Scope and limitation

This project will be conducted within the following scope of limitations:

1. A hypothetical scenario of mineralogical and geochemical composition of mine tailings from the West Rand, Witwatersrand region of South Africa will form the basis for the process design. Comprehensive data for gold deportment in literature for the West Rand, Witwatersrand region is limited and therefore information from different studies are collated to create a hypothetical scenario.

2. Secondary mineralogical and geochemical data will be used to create a representative sample of the composition of tailings dams in the West Rand, Witwatersrand region. The sampling error, bias, reliability, and validity of the secondary data can thus only be assessed based on information quoted within literature sources.

3. The environmental impacts of the processes developed will form the basis of process recommendations and not social and economic impacts. The methodology for Life Cycle Assessment has been standardised as part of ISO 14 000 series to only account for environmental impacts and not social and economic impacts (ISO, 2006a).

4. Environmental impact of capital goods manufacture will not be considered.

5. The life cycle impact assessment methodology will not include the optional steps of *normalisation* and *grouping and weighting* as quoted in ISO 14 040.

1.6. Thesis outline

This work will be presented in the following manner:

Chapter 1: Introduction – overview of the background, problem statement, research question, aims and objectives of this work.

Chapter 2: Literature review – characterisation and composition of gold mine tailings, to develop a representative sample of tailings dams within the West Rand, Witwatersrand region and understand gold deportment as an input into process flowsheet development. Evaluation of gold recovery unit processes to formulate flowsheets for optimum recovery of gold that minimise environmental consequences.

Chapter 3: Methodology – explanation of the methodology that will be followed to realise the aims and objectives.

Chapter 4: Process development and description – summary of reagents, process conditions and areas of concern for each unit process in the process flowsheets proposed

Chapter 5: Results and discussion – reporting and evaluation of results obtained

Chapter 6: Conclusions and recommendations – collate findings compiled at the current stage of the research.

2 LITERATURE REVIEW

2.1. South African gold industry and the legacy of mine tailings

Since 1990, the South African gold production rate has decreased by 85% to 90 tonnes per annum in 2019 and forfeited its position to China as the world leader in gold manufacturer in 2008 (CEIC, 2019). This is attributed to the decline in gold grade of primary ores mined from the earth as depicted in Figure 2-1. Figure 2-1 also displays an exponential trendline for average gold grade in South Africa along with a line representing average gold grade in South African tailings dams (represented in Figure 2-2) of 0.7 g/tonne as determined from the tailings dams investigated in the current study. Extrapolation of the exponential trendline in Figure 2-1 reveals an intersection with the line representing the assumed tailings grade in the year 2044. This signifies that primary gold ore resources purity will decline until mine tailings become an economically viable alternative resource.
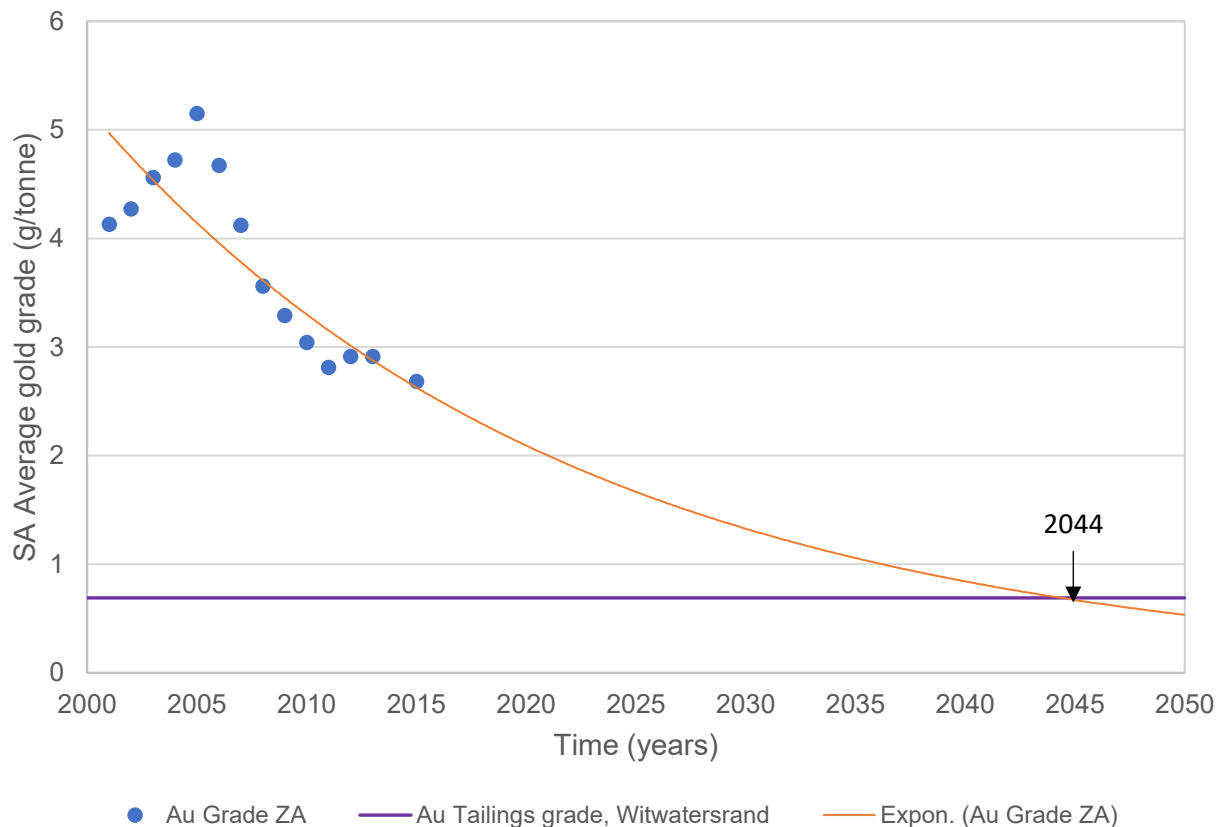
Figure 2-1: Decline in average gold head grade in primary ore resources in South Africa [Adapted from: (Chamber of Mines South Africa, 2017, 2011; Minerals Council South Africa, 2013; Neingo and Tholana, 2016)]

In 1997, Gold mine tailings accounted for 47% (220 million tonnes) of the solid waste pollution in South Africa (Adler et al., 2007, cited Department of Water Affairs and Forestry, 2001). Assuming an average head grade of 0.7 g/tonne, along with 90% gold recovery from process this equates to 138 tonnes of pure gold can be recovered from mine tailings. At the current gold price of ZAR 1 034 347.52 per kg on 7 September 2020, this equates to a market value of ZAR 143 billion that

is essentially captured within discarded waste (GoldPrice.Org, 2020). Tailings dams at the higher end of the distribution, such as the Crown mine (marker 17 in Figure 2-2) with a gold head grade of 3.91 g/tonne, already exceed the nominal grades from primary ores depicted in Figure 2-1. Therefore, the large quantity of tailings together with the gold grades present within them demonstrate the potential of gold mine tailings to be a secondary gold resource in South Africa. Paired with this, there is an opportunity to reduce the environmental impacts of gold mine tailings. Figure 2-2 illustrates the location of a non-exhaustive list of gold mine tailings dams in the Witwatersrand region of South Africa reported in literature. The mineralogical details of tailings dams in Figure 2-2 are summarised in Appendix C and Appendix D.
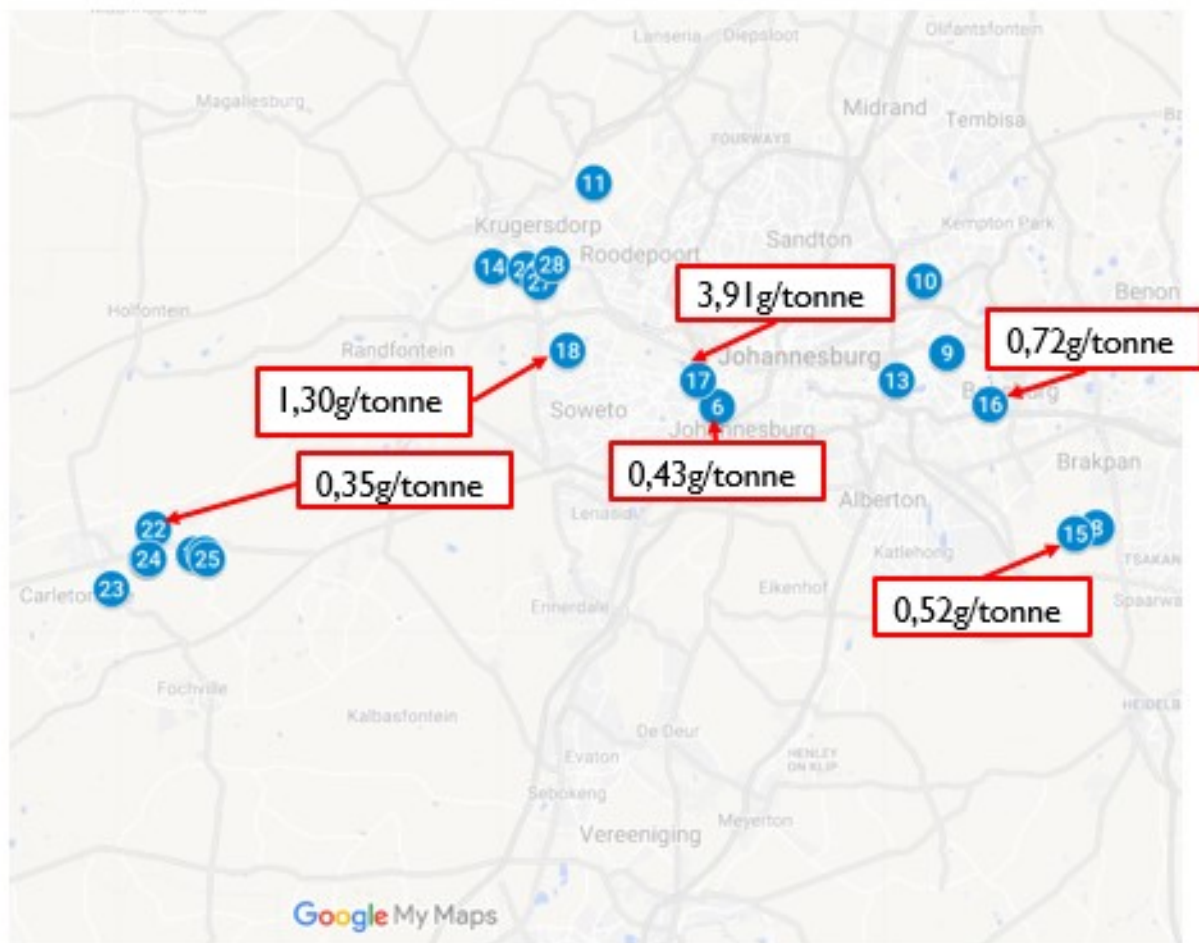
Figure 2-2: Location of tailings dams and gold head grades for which geochemical and mineralogical data is available in literature

2.2. Environmental impact of gold mine tailings

During the service life of a mine, the tailings are stockpiled where they pose a great environmental impact owing to the hazardous materials that are contained within mine tailings. The tailing stockpiles remain untouched unless an opportunity arises for the reprocessing of tailings for the recovery of valuable minerals or use of backfill for decommissioned mines. Many of the tailing's reservoirs in the Witwatersrand have not been processed for more than a century and the impacts associated with

the pollutants has increased with time such as the impact of acid mine drainage run-off (Fashola et al., 2016; Naicker et al., 2003; Venkateswarlu et al., 2016). Table 2-1 demonstrates that the concentration of carcinogens and acute toxic metals (i.e. arsenic, copper, nickel, vanadium) in the West Rand, Witwatersrand region are high (complete dataset summarised in Appendix C). Communities and ecosystems in close proximities to tailings dams are at risk of exposure to pollutants. Research has shown that heavy metal contamination from gold mines are at the levels where residents in adjacent communities can develop cancer if the food sources are contaminated by tailings dams dusts (Fashola et al., 2016; Kamunda et al., 2016a; Maseki, 2017; Ngole-Jeme and Fantke, 2017). An explanation of how the mineralogical data in Table 2-1 was gathered will follow in section 2.3.

Table 2-1: Average pollutant concentration (mg/kg) of tailings dams West Rand region in Witwatersrand Basin compared to regulatory soil screening values for protection of land and water resources.

Pollutant (mg/kg)	West rand, Witwatersrand	Maximum concentration [b]	% above regulatory limit
Cr	225	46 000	n/a
Cyanide	-	14	n/a
As	78.1	5.8	1247%
Mn	1982	740	168%
Cu	42.2	16	164%
U	54.9	23[a]	139%
Pb	24.8	20	24%
Ni	96.6	91	6%
Zn	65.5	240	n/a
Hg	0.2	0.93	n/a
Co	25	300	n/a
Cd	0.51	7.5	n/a
V	5.89	150	n/a
-: No data available Red values = above regulation limit; Green values = compliant.			
n/a: Not applicable			
[a] : (Canadian Council of Ministers of the Environment, 2007)			
[b]: (Department of Environmental Affairs, 2014)			

The South African environmental regulations do not stipulate any guidelines for uranium mass concentrations (in mg/kg) within soils but instead require site based activity concentration measurements using broad energy germanium (BEGe) detectors with radioactive waste being regulation by the National Nuclear Regulator (Department of Environmental Affairs, 2014; Kamunda et al., 2016b). The Canadian environmental regulations has, however, established a limit of 23 mg/kg for uranium in soils designated for agricultural and residential use, which the West Rand tailings dams exceed by 130% (Table 2-1) (Canadian Council of Ministers of the Environment, 2007). There is consensus among researchers that uranium has leached from tailings dams in the Randfontein

and Wonderfonteinspruit areas and has caused radionuclide contamination of water bodies (Coetzee et al., 2006; Winde, 2010; Winde and Sandham, 2004). Once water resources are contaminated, the health of fauna, flora and people who use polluted water resources are affected.

Collapsing tailings dams is another inherent risk that accompany the stockpiling of mine waste. Tailings dam failures cause the loss of life, destruction of villages and desolation of the affected ecosystems. The tailings dams failure in Merriespruit during 1994 is an example of a notable failure that has affected South Africa (Van Niekerk and Viljoen, 2005). The mitigation of tailings dams collapsing and associated environmental impacts is dependent on the proper design, monitoring and management of these geotechnical structures. But this is only a containment strategy for the environmental impacts and does not reduce nor eliminate the risk.

In literature, containment strategies for managing the environmental impact of mine tailings is often employed. Examples include lining of tailings dams, ground cover and using tailings as backfill. Direct treatment technologies to remove toxic substances from tailings, such as "soil-washing", are costly while phytoremediation technologies are not employed on a large scale owing to a technology shortage of selective, hyperaccumulator plants (Arab and Mulligan, 2018; Nemutandani et al., 2006; Odoh et al., 2019; Rösner and van Schalkwyk, 2000). The aforementioned approaches are useful for addressing the environmental and health impacts of older, inherited tailings dams. It does not, however, address the root cause of the problem which is to prevent the discharge of tailings exceeding the regulatory limits from mining operations.

In addition to this, research has demonstrated that in South Africa there is a loop-hole around the externalised costs associated with mine closure whereby larger corporations sell mines to smaller, less-well-resourced companies who then inherit the problems associated with the management of non-compliant tailings (Bainton and Holcombe, 2018; van Druten, 2017; Watson and Olalde, 2019). The implementation of sustainable countermeasures and the enforcement of legislation requires a collaborative effort from civil society, industry and the government to prevent the formation of non-compliant tailings emissions. The researcher acknowledges that environmental concerns caused by tailings dams are an inherited problem from malpractice in the past. But as the average gold head grade in South Africa declines to the point where secondary resources become a viable source of gold and other minerals, there is an opportunity to reduce and mitigate the long-term environmental impacts of mine tailings as well.

2.3. Characterisation of West Rand, Witwatersrand region gold mine tailings

The mineralogical composition of gold mine tailings can vary significantly depending on characteristics of the excavated ore, extraction regime followed, the efficiency of the prior extraction cycle and the age of the tailings reservoir. Proper characterisation of gold associations with minerals

is imperative to develop and optimise processes for efficient gold recovery. In this work, a case study of a hypothetical tailing's dam in West rand region of the Witwatersrand basin was made using descriptive statistics. Descriptive statistics, especially when using an arithmetic mean, has the flaw that outliers may skew data and the arithmetic mean is not representative of the entire population for this case study. Mineralogical surveys of specific tailings reservoirs as described in section 3.1 can be used to verify the results reported in literature, but this was beyond the scope of the present study.

Gold deportment and diagnostic studies are process mineralogical tools used to develop process flowsheets and to optimise desired metal recovery from ore resources. Limited gold deportment and diagnostic leaching studies have been reported in literature for tailings dams in the Witwatersrand region. XRD data is summarised in Table 2-2 along with trace elemental data in Table 2-3 and Table 2-4 for the West Rand, Witwatersrand region case study. The information from the individual literature sources are summarised in Appendix A to Appendix D.

Table 2-2: Collation of X-ray Diffraction (XRD) data of tailings dams of West Rand, Witwatersrand region in South Africa compiled from literature [Collated from (Janse van Rensburg, 2016; Ngole-Jeme and Fantke, 2017)]

Mineral	Mass (%)
Quartz	58.5
Pyrophyllite	24.9
Mica	5.8
Aluminite	3.8
Kaolinite/Chlorite	2.8
Serpentine	2.0
Gypsum	1.3
K-feldspar/ Rutile	0.9

Table 2-3: Summary of combustion analysis tailings dams of West Rand, Witwatersrand region in South Africa compiled from literature [Collated from:(Janse van Rensburg, 2016)]

	Sulphide	Sulphate	Total sulphur	Total carbon
Concentration (Mass %)	0.5	0.3	0.8	0.26

Table 2-4: Summary of trace elemental data by ICP-MS/OES (units g/tonne) tailings dams of West Rand, Witwatersrand region in South Africa compiled from literature [Collated from:(Janse van Rensburg, 2016; Kamunda et al., 2016; Mphinyane, 2018)]

Fe	16850	Cr	225.2	U	54.9	Ag	5.88	Sb	0.56
Al	15010	Ni	96.6	Na	53.6	Rb	4.63	Cd	0.51
Ca	7166	P	95.6	Cu	42.2	Bi	4.09	Au	0.28
Mg	2846	Th	82.8	Co	25	Se	3	Be	0.28
Mn	1982	Ba	82.4	Pb	24.8	Mo	2.4	Hg	0.2
Ti	651.5	As	78.1	Sr	11.5	Pd	1.73	Tl	0.15
K	554.9	Zn	65.5	V	5.89	B	0.73	Pt	0.01

XRD data (Table 2-2) does not identify auriferous sulphide minerals in tailings (e.g. pyrite, pyrrhotite etc.), while the presence of sulphides and sulphates was observed in combustion analysis (Table 2-3 and Table 2-4). A possible explanation for this could be the oxidation of auriferous sulphide minerals in tailings dams over time resulting in an amorphous crystal structure (Bhakta and Arthur, 2002; O'Connor and Dunne, 1991). Since XRD only detects crystalline and not amorphous phases of minerals, the sulphide minerals may not have been detected in low concentrations in the studies reviewed when compiling mineralogical data for the West Rand region. In a study on run of mine ore resources in the West rand, pyrite (FeS_2) accounted for over 97% of the sulphides and uranium minerals present consisted of 65.3% uraninite (UO_2) and 34.7% brannerite ($UTiO_2$) (Mngoma, 2012). Thus, the sulphides in the ore resource Table 2-3 was assumed to be only pyrite and the uranium composition was assumed to be the same as the run of mine ore as found in the study by Mngoma (2012).

Janse van Rensburg (2016) conducted lab scale, diagnostic leaching experiments on Witwatersrand region mine tailings to estimate the maximum gold recovery that may be attained in different unit processes. In this study, it was assumed that the cyanide diagnostic leaching results can be applied to non-cyanide lixiviants as well. This decision was justified by the fact that the diagnostic leaching steps decompose gangue minerals that encapsulate gold to be able to expose them to the lixiviant. This assumption can be verified by repeating the diagnostic leaching experiments with lixiviants that are promising alternatives to cyanide. This is a gap in literature that needs to be concluded for accurate process development on Witwaterstand mine tailings resources.

Figure 2-3 summarised the diagnostic leaching results along with guidelines from Lorenzen and Van Deventer (1993) and Lorenzen (1995) on which types of minerals may be digested at each stage of the evaluation. Janse van Rensburg (2016) substituted the hydrofluoric acid leaching step to determine gold occluded in silicate minerals as recommended by Lorenzen (1995) in favour of estimation by difference. This may introduce a discrepancy in the results obtained and verification with another analytical technique could reduce the uncertainty of the results. For example, QEMSCAN analysis or a hydrofluoric acid leaching step may be used to verify if 19.8% of gold is associated with quartz minerals (Figure 2-3 and Figure 2-4) since silica quartz makes up the bulk of the West Rand, tailings ore mineralogy as summarised in Table 2-2.
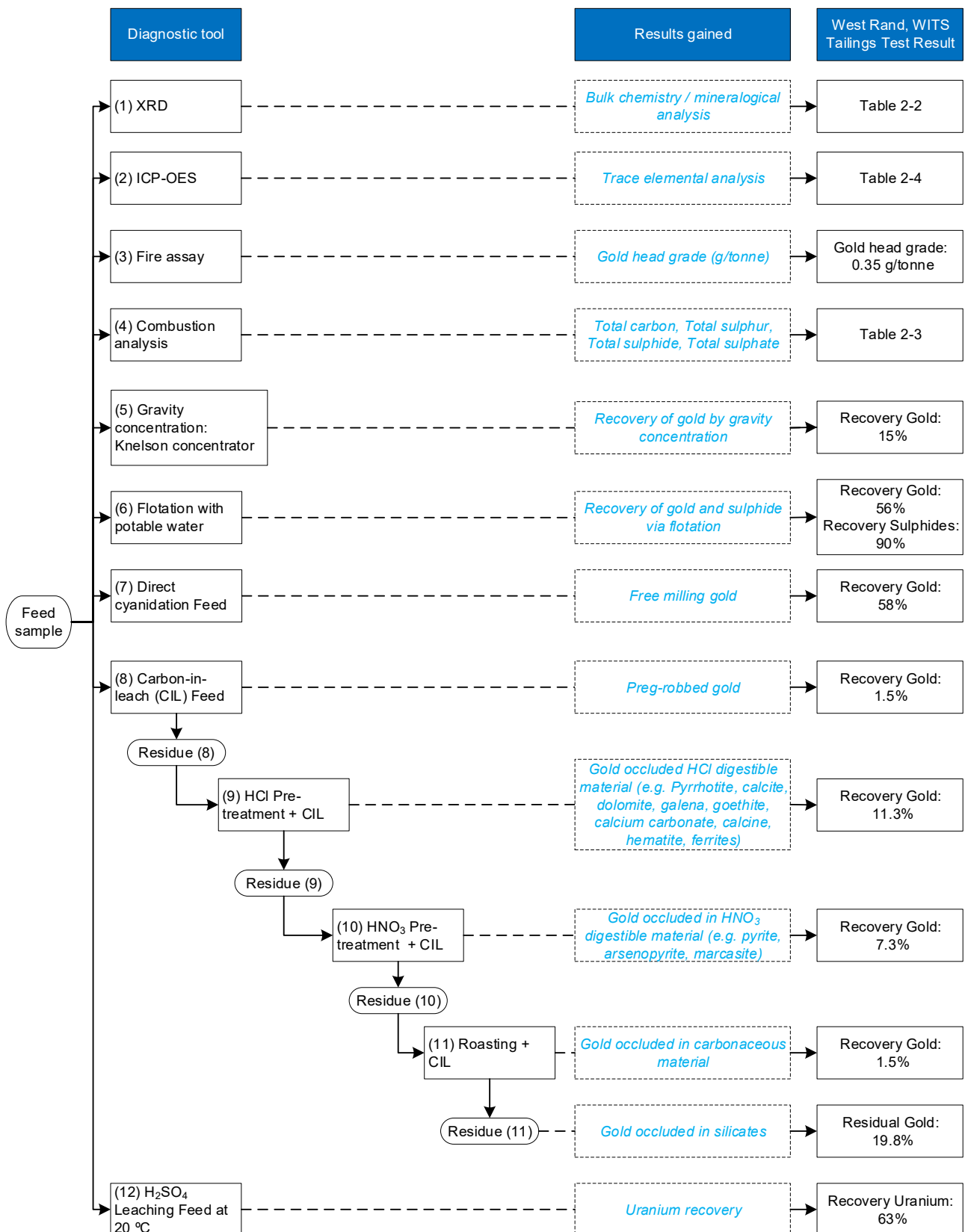
Figure 2-3: Diagnostic procedure and results of west rand, Witwatersrand mine tailings samples [Adapted from: (Janse van Rensburg, 2016; Lorenzen, 1995; Lorenzen and Van Deventer, 1993)]
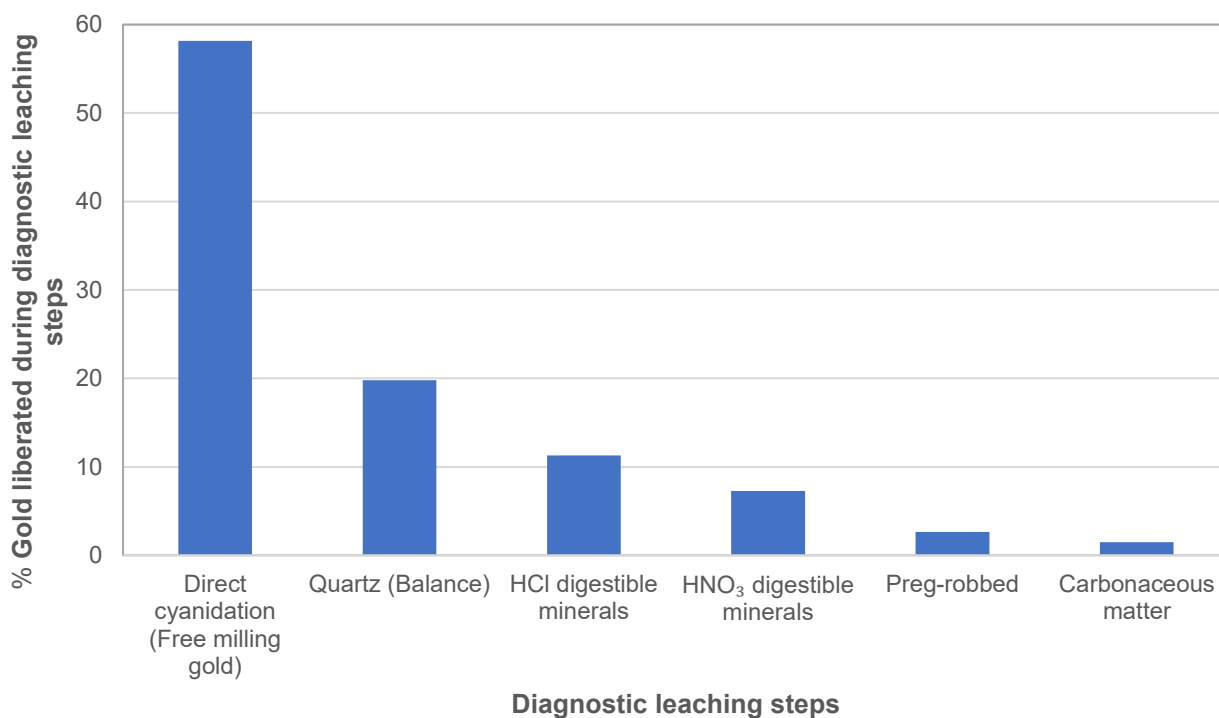
Figure 2-4: Diagnostic leach results of West Witwatersrand tailings dam samples. [Adapted from (Janse van Rensburg, 2016)]

The diagnostic leaching results reveal that 58% of gold is free-milling (Figure 2-4) and the ore resource can be considered as moderately refractory according to La Brooy's et al. (1994) definition of refractoriness of gold ores. HCl digestible minerals represented in Figure 2-3 labile sulphides (e.g. PbS, $Ee_{(1-x)}S$ ($x = 0 - 0.17$)), carbonates (CO_3^{2-}) or iron oxides (e.g. Fe_2O_3) that encapsulates gold within the tailings. These minerals may be decomposed by an acid pre-treatment step to recover the 11.3% of gold in carbonate minerals. Chlorine is a gold lixiviant therefore using hydrochloric acid may cause gold-halide complexes to report to the leachate and another gold recovery step will be required. Alternatively, sulphuric acid (not a gold lixiviant) may be considered for acid leaching and has the benefit that sulphuric acid leaches uranium as well. The recovery of uranium as a by-product will be necessary to prevent harmful radionuclide emissions.

Oxidative pre-treatment technologies may be required to liberate the 7% of gold within HNO₃ digestible minerals which can be sulphides and carbonaceous minerals (Figure 2-3). For the remaining 4% of gold locked within preg-robbing and carbonaceous mineralogy (Figure 2-3) require passivation of the preg-robbing minerals through chlorination, roasting, biological oxidation or the addition of specialised blanking agents to inhibit preg-robbing effects on gold recovery. Preg-robbing refers to the phenomenon occurring during leaching operations whereby precious metal complexes (specifically gold in this context) are adsorbed onto gangue minerals constituents of the ore, such as carbonaceous matter or other impurities, such as clays and elemental carbon. (Adams and Burger, 1998; Afenya, 1991; Dimov and Hart, 2016; Rees and van Deventer, 2000).

The balance of gold (19.8%) was associated with quartz minerals which can be liberated through grinding (Coetzee et al., 2010). Grading analysis data for West Rand mine tailings was not encountered in literature, therefore run-of-mine ore data from the West Rand was consulted to determine the particle size required for gold liberation from quartz minerals (depicted in Figure 2-5). The West Rand mine tailings has a particle size distribution of P_{80} = 106 µm which indicates that only 3.5% of gold occluded in quartz was liberated during the process of run-of-mine gold ore. Therefore, additional comminution is necessary to liberate gold occluded within quartz. The prior processing of gold mine tailings is an important consideration in subsequent tailings flowsheet development for gold recovery. Considering that the particle size distribution (PSD) of West rand tailings is P_{80} = 106 µm , when a P_{80} = 75 µm is recommended to ensure sufficient gold liberation prior to cyanidation (Coetzee et al., 2010; La Brooy et al., 1994) implies that prior processing focussed on recovery of coarse gold particles in higher size fractions was prioritised.
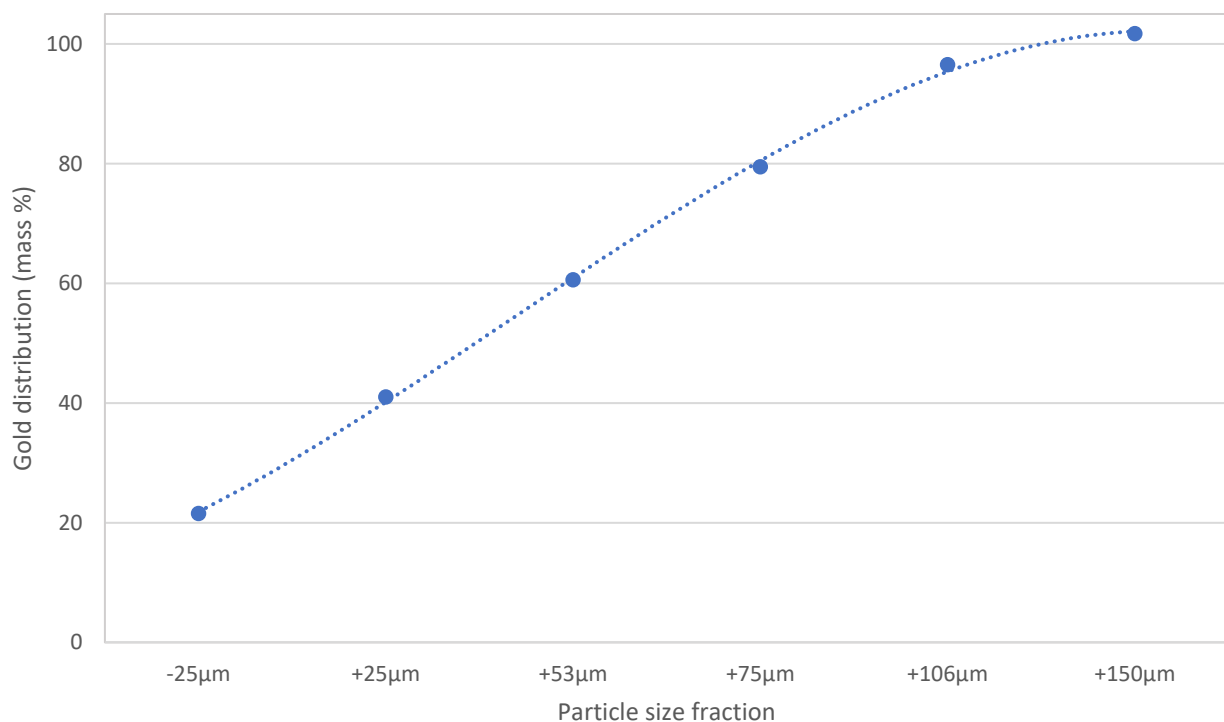
Figure 2-5: Grading analysis - gold distribution by particle size fraction for a run-of-mine, primary gold ore resource [Adapted from: (Mngoma, 2012)]

There is consensus between authors that Knelson® concentrator is an effective gravity separation technology for gold recovery. Gravity separation operations for gold recovery are more economical and reduce the size of downstream unit processes for gold recovery. Placing three or more Knelson® gravity concentrators in series may allow for close to 100% gold recovery as demonstrated in the work by Meza et al. (1994). This solution may be impractical owing to increased capital and operating costs (Gül et al., 2012; Laplante et al., 1995).

Gravity separation scouting tests conducted by Janse van Rensburg (2016) indicate that only 15% of gold by mass can be recovered using a Knelson® gravity concentrator from the West rand tailings ore body. It is unfortunate that the West Rand tailings are not amenable to gravity concentration techniques especially considering the low gold grade of 0.27 g/tonne is very low. Based on this information, gravity separation is not a viable process option for gold recovery for the tailing's feedstock considered in this study. The aforementioned results by Janse van Rensburg (2016) needs to be verified by heavy liquid separation as recommend by Coetzee *et al.* (2011) to determine if other gravity separation technologies can improve gold recovery.

Surface water supplied by Rand water was used in the gold and sulphide flotation trials conducted by Janse van Rensburg (2016) without the addition of collectors, activators, depressants or frothers (i.e. natural flotation). The potential sulphide and gold recoveries of 90% and 54% by mass respectively, reported in Figure 2-3. Literature on flotation used in tailings reprocessing plants in Witwatersrand region of South Africa report sulphide and gold recoveries of 85% and 46% respectively with the addition of flotation reagents (the reagent scheme employed is discussed in section 2.4.4) (O'Connor and Dunne, 1994, 1991). The discrepancy in results from aforementioned literature sources may be attributed to variations in tailings mineralogy since a time period of 15 years transpired between studies. The conservative sulphide and gold recoveries of 85% and 46% respectively was used in the present study to align with commercial plant data and to not base a study on data collected from a controlled laboratory environment.

Ambient condition sulphuric acid leaching trials revealed a 63% uranium recovery for West Rand mine tailings in the diagnostic leaching experiments conducted by Janse van Rensburg, (2016). The experimental conditions can be optimised for improved uranium recovery as recommend in the work by Lottering et al. (2008) by increasing leaching temperature to 40 - 60 °C, introducing an oxidising agent such as manganese dioxide (MnO_2) and increasing sulphuric acid consumption. Uranium recovery provides an opportunity for an additional value stream while reducing radionuclide emissions to the environment.

2.4. Process flowsheet development

2.4.1. Gold recovery flowsheets

Tailings reclamations plants for the recovery of gold have been commissioned in the Witwatersrand region. Figure 2-6 and Figure 2-7 illustrate gold recovery flowsheets proposed in literature for East Rand Gold Company (ERGO) and Driefontein. In South Africa, gold mining operations are separated into the initial gold recovery from ore to produce a crude gold bullion (gold purity ≈ 60%) by mine houses, followed by subsequent refining to a pure gold bullion (gold purity minimum 99.5%) by refineries (Auerswald and Radcliffe, 2005; Feather et al., 1997; Rapson, 1992). Hence, the gold recovery flowsheets in Figure 2-6 and Figure 2-7 exclude refining operations. The ERGO plant is

one of the largest gold-from-tailings plants in South Africa, while the Driefontein operation was a pilot plant study conducted by Fleming et al. (2010).

The flowsheets for gold mine tailings represented in Figure 2-6 and Figure 2-7 have the following features in common:

- flotation for gold and sulphide recovery
- production of yellow-cake uranium using sulphuric acid leaching
- cyanide as the gold lixiviant combined with activated carbon circuits for gold recovery
- flotation tailings subjected to carbon-in-leach cyanidation and gold recovery to increase gold recovery
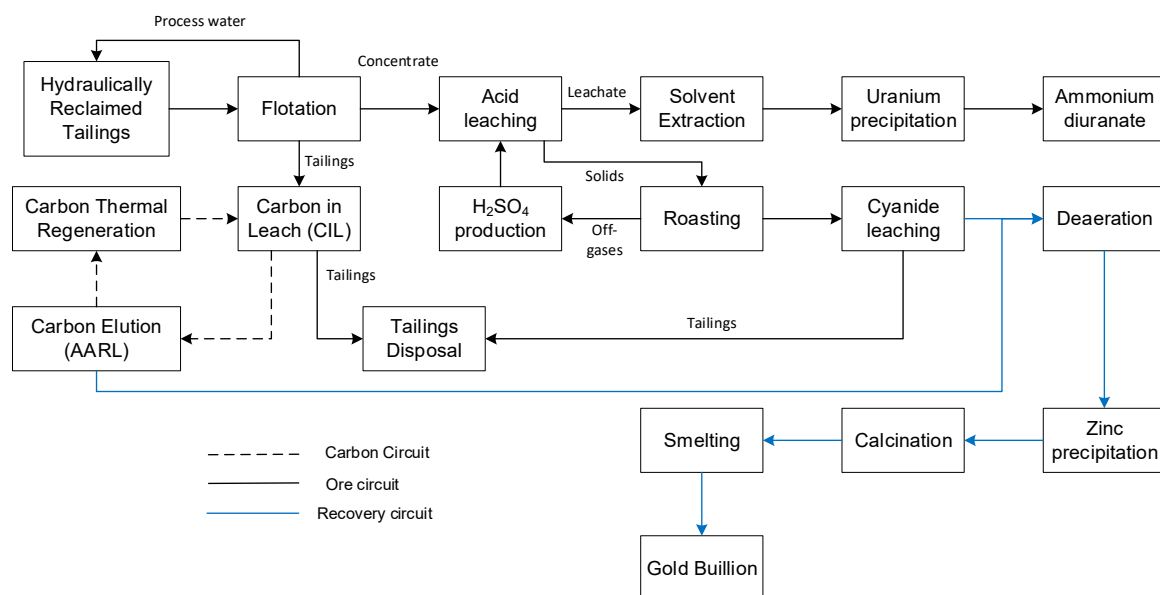
Figure 2-6: ERGO tailings reclamation process flowsheet [Adapted from: (Bosch, 1987; Marsden and House, 2006)]
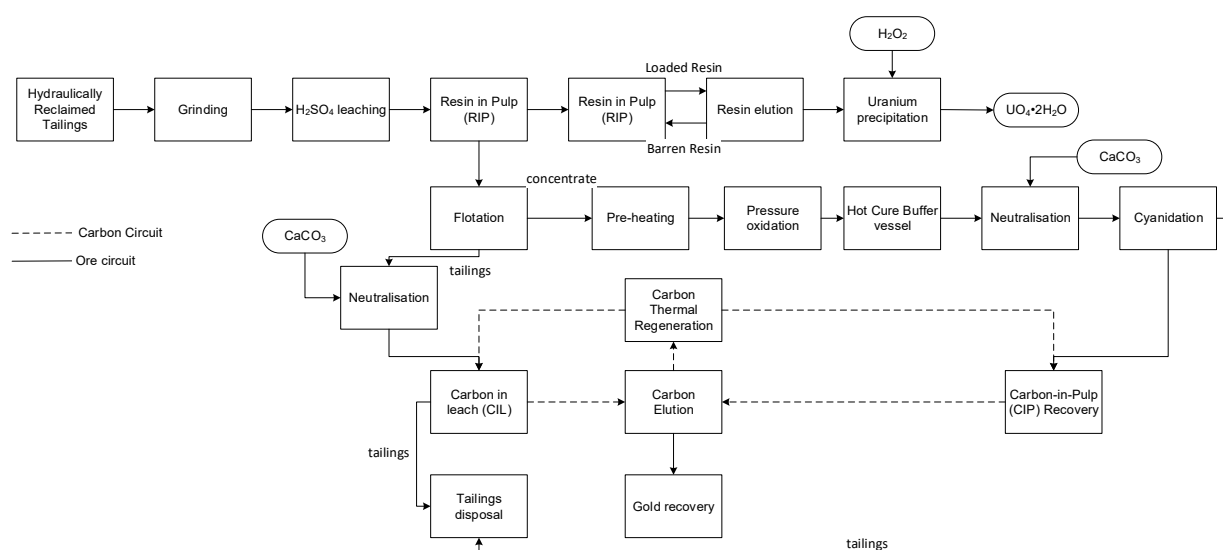
Figure 2-7: Driefontein tailings reclamation pilot plant flowsheet [Adapted from: (Fleming et al., 2010)]

The remaining sections of this chapter will compile a repository of unit processes that may be used in the recovery of value from tailings. This was done by summarising recent findings to maximise gold recovery from tailings in commercial processes and identify approaches to address the challenges of recovering value and reducing environmental impacts of mine tailings.

2.4.2.   Comminution

Comminution is the practice of size reduction of ores to attain desired particle size distribution for mineral recovery operations. Examples of these are: coarse material jaw crushers (1m > P_{80}>100 mm), intermediate material autogenous or semi-autogenous mills (SAG) (100mm > P_{80}> 20mm), fine material (20mm > P_{80}> 30μm) ball mill, vertical roller mill, hammer mill, jet mill, ultrafine grinding (P_{80}>20 μm) with IsaMill (Adams, 2016; Ellis and Gao, 2003; González-Anaya et al., 2011; Marsden and House, 2006; Orumwense and Forssberg, 1992). The balance of gold (19.8%) is occluded with quartz minerals which can be liberated through ultrafine grinding (Coetzee et al., 2010). The mine tailings feedstock has already been exposed to comminution, therefore regrinding of the bulk tailings feedstock was not recommended and would be costly. Ultrafine grinding is, however, reserved for grinding flotation concentrates to increase downstream process efficiency by increasing the surface areas exposed on particles.

Literature comparing ultrafine grinding of flotation concentrates (at P_{80} = 10μm) to sulphide oxidation technologies (i.e. roasting, pressure oxidation and biological oxidation) have concluded that ultrafine grinding provided the best metallurgical and economical results for gold recovery from high grade ores (94 g/tonne gold) (Ellis and Gao, 2003). Conversely, evidence in literature of sulphide oxidation technologies achieving greater than 99% decomposition of sulphide along with greater than 95% gold recovery has been reported (discussed in section 2.4.5) (Fomenko et al., 2013; Runkel and Sturm, 2009; Thomas and Pearson, 2016; van Niekerk, 2015). Therefore, including an additional unit process for ultrafine grinding of sulphide concentrate was not considered in flowsheet development. Furthermore, sulphide oxidation renders the residual mineral porous that permits penetration by lixiviant to access gold that was not liberated before oxidation (Chan et al., 2015; González-Anaya et al., 2011; Runkel and Sturm, 2009).

A disadvantage of ultrafine grinding is that sulphide minerals, which are precursors to acid mine drainage (AMD), are not converted to an inert form and report to solid waste emissions from mining process. Furthermore, lixiviant consuming effects of sulphides may increase reagent consumption costs during leaching. AMD is a significant environmental problem associated with mine tailings therefore sulphide oxidation technologies was prioritised above ultrafine grinding since environmental impact is a priority in this study. The economic versus environmental trade-off of process technologies is an important design consideration that plays a role in determining whether

technology will be implemented or not. The economic feasibility of flowsheets is beyond the scope of this study but remains an important decision-making criterion.

2.4.3. Uranium leaching and recovery

Alkaline and acidic uranium leaching technologies have both been employed commercially. Alkaline leaching operations are preferred in operations where acid-digestible minerals (as summarised in Figure 2-3) result in excessive reagent consumption during acidic leaching (Youlton and Kinnaird, 2013). The destruction of gangue minerals during acidic leaching can liberate occluded gold, a process known as reverse leaching. A 3-4% gold recovery increase was observed in the work by Lottering et al. (2008) following sulphuric acid leaching. The economic feasibility of uranium recovery is dependent on the market demand for this commodity. Therefore, although uranium recovery helps to reduce environmental impact of mine tailings the economic feasibility of the additional unit processes needs to be quantified and proven.

In literature, uranium recovery from leach liquors occurs predominantly by solvent extraction. The selection of extraction agent was based on a study on low-grade uranium ores investigating the extraction efficiency of common extraction agents within the uranium processing industry are ranked in descending order as: tri-n-octylamine > di(2-ethylhexyl) phosphoric acid > trioctylphosphine oxide > tributylphosphate > triethylamine. Following stripping of organic phase, a "yellow-cake" product of either ammonium or sodium diuranate is precipitated using ammonia or sodium chloride respectively (Kiegiel et al., 2017; Sole et al., 2011; Zhu and Cheng, 2011).

2.4.4. Flotation

Flotation is the practice of separating desired minerals from gangue within a pulp based on their hydrophobicity. Reagents such as collectors, activators, depressants and frothers alter the surface chemical properties of minerals such that desired minerals are rendered hydrophobic and report to the concentrate while undesired minerals are rendered hydrophilic and report to the tailings. O'Connor and Dunne (1991) compiled a summary of flotation conditions and reagents used by various tailings reclamation plants in South Africa (tabulated in Table 2-5). President Brand reagent scheme achieved the highest sulphide and gold recoveries of the tailing reclamation sites in Table 2-5 of 85% and 46 % respectively. When the uranium market price was high, the simultaneous flotation of pyrite, gold and uranium was practiced by ERGO and President Brand using the flotation reagent conditions in Table 2-5 which yielded flotation recoveries of 86% sulphur, 54% gold and 20% uranium for ERGO (Lloyd, 1981; Ruhmer et al., 1977). Table 2-5 summarises a later study done by O'Connor and Dunne (1991) where ERGO sulphide and gold recoveries dropped to 77% and 40% respectively. The difference in the results may be attributed to composition differences in the tailings samples evaluated owing to the 10 year time period between the studies, and the results of the most recent empirical data was considered in this study.

Table 2-5: Pyrite flotation empirical data and reagents from South African tailings reclamation plants [Adapted from: (O'Connor and Dunne, 1991)]

Plant	Head grade		Recoveries (%)		Feed Particle size	Pulp density	Residence time	Collectors	Activators	Depressant	Frother	pH
	% S	Au (g/t)	S	Au	% < 75 µm							
Ergo	1.0	0.56	77	40	73	45	22	85 g/t NaMBT[a] 10 g/t DTP[b]	35 g/t CuSO$_4$	8 g/t Acrol ® J2P 350 [c]	17 g/t Dow[d] ® 200	3.8
Free State Geduld	1.0	0.45	85	42	75	37	29	85 g/t NaMBT	30 g/t CuSO$_4$	40 g/t Acrol ® J2P 350	21 g/t Senfroth ®[e]	3.8
President Brand	1.0	0.35	85	46	70	40	19	85 g/t NaMBT	40 g/t CuSO$_4$	40 g/t Acrol ® J2P 350	35 g/t Dow ® 200	3.8
President Steyn	1.2	0.3	80	42	57	40	25	95 g/t NaMBT	55 g/t CuSO$_4$	40 g/t Acrol ® J2P 350	25 g/t Dow ® 200	3.8
Crown Mines	0.5	0.55	60	20	10	45	24	30 g/t NaMBT	-	-	10 g/t Senfroth ®	3-6

[a]: NaMBT - Sodium Mercaptobenzothiazole

[b]: DTP – Dithiophosphate

[c]: Acrol® J2P 350 – Trademarked chemically modified guar gum

[d]: Dow® 200 - Trademarked methoxy polypropylene glycols

[e]: Senfroth – Variant not specified.

2.4.5. Oxidative pre-treatment

Pre-treatment processes are often required to liberate gold particles encapsulated within gangue minerals. These minerals consume excess lixiviant or are preg-or leach- robbing minerals that adsorb gold during leaching operations thereby reducing gold recovery.

2.4.5.1. Pyrometallurgical roasting

Pyrometallurgical roasting involves exposure of sulphide and carbonaceous gangue minerals to oxygen at elevated temperatures (450-820 ºC) resulting in their decomposition into their respective oxides. The off gases are emitted during pyrometallurgical roasting such as sulphur dioxide (SO_2) and carbon dioxide (CO_2) along with particulate solids such as arsenic trioxide (As_2O_3). Scrubbing units and electrostatic precipitators are placed downstream of roasting equipment to sequester harmful emissions. Sulphur oxide emissions may be captured within a scrubber for the in-situ manufacture of sulphuric acid if economically viable. (Lottering et al., 2008; O'Connor and Dunne, 1994, 1991; Runkel and Sturm, 2009).

2.4.5.2. Pressure oxidation (POX)

The Driefontein tailings reclamation pilot plant achieved 99% sulphide oxidation of flotation concentrate in a pressure oxidation (POX) autoclave conditions of an oxygen partial pressure at 0.68 MPa and temperatures between 190 – 230 °C for 1 hour. One historical challenge of pressure oxidation is the conversion of iron sulphide minerals to basic iron sulphate ($Fe(OH)SO_4$) which dissociates above pH 7 to form sulphuric acid. Sulphuric acid affects pH control and increases reagent consumption in leaching operations. In cyanide flowsheets, if pH descends below pH 10 there is a risk of toxic hydrogen cyanide gas evolution and can be detrimental to human health. Sulphuric acid formation can be mitigated by neutralisation (Chan et al., 2015; Fleming et al., 2010; Thomas, 1991). Gold recovery is higher during acid POX than alkaline POX (Gertenbach, 2016; Gorain et al., 2016). Acidic POX is amenable to high sulphide ores because of the autogenous production of sulphuric acid. Alkaline POX is preferred when ores have high carbonate contents (Gorain et al., 2016; Thomas and Cole, 2005; Thomas and Pearson, 2016).

2.4.5.3. Biological oxidation

Dew et al. (1997) concluded that bio-oxidation of sulphide minerals is more economically feasible than pressure oxidation and roasting. Cost of technologies are ranked in Table 2-6 from cheapest to most expensive. This is promising bearing in mind the low head grade of gold in mine tailings.

Table 2-6: Relative cost comparison for oxidation plants compared to BIOX® process [Adapted from: (Dew et al., 1997)]

	Capital expenditure	Operating costs
BIOX® biological oxidation	1.00	1.00
Pyrometallurgical roasting	1.92	1.11
Pressure oxidation	2.38	1.14

A drawback of biological process is the slow reaction kinetics. Attia & El-Zeky (1989) reported a residence time of 28 days for 98% pyrite bio-oxidation with *Thiobacillus ferrooxidans* of a flotation concentrate produced from mine tailings. In the BIOX® process, the residence time was reduced by inoculation with multiple bacterial strains (*Thiobacillus ferrooxidans, Thiobacillus thiooxidans, Leptospirillum ferrooxidans).* In the Biox® process, 90% sulphide oxidation residence time of 5 days is required for arsenopyrite and 11 days for pyrite (Dew et al., 1997). Heap bio-oxidation of low-grade tailings (1.3-3.4 g/t gold) process followed by cyanide leaching has been commissioned at Newmont and reported a 58% gold recovery. Unfortunately, the slow kinetics of bio-oxidation only permitted 38% gold liberation from pyrite (Bhakta and Arthur, 2002). Non-cyanide lixiviant trials were conducted at the same plant in 2017 and concluded that thiourea was ineffective at gold leaching while ammonium thiosulphate was recommended as a cyanide alternative (Roberto, 2017).

The slow reaction kinetics (i.e. residence time of 11 days to achieve 90% sulphide oxidation for biological oxidation compared to 1 hour for 99% sulphide oxidation during pressure oxidation), robust process control requirements to maintain optimum process conditions and sensitivity of bacteria to pollutants such as chlorine and arsenic can reduce the plant throughput. Several bio-oxidation units operated in parallel will be required to remove this unit process as a bottleneck. Bio-oxidation was therefore not considered for sulphide oxidation technology in flowsheet development.

2.4.6. Leaching

Leaching is a hydrometallurgical process that involves the dissolution of soluble metals captured within the ore. Cyanide is commonly applied in industry as lixiviant for gold recovery, but ecological toxicity of cyanide has established a research field into alternative lixiviants. Unfortunately, alternative lixiviants are costly, require high concentrations for optimum gold leaching, unstable under certain conditions or research progress has not met criteria for profitable industrialisation (Ospina-Correa et al., 2018; Syed, 2012).

2.4.6.1. Cyanide and Thiosulphate

Cyanide is favoured for gold recovery (equation [2-1]) today because it is a well understood reaction system while being highly selective for gold and silver leading to profitable gold recovery (Dorin and Woods, 1991). Some disadvantages include toxicity, slow dissolution kinetics of gold and excessive lixiviant consumption owing to presence of gangue minerals such as sulphides and base metals that form more stable complexes than with gold. The presence of sulphides in the gangue cause passivation of solid gold surfaces reducing gold recovery. Challenges with sulphide passivation may be overcome by feeding cyanide in excess, lead nitrate addition to enhance gold dissolution kinetics, oxygen enrichment of the pulp or upstream destruction of cyanide consuming minerals. These approaches need to be assessed for their economic and environmental feasibility (Adams, 2016; Aylmore, 2016; Dorin and Woods, 1991; La Brooy et al., 1994; Marsden, 2006; Muir and Aylmore, 2004).

$$4Au^{\circ} + 8NaCN(aq) + O_2(g) + 2H_2O \rightarrow 4NaAu(CN)_2(aq) + 4NaOH(aq) \qquad \text{[2-1]}$$

The gold leaching kinetics of thiosulphate in aqueous alkaline solutions with dissolved oxygen is slow at normal temperature and pressure (NTP) conditions (Naito et al., 1970). A reaction system consisting of an oxidant (e.g. Cu^{2+} or Fe^{3+} ions) along with an oxidant stabiliser (e.g. ammonia, polyamine ligands paired with copper, oxalate paired with iron) is required to catalyse the dissolution of gold. Strict control of process parameters is required to prevent reduction in process efficiency by the oxidation of thiosulphate into polythionates that compete with gold for adsorption onto activated carbon or resins during recovery (Muir and Aylmore, 2004).

Ammoniacal copper–thiosulfate reaction system in solution and can be described by the following system of equations (Abbruzzese et al., 1995; Wan, 1997):

$$Au + 5S_2O_3^{2-} + Cu(NH_3)_4^{2+} \rightarrow Au(S_2O_3)_2^{3-} + 4NH_3 + Cu(S_2O_3)_3^{5-} \qquad \text{[2-2]}$$

$$4Cu(S_2O_3)_3^{5-} + 16NH_3 + O_2 + 2H_2O \rightarrow 4Cu(NH_3)_4^{2+} + 4OH^- + 12S_2O_3^{2-} \qquad \text{[2-3]}$$
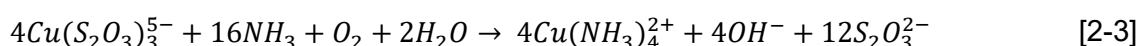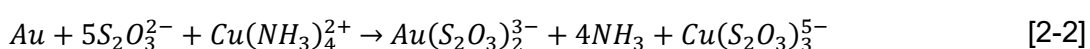
In an effort to identify leaching conditions that yielded reduced thiosulphate consumption, Lampinen et al. (2015) achieved 89% gold recovery of pressure oxidised flotation concentrate from a leaching solution consisting of 0.2 M $S_2O_3^{2-}$, 0.2 M NH_3, and 0.1 g/L Cu at operating conditions of 30 °C, 30 mass% solids under continuous air sparging of 0.2 L/min. Strict process controls are required to maintain ammonia and oxygen concentrations and Eh – pH conditions in the leach solution when applied commercially to deter passivation of gold surfaces by copper sulphate and sulphur (Aylmore and Muir, 2001; Birich et al., 2019; Gökelma et al., 2016).

Ammonia contributes to eutrophication in aquatic environments and therefore additional unit processes are required for ammonia destruction before discharge to the ecosphere. Laboratory investigations of other non-ammoniacal systems are summarised in Table 2-7. Commercialised processes by Barrick Goldstrike have opted for non-ammoniacal thiosulphate leaching (Choi, 2016; Choi et al., 2013). Thiosulphate is environmentally inert, however, the polythionate decomposition products increase the chemical oxygen demand in aquatic ecosystems therefore oxidation of polythionates into sulphate is required before discharge (Aylmore and Muir, 2001). Thiosulphate is an alternative lixiviant to cyanide that has been applied commercially (flowsheet depicted in Figure 2-8) to treat double refractory gold ores at Newmont and Barrick gold manufacturers to achieve a gold recovery of 79.6% (Choi, 2016; Muir and Aylmore, 2004).

Table 2-7: Summary of non-ammoniacal thiosulphate gold leaching systems

Reagent system	Reference	Commer-cialised
Oxygen-thiosulphate	(Choi et al., 2013; Sitando et al., 2015)	Yes
Oxygen saturated - Thiosulphate - Copper	(Sitando et al., 2015)	No

Reagent system	Reference	Commer-cialised
Copper ammonia thiosulphate	(Birich et al., 2019)	No
Air-saturated cyanide solution	(Sitando et al., 2015)	No
Copper organic amines	(Feng and van Deventer, 2011)	No
Iron -ethylenediaminetetraacetic acid (EDTA)	(S. Zhang et al., 2005)	No
Iron oxalate	(Chandra and Jeffrey, 2005)	No
Alkaline oxygenated thiosulphate	(Zhang and Nicol, 2003)	No
Alkaline oxygenated thiosulphate with Copper	(Zhang and Nicol, 2005)	No
Oxygen thiosulphate thallium	(Csicscovszki and Salminen, 2011; Bek and Shevtsova, 2012).	No
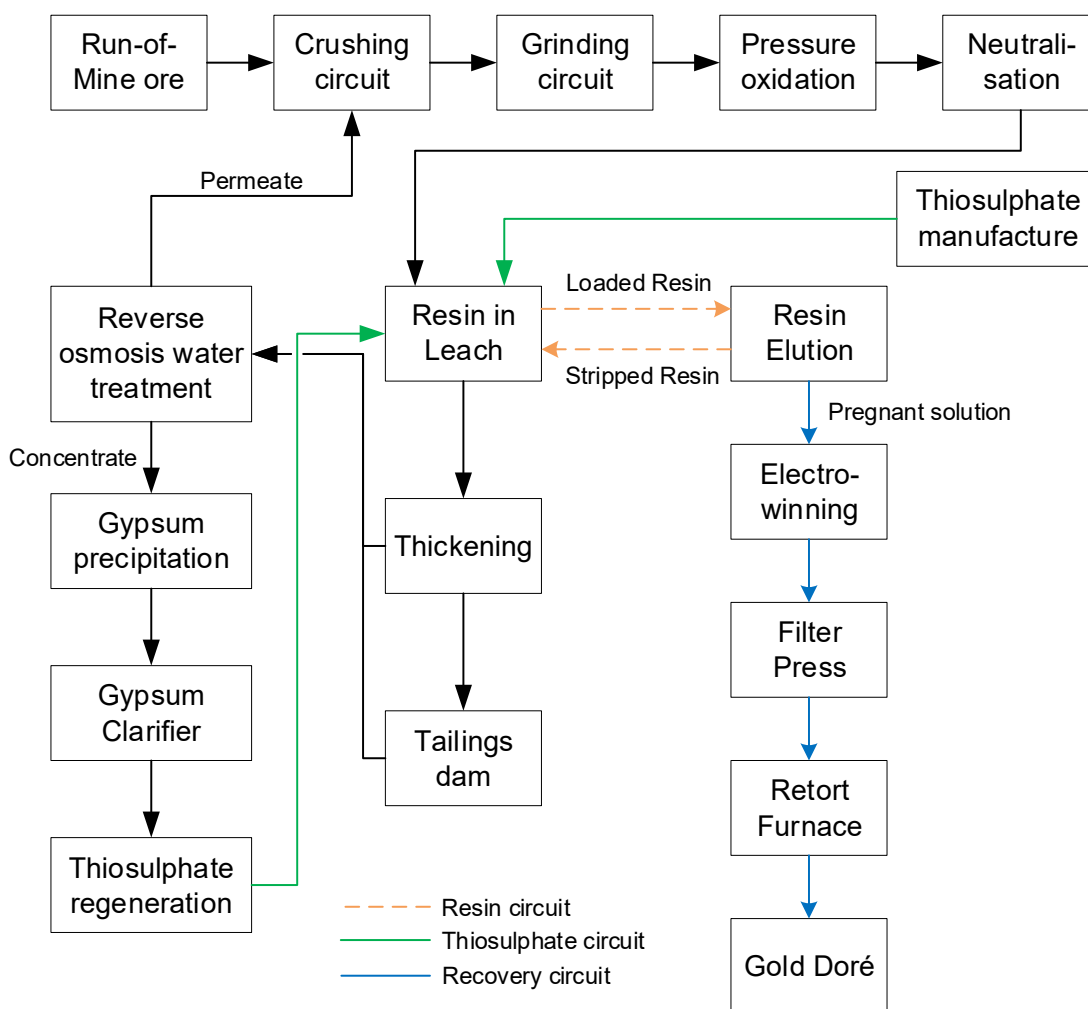
Figure 2-8: Barrick-Goldstrike thiosulphate leaching - gold recovery flowsheet for run of mine ore [Adapted from: (Choi, 2016)]

2.4.6.2. Glycine, Halides, Thiourea and Thiocyanate

Glycine (NH_2-CH_2-COOH) based leaching systems (equation [2-4] adapted from Tauetsile et al., 2019a) are currently being explored in literature as an environmentally friendly novel lixiviant for gold, copper and silver. Glycine is an amino acid species therefore a bioavailable resource within aquatic environments and can be decomposed by natural attenuation (Wheeler et al., 1977). Glycine ions are also more stable than thiosulphate and there is potential to recycle glycine solutions which is

23

beneficial from a resource consumption perspective. The current progress of glycine leaching technology is research into carbon adsorption that has proven that metal-glycinate complexes have a stronger affinity for activated carbon compared to metal-cyanide complexes (Tauetsile et al., 2019a, 2019b). At lab scale, electrodeposition-redox replacement (EDRR) technology for gold-glycinate system reported 88% and 35% gold recovery from synthetic pregnant solutions and real pregnant solutions respectively. The low gold recovery from real pregnant solutions was attributed to competing dissolved metal ion species that deposit on cathode preferentially to gold. Further research into EDRR reaction mechanisms for gold recovery from real solutions together with process optimisation studies are required in literature before technology can be promoted to pilot and commercial trials (Altinkaya et al., 2020). Seaman et al. (2019) developed a GlyCat[TM] pilot scale plant with glycine leaching technologies for gold and copper recovery from 10-15 g/t gold ore resources. The flowsheets in the study by Seaman et al. (2019) only demonstrated gold and copper recovery up to carbon adsorption. Therefore, research into viable eluants, optimum process conditions for carbon elution and gold recovery from pregnant leach solutions using technologies such as electrowinning and cementation still need to be demonstrated in literature. Glycine lixiviant systems will therefore not be considered for gold flowsheet development in this work (Eksteen et al., 2018; Oraby and Eksteen, 2015, 2014).

$$2Au + 4NH_2CH_2COOH + 2OH^- + O_2 \rightarrow 2[Au(NH_2CH_2COO)_2]^- + 3H_2O \qquad [2\text{-}4]$$
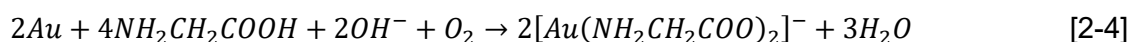
Birich et al., (2019) conducted gold dissolution kinetics studies that concluded that the dissolution rate of gold in halides arranged in descending order of dissolution rate is: aqua regia > iodine > bromine > cyanide. Iodine/iodide leaching systems have only been explored in lab-scale investigations. To date, no commercial processes have utilised this system owing to the cost of iodine relative to cyanide. In-situ bromine leaching processes at lab-scale have been investigated for gold leaching but have not been proven on pilot and commercial scale (Sousa et al., 2018). Bromine as a gold lixiviant has not seen further development into commercial applications. Aqua regia (3:1 HCl: HNO$_3$ ratio) was employed for gold leaching before the advent of cyanide (equation [2-5] adapted from (Syed, 2012)). The drawbacks of chlorine leaching systems include: poor gold selectivity, costly infrastructure to contend with low pH conditions and evolution of toxic chlorine gas that is an environmental and human health hazard. High reagent consumption is another disadvantage since aqua regia dissolves gangue minerals, precious metals and base metals. This increases downstream recovery demands to isolate desired precious metals from leachate (Baghalha, 2012; Birich et al., 2019; Sousa et al., 2018; Syed, 2012).

$$Au + 11HCl + 3HNO_3 \rightarrow 2HAuCl_4 + 3NOCl + 6H_2O \qquad [2\text{-}5]$$
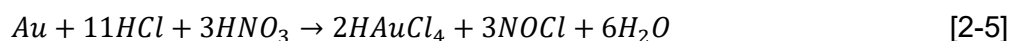
The *Kell-gold* hydrometallurgical leaching process for high-grade, run-of-mine ore containing platinum group metals has been demonstrated at pilot plant scale and relies on pressure oxidation, heat treatment and leaching with hydrochloric acid. The results of this study are misleading since

aqua regia decomposes gangue minerals, base metals and precious metals while cyanide can only dissolve free-milling gold in the ore. This process is therefore not selective of metals of interest in low-grade ore resources. Furthermore, no recommendations were made for downstream recovery processes required to isolate precious metals from leachate. Insufficient evidence has been presented in support of the Kell-Gold process to be considered as a viable technology for gold recovery from low-grade tailings (Adams et al., 2015; Liddell et al., 2019).

Thiourea ($CS(NH_2)_2$) is an organo-sulphur compound that is less toxic than cyanide, but it is under investigation as a potential carcinogen. Thiourea has seen limited commercial application and is predominantly considered for gold recovery (equation [2-6] adapted from (Syed, 2012)) from stibnite (Sb_2S_3) concentrates. Newmont concluded that thiourea was ineffective for gold recovery owing to large ore particle size (P_{80} = 1.27 cm) during a commercial scale trial on heap bio-oxidation followed by thiourea gold recovery. Thiourea requires an acidic medium (pH 1.4 to 1.8) and oxidants such as iron (III), hydrogen peroxide and ozone to create the high solution potentials required for gold dissolution. The predominant reasons for poor adoption of thiourea technology is the specialised materials of construction required to contend with low pH environment and high lixiviant consumption owing to decomposition to formamidine disulphide, elemental sulphur and dicyanamide [$CN(NH_2)$] that do not form complexes with gold. (Bhakta and Arthur, 2002; Deng and Liao, 2002; Dunne et al., 2009; Hiskey, 1984; La Brooy et al., 1994; Örgül and Atalay, 2002; Syed, 2012).

$$Au + 2CS(NH_2)_2 \rightarrow Au(CS[NH_2]_2)_2^+ + e^- \qquad [2\text{-}6]$$
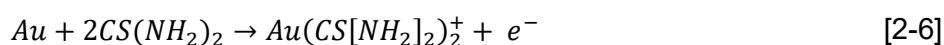
Thiocyanate (SCN^-) in combination with Fe (III) at pH 1-2 has been demonstrated on lab-scale for gold dissolution (equation [2-7] adapted from (Aylmore, 2016)) but not commercialised. Thiocyanate produces intermediates (($SCN)_3^-$ and $(SCN)_2$) that form complexes with gold. Thiocyanate is more stable to oxidation than thiourea and there is consensus among authors that thiocyanate is a more effective lixiviant for gold than thiourea and comparable to that of cyanide leaching systems. Lab scale investigation on run-of-mine and concentrate ores along with pilot-plant demonstrations of thiocyanate leaching systems are required to account for the commercial viability of thiocyanate as a cyanide alternative. In addition to no evidence of industrial manufacture of gold, thiocyanate has a Globally Harmonized System of Classification and Labelling of Chemicals (GHS) classification as acutely toxic to humans and long term hazard to aquatic environments and therefore will not be considered in this study. (Gökelma et al., 2016; Hilson and Monhemius, 2006; Li et al., 2012; National Library of Medicine, 2020; Syed, 2012).

$$Au + 4SCN^- + 3Fe^{3+} \rightarrow Au(SCN)_4^- + 3Fe^{2+} \qquad [2\text{-}7]$$
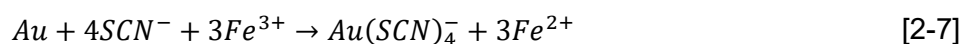
2.4.6.3. Conclusions

The lixiviant options for gold process flowsheets that have been demonstrated on a commercial scale is cyanide and thiosulphate. Synergistic glycine cyanide leaching systems are another viable

leaching alternative but require process development for carbon elution and subsequent recovery of gold from pregnant leach solutions. Commercial chlorine lixiviants systems focused on aqua regia in the past. Although chlorine lixiviants have a high gold leaching rate, they are not selective for gold. Considering the low gold head grades of mine tailings, high chlorine lixiviant consumption can be expected while downstream processing demands to isolate precious metals from base metals will be greater. The Kell-gold process has been demonstrated at pilot scale, but this technology still needs to be demonstrated to produce crude bullions of platinum group metals. This work will focus on flowsheet development for cyanide and thiosulphate gold recovery processes. Cyanide is the classic gold lixiviant that has been proven worldwide while thiosulphate is the leading contender for an environmentally friendly lixiviant that has been implemented commercially.

2.4.7. Solution purification

2.4.7.1. Carbon adsorption and elution

Cyanide leaching of gold ores followed by carbon adsorption and elution have been employed for free-milling gold recovery operations worldwide. Carbon-in-Leach processes are preferred in applications with low-grade pregnant solutions and when the ore source contains a large quantity of preg-robbing minerals (La Brooy et al., 1994; Marsden, 2006). Amidst the landscape of declining gold ore grades, the recent research on carbon adsorption focuses on approaches to improve gold recovery by investigating alternative activated carbon sources that yield improved gold recovery and elution together with process optimisation to reduce operational cost (Khosravi et al., 2017; Snyders et al., 2013; Syed, 2012). With reference to carbon elution technologies (Table 2-8) employed in industry, Anglo American Research Laboratory (AARL) and Pressure Zadra carbon elution processes are commonly employed while the Micron research process is used selectively in South African and Australian plants (Marsden and House, 2006).

Table 2-8: Summary of carbon elution process conditions employed in industry [Adapted from: (Marsden and House, 2006)].

	Pre-treatment	Elution agent	Temperature (°C)	Pressure (kPa)	Time (hours)	Maximum gold concentration (mg/L)
Zadra (Zadra, 1950)	-	10 g/L NaOH 1-2 g/L NaCN	90-100	101.3	36-72	150
AARL (Davidson and Duncanson, 1977)	20-50 g/L NaOH 10-20 g/L NaCN	Deionized H2O	110-120	170-200	8-14	1500
Murdoch University (Muir et al., 1985a)	80% acetonitrile (aqueous)	20-40% acetonitrile 2g/L NaOH 10g/L NaCN	25-70	101.3	8-14	1500-6000

Micron Research (Muir et al., 1985b)	20-50 g/L NaOH 50-100 g/L NaCN	60% - 80% Methanol (aqueous)	60-80	101.3	8-80	300

Based on the study by Muir et al. (1985b), within 1 hour, the percentage of gold eluted from carbon achieved by different elution procedures is ranked as Micron AN > Micron MeOH > AARL > Zadra. A reduced elution time period increases plant productivity, but the typical residence times for gold leaching is 12-48 hours. Carbon elution therefore will not be a process bottleneck when compared to leaching residence times. The AARL elution procedure elutes gold to the same levels as the Micron AN process within 6 hours elution time and has been commonly employed in flowsheets in South Africa. The decision for carbon elution process is therefore mainly based on economic considerations and the influence on gold loading capacity as summarised in Table 2-8. Thus the AARL elution procedure was employed along with carbon adsorption for the cyanide flowsheet in this study (Marsden and House, 2006; Muir et al., 1985a; Stange, 1999).

Acid washing with a 1-5% HCl solution prior to elution and reactivation is required to remove adsorbed organic materials and calcium that promotes gold complex adsorption to carbon. HCl corrodes reaction vessels and HNO_3 may be used as an alternative, but oxidation and deactivation of carbon occurs when HNO_3 is used requiring carbon inventories to be replenished more frequently (Stange, 1999).The activity of carbon deteriorates with each cycle of gold adsorption and elution and reactivation is required to maximise recovery of gold from pregnant solutions. Activated carbon is regenerated by live steam heating to 650-750 ºC in non-oxidising atmosphere (Marsden and House, 2006; Muir et al., 1985b; Stange, 1999).

2.4.7.2. Resin adsorption – cyanide leaching systems

The advantages and disadvantages of resin adsorption are summarised in Table 2-9 for cyanide leaching systems.

Table 2-9: Evaluation of the advantages and disadvantages of resin when compared to activated carbon as an absorption agent (Collated from: Adams, 2016; Fleming and Cromberget, 1984; Green et al., 2002)

Advantages	Disadvantages
Higher equilibrium gold loading rates of aurocyanide complexes (therefore smaller sized processes).	Specialised resins are expensive
Reduced susceptibility to poisoning by organic compounds (e.g. flotation reactants, solvents etc.)	Low selectivity for aurocyanide complexes compared to base metal cyanides complexes.
Does not require thermal regeneration unit process (reducing capital and operating expenses)	Resin particles are smaller than carbon. Retaining high cost resins in process requires investment retain them during filtration (i.e.

Advantages	Disadvantages
	acquisition and maintenance of narrow aperture sieves).
High recovery of gold in pre-robbing, leach-robbing and refractory ore pregnant solutions.	Resins have a lower density than carbon, therefore increased agitation costs are required to maintain suspension in solution.
Resin elution can occur at ambient conditions while carbon elution needs to be conducted at elevated temperatures and pressures (120 – 130 °C) increasing operating costs.	Complex multi-stage elution process to desorb base metals from resin before eluting gold.
Calcium carbonate has a greater affinity for carbon than resin. Therefore acid-washing treatment unit process will not be required following RIP adsorption.	Some elution reagents are expensive and have high consumption rates (e.g. thiourea irreversibly decomposes during the process).
Hematite, shales, clay category minerals reduce gold loading on carbon to a greater extent than it hampers adsorption on resin.	

For a typical mine tailings pregnant solution of 0.5 g/L gold in solution as determined by Green *et al.* (2002), an increase of 28% more gold is adsorbed on resin compared to activated carbon (Table 2-10). This increase in gold adsorption by resins may be compensated for by increasing the carbon inventory. Furthermore, gold selectivity on resins is poor (ratio of gold to metals: 0.14 in Table 2-10) and this increases downstream processing requirements to isolate gold from low-value base metals. The most significant disadvantage of resin adsorption over carbon is the low gold-selectivity and greater reagent cost compared to activated carbon.

Table 2-10: Comparison of Minix® to activated carbon as adsorption agent in pregnant solutions during counter-current plant campaigns [Adapted from: (Green et al., 2002)]

Adsorbent	% increase gold adsorbed on resin compared to carbon	Au/M[a]
High grade solution initial concentration		
Minix®	98.4%	0.64
Carbon		0.97
Standard solution initial concentration		
Minix®	44.1%	0.56
Carbon		0.90
Tailings solution initial concentration		
Minix®	28.8%	0.14
Carbon		0.64
[a]: Gold adsorption to total metal adsorption on loaded resin ratio. Indication of gold selectivity of resin.		

2.4.7.3. Recovery of thiosulphate complexes from pregnant solutions

Activated carbon reports poor gold loadings for gold-thiosulphate complexes $(Au(S_2O_3)_2^{3-})$. Gold recovery from ammoniacal-thiosulphate solutions with activated carbon from 51.6 g gold/tonne run of mine ore reported a maximum gold recovery of 43% at a carbon concentration of 60 g/L (1 hour adsorption) and was improved to 95% with 120 g/L carbon (6 hour adsorption) (Abbruzzese et al., 1995). Aylmore and Muir's (2001) interpretation of Abbruzzese et al.'s (1995) results conclude that the gold-carbon loadings are too poor to justify an economically viable process despite the high-grade ore being processed. Research on thiosulphate complex recovery from leach solutions focusses on the implementation of strong-base, ion exchange resins (with quaternary ammonium functional groups) in commercial processes over cementation and solvent extraction technologies. Barrick-Goldstrike has commissioned a gold recovery process with Purolite® A500C resin. Amberjet® 4200 resin systems were able to achieve 99% gold recovery while being more selective for gold than copper when compared to Purolite® A500C resin (Table 2-11). Dong et al. (2017) commented that most resins available were designed for $Au(CN)_2^-$ complexes. Therefore, a gap in literature exists for resin technology development that are selective for $Au(S_2O_3)_2^{-3}$ (Ahlatci et al., 2017; Aylmore and Muir, 2001; Choi, 2016; Dong et al., 2017; Grosse et al., 2003; Xu et al., 2017; Yu et al., 2015).

Table 2-11: Gold recovery of different strong base ion exchange resins from ammoniacal - thiosulphate pregnant solutions [Adapted from: (Grosse et al., 2003)]

Resins	$[S_2O_3^{2-}]$ (M)	[NH₃] (M)	pH	%Au	[Au]/[Cu] [a]	Reference
A500C (Purolite)	0.05 M	0.1 M	8.0	99.5	0.034	(Ferron et al., 1998)
AV-17-10P	0.5 M	0.5 M	5 – 11	94.2	-	(Kononova et al., 2001)
Amberjet® 4200	0.05 M	0.8 M	8 – 9.5	>99.4	2.39	(O'Malley, 2002)
Amberjet® 4200	0.05 M	0.2 M	9.5	99	-	(Nicol and O'Malley, 2002)
IRA-400 (R&H)	1.0 M	0.1 M	9 – 12	94.7	-	(Mohansingh, 2000)
Dowex (Dow)	-	-	-	76	-	(Mohansingh, 2000)
[a]: Gold to copper selectivity ratio on resin -: No data available						

Resins are not selective for gold as demonstrated in section 2.4.7.2 therefore multistage elution is recommended to separate base metals from precious metals. Table 2-12 summarises elution schemes proposed in literature. The trithionate-sulphite system is the only system implemented commercially by Barrick Gold while the rest are laboratory and pilot plant trials (Choi, 2016; Choi et al., 2015).

Table 2-12: Resin elution systems for gold and copper elution

	Copper Elution	Gold Elution	Reference
1	Thiosulphate $[S_2O_3^{2-}]$: 0.5 M Copper elution: 100% Ambient conditions. Eluant flowrate: 5 BV/h*. Bed volumes:14	Trithionate and sulphite $[S_3O_6^{2-}]$: 0.2 M and $[SO_3^{2-}]$: 0.1 M Gold elution: 100%. Ambient conditions. Eluant flowrate: 5 BV/h*. Bed volumes:13.	(Jeffrey et al., 2010)
2	Thiosulphate $[S_2O_3^{2-}]$: 0.5 M Copper elution: 100% Ambient conditions. Eluant flowrate: 5 BV/h*. Bed volumes:14	Sodium chloride and sulphite $[NaCl]$: 2 M and $[SO_3^{2-}]$: 0.1 M Gold elution: 100%. Ambient conditions. Eluant flowrate: 2 BV/h*. Bed volumes: 8.	(Jeffrey et al., 2010)
3	Thiosulphate $[S_2O_3^{2-}]$: 0.5 M Copper elution: 100% Ambient conditions. Eluant flowrate: 5 BV/h*. Bed volumes:14	Nitrate and sulphite $[NO_3^-]$: 1 M and $[SO_3^{2-}]$: 0.1 M Gold elution: 100%. Ambient conditions. Eluant flowrate: 2 BV/h*. Bed volumes: 6.	(Jeffrey et al., 2010)
4	Thiosulphate elution. $[S_2O_3^{2-}]$: 150 g/L (1.33 M) Copper elution: 99.9%. Ambient conditions. Eluant flowrate: 2 BV/h*. Bed volumes: 4.	Thiocyanate elution $[SCN^-]$: 100 g/L (1.72 M) Gold elution: 99.5%. Ambient conditions. Residence time: 2-4 hours. pH 6-8 (natural). Eluant flowrate: 2 BV/h*. Bed volumes: 6.	(Fleming et al., 2003)
5	Thiosulphate elution. $[S_2O_3^{2-}]$: 150 g/L (1.33 M) Copper elution: 99.9%. Ambient conditions. Eluant flowrate: 2 BV/h*. Bed volumes: 4.	Polythionate $[S_3O_6^{2-}]$: 40 g/L (0.21 M) and $[S_4O_6^{2-}]$: 80 g/L (0.36M) Gold elution: 99.4%. Ambient conditions. pH: 1.5 (natural). Eluant flowrate: 2 BV*/h. Bed volumes: 8.	(Fleming et al., 2003)
6	Thiosulphate elution. $[S_2O_3^{2-}]$: 150 g/L (1.33 M) Copper elution: 99.9%. Ambient conditions. Eluant flowrate: 2 BV/h*. Bed volumes: 4.	Trithionate $[S_3O_6^{2-}]$: 200 g/L (1.04 M) Gold elution: 99.8%. Ambient conditions. pH: 6-8 (natural). Eluant flowrate: 2 BV/h*. Bed volumes: 8.	(Fleming et al., 2003)
7	Ammonia. $[NH_3]$: 2 M Copper elution: 90%. Ambient conditions. Eluant flowrate: 5 BV/h*. Bed volumes: 30.	Ammonium nitrate. $[NH_4NO_3]$: 2 M Gold elution: 90%. Ambient conditions. Eluant flowrate: 5 BV/h*. Bed volumes: 30.	(Nicol and O'Malley, 2002)

For copper elution, thiosulphate is an effective eluant in three separate studies and has been implemented commercially at Barrick Gold (Table 2-12) and was recommend for the thiosulphate flowsheet in this work. Ammonia poses an environmental burden and requires water treatment before release to environment and was not considered in this work (Choi, 2016; Choi et al., 2015). Thiocyanate and ammonium nitrate eluants for gold elution pose an environmental impact and needs to be destroyed before discharge to the ecosphere and therefore was not considered in this work

(Muir and Aylmore, 2004; Syed, 2012). Sodium chloride accumulates in the leaching unit process and therefore introduces an additional impurity into the process. Sodium chloride elution does have the benefit that resin regeneration is not required since chlorides are readily displaced by metal complexes on ion exchange resins. Sodium chloride-sulphite elution has not been demonstrated commercially and therefore was not considered (Jeffrey et al., 2010).

Trithionate and tetrathionate eluants can be generated in-situ thereby reducing the costs associated with raw material acquisition. Furthermore, thiosulphate waste streams may be recycled to manufacture eluant. The drawback of tri- and tetra- thionate elution is that resin regeneration is required since polythionates adhere more strongly to ion exchange resin than precious metal complexes (Fleming et al., 2003). The natural degradation of thiosulphate to produce polythionates is a drawback of thiosulphate leaching systems. Therefore, resin elution and regeneration allow an opportunity to firstly manage polythionate build up in the leaching- and elution- circuit. In addition, converting polythionates back to thiosulphate reduces thiosulphate resource consumption. Excessive thiosulphate reagent consumption is one of the predominant reasons for not implementing thiosulphate over cyanide as a lixiviant in flowsheets. Following the above discussion, entry 1 in Table 2-12 was recommended for gold elution and has been demonstrated commercially (Choi, 2016; Choi et al., 2015).

2.4.7.4. Conclusion

For the cyanide flowsheet, carbon adsorption and AARL elution technology for gold recovery was selected. The basis for the decision was the poor selectivity of gold in resin systems which requires a multiple-step elution process. The costs associated with additional unit processes along with additional by-product streams that need to be scavenged for value or disposed of increases downstream processing requirements. For the thiosulphate flowsheet, Amberjet® 4200 ion exchange resin recovery from pregnant solutions was recommended owing to the poor gold recovery by carbon adsorption technologies which require uneconomical carbon inventories to improve gold recovery. Resin adsorption is followed by multi-stage elution (entry 1 in Table 2-12). Resins commercially available are not selective for $Au(S_2O_3)_2^{-3}$ complexes. Thus, there is a demand in literature for ion exchange resins that are tailored to adsorption of $Au(S_2O_3)_2^{-3}$ complexes.

2.4.8. Recovery

Following leaching unit processes, gold needs to be recovered from the pregnant solution. Zinc precipitation and electrowinning are technologies employed commercially to recover gold from solution and will be discussed in this section.

2.4.8.1. Zinc Precipitation

Recommended zinc precipitation process conditions are summarised in Table 2-13. Dissolved oxygen concentration and soluble sulphide in the pregnant leach solution should be minimised and

free NaCN concentration be maintained to prevent the formation of passivating zinc hydroxide and zinc sulphide layer on zinc surfaces. The passivation of zinc surfaces reduces precipitation efficiency and thereby increasing zinc powder consumption. The maximum dissolution of zinc occurs at NaCN concentration of 0.1 to 0.3 g/L within the pH range pH 10.5 to 11.2. Free cyanide levels exceeding the recommended range causes dissolution of zinc within the reaction solution and increases zinc consumption. The addition of lead nitrate increases redox potential by 0.2 V in the positive direction creating favourable cathodically charged, redox conditions for the precipitation of precious metals on zinc surfaces (Fleming, 1992). The lead polarized regions on zinc particles inhibit the formation of zinc hydroxide thereby allowing the entire surface of the particle to be used for precipitation (Adams, 2016; Chi et al., 1997; Mpinga et al., 2015; Nicol et al., 1979). The presence of silver cyanide complexes in solution reduces the maximum loading capacity for gold on activated carbon leading to reduced gold recovery. Therefore, in pregnant leach solutions, where the fineness (silver: gold ratio) exceeds 2 or 3:1 Merrill-Crowe zinc precipitation is favoured over carbon adsorption and recovery.

Table 2-13: Summary of recommended zinc precipitation process conditions for optimum recovery

Reference Property	(Wartenweiler, 1949)	(Chi et al., 1997)	(Adams, 2016)	(Mpinga et al., 2014)
Dissolved oxygen concentration	0.5 - 1.3 mg/L	0.5 mg/L	1 ppm	< 1 ppm
Cyanide concentration	-	100 - 300 mg/L	150 - 200 mg/L	150 ppm
Zinc concentration	-	-	2 - 4 kg zinc per metal of interest precipitated	1.5 - 3.4 times stoichiometric requirement
Soluble sulphide	-	-	20 mg/L	-
Lead nitrate	-	10-15 mg/L	0 - 0.2 kg per kg zinc	-
Temperature	-	-	-	60 °C

2.4.8.2. Electrowinning

Electrowinning of metals follows the elution step of resins and activated carbon. Aqueous eluates are subjected to an induced voltage within an electrochemical cell whereby redox reactions occur resulting in the reduction of metals at the cathode. For electrolytes with low gold concentrations compared to base metals, the co-deposition of base metals occurs on the cathode. If copper to gold ratio exceeds 4:1 in the electrolyte, copper cyanide complexes will deposit preferentially on cathodes over gold-cyanide complexes. Furthermore, the qualitative assessment of gold adhesion to the stainless steel (SS 304) cathodes revealed that the adhesion of gold to cathodes increased in electrolytes with high copper concentration (Steyn and Sandenbergh, 2004). This occurrence complicates the subsequent refining operations to produce a pure gold bullion, since copper cannot be recycled effectively during hydrometallurgical refining operations (Sole and Paul, 1986). In

addition, during Miller chlorination refining, copper chlorides have a higher boiling point than other base metal chlorides and therefore increases energy requirements to purify gold bullion (Auerswald and Radcliffe, 2005) . Marsden and House (2006) recommend zinc cementation over electrowinning for gold recovery when copper concentrations in pregnant solutions exceed 100 – 200 ppm.

For thiosulphate leaching systems, electrowinning has been favoured over cementation and solvent extraction for gold recovery. Electrowinning has been used in the commercial flowsheet proposed by Choi (2016). In lab-scale studies in literature on recovery of gold from eluted resins and carbon 98-99% gold recovery was achieved after electrowinning (Abbruzzese et al., 1995; Aylmore and Muir, 2001; Marsden and House, 2006; Steyn and Sandenbergh, 2004; Sullivan and Kohl, 2019).

2.4.8.3.   Conclusions

For the West Rand tailings feedstock, the silver to gold ratio is 21:1, therefore there is a strong likelihood that the silver to gold ratio will exceed 2:3 to 1 in the pregnant leach solution. Zinc precipitation technology was thus recommended over carbon adsorption for the main gold recovery stream for cyanide flowsheet (Adams, 2016; Bosch, 1987; Marsden and House, 2006). The copper to gold ratio is 150:1 in the feedstock (Table 2-3) and may exceed the 4:1 copper to gold ratio in the pregnant leach solution, a criteria recommended by Steyn and Sandenbergh (2004). Based on this, zinc cementation was selected over electrowinning for gold recovery from pregnant leach solution in the cyanide flowsheet. For the thiosulphate flowsheets, after dual-stage elution of loaded resins, electrowinning was recommended. Dual-stage elution minimises copper reporting to the electrowinning circuit and improves overall crude gold bullion purity.

2.4.9.   Calcination and smelting

Calcination follows cementation operations to oxidise lower value base metals to their respective oxides. This allows the lower value metals to report to the slag phase during smelting operations, thereby producing a higher value crude bullion and reduces the sizing requirements of upstream unit processes. Calcination control parameters of excess oxygen and temperature (600-700 °C) are optimised to ensure precious metals are not oxidised which result in process loss (Nicol et al., 1987). Smelting produces a crude gold bullion between 1 200-1 400 °C that is a precious metal alloy (Adams, 2016; Marsden and House, 2006). The crude bullion produced is a gold metal alloy of approximately 60% gold by mass depending on precious and base metals present in the ore feedstock. In South Africa mine houses sell crude bullion to Rand Refinery to produce a pure gold bullion with minimum 99.5% gold by mass for niche applications or to 99.99% by mass for use as a monetary commodity (Auerswald and Radcliffe, 2005; Sole, 2007). This work focussed on technologies for gold extraction from mine tailings and therefore the system boundary was drawn up until the production of a crude gold bullion to align with the practices in South Africa of mine houses selling crude gold bullions to the Rand refinery.

2.4.10.   Effluent treatment

The effluent of gold recovery processes to tailings dams contains toxic substances that need to be removed to comply with environmental regulations. In flowsheets that rely on cyanide as a lixiviant for gold recovery, cyanide needs to be removed or recovered before discharge to tailings dams. In applications where the tailings will be used for backfilling of decommissioned mines, the complexation using iron precipitation (with $FeCl_3$ or $FeSO_4$) to capture cyanide species as environmentally benign precipitate Prussian blue [$Fe_4(Fe(CN)_6)_3$]. Iron precipitation does not reduce cyanide concentrations in effluent water to below 5 mg/L and therefore is not suitable for applications where cyanide waste streams are discharged to the environment. Unfortunately, Prussian blue is unstable in alkaline environments (pH > 7) and dissociates back to ferrocyanide complexes ($Fe(CN)_6^{4-}$ and $Fe(CN)_6^{3-}$). With iron precipitation, cyanide is not destroyed but merely captured and the potential for cyanide release still exists if effluent tailings and water is transferred to an alkaline environment. The context of the current work is to minimise cyanide in the process effluent streams and to allow tailings solids to be used as a raw material within the circular economy for aggregate, bricks and ceramics. Thus, iron precipitation as a cyanide removal technology merely contains the environmental burden it does not reduce environmental impact and will not be viable as a process solution unless the application for depleted mine tailings is for mining backfill (Adams, 1992; Mudder et al., 2001; SGS Mineral Services, 2005).

 Kuyucak & Akcil (2013) reviewed cyanide removal technologies for extractive mineral processes and concluded that alkaline chlorination is the traditional process technology for cyanide destruction in gold flowsheets. Alkaline chlorination decomposes cyanide species to carbon dioxide and nitrogen which are bio-available products along with sodium chloride as a by-product. Carbon dioxide may contribute to greenhouse gas emissions, but it is a bioavailable resource that can be captured through natural attenuation therefore alkaline chlorination reduces the environmental impact of cyanide. One notable drawback of alkaline chlorination is that ferrocyanide complexes cannot be destroyed through alkaline chlorination and therefore this technology needs to be combined with iron precipitation to complement this technology. Iron precipitation ensures strong cyanide metal species are not bioavailable provided the environment pH does not become alkaline. Furthermore, the intermediates of the alkaline chlorination are toxic and therefore process control measures need to be implemented to ensure reaction completion. Free chlorine in solution is another challenge, therefore aeration and polishing ponds can be implemented after treatment to ensure complete conversion of reagents. The use of sodium hypochlorite is favoured over chlorine gas owing to the occupational health and safety risk of chlorine gas (Botz et al., 2016; Mudder et al., 2001).

Recent research has focused on the implementation of SO_2/Air, biological and hydrogen peroxide cyanide destruction technologies and has been employed in commercial plants. The SO_2/Air process is viable; however, strict process control is required therefore the ease of use of this technology to

ensure cyanide destruction still requires optimisation research before it can be considered mature enough for implementation. Cyanide destruction such as hydrogen peroxide, INCO SO_2/Air, Caro's acid and ozonation treatment have the same flaw of not being able to destroy ferrocyanide complexes therefore iron precipitation needs to be recommended for these processes as well. (Breuer and Hewitt, 2020; Kuyucak and Akcil, 2013).

Thiosulphate is considered environmentally benign, but thiosulphate can decompose to polythionate species under reducing conditions and can cause oxygen depletion in aquatic ecosystems. Hydrogen peroxide is used extensively in industrial and domestic wastewater treatment to reduce biological oxygen demand (BOD) and chemical oxygen demand (COD). Ammonia gas has the same class of threshold limiting value for air emissions as hydrogen cyanide gas (14 mg/m^3) and in aquatic environments is in the same category as chlorine. Ammonia decomposes to form nitrates in aquatic ecosystems leading to eutrophication and groundwater pollution (Ahmad, 2012; Ilyas and Lee, 2018; Muir and Aylmore, 2004).

Barrick Goldstrike relied on non-ammoniacal thiosulphate leaching and recycled thiosulphate by reverse osmosis water treatment of thickened tailings overflow (flowsheet illustrated in Figure 2-8) to create a permeate. The permeate is subjected to thiosulphate regeneration to recover thiosulphate. Within the closed flowsheet, the thiosulphate and polythionate are decomposed to sulphate (SO_4^{2-}) which is precipitated as gypsum ($CaSO_4.2H_2O$) thereby mitigating the emission of harmful thiosulphate by-products to the ecosphere. Reverse osmosis (RO) is a costly water treatment technology and requires microfiltration water treatment technology of feed to prevent fouling of RO membranes. Barrick gold reported RO membrane fouling as a bottleneck to operate at design capacity of gold recovery plant (Choi, 2016; Choi et al., 2013).

For the cyanide flowsheet, a combination process of alkaline chlorination and iron precipitation with ferrous sulphate was recommended for the cyanide process flowsheet. The basis for this recommendation is its abundance in the commercial gold processing industry while breaking down free-, WAD- and strong- cyanide complexes to environmentally inert chemical species and the ferrocyanide unaffected by chlorination is converted to Prussian blue by iron precipitation. For the thiosulphate reverse osmosis was recommended for water treatment to recover thiosulphate, reducing freshwater consumption and emission of thiosulphate as sulphate and gypsum to the ecosphere from the flowsheet. The use of non-ammoniacal thiosulphate system has the benefit of not requiring ammonia destruction unit processes and thiosulphate can be regenerated thereby reducing overall thiosulphate consumption.

2.5. Environmental impact assessment tools

The present study endeavours to evaluate the environmental impacts of process flowsheets. This requires environmental impact assessment (EIA) tools to quantify the environmental consequences

as illustrated in Table 2-14. Loiseau *et al*. (2012) conducted a qualitative review of EIA tools and summarised which territories in the scope of life cycle thinking are addressed by specific tools. The qualitative framework rates each EIA tool on a scale from "-" (tool does not address criteria) to "+++" (the criterion is comprehensively covered by tool).

From Table 2-14 it is clear that there isn't a single EIA tool that addresses all the life cycle thinking criteria strongly. This research project would benefit from a thorough inventory of pollutants, resource usage, water usage, greenhouse gas emissions, and a strong focus on life cycle thinking along with a robust, well-established methodological framework that is supported within research and industrial application. The qualitative assessment in Table 2-14 indicates that the only EIA that ranks the highest based on the framework developed by Loiseau *et al.* (2012) is life cycle assessment. Emergy, Ecological footprint, Physical input – output table EIA tools are next in line for consideration, but the focus on pollutants, greenhouse gas emissions needs to be improved and the methodologies need to be standardised universally to be considered for this study.

A limitation of LCA methodology is that it only considers the environmental impacts of the elementary inputs and outputs of a defined system boundary and does not offer an account of the resource usage efficiency for the system boundary. The optimisation of resource usage is an important design criterion when developing chemical processes and is an opportunity for improvement in the LCA methodology. Exergetic life cycle assessment (ELCA) incorporates the EIA tool of exergy analysis to assess the thermodynamic efficiency of individual unit processes and thereby accounting for the efficient utilisation of resources with two additional performance criteria, namely the life cycle irreversibility and exergy loss (Amini et al., 2007; Cornelissen and Hirs, 2002; Gößling-Reisemann, 2008)

Table 2-14: Qualitative assessment of environmental assessment methods [Reproduced and modified from: (Loiseau et al., 2012)]

Key features and Criteria	Environmental impact assessment tools								
	HERA	**EF**	**MFA**	**SFA**	**PIOT**	**ENA**	**Exergy**	**Emergy**	**LCA**
Formalisation									
Methodological Framework	+++	++	++	+	+	+	++	++	+++
System modelling									
Life cycle thinking	-	+	+	-	+++	+++	+	+++	+++
Inventoried flows									
Non-renewable resources	-	-	+++	++	++	++	+++	+++	+++
Renewable resources	-	++	++	-	++	++	++	+++	+
Water resources	-	-	+	-	++	++	+	++	++
Land use	-	+++	-	-	+	-	+	+	+
Greenhouse gas emissions	-	++	++	-	++	+	+	+	+++
Pollutant emissions	+++	-	+	+++	+	+	+	+	+++
Indicators provided									
Multi-criteria assessment	-	++	++	-	++	+	++	++	+++
Spatial differentiation	+++	+	-	++	-	-	+	+	++
Usability									
Feasibility	+	++	++	+	+	+	+	+	++
Understanding	+	+++	++	++	++	+	+	+	+
Total + (out of 36)	**11**	**19**	**18**	**11**	**19**	**15**	**17**	**21**	**27**
Abbreviations for EIA tools: HERA: Human and environmental risk assessment; EF: Ecological footprint; MFA: Material flow analysis; SFA: Substance flow analysis; PIOT: Physical input – output table; ENA: Ecological network analysis;									

2.6. Life cycle assessment (LCA)

LCA is a quantitative modelling tool employed to assess the potential environmental impacts during the manufacture of products by evaluating the inputs and outputs of processes assimilated to yield a product. LCA has a focus on reducing environmental impacts of the entire life cycle of products by employing a "cradle-to-grave" EIA philosophy to account for the full life cycle impacts of a product as opposed to the "gate-to-gate" philosophy. The "gate-to-gate" perspective allows product manufacturers to disregard the environmental impacts related to resource manufacture, product use and disposal and only accounts for 35% of environmental impacts related to product manufacture. The "cradle-to-gate" perspective is another partial LCA that considers ecological impacts from resource extraction to the factory gate before dispatch to customer. Therefore, usage and final disposal is excluded from system boundary and has a 67% as compared to a "cradle-to-grave" detailed LCA in the "cradle-to-gate" perspective. The "gate-to-grave" perspective only considers the downstream environmental impacts of the use and disposal phases related to the main product produced (Awuah-Offei and Adekpedjou, 2011; Guinée et al., 2011; Hunt et al., 1998; Klöpffer and Grahl, 2014; Rebitzer et al., 2004).

2.6.1. LCA methodology

A description of the methodology for LCA is illustrated in the diagram below (Pennington et al., 2004; Rebitzer et al., 2004):
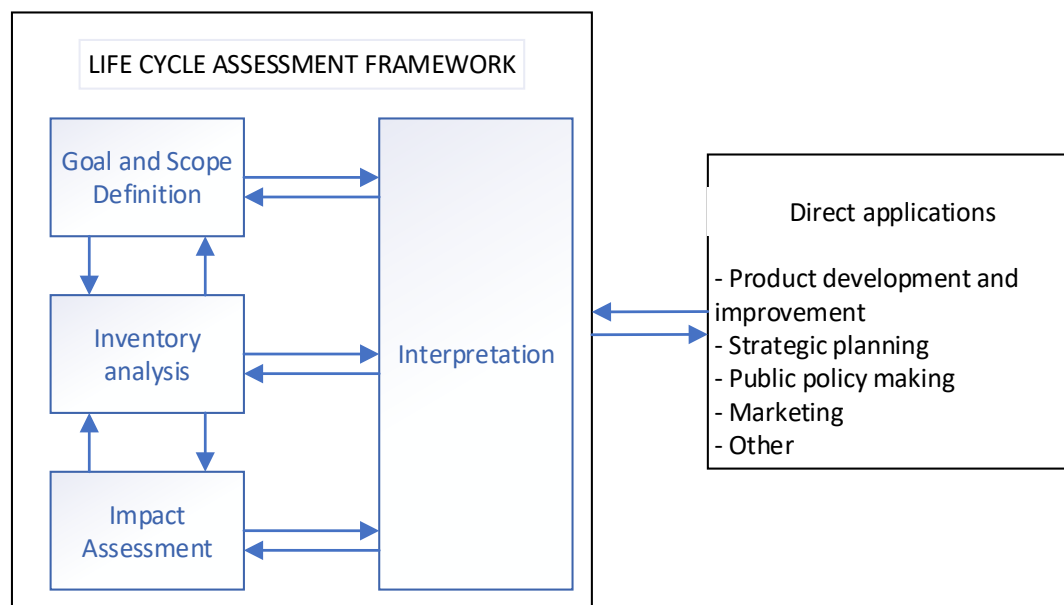
Figure 2-9: LCA stages [Adapted from: (ISO, 2006b)]

Where the following definitions apply to Figure 2-9:

1. **Goal and scope definition:** Definition of the intention of the study, assumptions, product/system boundaries, selection of impact categories, target audience, and confidentiality rating

2. **Life cycle inventory analysis (LCI phase):** Collection of inventory data for inputs (e.g. raw material, energy, water resources requirements, etc.), outputs (e.g. major products and by-products), and waste (e.g. air, water and land emissions, and waste streams)

3. **Life cycle impact assessment phase (LCIA):** the compulsory steps of *classification and characterisation* involves the translation of elementary inputs and output data from LCI phase into respective indicators that contribute to impact categories. The optional steps of *normalisation* involve relating LCIA results to a baseline to a reference while *grouping and weighting* involves the qualitative or semi-quantitative approach of ranking LCIA results in order of importance

4. **Life cycle interpretation phase:** Discussion of outcomes from LCI and LCIA phases of study as a basis for decision making based on criteria established during the scoping phase.

The optional LCIA steps of *grouping and weighting* is a subjective approach determined by the LCA practitioner as to which impact categories are most relevant to the LCA study. This practice is contested in the scientific community since the basis for *grouping and weighting* is qualitative and the methodologies employed for this have not been standardised. For this reason, according to the ISO 14 040 framework, LCA results that are disclosed to public must exclude any deductions and conclusions made based on the results obtained through *grouping and weighting* of impact categories. There are also some ethical implications where impact categories may be misrepresented and omitted through *grouping and weighting* and thereby resulting in incorrect and fraudulent conclusions being made (ISO, 2006b; Pennington et al., 2004). According to Segura-Salazar *et al.* (2019), the *normalisation, grouping and weighting* steps have been ignored in many mining LCA's. For these reasons, *normalisation*, *grouping and weighting* shall be excluded from the scope of this work.

LCIA methodologies are tailored to specific geographical locations, time periods, industries, or strongly focused on specific environmental impact categories while having weak predictions for other impact categories and therefore require further development (Hauschild et al., 2013). In literature, there is no universal LCIA methodology that can be applied to all contexts. The philosophies, assumptions, and uncertainties employed when developing models and characterisation factors vary greatly between LCIA methodologies for the same chemical species. The ReCiPe® 2016 LCIA methodology groups characterisation factors into 3 distinct perspectives to account for different assumptions and decisions regarding uncertainties in the estimation of the characterisation factors. These cultural perspectives are named Individualistic, Hierarchist, and Egalitarian perspectives. In the present study, the LCIA results for the ReCiPe® Hierarchist perspective were employed because there is consensus within the scientific community concerning the chronological period and credibility of the impact pathways proposed that contribute to the calculation of environmental indicators (Huijbregts et al., 2016).

LCA modelling is based on a cause-and-effect chain between the effects of anthropogenic activities on the environment as illustrated in Figure 2-10. LCA results can be reported at either midpoint or endpoint level along the cause-and effect-chain. Midpoint impact assessment reflects the change that emissions cause to targeted systems in the ecosphere such as the increase of pollutants in water, land and air environments. Endpoint impact assessment predicts the ultimate effect that the collective changes to habitats have on the overall ecosystem such as reduction in quality of human life, extinction of fauna and flora and depletion of non-renewable resources. Midpoint analysis is more detailed but requires LCA domain knowledge. While endpoint analysis has fewer impact categories but facilitates communication of environmental impacts by expressing results in terms of three key areas of protection. The statistical uncertainty of endpoint results is higher since data shortages and assumption made accumulate as one progresses down the LCA cause-and-effect chain. A combination of midpoint and endpoint reporting modes is recommended in literature to appeal to a variety of audiences both knowledgeable about LCA and for those who only require an overview (Bare et al., 2000; Hauschild et al., 2013; Huijbregts et al., 2017)
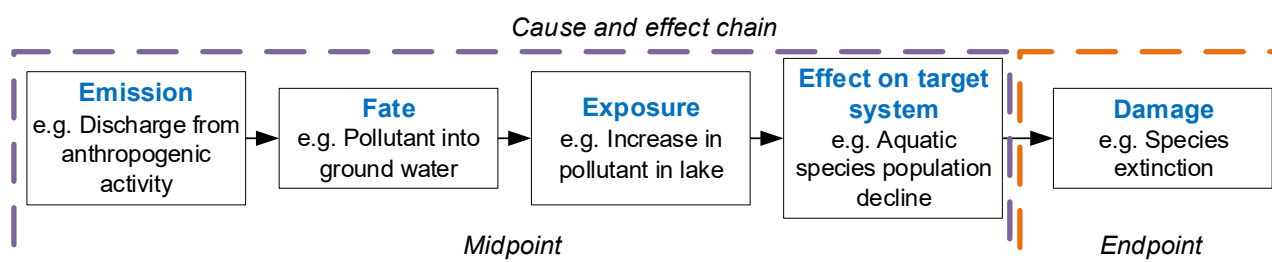
Figure 2-10: LCA cause-and-effect chain indicating the areas that midpoint and endpoint analysis report [Adapted and modified from: (Klöpffer and Grahl, 2014; Rebitzer et al., 2004)

2.6.2. LCA use in the mineral processing industry

In an attempt to standardise LCA practices within the mining industry, Segura-Salazar *et al.* (2019) conducted a review of LCA practices and identified that the ReCiPe® LCIA methodology is frequently applied in mineral processing for predicting all impact categories other than climate change. The ReCiPe® LCIA methodology was also used in LCA studies on gold mining from primary gold ore resources conducted by Abadías Llamas *et al.(*2019), Chen *et al.* (2018) and Elomaa *et al.* (2019). In 2016, the ReCiPe® LCIA methodology has been updated to account for shortcomings therefore the developers endeavour to update this methodology based on current research relating to LCA (Hauschild et al., 2013; Huijbregts et al., 2016). Therefore, to aid in comparison of results with literature data and to align with the approach followed in the mineral processing industry the ReCiPe® 2016 LCIA methodology will be used in this work.

Table 2-15 presents the incidence of LCA studies related to the mining industry when a query was conducted using specific keywords on the Compendex database. This indicates that less than 3% of the studies are related to the mining discipline and less than 0.1% is dedicated to the gold mining discipline. Awuah-Offei and Adekpedjou (2010) conducted a similar database search and reports

that 0.84% of studies conducted are related to the mining industry. This suggests that there is a low frequency of environmental life cycle, systems thinking methodology being applied to process design of mining processes in academic literature. An alternative perspective may be that LCA's are proprietary information and aren't disclosed, other EIA tools are used, or the methodology has specific limitations that has rendered its application to the mining industry ineffective(Awuah-Offei and Adekpedjou, 2011; Segura-Salazar et al., 2019).

Table 2-15: Results for keywords related to Life Cycle Assessment in the mining industry on database search of Compendex database conducted.

Database	Keyword (s)	Results	Percentage[a]
Compendex (Engineering Village)	"Life Cycle Assessment"	33 487	-
	"Life Cycle Assessment" AND "Mining"	1 107	3.31
	"Life Cycle Assessment" AND "Mining" NOT "Data"	737	2.20
	"Life Cycle Assessment" AND "Mining" NOT "Data" AND "Gold"	29	0.09
[a]: $Percentage = \dfrac{Results\ for\ [Keyword\ (s)]\ search\ on\ database}{Results\ for\ "Life\ cycle\ assessment"\ search\ on\ database} \times 100.$			

Based on the aforementioned information, more LCA's need to be conducted within the mineral processing discipline to understand what deficiencies are present in the methodology. LCA's for gold recovery processes from mine tailings have not been conducted and is an area where this work contributes to the academic body of knowledge. Considering that LCA has been adopted into the ISO 14 000, LCA could benefit mine houses in complying with global standards to aid stakeholders in gaining confidence in their business operations.

2.7. Process modelling

The benefit of a modelling approach is that it allows the evaluation of many different process flowsheet solutions. The disadvantages of the modelling approach are that the results obtained are subject to the accuracy, reliability and validity of the databases used and the inventory data gathered. It is the responsibility of the researcher to ensure due diligence and that limitations are documented in the relevant sections of the work (section 3.4.4). HSC Sim® and GaBi® software has been used in literature for process modelling to generate inventory data and assess the environmental aspects of processes (as summarised in Table 2-16). This methodological approach is promising and will be used to achieve the objectives in this study.

Table 2-16: Summary of papers using HSC Sim® together with GaBi® for conducting life cycle assessment

Study	Reference
Life Cycle Assessment of Metallurgical Processes Based on Physical Flowsheet Models	(Reuter et al., 2016)
A thermodynamic-based life cycle assessment of precious metal recycling out of waste printed circuit board through secondary copper smelting	(Ghodrat et al., 2017)
Simulation-based exergy, thermo-economic and environmental footprint analysis of primary copper production	(Abadías Llamas et al., 2019)
Process simulation and gate-to-gate life cycle assessment of hydrometallurgical refractory gold concentrate processing	(Elomaa et al., 2020)

2.8. Conclusions

More comprehensive inventory of geochemical and mineralogical data of South African tailings dams are necessary as outlined by authors quoted in this section. This will aid in prioritising the most lucrative projects for mine tailing's reclamation for minerals of interest along with the tailings dams that pose high environmental burden. This data can be the initiating point for stakeholders in the mineral processing industry to identify opportunities and develop solutions to ensure the resilience of this industry by chartering a course of action when market demands for minerals are favourable. Action plans for tailings dams that pose a large high environmental threat need to be established for at a minimum the containment of exposure to prevent the spread of toxic substances beyond the tailings dams.

Specifically, more comprehensive gold deportment studies are necessary for accurate flowsheet development to specific mineral phases for Witwatersrand mine tailings. Particle size fraction grading analysis data of mine tailings of gold recovery below 53 µm along with additional diagnostic leaching tests and QEMSCAN analysis is necessary in literature to verify if gold is associated within silica quartz matrix. Heavy liquid separation and gravity separation trials are necessary to rule out gravity separation techniques as flowsheet alternatives. The ideal of developing a process flowsheet for 100% gold recovery has the disadvantage of increasing capital and operational expenditure and owing to the low gold grade in tailings resources, it may render proposed flowsheet unprofitable. Therefore, the process flowsheet development in the next section needs to prioritise gold recovery in the following order: free-milling gold > gold in silicates > gold in carbonates > gold in sulphides > gold in preg-robbing minerals > gold in carbonaceous minerals.

There is a shortage in literature of commercially employed alternatives to ultra-fine grinding to liberate gold occluded in finer particle size fractions of quartz minerals below 75 µm. Minerals such as quartz have a mineral hardness of 7 on the Mohr scale and require energy intensive processes to liberate occluded minerals. The economically feasible, technological capability to liberate finely

disseminated gold within minerals below 75 µm will allow access to 11.8% of gold occluded in silicate minerals.

43

3 METHODOLOGY

This chapter explains the methodological framework employed for the development- and assessment of- environmental impacts of processes to achieve the aim of reducing ecological impacts while recovering value from mine tailings (as illustrated in Figure 3-1). A mineralogical survey of tailings dams in the case study region was performed to identify potential value streams and environmental impacts of mine tailings (discussed in section 3.1). The mineralogical data was then evaluated to identify baseline processes in literature as the current state of art employed in commercialised processes to meet the aim of this study. Mass and energy balances were conducted for the proposed baseline flowsheet followed by life cycle assessment (LCA) to quantify the environmental impacts of technology currently employed in industry for recovery of value from mine tailings. After the baseline environmental impact of the conventional process was established, an alternative flowsheet was developed based on literature to identify opportunities to further reduce environmental impacts. The simulation and LCA results of both flowsheets were compared against one another and environmental regulations to identify the process that minimises the burden to the environment. Opportunities to further reduce environmental impact of the process flowsheets proposed were identified for and an account of emissions to ecosphere provided.
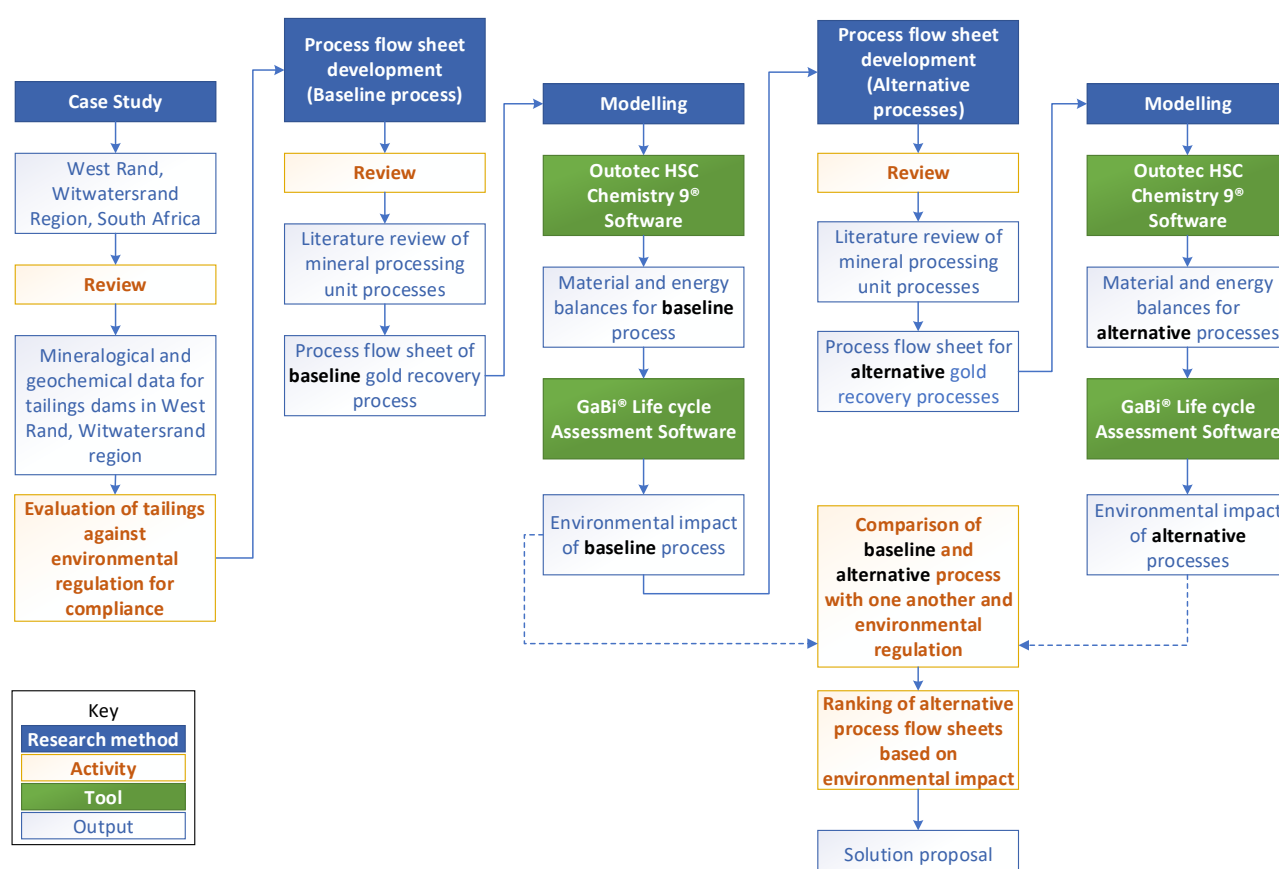
Figure 3-1: Methodological framework for project

44

3.1. Case study

Process flowsheet design requires mineralogical data of the mine tailings to determine the sequence of unit processes that will yield the greatest recovery of a mineral of interest (MOI). The characteristics of mine tailings varies depending on run-of-mine ore composition, process flowsheet used to recover MOI on the first-pass recovery, reagents used and the age of the tailings dam. A literature survey was conducted to source mineralogical data as depicted in Figure 3-2 to understand the deportment of minerals of interest and quantifying environmental impact of mine tailings by assessing results for compliance with National Environmental Management: Waste act 2008 (Act No. 59 of 2008) (NEMA)(Department of Environmental Affairs, 2014). In this work, a case study approach was used to create a hypothetical sample as a process feedstock which consists of West Rand, Witwatersrand region mine tailings. A case study approach has the benefit of tailoring a process flowsheet solution to a specific region where mine tailings have similar properties. The methodology employed in this work may be followed as a framework for other regions provided the mineralogical information for the mine tailings are known.
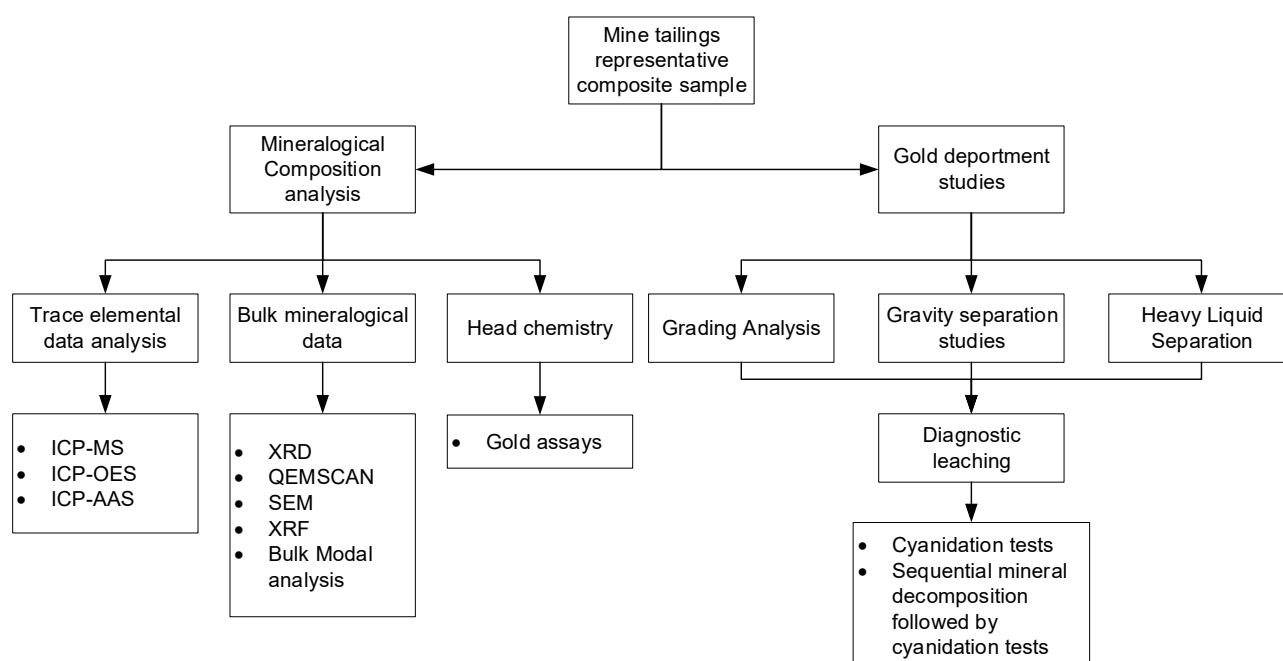
Figure 3-2: Flow diagram of mineralogical and geochemical data that will be considered for process flowsheet developed for mine tailings in Witwatersrand region [Adapted from: (Coetzee et al., 2010; Lorenzen, 1995)]

3.2. Process flowsheet development

By 2031, South African's secondary ore resource of mine tailings may have higher gold grades than primary ore resources as discussed in section 2.1 and illustrated in Figure 2-1. The short timeline until 2031 requires the prioritisation of gold recovery technologies that have been proven in pilot plant studies and commercial manufacturing campaigns over technologies that are still in laboratory scale development. A literature review was conducted of the available technologies for gold recovery from mine tailings to propose a baseline process for gold recovery along with an alternative process

for environmentally friendly gold recovery. The baseline process was developed from literature information concerning commercial gold recovery from mine tailings resources in South Africa with similar mineralogy to the case study region considered in this work.

3.3. Process simulation and modelling

Outotec® HSC chemistry 9.4 (HSC Sim®), a simulation package that allows the computation of mineral processing and thermodynamic properties for minerals, was used to conduct mass and energy balances on identified process flowsheets. HSC Sim®, allows the user to construct static models that allow calculations on chemical systems taking chemical reactions and equilibrium compositions but not reaction kinetics into account. The shortcoming of the software to model reaction kinetics was overcome by sourcing empirical data from literature for metallurgical unit processes (Outotec, 2018).

The HSC Sim® simulation module distinguishes between hydrometallurgical and pyrometallurgical unit processes. The hydrometallurgical module is selected if a defined set of chemical reactions within the process unit is being modelled. The pyrometallurgical module relies on the user understanding the effluent species being formed from literature and HSC Sim® relies on allocating elemental distributions such that the mass and energy balance is satisfied. HSC Sim® also has the functionality of modules for common mineral processing unit operations that have been developed by Outotec (2018). This allows for the input of fundamental chemical engineering parameters if this empirical data is available to determine chemical and physical properties of effluent streams leaving unit process modules. In this study, equilibrium chemical compositions of effluent streams from unit processes was calculated by using the following approaches (in descending order of preference): user-defined empirical data from literature, using the Gibb's equilibrium functionality in HSC Sim® that determines equilibrium compositions through Gibbs' Energy minimisation methodology or reaction stoichiometry (Outotec, 2018; White et al., 1958). The availability of empirical data in literature for the ore resource used in this case study was a limitation and where possible the was sourced or reasonable assumptions made and discussed in chapter 4.

3.4. Life cycle assessment

The environmental impacts of process flowsheets were modelled using Life cycle assessment (LCA) with Thinkstep's GaBi® software. Material and energy balances were conducted with Outotec HSC Chemistry 9® to calculate inventory data not available in GaBi®. HSC Sim® has an LCA evaluation module that allows the elementary inputs and outputs to be exported to GaBi®. This involves associating the elementary flows (inputs and outputs) in HSC Sim® flowsheet with species in GaBi® to represent the respective elementary flows. Once exported to the GaBi® environment, the selections made in HSC Sim® need to be verified to determine if an alternative species exists that is a more accurate representation of the elementary flow.

46

3.4.1. Goal and scope

A "cradle-to-gate" LCA study was conducted for production of a crude gold bullion from mine tailings. In South Africa, "gold-from-ore" manufacturers produce crude gold bullion as a product. The crude bullion is sold to the Rand Refinery for production of a pure gold bullion (Auerswald and Radcliffe, 2005). The system boundaries together with distinction between background and foreground processes for the cyanide and thiosulphate flowsheets is illustrated in Figure 5-1 and Figure 5-2 respectively. A thorough process description for both flowsheets was provided in chapter 4. Mine tailings which make up the feed are a waste stream and thus the "zero-burden assumption" was used in this study. This assumption is commonly made in LCA's on waste streams and states that the waste stream does not carry the environmental consequences of upstream processes along with it into the subsequent system boundary (Ekvall et al., 2007).

The functional unit for this study was defined as 1 kg of gold manufactured at steady state conditions for calculation of environmental indicators. The choice in functional unit was made since the context of this study considers mine tailings as a source of value. Furthermore, the target audience of this study (investors, community members and policy developers) will be able to relate to results expressed in terms of a commodity (i.e. gold) whose price drives economic decisions globally. LCA studies in literature on gold recovery from primary ore resources have also made this functional unit selection. Therefore, the results of this study can then be compared to studies with similar system boundaries (Chen et al., 2018; Elomaa et al., 2020; Norgate and Haque, 2012). A gap in literature for LCA's conducted for gold recovery from mine tailings as a resource has been noted. A mass allocation procedure has been used in this study and refers to the procedure whereby LCA environmental impacts are attributed to product outputs from the system boundary using their mass as basis for calculating the weighted average of life cycle indicators. Gold was the primary product and life cycle impacts were not attributed to by-products produced. The ReCiPe® 2016 life cycle impact assessment (LCIA) methodology with the Hierarchist cultural perspective that was used in this study (discussed in section 2.6.1). The environmental impact categories that were investigated in this study are summarised in section 5.2.1.

3.4.2. Life cycle inventory (LCI) for flowsheets

The life cycle inventory is a database of raw material, energy, product and emissions from the defined system boundary for an LCA study (Rebitzer et al., 2004). The methodology for the compilation of the LCI was summarised in section 2.6 and section 3.3. The condensed LCI for the cyanide and thiosulphate leaching flowsheet is summarised in Table 3-1 and Table 3-2 respectively. The extended LCI for the proposed flowsheet can be found in Appendix K.

Table 3-1: LCI for cyanide leaching process (presented per function unit – 1 kg gold produced)

Input	Quantity	Unit	Output	Quantity	Unit
Activated carbon	2,711.5	kg	Alamine® 336 (Proxy: Hexamethylene diamine (HMDA))	45.9	kg
Alamine® 336 (Proxy: Hexamethylene diamine (HMDA))	45.9	kg	Ammonia	192.5	kg
Ammonia	192.5	kg	Ammonium diuranate	51.1	kg
Calcium carbonate	61,299.5	kg	Arsenic trioxide [Emissions to air]	3.8	kg
Calcium hydroxide	51,562.9	kg	Arsenic trioxide [Product]	378.3	kg
Compressed air	121,563.6	Nm³	Carbon dioxide	69,843.0	kg
Copper sulphate	466.7	kg	Dodecanol	76.4	kg
Dodecanol	76.4	kg	Gold	1.0	kg
Dow® 200	205.0	kg	Nitrate	9,089.7	kg
Electricity	5,736,465.1	MJ	Nitrogen	566,720,775.5	kg
Ferrous sulphate	395.1	kg	Oxygen	41,148.0	kg
Gold	1.3	kg	Kerosene	1,219.2	kg
Hydrochloric acid	3,440.5	kg	Sodium chloride	10,579.4	kg
Iron (II) oxide	3,889.5	kg	Solids emissions (grouped)	12,299,751.0	kg
Kerosene	1,219.2	kg	Sulphur dioxide	4.0	kg
Lead nitrate	23,841.0	kg	Sulphuric acid	30,843.6	kg
Manganese dioxide	578.1	kg	Water (waste water, untreated)	16,607,772.4	kg
Potassium permanganate	335.6	kg	Water recycle	10,618,084.0	kg
Process Steam (from hard coal)	1,119,579.1	MJ	Water vapour	18,847.6	kg
Sodium cyanide	112,355.3	kg			
Sodium hydroxide	2,604.3	kg			
Sodium hypochlorite	13,478.7	kg			
Sodium Mercaptobenzothiazole	1,025.1	kg			
Sulphuric acid	12,658.4	kg			
Tailings Feed	12,092,202.8	kg			
Thermal energy (MJ)	1,051.0	MJ			

Input	Quantity	Unit	Output	Quantity	Unit
Water (desalinated; deionised)	16,626,620.0	kg			
Water recycle	10,618,084.0	kg			
Zinc powder	303.2	kg			

Table 3-2: LCI for thiosulphate leaching process (presented per function unit – 1 kg gold produced)

Input	Quantity	Unit	Output	Quantity	Unit
Alamine® 336/Amberjet® 4200 (Proxy: Hexamethylene diamine (HMDA))	1,394.7	kg	Alamine® 336/Amberjet® 4200 (Proxy: Hexamethylene diamine (HMDA))	1,394.7	kg
Ammonia	192.4	kg	Ammonia	192.4	kg
Calcium carbonate	62,161.8	kg	Ammonium diuranate [Product]	51.1	kg
Compressed air	3,542.2	Nm^3	Carbon dioxide	27,275.1	kg
Copper sulphate	444.5	kg	Dodecanol [Organic emissions to fresh water]	76.4	kg
Dodecanol	76.4	kg	Dow® 200	205.0	kg
Dow® 200	205.0	kg	Gold	1.0	kg
Electricity	1,570,973.9	MJ	Manganese dioxide	48,481.4	kg
Ferrous sulphate	395.0	kg	Nitrogen	37,031.8	kg
Hydrated Lime	120.9	kg	Oxygen	9,258.0	
Hydrogen peroxide	4,996.8	kg	Kerosene	1,218.84	kg
Iron oxide	3,888.3	kg	Silver	14.7	kg
Kerosene	1,218.8	kg	Sodium Mercaptobenzothiazole	1,024.8	kg
Manganese dioxide	577.9	kg	Solid emissions from flowsheet	12,073,599.5	kg
Mine tailings feed	12,092,202.8	kg	Sulphuric acid	30,834.6	kg
Oxygen	37,290.9	kg	Water (desalinated; deionised)	15,535,783.2	kg
Potassium permanganate	222.2	kg	Water recycle	9,932,713.8	kg
Sodium hydroxide	2,644.0	kg			
Sodium Mercaptobenzothiazole	1,024.8	kg			
Sodium sulphide	3,442.8	kg			
Sodium sulphite	816.6	kg			
Sodium thiosulfate	23,237.5	kg			

Input	Quantity	Unit	Output	Quantity	Unit
Sulphuric acid	12,110.6	kg			
Thermal energy	131,441.9	MJ			
Water (desalinated; deionised)	15,535,783.2	kg			
Water recycle	9,932,713.8	kg			

3.4.3. Methodology for determination of category indicators

The LCA methodology involves conducting a life cycle inventory (LCI) in HSC Sim® which is exported to GaBi® for life cycle impact assessment (LCIA) to evaluate environmental impact of elementary inputs and outputs. This approach does not allow the contribution analysis of contributions to impact categories by specific unit processes and chemical species. To overcome this, a supplementary LCIA was conducted simultaneously in Microsoft Excel® for the gold recovery flowsheets identified. This involved exporting ReCiPe® 2016 characterisation factors from the GaBi® database along with the flowsheet LCI to calculate the contribution of individual chemical species to environmental impact categories using equation [3-1] adapted from Pennington et al. (2004) where "*s*" represents a specific chemical species on the LCI. The accuracy of the results of the LCIA conducted in Microsoft Excel® was compared against the LCIA results output by GaBi® as a control measure.

$$Category\ indicator = \sum_{s} Characeterisation\ factor(s) \times Emission\ inventory(s) \qquad \text{[3-1]}$$

This approach allows the traceability to understand the quantitative impact of specific chemical species emitted to the ecosphere on environmental impact categories. The origin of the chemical species can then be traced back to the unit process where the chemical species was produced. This allows the assessment of individual unit processes on environmental impacts. This also allows the identification of opportunities to reduce environmental impacts within a process flowsheet or areas where environmental impacts are elevated (i.e. environmental "hotspots").

3.4.4. Data shortages and data requirements

Datasets for background processes are quintessential for development of an LCA, since the datasets for foreground processes contribute to less than 5% of the life cycle environmental impacts (Steubing et al., 2016). The GaBi® professional database did not include all the datasets required to complete an LCA for the flowsheets developed. The life cycle indicators of missing datasets background processes were accounted for by manually constructing the missing datasets in GaBi® from literature. The manufacturing processes, chemical reactions and assumptions made for the development of each manually created dataset is discussed in this section.

Iron oxide is used in combination with manganese dioxide to form Fe^{3+} ions to leach uranium from ore. Reagent grade iron oxide used in the pigment industry is produced through the neutralisation and calcination of ferrous sulphate at 300 °C as represented by equations [3-2] and [3-3] (Hayashi, 2012). Mass and energy balances were constructed in HSC Sim® which was then exported to GaBi® to create the iron oxide background process dataset. The LCIA results may be underestimated and accuracy of the LCA results affected when predicting the life cycle impacts associated with iron oxide manufacture.

$$2FeSO_4 + 4NaOH \rightarrow 2Na_2SO_4 + 2Fe(OH)_3 + 3H_2 \qquad \text{[3-2]}$$

$$2Fe(OH)_3 \xrightarrow{\Delta} Fe_2O_3 + 3H_2O \qquad\qquad [3\text{-}3]$$
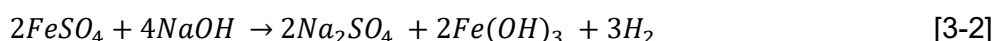
Manganese dioxide (MnO_2) is used as an oxidising agent during uranium leaching. Electrolytic manganese dioxide is prepared by calcining pyrolusite ores at 700-950 °C followed by sulphuric acid leaching (H_2SO_4 concentration = 50 g/L) at 80 °C for 3 hours to produce an electrolyte containing manganese sulphate ($MnSO_4$). The manganese dioxide is then recovered from electrolyte by electrowinning followed by subsequent neutralisation and filtration steps to neutralise residual acid (Biswal et al., 2015). A background dataset for the manganese dioxide was created in GaBi® and HSC Sim® to account for the environmental indicators associated with this species from the process described. The LCIA results may be underestimated and accuracy of the LCA results affected when predicting the life cycle impacts associated with manganese dioxide manufacture.

Potassium permanganate ($KMnO_4$) is an oxidising agent used during effluent water treatment for arsenic removal. Potassium permanganate is synthesised by charging MnO_2 to a solution of concentrated potassium hydroxide (KOH) solution (70-90% by mass) 200-350 °C which is then isolated by filtration (equation [3-4]) (Reidies, 2000). The potassium hydroxide required in equation [3-4] is manufactured by the electrolysis of potassium chloride (equation [3-5]) (Schultz et al., 2000). Background processes for potassium permanganate and potassium hydroxide are not part of the GaBi® professional database and therefore they were developed in GaBi® and HSC Sim® to complete the dataset to model gold metallurgical processes. The LCIA results may be underestimated and accuracy of the LCA results affected when predicting the life cycle impacts associated with potassium permanganate and potassium hydroxide manufacture.

$$MnO_2 + 2KOH + 0.5O_2 \rightarrow K_2MnO_4 + H_2O \qquad\qquad [3\text{-}4]$$
$$KCl + 2H_2O \rightarrow 2KOH + Cl_2 + H_2 \qquad\qquad [3\text{-}5]$$
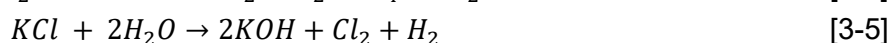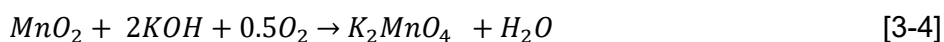
Sodium mercaptobenzothiazole (Na-MBT) was recommended as a flotation collector. The active component 2-mercaptobenzothiazole is produced by reacting aniline with carbon disulphide (CS_2) and sulphur at 3.1 MPa and between 250-300 °C (Equation [3-6]). The resulting 2-mercaptobenzothiazole which is then neutralised with sodium hydroxide to produce sodium mercaptobenzothiazole (Lay et al., 2000). Carbon disulphide background process datasets are not present in GaBi® professional database but have been developed by Kunene (2014) (Table 3-3) as part of their LCA investigation into xanthate salt manufacture. A background dataset was constructed in GaBi® and HSC-Sim® for sodium mercaptobenzothiazole with the aforementioned information. Primary data from literature was used for the development of the carbon disulphide background process dataset therefore the estimation of life cycle impacts for carbon disulphide may be accurate. The dataset created for sodium mercaptobenzothiazole may, however, underestimate life cycle impacts associated with its manufacture and affect the accuracy of the LCIA results.

$$+ CS_2 + S \xrightarrow{250-300\,°C} \qquad\qquad +H_2S \qquad\qquad\qquad [3\text{-}6]$$
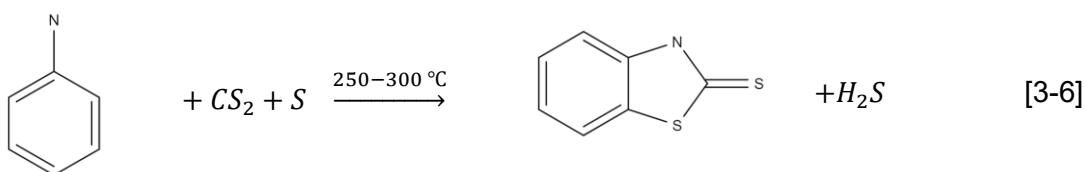
Table 3-3: LCA background dataset for 1 tonne of CS$_2$ manufacture [Adapted from: (Kunene, 2014)

Description			Unit	Quantity per tonne CS$_2$ manufactured
Inputs	Energy	Electricity	GJ	0.81
		Steam, net	GJ	6.54
		Fuel gas	GJ	7.51
	Materials	Water net	m^3	1.38
		Nitrogen	m^3	13.1
		Natural gas	kg	224
		Sulphur	kg	882
Outputs	Waste	Effluent	m^3	0.56
		Solid waste	kg	1.86
		Hazardous waste	kg	2.40
	Emissions	SO$_2$	kg	40.5
		CO$_2$	kg	362
	Product	CS$_2$	tonne	1.00

An LCA for activated carbon manufacture from coconut shells was conducted by Arena et al. (2016) and the life cycle inventory data is summarised in Table 3-4. Activated carbon from coconut shells is most widely used for gold recovery from cyanide liquors and therefore the inventory data in Table 3-4 is appropriate for developing a background dataset for activated carbon (Khosravi et al., 2017; Yalcin and Arol, 2002). Primary data from literature was used to calculate the life cycle impacts of this dataset therefore the LCA results for this dataset may be accurate.

Table 3-4: LCA background dataset for 1 tonne of activated carbon manufacture [Adapted from: (Arena et al., 2016)]

Inputs and outputs		Units	Quantity
Input	**Resources consumption of activated carbon production process**		
	Coconut shells that could have been used as biofuel	kg	0
	Electric energy (Indonesian energy mix)	MJ	2 160
Output	**Air emissions of activated carbon production process**		
	Carbon dioxide, biogenic (kg)	kg	6 460
	Water	kg	4 220
	Oxygen	kg	880
	Nitrogen	kg	19 000
	Carbon monoxide	g	2 440
	Nitrogen oxides, as NO$_2$	g	1 830

	Dust (g)	g	61
	Tar (as Naphthalene)	kg	3.9
	Activated carbon product	tonne	1

In the cyanide flowsheet, lead nitrate was recommended in zinc cementation to improve gold recovery. Lead nitrate manufacture predominantly occurs through the reaction of lead oxide with nitric acid. Unfortunately, lead oxide does not have background datasets in GaBi® professional database. Therefore, an alternative mechanism for lead nitrate manufacture was used to create a dataset which involves the reaction of lead with nitric acid (equation [3-7])(Lewis, 2007). Since the primary manufacturing means for lead nitrate production was not used to create the required dataset in HSC Sim® and GaBi®, the life cycle impact of this process may be underestimated and therefore affect the accuracy of LCIA results reported.

$$Pb + 2HNO_3 \ (aq) \ \rightarrow Pb(NO_3)_2 + 2H_2(g) \tag{3-7}$$
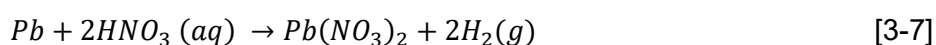
Copper sulphate was recommended as an activator in flotation for both flowsheets and a catalyst in thiosulphate leaching is manufactured as a by-product of copper electrolysis and no background dataset exists in GaBi® professional database for copper sulphate. The correct approach for the creation of background process datasets for by-products is to allocate a fraction of the life cycle impacts of the main product (i.e. copper) to the by-product (i.e. copper sulphate) through mass or economic allocation. Mass or economic allocation of LCA results refers to the procedure whereby life cycle impacts of by-products are calculated based on a weighted average of the mass or economic value of a by-product relative to the main product (Hauschild et al., 2013; Pennington et al., 2004). The research of copper production processes was beyond the scope of this study and therefore a background dataset was constructed for copper sulphate based on equation [3-8]. A background dataset was therefore constructed for copper sulphate in GaBi® and HSC Sim® based on equation [3-8] for copper sulphate formation during electrowinning of copper followed by evaporation to produce a copper sulphate pentahydrate product (Lossin, 2001). The background process dataset created for copper sulphate may overestimate the life cycle impacts of copper sulphate and affect the accuracy of the LCIA results. The overestimation of life cycle impacts is because the GaBi® dataset to produce pure copper was used as an input to create the dataset for copper sulphate. Therefore the life cycle impacts may be inflated since the full life cycle impacts of pure copper was carried forward to the creation of the copper sulphate dataset (equation [3-8]).

$$Cu + \ H_2SO_4 \ + 5H_2O \rightarrow CuSO_4 \cdot 5H_2O + H_2(g) \tag{3-8}$$
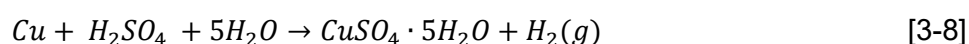
Sodium sulphite (Na_2SO_3) is used in the thiosulphate flowsheet as a reagent in gold elution from ion exchange resin to produce a gold-rich pregnant solution. Sodium sulphite may be manufactured by the sparging of sulphur dioxide through aqueous sodium hydroxide solution at 60-80 °C (equation [3-9]) followed by crystallisation to isolate the salt (Barberá et al., 2000). The background process datasets for sodium sulphite was not present in the GaBi® professional database and therefore

datasets were developed in HSC Sim® and GaBi® using the aforementioned literature information. The accuracy of the LCA results may be affected owing to the underestimation of the life cycle indicators of sodium sulphite.

$$SO_2(g) + 2NaOH \rightarrow Na_2SO_3 + H_2O \qquad [3\text{-}9]$$
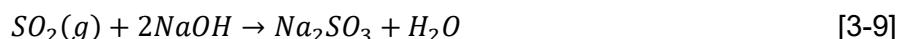
Sodium sulphide (Na_2S) is used for the regeneration of thiosulphate in the thiosulphate flowsheet to convert polythionates to thiosulphate. Sodium sulphide is produced through the reaction of sodium sulphate with carbon at 1 100°C (equation [3-10]) (Lange and Triebel, 2000). 70% of sodium sulphate production is derived from natural sources and 30% as a by-product of the Mannheim process involving the reaction of so at 600-700°C (equation [3-11]) (von Plessen, 2000). The primary data for the development of a background LCA dataset for sodium sulphate could not be identified in literature, therefore the Mannheim process (equation [3-11]) was used instead. Background process datasets for sodium sulphide and sodium sulphate are not part of the GaBi® professional database and therefore datasets were constructed for both species using HSC Sim® and GaBi®. The life cycle indicators for sodium sulphide and sodium sulphate may be underestimated and the accuracy of the LCA results affected.

$$Na_2SO_4(s) + 2\,C(s) \rightarrow Na_2S + 2CO_2 \qquad [3\text{-}10]$$
$$NaCl + H_2SO_4 \rightarrow Na_2SO_4(s) + HCl \qquad [3\text{-}11]$$
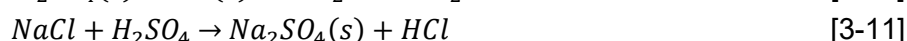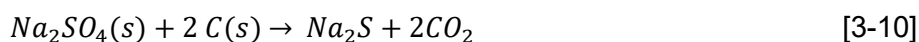
Dow®200 is a polypropylene glycol methyl ether recommended as frother for flotation in this work (Wiese and Harris, 2012). The monomer unit propylene glycol methyl ether is synthesised from propylene oxide and methanol over a calcium oxide catalyst at 120 °C for 5 hours (equation [3-12]) (Zhang et al., 2005). A background dataset for the monomer was created in GaBi® and HSC Sim® for the monomer species as estimation of the environmental indicators of the polymer Dow®200. A background dataset for methanol was not available and therefore ethanol was used instead as a proxy. Several assumptions were used to create the dataset to construct the Dow®200 dataset and the environmental indicators may be underestimated affecting the accuracy of the LCA results reported.

$$CH_3OH + CH_3CHCH_2O \xrightarrow{CaO} CH_3CH(OH)CH_2OCH_3 \qquad [3\text{-}12]$$
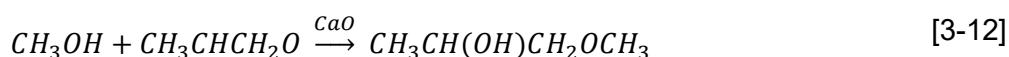
The dodecanol background dataset is required to calculate environmental indicators for uranium solvent extraction and is not included in the GaBi® professional database. Dodecanol belongs to the "fatty alcohol" chemical family and is a derivative produced from coconut oil (Falbe et al., 2000). Background process datasets for raw materials required to manufacture dodecanol are also not included in the GaBi® professional database and therefore a dodecanol background process dataset could not be created in GaBi®. In order to account for environmental impacts of the dodecanol background process, "*C16-18 fatty alcohol from palm oil*" was assumed to be the closest proxy for dodecanol. Dodecanol is a 12-carbon atom fatty alcohol derived from natural sources therefore the

selection of the "*C16-18 fatty alcohol from palm oil*" background process dataset as a proxy was a good representation for dodecanol since it belongs to the same chemical family. This assumption may report inaccurate life cycle indicators associated with dodecanol manufacture and affect the accuracy of the LCA results reported.

Ferric chloride ($FeCl_3$), used during arsenic removal process to capture arsenic as scorodite ($FeAsO_4.2H_2O$) in both flowsheets, background dataset was not present in the GaBi® professional database. Iron (II) sulphate ($FeSO_4$) was used as a proxy to represent the life cycle impacts associated with ferric chloride. This assumption may underestimate the life cycle indicators linked to ferric chloride manufacture and affect the accuracy of the LCA results reported.

Trademarked products were recommended during process flowsheet development. Amberjet® 4200 and Alamine® 336 are proprietary strong base resins used for gold recovery and uranium recovery respectively. The molecular structure of Amberjet® 4200 resin consists of a styrene divinylbenzene copolymer with a trimethyl ammonium functional group ($C_{22}H_{28}NCl$), while Alamine® 336 is an amine with the IUPAC name N,N-dioctyl-1-octanamine ($C_{24}H_{51}N$) (Friess, 2016; Grosse et al., 2003; Nicol and O'Malley, 2002). The manufacturing processes of these species are the intellectual property of the parent companies that hold the intellectual property and therefore the manufacturing processes for the materials would not be part of the GaBi® professional database. It was decided to use "*Hexamethylenediamine*" ($C_6H_{16}N_2$) as a proxy to represent these species. This is the only species in the GaBi® professional database that has a hydrocarbon backbone with amine functional groups and was therefore the only proxy that could be selected. The life cycle indicators associated with Alamine® 336 and of Amberjet® 4200 may be underestimated since the proxy selected has a shorter carbon backbone without alkyl functional groups and will reduce the accuracy of the LCA results reported.

Acrol®J2P 350 is a guar gum, polysaccharide consisting of mannose and galactose moieties used as a depressant during flotation (Voragen et al., 2000). The GaBi® professional database does not include any food and beverage products manufactured from renewable resources and therefore a suitable background process to represent Acrol®J2P could not be identified nor created in GaBi® as a proxy. In the absence of a suitable proxy it was decided to neglect the contribution of this species to the LCA because of the small amount of this species used in the flowsheet. The life cycle impacts associated with the manufacture of Acrol®J2P 350 was therefore not accounted for in the LCA and the environmental impacts of the flowsheets proposed are underestimated where this species is concerned.

Dithiosulphate ($Na_3PS_2O_2$) is a promoter in gold flotation and is produced by the reaction of phosphorous pentasulphide (P_2S_5) with sodium hydroxide (equation [3-13]). Phosphorous pentasulphide is produced by the reaction of molten sulphur and phosphorous at 280-515 °C in the

presence of molten phosphorous pentasulphide followed by distillation to isolate Phosphorous pentasulphide (Bettermann et al., 2000). The raw materials required for the Dithiosulphate does not exist in the GaBi® professional database therefore the construction of a background dataset could not be conducted. A representative species with background datasets that has a chemical structure akin to dithiosulphate could not be identified. In the absence of a suitable proxy it was decided to neglect the contribution of this species to the LCA because of the small amount of this species used in the flowsheet. The life cycle impacts of this species are therefore not represented in this study and the corresponding environmental impacts of the proposed flowsheets.

$$P_2S_5 + 6NaOH \rightarrow Na_3PS_2O_2 + H_2S + 2H_2O \qquad \text{[3-13]}$$
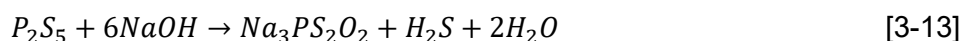
Complete sulphur recovery is not obtained during flotation and has potential to generate sulphuric acid in the depleted tailings effluent (a phenomenon known as acid mine drainage (AMD)). The ReCiPe® 2016 LCIA methodology used in this work does not calculate environmental indicators for sulphur, sulphide or pyrite emissions but indicators for sulphuric acid emissions are estimated (Huijbregts et al., 2016). Therefore, to account for life cycle impacts of acid mine drainage the reaction system in equations [3-14] to [3-18] was created as a unit process in HSC Sim® for both flowsheets (Singer and Stumm, 1970; Tutu et al., 2008). The complete oxidation of pyrite to sulphuric acid was assumed and therefore the life cycle impacts of AMD may be overestimated since only sulphide minerals where oxygen and water concentrations are enough to meet the stoichiometric requirements (i.e. in oxidation zone of mine tailings) will be converted to sulphuric acid (Naicker et al., 2003).

$$FeS_2(s) + 3.5\,O_2(g) + H_2O \rightarrow Fe^{2+}(aq) + 2H^+(aq) + 2SO_4^{2-}(aq) \qquad \text{[3-14]}$$
$$Fe^{2+}(aq) + 0.25\,O_2(g) + H^+(aq) \rightarrow Fe^{3+}(aq) + 0.5H_2O \qquad \text{[3-15]}$$
$$FeS_2(s) + 14Fe^{3+}(aq) + 8H_2O \rightarrow 15Fe^{2+}(aq) + 2SO_4^{2-}(aq) + 16H^+(aq) \qquad \text{[3-16]}$$
$$Fe^{3+}(aq) + 3H_2O \rightarrow Fe(OH)_3(s) + 3H^+(aq) \qquad \text{[3-17]}$$
$$2H^+(aq) + SO_4^{2-}(aq) \rightarrow H_2SO_4(aq) \qquad \text{[3-18]}$$
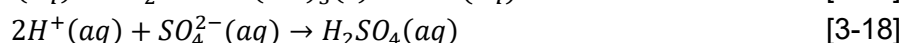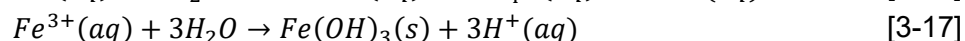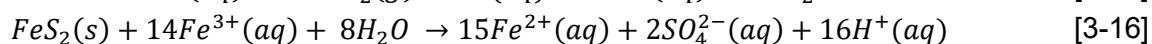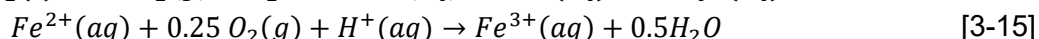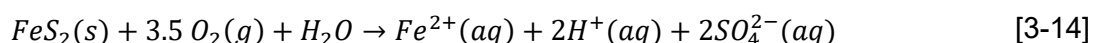
No GaBi® professional database background datasets exist for the effluent water treatment of mining wastewaters. Primary data from reverse osmosis and wastewater treatment plants commissioned at tailings reclamation facilities are required to develop accurate background datasets for wastewater treatment of mining effluent water discharges. For exercises such as these, however, this data is not readily available and is a literature gap for comprehensive metallurgical LCA's to be conducted for mining wastewater treatment. Background datasets for the reverse osmosis water treatment and wastewater treatment of municipal water do, however, exist in the GaBi® professional database and therefore were used as a proxy for mining wastewater treatment. Pre-treatment processes such as cyanide destruction and arsenic removal were included in the proposed flowsheets to bring the effluent water discharge stream from the flowsheets closer to an approximation of the input to municipal wastewater treatment. This may improve the accuracy of the assumption made for selecting the aforementioned effluent water treatment proxies. The environmental impacts

associated within mining wastewater treatment may be underestimated and affect the reliability of the LCA results.

Arsenic trioxide (As_2O_3) is a harmful emission to environment from the pyrometallurgical roasting and was not accounted for in any of the impact categories in the ReCiPe® 2016 LCIA methodology. Other LCIA methodologies such as the CML 2001 LCIA database have derived characterisation factors for As_2O_3. As_2O_3 ecological burden was accounted for by comparing arsenic emissions to ecosphere against environmental regulation specifically NEMA: Air Quality Act (Act No.29 of 2004).

Another point of discussion is that of uranium. Within the ReCiPe® LCIA, the fate models used to derive characterisation factors for radionuclides make the assumption of a 100 000-year time horizon. Therefore, there are no characterisation factors for uranium within the Individualist (time horizon: 20 years) and Hierachist approaches (time horizon: 100 years). The characterisation factors for uranium isotopes are only considered under the Egalitarian perspective (time horizon: 100 000 years) as summarised in the work from Huijbregts et al. (2016) in Table 3-5. Therefore, for certain chemical species an analysis in the Individualist or Egalitarian perspectives needs to be adopted to account for their environmental impacts.

Table 3-5: Uranium radionuclide characterisation factors at midpoint level for the ReCiPe 2016 LCIA methodology [Adapted from: (Huijbregts et al., 2016)]

Radionuclides	Cultural perspectives		
	Individualist	Hierarchist	Egalitarian
Emissions to air			
U-234	-	-	5.82
U-235	-	-	1.27
U-238	-	-	0.491
Emissions to fresh water			
U-234	-	-	0.145
U-235	-	-	0.136
U-238	-	-	0.136

Environmental regulation was consulted to assess if the uranium concentration in depleted tailings emissions was compliant. The South African National environmental management: Waste act 2008 does not report emissions limits for uranium emissions to rehabilitated land (Department of Environmental Affairs, 2014). Therefore, Canadian environmental regulations were consulted instead for uranium emissions to the ecosphere (Canadian Council of Ministers of the Environment, 2007).

In instances where background datasets for raw materials were developed in HSC Sim® along with GaBi® only the material and energy requirements to produce the species were considered. Therefore, the associated material and energy costs to separate the desired species from waste products such as transport, separation, packaging and refining were not considered. This implies

that the background datasets developed are incomplete but represent an estimation of the material and energy requirements to produce the desired species. These assumptions were made because the scope of the present study was to thoroughly develop metallurgical processes for gold manufacture and not to develop background process datasets for the raw materials required for gold manufacture. These assumptions result in an underestimation of the life cycle impacts of the created datasets and reduce the overall life cycle impacts associated with a unit process and the overall flowsheet presented. The contribution of assumptions made to create the background datasets on the LCA results was quantified in a sensitivity analysis. The sensitivity analysis determines the extent to which LCA results are influenced by an input to the LCA study and will be discussed in section 5.2.2.

Some environmental impacts of background datasets developed in this work may not be accounted for in the results. Additionally, the downstream environmental consequences of the developed species were not calculated since extensive toxicological research and modelling are required to develop characterisation factors for species and is beyond the scope of this work. Data limitations due to emissions from the foreground process have minimal impact on the overall LCA results. According to Steubing et al. (2016), the background process datasets account for 99% of the environmental indicators during an LCA investigation and therefore the omission of the foreground process impacts are low. As discussed in 2.6.2, LCA's in the field of mining only account for 3.3% of the academic journal entries in the Compendex database and the data shortages discussed in this section may be one of the reasons for the low representation. LCA is a developing field and researchers within the field improve the databases that software is based on regularly (Huijbregts et al., 2017; Steubing et al., 2016). Despite the data shortages, the information gleaned from datasets that are available in the GaBi® professional database provide meaningful information about the life cycle environmental impacts of products and processes. Furthermore, identification of environmental hotspots in processes is a valuable design tool for minimising the life cycle environmental impacts of a process. A retrospective commentary of the use of LCA as a design tool is discussed in section 5.3.

4 PROCESS FLOWSHEET DEVELOPMENT AND DESCRIPTION

The outcome of objective two of this work is presented in this chapter which collates all the findings from literature in the previous sections to propose gold recovery flowsheets. The scenario of a hypothetical tailings dam mineralogy was considered, and process design decisions were made to develop two process flowsheets for the recovery of gold along with opportunities to reduce the environmental impact of mine tailings. Specifically, a cyanide flowsheet (Figure 4-1) as a baseline for the traditional process for gold recovery and a thiosulphate flowsheet (Figure 4-2) as a prospective environmentally friendly alternative process for gold recovery was proposed. The basis for the flowsheets developed focussed on technology that has been demonstrated for commercial gold recovery and therefore is technologically feasible. This chapter lists assumptions, reagent concentrations and process conditions to provide a thorough process description of each unit process recommended. This information was used to develop process simulations conducted in HSC Sim$^\circledR$ to build the life cycle inventory (LCI) which was used to assess the environmental consequences associated with each process flowsheet through life cycle assessment (LCA). LCA provides an objective platform for the assessment of technologies proposed in literature against the current state of the art employed in industry. In this chapter, only the main reactions occurring in unit processes are reported with the extended reaction systems considered in HSC Sim® being reported in Appendix I.

The flowsheets proposed have unit processes that are in common with one another since the cyanide flowsheet formed the basis for design of the thiosulphate flowsheet. The main differences between the flowsheets are in the oxidative decomposition of sulphide flotation concentrate, the gold leaching lixiviant and accompanying recovery unit processes and finally how the residual lixiviant is managed to prevent lixiviant discharge to the ecosphere. For the cyanide flowsheet (Figure 4-1), sulphide roasting was recommended followed by cyanide leaching and gold recovery by carbon adsorption together with zinc cementation and finally smelting to produce a crude gold bullion. The residual cyanide was destroyed, and arsenic removed from the tailings as scorodite before the solid waste emissions were discharged to the ecosphere and wastewater is subjected to wastewater treatment. For the thiosulphate flowsheet (Figure 4-2), pressure oxidation was proposed for sulphide mineral oxidation followed by precious metal thiosulphate leaching and recovery by ion-exchange resin adsorption and elution to produce a pregnant leach solution. The pregnant leach solution is then electrowon and smelted to produce a crude gold bullion. The residual thiosulphate lixiviant is recovered from the spend leach liquor through reverse osmosis water treatment and thiosulphate lixiviant regenerated to reduce overall lixiviant consumption.

Figure 4-1: Process flow diagram of cyanide leaching flowsheet of the foreground process
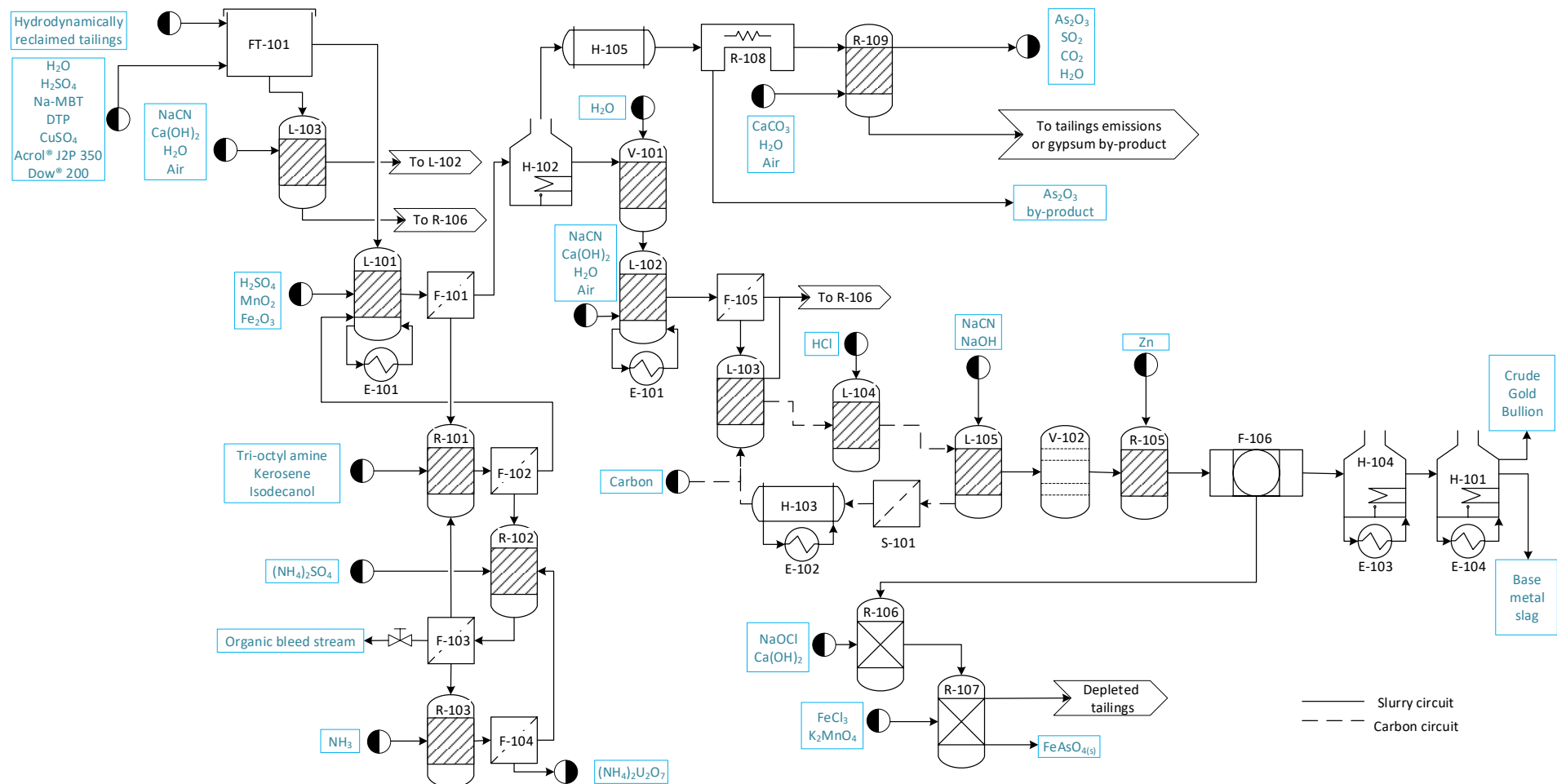
FT-101	Flotation	R-102	Uranium stripping	R-108	SOx scrubber	V-101	Quenching and air cooling	F-102	Aqueous/Organic separator
L-101	H₂SO₄ leaching	R-103	ADU precipitation	R-109	Electrostatic precipitator	V-102	De-aeration	F-103	Aqueous/Organic separator
L-102	Cyanide leaching reactor	R-104	Carbon adsorption	E-101	Live steam heater	H-101	Smelter	F-104	ADU Precipitation
L-103	Carbon in Leach Reactor (Tailings)	R-105	Zinc cementation	E-102	Live steam heater	H-102	Pyrometallurgical sulphide roasting	F-105	Solid/Liquid separator
L-104	Acid washing	R-106	Cyanide destruction	E-103	Coal burner	H-103	Carbon regeneration	F-106	Filter press
L-105	Carbon Elution	R-106	Cyanide destruction	E-103	Coal burner	H-104	Calcination	S-101	Carbon screen
R-101	Uranium solvent extraction	R-107	Arsenic removal	E-104	Coal burner	F-101	Solid/Liquid separator		

61

Figure 4-2: Process flow diagram of thiosulphate leaching flowsheet of the foreground process

FT-101	Flotation	R-101	Uranium solvent extraction	R-108	Reverse Osmosis/Nanofiltration Water Treatment	E-103	Live steam heater	E-104	Coal burner	H-101	Smelter
L-101	H$_2$SO$_4$ leaching	R-102	Uranium stripping	R-109	Thiosulphate regeneration	E-104	Cooling tower	V-101	Thickener	H-102	Pyrometallurgical sulphide roasting
L-102	Resin in Leach Reactor (Concentrate)	R-103	ADU precipitation	R-110	Resin regeneration	E-105	Cooling tower	V-102	Splash Vessel	H-103	Carbon regeneration
L-103	Copper Elution	R-104	Autoclave	R-111	Polythionate synthesis	E-106	Cooling tower	V-103	Tailings thickener	H-104	Calcination
L-104	Thiosulphate elution	R-105	Hot-cure	R-112	Arsenic Removal	E-101	Live steam heater	V-104	Gypsum precipitation	F-101	Solid/Liquid separator
L-105	Gold Elution	R-106	Neutralisation	E-101	Live steam heater	E-102	Live steam heater	V-101	Thickener	F-102	Aqueous/Organic separator
L-106	Resin in Leach Reactor (Tailings)	R-107	Electrowinning	E-102	Live steam heater	E-103	Coal burner	V-102	De-aeration	F-103	Aqueous/Organic separator
										F-104	ADU Precipitation

4.1. Process Feed

This chapter collates all the findings from literature in the previous sections to propose flowsheets for the recovery of value and reduction of environmental impacts of mine tailings. Two process flowsheets were developed (cyanide flowsheet represented in Figure 4-1 and a thiosulphate flowsheet illustrated in Figure 4-2), and this section provides a thorough process description, lists assumptions and provides process descriptions and reagents used in unit processes. This information forms the basis of the process simulations conducted in HSC Sim® to develop the life cycle inventory which will be used to assess the environmental consequences associated with each process flowsheet and each unit operation chosen.

Table 4-1: Process feed assumption for tailings reclamation plant

Property	Value	Reference
Solids feed rate (tons per day)	50 000	(Marsden and House, 2006)
Slurry Density (kg/m^3)	1.45	
Gold head grade (g/tonne)	0.28	(Table 2-2)
Plant availability (%)	95%	Assumption
Operating days per year	320	Assumption
Plant operating hours per day	24	Assumption
Feed composition	Summarised in Table 2-2 and Table 2-3	-

4.2. Flotation

Flotation of reclaimed tailings was recommended to produce a high value concentrate containing gold, sulphide minerals and uranium thereby reducing the size and processing requirements of downstream recovery operations. Oxidation of sulphide minerals in tailings dams causes the passivation of reactive surfaces and therefore conditioning of the pulp within an acidic medium is required before flotation (O'Connor and Dunne, 1991). The environmental impact of flotation is associated with the reagents used and electricity consumed. The West Rand tailings contain 25% pyrophyllite (Table 2-2), which is hydrophobic and can report to the concentrate and thus a depressant was recommended (Cabassi et al., 1983). Together with the copper sulphate activator, dithiophosphate (DTP) functions as a gold promotor to increase gold recovery (Grund et al., 2000; Lloyd, 1981; Ruhmer et al., 1977).

Table 4-2: Flotation process conditions for west rand mine tailings

Reagents [c]	Process conditions [c]	Recoveries
Collectors: 85 g/t NaMBT[a] and 10 g/t DTP [b] Activators: 40 g/t CuSO$_4$ Depressant: 40 g/t Acrol® J2P 350 Frother: 35 g/t Dow® 200 pH modifier: Sulphuric acid	Temperature: 25 °C Pressure: 101.3 kPa pH: 3.8 Pulp density: 40% by mass. Residence time: 19 minutes Mass pull: 3.5%	Sulphur: 85% [c] Gold: 46% [c] Uranium: 20% [d]
[a]: NaMBT: Sodium Mercaptobenzothiazole [b]: DTP: Dithiophosphate [c]: (O'Connor and Dunne, 1991) [d]: (Lloyd, 1981)		

4.3. Uranium leaching and recovery

By removing uranium from mine tailings as a value stream the impact associated with ionising radiation and carcinogenic emissions in depleted tailings is reduced. There is consensus among authors that acid leaching followed by solvent extraction is the preferred mode for recovery of uranium from low-grade ores. The use of manganese dioxide as an oxidising agent for uranium leaching will however increase manganese in the effluent. The tailings feed is already non-compliant with environmental regulation for manganese as summarised in Table 2-1 therefore there is an economic and environmental trade-off when improving uranium recovery efficiency. An alternative is the use of pressure leaching for uranium recovery, but this is a high cost operation reserved for high-grade, complex uranium ores. The South Uranium plant in South Africa recovers uranium from ore deposits containing a uranium grade of 450 g/tonne. In the present study, the mine tailings uranium grade was 54.9 g/tonne (Table 2-1). Therefore the uranium grade in the feedstock in this study is too low to consider high cost technologies such as pressure leaching (Edwards and Oliver, 2000; Sole et al., 2011; Zhu and Cheng, 2011).

Table 4-3: Uranium sulphuric acid leaching reactions for both process flowsheets

Progress (%)	Chemical Reaction	
100	$Fe_2O_3 + 5H_2SO_4(aq) + MnO_2$ $\rightarrow 2Fe^{3+}(aq) + Mn^{2+}(aq) + 5SO_4^{2-}(aq) + 5H_2O$	[4-1]
84.8	$UO_2 +\ 2Fe^{3+}(aq) + 3H_2SO_4(aq)$ $\rightarrow UO_2(SO_4)_3^{4-}(aq) + 2Fe^{2+}(aq) + 6H^+(aq)$	[4-2]
84.8	$UTi_2O_6 + 2Fe^{3+}(aq) + 7H_2SO_4(aq)$ $\rightarrow 2FeSO_4 +\ 2TiOSO_4 + UO_2(SO_4)_3^{4-}(aq) + 10H^+(aq) + H_2O$	[4-3]
100	$CaCO_3(s) + H_2SO_4(aq) \rightarrow CaSO_4(s) + CO_2(g) + H_2O$	[4-4]
0	$FeS_2 +\ 14Fe^{3+}(aq) + 8H_2O \rightarrow 2Fe^{2+}(aq) + 2SO_4^{2-}(aq) + 16H^+(aq)$	[4-5]

Table 4-4: Uranium leaching process conditions

Reagents [a]	Process conditions [a]	Extent of reactions [a]
MnO_2 consumption: 4 kg/tonne dry solids Fe_2O_3 consumption: 5 kg/tonne dry solids H_2SO_4 consumption: 25 kg/tonne dry solids	Temperature: 60 °C Pressure: 101.3 kPa Residence time: 20-24 hours [b] Pulp density: 40% by mass[b] pH: 1.5[b] Redox potential (SHE): greater than 450 mV [b]	UO_2 and UTi_2O_6 leaching extent: 84.8%
[a]: (Lottering et al., 2008) [b]: (Janse van Rensburg, 2016)		

The extents of reaction in Table 4-3 (full reaction scheme in Appendix I) were assumed to be 100% as a conservative estimate and to represent a scenario of maximum reagent consumption. It was assumed that all sulphides and liberated gold report to subsequent oxidative pre-treatment processes and not to the leachate. The decomposition of pyrite was accounted for in subsequent oxidative treatment unit processes and not by acid leaching as defined by equation [4-5]. The presence of brannerite (UTi_2O_6) makes the uranium ore complex and requires elevated temperature, oxidising agent and more severe acidic conditions along with longer residence time (20-24 hours) to achieve up to 84.8% uranium recovery (Gilligan and Nikoloski, 2017, 2015; Lottering et al., 2008).

To recover the uranium, the leachate from acid leaching was subjected to solvent extraction with tri-n-octylamine (trademarked name Alamine 336®) in the organic phase using the reagents and process conditions summarised in Table 4-5 and Table 4-6 respectively to produce ammonium diuranate. Dilution of the leachate to 5% sulphuric acid by mass was implemented to maintain the uranium extraction efficiency of tri-n-octylamine at 98% (Kiegiel et al., 2017). 5% bleed stream of organic phase and stripping medium was recommended to prevent the accumulation of contaminants in the solvent extraction and stripping circuit.

Table 4-5: Uranium solvent extraction reactions for both process flowsheets

Progress (%)	Chemical Reaction	
100	$2C_{24}H_{51}N + H_2SO_4(aq) \rightarrow [(C_{24}H_{51}NH)_2\,SO_4]$	[4-6]
98	$2[(C_{24}H_{51}NH)_2\,SO_4] + UO_2(SO_4)_3^{4-}(aq)$ $\rightarrow \{[(C_{24}H_{51}NH)_4]^{4+}[UO_2(SO_4)_3]^{4-}\} + 2SO_4^{2-}(aq)$	[4-7]

Table 4-6: Uranium solvent extraction process conditions

Reagents [a]	Process conditions [a]	Recoveries [a]
H_2SO_4 mass concentration = 5% Aqueous/organic volume ratio = 1:1 Organic phase composition: 0.2 M tri-n-octylamine in kerosene	Temperature: 25 °C Pressure: 101.3 kPa	Uranium extraction efficiency: 98% [a]
[a]:(Kiegiel et al., 2017)		

Following the simulations in HSC Sim®, it was observed that an 8 000% stoichiometric excess of tri-n-octylamine was proposed in literature for uranium solvent extraction when using the process conditions in Table 4-6. The literature recommendations were based on uranium being recovered as the main product and not a by-product from uranium primary ores. Furthermore, molybdenum and vanadium was recovered along with uranium when using tri-n-octylamine as a solvent and may occupy the solvent functional groups thereby reducing uranium solvent extraction efficiency (Edwards and Oliver, 2000; Sole et al., 2011; Zhu and Cheng, 2011). Therefore, the excess reagent recommendations in literature were followed to ensure the maximum recovery of uranium since only 20% of uranium was recovered during flotation and reduce the environmental impacts related to uranium. Tri-n-octylamine is recycled within the solvent extraction circuit therefore only the 5% bleed stream is replenished in the flowsheet.

Following solvent extraction, the organic phase was stripped with ammonium sulphate and ammonia to recover uranium. The optimum process conditions and stripping reactions are summarised in Table 4-7 and Table 4-8 respectively, along with ammonium diuranate precipitation conditions in Table 4-9. The efficiency of uranium stripping with ammonium sulphate was improved by increasing concentration of stripping agent and maintaining pH below pH 5 which prevents the formation of an emulsion and the premature precipitation of uranium diuranate. Ammonium sulphate was recycled to the stripping stage to reduce reagent consumption.

Table 4-7: Uranium stripping and precipitation reactions for both process flowsheets

Progress (%)	Chemical Reaction	
99.9	$\{[(C_{24}H_{51}NH)_4]^{4+}[UO_2(SO_4)_3]^{4-}\} + 2(NH_4)_2SO_4$ $\rightarrow (NH_4)_4UO_2(SO_4)_3 + 2\{[(C_{24}H_{51}NH)_2]^{2+}SO_4^{2-}\} + 2H_2SO_4$	[4-8]
100	$2(NH_4)_4UO_2(SO_4)_3 + 6NH_4OH \rightarrow (NH_4)_2U_2O_7 + 6(NH_4)_2SO_4 + 3H_2O$	[4-9]

Table 4-8: Uranium stripping process conditions

Reagents [a,b]	Process conditions [a,b]	Extent of reaction
$[(NH_4)_2SO_4]$: 2.0 M NH_3 gas to maintain pH maintained at pH 4.4 - 4.6	pH: 4.4 - 4.6 Temperature: 25 °C Pressure: 101.3 kPa Number of stripping stage: 5	Stripping efficiency: 66%[b]
[a]:(Zhu and Cheng, 2011) [b]: (Morais and Gomiero, 2005)		

Table 4-9: Ammonium diuranate precipitation process conditions

Reagents	Process conditions	Extent of reaction
NH_3 gas to maintain pH maintained at 7.2 – 7.4	pH: 7.2-7.4 Temperature: 35 °C Pressure: 101.3 kPa	100% precipitation ammonium diuranate
[a]:(Ford, 1993)		

4.4. Oxidative pre-treatment of flotation concentrate

Following uranium extraction, the solids filtered off after acid leaching is transferred to oxidative pre-treatment to decompose lixiviant robbers, preg-robbing and other host minerals that gold is occluded within. Oxidative pre-treatment is the point of departure where the proposed process flowsheets for cyanide and thiosulphate leaching differ.

4.4.1. Pyrometallurgical roasting

As discussed in 2.3, the sulphide concentrate consists predominantly of pyrite (FeS_2). For the cyanide flowsheet (Figure 4-1), a single stage, circulating fluidised bed (CFB) roasting was recommended since arsenic concentration in feed was less than 1% by mass. At greater than 1% arsenic a dual stage CFB is recommended in literature. CFB has the benefit of increased sulphur and carbon oxidation, greater specific throughput and more uniform temperature control when compared to the conventional bubbling fluidised bed roasting equipment (Hammerschmidt et al., 2016; Runkel and Sturm, 2009). Following roasting, the solids (referred to as the calcine) are cooled within an air cooler followed by a quenching vessel to reach the process temperature and pulp density required for subsequent gold leaching process. HSC Sim® has a pyrometallurgical module that relies on Gibbs energy minimisation to estimate the composition of the effluent species as discussed in section 3.3. For the inputs listed in Table 4-10, 100% conversion of sulphide, carbon and arsenic to sulphur dioxide, carbon dioxide and arsenic trioxide respectively was reported in the simulations.

Table 4-10: Pyrite roasting oxidation process conditions

Process conditions	Reaction extent
Temperature: 640 °C [b] Excess oxygen: 5%[b] Average solid residence time: 30 minutes [b]	Sulphide oxidation: 100% Carbon oxidation: 100%
[a]: (Runkel and Sturm, 2009) [b]: (Hammerschmidt et al., 2016)	

The off-gases of pyrometallurgical roasting are first cooled to 300 °C in a heat exchanger in the form of a waste heat boiler to generate superheated steam or an evaporative cooler with water depending on the plant's demands. The cooled off-gases are fed to a hot electrostatic precipitator to separate 99% solids, including hazardous arsenic trioxide, from the gas stream before being fed to the SO$_x$ scrubbing units (Hammerschmidt et al., 2016; Thomas and Cole, 2016). Wet limestone flue gas desulphurisation is the technology most widely employed in industry for SO$_2$ extraction. Forced oxidation methods (i.e. direct injection of air into the liquor) can achieve up to 99% desulphurisation compared to natural oxidation methods where 90% desulphurisation is achieved. Forced oxidation also allows for conversion of SO$_x$ to gypsum product at sufficient quality to be used in the gypsum drywall industry thereby contributing to the circular economy (Córdoba, 2015a; Koralegedara et al., 2019a; Srivastava and Jozewicz, 2001a).

Table 4-11: Flue gas desulphurisation reactions for cyanide leaching flowsheet

Progress (%)	Chemical Reaction	
99	$CaCO_3(s) + SO_2(g) + 2H_2O(aq) \rightarrow CaSO_3 \cdot 2H_2O(aq) + CO_2(g)$	[4-10]
90	$2CaSO_3 \cdot 2H_2O(aq) + O_2(g) \rightarrow 2CaSO_4 \cdot 2H_2O(s)$	[4-11]

Table 4-12: Flue gas desulphurisation process conditions

Process conditions [a]	Reaction extent
CaCO$_3$ feed: stoichiometric quantity to achieve 99% SO_2 sequestration O$_2$ feed: air feed rate required to achieve 90% $CaSO_3 \cdot 2H_2O$ to $CaSO_4 \cdot 2H_2O$ conversion pH: 5.5 to 6.5[a]	SO_2 sequestration: 99% [a] $CaSO_3 \cdot 2H_2O$ to $CaSO_4 \cdot 2H_2O$ conversion: 90%[a]
[a]:(Córdoba, 2015b; Koralegedara et al., 2019b; Srivastava and Jozewicz, 2001b)	

4.4.2. Pressure oxidation

The upstream acid leaching step benefited the pressure oxidation process through the removal of carbonates. Carbonates consume oxygen and evolve CO$_2$ reducing oxygen utilisation and oxygen overpressure in the autoclave (Chan et al., 2015). The reactions and process conditions to oxidise sulphides and liberate gold are summarised in Table 4-13 (full reaction scheme in Appendix I) and Table 4-14.

Table 4-13: Pressure oxidation reaction for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
99	$FeS_2 + 3.5O_2(g) + H_2O \rightarrow 2FeSO_4(aq) + H_2SO_4(aq)$	[4-12]
100	$2FeSO_4(aq) + 2H_2SO_4(aq) + O_2(g) \rightarrow 2Fe_2(SO_4)_3(aq) + 2H_2O$	[4-13]
100	$2As + 1.5O_2(g) + 2H_2O + 2FeSO_4(aq) + 2H_2SO_4(aq) + O_2(g)$ $\rightarrow 2FeAsO_4(aq) + H_2SO_4(aq)$	[4-14]
26	$Fe_2(SO_4)_3(aq) + 2H_2O \rightarrow 2Fe(OH)SO_4(aq) + H_2SO_4(aq)$	[4-15]
67	$Fe_2(SO_4)_3(aq) + 3H_2O \rightarrow Fe_2O_3(s) + 3H_2SO_4(aq)$	[4-16]
0	$2Fe_2(SO_4)_3 + 14H_2O \rightarrow 2H_3OFe_3(SO_4)_2(OH)_6 + H_2SO_4$	[4-17]

Table 4-14: Pressure oxidation process conditions

Process conditions [a]	Reaction extent
Temperature: 230 °C[a] Residence time: 2 hours Pulp density: 30% Oxygen partial pressure: 690 kPa	Sulphide oxidation: 99%[a] <u>Iron distribution of discharge</u> [a] $Fe_2O_3(s)$: 67% $Fe(OH)SO_4$: 26% $Fe_2(SO_4)_3$: 7%
[a]:(Fleming, 2010)[b]: (Gertenbach, 2016)	

Following pressure oxidation, goethite, basic iron sulphate, jarosite and sulphuric acid by-products need to be removed to minimise impact on downstream processes. The elevated temperature neutralisation process converts basic iron sulphate and jarosite to hematite (equation [4-19]) instead of goethite ($FeO(OH)$) or iron (III) hydroxide ($Fe(OH)_3(s)$) (equation [4-22] and [4-23]) (which is a precursor to jarosite and goethite formation). Hematite production is favoured at temperatures above 100 °C and goethite is produced at temperatures between 60-100 °C (Fleming, 2010). Process conditions are summarised in Table 4-15, Table 4-16 and Table 4-17. Iron oxyhydroxides such as goethite and jarosite inhibit gold recovery through "preg-borrowing" a phenomenon whereby reversible adsorption of precious metals, especially silver, occurs (Chan et al., 2015). Furthermore, these species increase pulp viscosity of leaching liquors that inhibit mass transfer during gold leaching. The elevated temperature neutralisation process also allows the use of low-cost neutralisation agents such as limestone to negate the effects of basic iron sulphate, ferric sulphate and goethite (Brittan, 2008; Fleming, 2010; Thomas, 1991; Venter et al., 2004; Zhang and Peng, 2015).

Table 4-15: Hot cure process and neutralisation reactions for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
90	$2Fe(OH)SO_4(aq) + H_2SO_4(aq) \rightarrow Fe_2(SO_4)_3(aq) + 2H_2O$	[4-18]
100	$Fe_2(SO_4)_3(aq) + 3CaCO_3 \rightarrow Fe_2O_3(s) + 3CaSO_4(s) + 3CO_2$	[4-19]
100	$H_2SO_4(aq) + 3CaCO_3 \rightarrow CaSO_4(s) + H_2O + 3CO_2(g)$	[4-20]
0	$Fe_2(SO_4)_3(aq) + 3CaCO_3 + 3H_2O \rightarrow Fe(OH)_3(s) + 3CaSO_4(s) + 3CO_2(g)$	[4-21]

0	$Fe_2(SO_4)_3(aq) + 14H_2O + 2H_3OFe_3(SO_4)_2(OH)_6 + 5H_2SO_4$	[4-22]
0	$2H_3OFe_3(SO_4)_2(OH)_6 + H_2SO_4 \rightarrow 2Fe_2(SO_4)_3 + 14H_2O$	[4-23]

Table 4-16: Hot cure process conditions for thiosulphate leaching flowsheet

Reagents [a]	Process conditions [a]	Reaction extent
$CaCO_3$ addition to maintain pH > 2	Temperature: 100 °C Pressure: 101.3 kPa Residence time: 12 hours pH: pH > 2	Conversion of $Fe(OH)SO_4$ to $Fe_2(SO_4)_3$: 90%[a]
[a]:(Chan et al., 2015; Fleming, 2010)		

Table 4-17: Neutralisation process conditions for thiosulphate leaching flowsheet

Reagents	Process conditions	Reaction extent
$CaCO_3$ addition to maintain pH > 7	pH: pH > 7	$Fe_2(SO_4)_3$, H_2SO_4 neutralisation: 100%

Assumptions

1. Complete conversion of $Fe_2(SO_4)_3$ and H_2SO_4 by neutralisation with calcium carbonate by ensuring solution pH is above pH 7 to form hematite (Table 4-17).

2. No jarosite $(H_3OFe_3(SO_4)_2(OH)_6)$ is formed during pressure oxidation and is represented by its precursor basic iron sulphate in the system (equation [4-17]). During the elevated temperature neutralisation process the assumption is made that jarosite is converted to basic iron sulphate (equation [4-23]) and subsequently hematite (equation [4-19]) if conditions in Table 4-16 and Table 4-17 are met (Fleming, 2010).

4.5. Cyanide gold leaching circuit and recovery

Following quenching of the roasted calcine emanating from pyrometallurgical roasting (Figure 4-1), a slurry is formed to create a pulp for gold leaching using the cyanide-in-pulp (CIP) leaching mode. The flotation tailings were subjected to carbon-in-leach (CIL) processes for gold recovery of free-milling while the remaining gold is occluded within quartz, sulphide, carbonate and carbonaceous minerals. The extents of reaction and process conditions are summarised in Table 4-18 (full reaction scheme in Appendix I) and Table 4-19 respectively.

Table 4-18: Cyanide leaching reaction for cyanide leaching flowsheet

Progress (%)	Chemical Reaction	
96.71/58.15	$4Au^{°} + 8NaCN(aq) + O_2(g) + 2H_2O \rightarrow 4NaAu(CN)_2(aq) + 4NaOH(aq)$	[4-24]

Table 4-19: Cyanide leaching process conditions

Reagents	Process conditions	Recoveries
$NaCN$ consumption = 1.1 kg/tonne dry solids [c] $Ca(OH)_2$ consumption to maintain pH 10.5 = 4 kg/tonne [c] CIL activated carbon inventory = 1kg/tonne dry solids[d]	Dissolved oxygen concentration: 8 ppm [a] pH > 10.5 Temperature: 25 °C [b,c,d] Pressure: 101.3 kPa [b,c,d] Pulp density: 50% by mass[d] Residence time: 12-48 hours [d]	Roasted calcine precious metal and base metal recovery: 96.71% [c] Flotation tailings precious metal and base metal recovery: 58.15% [c]
[a]:(Deschênes et al., 2010) [b]:(Anderson, 2016) [c]:(Janse van Rensburg, 2016) [d]: (Stange, 1999)		

Although the particle size of tailings is P_{80} = 106 µm (optimum particle size for cyanidation is P_{80} = 75 µm), the empirical data provided in the diagnostic leaching study by Janse van Rensburg (2016) reflect empirical gold recovery data for the tailings resource being investigated in this study. Gold recovery during cyanidation leaching is 96.7% and 58.2% (Table 4-19) for sulphide roasting calcine and flotation tailings respectively.

Metal cyanide complexes are grouped into strong, moderate and weak categories based on their stability constants. Equilibrium stability constants were used to determine which metal cyanide species were likely to report to the pregnant leach solution. Metal cyanide complexes with stability constants greater than the stability constant of $Zn(CN)_4^{2-}$ were considered during simulations (equilibrium stability constants are summarised in Table 0-15 in Appendix J). These base, precious and heavy metals are more likely to consume reagents during cyanidation- and zinc cementation- processes and report to downstream operations for gold recovery.

4.5.1. Carbon elution and generation and regeneration

After the cyanide leaching of flotation tailings in the cyanide flowsheet (Figure 4-1), acid washing with hydrochloric acid (Table 4-20) was recommended to dissolve gangue minerals that have adsorbed to activated carbon. The adsorbed gangue minerals reduce active sites available for precious metal complex adsorption and certain cations (e.g. Ca^{2+} , $Na^{2+,}$ K^+) deter the elution of gold from carbon since gold-cyanide complexes form bonds with cations that adhere strongly to carbon. Elution with reagents and process conditions in Table 4-21 follows for desorption of metal-cyanide complex from activated carbon. Carbon regeneration to restore gold adsorption capacity of activated carbon inventory takes place in a rotary kiln with process conditions stipulated in Table 4-22. Following the elution and regeneration steps, the carbon is screened at 0.8 mm and replenished with fresh carbon to minimise precious metal discharge to tailings dam. An elution efficiency of 91.4% is assumed based on empirical data reported in literature (Snyders et al., 2015, 2013; Stange, 1999).

Table 4-20: Acid washing process conditions

Reagents [a]	Process conditions [a]
Aqueous HCl mass concentration: 3% by mass.	Temperature: 25 °C Pressure: 101.3 kPa
[a]: (Stange, 1999)	

Table 4-21: Carbon elution process conditions

Reagents [a]	Process conditions [a]	Extent
Deionised water feed with $NaCN$ and NaOH at the following mass concentrations: $NaCN$: 20 g/L $NaOH$: 35g/L	Temperature = 110 °C Pressure = 185 kPa Residence time = 14 hours	Precious metal and base metal elution: 91.4%[b]
[a]: (Davidson and Duncanson, 1977; Snyders et al., 2013; Stange, 1999) [b]: (Muir et al., 1985b)		

Table 4-22: Carbon regeneration process conditions

Reagents	Process conditions
Live steam carbon regeneration	Temperature: 750 °C [a]
[a]: (Stange, 1999)	

4.5.2. Zinc cementation

The pregnant leach solution emanating from cyanidation leaching and carbon elution was passed through a Crowe vacuum, deaeration column to reduce dissolved oxygen that consumes zinc (equation [4-26). Lead nitrate aids in increasing gold recovery in zinc cementation as discussed in section 2.4.8.1. Deaeration was not required for CIL eluted streams because of the oxygen consumption by cyanide in these solutions (Marsden and House, 2006; Walton, 2005). Nevertheless, deaeration was recommended to prevent excessive zinc consumption and passivation caused by zinc hydroxide (equation [4-26]) resulting in the reduction of precious metal recovery. Following precipitation (process conditions and reactions in Table 4-23 and Table 4-24 (full reaction scheme in Appendix I)), a plate and frame filter press separates the high value precipitate from the barren solution (Adams, 2016). Mercury is an occupational health hazard and toxic to the environment. Mercury retorting is necessary at concentrations above 0.5% mercury and would follow zinc cementation to prevent promoting mercury up the gold value chain (Marsden and House, 2006). Fortunately, the zinc precipitated sludge contains less than 0.001% mercury therefore a mercury retort furnace was not recommended in the cyanide flowsheet. In addition to this, the concentration of mercury in the mine tailings feedstock complies with environmental regulation before processing (Table 2-1 in section 2.2) and therefore the sequestration of mercury was not required in both flowsheets.

Table 4-23: Deaeration process conditions

Process conditions [a]
Pressure: 3-10 kPa (absolute) to meet dissolved oxygen concentration
Dissolved oxygen concentration effluent: < 1 ppm
[a]:(Adams, 2016)

Table 4-24: Zinc cementation reaction for cyanide leaching flowsheet

Progress (%)	Chemical Reaction	
100	$Zn° + 2Au(CN)_2^-(aq) \rightarrow 2Au° + Zn(CN)_4^{2-}(aq)$	[4-25]
0	$2Zn° + O_2(g) + 2H_2O \rightarrow 2Zn(OH)_2(s)$	[4-26]

Table 4-25: Zinc cementation process conditions

Reagents [a]	Process conditions [a]
Zn stoichiometric excess: 1.5 times	Dissolved oxygen concentration: < 1 ppm
Free cyanide concentration: 150 ppm	Temperature: 60 °C
Lead nitrate addition: 0.2 kg/kg zinc	Pressure: 101.3 kPa
	Residence time:
[a]: (Mpinga et al., 2014)	

Assumptions

1. The formation of $Zn(OH)_2(s)$ was neglected owing to the removal of oxygen during deaeration (Table 4-23).
2. The zinc stoichiometric excess was enough to completely precipitate all metal cyanide complexes in solution.
3. Zinc precipitate moisture content of 50% by mass after filter press filtration (Adams, 2016).

4.6. Thiosulphate gold leaching and recovery

The pressure oxidised concentrate and the flotation tailings was subjected to resin-in-leach (RIL) thiosulphate leaching to simultaneously leach and recover precious metals from the liquor. Table 4-26 summarises the chemical reactions considered for precious metal recovery (full reaction scheme in Appendix I) and the process conditions are summarised in Table 4-27. Following the discussion in section 2.4.7.3, Amberjet® 4200 strong base ion exchange resin was chosen to achieve 99% gold adsorption. Strong base ion exchange resins have an affinity for thiosulphate complexes in the following order: Au > Pb >> Ag > Cu >> Zn. Therefore, gold displaces other metals on the resin (Grosse et al., 2003; O'Malley, 2002). The formation of polythionates produced during the degradation of thiosulphate can displace gold thiosulphate complexes (equation [4-29]) on the resin therefore the formation of polythionates needs to be minimised. An ion exchange resin concentration of 0.01 g/L in the RIL liquor was recommended in literature (Zhang and Dreisinger, 2003). After HSC Sim® simulations, it was revealed that a stoichiometric resin concentration of 9.66

g/L and 1.73 g/L was required for precious metal recovery from the pressure oxidised concentrate and flotation tailings respectively (Table 4-27). The increase in resin concentration was to compensate for copper, silver, tetrathionate and trithionate adsorption onto resin from the leaching liquor.

Table 4-26: Thiosulphate leaching reactions for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
100	$$CuSO_4 + 3CaS_2O_3(aq) + O_2(g)$$ $$\rightarrow [O_2 \cdot Cu(S_2O_3)_3]^{5-}(aq) + 3Ca^{2+}(aq) + SO_4^{2-}(aq)$$	[4-27]
96.71/58.15	$$4Au + [O_2 \cdot Cu(S_2O_3)_3]^{5-}(aq) + 8CaS_2O_3(aq) + 2H_2O$$ $$\rightarrow 4[Au(S_2O_3)_2]^{3-}(aq) + [Cu(S_2O_3)_3]^{5-}(aq) + 8Ca^{2+}(aq)$$ $$+ 4OH^-(aq)$$	[4-28]
5	$$4CaS_2O_3(aq) + [O_2 \cdot Cu(S_2O_3)_3]^{5-}(aq) + 2H_2O$$ $$\rightarrow 2S_4O_6^{2-}(aq) + [Cu(S_2O_3)_3]^{5-}(aq) + 4Ca^{2+}(aq)$$ $$+ 4OH^-(aq)$$	[4-29]
100	$$2(C_{22}H_{28}N)Cl + CaS_2O_3(aq)$$ $$\rightarrow [(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}] + Ca^{2+}(aq) + 2Cl^-(aq)$$	[4-30]
100	$$2[Au(S_2O_3)_2]^{3-}(aq) + 3[(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}]$$ $$\rightarrow 2[(C_{22}H_{28}N)_3{}^{3+}[Au(S_2O_3)_2]^{3-}] + 3S_2O_3^{2-}(aq)$$	[4-31]
50	$$2[Cu(S_2O_3)_3]^{5-}(aq) + 5[(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}]$$ $$\rightarrow 2[(C_{22}H_{28}N)_5{}^{5+}[Cu(S_2O_3)_3]^{5-}] + 5S_2O_3^{2-}(aq)$$	[4-32]

Table 4-27: Thiosulphate leaching process conditions for thiosulphate leaching flowsheet

Reagents	Process conditions	Recoveries
Reagent concentrations: $[CaS_2O_3]$ = 0.1 M [a,c] $[Cu^{2+}]$ = 2 mM [c] [Amberjet® 4200 Resin] = 9.66 g/L and 1.37 g/L for concentrate leaching and flotation tailings leach respectively $0.1\,M\,NaOH$ solution to maintain pH at pH 10 [c]	Temperature: 50 °C [a,c] Pulp density: 35% by mass[c] Air sparging rate: 0.1 L/min[c] Pressure: 101.3 kPa pH 10 [c] Residence time: 24 hours [c]	Gold recovery = 96.71% and 58.15% for pressure oxidised concentrate and flotation tailings respectively
[a]: (Choi, 2016; Choi et al., 2013) [b]:(Janse van Rensburg, 2016) [c]:(Sitando et al., 2020, 2015)		

The process conditions and reagents summarised in Table 4-27 were based on the results achieved by Lampinen et al. (2015) who reported a gold recovery of 89% for a 32 g/tonne pressure oxidised, concentrate. Zhang et al. (2005) proposed the reaction mechanism for oxygen, copper and thiosulphate leaching in equations [4-28] and [4-29] which were used. The diagnostic leaching empirical data in section 2.3 shows that after pressure oxidation and acid leaching 96.71% gold is

liberated and amenable to leaching. The empirical diagnostic leaching data for gold recovery reported by Janse van Rensburg's (2016) was selected over the results reported by Lampinen et al. (2015). The choice to use Janse van Rensburg's (2016) empirical data best represents the tailings resource investigated in this study and accounts for the influence of gangue minerals on gold recovery by lixiviants..

There is potential for metal-thiosulphate complexes with stability constants greater than copper (equilibrium stability constants summarised in Table 0-16 in Appendix J) to be stable in the leaching liquor and may report to resin recovery circuit to be recovered along with gold. Although the stability constants of platinum- and palladium- thiosulphate complexes are strong, they are not thermodynamically stable and form S-bridged oligomers that precipitate out of aqueous solution (Anthony and Williams, 1993). There is a gap in literature preventing a conclusion on whether platinum and palladium thiosulphate complexes can possibly adsorb onto ion exchange resins before precipitation occurs. Literature resources that demonstrate evidence of gold-, silver- and copper- thiosulphate complex adsorption onto- and elution from- ion exchange resins were sourced (Aylmore and Muir, 2001; Aylmore, 2016; Dong et al., 2017; Grosse et al., 2003; Jeffrey et al., 2010). It was assumed that only gold-, silver- and copper- thiosulphate complexes would be recovered from the thiosulphate leaching liquor.

Only the influence of thiosulphate ($S_2O_3^{2-}$) decomposition products trithionate ($S_3O_6^{2-}$) and tetrathionate ($S_4O_6^{2-}$) were considered in this work. The literature focuses on tri- and tetra-thionate adsorption onto ion exchange resins (Fleming et al., 2003; Jeffrey et al., 2010; Jeffrey and Brunt, 2007). The assumption was therefore made that the impact of higher order polythionates did not impact gold dissolution and resin adsorption enough to warrant mention in the literature. Furthermore, in the subsequent resin regeneration process, tri- and tetra-thionate were converted back to thiosulphate (discussed in section 2.4.7.3). The assumption was thus made that because the precursors to higher order polythionates molecules (tri- and tetra-thionate) were converted back to thiosulphate, the higher order polynomials would either be converted to sulphate (SO_4^{2-}) or back to tri- and tetra-thionate.

4.6.1. Multi-stage recovery of copper and gold from resin

Multi-stage elution of the loaded resin followed for the thiosulphate process and began with copper elution from resin and then gold elution. Copper is undesirable in the crude gold bullion because it increases resource consumption during the subsequent refining operations (Sole and Paul, 1986; Steyn and Sandenbergh, 2004).

Table 4-28: Copper elution reactions for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
96.9	$2\left[(C_{22}H_{28}N)_5{}^{5+}[Cu(S_2O_3)_2]^{5-}\right] + 5S_2O_3^{2-}(aq)$ $\rightarrow 5\left[(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}\right] + 2[Cu(S_2O_3)_3]^{5-}(aq)$	[4-33]
0.3	$2\left[(C_{22}H_{28}N)_3{}^{3+}[Au(S_2O_3)_2]^{3-}\right] + 3S_2O_3^{2-}(aq)$ $\rightarrow 3\left[(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}\right] + 2[Au(S_2O_3)_2]^{3-}(aq)$	[4-34]

Table 4-29: Copper elution process conditions for thiosulphate leaching flowsheet

Reagents [a]	Process conditions [a]	Extent of reaction [a]
$[S_2O_3^{2-}]$= 0.4 M	Temperature: 20 °C Pressure: 101.3 kPa Bed volumes (BV): 5 Eluant flowrate: 5 BV/h (0.04 m^3/h) Eluant pH: natural	$[Cu(S_2O_3)_3]^{5-}$ elution: 96.9% $[Au(S_2O_3)_2]^{3-}$ elution: 0.3%
[a]:(Choi, 2016; Fleming et al., 2003; Jeffrey et al., 2010)		

The recommended gold elution scheme was the in-situ generation of trithionate eluant supplemented with sodium sulphite as discussed in section 2.4.7.3 and demonstrated in Table 4-30 (full reaction scheme in Appendix I) and Table 4-31. The addition of sulphite reduces the number of bed volumes required for complete gold elution while reducing hydrolysis of trithionate to sulphate. Sulphite also converts tetrathionate to trithionate by the mechanism in equation [4-35]. The in-situ generation of the trithionate eluant is explained in section 4.6.3. The assumption was made that the elution mechanism for silver was the same as gold, but this mechanism is not verified and supported by evidence in literature.

$$S_4O_6^{2-}(aq) + 2SO_3^{2-}(aq) \rightarrow 2S_2O_3^{2-}(aq) + S_3O_6^{2-}(aq) \qquad [4-35]$$
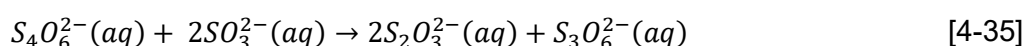
Table 4-30: Gold elution reactions for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
49.55	$2\left[(C_{22}H_{28}N)_3{}^{3+}[Au(S_2O_3)_2]^{3-}\right] + 3S_3O_6^{2-}(aq) + 2SO_3^{2-}(aq)$ $\rightarrow 3\left[(C_{22}H_{28}N)_2{}^{2+}S_3O_6^{2-}\right] + 2[Au(S_2O_3)(SO_3)]^{3-}(aq)$ $+ 2S_2O_3^{2-}(aq)$	[4-36]
49.55	$2\left[(C_{22}H_{28}N)_3{}^{3+}[Au(S_2O_3)_2]^{3-}\right] + 3S_4O_6^{2-}(aq) + 2SO_3^{2-}(aq)$ $\rightarrow 3\left[(C_{22}H_{28}N)_2{}^{2+}S_4O_6^{2-}\right] + 2[Au(S_2O_3)(SO_3)_2]^{3-}(aq)$ $+ 2S_2O_3^{2-}(aq)$	[4-37]
49.55	$2\left[(C_{22}H_{28}N)_3{}^{3+}[Au(S_2O_3)_2]^{3-}\right] + 3S_3O_6^{2-}(aq) + 2SO_3^{2-}(aq)$ $\rightarrow 3\left[(C_{22}H_{28}N)_2{}^{2+}S_3O_6^{2-}\right] + 2[Au(S_2O_3)(SO_3)]^{3-}(aq)$ $+ 2S_2O_3^{2-}(aq)$	[4-38]

Table 4-31: Gold elution process conditions for thiosulphate leaching flowsheet

Reagents [a]	Process conditions [a]	Extent of reaction [a]
Aqueous solution* $[S_2O_3^{2-}]$: 75 g/L $[S_3O_6^{2-}]$: 200 g/L $[S_4O_6^{2-}]$: 3.3 g/L $[SO_4^{2-}]$: 103 g/L Additional reagents: $[SO_3^{2-}]$: 8.0 g/L	Temperature: 20 °C Pressure: 101.3 kPa Bed volumes (BV): 8 Eluant flowrate: 2 BV/h Eluant pH: natural (≈ pH 1.5)	Gold and silver percentage elution: 99.1%
[a]:(Choi, 2016; Fleming et al., 2003; Jeffrey et al., 2010) *: Eluant composition after in-situ generation of trithionate generation		

4.6.2. Ion exchange resin regeneration

Polythionate species, for example trithionate and tetrathionate, have a stronger affinity for ion exchange resins compared to base metal and precious metal complexes and are produced through the natural degradation of thiosulphate. This causes the deterioration of the loading capacity of ion exchange resins with each sequential loading and elution cycle. At Barrick Goldstrike, sodium sulphide solutions have shown success in regeneration of gold loading capacity of resins by up to 400% through the conversion of tri-and tetra-thionate species to thiosulphate (equations [4-39] to [4-41]) with the conditions in Table 4-33 (Choi, 2016; Choi et al., 2013; Fleming et al., 2003; Jeffrey et al., 2010).

Table 4-32: Resin regeneration reactions for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
100	$4\left[(C_{22}H_{28}N)_2{}^{2+}S_4O_6^{2-}\right] + 2S^{2-}(aq) + 6OH^-(aq)$ $\rightarrow 4\left[(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}\right] + 5S_2O_3^{2-}(aq) + 3H_2O$	[4-39]
100	$\left[(C_{22}H_{28}N)_2{}^{2+}S_3O_6^{2-}\right] + S^{2-}(aq) \rightarrow \left[(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}\right] + S_2O_3^{2-}(aq)$	[4-40]
100	$Na_2S \rightarrow 2Na^+(aq) + S^{2-}(aq)$	[4-41]

Table 4-33: Resin regeneration process conditions for thiosulphate leaching flowsheet

Reagents [a]*	Process conditions [a]	Extent of reaction [a]
Aqueous solution: $[Na_2S]$: 120% of the stoichiometric excess required to convert polythionate to $S_2O_3^{2-}$	Temperature: 20 °C Pressure: 101.3 kPa	Polythionate to $S_2O_3^{2-}$ conversion: 100%
[a]:(Fleming et al., 2003)		

4.6.3. In-situ generation of trithionate eluant for recovery

The supply of tri- and tetra-thionate species as sodium salts is costly, therefore the in-situ generation of the eluant solution as demonstrated in Table 4-34 and Table 4-35 was recommended. Maintaining pH between pH 6-8 minimises thiosulphate formation during trithionate production and reducing tetrathionate formation (Fleming et al., 2003). The production of sulphate in the recovery and elution circuit is then managed through gypsum precipitation of the reverse osmosis concentrate stream (Figure 4-2).

Table 4-34: Gold eluant in-situ generation reactions for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
0.52	$2S_2O_3^{2-}(aq) + H_2O_2(aq) + 2H^+(aq) \rightarrow S_4O_6^{2-}(aq) + 2H_2O$	[4-42]
36.93	$3S_2O_3^{2-}(aq) + 4H_2O_2(aq) + 2H^+(aq) \rightarrow 2S_3O_6^{2-}(aq) + 5H_2O$	[4-43]
38.05	$S_2O_3^{2-}(aq) + 4H_2O_2(aq) \rightarrow 2SO_4^{2-}(aq) + 3H_2O + 2H^+(aq)$	[4-44]
0.75	$2S_2O_3^{2-}(aq) + 6OH^-(aq) \rightarrow 4SO_3^{2-}(aq) + 2S^{2-}(aq) + 3H_2O$	[4-45]

Table 4-35: Gold eluant in-situ generation process conditions for thiosulphate leaching flowsheet

Reagents [a]	Process conditions [a]	Extent of reaction [a]
H_2O_2 stoichiometric excess: 100% [NaOH]: 2.51 g/L	Temperature: 20 °C Pressure: 101.3 kPa pH: 6-8	$S_2O_3^{2-}$ to $S_4O_6^{2-}$: 0.52% $S_2O_3^{2-}$ to $S_3O_6^{2-}$: 36.93% $S_2O_3^{2-}$ to SO_4^{2-}: 38.05% $S_2O_3^{2-}$ to SO_3^{2-}: 0.75%
[a]:(Fleming et al., 2003)		

4.6.4. Regeneration of thiosulphate lixiviant

Following the solid-liquid separation of the depleted tailings from the spent leach liquor, the spent thiosulphate liquor was treated by reverse osmosis water treatment to produce a thiosulphate-rich concentrate stream (Figure 4-2). This stream was divided between the thiosulphate regeneration and trithionate generation steps. During the thiosulphate regeneration step, Choi et al. (2015) proposed the use of sodium sulphite and sodium sulphide to regenerate thiosulphate from the recycled concentrate stream while Fleming et al. (2003) proposed thiosulphate regeneration with sodium sulphide and ammonia. There are negative environmental impacts associated with ammonia emissions. Therefore, sodium hydroxide was recommended for pH control at conditions summarised in Table 4-27 and Table 4-37 as opposed to ammonia as applied in the work by Fleming et al. (2003). Based on equation [4-46], the hydroxide ion is the active reagent in tetrathionate conversion to thiosulphate. Calcium oxy-hydroxides such as CaO and Ca(OH)$_2$ increase the likelihood of gypsum precipitation which causes fouling of equiment in the leaching and elution circuit and thus sodium hydroxide was selected over ammonia and calcium oxy-hydroxides for pH control. The sodium hydroxide concentration required was determined by mole balances based on the ammonia concentration recommended in Fleming et al. (2003).

Sulphides in the leaching system can cause precipitation of gold as gold sulphide which will report to tailings stream and not be adsorbed onto ion exchange resin. Therefore regeneration of thiosulphate should occur in a separate unit process (Nicol et al., 2014; Sitando et al., 2018). The process feed of fresh calcium thiosulphate was still required to compensate for the lixiviant losses as thiosulphate decomposes to trithionate ($S_3O_6^{2-}$) and sulphate (SO_4^{2-}).

Table 4-36: Thiosulphate regeneration reactions for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
96.4	$4S_4O_6^{2-}(aq) + 2S^{2-}(aq) + 6OH^-(aq) \rightarrow 9S_2O_3^{2-}(aq) + 3H_2O$	[4-46]
51.2	$S_3O_6^{2-}(aq) + S^{2-}(aq) \rightarrow 2S_2O_3^{2-}(aq)$	[4-47]
100	$Na_2S \rightarrow 2Na^+(aq) + S^{2-}(aq)$	[4-48]
100	$NaOH \rightarrow Na^+(aq) + OH^-(aq)$	[4-49]

Table 4-37: Thiosulphate regeneration process conditions for thiosulphate leaching flowsheet

Reagents [a]	Process conditions [a]	Extent of reaction [a]
$[Na_2S]$: 100% stoichiometric requirement to convert $S_4O_6^{2-}$ and $S_3O_6^{2-}$ to $S_2O_3^{2-}$. $[NaOH]$: 2.51 g/L	Temperature: 25 °C Pressure: 101.3 kPa pH: 8	$S_4O_6^{2-}$ to $S_2O_3^{2-}$ [Equation [4-46]]: 96.4% [a] $S_3O_6^{2-}$ to $S_2O_3^{2-}$: 51.2% [a]
[a]: (Fleming et al., 2003)		

4.6.5. Electrowinning of gold from thiosulphate pregnant solution

Following gold elution, electrowinning onto woven stainless-steel cathodes at low current densities of 5A/m^{2} to recover precious metals from the eluate was recommended. The low current density enables gold bullion detachment following electrowinning (Adams, 2016). An assumption was made that the eluate was recirculated through the electrowinning cycle until complete electrodeposition of precious metals occurred. The sulphite elution system produces a $[Au(S_2O_3)(SO_3)]^{3-}$ complex that deters the formation of gold sulphide precipitate that inhibit electrodeposition of precious metals onto the cathode. The increased stability of the $[Au(S_2O_3)(SO_3)]^{3-}$ complex compared to $[Au(S_2O_3)_2]^{3-}$ was proposed as the main reason for deterring the formation of gold sulphide precipitates in literature (Jeffrey et al., 2010). An assumption was made that neither thiosulphate decomposition (equation [4-52]) nor parasitic reactions (equation[4-53] to [4-55]) at the electrode took place in the electrowinning cell. Although in practice, these reactions consume the operating current and reduce current efficiency and corrode the anode.

Table 4-38: Electrowinning reactions for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
100	$[Au(S_2O_3)(SO_3)_2]^{3-}(aq) + e^- \rightarrow Au^\circ + S_2O_3^{2-}(aq) + 2SO_3^{2-}(aq)$	[4-50]
100	$[Ag(S_2O_3)(SO_3)_2]^{3-}(aq) + e^- \rightarrow Ag^\circ + S_2O_3^{2-}(aq) + 2SO_3^{2-}(aq)$	[4-51]

0	$S_2O_3^{2-}(aq) + 6OH^-(aq) \rightarrow 2SO_3^{2-}(aq) + 3H_2O + 4e^-$	[4-52]
0	$4OH^-(aq) \rightarrow 2H_2O + O_2(g) + 4e^-$	[4-53]
0	$2H_2O + 2e^- \rightarrow H_2(g) + 2OH^-(aq)$	[4-54]
0	$O_2(g) + 2H_2O + 4e^- \rightarrow 4OH^-(aq)$	[4-55]

4.7. Formation of crude gold bullion

4.7.1. Cyanide flowsheet

After zinc cementation, calcination followed which was described in section 2.4.9. The zinc precipitate was calcined in an air circulated furnace maintained at 750 °C and 5% excess oxygen to convert base metals to their respective oxides as determined by Gibbs Energy minimisation in HSC Sim® (Adams, 2016; Marsden and House, 2006). Following calcination, a crude gold bullion was produced by smelting at 1 300 °C with the low-value metal oxides reporting to the slag phase.

4.7.2. Thiosulphate flowsheet

After electrowinning onto stainless steel cathodes in the thiosulphate flowsheet, high pressure water jets are used to detach gold from the cathode forming a gold bullion-sludge. The bullion sludge was smelted at 1 300 °C to produce a crude gold bullion (Adams, 2016).

4.8. Wastewater treatment

4.8.1. Cyanide leaching flowsheet

For the cyanide flowsheet (Figure 4-1), the solid waste emissions from cyanide leaching along with the filtrate separated from precious metal zinc precipitated sludge is routed for effluent treatment. Specifically, cyanide destruction and arsenic removal first need to be completed before wastewater can be discharged to wastewater treatment. The chemical reactions and process conditions for cyanide destruction is summarised in Table 4-39 (full reaction scheme in Appendix I) and Table 4-40 respectively. Alkaline chlorination together with iron precipitation was recommended for cyanide destruction. None of the cyanide destruction technologies in literature can break down strong ferrocyanide complexes (e.g. $Fe(CN)_6^{3-}$) and the solution recommended in literature was to precipitate and capture ferrocyanide complexes as inert Prussian blue ($Fe_4(Fe(CN)_6)_3$) through iron precipitation (equation [4-58]) (as discussed in section 2.4.10). Ferrocyanide destruction presents a gap in literature that requires additional investigation to ensure zero cyanide emissions to environment. It is a requirement of alkaline chlorination that solution pH must exceed pH 10 to completely convert harmful cyanogen chloride ($CNCl$) to cyanogen (CNO^-). Cyanide is broken down to nitrogen and carbon dioxide (equation [4-57]), when the chlorine concentration in solutions exceeds the stoichiometric requirement for complete destruction of cyanide of 2.73 grams Cl_2 per gram of CN^-(referred to as "breakpoint chlorination"). Sodium hypochlorite solution was recommended as the chlorine reagent to mitigate the health impacts associated with using toxic chlorine gas as a reagent (Botz et al., 2016; Kuyucak and Akcil, 2013; Mudder et al., 2001).

Table 4-39: Cyanide destruction reaction for cyanide leaching flowsheet

Progress (%)	Chemical Reaction	
100	$CN^-(aq) + NaClO(aq) \rightarrow CNO^-(aq) + NaCl(aq) + e^-$	[4-56]
100	$2CNO^-(aq) + 3NaClO(aq) + 2H^+(aq) + 2e^-$ $\rightarrow N_2(g) + 2CO_2(g) + 3NaCl(aq) + H_2O$	[4-57]

Table 4-40: Cyanide destruction process conditions for cyanide leaching flowsheet

Reagents	Process conditions	Extent of reaction
$NaClO$ solution concentration: 12.5% by mass [b] $NaClO$ requirement: 5.5 Cl_2 per gram CN^- oxidised [b] $NaOH$ recommended to maintain pH at 10.5	pH = 10.5 [b]	100% for cyanide species destruction except for ferrocyanide.
[a]: (Adams, 1992) [b]: (Botz et al., 2016)		

Arsenic present in the mine tailings feedstock was non-compliant according to environmental regulation (Table 2-1 in section 2.2) and therefore an arsenic sequestration process was recommended. For arsenic removal the chemical reactions and process conditions are summarised in Table 4-41 and Table 4-42 respectively.   Arsenic is first hydrolysed into solution using potassium permanganate followed by precipitation with ferric chloride to produce scorodite (FeAsO₄) which was flocculated to remove it from solution. Scorodite is considered the most stable arsenate compound for long term arsenic disposal of arsenic wastes. The industry practice for arsenic trioxide and arsenic wastes involves the sequestration and underground storage of wastes to prevent surface exposure of arsenic (Adams, 2016; Bowell et al., 1994; Jadhav and Fan, 2001; Swash and Monhemius, 1998; Thomas and Cole, 2016).

Table 4-41: Arsenic removal reactions for cyanide leaching flowsheet

Progress (%)	Chemical Reaction	
95	$As + 3H_2O \rightarrow H_3AsO_3(aq) + 3H^+(aq) + 3e^-$	[4-58]
100	$3H_3AsO_3(aq) + 2KMnO_4$ $\rightarrow 3H_2AsO_4(aq) + 2MnO_2 + H_2O + H^+(aq) + 2K^+(aq)$	[4-59]
100	$FeCl_3 + H_2O \rightarrow Fe(OH)_3(aq) + 3HCl(aq)$	[4-60]
100	$H_2AsO_4(aq) + Fe(OH)_3(aq) \rightarrow FeAsO_4(s) + 2H_2O + OH^-(aq)$	[4-61]
100	$4FeSO_4 + 3Fe(CN)_6^{3-}(aq) \rightarrow Fe_4(Fe(CN)_6)_3(s) + 4SO_4^{2-}(aq)$	[4-62]

Table 4-42: Arsenic removal process conditions for cyanide leaching flowsheet

Reagents	Process conditions	Recoveries
$KMnO_4$ concentration = 1.0 g/L [a]	Temperature: 25 °C	Arsenic removal = 95%

$FeCl_3$ concentration = 405.5 g/L[b] $FeSO_4$ added to satisfy Fe:CN ratio = 0.5 by mass [c]	Pressure: 101.3 kPa pH 5.5 to 6.5 [a]	
[a]: (Sorlini and Gialdini, 2010)	[b]: (Hering et al., 1997)	[c]: (Adams, 1992)

4.8.2. Thiosulphate leaching flowsheet

The thiosulphate leaching flowsheet was based on the process by Barrick gold and relies on microfiltration, ultrafiltration and reverse osmosis water treatment technologies to treat effluent water from thickened tailings. A lixiviant-rich concentrate stream is produced that can be treated with sodium sulphite and sodium sulphide to regenerate the thiosulphate lixiviant. The fouling of microfiltration and ultrafiltration membranes is a bottleneck in the process flowsheet and prevents the process from operating at its design capacity (Choi, 2016). The permeate water may be reused in the process or discharged to surface water systems thereby reducing freshwater consumption of the gold recovery process.

4.9. Heating and agitator energy requirements

Energy balances were conducted in HSC® Sim on each unit process in the flowsheets developed to determine thermal energy and process steam requirements. The heating and cooling requirements to maintain temperature set points were transferred to GaBi® for the LCA.

Tailings reprocessing facilities require large agitation vessels for specific unit processes. With large vessels (≈ 380 m³) and where slurry viscosities are less than 0.01 Pa.s, two side-entering impellers with hydrofoil impellers were recommended in literature for maintaining solid in suspension in applications because they are more economical (Green and Perry, 2007). The slurry density and kinematic viscosity for pulps were estimated from literature data reported in Mangesana et al. (2008) and Marsden (1962) respectively. At turbulent conditions (N_{Re} > 10 000) power number remains constant. For hydrofoil impellers the power number of 0.3 which can be used to determine the agitator electricity consumption. The electricity consumption associated with agitation was calculated using the system of equations [4-63] to [4-68] (Green and Perry, 2007; Hall, 2012). Agitation energy requirements for each flowsheet are summarised in Table 4-3 and Table 4-4 on the basis of processing 50 000 tonnes of ore per day. The calculations for each unit process are tabulated in Appendix E. The flotation and electrowinning electricity requirements were based on those reported in an LCA study on gold recovery from run-of-mine ore which was 3 kWh per ton ore for flotation and 3 100 kWh/tonne gold (Norgate and Haque, 2012).

$$\text{Tank volume: } V = \frac{\pi D_T{}^2 H}{4} \qquad \text{[4-63]}$$

$$\frac{H}{D_T} = 1.125 \therefore H = 1.125 \times D_T \qquad \text{[4-64]}$$

Combine [4-63] and [4-64] and make D_T the subject to derive [4-65]:

$$\text{Tank diameter: } D_T = \sqrt[3]{\frac{V}{0.28125 \times \pi}} \qquad [4\text{-}65]$$

$$\text{Agitator diameter: } D_a = 0.6 \times D_T \qquad [4\text{-}66]$$

$$\text{Reynolds number: } N_{Re} = \frac{\rho.N.D_a{}^5}{\mu} \qquad [4\text{-}67]$$

$$\text{Agitator power: } P = \frac{N_p.\rho.N^3.D_a{}^5}{g_c} \qquad [4\text{-}68]$$

Figure 4-3: Cyanide flowsheet electricity consumption (kWh)

Acid Leaching	Arsenic removal	Carbon Elution	Carbon in Leach (CIL)
13.6	326 820.0	26.7	46 688.6
Cyanide leaching (concentrate)	Cyanide Removal Reactor	De Aeration	Uranium solvent extraction
84.8	326 820.0	62.6	62.6
Uranium stripping	Zinc Cementation	Flotation	
26.7	62.6	16 280.8	
Total electricity for cyanide flowsheet (kW)			716 948.84

Figure 4-4: Thiosulphate flowsheet electricity consumption (kWh)

Acid Leaching	Uranium solvent extraction	Uranium stripping	Arsenic removal
13.6	62.6	62.6	58 360.7
Collection vessel	Copper elution	Flash cooling unit	Gold Elution
58 360.7	198.7	198.7	198.7
Gypsum precipitation	Hot Cure Process	Neutralisation	Pressure oxidation
1 239.7	198.7	198.7	198.7
Resin in Leach (concentrate)	Resin in Leach (tailings)	Resin Regeneration	Splash Vessel
630.7	58 360.7	198.7	198.7
Thiosulphate regeneration	Trithionate Manufacture	Flotation	Electrowinning
1 239.7	198.7	16 280.8	0.93
Total electricity per unit process (kWh)			196 400.44

Cyanide flowsheet electricity requirements for agitation is 3.7 times larger than thiosulphate flowsheet (indicated in Table 4-3 and Table 4-4). Cyanide destruction and arsenic removal collectively accounts for 90% of flowsheet electricity requirements. Before cyanide destruction, the merging of slurry effluent streams from gold leaching and zinc cementation to pass through the effluent treatment unit processes increased the raw material and electricity requirements. Furthermore, a 12.5% aqueous solution of sodium hypochlorite was recommended for cyanide

destruction in literature and the dilute raw material stream further increased the liquid volume and consequently increased the liquid volume and agitation electricity requirements.

5 RESULTS AND DISCUSSION

In this chapter objectives three, four and five will be met by reporting and interpreting HSC Sim® simulation and LCA results for the process flowsheets solutions proposed. The goal of this study was to identify strategies to recover value from gold mine tailings while reducing the environmental impacts of mine waste. The flowsheets developed employ technology that has been demonstrated for the commercial manufacture of gold. Technologies that have not matured beyond laboratory development were not considered for flowsheet development. Two gold recovery flowsheets were identified which include a cyanide flowsheet as the baseline process for gold recovery and a thiosulphate flowsheet as an alternative flowsheet. A comparative LCA assessment was conducted to identify which process solution minimises environmental burdens associated with gold recovery while reducing the environmental impacts of the tailings feedstock. Environmental hotspots in the respective processes were identified along with recommendations to reduce environmental impacts of processes. Retrospective consideration of LCA as a design tool along with a discussion of the significance of this project in terms of its alignment with the United Nations' sustainable development goals (UN SDG) will be discussed after the interpretation of results.

5.1. Process simulation results and discussion

The information in the prior section of this work in the flowsheet development has yielded a cyanide flowsheet Figure 5-1 and a thiosulphate flowsheet in Figure 5-2. The process flowsheets were designed to maximise gold recovery efficiency while identifying solutions to reduce the environmental impact of solid waste emissions from gold recovery processes. The techno-economic feasibility of the process flowsheets was beyond the scope of this study although both flowsheets have been used for commercial recovery of gold. The profitability of a process flowsheets is an imperative criterion for the selection of technology for gold recovery. Therefore, a trade-off between economic and environmental feasibility needs to be made when developing a process for gold recovery.

Figure 5-1: System boundary baseline process (cyanide leaching)
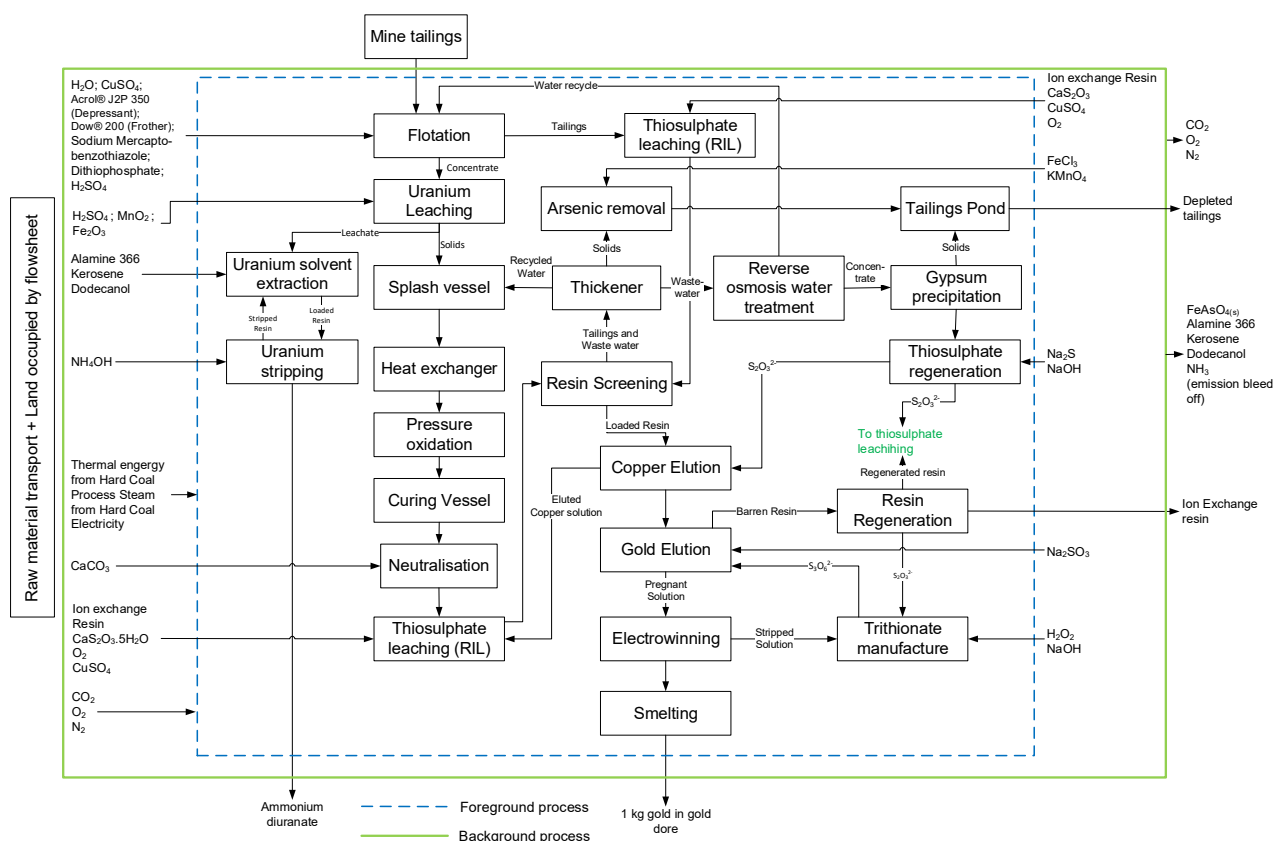
Figure 5-2: System boundary of thiosulphate leaching flowsheet

5.1.1. Solid effluent emissions

The process simulation results and corresponding discussion of the solid effluent emissions will follow in this section. The simulation results of the process flowsheets' depleted tailings emissions and the comparison to environmental regulation for the protection of water resources are listed in Table 5-1.

Table 5-1: Average pollutant concentration (mg/kg) for depleted solids effluent from flowsheets compared to regulatory soil screening values for protection of water resources

Pollutant	Tailings before processing (g/tonne)	Thiosulphate flowsheet tailings (g/tonne)	Cyanide flowsheet tailings (g/tonne)	Maximum concentration (Department of Environmental Affairs, 2014)
Sulphide (%)	0.18	0.03	0.03	-
Cr (mg/kg)	225	85.6	86.7	46 000
As (mg/kg)	78.1	3.95[b]	3.95[b]	5.8
Mn (mg/kg)	1982	3260	799	740
Cu (mg/kg)	42.2	18.9	27.4	16
U (mg/kg)	54.9	13.0	13.6	23[a]
Pb (mg/kg)	24.8	9.40	1250	20
Ni (mg/kg)	96.6	36.6	30.0	91
Zn (mg/kg)	65.5	24.8	93.8	240

Pollutant	Tailings before processing (g/tonne)	Thiosulphate flowsheet tailings (g/tonne)	Cyanide flowsheet tailings (g/tonne)	Maximum concentration (Department of Environmental Affairs, 2014)
Hg (mg/kg)	0.2	0.08	0.03	0.93
Co (mg/kg)	25	9.49	9.39	300
Cd (mg/kg)	0.51	0.19	0.15	7.5
V (mg/kg)	5.89	2.23	2.26	150
-: No data available; Red values = above regulation limit; Green values = compliant. [a]: (Canadian Council of Ministers of the Environment, 2007) [b]: Arsenic in the form of scorodite ($FeAsO_4.2H_2O$) was neglected from calculation since this species is environmental inert and stable.				

Assessment of the depleted tailings solids from both flowsheets (Table 5-1) were not compliant with NEMA environmental regulation (Department of Environmental Affairs, 2014). South African NEMA regulation does not account for uranium emissions in concentration terms and is controlled by the National Nuclear Regulator in terms of activity concentration limits and therefore Canadian regulation was consulted for uranium emissions threshold of 23 mg/kg (Canadian Council of Ministers of the Environment, 2007; Kamunda et al., 2016b). The compliance failure was brought about by manganese, lead and copper emissions in the tailings. There was, however, an improvement in compliance of the depleted solid waste compared to the mine tailings feedstock. This improvement was in the form of sulphide removal (83% reduction in sulphur when compared to tailings feed as represented in Table 5-1), uranium recovery (75% reduction) and arsenic removal. Furthermore, the dilution effect produced by solid precipitates such as gypsum, scorodite, prussian blue $[Fe_4(Fe(CN)_6)_3]$, hematite, goethite, basic iron sulphate and sodium chloride, formed during the processes also aided in reducing the effluent concentration of pollutants. Sulphide, uranium and arsenic concentrations in the depleted tailings were reduced owing to oxidative pre-treatment, uranium recovery and arsenic removal. Therefore, the environmental impact of acid mine drainage, arsenic and radionuclide emissions of mine tailings was reduced.

Additional interventions are required to reduce heavy metals to comply with environmental regulation as illustrated in Table 5-1. Literature concerning rehabilitation of the post-mining landscape focuses on physical, chemical and biological initiatives to either sequester pollutants or inhibit the migration of hazardous pollutants into the ecosphere. Physical strategies involve soil amendments with salvaged topsoil or organic materials (e.g. manure, sewage sludge, bio-char etc.). Chemical methods include soil washing and raising pH by amending tailings with limestone and dolomite to counteract AMD effects. Finally, phytoremediation is a biological intervention where plants and bacteria are used to minimise influence of pollutants on ecosphere. Phytoremediation has the potential to remove pollutants and restore the soil's ability to cultivate plants. Furthermore, certain plant species have

the potential for phytomining. Phytomining involves the bioaccumulation of metals from tailings followed by extractive metallurgical operations on biomass to recover base-, heavy- and precious-metals as products. Phytomining presents a solution for the sustainable development of the mining industry by reducing environmental impacts of mine tailings while generating a revenue stream. The recommendation of phytomining was not evaluated in the flowsheets presented but aligns with the aims and objectives of this study and is an opportunity to extend the scope of this study (Chaney and Baklanov, 2017; Dermont et al., 2008; Festin et al., 2019).

Although the tailings emissions do not meet the environmental regulation for all land uses, there is potential for the tailings to be used as a soil resource in industrial applications. By comparing the solid emissions from the flowsheets in Table 5-1 with the extended environmental legislation specification in Table 5-2, the depleted tailings emissions can be used as soil resource for industrial applications such as aggregate and mining backfill thereby offering a limited contribution to the circular economy as long as the region is zoned for industrial use (Sibanda and Broadhurst, 2018).

Table 5-2: Excerpt from environmental regulation for soil screening values for pollutants in rehabilitated land (Department of Environmental Affairs, 2014)

Pollutants	All land uses [a]	Informal residential	Standard residential	Industrial
Mn	740	740	1 500	12 000
Cu	16	1 100	2 300	19 000
Pb	20	110	230	1 900
Cyanide	14	620	1 200	10 000
[a]; protective of the water resource				

The environmental regulation for soil quality of rehabilitated land (presented in Table 5-2) has a tiered structure arranged in order of increasing stringency: industrial, informal residential, formal residential regions and finally regions for the protection of water resources (Department of Environmental Affairs, 2014). The solid effluent emissions from the flowsheets was compared to the most stringent criteria which is for the protection of water resources. Comparing the solid effluent emissions to the most stringent environmental regulation ensures that environmental burdens are not promoted to other areas of the South African ecosphere if depleted tailings are destined for use within the circular economy as construction materials.

Elements contributing towards compliance failure in solid emissions (listed in Table 5-1) are associated with gold recovery and contaminant (i.e. arsenic) removal. For the cyanide flowsheet, lead emissions increased owing to the use of lead nitrate to improve gold recovery during zinc cementation. There is an economic versus environmental trade-off for the use of lead nitrate for improvement of gold recovery by cyanidation. For the thiosulphate flowsheet, the increase in manganese was attributed to the production of manganese dioxide during arsenic removal unit process using potassium permanganate as an oxidant. The optimum thiosulphate leaching

conditions require a pulp density of 30% as compared with 50% in cyanide leaching. Thus, the subsequent water treatment for arsenic removal in the thiosulphate flowsheet required higher potassium permanganate consumption. The control of emissions in both flowsheets investigated needs to be managed and suggestions will be presented in the discussion that follows.

Cyanide destruction needs to be optimised such that all cyanide species are destroyed and not just captured within an inert prussian blue $[Fe_4(Fe(CN)_6)_3]$. Prussian blue precipitate decomposes above pH 7 to liberate ferrocyanide complexes thereby liberating cyanide to the ecosphere (Adams, 1992). Ferrocyanide complexes are also not destroyed by alternative cyanide destruction technologies such as biological oxidation, hydrogen peroxide, SO_2/air with copper catalyst, ferrous sulphate complexation, ozonation and Caro's acid (Breuer and Hewitt, 2020; Kuyucak and Akcil, 2013; Munive et al., 2020; Oleson et al., 2005; SGS Mineral Services, 2005). Therefore, research into technology for the destruction of iron cyanide complexes is necessary to mitigate cyanide emissions. At present, dilution and containment strategies within lined tailings dams are employed in industry to comply with regulation. This approach is merely a containment strategy for cyanide emissions. Using an alternative lixiviant that does not produce harmful emissions to the ecosphere, such as thiosulphate, is one solution that can prevent the emission of harmful lixiviants to the ecosphere (provided the process flowsheet recommended is profitable). The thiosulphate flowsheet had the disadvantages of consisting of several costly unit processes and may influence its economic feasibility, such as: pressure oxidation, resin adsorption and elution, and reverse osmosis water treatment. In addition to this, the chemical reaction mechanisms for thiosulphate leaching are complex require robust process control to maintain optimum process conditions for gold recovery.

5.1.2. Gaseous emissions

The thiosulphate flowsheet did not emit any hazardous gaseous emissions from the foreground process that need to be controlled according to the NEMA: Air Quality Act (Act No.29 of 2004). For the cyanide flowsheet, the sulphide roasting gaseous emissions from the foreground process was not compliant with the NEMA: Air Quality Act as demonstrated in Table 5-3. Therefore, electrostatic precipitation for the removal of arsenic trioxide followed by flue gas desulphurisation for SO_2 gas sequestration was incorporated into the cyanide flowsheet design. Although the SO_2 and particulate matter emissions meet environmental regulation in the cyanide flowsheet, the SO_2 and arsenic trioxide emissions still pollute the ecosphere. In the thiosulphate flowsheet, arsenic was hydrolysed and converted to inert scorodite ($FeAsO_4.2H_2O$) during pressure oxidation as discussed in 4.4.2.

Table 5-3: Comparison of sulphur dioxide emissions from cyanide flowsheet with environmental legislation for air emissions (Department of Environmental Affairs, 2010)

Substance (Units)	Before electrostatic precipitation and SO$_2$ scrubbing	After electrostatic precipitation and SO$_2$ scrubbing	Regulation upper limit [b]
Particulate matter (mg/Nm^3 [a]) as arsenic emissions	686.5	6.86	20
Sulphur dioxide (mg/Nm^3 [a])	105 500	1 055	1200
[a]: under normal conditions of 273 K and 101.3 kPa			
[b]: (Department of Environmental Affairs, 2010)			
[c]: Red values = above regulation limit; Green values = compliant			

Based on these results, pressure oxidation is the superior technology for sulphide mineral oxidation to liberate gold and has the benefit of no hazardous emissions to the atmosphere in addition to converting sulphide minerals to gypsum and thereby mitigating the impact of acid mine drainage. The economic feasibility of implementing pressure oxidation technology for low gold grade, ore resources needs to be assessed to ensure a profitable operation can be run with this technology. Carbon dioxide emissions from both flowsheets and the implications of greenhouse gas emissions will be discussed in section 5.2.1.6.

5.1.3. Flowsheet evaluation

The overall gold recovery efficiency was 71.2% for both process flowsheets. The identical gold recovery from both flowsheets was attributed to the fact that diagnostic leaching data was used to estimate gold recovery based on unit processes employed to break down minerals that occlude gold. The gold that could not be recovered by the proposed flowsheets was captured in refractory minerals (i.e. gold occluded within quartz, sulphide minerals and carbonaceous matter) that reported to the flotation tailings stream and discharged as solid waste effluent.

In the proposed flowsheets, the flotation tailings reported a low gold grade of 0.12 g/tonne of which 58% was free-milling gold. Research into flotation technology to improve sulphide and gold recovery above 85% and 46% respectively is beneficial to improve gold recovery and further reduce the impact of acid mine drainage. The flotation recovery maximisation could potentially eliminate the need for leaching of flotation tailings which consumes the bulk of the lixiviant used in both flowsheets. As an alternative solution, heap leaching of flotation tailings instead of agitated leaching for leaching of free-milling gold may reduce electricity costs brought about by agitation during leaching flotation tailings. Heap leaching has been demonstrated at commercial scale for thiosulphate and cyanide leaching (Brierley, 1997; Dunne et al., 2009; Roberto, 2017; Wan and Brierley, 1997). The disadvantage of heap leaching is the residual lixiviant remaining in depleted tailings that are toxic to the environment Therefore, processes to breakdown harmful emissions need to accompany heap

leaching solutions. The aforementioned solutions can reduce electricity and lixiviant consumption and consequently the life cycle impacts of both flowsheets.

Cost-effective technologies for the liberation of fine-grained gold in quartz and silicate minerals smaller than 75 µm need to be identified and presents a gap in literature. In the present study, 11.8% gold was occluded in quartz and at present ultrafine grinding is the only technology available to liberate gold in quartz. Unfortunately, the energy cost of comminution required to reduce the particle size increases exponentially towards the particle size required for complete liberation of gold and may render gold recovery unprofitable (Orumwense and Forssberg, 1992). According to literature, ultra-fine grinding is reserved for the treatment of gold concentrates (Coetzee et al., 2010; Corrans and Angove, 1991; Senchenko et al., 2016).

In thiosulphate leaching systems, palladium and platinum form more stable metal-thiosulphate complexes than gold. Therefore, there is potential for co-adsorption and elution of palladium and platinum metal thiosulphate complexes along with gold. As far as the current study has determined there is no evidence of platinum and palladium recovery from thiosulphate liquor with ion exchange resins to support this hypothesis, but this may be a basis for a subsequent study. Undesirable metals such as titanium and mercury also form more stable metal-thiosulphate complexes than gold. Research is required to minimise the quantity of these metals reporting to the gold eluate before electrowinning to increase purity of the crude gold bullion produced from the thiosulphate flowsheet.

This section concludes objective two of this work which was to identify process flowsheets for value recovery from hypothetical mine tailings of the West Rand region. The next steps involve the collation of a life cycle inventory followed by life cycle assessment of the two process options that have been proposed and to assess the environmental impacts of both flowsheets.

5.2. Life cycle assessment (LCA)

5.2.1. Life cycle impact assessment (LCIA)

Objectives three and four will be addressed in this section, which involves quantifying the environmental impacts of the proposed flowsheets through LCA to identify a process that yields reduced environmental impacts for gold recovery from mine tailings. The LCA results were interpreted by means of comparative analysis followed by a sensitivity analysis for each flowsheet as described in the work by Segura-Salazar et al. (2019). The comparative analysis between the cyanide and thiosulphate flowsheets involved two levels. Firstly, a contribution analysis of LCA results using midpoint and endpoint impact categories was done to identify environmental hotspots in background and foreground processes along with emissions to the ecosphere. The background process evaluation assesses the cradle-to-gate upstream environmental impacts associated with the manufacture of raw material and energy resources utilised in the proposed flowsheets. The foreground process analysis groups environmental impacts according to the unit processes in a flowsheet to identify where environmental hotspots are concentrated in the flowsheet and associated with specific technology selections as a basis for process design decisions. The environmental assessment of direct emissions from foreground process to ecosphere quantifies and ranks the ecological impact of chemical species in process waste streams. The second level of the comparative analysis involved a comparison of the baseline gold recovery technology of cyanide leaching with an alternative lixiviant system, namely thiosulphate. The purpose of the second level of comparative analysis will determine which flowsheet results in reduced environmental consequences overall. In both levels of comparative analysis, an account of the environmental impacts of the current state-of-the-art for gold recovery from tailings was provided along with opportunities to further reduce environmental impacts will be identified in this section.

5.2.1.1. Hotspot analysis for background processes

The LCIA results for both flowsheet options are summarised in Table 5-4. The thiosulphate flowsheet generates a reduced environmental burden for 18 out of 19 environmental impact categories except for freshwater consumption when compared to the cyanide flowsheet and will be discussed further in the ensuing sections. The background processes reflect the life cycle impacts associated with raw material and energy requirements of the flowsheets proposed. The contribution of background processes for the cyanide and thiosulphate processes are summarised in Figure 5-3 and Figure 5-4 respectively.

Table 5-4: LCIA midpoint impact category results (results expressed per 1 kg gold (functional unit)

Impact category	Abbreviation	Thiosulphate leaching	Cyanide leaching	Units
Climate change, default, excl biogenic carbon	CC (non-biogenic)	763,917.92	2,684,671.06	kg CO_2 eq.
Climate change, incl biogenic carbon	CC (Biogenic)	764,666.03	2,702,139.00	kg CO_2 eq.
Fine Particulate Matter Formation	FPM	1,658.76	5,522.95	kg $PM_{2.5}$ eq.
Fossil depletion	FD	2,267,884.61	3,804,331.79	kg oil eq.
Freshwater Consumption	FW, consump.	15,196.25	9,631.59	m^3
Freshwater ecotoxicity	FW, ecotoxi.	56,693.34	104,714.36	kg 1,4-DB eq.
Freshwater Eutrophication	FW, eutroph.	47.34	680.62	kg P eq.
Human toxicity, cancer	Htox, cancer	346.94	339,554.10	kg 1,4-DB eq.
Human toxicity, non-cancer	Htox,non-cancer	318,132.47	15,303,601.14	kg 1,4-DB eq.
Ionizing Radiation	IR	4,594.28	10,822.17	Bq C-60 eq. to air
Land use	Land use	5,216.27	14,737.50	Annual crop eq.·y
Marine ecotoxicity	Marine, Ecotox.	21,633.21	83,269.12	kg 1,4-DB eq.
Marine Eutrophication	Marine, Eutroph	54.71	9,003.09	kg N eq.
Metal depletion	Metal Deplete.	38,903.78	82,014.34	kg Cu eq.
Photochemical Ozone Formation, Ecosystems	Photo Ozone, eco	2,497.70	8,610.59	kg NO_x eq.
Photochemical Ozone Formation, Human Health	Photo Ozone, human	2,492.72	8,596.52	kg NO_x eq.
Stratospheric Ozone Depletion	Strato Ozone Deplet.	0.18	0.77	kg CFC-11 eq.
Terrestrial Acidification	Ter. Acidify	5,391.55	17,958.77	kg SO_2 eq.
Terrestrial ecotoxicity	Ter. Ecotox.	194,003.52	975,687.40	kg 1,4-DB eq.
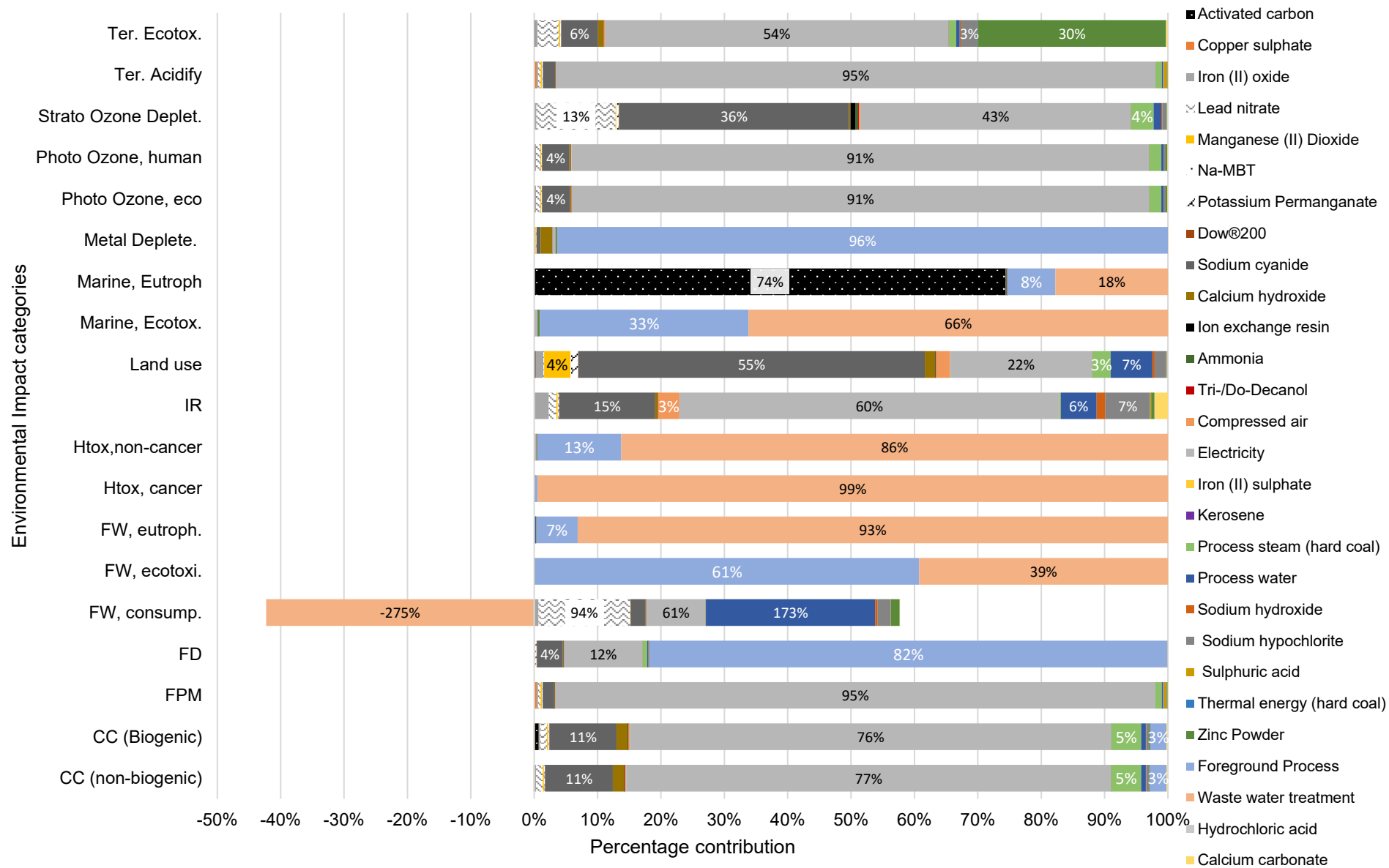
Figure 5-3: Contribution of background processes to each impact category for cyanide leaching flowsheet
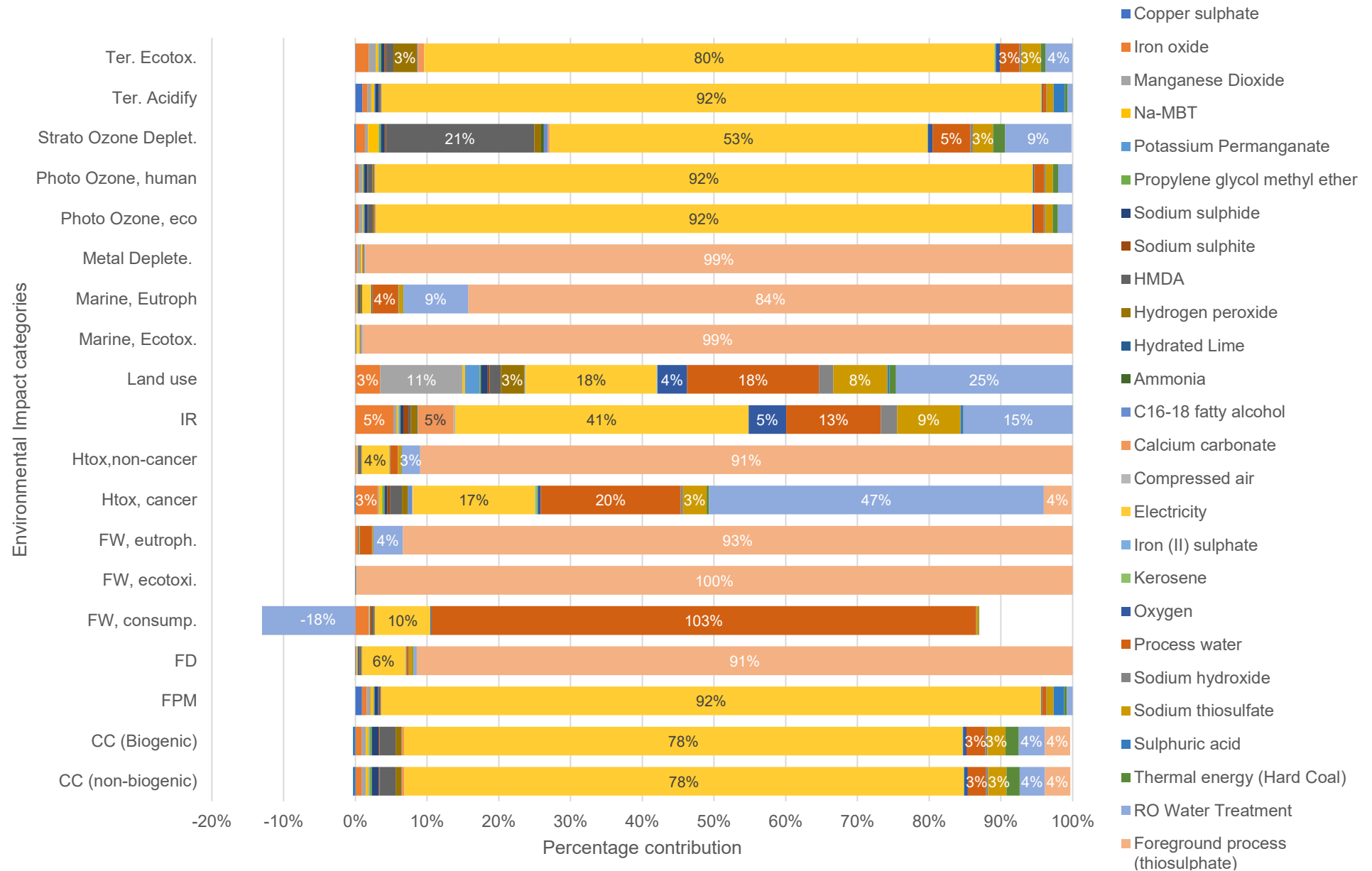
Figure 5-4: Contribution of background processes to each impact category for thiosulphate leaching flowsheet

Electricity was the majority contributor for 9 out of 19 impact categories for the cyanide flowsheet and 9 out of 19 impact categories for the thiosulphate flowsheet (Figure 5-3 and Figure 5-4) and was linked to agitation requirements for process vessels (as discussed in section 4.9). Electricity consumption for cyanide flowsheet was higher than the thiosulphate flowsheet because of agitation requirements required for cyanide destruction and arsenic removal (section 4.9). Chlorine gas may be used as an alternative to 12.5% aqueous sodium hypochlorite to reduce the amount of water introduced, provided safety measures are employed to mitigate occupational health and environmental impacts. The biological treatment, hydrogen peroxide or SO_2/Air (i.e. INCO process) cyanide destruction technologies also need to be evaluated to ascertain if these technologies will reduce electricity consumption and life cycle environmental impacts. In summary, the requirement for lixiviant destruction in the cyanide flowsheet increases the electricity requirements causing a corresponding increase in life cycle environmental impacts. By recycling and regenerating lixiviant in the thiosulphate flowsheet, the requirement for lixiviant destruction was avoided.

The environmental implications of electricity production in both flowsheets may be reduced by incorporating more renewable energy resources into the electricity grid mix (Pehnt, 2006). In South Africa 90% of electricity is generated from coal. Considering the rising electricity costs in South Africa, incorporating more renewable energy resources may reduce the operating costs linked with electricity consumption and increase the profitability of processes (IEA, 2017; Votteler and Brent, 2017). The South African government has encouraged partnerships with the Department of Minerals and Energy to increase the renewable energy adoption thereby reducing the strain on the national electricity grid. Considering the large land occupied by mines there is potential for a hybrid electricity grid supply by introducing more solar photovoltaic, onshore wind and geothermal energy plants (Department of Minerals and Energy, 2002) The realisation of a return on investment for implementing renewable energy will however be dependent on the service life of the mine. Therefore, until the initial investment cost is recovered, the commissioning of renewable energy may increase the unit cost of electricity and a cost-benefit analysis needs to be conducted on hybrid electricity supplies for mine operations.

Wastewater treatment greatly affected 5 out 19 impact categories for cyanide flowsheet (Figure 5-3) and 3 out of 19 impact categories for the thiosulphate flowsheet (Figure 5-4). According to GaBi® professional database, in the cyanide leaching flowsheet the wastewater treatment dataset produces a sludge that consists of nitrogen, phosphorous and potassium (represented as N, P_2O_5, K_2O). The activated sludge may be used as a mineral fertiliser in agriculture to reduce the environmental impacts on human toxicity, freshwater and marine eutrophication or incinerated as a thermal energy source for the water treatment plant. The incinerated ash is a base metal ore resource of copper, iron, zinc, lead and aluminium (Karri et al., 2018; Sphera Solutions GmbH, 2020).

In the thiosulphate leaching circuit, reverse osmosis (RO) water treatment produced a concentrate that was recycled in the process. In the reverse osmosis dataset modelled in GaBi®, the concentrate stream is discharged to municipal wastewater treatment which in turn produced an incinerated ash (containing base and heavy metals) and activated sludge that may be used as a mineral fertiliser as discussed above. Therefore, the environmental impacts related to RO water treatment are overestimated for the thiosulphate flowsheet since the concentrate stream was recycled to regenerate thiosulphate and return to the gold leaching process steps.

Water consumption in the water scarce region of South Africa has led to the government introducing a water resource management charge and establish benchmarks for water reuse and recycle on mining industries to reduce freshwater consumption. Therefore technology solutions that reduce freshwater consumption will be favoured (Gunson et al., 2012; Letsoalo et al., 2007; Mudd, 2007; Naicker et al., 2003). The cyanide flowsheet consumed half the amount of freshwater when compared to the thiosulphate flowsheet (Table 5-4). This may be attributed to the 30% pulp density required for thiosulphate leaching compared to 50% pulp density in the cyanide flowsheet required for optimum gold leaching. Furthermore, wastewater treatment results in a negative contribution to the freshwater consumption impact category in Figure 5-3 and Figure 5-4 because of treated water produced as a product of wastewater treatment processes. The GaBi® datasets for wastewater treatment was not developed for mining industry and therefore the reverse osmosis (RO) water treatment dataset does not result in a large negative contribution when compared to the wastewater treatment dataset (275% versus 18% versus reduction in freshwater consumption Figure 5-3 and Figure 5-4 respectively). There is potential that more treated water can be produced through RO water treatment although this needs to be proven during the development of GaBi® datasets for wastewater treatment of mining wastewater specifically. In industry, additional approached to reduce mine freshwater consumption such as: ore pre-concentration, evaporation reduction by cover tanks, thickeners and tailings ponds along with filtered tailings disposal can aid in reducing water consumption by 74% (Gunson et al., 2012). These strategies were not considered in this study but are worthwhile mentioning as freshwater consumption reduction strategies.

The land occupied by flowsheets was not considered in this LCA. Therefore, land use reflected in LCA results is linked to land occupied to supply raw material and energy resources to the proposed flowsheets. The resources required for the thiosulphate flowsheet occupied 2.8 times less land when compared to the cyanide flowsheet (Table 5-4). This was linked to the life cycle impacts of sodium cyanide manufacture (as illustrated in Figure 5-3) and the large electricity consumption required for cyanide destruction and arsenic removal as discussed in this section.

5.2.1.2.   *Hotspot analysis on unit processes*

An analysis of unit processes in the flowsheets can aid in identifying opportunities to reduce environmental impacts. The contribution per unit process within the cyanide and thiosulphate

flowsheets are summarised in Figure 5-5 and Figure 5-6 respectively. In the cyanide flowsheet (Figure 5-5), the leaching of flotation tailings, the depleted tailings emissions to the ecosphere along with cyanide destruction and arsenic removal accounted for the bulk of the environmental impacts (in descending order of environmental impact). For the thiosulphate flowsheet (Figure 5-6), arsenic removal, flotation tailings gold leaching, depleted tailings emissions to ecosphere, flotation, acid leaching and uranium recovery contribute to most of the environmental impacts of this flowsheet (in descending order of environmental impact).

The leaching of flotation tailings increased gold recovery from 44.5% to 71.2%, therefore leaching of flotation tailings increased the revenue generated from the process although an economic evaluation needs to determine if this improvement is profitable. In both flowsheets, leaching of flotation tailings for precious metal recovery increased the overall life cycle impacts. The large mass flowrate of the flotation tailings stream compared to the flotation concentrate stream means that the lixiviant consumption and agitation electricity requirements increased to maintain optimum lixiviant conditions for gold leaching. Consequentially, the increased lixiviant consumption increases overall life cycle impact of background processes associated with lixiviant production.

Life cycle impacts can be reduced by minimising lixiviant consumption through process optimisation and lixiviant regeneration and recycling. Alternatively, lixiviants can be selected that have reduced life cycle environmental impacts. Based on the LCIA results demonstrated in Table 5-4, the thiosulphate flowsheet proposed was a viable replacement for cyanide leaching. The environmental benefits held by the thiosulphate flowsheet is brought about by the reduced life cycle impacts inherent with thiosulphate leaching technology and the reduction in lixiviant consumption by recycling and regenerating lixiviant.
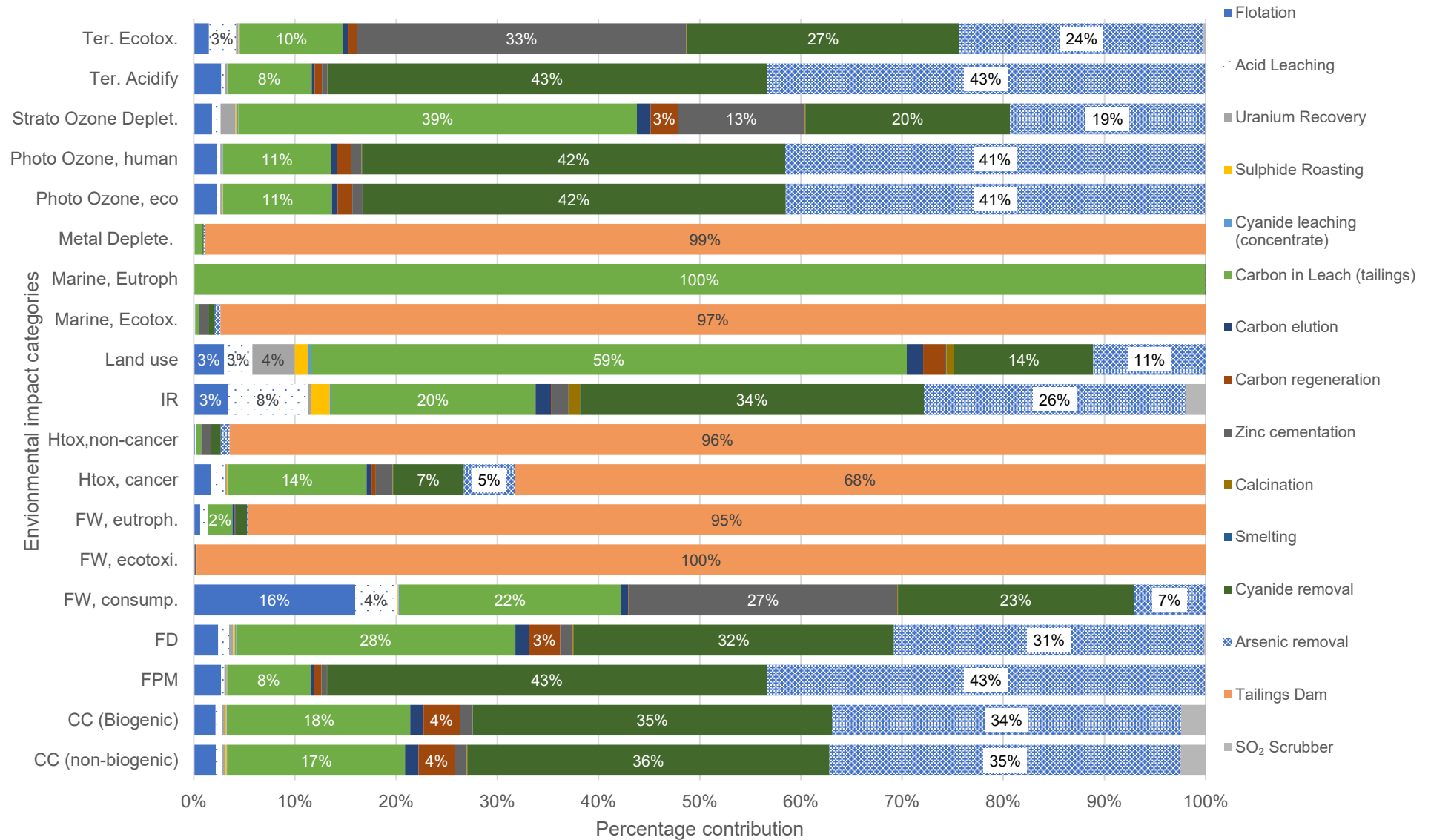
Figure 5-5: Cyanide flowsheet unit process contribution to each impact category
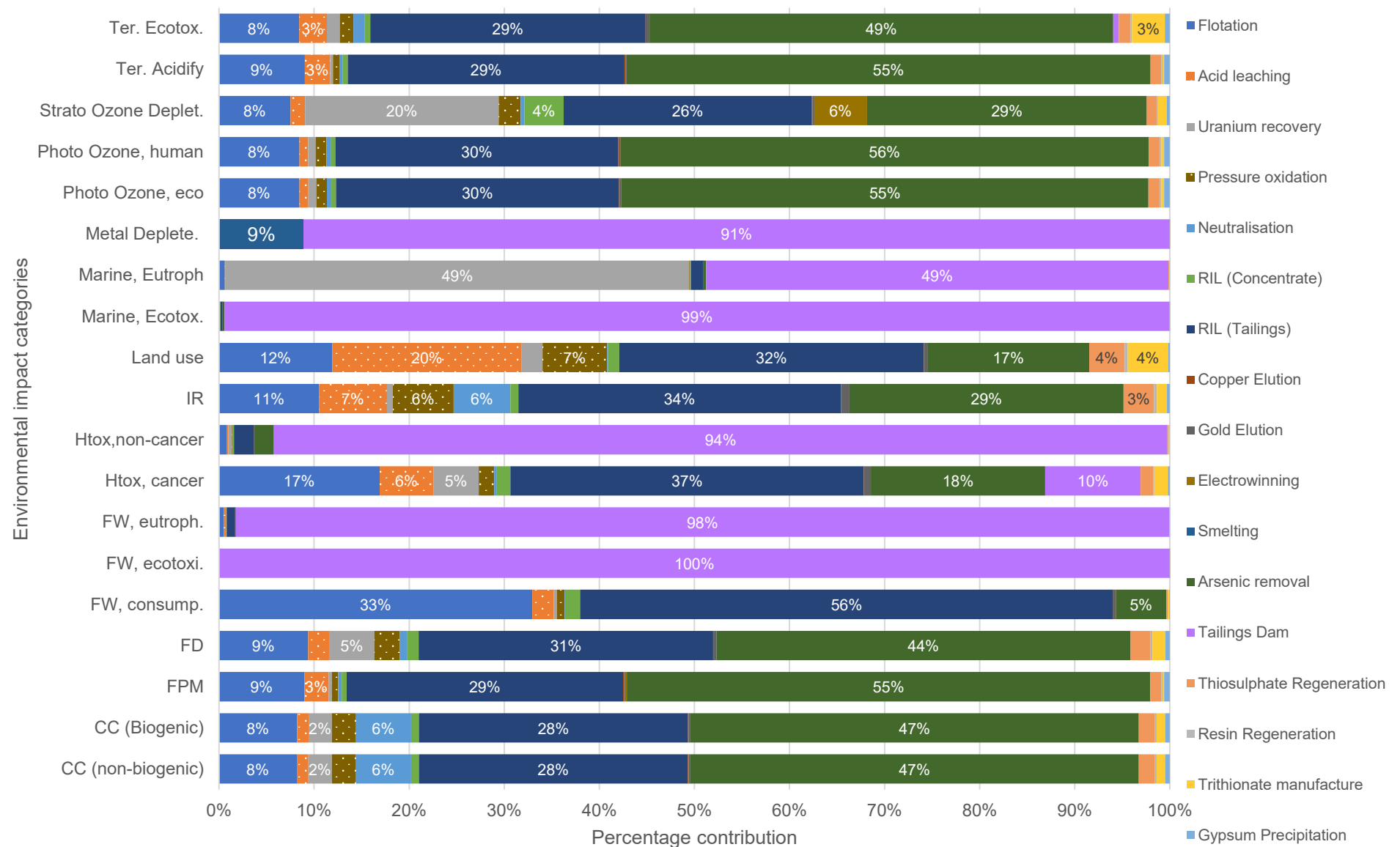
Figure 5-6: Thiosulphate flowsheet unit process contribution to each impact category

Table 5-5: Endpoint LCIA results (functional unit on functional unit 1kg gold) for cyanide and thiosulphate flowsheets

Cyanide flowsheet		Thiosulphate flowsheet	
Impact category [Unit]	**Magnitude**	**Impact category [Unit]**	**Magnitude**
Human Health [DALY]	13.1	Human Health [DALY]	2.57
Ecosystem quality [species.yr]	2 368	Ecosystem quality [species.yr]	2 368
Resource scarcity [$]	91 374	Resource scarcity [$]	29 213

Table 5-6: Contributional analysis and ranking of unit processes by endpoint LCIA results (functional unit on functional unit 1kg gold)

Endpoint impact categories [units]								
Human Health [DALY]			**Ecosystem quality [species.yr]**			**Resource scarcity [$]**		
1	Cyanide Wastewater treatment	7.33	1	Uranium leaching and recovery	2 368	1	Cyanide Gold Leaching	45 345
2	Cyanide Lixiviant management	3.32	2	Uranium leaching and recovery	2 368	2	Cyanide lixiviant management	11 068
3	Cyanide Gold Leaching	1.28	3	Cyanide lixiviant management	7.66E-03	3	Thiosulphate Gold Leaching	10 225
4	Wastewater treatment	1.27	4	Cyanide Wastewater treatment	7.54E-03	4	Cyanide Wastewater treatment	9 934
5	Thiosulphate Gold Leaching	0.73	5	Cyanide Gold Leaching	3.61E-03	5	Thiosulphate Wastewater treatment	7 884
6	Cyanide Gold Recovery	0.30	6	Thiosulphate Wastewater treatment	3.09E-03	6	Cyanide Gold Recovery	2 289
7	Flotation	0.22	7	Thiosulphate Gold Leaching	1.89E-03	7	Flotation	2 170
8	Flotation	0.22	8	Cyanide Gold Recovery	8.61E-04	8	Flotation	2 169
9	Sulphide Roasting	0.13	9	Flotation	5.42E-04	9	Uranium leaching and recovery	1 812
10	Pressure oxidation	0.10	10	Flotation	5.42E-04	10	Uranium leaching and recovery	1 811
11	Uranium leaching and recovery	0.06	11	Sulphide oxidation	3.88E-04	11	Thiosulphate management	1 053
12	Uranium leaching and recovery	0.05	12	Sulphide oxidation	3.07E-04	12	Thiosulphate Gold Recovery	1 000
13	Thiosulphate management	0.05	13	Thiosulphate Lixiviant management	1.23E-04	13	Pressure oxidation	875
14	Thiosulphate Gold Recovery	0.02	14	Thiosulphate Gold Recovery	6.83E-05	14	Sulphide Roasting	416

As discussed in section 2.6.1, endpoint analysis reduces complexity of analysis by predicting the ultimate effect of emissions on only three key areas of protection as summarised in Table 5-5. The human health category reflects the shortening of human life leading to premature death brought about by disease and disability caused by pollution of the ecosphere. Ecosystem quality refers to the extinction of fauna and flora species because of influence of pollutants on habitat contamination and destruction. Finally, the resource scarcity category reflects the loss of non-renewable resources such as fossil fuels and minerals. The endpoint analysis information was further stratified in Table 5-6 to rank unit processes according to how the life cycle impact affect the key areas of protection from highest to lowest. This ranking system allowed gold processing technologies to be compared against one another (cyanide technologies in blue and thiosulphate technologies in green).

The impact on ecosystem quality for both flowsheets was equal (i.e. 2 368 species.yr in Table 5-5) because dodecanol emissions (used as an additive during uranium solvent extraction) to ecosphere account for more than 99% of contribution to ecosystem quality impact category for both flowsheets. Dodecanol emissions management will be discussed in section 5.2.1.5 along with other emissions to ecosphere.

Based on the ranking system in Table 5-6, for impact on ecosystem quality, the cyanide flowsheet resulted in 2.3 times more impact for gold leaching and recovery, 4.7 times more impact for effluent treatment and lixiviant management and 1.26 times more impact for sulphide oxidation when compared to thiosulphate flowsheet. For the human health category, the cyanide flowsheet resulted in 2.1 times more impact for gold leaching and recovery, 4.7 times more impact for effluent treatment and lixiviant management and 1.3 times more impact for sulphide oxidation when compared to thiosulphate flowsheet (as summarised in Table 5-6). For impact on resource scarcity, the cyanide flowsheet resulted in 4.2 times more impact for gold leaching and recovery and 2.4 times more impact for effluent treatment and lixiviant management when compared to thiosulphate flowsheet (as summarised in Table 5-6). Sulphide roasting, however, results in 0.48 times less impact on resource scarcity than pressure oxidation.

The general trend in Table 5-6 using endpoint analysis was that the thiosulphate flowsheet unit processes results in diminished environmental impacts for the three areas of protection human health, ecosystem quality and resource scarcity. The reduction in resource scarcity of sulphide roasting compared to pressure oxidation is attributed to the $CaCO_3$, oxygen and heating requirements for pressure oxidation. Therefore, the trade-off for a reduction in environmental impacts that benefit human health and ecosystem quality is that more resources are consumed to employ pressure oxidation to decompose sulphide minerals. Aspects and impacts related to air quality from sulphide oxidation processes was discussed in section 5.2.1.3 along with other emissions to ecosphere. The environmental impact of effluent treatment and lixiviant management in the cyanide flowsheet was exacerbated by cyanide destruction and arsenic removal technologies requiring a

dilute solution of 12.5% sodium hypochlorite together with the electricity consumption for agitation (as discussed in section 5.2.1.1). The increased environmental impacts for gold leaching and recovery in the cyanide flowsheet was brought about by the life cycle impacts of raw materials and electricity consumption during cyanide leaching as indicated in Table 5-4. The recycling and regeneration of lixiviant in the thiosulphate flowsheet aids in reducing the raw material consumption which is evidenced by the resourced scarcity impact category in Table 5-4.

Table 5-7: Gold leaching and recovery contributional analysis of endpoint results for thiosulphate and cyanide flowsheets

	Thiosulphate flowsheet			Cyanide flowsheet		
	Human Health [DALY]	Ecosystem quality [species.yr]	Resource scarcity [$]	Human Health [DALY]	Ecosystem quality [species.yr]	Resource scarcity [$]
Raw materials	0.13	4.40E-04	6 363	0.85	2.64E-03	44 851
Electricity and utility	0.66	1.51E-03	1 988	0.74	1.85E-03	3 029
Total	**0.79**	**1.95E-03**	**8 351**	**1.65**	**4.49E-03**	**47 880**

The proposed cyanide flowsheet does not include cyanide regeneration (Figure 5-1). Including the regeneration and recycling of cyanide in Figure 5-1 may have been a less biased comparison when comparing it to a flowsheet where thiosulphate was regenerated. Cyanide recovery technologies have been demonstrated in commercial processes such as acidification-volatilisation and regeneration (AVR) (Kuyucak and Akcil, 2013). Only 50% cyanide recovery can be achieved in practice and the evolution of hydrogen cyanide gas requires strict occupational health and safety controls (Eksteen et al., 2018). Therefore, cyanide recovery processes were not incorporated into gold recovery flowsheets in literature and the baseline flowsheet also did not consider this. It should be noted that recycling and recovering cyanide will reduce cyanide consumption and reduce the resource requirements for cyanide destruction (specifically reagents and electricity required for agitation). By reducing resource consumption, the corresponding environmental impacts that are linked to resources are also reduced. The opportunity for reduction of environmental impacts through cyanide recycling and regeneration was not proven in this study and the benefit needs to be compared with the cost of additional resource requirements that accompany the additional unit process of cyanide recycling and regeneration.

5.2.1.3. Hotspot analysis on emissions to ecosphere

The life cycle impact of emissions to ecosphere from the foreground process for both flowsheets was calculated in Microsoft Excel® as explained in section 3.4.1. The LCIA midpoint results of emissions to ecosphere from each flowsheet is summarised in Table 5-8 with the percentage contribution of each emission to environmental impact categories stratified for the cyanide and thiosulphate

flowsheets in Figure 5-7 and Figure 5-8 respectively. Only the 11 out of 19 impact categories were affected by direct emissions from foreground processes were listed in Table 5-8. For comparison, an LCIA was conducted on a "No action" scenario where the mine tailings were left undisturbed and not subjected to processing by the flowsheets developed to assess how the flowsheets affect the environmental burden of mine tailings.

Table 5-8: LCIA midpoint impact assessment results for emissions to ecosphere for flowsheets (results expressed per 1 kg gold (functional unit)

Impact category	No action	Thiosulphate leaching	Cyanide leaching
CC (non-biogenic) (kg CO_2 eq.)	0.00	27,275.13	69,842.99
CC (Biogenic) (kg CO_2 eq.)	0.00	27,275.13	69,842.99
FPM (kg $PM_{2.5}$ eq.)	0.00	0.00	1.15
FW, ecotox. (kg 1,4-DB eq.)	79,333.99	56,617.40	63,432.32
FW, eutroph. (kg P eq.)	34.34	44.25	44.26
Htox, cancer (kg 1,4-DB eq.)	5,373.12	13.76	1,263.13
Htox,non-cancer (kg 1,4-DB eq.)	1,593,145.87	289,581.72	2,027,323.97
Marine, Ecotox. (kg 1,4-DB eq.)	18,155.40	21,410.52	27,417.42
Marine, Eutroph (kg N eq.)	0.00	46.19	682.48
Ter. Acidify (kg SO_2 eq.)	0.00	0.00	3.96
Ter. Ecotox. (kg 1,4-DB eq.)	5.75E-05	3.23E-05	3.24E-05

The thiosulphate flowsheet reduced emissions to the ecosphere for 9 out of 11 impact categories in Table 5-8 compared to the cyanide flowsheet, while the impact to the remaining two impact categories was equal. When compared to the "No action" scenario, the thiosulphate flowsheet reduced the burden to freshwater ecotoxicity by 28% through sulphide mineral removal, human toxicity (non-cancer) by 99% and human toxicity through arsenic (cancer) by 81% through arsenic removal. Although the cyanide flowsheet also incorporated the aforementioned unit processes, the additional emissions by reagents fed increased environmental impact. Unfortunately, the flowsheets proposed caused an increase in environmental impacts for the remaining impact categories when compared to the "No action" scenario. The origins of these impacts will be explained in the ensuing sections addressing impacts on human health, aquatic and terrestrial ecosystems and atmosphere (sections 5.2.1.4 to 5.2.1.6).
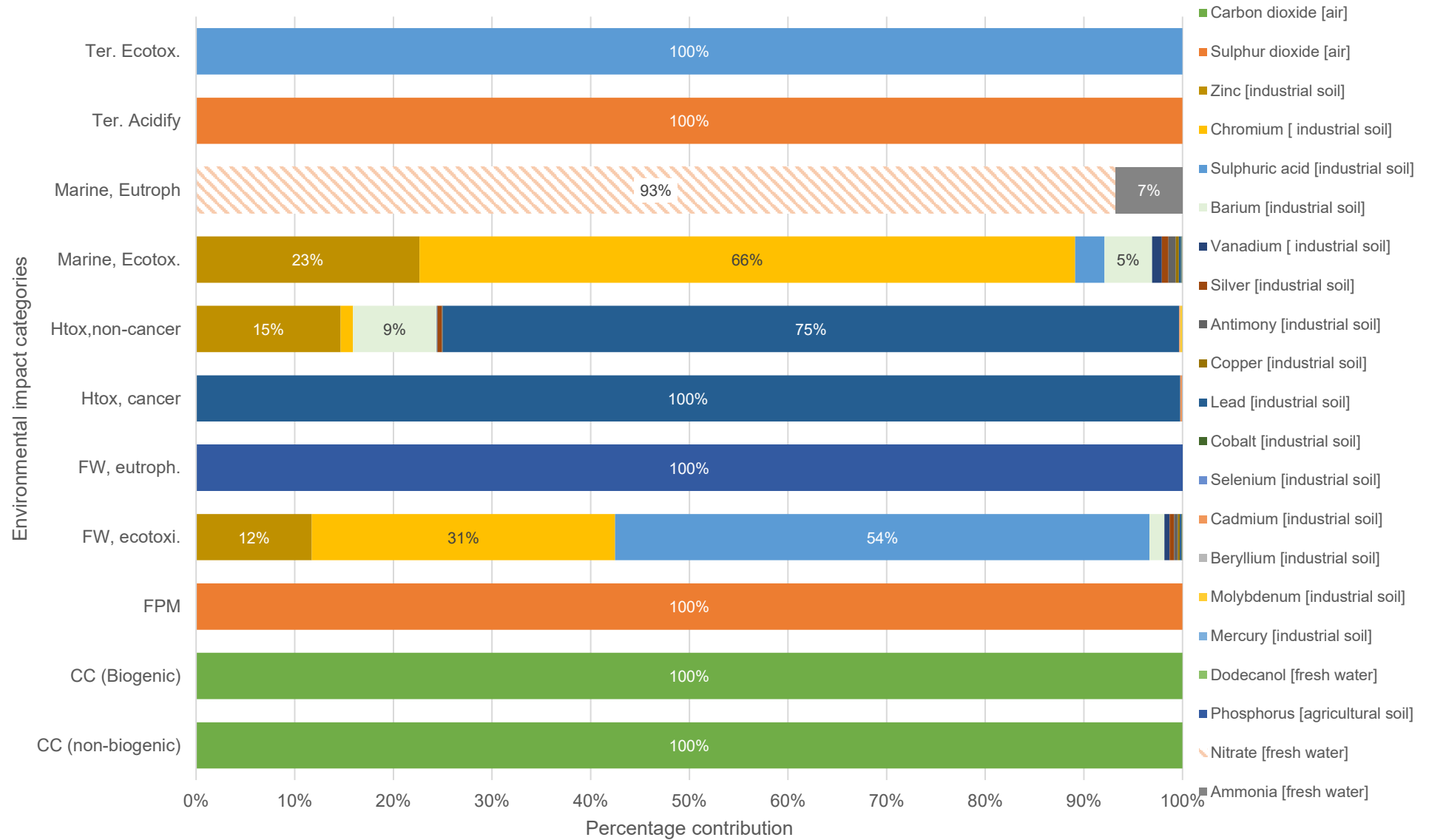
Figure 5-7: Cyanide flowsheet emissions to ecosphere from foreground process contribution to each impact category
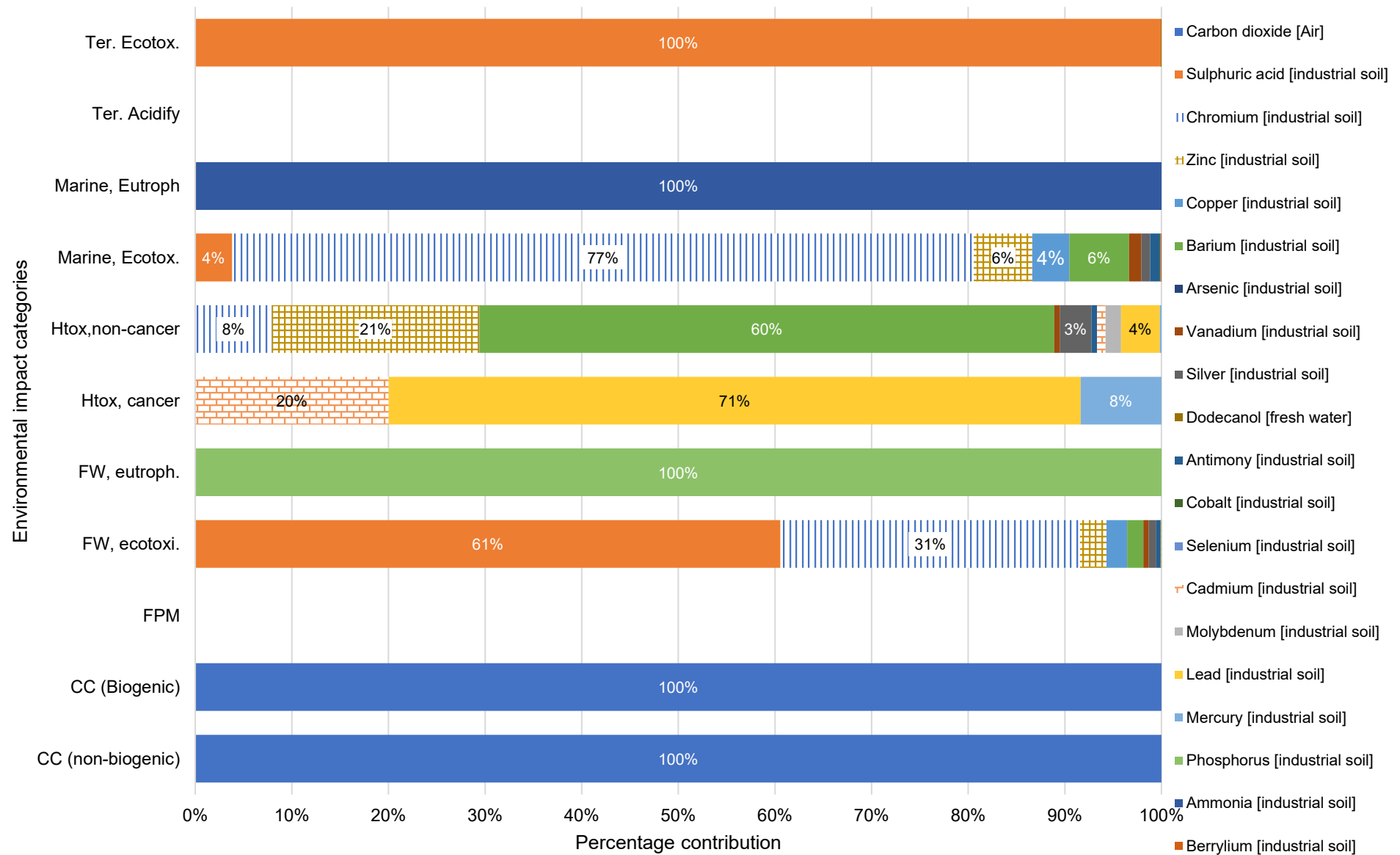
Figure 5-8: Thiosulphate flowsheet emissions to ecosphere from foreground process contribution to each impact category

5.2.1.4.   Process impact on human health

The human toxicity impact category quantifies the cause-and-effect relationship between the emission of pollutants on the health of the general population (Huijbregts et al., 2016). Following on from the discussion in section 2.2, literature has reported that communities near mine tailings dams in the Witwatersrand region are at risk of developing cancer and non-cancer related health complications because of radionuclide and heavy metal aeolian emissions. The risk is exacerbated for subsistence farmers who cultivate food to support their families in these regions (Kamunda et al., 2016a; Ngole-Jeme and Fantke, 2017; Tutu et al., 2008). Therefore, hazardous emissions from process flowsheets need to be minimised to ensure that mineral processing activities do not affect the health of the general population.
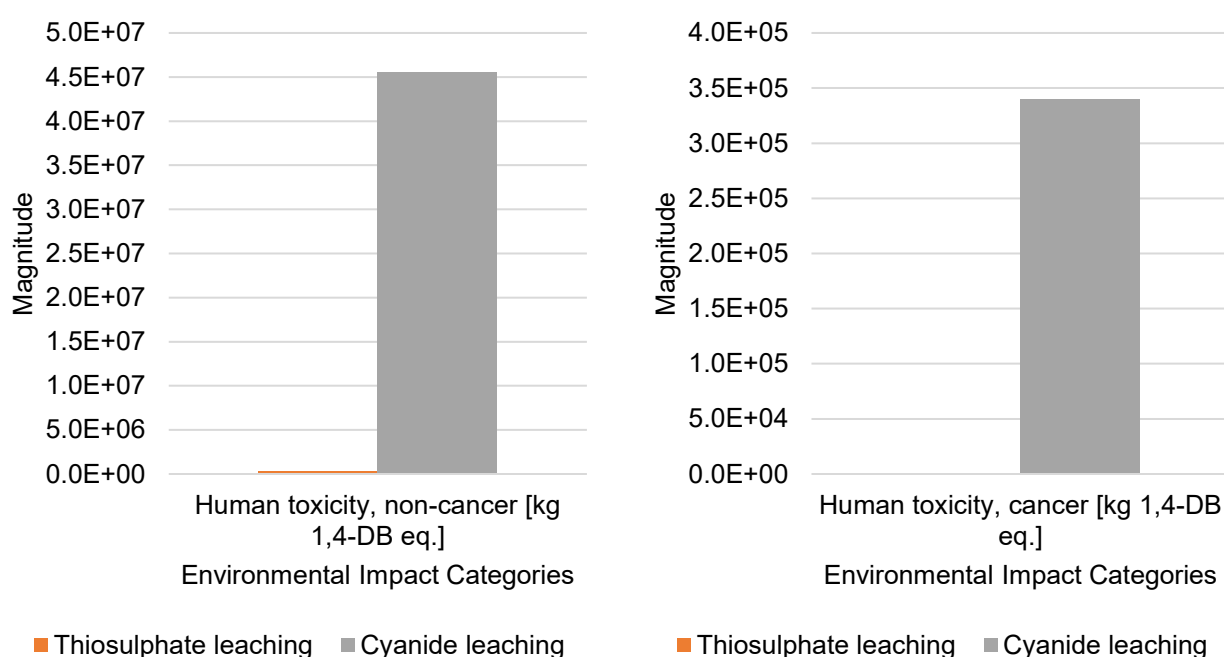
Figure 5-9: Life cycle human toxicity (cancer) and (non-cancer) impact category results for proposed flowsheets

In an analysis at background process level, the thiosulphate flowsheet resulted in environmental impacts that are three- and two- orders of magnitude less for human toxicity (non-cancer) and human toxicity (cancer) impact categories respectively when compared to the cyanide flowsheet (Figure 5-9). For the cyanide flowsheet, the emissions from wastewater treatment contributed to 99% and 86% of the life cycle impacts to human toxicity (cancer) and human toxicity (non-cancer) impact categories respectively. Opportunities for how this may be managed was discussed in section 5.2.1.1. At foreground process level, in the cyanide flowsheet lead (Pb) emissions to industrial soil contributed towards 100% and 75% of the effects to the human toxicity (cancer) and human toxicity (non-cancer) impact categories respectively (Figure 5-7).Zinc emissions accounted for 15% of human toxicity (non-cancer) impact category contributions (Figure 5-7).

The origin of lead and zinc was the addition of lead nitrate and powdered zinc during precious metal recovery during zinc cementation. A cost-benefit analysis of lead nitrate as a performance additive for gold recovery is necessary to determine if externalised costs associated with environmental pollution and depleted tailings management can be justified by the gold recovery improvement. The assumption of 1.5 times the stoichiometric requirement of zinc powder fed during the zinc cementation unit process has flagged zinc as a hazardous emission. The stoichiometric excess allowed optimisation of precious metal cementation and compensates for base and heavy metals being preferentially cemented (Marsden and House, 2006). Opportunities to recover zinc and lead before it reports to the tailings dam may aid in reducing the contribution of zinc and lead emissions to human toxicity impact categories.

Alternatively, electrowinning may be considered for gold recovery for cyanide flowsheet since it was not flagged to have a large environmental impact for the thiosulphate flowsheet in the midpoint level assessment of unit processes Figure 5-6 (section 5.2.1.2). Furthermore, thiosulphate gold recovery unit processes (including electrowinning) was ranked 10th in Table 5-6 for endpoint level human health and ecosystem quality areas of protection. The aforementioned results were observed because the electricity consumption for electrowinning was calculated on the assumption of 3 100 kWh/tonne gold from Norgate and Haque (2012) (as stated in section 4.9). Therefore, the electricity consumption for electrowinning was negligible compared to upstream unit processes in the thiosulphate flowsheet (section 4.9). There is, however, a trade-off in terms of the purity of the crude gold bullion that can be formed owing to the co-deposition of copper on the cathode along with gold which increases upstream refining process demands to remove copper and form a 99.99% pure gold bullion as discussed in 2.4.8.2. Despite this, electrowinning does have the potential environmental benefit of not liberating hazardous emissions to the ecosphere as was the case with zinc cementation. The life cycle impacts of electrowinning for cyanide flowsheet was not evaluated in this study and a supplementary study needs to be done to provide evidence of hypothesised benefit.

The thiosulphate process reduced environmental impact on human toxicity compared to the "No action" scenario for solid effluent emissions from foreground process because of the arsenic removal unit process which captured arsenic as scorodite ($FeAsO_4.2H_2O$) as mentioned in section 5.2.1.3. At foreground process level for the thiosulphate flowsheet, the hazardous emissions (i.e. lead, barium, zinc and cadmium) represented for human toxicity in Figure 5-8 were all present in the tailings feedstock therefore the environmental burden of these species was not reduced compared to the "No action" scenario. Although barium, vanadium and zinc contribute to human toxicity indicators in the LCIA results, the solid waste emissions complied with environmental regulation for the protection of water resources in Table 5-1. Except for lead, manganese and zinc both flowsheets reduced the environmental impact of the gold mine tailings feedstock through the dilution effect and

process interventions. Additional processing of solid effluent emissions, for example phytoremediation, is required to reduce burden to human toxicity as discussed in section 5.1.1.

5.2.1.5. *Process impact on aquatic and terrestrial ecosystems*

Freshwater-, marine- and terrestrial- ecotoxicity impact categories reflects the effect emissions will have on fauna and flora that reside within these ecosystems (Huijbregts et al., 2017). The idealised goal of mining operations is to extract beneficial materials from the earth while ensuring that the surrounding ecosystem is returned to the condition before the mine was commissioned. Unfortunately, the impact of mining activities may result in habitat destruction, biodiversity loss and in severe cases irreversible damage to ecosystems caused by acid mine drainage (Fashola et al., 2016; Naicker et al., 2003; Ngole-Jeme and Fantke, 2017).
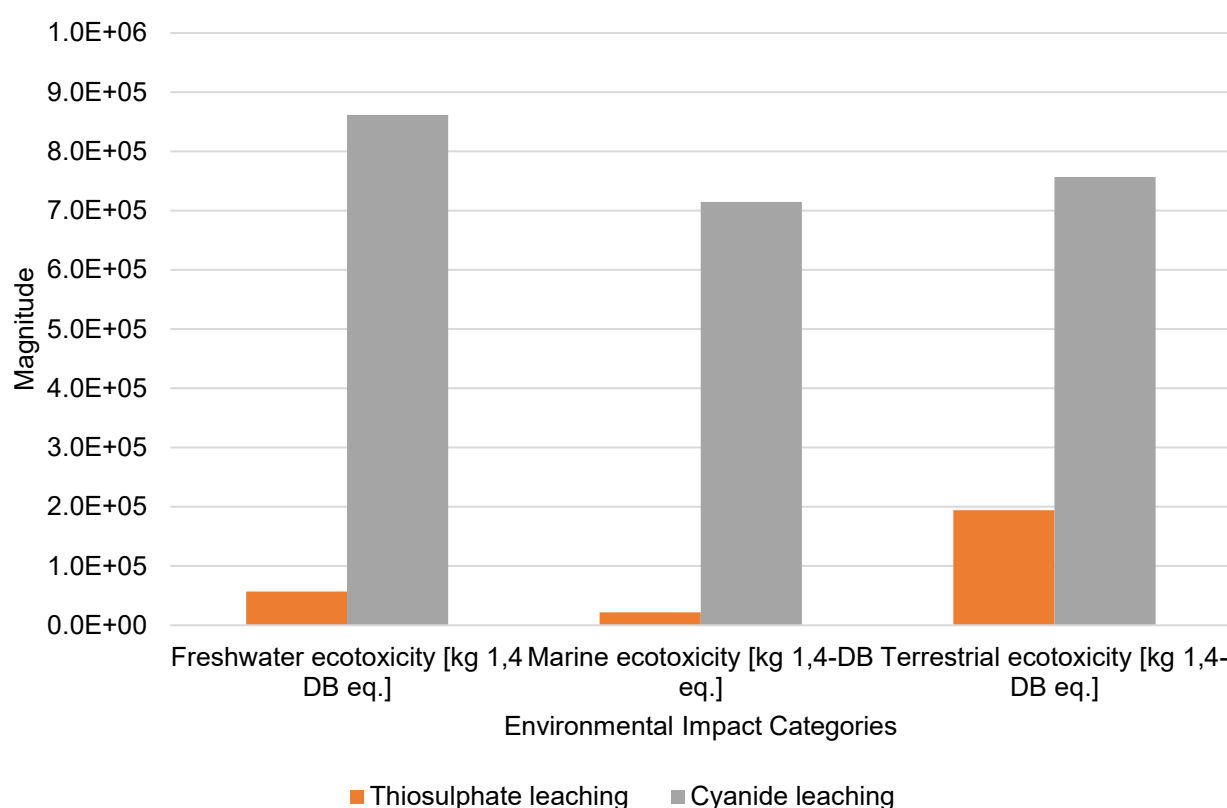
Figure 5-10: Influence on life cycle freshwater, marine and terrestrial toxicity impact categories by proposed flows sheets for gold recovery from mine tailings

The thiosulphate flowsheet reported a reduced environmental impact on freshwater-, marine- and terrestrial-ecotoxicity impact categories by 46%, 74% and 80% respectively compared to the cyanide flowsheet (Figure 5-10 in combination with Table 5-4). At background process level for both flowsheets, the wastewater treatment activated sludge emissions contribute to the majority impacts on the freshwater- and marine ecotoxicity impact categories while electricity was the majority contributor for terrestrial ecotoxicity (Figure 5-3 and Figure 5-4). The ecological impact of thiosulphate flowsheet emissions to freshwater- and marine- ecosystems was less than the cyanide

flowsheet by 12% and 28% respectively while the influence on terrestrial ecosystem was equal as summarised in Table 5-8 of section 5.2.1.3.

In the cyanide flowsheet, the background processes of electricity and zinc powder manufacture accounted for 54% and 30% of the life cycle impacts on terrestrial ecotoxicity. Electricity consumption for the cyanide flowsheet was 3.7 times larger than for the thiosulphate flowsheet (for which an explanation was presented in section 5.2.1.1) hence the large disparity reported for terrestrial ecotoxicity burden for the cyanide flowsheet. The environmental impact of zinc cementation as a unit process was increased by the life cycle impacts associated with zinc powder manufacture. As discussed in section 5.2.1.4, electrowinning may be considered as an alternative to zinc cementation to reduce effect on terrestrial ecosystems for the cyanide flowsheet. The drawback of this recommendation is that the crude gold bullion purity may have to be sacrificed because of copper plating preferentially to gold during electrowinning as discussed in section 2.4.8.2.

For the thiosulphate flowsheet, emissions from the foreground process represented 99% and 100% of the influence on marine- and freshwater- ecotoxicity respectively while electricity accounted for 80% of the environmental indicators on terrestrial ecotoxicity (Figure 5-4). The emissions from the thiosulphate foreground process that were major contributors to freshwater- and marine- ecotoxicity impact categories were chromium and sulphuric acid (Figure 5-8).

Metals in solid waste emissions from both flowsheets that were highlighted by the LCA to affect human health and ecotoxicity include chromium, zinc, lead, barium, cadmium and mercury (Figure 5-7 and Figure 5-8). The evaluation of solid emissions against environmental regulation in Table 5-1 in section 5.1.1, revealed that manganese and copper emissions from the thiosulphate flowsheet; and lead, manganese and copper emissions from the cyanide flowsheet were non-compliant. When comparing LCA results with environmental regulation for solid waste emissions one may conclude that chromium, zinc, cadmium and mercury emissions are not of concern because they were compliant during the evaluation in Table 5-1. Additional interventions, such as phytoremediation, are required to sequester metals from mine tailings and render mine tailings benign to the environment (as discussed in section 5.1.1). Influence of phytoremediation was not tested in this study but is offered as a suggestion to reduce life cycle environmental impacts.

Manganese emissions originates from manganese dioxide used as an oxidant during uranium leaching and potassium permanganate used as an oxidant during arsenic removal in both flowsheets. For the arsenic removal unit process, the environmental burden of manganese may be reduced by investigating the life cycle impact of alternative oxidants to potassium permanganate such as chlorine dioxide, sodium hypochlorite and monochloramine (Sorlini and Gialdini, 2010). A cost-benefit analysis of using manganese dioxide for uranium recovery needs to be done to determine if the environmental impacts can be justified by the improvement in uranium recovery.

Lead emissions to ecosphere originated from lead nitrate used as a zinc cementation reagent to increase gold recovery in the cyanide flowsheet. A cost-benefit analysis evaluating the trade-offs between the improvements to gold recovery that lead nitrate offers and ecological impacts of lead for human and ecosystem health is necessary (as discussed in section 5.2.1.4). Evidence to support the aforementioned proposals to reduce environmental impacts was not determined in this study and are offered as suggestions for subsequent studies.

Sulphuric acid produced from residual sulphide in solid waste emissions from both flowsheets were highlighted as a hazard to freshwater- and terrestrial- ecotoxicity (Figure 5-7 and Figure 5-8). The flowsheet solutions proposed reduced the impact of acid mine drainage by 83% (discussed in section 5.1.1). Improvements in flotation technology to increase sulphide mineral recovery beyond 85% may be considered to further reduce environmental impacts of sulphuric acid produced during acid mine drainage. This recommendation has the added benefit of increasing gold recovery since 7.27% of gold is occluded within sulphide minerals (as discussed in section 2.3). The acid mine drainage potential of the solid effluent emissions should first be quantified to ascertain if flotation technology improvements for sulphide recovery are necessary.
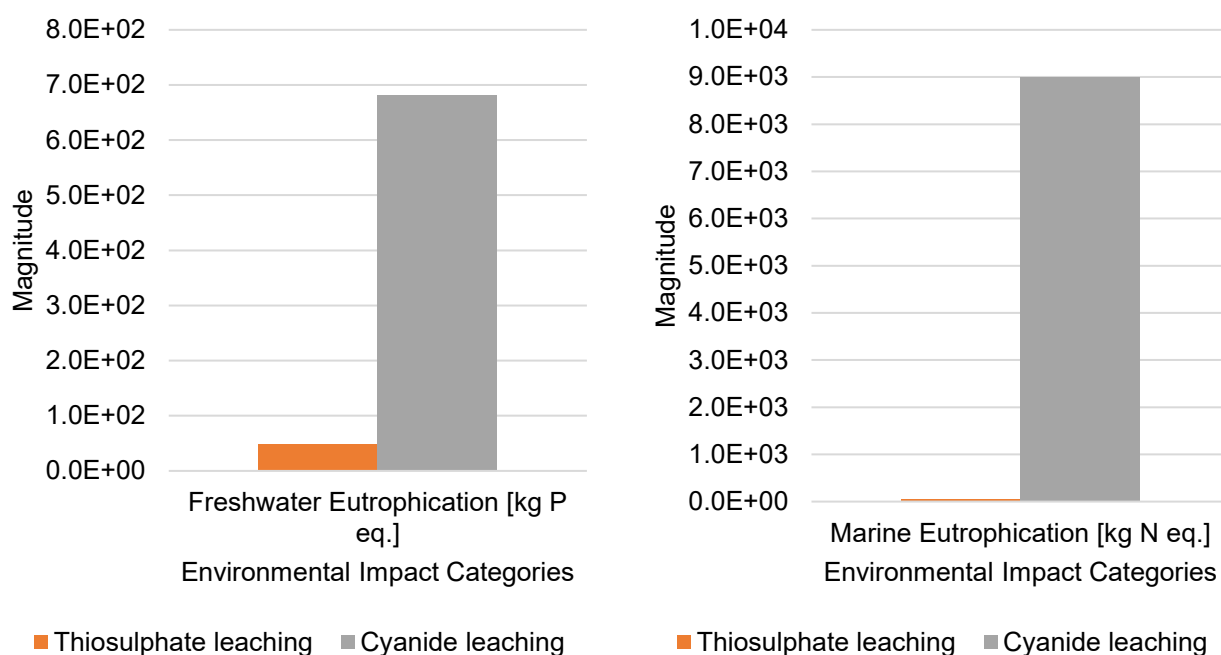
Figure 5-11: Influence on life cycle freshwater and marine and eutrophication impact categories by flowsheets for gold recovery from mine tailings

Freshwater- and marine- eutrophication reflects the rise in nutrients, specifically nitrogen and phosphorous, in aquatic ecosystems that culminate in species loss owing to hypoxia (oxygen depletion) and aquatic ecosystem destruction (Huijbregts et al., 2017). The cyanide flowsheet affected freshwater and marine eutrophication by more than one order of magnitude compared to the thiosulphate process (Figure 5-11).

In the cyanide flowsheet, the wastewater treatment emissions of activated sludge along with the emissions from the activated carbon dataset created are the most notable contributors to eutrophication (Figure 5-3). The activated sludge emanating from wastewater treatment may be used as agricultural fertiliser for energy crops or incinerated to provide thermal energy and is an opportunity to reduce environmental impact as discussed in section 5.2.1.1. Activated carbon background dataset was created from data reported by Arena et al. (2016) (as discussed in section 3.4.4.). Tar produced as a by-product of activated carbon manufacture accounted for 75% of for marine eutrophication impact (Figure 5-3). Sourcing of activated carbon from ISO 14 000 certified suppliers that minimise environmental impact may aid in reducing marine eutrophication burden for the cyanide flowsheet.

Ammonia (uranium recovery reagent) emissions from both flowsheets contribute to marine eutrophication while phosphorous present in the tailings feedstock added to the eutrophication burden in freshwater resources (Figure 5-7 and Figure 5-8). Furthermore, nitrate emissions from lead nitrate use in the cyanide flowsheet increases marine eutrophication burden (Figure 5-7). Although the land-locked region of the Witwatersrand is 575 kilometres from the nearest ocean, the life cycle impact of these emissions needs to be managed since surface- and ground-water systems can transport hazardous species to marine ecosystems. Struvite ($NH_4MgPO_4 \cdot 6H_2O$) chemical precipitation has been used in semi-conductor and animal processing industries remove nitrogen and phosphorous from wastewaters (Altinbaş et al., 2002; Karri et al., 2018; Ryu et al., 2008). The function of the ammonia bleed stream was to prevent accumulation of waste products in recycle streams. The requirement for a bleed stream should be re-evaluated owing to the environmental burden of ammonia discharge. Sodium chloride precipitation may be used as an alternative to ammonia precipitation with the disadvantage of reducing purity of the yellow-cake uranium produced by 10% U_3O_8 (Zhu and Cheng, 2011). The recommendations of struvite chemical precipitation for ammonia and phosphorous removal along with sodium chloride precipitation to produce sodium diuranate was not evaluated in this study but is offered as a suggestion for supplementary studies.

Dodecanol (additive used to improve uranium recovery during solvent extraction) bleed streams to the ecosphere was flagged as a burden to ecosystem quality in section 5.2.1.2. The environmental impacts associated with dodecanol (a fatty alcohol) emissions may be reduced by incinerating dodecanol to produce CO_x and methane by-products to mitigate hazard of exposure to aquatic ecosystems (Choi and Watanabe, 2012). The incineration of bleed streams was not incorporated into proposed flowsheets but is a possible solution to reduce environmental impacts. Dodecanol is in-soluble in water therefore physical sequestration of dodecanol from aqueous streams with organic phase adsorbents such as activated carbon with soil adsorption coefficient (K_{oc}) values > 7700 is quintessential. Adsorption is then followed by incineration of the carbon once adsorption capacity has been reached is required (National Center for Biotechnology Information, 2020). A cost-benefit

analysis for the use of dodecanol in uranium recovery needs to be conducted to ascertain if there is a significant improvement in uranium recovery considered the environmental impact it poses to ecosystem quality.

5.2.1.6. *Process impacts on atmosphere and ionising radiation*

Fine particulate matter represents the risk for human respiratory morbidity caused by aerosols (finely dispersed liquids and solids in the air). Ionising radiation refers to the effects of radionuclides on the ecosystem such as the carcinogenic effects on human and animal life. Finally, terrestrial acidification reflects the shift in optimum acidity of soils for plant species because of the effects of acid rain brought about by atmospheric emissions of sulphates, nitrates and phosphates (Huijbregts et al., 2017).
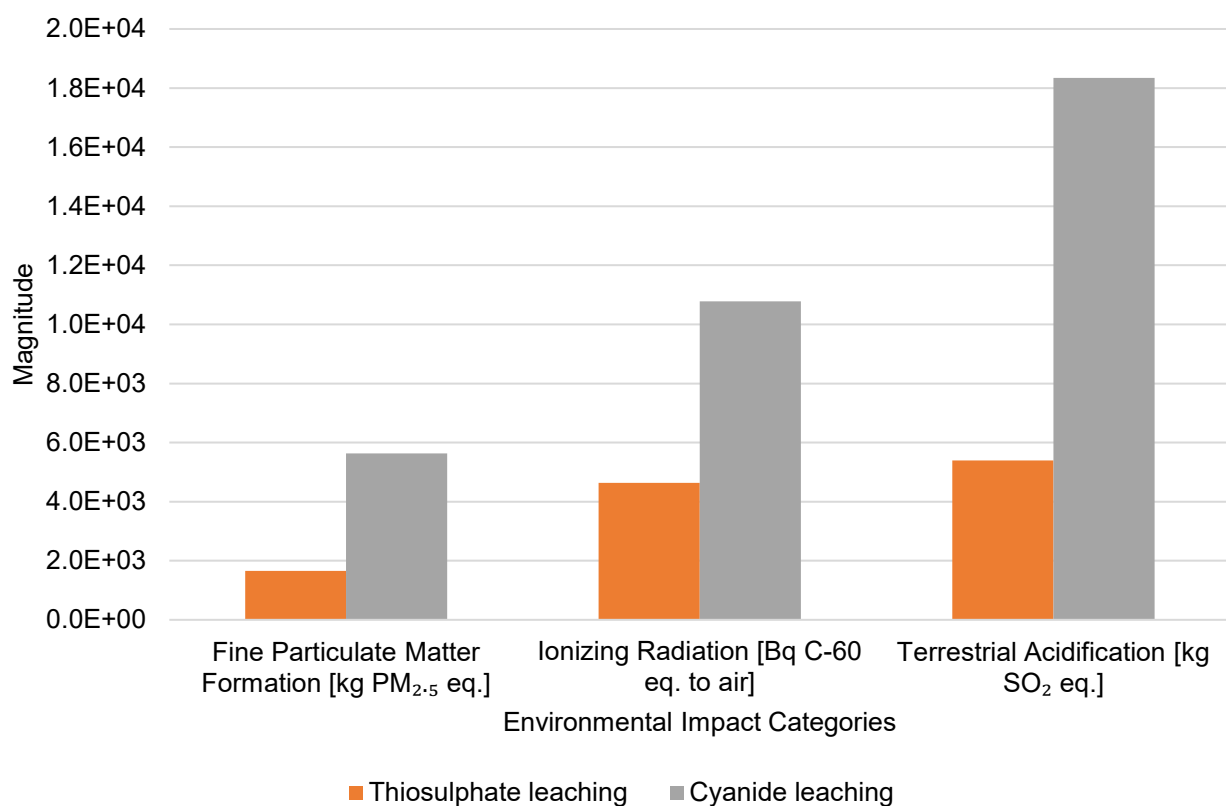
Figure 5-12: Effect on life cycle fine particulate matter, ionizing radiation and terrestrial acidification impact category for process flowsheets proposed

The thiosulphate flowsheet has reduced impact compared to the cyanide flowsheet for fine particulate matter formation (3.3 times), ionising radiation (2.3 times) and terrestrial acidification (3.3 times) as illustrated in Figure 5-12. The ecological impacts for fine particulate matter and terrestrial acidification were linked to electricity consumption in both flowsheets (Figure 5-3 and Figure 5-4). The reduced impacts by the thiosulphate flowsheet to the impact categories in Figure 5-12 was because of agitation electricity requirements for cyanide destruction and arsenic removal before wastewater treatment (as discussed in section 5.2.1.1). Ionising radiation is linked to nuclear power being incorporated into electricity grid mix for a specific region (Huijbregts et al., 2017). This may be

minimised by incorporating renewable electricity production into the electricity grid as discussed in section 5.2.1.1.

Photochemical ozone formation refers to tropospheric ozone which affects human and animal respiratory health causing chronic obstructive pulmonary diseases while stratospheric ozone depletion is representative of the increase in ultraviolet-B radiation leading to human health related concerns such as melanoma and cataracts (Huijbregts et al., 2017). The cyanide flowsheet produced 3.4 times the amount of photochemical ozone emissions and 4.2 times the quantity of stratospheric ozone emissions compared to the thiosulphate flowsheet (Figure 5-13 in combination with Table 5-4). This large difference between flowsheets was brought about by the background processes of sodium cyanide and electricity manufacture as resource requirements for CIL leaching of flotation tailings together with the cyanide destruction and arsenic removal processes (Figure 5-3). Electricity production was also the major contributor to these impact categories for the thiosulphate flowsheet (Figure 5-4). The effect on these impact categories may be managed by employing environmentally friendly technologies for electricity and lixiviant production, with an emphasis on photochemical ozone emission minimisation. The recommendation of the thiosulphate flowsheet minimised the impacts to human and animal health caused by photochemical ozone emissions and ozone layer destruction.
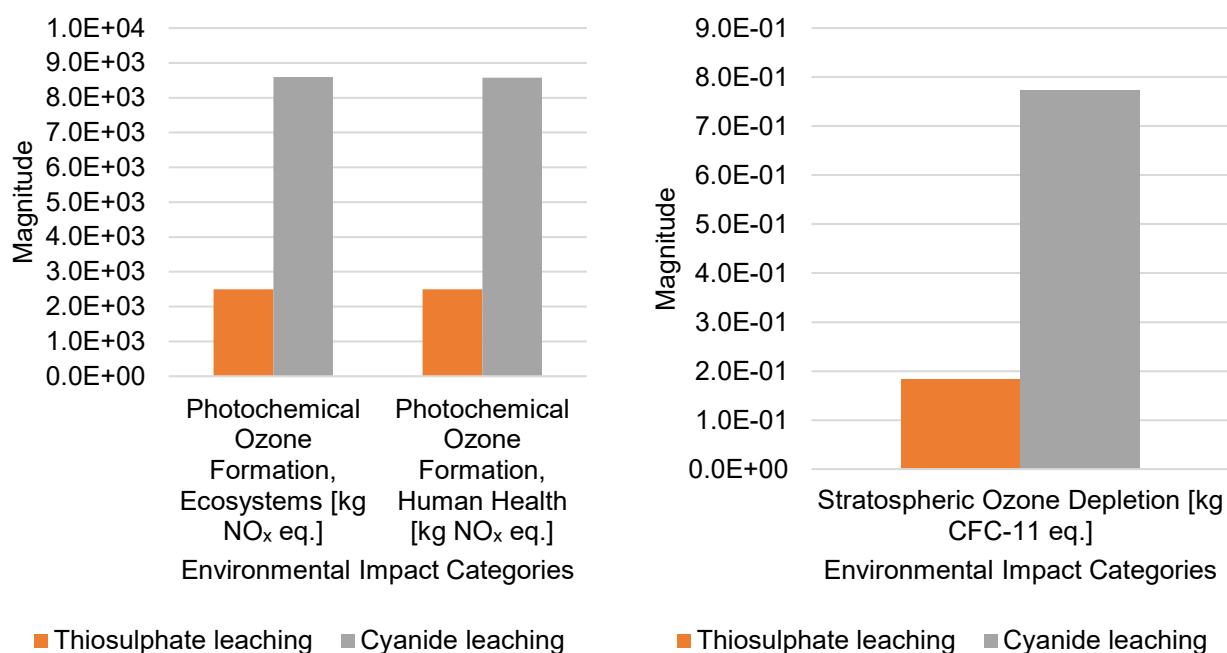
Figure 5-13: Effect on photochemical ozone formation and stratospheric ozone depletion for process flowsheets proposed
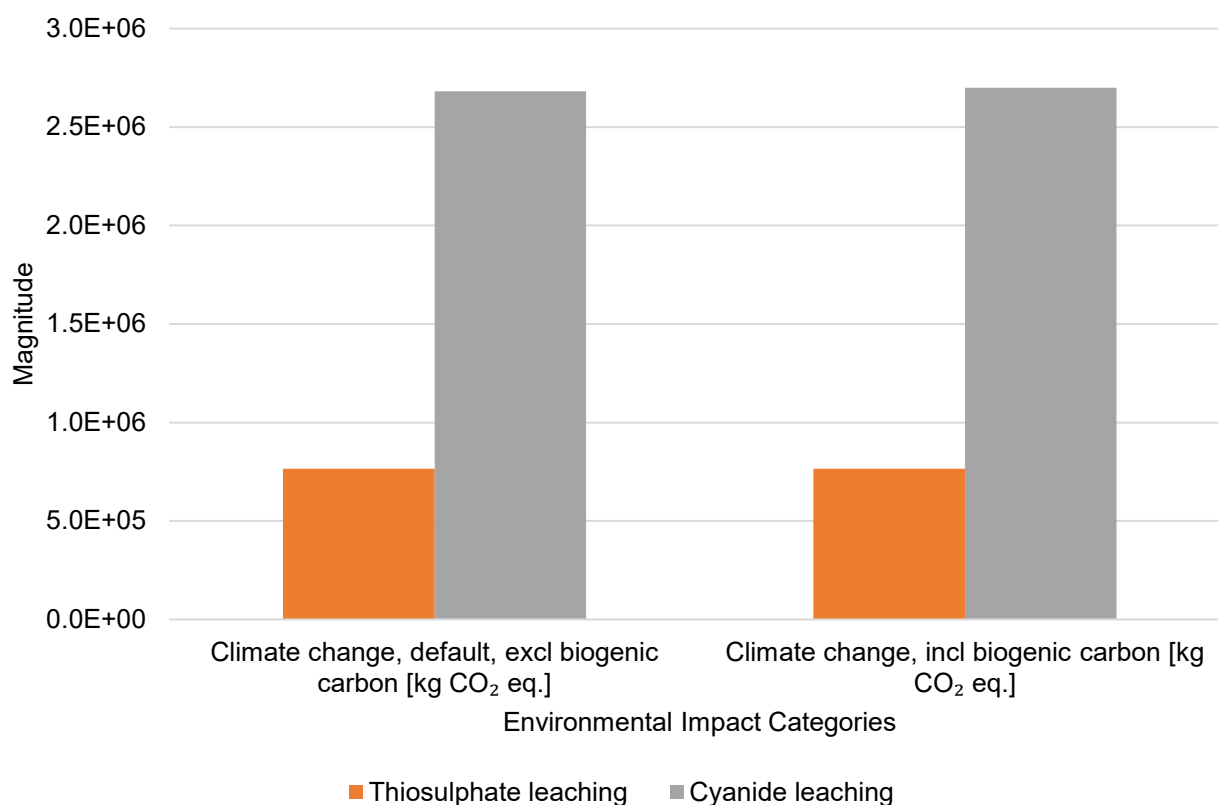
Figure 5-14: Influence on life cycle climate change impact categories of proposed flowsheets for gold recovery from mine tailings

In response to South Africa's commitment to reduce greenhouse gas emissions by 34% in 2020 and by 42% in 2025 under the Copenhagen accord, the National Development Plan and draft Carbon Tax Act policies have been implemented in South Africa to combat the burden of climate change (Department of Environmental Affairs, 2011; Republic of South Africa, 2019). Considering South Africa's dependency on electricity from coal, the mining industry may benefit from adopting strategies to reduce electricity consumption and corresponding greenhouse gas emissions to avoid implications of the carbon tax act on the profitability of their operations.

The greenhouse gas emissions to the ecosphere for the thiosulphate flowsheet was 3.5 times less than the cyanide flowsheet (Table 5-4). For the thiosulphate process, electricity consumption accounted for more than 78% of the contributions to climate change (Figure 5-4). For the cyanide flowsheet, electricity and sodium cyanide manufacture accounted for 86% of the contributions to this impact category (Figure 5-3). This observation stems from the fact that the cyanide flowsheet required 3.7 times more electricity for agitation to maintain optimum conditions for hydrometallurgical processes, specifically cyanide destruction and arsenic removal (as discussed in section 5.2.1.1). The leaching of flotation tailings required large quantities of lixiviant and electricity consumption for optimum gold recovery and to maintain solids in suspension because of the large mass flowrate of solids originating from flotation. In both flowsheets, the increased lixiviant and electricity consumption was necessary for leaching of flotation tailings to improve overall gold recovery from 44.5% to 71.2%.

In summary, the thiosulphate flowsheet offers an opportunity for mine houses to reduce greenhouse gas emissions compared to the conventional cyanide flowsheet proposed. Incorporating renewable energy into the electricity grid of mines may further reduce the greenhouse gas emissions (as discussed in section 5.2.1.1) and consequently is an opportunity to avoid penalties associated with the Carbon Tax Act.

5.2.2. Sensitivity analysis

Sensitivity analysis involves an assessment of the influence of methods, inputs and assumptions on the LCA model to identify the most influential parameters in the model and reduce the parameters required for uncertainty analysis (Björklund, 2002). A scenario analysis was adopted for sensitivity analysis in this study and has been employed by other researchers in the field of gold process LCA (Chen et al., 2018; Elomaa et al., 2020, 2017; Norgate and Haque, 2012; Segura-Salazar et al., 2019). The scenarios considered in the sensitivity analysis and the variations from the base case are summarised in Table 5-9.

Table 5-9: Scenarios concerned for sensitivity analysis for the flowsheets developed

Scenario number	Change implemented
S1	+10% increase in electricity consumption from base case
S2	+10% increase in Lixiviant consumption from base case
S3	+10% increase effluent to wastewater treatment from base case
S4	Electricity grid change to EU-28: Electricity grid
S5	+25% increase in reagent consumption for created datasets

Of the process hotspots that were identified in section 5.2.1, electricity (S1), lixiviant consumption (S2) and wastewater treatment (S3) accounted for the bulk of the contributions to the LCA results. Therefore, it was decided to assess the effect a 10% change from base case values would have on the environmental impact of processes. The results of scenarios S1, S2 and S3 for the cyanide and thiosulphate flowsheet are illustrated in Figure 5-15 and Figure 5-16 respectively. From Figure 5-15 and Figure 5-16 it can be seen that there is a sizeable correlation between electricity consumption and the environmental impact categories. Any approaches to reduce the electricity usage can benefit human health and help improve ecosystem quality areas of protection in both flowsheets. Increasing lixiviant consumption in both flowsheets results in opposing effects on land use and ionising radiation. Increasing effluent water treatment discharge from the cyanide flowsheets results in 29% reduction in freshwater consumption impact category. This is attributed to water treatment allowing freshwater to re-enter the ecosystem.
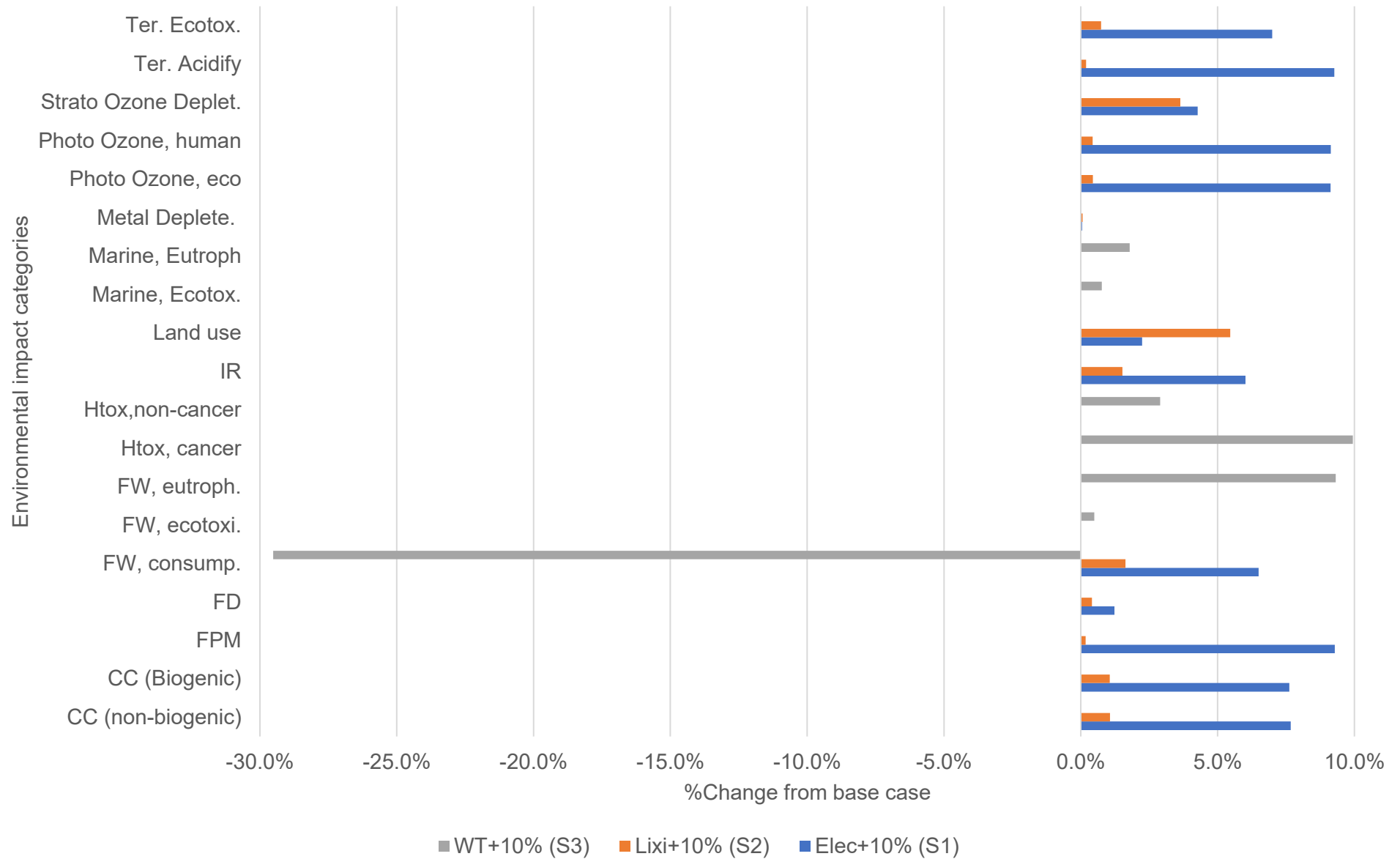
Figure 5-15: Cyanide flowsheet sensitivity analysis (10% change from base case)
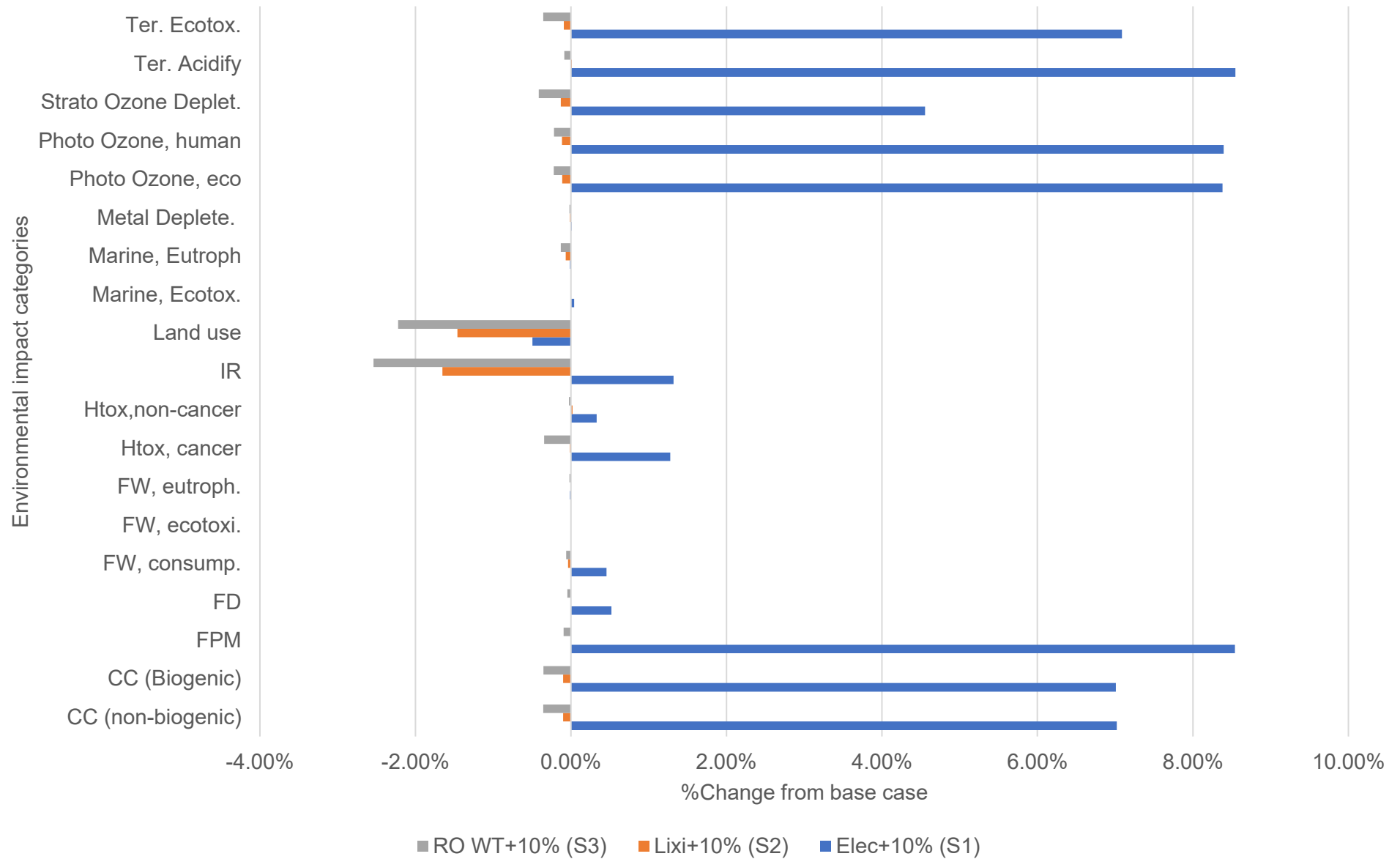
Figure 5-16: Thiosulphate flowsheet sensitivity analysis (+10% change from base case)

Owing to the correlation of the LCA results with electricity consumption, the selection of an electricity grid (S4) less reliant on coal and with more renewable energy was selected to investigate the impact that the electricity grid mix has on the LCA results. Since most of the datasets used in the LCA model were developed based on primary data from the European Union (EU), the "EU-28" electricity grid was selected for comparison with the South African (RSA) electricity grid. As demonstrated in Figure 5-17, there was a reduction in environmental impacts for 10 out of 19 impact categories. The changes may be explained by the combination of electricity generation technologies used in RSA and the EU displayed in Figure 5-18 and Figure 5-19 respectively. The reduction in the 10 impact categories highlighted was attributed to RSA's dependency on fossil fuel coal (90%) resulting in greenhouse gas emissions and environmental pollution. The increase in ionising radiation was brought about by the 25% dependency on nuclear power by the EU versus 5% in RSA. The radionuclide emissions associated with nuclear energy results in an increase between 200% and 300% from the base case in carcinogenic effects on the ecosphere. The land use impacts are inherent with the requirements of the technology and the number of plants in commission within a country. The environmental impacts associated with electricity generation in RSA and EU electricity grids may be circumvented by incorporation of more renewable energy technologies into the electricity grid as discussed in 5.2.1.1.
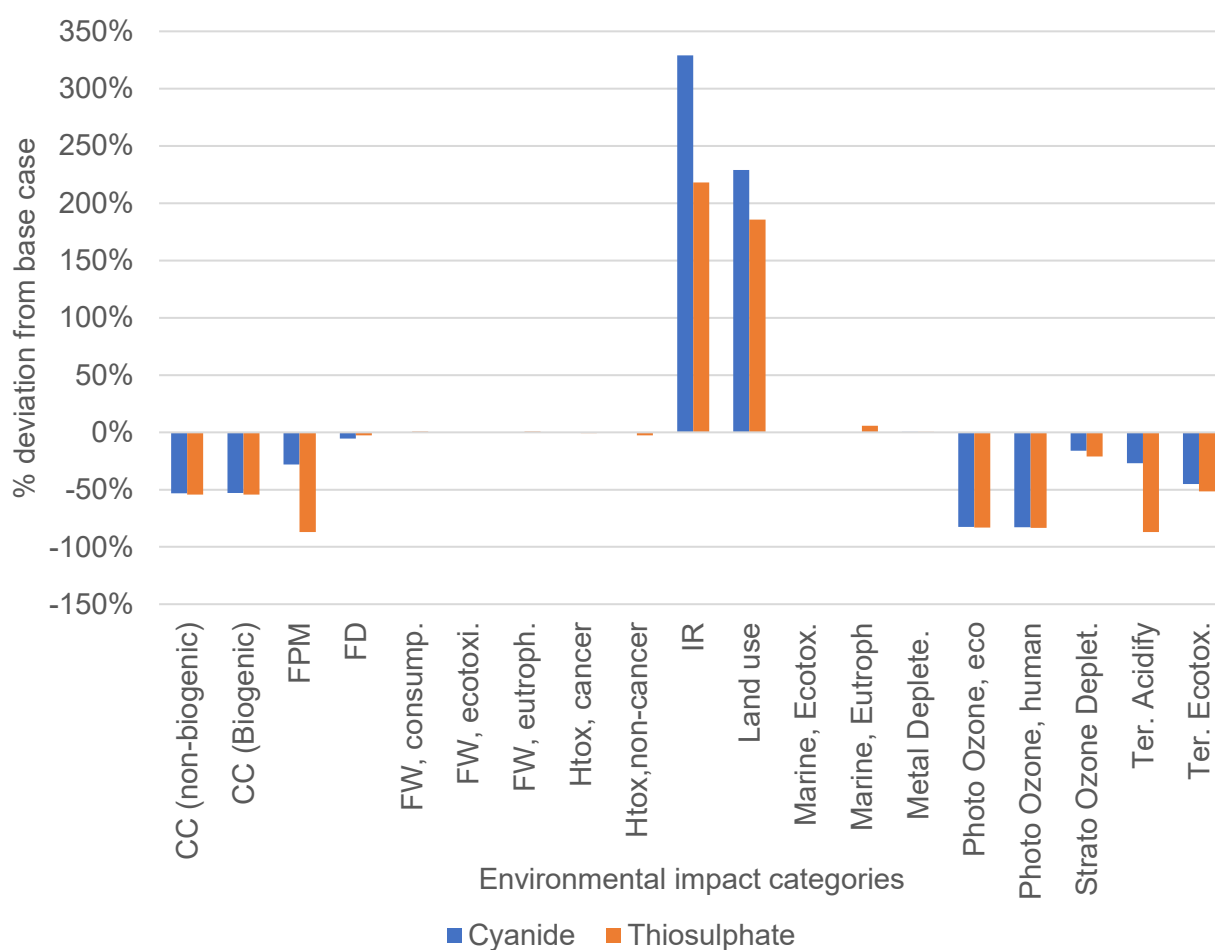
Figure 5-17: Change in environmental impact when EU-28: Electricity grid mix is applied instead of ZA: Electricity grid
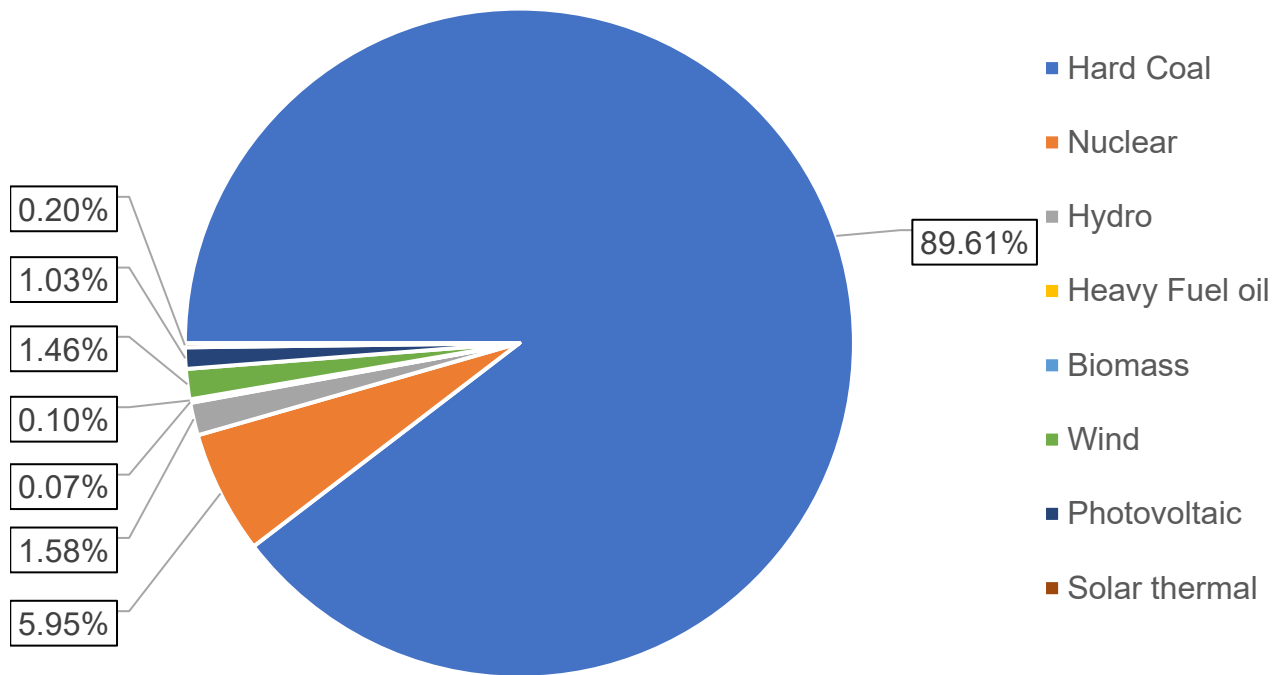
Figure 5-18: Contribution of different energy generation techniques to the South African Electricity grid (Dataset: ZA: Electricity Grid) [Adapted from: (Sphera Solutions GmbH, 2020)]
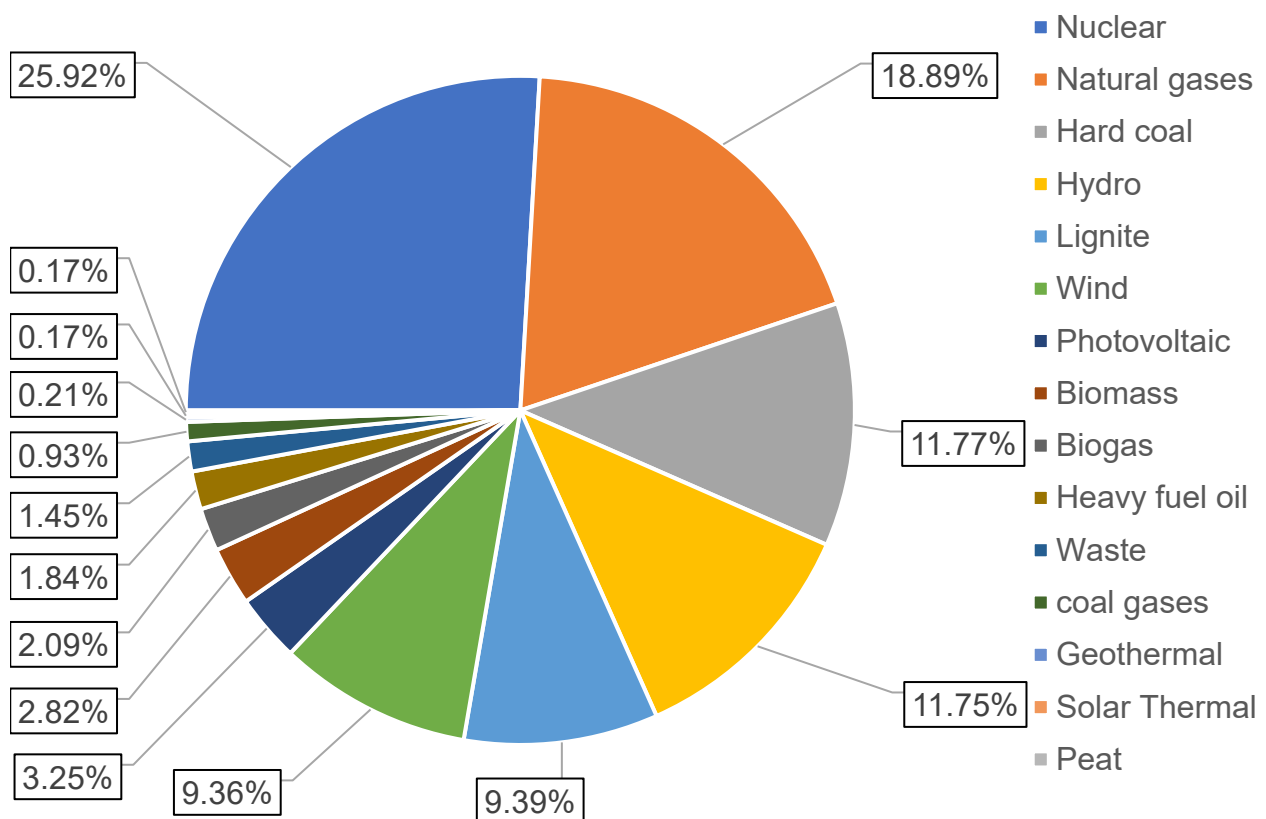
Figure 5-19: Contribution of different energy generation techniques to the 28 countries in the European Union Electricity grid (Dataset: EU-28: Electricity Grid mix) [Adapted from: (Sphera Solutions GmbH, 2020)]

121

Table 5-10: Sensitivity analysis on datasets created for the cyanide flowsheet (25% change from base case)

	Activated carbon	Copper sulphate	Ferrous sulphate (dissolved)	Dodecanol	Hexamethylene diamine (HMDA)	Lead nitrate	Manganese dioxide	Potassium permanganate	Propylene glycol methyl ether acetate	Sodium Mercaptobenzothiazole
CC (non-biogenic)	0.014%	0.001%	0.001%	0.002%	0.027%	0.258%	0.046%	0.014%	0.021%	0.027%
CC (Biogenic)	0.176%	0.001%	0.001%	-0.001%	0.027%	0.266%	0.045%	0.014%	0.021%	0.027%
FPM	0.025%	0.073%	0.002%	0.002%	0.003%	0.104%	0.041%	0.011%	0.003%	0.029%
FD	0.002%	0.001%	0.000%	0.000%	0.008%	0.050%	0.008%	0.003%	0.010%	0.012%
FW, consump.	0.028%	0.002%	0.001%	0.005%	0.028%	25.261%	0.040%	0.014%	0.018%	0.045%
FW, ecotoxi.	0.000%	0.000%	0.000%	0.000%	0.000%	0.000%	0.000%	0.000%	0.000%	0.000%
FW, eutroph.	0.000%	0.000%	0.000%	0.000%	0.000%	0.001%	0.000%	0.000%	0.000%	0.000%
Htox, cancer	0.010%	0.000%	0.000%	0.000%	0.000%	0.001%	0.000%	0.000%	0.000%	0.000%
Htox,non-cancer	0.000%	0.000%	0.000%	0.000%	0.000%	0.003%	0.000%	0.000%	0.000%	0.000%
IR	0.008%	0.005%	0.002%	0.004%	0.007%	0.299%	0.053%	0.032%	0.006%	0.025%
Land use	0.055%	0.003%	0.008%	0.000%	0.023%	0.029%	1.030%	0.277%	0.016%	0.027%
Marine, Ecotox.	0.000%	0.000%	0.000%	0.000%	0.000%	0.001%	0.000%	0.000%	0.000%	0.000%
Marine, Eutroph	18.694%	0.000%	0.000%	0.000%	0.000%	0.001%	0.000%	0.000%	0.000%	0.000%
Metal Deplete.	0.000%	0.000%	0.002%	0.000%	0.000%	0.012%	0.035%	0.010%	0.001%	0.000%
Photo Ozone, eco	0.031%	0.001%	0.001%	0.001%	0.008%	0.185%	0.036%	0.010%	0.007%	0.009%
Photo Ozone, human	0.031%	0.001%	0.001%	0.001%	0.008%	0.184%	0.036%	0.010%	0.007%	0.009%
Strato Ozone Deplet.	0.010%	0.002%	0.001%	0.031%	0.204%	3.130%	0.025%	0.008%	0.016%	0.084%
Ter. Acidify	0.021%	0.077%	0.002%	0.001%	0.002%	0.103%	0.041%	0.011%	0.003%	0.030%
Ter. Ecotox.	0.048%	0.003%	0.001%	0.004%	0.011%	1.091%	0.065%	0.018%	0.013%	0.023%

Table 5-11: Sensitivity analysis on datasets created for the thiosulphate flowsheet (25% change from base case)

	Copper sulphate	Iron oxide	Do-decanol	Hexame-thylene diamine	Manganese dioxide	Propy-lene glycol methyl ether acetate	Sodium Mercapto-benzothi azole	Sodium sulphide	Sodium sulphite	Potas-sium perman-ganate
CC (non-biogenic)	0.005%	0.212%	0.005%	0.572%	0.160%	0.075%	0.094%	0.243%	0.026%	0.032%
CC (Biogenic)	0.005%	0.212%	-0.002%	0.572%	0.160%	0.075%	0.094%	0.243%	0.026%	0.032%
FPM	0.236%	0.162%	0.008%	0.051%	0.140%	0.009%	0.097%	0.129%	0.007%	0.025%
FD	0.002%	0.034%	0.000%	0.079%	0.014%	0.016%	0.021%	0.026%	0.004%	0.003%
FW, consump.	0.000%	0.334%	0.002%	0.048%	0.012%	0.005%	0.013%	0.005%	0.045%	0.003%
FW, ecotoxi.	0.000%	0.002%	0.000%	0.001%	0.000%	0.000%	0.000%	0.000%	0.000%	0.000%
FW, eutroph.	0.000%	0.088%	0.002%	0.007%	0.001%	0.002%	0.003%	0.003%	0.011%	0.000%
Htox, cancer	0.009%	0.777%	0.146%	0.430%	0.050%	0.080%	0.107%	0.093%	0.096%	0.011%
Htox,non-cancer	0.002%	0.045%	0.003%	0.073%	0.011%	0.014%	0.021%	0.024%	0.005%	0.002%
IR	0.010%	1.322%	0.009%	0.093%	0.124%	0.014%	0.059%	0.103%	0.175%	0.050%
Land use	0.008%	0.869%	0.000%	0.394%	2.902%	0.044%	0.077%	0.242%	0.066%	0.516%
Marine, Ecotox.	0.000%	0.010%	0.000%	0.008%	0.002%	0.002%	0.003%	0.003%	0.001%	0.000%
Marine, Eutroph	0.001%	0.043%	0.001%	0.089%	0.007%	0.008%	0.020%	0.015%	0.004%	0.002%
Metal Deplete.	0.000%	0.076%	0.000%	0.006%	0.073%	0.001%	0.001%	0.001%	0.000%	0.014%
Photo Ozone, eco	0.002%	0.113%	0.004%	0.174%	0.123%	0.025%	0.031%	0.112%	0.011%	0.023%
Photo Ozone, human	0.002%	0.112%	0.004%	0.163%	0.123%	0.023%	0.030%	0.111%	0.011%	0.023%
Strato Ozone Deplet.	0.008%	0.332%	0.131%	5.210%	0.106%	0.069%	0.358%	0.132%	0.041%	0.023%
Ter. Acidify	0.250%	0.166%	0.002%	0.051%	0.140%	0.009%	0.102%	0.127%	0.007%	0.025%
Ter. Ecotox.	0.010%	0.457%	0.014%	0.253%	0.252%	0.052%	0.088%	0.107%	0.059%	0.046%

Data shortages were encountered while conducting the LCA. Specifically, the background process datasets for some raw materials used in the flowsheets were not part of the GaBi® professional database and were created as discussed in section 3.4.4. The influence of the created datasets on the LCIA results need to be investigated. A 25% change to the base case raw material input value for both flowsheets were made and the influence on environmental impact categories are depicted in Table 5-10 and Table 5-11 for the cyanide and thiosulphate flowsheets respectively.

For the cyanide flowsheet, freshwater consumption for lead nitrate and marine eutrophication associated with activated carbon production were sensitive to raw material input changes. The accuracy of the LCA can be improved by verifying the results reported by the created datasets with purchased datasets developed by GaBi®. For the remaining datasets, the impact categories that are affected by greater than 0.5% when a 25% change is made from the base case are highlighted in red in Table 5-10 and Table 5-11. The LCIA results were therefore not greatly impacted by the assumptions made when creating additional datasets.

5.3. Life cycle assessment as a design tool

LCA facilitates quantifying upstream and downstream environmental impacts of process flowsheets. Comparisons of environmental impacts of process flowsheets and unit processes can be completed, with the provision that the system boundaries and functional unit selections are identical. Comparative LCA's that conduct hotspot analysis on background processes, unit process and effluent streams where a baseline process is used for comparison are particularly insightful. Strategies can be developed to address hotspots by developing processes to remove or render harmful species benign, selection of raw materials and minimisation of energy usage.

The methodology employed provided design criteria to assess and identify opportunities to reduce the life cycle environmental impact associated with gold recovery technologies. Life cycle thinking is important since most environmental impact assessments (EIA's) focus on the gate-to-gate and local ecological impacts of anthropogenic activities. The ecosphere is, however, an interconnected global system. Tools that assess the global (i.e. upstream and downstream) ecological impacts of specific process technologies within commercialised processes offers a big-picture perspective of the ecological consequences of engineering designs (Morrison-Saunders and Pope, 2013; Pope et al., 2004; Tukker, 2000). This therefore provides an objective platform for the assessment of "green"-technologies proposed in literature against conventional technologies employed in industry.

There are examples of LCA being used as a design tool for gold manufacture from primary ores in literature. Elomaa et al. (2019, 2017) used LCA as a design tool to investigate environmental footprint associated with different ore oxidation and leaching unit processes for gold recovery. Other studies have used LCA to provide quantitative insight on the environmental impacts of energy policy, determine the impact of ore resources (e.g. refractory versus non refractory) and the impact of unit

process selection for gold recovery (Chen et al., 2018; Li et al., 2014; Norgate and Haque, 2012). Considering these few examples along with the EIA review conducted in section 2.5, there is support for LCA within the academic- and metallurgical- community as a holistic EIA. In a globalised world during the Anthropocene, evaluations of how decisions affect the global ecosystem are necessary to support the ecosystem in regulating the effects of anthropogenic activities (Birich et al., 2019).

Background process environmental impacts account for 99% of environmental indicators calculated (Steubing et al., 2016). This may present a bias towards selection of raw materials, minimisation of energy usage and the selection of energy resources to reduce the overall life cycle impact of a process. This finding is relevant considering that local communities and environments are affected by the impact of direct emissions that make small contributions to the overall LCA results. Therefore, supplementing LCA results with local environmental regulations in addition to conducting an analysis on emissions from foreground processes is necessary.

Data availability is a key challenge in conducting LCA's as a design tool. The LCA practitioner needs to construct background process datasets for raw material manufacture from literature information. The link to missing background process datasets is complete when the practitioner has moved up the raw material supply chain to meet up with background process datasets that are present in the LCA software databases. Effluent stream to ecosphere hotspot analysis was particularly challenging if the effluent species are not present in the LCA database available to practitioners. For effluent species not available in the database, one course of action is to use a decomposition product or a species belonging to the same chemical family as a substitute.

The environmental impact of effluent species related to the mining industry such as thiosulphate, arsenic trioxide, Prussian blue, scorodite, sulphides and sulphur do not have characterisation factors in the ReCiPe ® 2016 database. The full scope of environmental impacts is not completely accounted for with regards to acid mine drainage (AMD) which releases sulphuric acid to the ecosphere. This challenge can be overcome by modelling a process for AMD within the system boundary of the LCA study. There is consensus among authors that an LCIA methodology for the mineral processing industry is required (Awuah-Offei and Adekpedjou, 2011; Segura-Salazar et al., 2019). Some researchers in the mining industry have even developed their own methodologies such as "LICYMIN^{TM}" by Durucan et al. (2006) to address impacts of processes related to the metallurgical industry.

The aforementioned approaches introduce a margin for error in the results of the study because the development of LCA databases for process and chemical species datasets are conducted by large research institutions who have access to more resources and information than an individual practitioner. Examples of this is the work done by Huijbregts et al.(2016) and Steubing et al.(2016) to develop the ReCiPe® and Eco-invent® LCIA methodologies. The field of LCA is addressing the

shortcomings of databases to include more species and update the libraries as human technology overcomes environmental issues (Ayres, 1995; Freidberg, 2018; Huijbregts et al., 2017; Steubing et al., 2016).

A techno-economic assessment is required to supplement the LCA results presented in this study. This work has also highlighted opportunities to reduce environmental impacts but are unfavourable towards product recovery. Examples of this include the recommendation of a stoichiometric excess of zinc and reagents such as lead nitrate to increase gold recovery or the use of dodecanol and ammonia for uranium recovery. The economic benefit needs to be compared with the environmental consequences along with the cost of additional unit processes required to break down toxic compounds before release to the ecosphere. This type of analysis would enable an evaluation of the gold cut-off grade required to ensure the profitability of the unit processes proposed while minimising the environmental consequences. This is where life cycle sustainability analysis (LSCA) provides a framework to assess economic, environmental and social aspects associated with a product that can be used by the metallurgical industry for sustainable mining and resource management while incorporating the principles of life cycle thinking (Fauzi et al., 2019; Gorman and Dzombak, 2018; Guinée, 2015; Heiskanen, 2002; Petit-Boix et al., 2017).

5.4. Project alignment with UN Sustainable development goals

The work conducted in this project supports the United Nations Sustainable Development Goals (UN SDG) by presenting solutions that reduce the extent to which mining operations affect the environment and communities (UN SDG 12). Mine tailings have not been rendered inert, but the potential for acid mine drainage has been reduced by removing sulphide minerals and reduction of uranium as a source of radionuclides. This supports biodiversity by reducing impact for aquatic, marine and terrestrial ecosystems by reducing the impact of acid mine drainage and carcinogenic effects of caused by radionuclide pollution and arsenic (UN SDG 14 and 15). UN SDG's 14 and 15 are further supported but employing green chemistry and green technology solutions that reduce the life cycle impact of mining technology on the ecosphere. Ideally the goal should be to strive towards a reality where the ecosystem is restored to the same or better condition after mine closure. The climate action goal (UN SDG 13) was addressed by identifying gold recovery processes that reduce greenhouse gas emissions over the cradle-to-gate life cycle and opportunities were highlighted to further reduce the burden of climate change. Consulting recent literature on gold mining activities to support the development of processes for treating mining solid waste supported the industry, innovation and infrastructure goal (UN SDG 9) together with sustainable cities and communities' goal (UN SDG 11). The aforementioned goals were addressed by identifying process technologies that reduced environmental footprint of mining activities while identifying mineral revenue streams. Furthermore, valorising a waste stream for value in light of the declining gold industry in South Africa helps to ensure the longevity of the gold industry while sustaining employment opportunities in the

gold sector and thereby supporting the decent work and economic growth goal (UN SDG 8). This is especially relevant in the context of the South African gold mining industry where gold from primary ore mines are approaching the gold cut-off grades to be able to operate profitable businesses (United Nations, 2019). By addressing these UN SDG's, this work has benefited the mining industry presenting solutions to promote environmental and societal well-being while generating revenue and thereby a more sustainable industry (Rockström et al., 2009).

6 CONCLUSIONS AND RECOMMENDATIONS

The aim of this study was to identify processes for the recovery of value and reduction of environmental impacts of gold mine tailings in the West Rand, Witwatersrand region. The significance of this study was twofold. Firstly, to reduce the environmental impacts related to stockpiled mine tailings which is the largest single source of pollution in South Africa. Secondly to identify revenue streams in mine tailings to support reducing environmental impact of mine tailings in light of the declining head grade of primary gold ore resources. The objectives of this study listed in section 1.4 were achieved and key findings for each objective will be summarised in this chapter.

6.1. Conclusions

Mineralogical and geochemical data was compiled for tailings dams within the West Rand, Witwatersrand region to create a hypothetical tailings dam scenario thereby completing the first objective in this study. The heavy metal concentrations in hypothetical tailings were not compliant with regulation for protection of water resources according to National Environmental Management: Waste act 2008 (Act No. 59 of 2008) (NEMA)(Department of Environmental Affairs, 2014). For the mine tailings ore resource considered the gold grade was 0.28 g/tonne of which 58% gold was free milling while the rest of gold is captured in refractory minerals.

The second objective involved identifying process flowsheets for recovering value and reducing environmental impacts and was achieved. Two process flowsheets were developed for gold recovery from mine tailings: a conventional gold recovery flowsheet employing cyanide leaching and an alternative process employing thiosulphate leaching. Thiosulphate was identified as an environmentally friendly replacement to cyanide that has demonstrated evidence that the technology is mature enough for commercial gold recovery. Cost-effective technologies for liberation of 11.8% gold occluded in quartz minerals could not be identified in literature.

Following mass and energy balances in HSC Sim®, both flowsheets proposed reported a gold recovery of 71.2% with the depleted solid emissions reporting a gold grade (dry basis) of 0.02 g/t and 0.03 g/t for thiosulphate and cyanide flowsheet respectively. The residual gold was occluded within quartz minerals smaller than P_{80} of 106 μm. Sulphide and uranium concentrations in the depleted tailings were reduced from 0.18% to 0.03% and from 54.9 g/t to 13.0 g/t respectively by the flowsheets proposed. Cyanide and thiosulphate solid emissions were not compliant with NEMA therefore mine tailings was not rendered inert for protection of water resources. Cyanide flowsheet solid emissions were non-compliant for lead, manganese and copper while thiosulphate flowsheet solids emissions were non-compliant for manganese and copper. The solid emissions from flowsheets may, however, be used in regions zoned as industrial according to NEMA.

128

A cradle-to-gate life cycle assessment of the proposed flowsheets was conducted to quantify the life cycle environmental impacts of flowsheets and identify the process that yielded the lowest environmental impact to complete the third and fourth objective of this study. The thiosulphate flowsheet had a reduced life cycle impact for 18 out of 19 midpoint impact categories when compared to the cyanide flowsheet apart from freshwater consumption impact category. The freshwater consumption was lower in the cyanide flowsheet because a pulp density of 50% was required for gold leaching in the cyanide flowsheet compared to 30% in the thiosulphate flowsheet and the wastewater treatment models selected in GaBi® were for municipal wastewater treatment and not mining wastewater treatment. At endpoint level, the thiosulphate flowsheet had reduced ecological impact on the human health and resource scarcity endpoint impact categories while the ecosystem quality was equal in both flowsheets because of dodecanol (an additive used during uranium solvent extraction) emissions to ecosphere.

Hotspot analysis on LCA results allowed the contributional analysis and prioritisation of contributions to environmental impact categories of background processes, unit process and emissions to ecosphere. For both process flowsheets, backgrounded processes accounted for the bulk of life cycle environmental impacts specifically electricity production, lixiviant manufacture, activated sludge generation during wastewater treatment and depleted tailings emissions to the ecosphere. Cyanide destruction and arsenic removal unit processes account for the bulk of the life cycle impacts for cyanide flowsheet because of the large water inventory introduced by recommending 12.5% by mass aqueous sodium hypochlorite solution as reagent for cyanide destruction. The dilute sodium hypochlorite solution increased water inventory and consequently increased agitation electricity consumption by 3.65 times compared to thiosulphate flowsheet. Leaching of flotation tailings increased gold recovery from 44.5% to 71.2% but increases the life cycle impacts of both flowsheets owing to increased reagent and electricity consumption to maintain optimum gold leaching conditions to compensate for the large mass flow rate of solids originating from flotation tailings stream. Concerning sulphide roasting technologies, pressure oxidation yielded no harmful emissions from foreground process to ecosphere compared to sulphide roasting. Technologies for SO_2 gas and arsenic trioxide sequestration from sulphide roasting off-gases only achieve 99% removal of these hazardous emissions and therefore the consequences of these emissions have the potential to accumulate in the environment over time.

The thiosulphate flowsheet resulted in reduced impact to human health over the entire life cycle by two orders of magnitude when compared to cyanide flowsheet. This discovery was attributed to activated sludge discharge from wastewater treatment along with zinc and lead nitrate emissions to ecosphere that was recommended to increase gold recovery during zinc cementation which culminated in an increase in zinc and lead emissions to ecosphere. The thiosulphate flowsheet offers

a solution to reduce the human health impacts associated with gold recovery from mine tailings when compared to the conventional cyanide flowsheet. This finding is relevant owing to the burdens that radionuclides and heavy metals present in Witwatersrand tailings dams have posed to human health as reported in literature (Kamunda et al., 2016a; Ngole-Jeme and Fantke, 2017; Tutu et al., 2008).

The proposed flowsheets benefited the aquatic and terrestrial ecosystems by removing sulphide mineral and uranium and therefore reduced the impact of acid mine drainage and radionuclide pollution by 83% and 75% respectively. A drawback for both processes was activated sludge release from wastewater treatment along with the emissions of ion exchange resins, ammonia and organic phase used during uranium recovery that negatively affected aquatic ecosystems. The thiosulphate flowsheet has a 46%, 74% and 80% reduction of burdens towards freshwater-, marine- and terrestrial-toxicity impacted categories compared to the cyanide flowsheet. The thiosulphate flowsheet also reduced the impact of eutrophication by one order of magnitude compared to the cyanide flowsheet. The cyanide flowsheets failure was attributed to the increased environmental burden of activated sludge released from wastewater treatment, activated carbon production, zinc powder manufacture together with the environmental consequences of 3.7 times larger electricity consumption compared to thiosulphate flowsheet. The thiosulphate flowsheet has greater potential to restore the ecosystem to its original state compared to cyanide flowsheet. This supports the mining industry in reducing externalised cost for environmental rehabilitation during mine closure.

The thiosulphate flowsheet yielded at least two times less influence on environmental impact categories that represent the atmosphere compared to the cyanide flowsheet. Specifically, these impact categories were: fine particulate matter, ionising radiation, terrestrial acidification, photochemical ozone formation and stratospheric ozone depletion. This finding was attributed to electricity consumption for cyanide flowsheet being 3.7 times larger than the thiosulphate flowsheet because of cyanide destruction and arsenic removal unit processes. This discovery is essential since air quality is linked to human and ecosystem health and ensuring that effects of terrestrial acidification does not render arable land incapable of supporting plant life. Furthermore, the impact on climate change by the thiosulphate flowsheet was three times less when compared to the cyanide flowsheet because of the reduced electricity consumption. Considering that the South African government has imposed a Carbon tax, mine houses may need to look towards solutions to reduce greenhouse gas emissions to avoid penalties which may affect the profitability of their operations.

Based on the conclusions drawn from the life cycle assessment on key areas of protection, the thiosulphate flowsheet was the more environmentally sustainable process for gold recovery from mine tailings. A supplementary finding that evolved from achieving the third objective, was that LCA was an effective design tool for comparative assessments on environmental impacts of process flowsheets provided that the system boundaries and functional units are identical and the LCIA

methodology accurately accounts for background processes and emissions. National environmental regulation for the case study region should be consulted during hotspot analysis for emissions to ecosphere during the LCA to determine if highlighted emissions require intervention or not. The present study has revealed that there are database gaps for background processes, raw materials and emissions specific to the metallurgical industry and were documented in section 3.4.4. LCA can effectively account for environmental efficiencies of elementary flows at the system boundary but the process and resource usage efficiency of each process inside the system boundary was not accounted for.

6.2. Recommendations

The fifth objective of identifying improvements to further reduce environmental impacts in both flowsheets was realised. The recommendations listed in this section were not incorporated into the proposed flowsheets. The recommendations are, however, considerations for future work that holds similar aims and objectives as the present study. The environmental impacts of background processes such as the upstream electricity production and lixiviant manufacture may be achieved by implementing green technologies to reduce environmental impacts such as the use of renewable energy electricity production, manufacturers limiting emissions to environment and valorising waste streams to produce inert by-products. The ecological impacts of activated sludge from wastewater treatment can be minimised by repurposing activated sludge as fertiliser for agriculture and energy crops.

Research into improving flotation technology for mine tailings to increase sulphur, uranium and gold recoveries above 85%, 46% and 20% respectively is needed to improve gold recovery, reduce lixiviant consumption and reduce impact of acid mine drainage and radionuclide pollution. Furthermore, investigations into low-cost technologies as an alternative to ultrafine grinding for liberating fine-grained gold within quartz minerals (P_{80}<106 µm) is necessary to increase gold recovery. The reagents used create species that increase the environmental impact of solid emissions from flowsheets. A cost-benefit analysis of these reagents needs to be concluded to understand if the environmental impact of using these reagents have a significant improvement on product recovery to justify their inherent environmental impact.

An investigation into reducing the environmental impacts of the organic phase for uranium recovery, ion exchange resins and ammonia needs to be concluded. An opportunity was identified for ammonia sequestration through struvite chemical precipitation, while the organic phase of uranium recovery and non-viable ion exchange resin bleed streams may be passed through high-performance activated carbon adsorption followed by incineration. The environmental impacts of using sodium

chloride for yellow-cake uranium precipitation as opposed to ammonia needs to be investigated to identify if this is an opportunity for environmental impact reduction.

The life cycle impact of alternative cyanide- destruction and -recovery technologies need to be investigated to determine if there is potential to further reduce environmental impact of cyanide flowsheet. Examples of these technologies include biological treatment, SO_2/air or H_2O_2 cyanide destruction technologies and acidification, volatilisation and re-neutralisation (AVR) cyanide recovery technology. An evaluation of the environmental impacts of electrowinning compared to zinc cementation for cyanide flowsheet gold recovery needs to be conducted. This recommendation stems from the human and ecosystem health concerns related to the zinc cementation unit process. This is a design trade-off where the purity of crude gold bullion produced must be weighed against the potential environmental benefit. Alternatively approaches to recycle zinc and lead nitrate in gold recovery processes should be investigated to identify opportunities to mitigate the environmental impact associated with zinc cementation.

The solid emissions from the process flowsheets proposed were non-compliant with NEMA. Therefore, subsequent processing, such as phytoremediation, is required to remove pollutants from tailings to render the tailings environmentally benign. Phytoremediation of depleted mine tailings has been demonstrated commercially to reduce pollutants in mine tailings and generate bio-ore for precious and base metal recovery. Identifying opportunities for profitable metal recovery enables remediation activities to fund themselves and therefore strives towards the goal of economic and environmental sustainability of the mineral processing industry (Chaney and Baklanov, 2017; Nkrumah et al., 2018; Sheoran et al., 2009).

Incomplete gold deportment studies were found in literature for mine tailings dams in South Africa to support accurate process flowsheet development. Specifically, a thiosulphate diagnostic leaching step should be included to test the assumption used in this work that thiosulphate may be used as a direct replacement for cyanide and achieve the same gold recoveries. This enables an empirical understanding of the association of gold with minerals and challenges to gold recovery for a specific ore resource. It was forecasted that the gold grade of the secondary ore resource of mine tailings will exceed primary ore gold grades by approximately 2031. Therefore, comprehensive gold deportment data will support research into process development to support the longevity of the gold industry in South Africa and quantify the economic value in mine tailings.

Shortcomings in the methodology used was identified as presented in section 6.1. Literature has noted the need for the development of an LCIA methodology specific to the metallurgical industry to account for environmental aspects and impacts of processes and chemical species related to the mining operations (Awuah-Offei and Adekpedjou, 2011; Segura-Salazar et al., 2019). Accuracy and

reliability of the LCA results in this study can be improved by developing background datasets and emissions fate models based on primary data together with conducting uncertainty analysis on inputs. Exergetic life cycle assessment (ELCA) allows the evaluation of thermodynamic efficiency of the individual unit processes and the overall flowsheet offering insight into the consumption and depletion of natural resources. The addition of ELCA will therefore further support reducing life cycle environmental impacts by optimising resource utilisation efficiency in processes (Amini et al., 2007; Cornelissen and Hirs, 2002; Gößling-Reisemann, 2008).

Exploring the environmental impacts of gold recovery flowsheets is beneficial. An analysis of the economic feasibility of the proposed flowsheets is, however, an important criterion for the design of recovery of value from low gold grade mine tailings processes. The thiosulphate flowsheet includes costly unit operations such as pressure oxidation, resin adsorption and elution, thiosulphate regeneration and reverse osmosis water treatment. A supplementary study needs to be conducted to determine the economic viability of the process flowsheets identified for gold recovery from West Rand mine tailings. Environmental sustainability was the focus of this work. But the economic and social aspects related to this case study could be investigated in a future study through life cycle cost analysis and social life cycle assessment respectively. This will aid in ensuring that a process is developed that addresses the core elements of a sustainable process and aid in developing a framework for life cycle sustainability assessment which is a continuation of the work done within the field of life cycle assessment.

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APPENDICES

Appendix A   XRF Data

Table 0-1:Collation of XRF data from literature sources for Witwatersrand region

Reference	(Abegunde et al., 2016)	(Malatse and Ndlovu, 2015)
GMaps ID	(G-14)	N/A
GPS Location	26°07' 49.1"S 27°45' 51.7"E	N/A
GPS Location (DMS)	-26.130 297, 27.764 361	
GPS Location (DD)		
Units	mg/kg	mg/kg
Element		
SiO_2	84,34	77,7
Al_2O_3	6,45	10,2
Fe_2O_3	3,03	4,51
CaO	0,52	1,93
MgO	0,4	1,79
Na_2O	0,34	0,61
K_2O	0,24	1,19
MnO	0,09	0,05
TiO_2	0,06	0,47
P_2O_5	0,04	-
Cr_2O_3	0,03	0,45
SO_3	-	0,91
P_2O_3	-	0,09
ZrO_2	-	0,03
NiO	-	0,02
SrO	-	0,02
As_2O_3	-	0,01
ZnO	-	0,01
CuO	-	0,01
U_3O_8	-	0,01
Co_2O_3	-	0,01
Pb_2O	-	0,00

Appendix B   XRD Data

Table 0-2: Collation of XRD Data from literature references for Witwatersrand region, South Africa.

Reference	(Rudzani et al., 2017)		(Nengovhela et al., 2007)					(Fashola et al., 2016)		
Place	**Nestor**	**Glynn Lydenburg**	**Brakpan**	**Knights**	**Edenvale**	**Roodepoort**	**Klerksdorp**	**Krugersdorp**	**Krugersdorp**	**Krugersdorp**
GMap tag	**(G-7)**	**(G-7)**	**(G-8)**	**(G-9)**	**(G-10)**	**(G-11)**	**(G-12)**	**(G-26)**	**(G-27)**	**(G-28)**
								26º 7' 56" S and 27º 48' 13"E	26º 8' 34" S and 27º 48' 28" E	26º 7' 40" S and 27º 48' 57"E
GPS Location	-24.91444, 30.74036	-24.91444, 30.74036	-26.34493, 28.31827	-26.20112, 28.18184	-26.14136, 28.16094	-26.0615, 27.85727	-26.90477, 26.69444	-26.131610, 27.794888	-26.142778, 27.807778	-26.127782, 27.818273
Jarosite [$KFe^{3+}_3(OH)_6(SO_4)_2$]	4,07	62,4	1,00	1,33	1,00	1,00	1,50	-	-	-
Calcite [$CaCO_3$]	0,44	19,8	-	-	-	-	-	-	-	-
Kaolinite/Chlorite [$Al_2Si_2O_5(OH)_4$]	0,99	11,8	-	-	-	-	-	5	-	1
Mica [$KAl_3Si_3O_{10}(OH)_2$]	9,12	2,20	9,50	9,75	6,33	5,33	5,50	13	3	3
Gypsum [$CaSO_4$]	1,76	2,20	0,25	-	-	-	-	< 1	-	-
Gibbsite [$Al(OH)_3$]	0,44	2,11	-	-	-	-	-	-	-	-
Goethite	0,44	1,83	-	-	-	-	-	-	-	-
Quartz [SiO_2]	70,2	0,37	80,5	70,3	69,3	64,7	71,0	27	84	70
Hematite[$CaMg(CO_3)_2$] / Geothite	3,63	0,37	-	-	-	-	-	-	-	-
Pyrite [FeS_2]	3,52	0,37	0,50	1,00	1,00	-	1,50	-	-	-
Dolomite [$CaMg(CO_3)_2$]	3,30	0,37	-	-	-	-	-	-	-	-
Plagioclase [$(Na,Ca)(Si,Al)_4O_8$]	3,19	0,37	-	-	-	-	-	-	-	-
K-feldspar/ Ruthile [$KAlSi_3O_8$]/[TiO_2]	0,55	0,28	0,75	1,00	1,00	1,00	-	-	-	1
Chloritoid [$(Fe,Mg,Mn)_2Al_4Si_2O_{10}(OH)_4$]	-	-	-	9,50	15,0	19,7	2,25	-	-	-
Chlorite [ClO_2]	-	-	7,25	8,25	6,67	5,00	3,00	-	-	-
Pyrophyllite [$Al_2Si_4O_{10}(OH)_2$]	-	-	-	-	-	5,67	16,5	55	12	26
Clay	-	-	0,00	-	1,00	-	-	-	-	-

Appendix C   ICP-MS Data

Table 0-3:Collated table of ICP-MS Data from literature references for Witwatersrand region, South Africa.

Reference	(Mphinyane, 2018)						(Tutu et al., 2009)	(Rudzani et al., 2017)		(Maseki, 2017)				(Kamunda et al., 2016b)						
Gmap Code	(G-6)	(G-3)	(G-5)	(G-4)	(G-1)	(G-2)	(G-13)	(G-7)	(G-7)	(G-15)	(G-16)	(G-17)	(G-18)	(G-19)	(G-20)	(G-21)	(G-22)	(G-23)	(G-24)	(G-25)
Units	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	ppm	ppm	ppm	ppm	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
Elements																				
Be	0.090064	0.461396	0,4949	0,0639	0,3098	0,1214	-	-	-	-	-	-	-	-	-	-	-	-	-	-
B	-	0.728	7,558	-	3,12	2,314	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Na	59.28	47.8972	722,2	292,5	50,74	180,1	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Mg	1332.24	706.16	>15500	15350	4502	410	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Al	4284.8	3759.08	13880	8338	8071	3231	-	-	-	-	-	-	-	-	-	-	-	-	-	-
P	44.0232	147.16	140,6	144,2	143,4	72,37	-	-	-	-	-	-	-	-	-	-	-	-	-	-
K	251.732	858	5725	2774	1746	1387	-	-	-	19700	21000	10670	2200	-	-	-	-	-	-	-
Ca	224.952	17472	41580	7109	340,2	2485	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ti	30.2848	224.12	879,6	468,4	41,32	41,9	-	-	-	-	-	-	-	-	-	-	-	-	-	-
V	5.8656	5.9124	77,3	35,1	16,67	7,218	-	140	82	-	-	-	-	-	-	-	-	-	-	-
Cr	45.2712	30.4616	364,3	517,3	97,95	38,79	-	172	140	410	550	230	120	441.52	270.76	77.50	104.17	97.50	861.67	98.18
Mn	65.728	4880.2	1082	120,9	190,5	96,06	-	-	-	205	220	325	160	-	-	-	-	-	-	-
Fe	8517.6	11055.2	48470	24870	20170	11380	-	-	-	40500	42200	44800	21300	-	-	-	-	-	-	-
Co	2.16112	44.1064	32,14	47,04	20,87	14,23	-	-	-	-	-	-	-	33.68	31.76	30.00	21.67	21.67	28.33	11.82
Ni	9.5108	75.192	292,4	438,9	59,61	28,02	-	53	66	106	60	76	16	131.04	115.08	152.50	99.83	125.83	68.33	91.82
Cu	18.5692	63.596	60,97	43,98	34,59	31,68	-	154	30	-	-	-	-	46.78	45.48	46.25	55.83	47.50	36.67	19.09
Zn	14.0712	216.268	104,1	41,19	132,9	39,58	-	73	150	40	30	70	30	46.15	51.95	82.50	60.00	48.33	48.33	21.82
As	40.534	106.86	243,4	2171	14,23	84,22	-	599	1471	150	140	90	110	94.17	115.19	71.33	73.18	67.08	65.17	69.69
Se	0.466388	5.538	0,5924	0,3672	0,6034	0,8845	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Rb	1.29376	7.9716	37,83	22,62	10,71	8,029	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Sr	3.3384	19.76	220,7	20,83	2,939	14,77	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Mo	0.369928	4.43768	0,9741	0,5326	0,8115	2,894	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Pd	0.059904	3.4008	0,5149	0,0805	0,1507	0,2008	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ag	1.29272	10.4676	5,148	0,6655	1,935	3,342	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Cd	0.01469	4.26712	0,0975	0,0499	1,01	0,2458	-	-	-	0,1	0,08	1,52	5,67	0.05	0.05	0.05	0.04	0.05	0.05	0.05
Sb	0.479284	0.65052	0,6466	85,36	0,8245	1,725	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Ba	12.1576	152.568	346,7	113,5	18,19	36,64	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Pt	0.00129688	0.0121628	-	-	0,0038	0,0022	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Au	0.224536	0.26026	0,2379	0,3785	0,0963	0,3115	-	-	-	0,52	0,72	3,91	1,3	-	-	-	-	-	-	-
Hg	1.1648	0.255944	1,333	0,6952	0,1419	0,6461	-	-	-	-	-	-	-	0.13	0.13	0.06	0.07	0.06	0.10	0.06
Tl	0.0168012	0.276068	0,4743	0,2349	0,1233	0,1324	-	-	-	-	-	-	-	-	-	-	-	-	-	2
Pb	9.3496	180.024	68,21	42,01	16,05	47,65	-	54	37	16	11	27	24	8.85	10.22	2.31	2.96	3.31	1.58	4.32
Bi	0.51116	7.6596	0,7085	0,2971	0,3077	1,945	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Th	2.45596	163.124	0,7322	1,012	9,646	28,13	20,2	-	-	-	-	-	-	-	-	-	-	-	-	-
U	1.495	104.312	0,534	0,2336	14,41	40,76	46,5	-	-	8	7	16	16	-	-	-	-	-	-	-

Appendix D   List of tailings dam literature for Witwatersrand region

Table 0-4: Summary of type of data available for tailings dams in literature

No	Tailings dam (sample depth)	XRD	ICP-MS/ OES	XRF	Reference
G-1	New Machavie- Black Reef (2.4 m)		x		(Mphinyane, 2018)
G-2	New Machavie – Shale (2.4 m)		x		(Mphinyane, 2018)
G-3	Dominion Reef (2.4 m)		x		(Mphinyane, 2018)
G-4	Klein Letaba (2.4 m)		x		(Mphinyane, 2018)
G-5	Louise Moore (2.4 m)		x		(Mphinyane, 2018)
G-6	Crown Mine (2.4 m)		x		(Mphinyane, 2018)
G-7	Glynn Lynderberg (10m) + Nestor Mine (4m)	x	x		(Rudzani et al., 2017)
G-8	Brakpan (10 m)	x			(Nengovhela et al., 2007)
G-9	Knights (10 m)	x			(Nengovhela et al., 2007)
G-10	Edenvale (10 m)	x			(Nengovhela et al., 2007)
G-11	Roodepoort (10 m)	x			(Nengovhela et al., 2007)
G-12	Klerksdorp (10 m)	x			(Nengovhela et al., 2007)
G-13	Jupiter, Germiston		x		(Tutu et al., 2009)
G-14	Mogale tailings dam (10 m)		x	x	(Abegunde et al., 2016)
G-15	Springs (0.15 m)		x		(Maseki, 2017)
G-16	Elsburg Tailing complex (0.15 m)		x		(Maseki, 2017)
G-17	Crown mine 2 (0.15 m)		x		(Maseki, 2017)
G-18	Valkfontein (0.15 m)		x		(Maseki, 2017)
G-19	Carltonville 1 (0.05 m)		x		(Kamunda et al., 2016a)
G-20	Carltonville 2 (0.05 m)		x		(Kamunda et al., 2016a)
G-21	Carltonville 3 (0.05 m)		x		(Kamunda et al., 2016a)
G-22	Carltonville 4 (0.05 m)		x		(Kamunda et al., 2016a)
G-23	Carltonville 5 (0.05 m)		x		(Kamunda et al., 2016a)
G-24	Carltonville 6 (0.05 m)		x		(Kamunda et al., 2016a)
G-25	Carltonville 7 (0.05 m)		x		(Kamunda et al., 2016a)
G-26	Krugersdorp 1 (0.15 m)	x	x		(Ngole-Jeme and Fantke, 2017)
G-27	Krugersdorp 2 (0.15 m)	x	x		(Ngole-Jeme and Fantke, 2017)
G-28	Krugersdorp 3 (0.15 m)	x	x		(Ngole-Jeme and Fantke, 2017)

Appendix E   Agitation energy calculations

Table 0-5: Thiosulphate flowsheet agitation electricity calculations

Parameters	Acid Leaching	Uranium solvent extraction	Uranium stripping	Arsenic removal	Collection vessel	Copper elution	Flash cooling unit	Gold Elution	Gypsum precipitation	Hot Cure Process	Neutralisation	Pressure oxidation (P.O.)	Resin in Leach (concentrate)	Resin in Leach (tailings)	Resin Regeneration	Splash Vessel	Thiosulphate regeneration	Trithionate Manc
Volume (m³/h)	6.18	20.70	14.78	3511.52	3876.21	30.78	42.09	48.66	122.16	42.55	32.12	32.27	83.60	3511.52	30.99	17.83	122.25	27.68
Tank volume (m³)	10.00	25.00	25.00	380.00	380.00	50.00	50.00	50.00	150.00	50.00	50.00	50.00	100.00	380.00	50.00	50.00	150.00	50.00
Number of tanks	1.00	1.00	1.00	9.24	10.20	0.62	0.84	0.97	0.81	0.85	0.64	0.65	0.84	9.24	0.62	0.36	0.81	0.55
Number of tanks (Rounded)	1.00	1.00	1.00	10.00	10.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	10.00	1.00	1.00	1.00	1.00
Agitator type	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator
D(Tank) (m)	2.25	3.05	3.05	7.55	7.55	3.84	3.84	3.84	5.54	3.84	3.84	3.84	4.84	7.55	3.84	3.84	5.54	3.84
D(agitator) (m)	1.35	1.83	1.83	4.53	4.53	2.30	2.30	2.30	3.32	2.30	2.30	2.30	2.90	4.53	2.30	2.30	3.32	2.30
H(Tank) (m)	2.53	3.43	3.43	8.49	8.49	4.32	4.32	4.32	6.23	4.32	4.32	4.32	5.44	8.49	4.32	4.32	6.23	4.32
Agitator Speed (rpm)	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17
Density (kg/m³)	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45
Viscosity (Pa.s)	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Re (-)	1.892E+05	8.711E+05	8.711E+05	8.125E+07	8.125E+07	2.766E+06	2.766E+06	2.766E+06	1.726E+07	2.766E+06	2.766E+06	2.766E+06	8.780E+06	8.125E+07	2.766E+06	2.766E+06	1.726E+07	2.766E+06
Np (-)	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30
Impeller power (kWh)	13.59	62.57	62.57	5836.07	5836.07	198.65	198.65	198.65	1239.65	198.65	198.65	198.65	630.69	5836.07	198.65	198.65	1239.65	198.65
Number of tanks	1.00	1.00	1.00	10.00	10.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	10.00	1.00	1.00	1.00	1.00
Total electricity per unit process(kW)	13.59	62.57	62.57	58360.71	58360.71	198.65	198.65	198.65	1239.65	198.65	198.65	198.65	630.69	58360.71	198.65	198.65	1239.65	198.65

Total electricity for agitators (kWh)	180118.76		Flotation electricity (kWh)	16280.75		Electrowinning electricity (kWh)	0.93
Total Electricity (kWh)	196400.44						

Table 0-6: Cyanide flowsheet agitation electricity calculations

Parameters	Acid Leaching 01	Arsenic removal	Carbon Elution	Carbon in Leach (CIL)	Cyanide leaching 2	Cyanide Removal Reactor	De Aeration	Uranium solvent extraction	Uranium stripping	Zinc Cementation
Volume (m^3/h)	6.18	21239.24	12.32	2853.83	26.08	21238.94	24.71	20.70	14.78	24.98
Tank volume (m^3)	10	380	15	380	30	380	25	25	15	25
Number of tanks	1	55.89	1	7.51	1	55.89	1	1.00	1	1.00
Number of tanks (Rounded)	1	56	1	8	1	56	1	1	1	1
Agitator type	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator	Side entering axial fluidfoil agitator
D(Tank) (m)	2.25	7.55	2.57	7.55	3.24	7.55	3.05	3.05	2.57	3.05
D(agitator) (m)	1.35	4.53	1.54	4.53	1.94	4.53	1.83	1.83	1.54	1.83
H(Tank) (m)	2.53	8.49	2.89	8.49	3.64	8.49	3.43	3.43	2.89	3.43
Agitator Speed (rpm)	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17	19.17
Density (kg/m^3)	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45	1.45
Viscosity (Pa.s)	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Re (-)	1.892E+05	8.125E+07	3.718E+05	8.125E+07	1.180E+06	8.125E+07	8.711E+05	8.711E+05	3.718E+05	8.711E+05
Np (-)	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30
Impeller power (kWh)	13.59	5836.07	26.71	5836.07	84.79	5836.07	62.57	62.57	26.71	62.57
Number of tanks	1	56	1	8	1	56	1	1	1	1
Total electricity per unit process(kW)	13.59	326820.00	26.71	46688.57	84.79	326820.00	62.57	62.57	26.71	62.57
Flotation (kW)	16280.75									
Total electricity for agitators (kW)	716948.84									

Appendix F   Sample calculations

An example for the calculation of agitation requirements for zinc cementation reaction vessel:

$$\text{Tank volume: } V = \frac{\pi D_T^2 H}{4} \qquad \text{[4-63]}$$

$$\frac{H}{D_T} = 1.125 \therefore H = 1.125 \times D_T \qquad \text{[4-64]}$$

Combine [4-63] and [4-64] and make D_T the subject to derive [4-65]:

$$\text{Tank diameter: } D_T = \sqrt[3]{\frac{V}{0.28125 \times \pi}} \qquad \text{[4-65]}$$

The volume in the vessel as determine by HSC Sim® simulation = 24.98 m³/h. The tank volume required is 25 m³ per vessel and therefore 1 vessel are require to meet the residence time requirements for cementation.

The diameter of the tank can then be calculated: $D_T = \sqrt[3]{\frac{V}{0.28125 \times \pi}}$

$$D_T = \sqrt[3]{\frac{25 \; m^3}{0.28125 \times \pi}}$$
$$D_T = 3.05 \; m$$

[4-66]

The agitator diameter can be calculated using the tank dimeter calculated using [4-65]

$$\text{Agitator diameter: } D_a = 0.6 \times D_T$$
$$D_a = 0.6 \times (3.05 \; m)$$
$$D_a = 1.83 \text{ m}$$

[4-67]

Density and viscosity of the slurry was determined based on results reported by Mangesana et al. (2008) and Marsden (1962) as discussed in section 4.9

$$\text{Reynolds number: } N_{Re} = \frac{\rho.N.D_a^5}{\mu}$$

$$N_{Re} = \frac{(1.45 \frac{kg}{m^3}).(1).(1.83 \; m)^5}{(8.38 \times 10^{-3} Pa.s)}$$
$$N_{Re} = 8.71 \times 10^5$$

N_{Re} > 10 000 therefore turbulent flow to ensure solid particles remain in suspension and for hydrofoil impellers a power number (N_p) of 0.3 (Green and Perry, 2007; Hall, 2012).

$g_c = 1$ for metric units. 2 impellers were recommended since slurry viscosity is less than 0.01 Pa.s as discussed in section 4.9 (Green and Perry, 2007; Hall, 2012) [4-68]

Agitator power: $P = \dfrac{N_p.\rho.N^3.D_a{}^5}{g_c}$

$$P = \frac{(0.3).(1.45\ \frac{kg}{m^3}).(19.17\ \frac{rev}{min})^3.(1.83\ \text{m})^5}{(1\frac{m}{s^2} * 1000)}$$

$$P = 62.57\ \text{kWh}$$

Appendix G   GaBi® Flowsheet screenshots
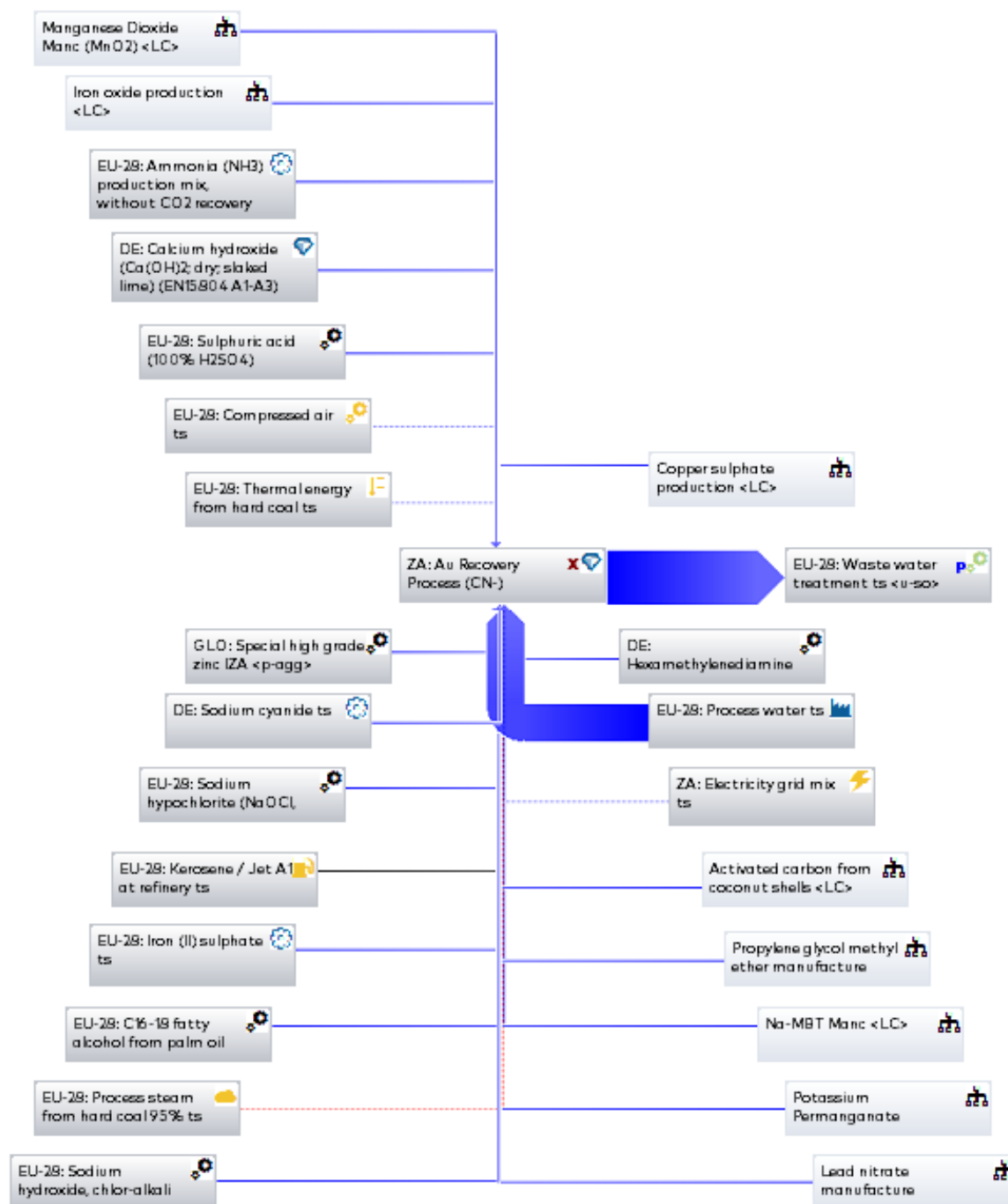
Figure 0-1: GaBi® plan for cyanide leaching flowsheet LCA

Figure 0-2: GaBi® plan for thiosulphate leaching flowsheet LCA

Appendix H   HSC Sim® Flowsheet screenshots
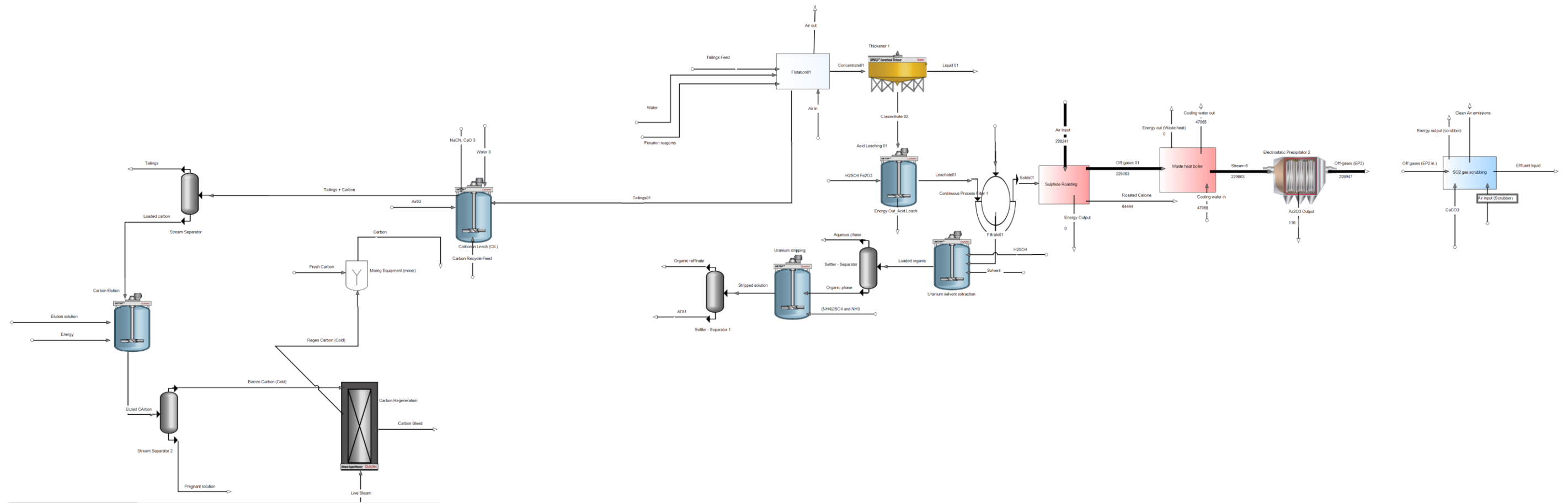
Figure 0-3: HSC Sim® Cyanide flowsheet (Part A: Flotation, Carbon-in-Leach, Uranium recovery, Pyrometallurgical roasting, Effluent gas scrubbing).
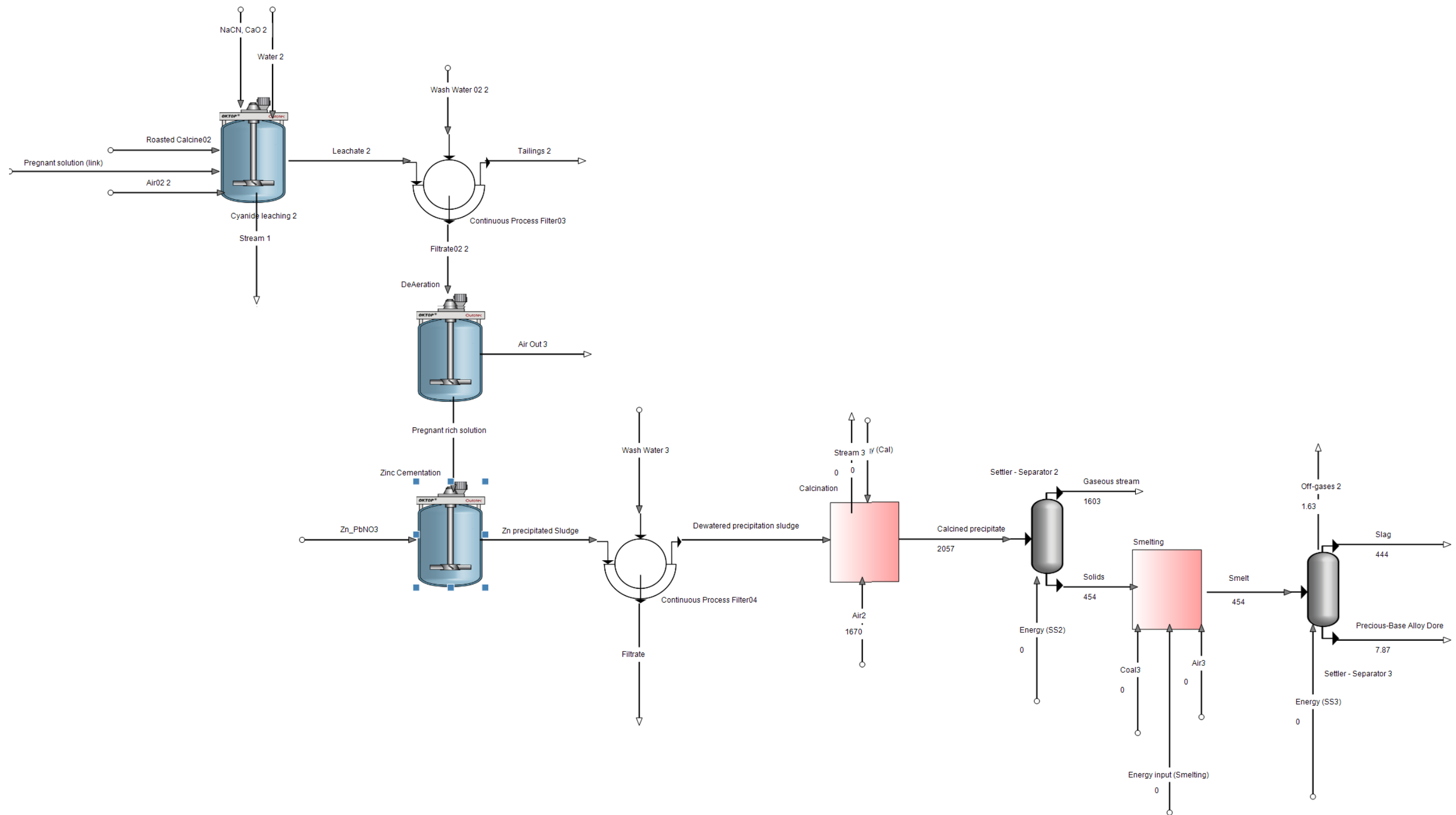
Figure 0-4: HSC Sim® Cyanide flowsheet (Part B: Gold cyanide leaching and zinc cementation recovery, calcination and smelting).

171
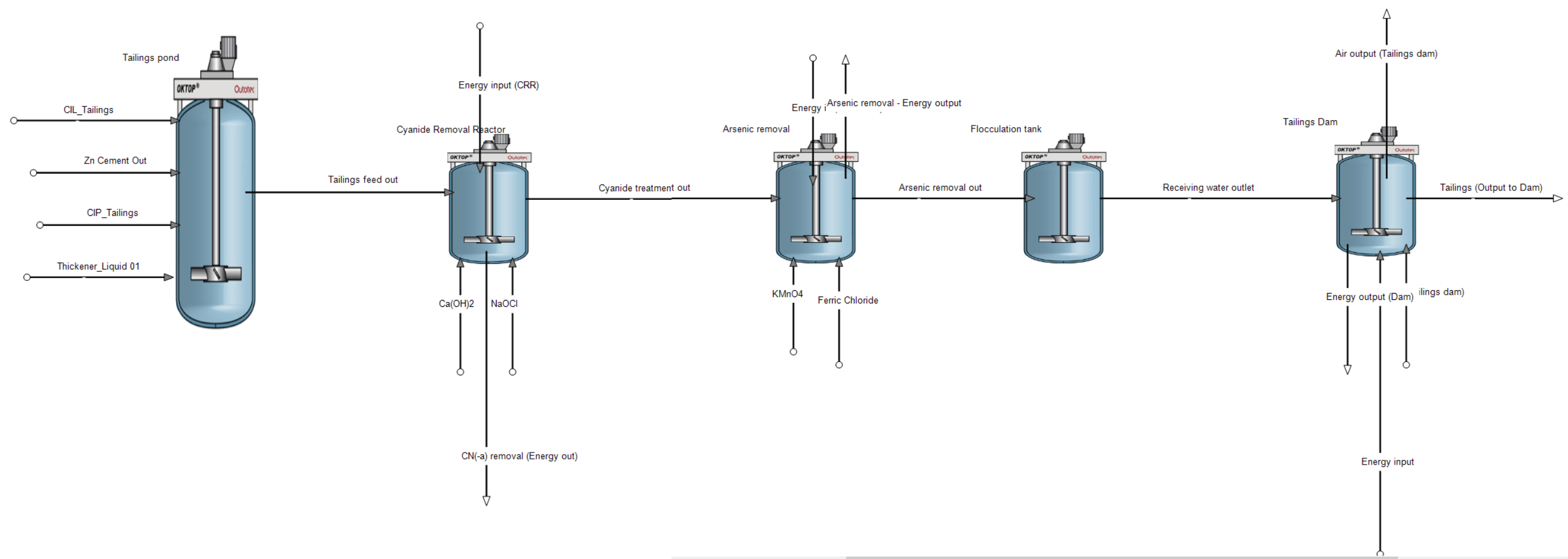
Figure 0-5: HSC Sim® Cyanide flowsheet (Part C: Cyanide destruction and arsenic removal).
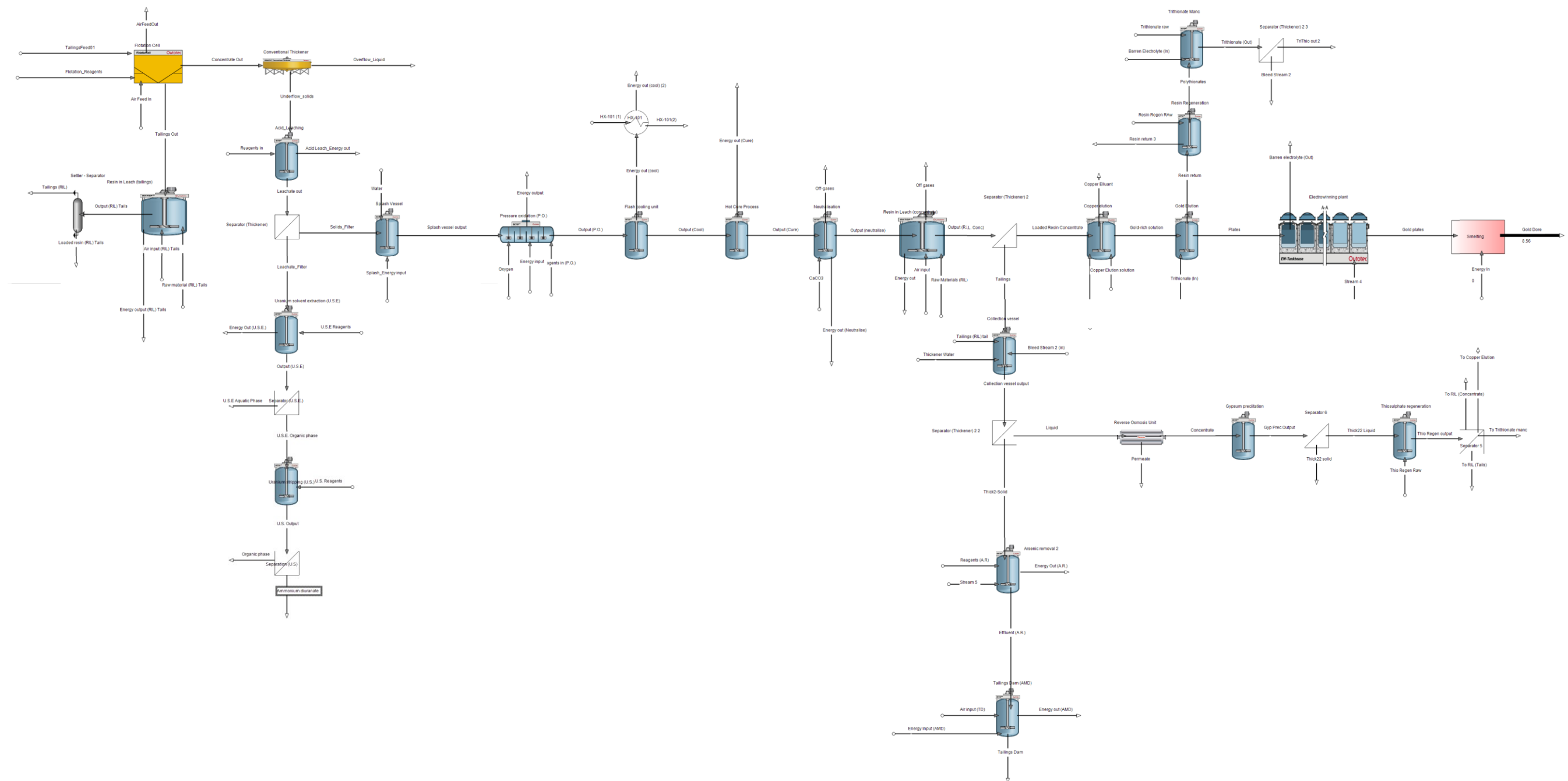
Figure 0-6: HSC Sim® complete thiosulphate flowsheet

Appendix I   Extended chemical reaction tables for HSC Sim ® simulations

Table 0-7: Uranium sulphuric acid leaching reactions for both process flowsheets

Progress (%)	Chemical Reaction	
100	$Fe_2O_3 + 3H_2SO_4(aq) \rightarrow 2Fe^{2+}(aq) + 3SO_4^{2-}(aq) + 3H_2O$	[0-1]
100	$2Fe^{2+}(aq) + MnO_2 + 2H_2SO_4(aq)$ $\rightarrow 2Fe^{3+}(aq) + Mn^{2+}(aq) + 2H_2O + 2SO_4^{2-}(aq)$	[0-2]
100	$H_2SO_4(aq) \rightarrow 2H^+(aq) + SO_4^{2-}(aq)$	[0-3]
84.8	$UO_2 + 2Fe^{3+}(aq) \rightarrow UO_2^{2+}(aq) + 2Fe^{2+}(aq)$	[0-4]
100	$UO_2^{2+}(aq) + SO_4^{2-}(aq) \rightarrow UO_2SO_4$	[0-5]
84.8	$UTi_2O_6 + 2Fe^{3+}(aq) + 3SO_4^{2-}(aq) \rightarrow 2FeSO_4 + 2TiO_2 + UO_2SO_4$	[0-6]
100	$TiO_2 + H_2SO_4(aq) \rightarrow TiOSO_4 + H_2O$	[0-7]
100	$UO_2SO_4 + 2SO_4^{2-}(aq) \rightarrow UO_2(SO_4)_3^{4-}(aq)$	[0-8]
100	$CaCO_3(s) + H_2SO_4(aq) \rightarrow CaSO_4(s) + CO_2(g) + H_2O$	[0-9]
0	$FeS_2 + 14Fe^{3+}(aq) + 8H_2O \rightarrow 2Fe^{2+}(aq) + 2SO_4^{2-}(aq) + 16H^+(aq)$	[0-10]

Table 0-8: Pressure oxidation reaction for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
99	$FeS_2 + 3.5O_2(g) + H_2O \rightarrow 2FeSO_4(aq) + H_2SO_4(aq)$	[0-11]
100	$2FeSO_4(aq) + 2H_2SO_4(aq) + O_2(g) \rightarrow 2Fe_2(SO_4)_3(aq) + 2H_2O$	[0-12]
100	$2As + 1.5O_2(g) + 2H_2O \rightarrow 2HAsO_2(aq)$	[0-13]
100	$2HAsO_2(aq) + 2FeSO_4(aq) + 2H_2SO_4(aq) + O_2(g)$ $\rightarrow Fe_2(SO_4)_3(aq) + 2H_3AsO_4(aq)$	[0-14]
100	$2H_3AsO_4(aq) + Fe_2(SO_4)_3(aq) \rightarrow 2FeAsO_4(aq) + H_2SO_4(aq)$	[0-15]
26	$Fe_2(SO_4)_3(aq) + 2H_2O \rightarrow 2Fe(OH)SO_4(aq) + H_2SO_4(aq)$	[0-16]
67	$Fe_2(SO_4)_3(aq) + 3H_2O \rightarrow Fe_2O_3(s) + 3H_2SO_4(aq)$	[0-17]
0	$2Fe_2(SO_4)_3 + 14H_2O \rightarrow 2H_3OFe_3(SO_4)_2(OH)_6 + H_2SO_4$	[0-18]

Table 0-9: Hot cure process and neutralisation reactions for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
90	$2Fe(OH)SO_4(aq) + H_2SO_4(aq) \rightarrow Fe_2(SO_4)_3(aq) + 2H_2O$	[0-19]
100	$Fe_2(SO_4)_3(aq) + 3CaCO_3 \rightarrow Fe_2O_3(s) + 3CaSO_4(s) + 3CO_2$	[0-20]
100	$H_2SO_4(aq) + 3CaCO_3 \rightarrow CaSO_4(s) + H_2O + 3CO_2(g)$	[0-21]
0	$Fe_2(SO_4)_3(aq) + 3CaCO_3 + 3H_2O \rightarrow Fe(OH)_3(s) + 3CaSO_4(s) + 3CO_2(g)$	[0-22]
0	$2H_3OFe_3(SO_4)_2(OH)_6 + H_2SO_4 \rightarrow 2Fe_2(SO_4)_3 + 14H_2O$	[0-23]

Table 0-10: Cyanide leaching reaction for cyanide leaching flowsheet

Progress (%)	Chemical Reaction	
96.71/58.15	$Au^° + 2CN^-(aq) \rightarrow Au(CN)_2^-(aq) + e^-$	[0-24]
96.71/58.15	$O_2(g) + 2H_2O + 4e^- \rightarrow 4OH^-(aq)$	[0-25]
100	$NaCN \rightarrow Na^+(aq) + CN^-(aq)$	[0-26]
58.15	$Pd^° + 4CN^-(aq) \rightarrow Pd(CN)_4^{2-}(aq) + 2e^-$	[0-27]
58.15	$Ni^° + 4CN^-(aq) \rightarrow Ni(CN)_4^{2-}(aq) + 2e^-$	[0-28]
58.15	$Cu^° + 4CN^-(aq) \rightarrow Cu(CN)_4^{3-}(aq) + e^-$	[0-29]
58.15	$Cd^° + 4CN^-(aq) \rightarrow Cd(CN)_4^{2-}(aq) + 2e^-$	[0-30]
58.15	$Fe^° + 6CN^-(aq) \rightarrow Fe(CN)_6^{3-}(aq) + 3e^-$	[0-31]
58.15	$Zn^° + 4CN^-(aq) \rightarrow Zn(CN)_4^{2-}(aq) + 2e^-$	[0-32]
58.15	$Hg^° + 4CN^-(aq) \rightarrow Hg(CN)_4^{2-}(aq) + 2e^-$	[0-33]
58.15	$Pt^° + 4CN^-(aq) \rightarrow Pt(CN)_4^{2-}(aq) + 2e^-$	[0-34]
58.15	$Co^° + 6CN^-(aq) \rightarrow Co(CN)_6^{3-}(aq) + 3e^-$	[0-35]
58.15	$Ag^° + 2CN^-(aq) \rightarrow Ag(CN)_2^-(aq) + e^-$	[0-36]

Table 0-11: Zinc cementation reaction for cyanide leaching flowsheet

Progress (%)	Chemical Reaction	
100	$Zn° + 2Au(CN)_2^-(aq) \rightarrow 2Au° + Zn(CN)_4^{2-}(aq)$	[0-37]
100	$Zn° + Pd(CN)_4^{2-}(aq) \rightarrow Pd° + Zn(CN)_4^{2-}(aq)$	[0-38]
100	$Zn° + Ni(CN)_4^{2-}(aq) \rightarrow Ni° + Zn(CN)_4^{2-}(aq)$	[0-39]
100	$Zn° + Cu(CN)_4^{3-}(aq) \rightarrow Cu° + Zn(CN)_4^{2-}(aq)$	[0-40]
100	$Zn° + Cd(CN)_4^{2-}(aq) \rightarrow Cd° + Zn(CN)_4^{2-}(aq)$	[0-41]
100	$1.5Zn° + Fe(CN)_6^{3-}(aq) \rightarrow Fe° + 1.5Zn(CN)_4^{2-}(aq)$	[0-42]
100	$Zn° + Hg(CN)_4^{2-}(aq) \rightarrow Hg° + Zn(CN)_4^{2-}(aq)$	[0-43]
100	$Zn° + Pt(CN)_4^{2-}(aq) \rightarrow Pt° + Zn(CN)_4^{2-}(aq)$	[0-44]
100	$Zn° + Co(CN)_6^{3-}(aq) \rightarrow Co° + Zn(CN)_4^{2-}(aq)$	[0-45]
100	$Zn° + 2Ag(CN)_2^-(aq) \rightarrow 2Ag° + Zn(CN)_4^{2-}(aq)$	[0-46]
0	$2Zn° + O_2(g) + 2H_2O \rightarrow 2Zn(OH)_2(s)$	[0-47]

Table 0-12: Thiosulphate leaching reactions for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
100	$CaS_2O_3 \rightarrow Ca^{2+}(aq) + S_2O_3^{2-}(aq)$	[0-48]
100	$CuSO_4 + 3S_2O_3^{2-}(aq) \rightarrow [Cu(S_2O_3)_3]^{5-}(aq) + SO_4^{2-}(aq)$	[0-49]
100	$Cu(s) + 3S_2O_3^{2-}(aq) \rightarrow [Cu(S_2O_3)_3]^{5-}(aq) + e^-$	[0-50]
100	$[Cu(S_2O_3)_3]^{5-}(aq) + O_2(g) \rightarrow [O_2 \cdot Cu(S_2O_3)_3]^{5-}(aq)$	[0-51]
96.71/58.15	$4Au + [O_2 \cdot Cu(S_2O_3)_3]^{5-}(aq) + 8S_2O_3^{2-}(aq) + 2H_2O$ $\rightarrow 4[Au(S_2O_3)_2]^{3-}(aq) + [Cu(S_2O_3)_3]^{5-}(aq) + 4OH^-(aq)$	[0-52]

Progress (%)	Chemical Reaction	
96.71/58.15	$4Ag + [O_2 \cdot Cu(S_2O_3)_3]^{5-}(aq) + 8S_2O_3^{2-}(aq) + 2H_2O$ $\rightarrow 4[Ag(S_2O_3)_2]^{3-}(aq) + [Cu(S_2O_3)_3]^{5-}(aq) + 4OH^-(aq)$	[0-53]
5	$4S_2O_3^{2-}(aq) + [O_2 \cdot Cu(S_2O_3)_3]^{5-}(aq) + 2H_2O$ $\rightarrow 2S_4O_6^{2-}(aq) + [Cu(S_2O_3)_3]^{5-}(aq) + 4OH^-(aq)$	[0-54]
100	$2(C_{22}H_{28}N)Cl + S_2O_3^{2-}(aq) \rightarrow [(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}] + 2Cl^-$	[0-55]
100	$2[Au(S_2O_3)_2]^{3-}(aq) + 3[(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}]$ $\rightarrow 2[(C_{22}H_{28}N)_3{}^{3+}[Au(S_2O_3)_2]^{3-}] + 3S_2O_3^{2-}(aq)$	[0-56]
100	$2[Ag(S_2O_3)_2]^{3-}(aq) + 3[(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}]$ $\rightarrow 2[(C_{22}H_{28}N)_3{}^{3+}[Ag(S_2O_3)_2]^{3-}] + 3S_2O_3^{2-}(aq)$	[0-57]
50	$2[Cu(S_2O_3)_3]^{5-}(aq) + 5[(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}]$ $\rightarrow 2[(C_{22}H_{28}N)_5{}^{5+}[Cu(S_2O_3)_3]^{5-}] + 5S_2O_3^{2-}(aq)$	[0-58]
100	$S_4O_6^{2-}(aq) + [(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}] \rightarrow [(C_{22}H_{28}N)_2^{2+}S_4O_6^{2-}] + S_2O_3^{2-}(aq)$	[0-59]
100	$S_3O_6^{2-}(aq) + [(C_{22}H_{28}N)_2{}^{2+}S_2O_3^{2-}] \rightarrow [(C_{22}H_{28}N)_2^{2+}S_3O_6^{2-}] + S_2O_3^{2-}(aq)$	[0-60]
100	$Ca^{2+}(aq) + SO_4^{2-}(aq) \rightarrow CaSO_4(s)$	[0-61]

Table 0-13: Gold elution reactions for thiosulphate leaching flowsheet

Progress (%)	Chemical Reaction	
49.55	$2[(C_{22}H_{28}N)_3{}^{3+}[Au(S_2O_3)_2]^{3-}] + 3S_3O_6^{2-}(aq) + 2SO_3^{2-}(aq)$ $\rightarrow 3[(C_{22}H_{28}N)_2{}^{2+}S_3O_6^{2-}] + 2[Au(S_2O_3)(SO_3)]^{3-}(aq)$ $+ 2S_2O_3^{2-}(aq)$	[0-62]
49.55	$2[(C_{22}H_{28}N)_3{}^{3+}[Au(S_2O_3)_2]^{3-}] + 3S_4O_6^{2-}(aq) + 2SO_3^{2-}(aq)$ $\rightarrow 3[(C_{22}H_{28}N)_2{}^{2+}S_4O_6^{2-}] + 2[Au(S_2O_3)(SO_3)_2]^{3-}(aq)$ $+ 2S_2O_3^{2-}(aq)$	[0-63]
49.55	$2[(C_{22}H_{28}N)_3{}^{3+}[Au(S_2O_3)_2]^{3-}] + 3S_3O_6^{2-}(aq) + 2SO_3^{2-}(aq)$ $\rightarrow 3[(C_{22}H_{28}N)_2{}^{2+}S_3O_6^{2-}] + 2[Au(S_2O_3)(SO_3)]^{3-}(aq)$ $+ 2S_2O_3^{2-}(aq)$	[0-64]
49.55	$2[(C_{22}H_{28}N)_3{}^{3+}[Ag(S_2O_3)_2]^{3-}] + 3S_4O_6^{2-}(aq) + 2SO_3^{2-}(aq)$ $\rightarrow 3[(C_{22}H_{28}N)_2{}^{2+}S_4O_6^{2-}] + 2[Ag(S_2O_3)(SO_3)_2]^{3-}(aq)$ $+ 2S_2O_3^{2-}(aq)$	[0-65]

Table 0-14: Cyanide destruction reaction for cyanide leaching flowsheet

Progress (%)	Chemical Reaction	
100	$CN^-(aq) + NaClO(aq) \rightarrow CNO^-(aq) + NaCl(aq) + e^-$	[0-66]
100	$2CNO^-(aq) + 3NaClO(aq) + 2H^+(aq) + 2e^-$ $\rightarrow N_2(g) + 2CO_2(g) + 3NaCl(aq) + H_2O$	[0-67]
0.00068	$H_2O \rightarrow H^+(aq) + OH^-(aq)$	[0-68]
100	$NaCN \rightarrow Na^+(aq) + CN^-(aq)$	[0-69]
100	$Zn(CN)_4^{2-}(aq) + 4NaClO(aq) \rightarrow Zn + CNO^-(aq) + 4NaCl(aq) + 2e^-$	[0-70]
100	$Pd(CN)_4^{2-}(aq) + 4NaClO(aq) \rightarrow Pd + 4CNO^-(aq) + 4NaCl(aq) + 2e^-$	[0-71]
100	$Ni(CN)_4^{2-}(aq) + 4NaClO(aq) \rightarrow Ni + 4CNO^-(aq) + 4NaCl(aq) + 2e^-$	[0-72]
100	$Cu(CN)_2^-(aq) + 2NaClO(aq) \rightarrow Cu + 2CNO^-(aq) + 2NaCl(aq) + e^-$	[0-73]
100	$Cd(CN)_4^{2-}(aq) + 4NaClO(aq) \rightarrow Cd + 4CNO^-(aq) + 4NaCl(aq) + 2e^-$	[0-74]
100	$Hg(CN)_4^{2-}(aq) + 4NaClO(aq) \rightarrow Hg + 4CNO^-(aq) + 4NaCl(aq) + 2e^-$	[0-75]
100	$Co(CN)_6^{3-}(aq) + 6NaClO(aq) \rightarrow Co + 6CNO^-(aq) + 6NaCl(aq) + 3e^-$	[0-76]
100	$Pt(CN)_4^{2-}(aq) + 4NaClO(aq) \rightarrow Pt + 4CNO^-(aq) + NaCl(aq) + 2e^-$	[0-77]
100	$Fe(CN)_6^{4-}(aq) + 4NaClO(aq) \rightarrow 4Fe(CN)_6^{3-}(aq) + NaCl(aq) + H_2O$	[0-78]
100	$4FeSO_4 + 3Fe(CN)_6^{3-}(aq) \rightarrow Fe_4(Fe(CN)_6)_3(s) + 4SO_4^{2-}(aq)$	[0-79]

Appendix J   Thiosulphate and cyanide equilibrium stability constants

Table 0-15: Equilibrium stability constants for metal cyanide complexes

Cyanide complex	Stability constant (log β)	Cyanide complex	Stability constant (log β)	Cyanide complex	Stability constant (log β)
$Co(CN)_6^{3-}$	64 [a]	$Ni(CN)_4^{2-}$	30.22 [c]	$Co(CN)_6^{3-}$	19.1 [a]
$Au(CN)_4^{-}$	56 [a]	$Cu(CN)_4^{3-}$	23.1 [c]	$Cd(CN)_4^{2-}$	17.92 [c]
$Pd(CN)_4^{2-}$	51.6 [b]	$Cu(CN)_3^{2-}$	21.66 [c]	$Cd(CN)_3^{-}$	16.65 [c]
$Fe(CN)_6^{3-}$	43.6 [c]	$Ag(CN)_3^{2-}$	21.40 [c]	$Cu(CN)_3^{2-}$	16.26 [c]
$Pt(CN)_4^{2-}$	41 [b]	$Ag(CN)_4^{3-}$	20.8 [c]	$Zn(CN)_3^{-}$	16.05 [c]
$Au(CN)_2^{-}$	39.3 [a]	$Ag(CN)_2^{-}$	20.48 [c]		
$Fe(CN)_6^{4-}$	35.4 [c]	$Zn(CN)_5^{3-}$	20.17 [a]		
$Hg(CN)_2$	32.8 [d]	$Zn(CN)_4^{2-}$	19.62 [c]		
[a] :(Sillen et al., 1964) cited by (Wang and Forssberg, 1990)					
[b]: (Muir and Ariti, 1991) cited by (Mpinga et al., 2014)					
[c]: (Smith and Martell, 1989) cited by (Wang and Forssberg, 1990)					
[d]: (Miltzarek et al., 2002)					

Table 0-16: Equilibrium stability constants of metal thiosulphate complexes of metal ions

Metal thiosulphate complex	Stability constant (log β)	Metal thiosulphate complex	Stability constant (log β)
$Pt(S_2O_3)_4^{6-}$	43.6 [a]	$Cd(S_2O_3)_4^{6-}$	7.10 [c]
$Ti(S_2O_3)_4^{6-}$	41.1 [a]	$Pb(S_2O_3)_4^{6-}$	6.2 [c]
$Pd(S_2O_3)_4^{6-}$	34.9 [a]	$Zn_2(S_2O_3)_2$	5.84 [c]
$Hg(S_2O_3)$	33.6 [a]	$Ni(S_2O_3)$	2.06 [c]
$Au(S_2O_3)_2^{3-}$	28 [b]	$Co(S_2O_3)$	2.05 [c]
$Ag(S_2O_3)_2^{3-}$	14.2 [a]	$Mn(S_2O_3)$	1.99 [a]
$Cu(S_2O_3)_3^{5-}$	13.6 [a]	$Mg(S_2O_3)$	1.99 [a]
$Fe(S_2O_3)_2^{-}$	8.72 [a]	$Fe(S_2O_3)_3^{3-}$	1.98 [a]
$Pb(S_2O_3)$	8.30 [a]	$Ca(S_2O_3)$	1.90 [a]
[a]:Cited by (Grosse et al., 2003) [b]:(Sullivan and Kohl, 2019) [c]:(Smith and Martell, 1989)			

Appendix K   Extended life cycle inventory (LCI) for LCA on process flowsheets

Table 0-17: Complete and extended life cycle inventory for cyanide flowsheet (functional unit 1kg gold)

Input	Quantity	#a	TFb	Output	Quantity	#a	TFb
Activated carbon [Organic intermediate products]	2,711.5	kg	X	Aluminium [Inorganic emissions to industrial soil]	69,522.8	kg	*
Aluminium [Non renewable elements]	69,522.8	kg		Aluminium silicate [Inorganic intermediate products]	1,420,598.0	kg	*
Aluminium silicate [Inorganic intermediate products]	1,420,598.0	kg		Aluminium sulphate [Inorganic intermediate products]	4,216,797.8	kg	*
Aluminium sulphate [Inorganic intermediate products]	4,216,797.8	kg		Ammonia [Inorganic emissions to fresh water]	192.5	kg	*
Ammonia [Inorganic intermediate products]	192.5	kg	X	Antimony [Heavy metals to industrial soil]	2.6	kg	*
Antimony [Non renewable elements]	2.6	kg		Arsenic [Heavy metals to industrial soil]	0.0	kg	*
Arsenic [Non renewable elements]	361.8	kg		Arsenic trioxide [Heavy metals to air]	3.8	kg	*
Barium [Non renewable elements]	381.4	kg		Arsenic trioxide [Product]	378.3	kg	*
Beryllium [Non renewable elements]	3.4	kg		Barium [Inorganic emissions to industrial soil]	381.4	kg	*
Bismuth [Non renewable elements]	18.9	kg		Beryllium [Inorganic emissions to industrial soil]	3.4	kg	*
Boron [Non renewable elements]	1.3	kg		Bismuth [Non renewable elements]	18.9	kg	*
Cadmium [Non renewable elements]	2.4	kg		Boron [Inorganic emissions to industrial soil]	1.3	kg	*
Calcium [Non renewable elements]	33,169.6	kg		Cadmium [Heavy metals to industrial soil]	2.1	kg	*
Calcium carbonate (> 63 microns) [Minerals]	61,299.5	kg	X	Cadmium oxide [Inorganic intermediate products]	0.3	kg	*
Carbon [Organic intermediate products]	12,035.3	kg		Calcium [Inorganic emissions to industrial soil]	6,635.9	kg	*
Chromium [Non renewable elements]	1,042.9	kg		Calcium hydroxide [ecoinvent long-term to air]	4,178.4	kg	*
Cobalt [Non renewable elements]	115.9	kg		Carbon (unspecified) [Organic emissions to industrial soil]	135.55	kg	*
Compressed air [Mechanical energy]	121,563.6	Nm³	X	Carbon dioxide [Inorganic emissions to air]	69,843.0	kg	*
Copper [Non renewable elements]	195.4	kg		Chromium [Heavy metals to industrial soil]	1,155.7	kg	*
Copper sulphate [Inorganic intermediate products]	466.7	kg	X	Cobalt [Heavy metals to industrial soil]	115.9	kg	*
Electricity [Electric power]	5,736,465.1	MJ	X	Copper [Heavy metals to industrial soil]	30.3	kg	*

Input	Quantity	#a	TFb	Output	Quantity	#a	TFb
Fatty alcohol (C16-18) [Valuable substances]	76.4	kg	X	Copper sulphate [Inorganic emissions to air]	466.7	kg	*
Feldspar (aluminium silicates) [Non renewable resources]	1,028,185.1	kg		Cyanide (unspecified) [Inorganic emissions to air]	0.0	kg	*
Ferrous sulphate (dissolved) [Inorganic intermediate products]	395.1	kg	X	Dodecanol [Organic emissions to fresh water]	76.4	kg	*
Gold [Non renewable elements]	1.3	kg		Feldspar (aluminium silicates) [Non renewable resources]	1,028,487.3	kg	*
Gypsum (natural gypsum) [Non renewable resources]	59,995.1	kg		Gold [Heavy metals to industrial soil]	0.5	kg	*
Hexamethylene diamine (HMDA) [Organic intermediate products]	232.0	kg	X	Gold [Metals]	1.0	kg	X
Hydrated lime dry slaked [Minerals]	51,562.9	kg	X	Gypsum [Waste for recovery]	183,666.8	kg	*
Hydrochloric acid (100%) [Inorganic intermediate products]	3,440.5	kg	X	Hexamethylene diamine (HMDA) [Hydrocarbons to sea water]	45.9	kg	*
Iron [Non renewable elements]	78,056.3	kg		Hydroxide [Inorganic emissions to fresh water]	0.0	kg	*
Iron oxide (II-oxide) [Inorganic intermediate products]	3,889.5	kg	X	Iron [Heavy metals to industrial soil]	6,556.7	kg	*
Kerosene [Refinery products]	1,219.2	kg	X	Iron oxide [Hazardous non organic waste for disposal]	115,284.3	kg	*
Lead [Non renewable elements]	114.7	kg		Lead [Heavy metals to industrial soil]	14,885.1	kg	*
Lead nitrate (Pb(NO$_3$)$_2$) [Plastics]	23,841.0	kg	X	Magnesium [Inorganic emissions to industrial soil]	13,178.6	kg	*
Magnesium [Non renewable elements]	13,178.5	kg		Magnesium silicate [Minerals]	2,391,232.9	kg	*
Magnesium silicate [Minerals]	2,391,232.9	kg		Manganese [Heavy metals to industrial soil]	9,174.5	kg	*
Manganese [Non renewable elements]	9,174.5	kg		Manganese dioxide [Inorganic intermediate products]	12,527.9	kg	*
Manganese dioxide [Inorganic intermediate products]	578.1	kg	X	Mercury [Heavy metals to industrial soil]	0.3	kg	*
Mercury [Non renewable elements]	0.9	kg		Mercury [Metals]	0.6	kg	*
Molybdenum [Non renewable elements]	11.1	kg		Molybdenum [Heavy metals to industrial soil]	11.1	kg	*
Nickel [Non renewable elements]	447.2	kg		Nickel [Heavy metals to industrial soil]	447.2	kg	*
Palladium [Non renewable elements]	8.0	kg		Nitrate [Inorganic emissions to fresh water]	9,089.7	kg	*
Phosphorus [Non renewable elements]	1,024.8	kg		Nitrogen (atmospheric nitrogen) [Inorganic emissions to air]	566,720,775.5	kg	*

180

Input	Quantity	#ᵃ	TFᵇ	Output	Quantity	#ᵃ	TFᵇ
Platinum [Non renewable elements]	0.0	kg		Oxygen [Inorganic emissions to air]	41,148.0	kg	*
Potassium [Non renewable elements]	2,569.2	kg		Palladium [Metals]	3.8	kg	
Potassium permanganate [Inorganic intermediate products]	335.6	kg	X	Palladium scrap [Waste for recovery]	3.4	kg	*
Propylene glycol methyl ether acetate (PGMEA) [Organic intermediate products]	205.0	kg	X	Paraffin oil [Organic emissions to industrial soil]	1,406.5	kg	*
Pyrite [Non renewable resources]	37,050.1	kg		Phosphorus [Inorganic emissions to agricultural soil]	1,024.8	kg	*
Quartz sand (silica sand; silicon dioxide) [Non renewable resources]	2,708,349.8	kg		Platinum [Metals]	0.0	kg	
Rubidium [Non renewable elements]	21.5	kg		Potassium [Inorganic emissions to industrial soil]	2,569.2	kg	*
Selenium [Non renewable elements]	13.9	kg		Quartz sand (silica sand; silicon dioxide) [Non renewable resources]	2,708,349.8	kg	*
Silver [Non renewable elements]	27.2	kg		Rubidium [Inorganic emissions to fresh water]	21.5	kg	*
Sodium cyanide [Inorganic intermediate products]	112,355.3	kg	X	Selenium [Heavy metals to industrial soil]	13.9	kg	*
Sodium hydroxide (100%; caustic soda) [Inorganic intermediate products]	2,604.3	kg	X	Silver [Heavy metals to industrial soil]	11.8	kg	*
Sodium hypochlorite [Inorganic intermediate products]	13,478.7	kg	X	Silver [Metals]	15.5	kg	
Sodium Mercaptobenzothiazole [Plastics] [Plastics]	1,025.1	kg	X	Sodium [Inorganic emissions to agricultural soil]	45,986.2	kg	*
Special high grade zinc [Metals]	303.2	kg	X	Sodium chloride (rock salt) [Inorganic emissions to fresh water]	8,855.4	kg	*
Steam (MJ) [steam]	1,119,579.1	MJ	X	Sodium hydroxide [ecoinvent long-term to air]	0.0	kg	*
Strontium [Non renewable elements]	53.5	kg		Sodium hypochlorite [Inorganic emissions to industrial soil]	0.0	kg	*
Sulphuric acid (100%) [Inorganic intermediate products]	12,658.4	kg	X	Strontium [Inorganic emissions to industrial soil]	53.5	kg	*
Tantalum [Non renewable elements]	0.7	kg		Sulphate [Inorganic emissions to industrial soil]	5,980.9	kg	*
Thermal energy (MJ) [Thermal energy]	1,051.0	MJ	X	Sulphur dioxide [Inorganic emissions to air]	4.0	kg	*
Thorium [Non renewable elements]	383.3	kg		Sulphuric acid [Inorganic emissions to industrial soil]	30,843.6	kg	*
Titanium [Non renewable elements]	3,016.5	kg		Tantalum [Heavy metals to fresh water]	0.7	kg	*

Input	Quantity	#a	TFb	Output	Quantity	#a	TFb
Uranium, in ground [Uranium (resource)]	254.4	kg		Thorium [Heavy metals to industrial soil]	383.3	kg	*
Vanadium [Non renewable elements]	27.3	kg		Titanium [Heavy metals to industrial soil]	3,016.5	kg	*
Water (desalinated; deionised) [Operating materials]	16,607,772.4	kg	X	Uranium [Radioactive emissions to fresh water]	2,452,818,319.6	Bq	*
Water (waste water, untreated) [Production residues in life cycle]	10,618,084.0	kg		Uranium, fuel grade, 2291 GJ per kg [Uranium products]	51.1	kg	
Zinc [Metals]	1,155.7	kg		Vanadium [Heavy metals to industrial soil]	27.3	kg	*
				Water (waste water, untreated) [Production residues in life cycle]	27,207,008.7	kg	*
				Water vapour [Inorganic emissions to air]	18,847.6	kg	*
				Zinc [Heavy metals to industrial soil]	1,459.0	kg	*
Total	**39,608,380.9**	**kg**		**Total**	**39,611,069.5**	**kg**	
% Error	**0.0068**	**%**					
a:# = units b: TF: Tracked flows							

Table 0-18: Complete and extended life cycle inventory for thiosulphate flowsheet (functional unit 1kg gold)

Input	Quantity	#a	TFb	Output	Quantity	#a	TFb
Aluminium [Non renewable elements]	69,502.3	kg		Aluminium [Inorganic emissions to industrial soil]	69,502.3	kg	*
Aluminium silicate [Inorganic intermediate products]	1,420,180.7	kg		Aluminium silicate [Inorganic intermediate products]	1,420,180.7	kg	*
Aluminium sulphate [Inorganic intermediate products]	4,215,558.9	kg		Aluminium sulphate [Inorganic intermediate products]	4,215,558.9	kg	*
Ammonia [Inorganic intermediate products]	192.4	kg	X	Ammonia [Inorganic emissions to fresh water]	192.4	kg	*
Antimony [Non renewable elements]	2.6	kg		Antimony [Heavy metals to industrial soil]	2.6	kg	*
Arsenic [Non renewable elements]	361.7	kg		Arsenic [Heavy metals to industrial soil]	0.0	kg	*
Barium [Non renewable elements]	381.3	kg		Barium [Inorganic emissions to industrial soil]	381.3	kg	*
Beryllium [Non renewable elements]	3.4	kg		Beryllium [Inorganic emissions to industrial soil]	3.4	kg	*
Bismuth [Non renewable elements]	18.9	kg		Bismuth [Non renewable elements]	18.9	kg	*
Boron [Non renewable elements]	1.3	kg		Boron [Inorganic emissions to industrial soil]	1.3	kg	*
Cadmium [Non renewable elements]	2.4	kg		Boron [Non renewable elements]	1.3	kg	*
Calcium [Non renewable elements]	33,169.6	kg		Cadmium [Heavy metals to industrial soil]	2.4	kg	*
Calcium carbonate (> 63 microns) [Minerals]	62,161.8	kg	X	Calcium [Inorganic emissions to industrial soil]	33,169.6	kg	*
Carbon [Organic intermediate products]	12,035.3	kg		Carbon (unspecified) [Organic emissions to industrial soil]	12,035.3	kg	*
Chromium [Non renewable elements]	1,042.6	kg		Carbon dioxide [Inorganic emissions to air]	27,275.1	kg	*
Cobalt [Non renewable elements]	115.8	kg		Chromium [Heavy metals to industrial soil]	1,042.6	kg	*
Compressed air [Mechanical energy]	3,542.2	Nm³	X	Cobalt [Heavy metals to industrial soil]	115.8	kg	*
Copper [Non renewable elements]	195.3	kg		Copper [Heavy metals to industrial soil]	195.3	kg	*
Copper sulphate [Inorganic intermediate products]	444.5	kg	X	Copper sulphate [Inorganic emissions to air]	444.5	kg	*
Electricity [Electric power]	1,570,973.9	MJ	X	Dodecanol [Organic emissions to fresh water]	76.4	kg	*
Fatty alcohol (C₁₆-C₁₈) [Valuable substances] (Dodecanol proxy)	76.4	kg	X	Feldspar (aluminium silicates) [Non renewable resources]	1,028,185.1	kg	*
Feldspar (aluminium silicates) [Non renewable resources]	1,028,185.1	kg		Gold [Heavy metals to industrial soil]	0.3	kg	*
Ferrous sulphate (dissolved) [Inorganic intermediate products]	395.0	kg	X	Gold [Metals]	1.0	kg	X
Gold [Metals]	1.3	kg		Gypsum [Waste for recovery]	117,041.0	kg	*

Input	Quantity	#a	TFb	Output	Quantity	#a	TFb
Gypsum [Waste for recovery]	59,977.5	kg		Hexamethylene diamine (HMDA) [Organic intermediate products] [Alamine® 336/Amberjet® 4200 Proxy]	1,394.7	kg	*
Hexamethylene diamine (HMDA) [Organic intermediate products] [Alamine® 336/Amberjet® 4200 Proxy]	1,394.7	kg	X	Iron [Heavy metals to industrial soil]	78,033.4	kg	*
Hydrated Lime [Minerals]	120.9	kg	X	Iron oxide [Hazardous non organic waste for disposal]	15,824.3	kg	*
Hydrogen peroxide (100%) [Inorganic intermediate products]	4,996.8	kg	X	Lead [Heavy metals to industrial soil]	114.7	kg	*
Iron [Non renewable elements]	78,033.4	kg		Magnesium [Inorganic emissions to industrial soil]	13,174.7	kg	*
Iron oxide (II-oxide) [Inorganic intermediate products]	3,888.3	kg	X	Magnesium silicate [Minerals]	2,390,530.4	kg	*
Kerosene [Refinery products]	1,218.8	kg	X	Manganese [Heavy metals to industrial soil]	9,174.5	kg	*
Lead [Non renewable elements]	114.7	kg		Manganese dioxide [Inorganic intermediate products]	48,481.4	kg	*
Magnesium [Non renewable elements]	13,174.7	kg		Mercury [Heavy metals to industrial soil]	0.9	kg	*
Magnesium silicate [Minerals]	2,390,530.4	kg		Molybdenum [Heavy metals to industrial soil]	11.1	kg	*
Manganese [Non renewable elements]	9,174.5	kg		Nickel [Heavy metals to industrial soil]	447.0	kg	*
Manganese dioxide [Inorganic intermediate products]	577.9	kg	X	Palladium scrap [Waste for recovery]	0.0	kg	*
Mercury [Non renewable elements]	0.9	kg		Paraffin oil [Organic emissions to industrial soil]	1,406.1	kg	*
Molybdenum [Non renewable elements]	11.1	kg		Phosphorus [Inorganic emissions to agricultural soil]	442.5	kg	*
Nickel [Non renewable elements]	447.0	kg		Potassium [Inorganic emissions to industrial soil]	2,568.5	kg	*
Oxygen gaseous [Inorganic intermediate products]	37,290.9	kg	X	Potassium permanganate [Inorganic intermediate products]	0.0	kg	*
Palladium scrap [Waste for recovery]	0.0	kg		Propylene glycol methyl ether acetate (PGMEA) [Organic intermediate products] [Dow® 200]	205.0	kg	*
Phosphorus [Non renewable elements]	1,024.8	kg		Quartz sand (silica sand; silicon dioxide) [Non renewable resources]	2,707,554.1	kg	*
Potassium [Non renewable elements]	2,568.5	kg		Rubidium [Inorganic emissions to fresh water]	21.4	kg	*
Potassium permanganate [Inorganic intermediate products]	222.2	kg	X	Selenium [Heavy metals to industrial soil]	13.9	kg	*
Propylene glycol methyl ether acetate (PGMEA) [Organic intermediate products] [Dow® 200]	205.0	kg	X	Silver [Heavy metals to industrial soil]	11.8	kg	*
Pyrite [Non renewable resources]	37,050.09	kg		Silver [Metals]	15.5	kg	

184

Input	Quantity	#[a]	TF[b]	Output	Quantity	#[a]	TF[b]
Quartz sand (silica sand; silicon dioxide) [Non renewable resources]	2,707,554.1	kg		Sodium [Inorganic emissions to industrial soil]	0.0	kg	*
Rubidium [Non renewable elements]	21.4	kg		Sodium Mercaptobenzothiazole [Plastics] [Plastics]	1,024.8	kg	*
Selenium [Non renewable elements]	13.9	kg		Sodium sulphate [Inorganic emissions to fresh water]	0.0	kg	*
Silver [Non renewable elements]	27.2	kg		Strontium [Inorganic emissions to industrial soil]	53.5	kg	*
Sodium hydroxide (100%; caustic soda) [Inorganic intermediate products]	2,644.0	kg	X	Sulphur [Inorganic emissions to industrial soil]	0.0	kg	*
Sodium Mercaptobenzothiazole [Plastics] [Plastics]	1,024.8	kg	X	Sulphuric acid [Inorganic emissions to industrial soil]	30,834.6	kg	*
Sodium sulphide [Inorganic intermediate products]	3,442.8	kg	X	Thorium [Heavy metals to industrial soil]	383.2	kg	*
Sodium sulphite (Na$_2$SO$_3$) [Intermediate products]	816.6	kg	X	Titanium [Heavy metals to industrial soil]	3,015.6	kg	*
Sodium thiosulfate [Inorganic intermediate products]	23,237.5	kg	X	Uranium, fuel grade, 2291 GJ per kg [Uranium products]	51.1	kg	
Strontium [Non renewable elements]	53.5	kg		Uranium, in ground [Uranium (resource)]	84.7	kg	*
Sulphuric acid (100%) [Inorganic intermediate products]	12,110.6	kg	X	Vanadium [Heavy metals to industrial soil]	27.3	kg	*
Tantalum [Non renewable elements]	0.7	kg		Water (desalinated; deionised) [Operating materials]	15,535,783.2	kg	*
Thermal energy (MJ) [Thermal energy]	131,441.9	MJ	X	Water (waste water, untreated) [Production residues in life cycle]	9,932,713.8	kg	*
Thorium [Non renewable elements]	383.2	kg		Zinc [Heavy metals to industrial soil]	303.2	kg	*
Titanium [Non renewable elements]	3,015.6	kg					
Uranium, in ground [Uranium (resource)]	254.3	kg					
Vanadium [Non renewable elements]	27.3	kg					
Water (desalinated; deionised) [Operating materials]	15,535,783.2	kg	X				
Water (waste water, untreated) [Production residues in life cycle]	9,932,713.8	kg					
Zinc [Non renewable elements]	303.2	kg					
Total	**37,709,479.3**	**kg**		**Total**	**37,699,128.4**	**kg**	
% Error	**0.027**	**%**					

[a]:# = units   [b]: TF: Tracked flows

185