Assessing Water Risks in the Mining Industry using Life Cycle Assessment Based Approaches

Stephen Alan Northey
BEnvEng(Hons), MAusIMM, MIEAust

A thesis submitted for the degree of *Doctor of Philosophy* at
Monash University in 2018

Department of Civil Engineering, Faculty of Engineering

Copyright notice

Abstract

The mining industry is significant to national economies. However, existing studies of the mining industries water consumption and hydrological impacts are typically limited to detailed case studies of individual mining operations. Consequently, there are few industry wide datasets and studies available to understand how the water use impacts of the mining industry vary across regions, industry sub-sectors and hydrological settings.

A detailed literature review was conducted to determine how life cycle assessment and water footprinting methods have been applied to the mining industry. From this review, the major limitations and the opportunities for further use of these methods for assessments of the mining industries water use were identified.

Detailed analysis was performed to understand how copper, lead-zinc and nickel resources are situated in the context of regional water resources, climate regimes and life cycle assessment impact characterisation factors. The analysis demonstrated the common industry narrative that several major copper producing regions are more acutely exposed to these water risks than other sub-sectors of the industry. Furthermore, there is also a likelihood that the climate in many regions containing base metal resources will undergo changes over the coming decade that may alter the risk profile of hydrological, water quality and infrastructure risks at mining operations.

The spatial distribution of mine production in relation to water use impact characterisation factors was also assessed for 25 mined commodities. From this analysis, it was found that the results of studies may be sensitive to the choice of characterisation factors used and also the spatial resolution of the study. Assessing the industries consumptive water use impacts through the use of national average factors for the water stress index (WSI) and the Available Water Remaining (AWaRe) factors for non-agricultural water use is likely to lead to, on average, an overestimation of impacts for the mining industry. This is compared to assessments using watershed specific factors. Therefore, watershed based inventory development and impact assessment is strongly recommended for future life cycle assessment studies of the mining industry. This is one of the major findings from this study that industry future research should address.

The results of the study provide a quantitative basis for advancing discussions of the mining industries water use and risks, whilst also providing avenues for improving future life cycle assessment studies that consider the water use impacts of the mining industry and its products. Datasets were developed that can aid in the assessment of the industries water use impacts at national and global boundaries for many mineral commodities. This includes detailed datasets for a large number of mineral deposits. Through the combination of these datasets and life cycle assessment methodology, a more holistic assessment can be made regarding the contribution of the mining industry to achieving sustainable development objectives.

Publications during enrolment

Journal Articles

Northey, S.A., Mudd, G.M., Saairvuori, E., Wessman-Jääskeläinen, H., Haque, N. (2016). Water footprinting and mining: Where are the limitations and opportunities? Journal of Cleaner Production, 135, pp. 1098-1116. http://dx.doi.org/10.1016/j.jclepro.2016.07.024

Northey, S.A., Mudd, G.M., Werner, T.T., Jowitt, S.M., Haque, N., Yellishetty, M., Weng, Z. (2017). The exposure of global base metal resources to water criticality, scarcity and climate change. Global Environmental Change, 44, pp. 109-124. http://dx.doi.org/10.1016/j.gloenvcha.2017.04.004

Northey, S.A., Mudd, G.M., Werner, T.T. (2017). Unresolved Complexity in Assessments of Mineral Resource Depletion and Availability. Natural Resources Research, volume unassigned, 15p. http://dx.doi.org/10.1007/s11053-017-9352-5

Werner, T.T., Ciacci, L., Mudd, G., Reck, B., Northey, S. (2018). Looking Down Under for a circular economy of indium. Environmental Science & Technology, in press. http://dx.doi.org/10.1021/acs.est.7b05022

Northey, S.A., Madrid Lopez, C., Haque, N., Mudd, G.M., Yellishetty, M. Production weighted water use impact characterisation factors for the global mining industry: A comparison of watershed and national average WSI and AWaRe factors. Submitted to the Journal of Cleaner Production.

Mudd, G.M., Roche, C.P., Northey, S.A., Jowitt, S.M. Mining in Papua New Guinea: Linking Key Trends and Complex Environmental-Social Issues. Submitted to Science of the Total Environment.

Conference Abstracts, Articles and Presentations

Northey, S., Haque, N., Mudd, G. (2015). Life-cycle based water footprinting methodology in the production of metal commodities. LCM Australia 2015, Melbourne, VIC, 23-27 November 2015, 2p.

Northey, S.A., Mudd, G.M., Haque, N. (2015). The Challenges in Estimating the Water Footprint of Mined Commodities. SENG 2015 National Conference, Adelaide, VIC, 9-10 September 2015, paper 15, 4p.

Browning, C., Northey, S., Haque, N., Bruckard, W., Cooksey, M. (2016). Life Cycle Assessment of Rare Earth Production from Monazite. REWAS 2016: Towards Material Resource Sustainability (Ed. Kirchain et al.), Springer International Publishing, pp. 83-88.

Northey, S.A., Haque, N., Mudd, G. (2016). Water-related Data Requirements for Improved Life Cycle Assessment of Mining, Mineral Processing and Tailings Management. Life-of-Mine 2016 Conference, Brisbane, QLD, 28-30 September 2016, pp. 16-18.

Northey, S.A., Mudd, G.M. (2016). Resource Depletion Scenarios – How should we address the limitations? 35th International Geological Congress, Cape Town, South Africa, paper 2731, 2p. ***Invited presentation.***

Hoekstra, D., Northey, S., Upton, M., Williamson, P. (2016) Mine water footprint as a primary tool for water stewardship. Industrial Water Use and Reuse Workshop – Strategies for Sustainability Water Management for Mining, American Institute of Chemical Engineers, Denver CO, United States, 24 October 2016.

Northey, S., Upton, M., Williamson, P., Hoekstra, D. (2017). Water Footprinting? Communicating mine site water performance in a circular economy. Engineering Solutions for Sustainability: Materials and Resources 3 symposium, Denver, CO, United States, 18-19 February 2017, 1p.

Berger, M., Sonderegger, T., Bach, V., Dewulf, J., Drielsma, J., Guinée, J., Helbig, C., Motoshita, M., Northey, S., Sala, S., Schmidt, M., Sonnemann, G., Thorenz, A., Vieira, B., Weidema, B. (2017). Harmonizing the assessment of resource use in LCA. SETAC Europe 27th Annual Meeting, Brussels, Belgium, 7-11 May 2017, poster.

Thesis including published works General Declaration

I hereby declare that this thesis contains no material which has been accepted for the award of any other degree or diploma at any university or equivalent institution and that, to the best of my knowledge and belief, this thesis contains no material previously published or written by another person, except where due reference is made in the text of the thesis.

This thesis includes 3 original papers published in peer reviewed journals and 1 unpublished publication. The core themes of the thesis are water, mining and life cycle assessment. The ideas, development and writing up of all the papers in the thesis were the principal responsibility of myself, the candidate, working within the Department of Civil Engineering under the supervision of Dr Mohan Yellishetty.

The inclusion of co-authors reflects the fact that the work came from active collaboration between researchers and acknowledges input into team-based research.

In the case of chapters 3, 4, 5 and 6.4 my contribution to the work involved the following:

Thesis chapter	Publication title	Publication status*	Nature and extent (%) of students contribution	Co-author name(s) Nature and % of Co-author's contribution	Co-author(s), Monash Student Y/N
3	Water footprinting and mining: Where are the limitations and opportunities?	Published (Q1, IF: 5.871)	75%, study design, literature review, analysis and manuscript preparation	1) Gavin Mudd, 7.5%, manuscript drafting 2) Elina Saarivuori, 7.5%, study design, manuscript drafting 3) Helena Wessman- Jääskeläinen, 5%, study design, manuscript drafting 4) Nawshad Haque, 5%, manuscript drafting	N N N N
4	The exposure of global base metal resources to water criticality, scarcity and climate change	Published (Q1, IF: 7.693)	65%, study design, data analysis and manuscript preparation	1) Gavin M. Mudd, 7.5%, study design 2) Timothy T. Werner, 7.5%, data preparation 3) Simon M. Jowitt, 5%, discussion 4) Nawshad Haque, 5%, discussion 5) Mohan Yellishetty, 5% discussion 6) Zhehan Weng, 5%, data preparation	N Y N N N Y
5	Production weighted water use impact characterisation factors for the global mining industry: A comparison of watershed and national average WSI and AWaRe factors	Returned for revision	75%, study design, data analysis and manuscript preparation	1) Cristina Madrid Lopez, 7.5%, study design, analysis and discussion 2) Nawshad Haque, 7.5%, study design 3) Gavin M. Mudd, 5%, discussion 4) Mohan Yellishetty, 5%, discussion	N N N N
6.4	Unresolved Complexity in Assessments of Mineral Resource Depletion and Availability	Published (Q2, IF: 1.309)	75%, manuscript preparation	1) Gavin M. Mudd, 15%, drafting & review 2) Timothy T. Werner, 10%, section drafting	N Y

I have renumbered sections of submitted or published papers in order to generate a consistent presentation within the thesis.

Student signature: **Date:** 8 / 11 / 2017

The undersigned hereby certify that the above declaration correctly reflects the nature and extent of the student and co-authors' contributions to this work.

Main Supervisor signature: **Date:** 8 / 11 / 2017

Acknowledgements

Acknowledgement is given to the valuable guidance and advice of my supervisors throughout the course of the project: Assoc. Prof. Gavin Mudd (RMIT University), Dr Nawshad Haque (CSIRO) and Dr Mohan Yellishetty (Monash University). The contributions of Prof. Sankar Bhattacharya (Monash University) early in the project are also valued.

This project was primarily funded by the Mineral Resources business unit of the Commonwealth Scientific and Industrial Research Organisation (CSIRO), with some minor funding also obtained through the Monash Institute for Graduate Studies travel grant program. This financial assistance, as well as access to internal CSIRO resources and staff throughout the duration of the project is gratefully appreciated. Various colleagues at CSIRO provided valuable guidance on the nature and direction of the research undertaken, including: Dr Mark Cooksey, Mr Roy Lovel, Mr Andreas Monch and Ms Anna Littleboy. The work of the following CSIRO staff to conduct internal review of articles prior to submission to journals was also appreciated, these include: Dr Mark Cooksey, Dr Justine Lacey, Dr Bradley Ridoutt, Dr Daniel Jewell and Mr Warren Bruckard. The assistance of Mr Sam Stoddart for compiling and analysing data for tailings management in the Australian mining industry is also appreciated.

The valuable contribution of researchers at other institutions is also acknowledged for collaborating on articles or facilitating research visits and seminars. These include: Dr Tim Werner (RMIT University), Dr Zhehan Weng (Ghent University), Dr Helena Wessman-Jääskeläinen and Ms Elina Saarivuori (VTT Technical Research Centre of Finland Ltd), Dr Kevin Harding and Assoc. Prof. Craig Sheridan (University of Witwatersrand), Prof. Upmanu Lall and Mr Luc Bonnafous (Columbia University), Dr Greg Keir (Sustainable Minerals Institute, University of Queensland), Dr Cristina Madrid Lopez (Universitat Autonoma de Barcelona & Yale University), and Dr Barbara Reck and Prof. Thomas Graedel (Yale University). Considerable insights have also been gained from participants in the WULCA sub-committee on the natural resources area-of-protection, as well as meetings of the UNEP-SETAC Life Cycle Initiative’s Global Guidance on Environmental Indicators project task force on natural resources.

The efforts of Prof. Neil McIntyre (Sustainable Minerals Institute, University of Queensland) and Assoc. Prof. Jenny Broadhurst (University of Cape Town) to review this thesis and provide critical feedback are highly appreciated. In addition, various journal editors and anonymous peer reviewers have greatly improved the quality of the research presented in this thesis.

Knowledge is cumulative and so the published work of the thousands of authors who have shaped my worldview and understanding over the past decade must also be recognised. To quote a heavily paraphrased translation of Lucius Annaeus Seneca, “I shall never be ashamed of citing a bad author if the line is good.”

Finally, I would like to thank my parents, Mr Alan and Mrs Helen Northey, for setting a foundation and instilling a perplexing sense of lackadaisical ambition in me. Ms Belinda and Mr Michael Northey for continued mischief. My late grandfather, Mr John F. Northey, for some timely words of encouragement mid-way through the project. Finally, my beautiful wife, Mrs Chloe Northey, for her continued support, reluctant proofreading and stern feedback.

Table of Contents

List of Figures

List of Tables

- - - - - PAGE LEFT INTENTIONALLY BLANK - - - - -

1. Introduction

The mineral resources provided by the mining industry contribute towards meeting society's desired standard of living and sustainable development objectives. Despite the value these resources provide, the industry is a magnet for controversy and public opposition, particularly as individual mining operations can cause significant impacts to local environments and water resources if not properly managed. Local communities can be impacted, both positively and negatively, by the development of mining operations. Many of the pressures, impacts and risks associated with mining are being altered or exacerbated overtime due to the changing nature of the industry. Some of the changes in the industry include increases in: the scale of mining operations, extracted waste rock, overburden and tailings volumes, mining and processing of lower grade ores, the intensity of resource use (e.g. energy and water) per unit of production, and implementation of low-cost bulk mining techqniues (Crowson, 2003; Mudd, 2010). The overall growth of the industry over the 20th and early 21st centuries has been substantial, and so the continuation of these trends would see a growth in the pressure placed by mining operations on surrounding environments and communities. Therefore, there is a strong incentive to improve our understanding of the impacts of the mining industry so that informed decision making, improved mine-site management, and better societal outcomes can be achieved. As part of this, it has been recognised that there is a need to develop criteria that can be used to assess the progress and contribution of the mining industry towards meeting sustainable development objectives (Fonseca et al., 2013; Moran et al., 2014).

Currently, there have been few studies of the regional scale impacts of mining (Moran et al., 2014). Most of the existing studies conducted for the mining industry are focused upon assessing the potential impacts of an individual mining operation, either as part of environmental impact assessment processes, monitoring of environmental performance targets, or for identifying opportunities to improve management practices and mitigate the environmental risks associated with the mining operation. Considering the cumulative impacts associated with multiple mine developments that occur in resource regions may lead to benefits for decision making, as well as an improved understanding of the system and how the aggregated impacts may evolve through time and across geographies (Franks et al., 2013). Some limited research has been undertaken to understand how the cumulative impacts of mining may be evaluated alongside other industries (e.g. Moran et al., 2013). However, these forms of study can still be considered exceptions to the rule. In order to facilitate these forms of assessment, methods are required that enable the fair comparison of positive and negative impacts occurring across industries and geographical regions.

Many of the environmental, cross-sectoral and social impacts of the mining industry are related to water resources. Water resources were highlighted by members of the mining industry as being in their top three sustainable development focuses during the ten year update report of the Mining, Minerals and Sustainable Development (MMSD) project (IIED, 2012). Even though the mining industry is generally considered a minor consumer of water globally compared to other industries (Gunson et al., 2013; Hejazi et al., 2014), individual operations may have the potential to cause large impacts to local hydrology or water quality. Thus there is often substantial reported data and hydrological modelling available to understand water use and management strategies of particular mines and mineral processing facilities. However, there is still a very poor understanding of how the mining industry as a whole intersects with water resources at national and industry-wide scales, partially due to a lack of suitable methodology for comparing water use at mining operations that are situated in very different hydrological and climate contexts. These contexts can differ substantially between mining operations and require very different approaches to water management (ICMM, 2012).

Methods for quantifying the water consumption or water use impacts associated with production systems have advanced significantly over the past two decades. For example, the concept of ‘virtual’ or ‘embodied’ water was established to describe the water consumption required to produce goods and services, to aide in the discussion of the benefits and detriments of trade (Allen, 1998). This concept was extended, when Hoekstra (2003) coined the term ‘water footprint’ to bring greater geographical and temporal specificity to virtual water discussions. Following this, the methods and data sources available to perform a water footprint assessments have developed considerably, particularly through standardisation efforts by the Water Footprint Network (Hoekstra et al., 2009; 2011). Also there have been significant advances in life cycle assessment (LCA) based methodologies for quantifying consumptive and degradative water use impacts over the past decade (Boulay et al., 2015). As a consequence, the developers of the ISO14046 (2014) standard ‘Environmental Management – Water Footprint – Principles, requirements and guidelines’ adopted an approach consistent with the broader framework methodology for LCA, ISO14044 (2006) ‘Environmental Management – Life Cycle Assessment – Requirements and Guidelines’. LCA based methods seek to enable consistent and fair assessment of consumptive water use impacts across industries, geographical locations and through time. Although there have been significant advances in the development of these various methods, the existing applications of these methods have mostly been limited to understanding the impacts of the major water consuming industries such as agriculture, forestry and power generation. As a result, there are limited examples of LCA methods being used to evaluate or study the ‘water footprint’ of mining operations or mined products.

As the mining industry continues to evolve, there has been a substantial increase in public disclosures by the mining industry regarding their environmental, social and economic performance (Fonseca et al., 2014; Jenkins and Yakovleva, 2006; Mudd, 2008) – which has been greatly aided by the rise of the internet facilitating improved and widespread communication of this information. These public disclosures are increasingly becoming standardised, with a prominent example being the adoption of the Global Reporting Initiative by mining companies (Jenkins and Yakovleva, 2006). However, even within these broad reporting standards, there are difficulties for individual companies to apply these standards consistently due to differences in interpretation, and the availability of resources and expertise within the company. In the context of water, these challenges have not gone unrecognised and so mining industry bodies are increasingly providing recommendations and best-practice guidelines for reporting to these schemes. For instance, water accounting and reporting guidance for the mining industry is provided by the Minerals Council of Australia (MCA, 2014), as well as the International Council on Mining & Metals (ICMM, 2017). Therefore, as the quantity of data being made available to analyse the industry is continually increasing, it is also likely that the overall quality of this data will increase overtime as additional standardisation and industry guidance is made available – and as companies continue to develop experience in these forms of disclosure. The increasing disclosure of industry performance through sustainability reporting and other disclosure mechanisms presents a unique and timely opportunity to gain broad-scale understanding of how the industry intersects with society and the environment.

This PhD project has sought to apply the recent advances in LCA methodology to study how the mining industry intersects with water resources. Existing data sources, relevant methodology and industry practices were reviewed to identify research gaps and the opportunities for improving our understanding of these issues. An exhaustive spatial analysis of global mineral resources and production was undertaken to understand how industry sub-sectors are distributed in relation to local water and climate contexts. Detailed datasets of water withdrawals, use and discharges from individual mining operations have also been compiled to improve the data available for use in life cycle inventories and industry assessments. The outcomes of the project represent a significant advance in our understanding of how the minerals industry interacts with water resources.

1.1. Thesis Structure

Chapter 1 introduces the research project, the structure of the thesis and the key research questions being addressed.

Chapter 2 outlines background methodology regarding life cycle assessment and its current application to the mining industry, as well as existing approaches for evaluating a water footprint.

Chapter 3 provides a detailed review of the complexity of mine-water interactions, key life cycle assessment based water footprinting methodology, 'water footprint' assessments conducted of the mining industry to date, and an identification of the data and methodology limitations that prevent further use of water footprinting methodology. Then an assessment of the opportunities of these studies to inform the industry or improve understanding of mine-water interactions is provided.

Chapter 4 details the local hydrological and climate context of regions containing the major base metal resources (copper, nickel and lead-zinc). The assessment is performed using a variety of spatial water indices used for life cycle assessment, as well as spatial and regional Köppen-Geiger climate classifications. The evolution of these climate classifications under published climate scenarios are assessed. A broad ranging discussion of the potential implications for hydrological and water quality risks associated with mining operations is provided.s

Chapter 5 provides a spatial assessment of global mine production in relation to two spatial characterisation factors for assessing water use impacts within life cycle assessment. This provides an understanding of the potential deviation between results produced from national and watershed scale assessments, as well as the relative exposure of industry sub-sectors to contextual water stress or scarcity risks.

Chapter 6 discusses the role that water resources play in the development of mineral resource projects. A discussion paper is presented that explores how long-term mineral resource depletion and availability are assessed in LCA, MFA and resource criticality studies. Improvements to studies in these domains can be aided by knowledge of the local social, environmental and economic contexts surrounding mineral deposits – and so understanding the local hydrological contexts of the mining industry may also play a role in informing these type of studies.

Chapter 7 outlines advances in industry water use reporting and shows how these disclosures can be used to compile detailed statistics that describe the water balances of mining and mineral processing operations, using the copper industry as a case study.

Chapter 8 outlines avenues for future research that would lead to improved life cycle assessment and water scarcity footprint assessments of mined products. Opportunities for improving life cycle inventory data are discussed, and the various conceptual models and datasets that could be developed to aide this are discussed. The suitability of existing life cycle assessment methods for quantifying water use impacts is also discussed. From this, a roadmap for future research to develop global and regionalised water footprint assessments of mined products is presented.

Chapter 9 presents the main conclusions of the research undertaken.

Appendices provide summaries of the detailed datasets and additional publications associated with this research project.

1.2. Research Questions

Following detailed literature review of existing data resources and studies, a range of open questions were identified that guided the research presented in this thesis. These questions have all been addressed in some capacity within the thesis, either quantitatively through detailed data analysis or qualitatively through detailed discussion (see Figure 1.1).

1: How does the mining industry interact with water resources?

(a) How variable are water management practices across the mining industry?
(b) What is the magnitude and variability of water use at individual mining operations?
(c) How do local water or climate contexts influence mine site water management?
(d) What water accounting and reporting standards are being used by the mining industry?

2: How exposed is the mining industry to local water stress and scarcity?

(a) What are the hydrological or water use contexts of regions containing mineral resources or production?
(b) What are the local climate in regions containing mineral resources?
(c) Are there differences in exposure to water related risks between mineral commodities?

3: How can 'water footprint' and life cycle assessment based approaches be used to understand the impacts of water consumption in the mining industry?

(a) How have water footprint and life cycle assessment based methods been previously used to quantify water use in the mining industry?
(b) Are there opportunities to improve life cycle inventory datasets for the mining industry?
(c) Are existing water use impact characterisation methods and factors suitable for application to the mining industry?

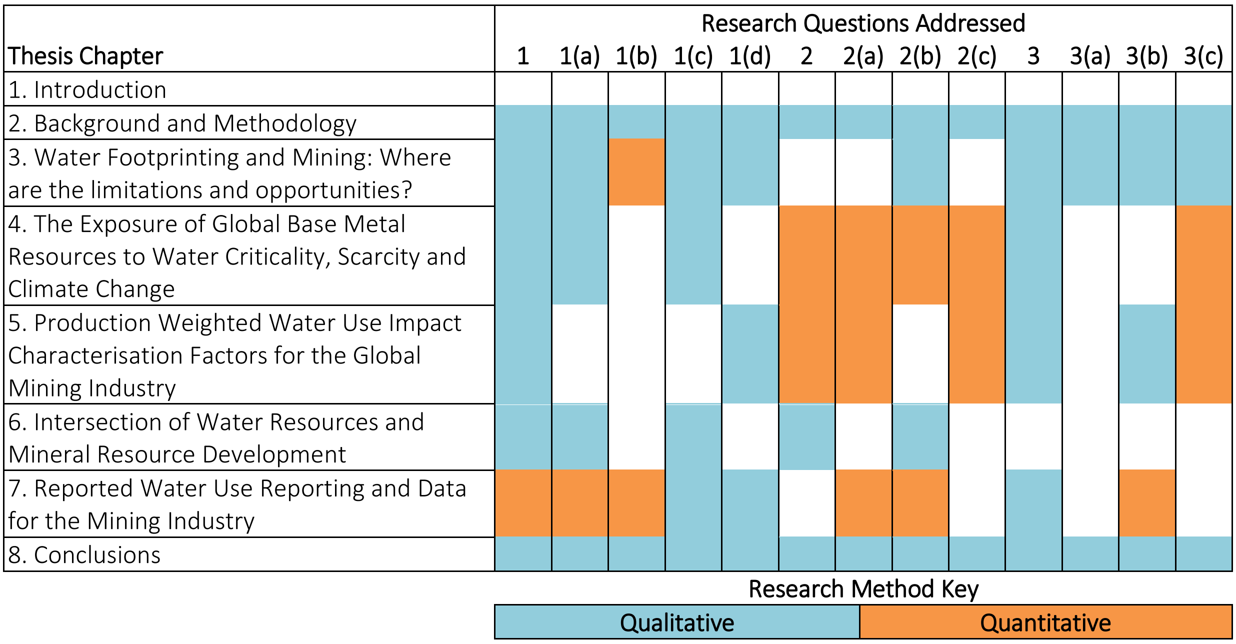

Figure 1.1: Research questions addressed either qualitatively or quantitatively in each chapter of this thesis.

2. Background Methodology

A particular emphasis of research presented in this thesis relates to how life cycle assessment and also water footprinting methodology can be used to develop a more informed understanding of how the mining industry relates to water resources. Many readers may be unfamiliar with these approaches to assessing environmental impacts and water resource burdens. Therefore, a brief overview of the life cycle assessment and water footprinting methodology is presented in this chapter to give the reader sufficient context to interpret the research presented throughout the thesis. Some examples of how life cycle assessment methods have been applied to the mining industry is also presented.

2.1. Life Cycle Assessment

Life cycle assessment is a framework methodology that enables evaluation of the environmental burdens or impacts associated with products and services. A key feature of life cycle assessment is the ability to consistently evaluate the indirect impacts that occur throughout supply chains and production systems. In doing so, life cycle assessment studies are able to evaluate the environmental impacts that occur at all stages of a products life cycle, from raw material acquisition and manufacturing of the product, through product usage and then finally the ultimate disposal or recycling of the product.

Life cycle assessment methodology has developed significantly over the past thirty years to enable evaluation of a wide variety of production systems and environmental impact categories. The basic stages, structure and approaches to communicating the results of life cycle assessment studies are outlined by the international standard, 'ISO14044:2006 – Environmental Management – Life Cycle Assessment – Requirements and Guidelines' (ISO, 2006). There are four general stages required for a full life cycle assessment study:

1. 'Goal and Scope Setting' to determine the purpose and intended use of the study, as well as the overall methodological approach and communication strategy taken by the study. Aspects of this may include the determination of a 'functional unit' that inventory and impact assessment will be conducted for (e.g. the production of 1kg of refined copper), as well as geographic and temporal system boundaries of assessment that will enable meaningful results to be reached.

2. 'Inventory Analysis' to develop a quantitative representation of the interactions of the product system with the natural environment. This may include evaluation of the energy, resource, pollution and materials flows between the production system, the natural environment and other parts of the 'techno-sphere' (the anthropogenic environment). The inventory data may also include other relevant information such as land occupation. Inventory data is typically expressed relative to the 'functional unit' of the system.

3. 'Impact Assessment' is then used to convert the inventory data into estimates of environmental impacts, along defined impact characterisation pathways. Impact characterisation procedures have been developed to assess a wide range of environmental impact categories. Such as contributions to climate change, ozone depletion, freshwater eutrophication and eco-toxicity. Some methods also allow evaluation of damage to major areas of human concern, such as human health, ecosystem quality and resource availability. One principle of life cycle assessment is that

multiple environmental impact categories should ideally be assessed simultaneously to enable evaluation of potential trade-offs between impact categories.

4. ‘Interpretation’ of the inventory and impact assessment are then conducted to determine how the outcomes of the study can be used to inform decision making. The framework methodology described by the ISO standard provides considerable flexibility to enable life cycle assessment studies to be tailored to support the needs of decision makers.

Organisations such as the UNEP-SETAC Life Cycle Initiative provide guidance on life cycle assessment methodology. An example of this is the recent global guidance project aimed at building consensus on life cycle impact assessment approaches, with task forces being developed to review and address issues pertaining to specific impact categories (Frischknecht et al., 2016).

2.2. Life Cycle Inventory Data for the Mining Industry

The goals and scope of a particular study will dictate the boundary of assessment used for inventory and impact assessment. A study may consider the full ‘cradle-to-grave’ life cycle of an end-consumer product – ranging from raw material acquisition, material fabrication, manufacturing, product use and then the final disposal of the product. Alternatively, studies may utilise a more narrow assessment boundaries to better understand environmental burdens associated with the production of intermediate products (e.g. mineral concentrates or refined metals) or individual unit processes. An example of potential boundaries of assessment are shown in Figure 2.1.

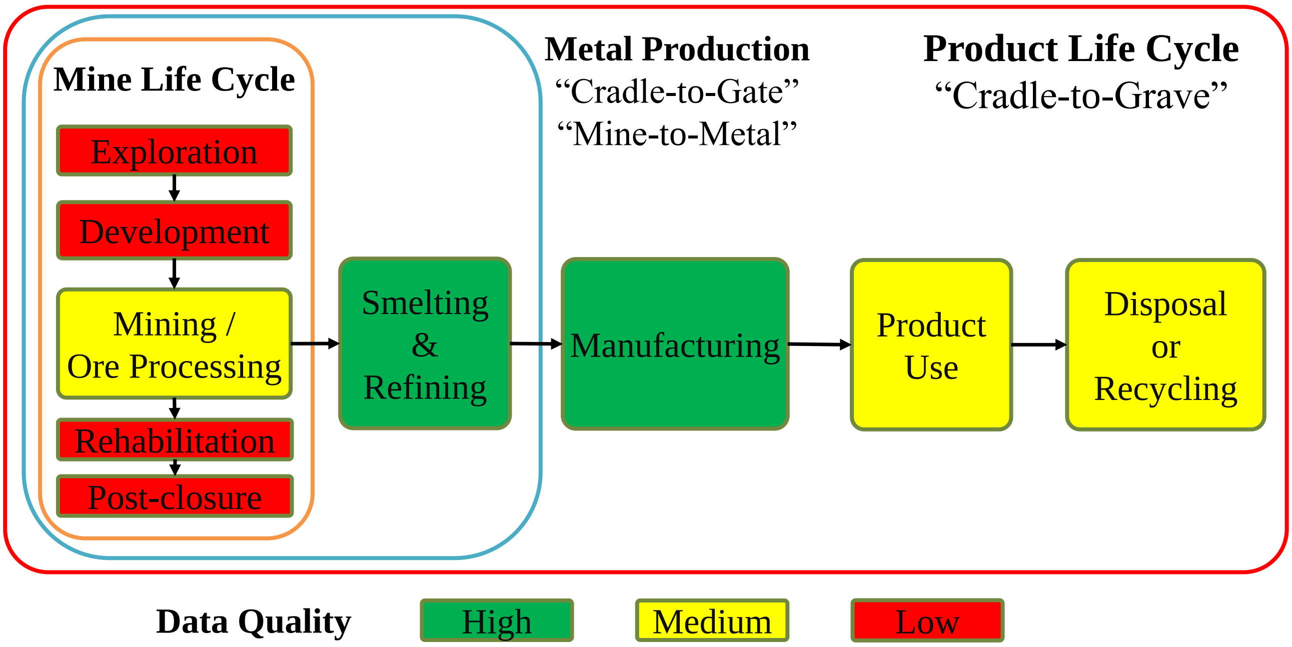

Figure 2.1: Several possible boundaries of assessment and the author’s perception of data quality of existing life cycle inventories.

The development of life cycle inventory data for mining and metal production requires overcoming of a range of conceptual limitations and also the compilation or modelling of data for inventory flows. The approaches to developing inventory data range from theoretical process modelling to industrial surveys and liaisons with industry bodies or organisations. Most datasets for mining and mineral processing are developed as part of broader inventory development for metal products. Overviews of available life cycle inventory data sources for metal products are provided by UNEP (2013) and Nuss and Eckelman (2014). The development of the Ecoinvent life cycle inventory database (Althaus and Classen, 2005; Classen et al., 2005) represented a significant advance in the sophistication of inventory data for mined and metal products. This database still forms the basis of much of the life cycle inventory data available for many mined mineral or metal commodities (refer to supplementary information of Nuss and Eckelman, 2014). Inventory data has also been developed by many of the international or regional metal associations (UNEP, 2013), such as: the World Steel Association, the European Copper Institute, the International Copper Association, the International Zinc Association, European Aluminium Association, International Aluminium Institute, the Nickel Institute, and the International Molybdenum Association. As reported by UNEP (2013), the forms of water use data provided by these various sources is inconsistent in terms of the specification of source, use (e.g. cooling water, process water) and quality parameters.

2.3. Applications of Life Cycle Assessment to the Mining Industry

Despite progress being made to overcome methodological challenges relevant to mining, mineral processing and metal production systems, the adoption and use of life cycle assessment to study the mining industry is still somewhat limited compared to the breadth of studies available for other industries (Awuah-Offei and Adekpedjou, 2011; Balanay and Halog, 2016; Santero and Hendry, 2016; Yellishetty et al., 2009). Life cycle assessment studies of mining, mineral processing and metal production are varied in terms of their scope, purpose and their intended use for decision making. For instance, some studies seek to evaluate the environmental impacts of competing production or processing routes, while other studies may seek to make a comparative evaluation of the environmental impacts associated with the primary (mined) and secondary (reused or recycled) supply of materials. A few other studies may seek to make comparisons of competing industries within a geographic region to aide in decision making regarding natural resource management.

Some studies present very detailed life cycle inventory data and impacts assessment for production processes or individual regions. For instance, Qi et al. (2017) provided a detailed analysis of primary zinc production in China, which included detailed inventory from the mining through the hydrometallurgical production chain. This enabled the contribution of mining to the overall environmental burden of zinc production to be evaluated across 18 impact categories.

Other studies compare the relative environmental impacts associated with producing mined commodities from different mining operations. For instance, Weng et al. (2016) evaluated the gross energy requirements and carbon footprint of rare earth production from 26 potential or already operating mining projects. Detailed analysis was undertaken to understand the influence that geological parameters – such as ore mineralogy and grade – will have on the environmental impacts of production. Results were also broken down by processing stage (mining, mineral beneficiation and refining) to enable identification of the processes that should be preferentially focused on to reduce overall energy consumption and greenhouse gas emissions.

Studies have also utilised life cycle assessment to evaluate potential tailings management options. For instance, Reid et al. (2009) assessed tailings storage options for a mine in Canada, such as mine backfill or cover and revegetation options. The results of the assessment highlighted that life cycle assessment can provide unique insights into tailings management options, however these forms of assessment are highly site specific and can be sensitive to factors such as the temporal boundaries of the study or the suitability of the impact assessment models to the local context. Another study conducted by Song et al. (2017) evaluated potential options for an underground copper mine in Norway, which included options for replacing diesel trucks with all-electric trucks, replacing fuel oil used to heat buildings with natural gas, and also an additional electrodialytic process to recover further copper from the flotation tailings.

Life cycle assessment has also been used to evaluate how the environmental impacts of mining operations can evolve through time. For instance, Memary et al. (2012) evaluated the evolution of greenhouse gas emissions, acidification potential and photochemical ozone creation potential overtime from five Australian copper mines. The results were able to be used to determine the influence that key factors such as changing ore grades, sources of electricity, technology or site infrastructure (for instance the closure of a smelters) have had on these three environmental impact categories.

Within Australia, the Commonwealth Scientific and Industrial Research Organisation (CSIRO) has conducted significant research over the past two decades to assess the environmental impacts of primary metal production systems using life cycle assessment, primarily as a means to identify 'hotspots' for process improvement. Some of the studies by CSIRO in this field include:

- Comparisons of mining approaches such as the energy and greenhouse gas trade-offs between conventional hauling and in-pit crushing and conveying (Norgate and Haque, 2013), or non-traditional mining approaches such as in-situ leaching (Haque and Norgate, 2014), or underground and open cut mining of different commodities (Norgate and Haque, 2010).
- Processing of nickel laterite ores (Norgate and Jahanshahi, 2011);
- Treatment options for low grade copper and nickel ores (Norgate and Jahanshahi, 2010);
- Mining and processing of refractory and non-refractory gold ores (Norgate and Haque, 2012);
- Utilisation of biomass to substitute coal use in steel and ferroalloy production (Haque and Norgate, 2013; Jahanshahi et al., 2015); and,
- Understanding the environmental impacts of the major processing routes (pyrometallurgical, hydrometallurgical, etc.) for metal commodities (Norgate et al., 2007).

And, of particular relevance for this doctoral thesis:

- Norgate and Lovel (2004; 2006) analysed the direct and indirect water consumption associated with metal production, assuming Australian conditions. The direct water use was estimated based upon data obtained from sustainability reporting, whereas indirect water use was estimated using life cycle assessment approaches.

During previous employment with CSIRO, prior to the commencement of doctoral studies, the author completed several studies to extend and update the work of Norgate and Lovel (2004; 2006):

- Northey et al. (2013) assessed the water use, energy consumption and greenhouse gas emissions of 31 copper mining operations, based upon a review of the corporate sustainability reporting of major copper producing companies. Copper mines were classified based upon mine type (i.e. underground and open pit) and processing configuration (e.g. heap leaching, flotation concentration, smelting, and refining). The results for energy consumption and greenhouse gas emissions displayed significant correlation with factors such as ore grades, which confirmed the expectations of prior life cycle assessment studies. However, the results for water use revealed significant variability in water use between individual mining operations, regardless of processing configuration, ore throughput and ore grades (Figure 2.2). Therefore further investigation to understand how water use varies across the industry was recommended.

- Northey and Haque (2013) modelled the direct and indirect water use associated with some of the major copper, gold and nickel production routes (Figure 2.3), based upon development of process models and a detailed review of available process data and literature. It was found that the results of Norgate and Lovel (2004; 2006) for direct water consumption were reasonable once ore grades were adjusted for, however it is likely that indirect water consumption was underestimated by previous studies – particularly for hydrometallurgical processing of ores via acid leaching.

- Northey et al., (2014a) extended the prior study by applying the single-indicator water footprint methodology developed by Ridoutt and Pfister (2013b) to assess these process systems. With additional analysis to understand uncertainty in the inventory data, assess case studies of several mine sites, and to evaluate the global distribution of primary mineral/metal production in relation to the national average water stress index (Figure 2.4).

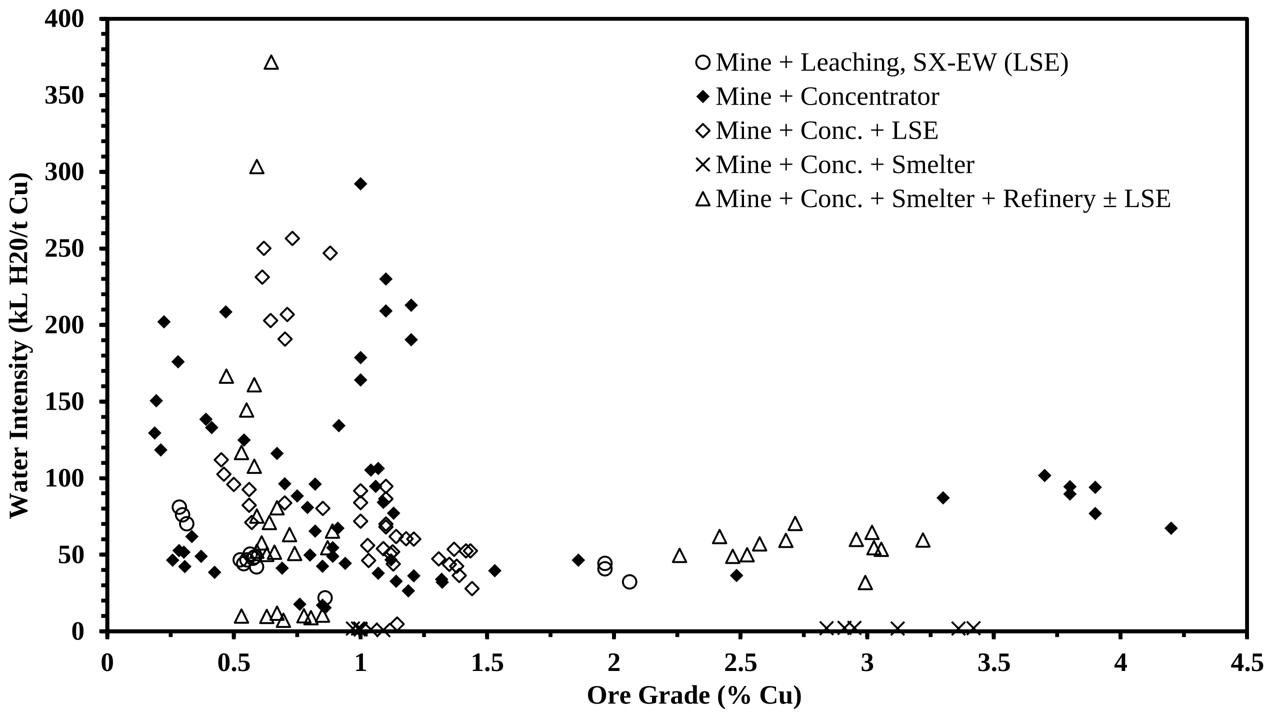

Figure 2.2: Water intensity as a function of ore grade for 31 copper producing operations (Northey et al., 2013).

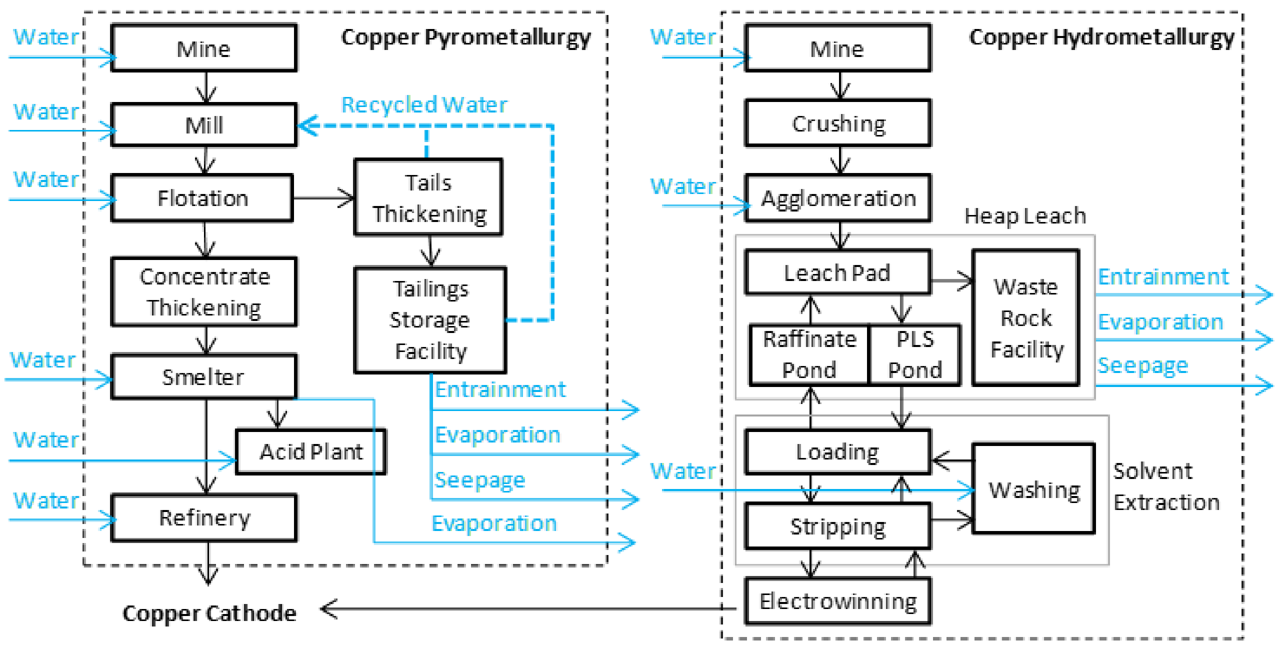

Figure 2.3: Example of the major water flows through the two dominant copper production routes (Northey and Haque, 2013; Northey et al., 2014a).

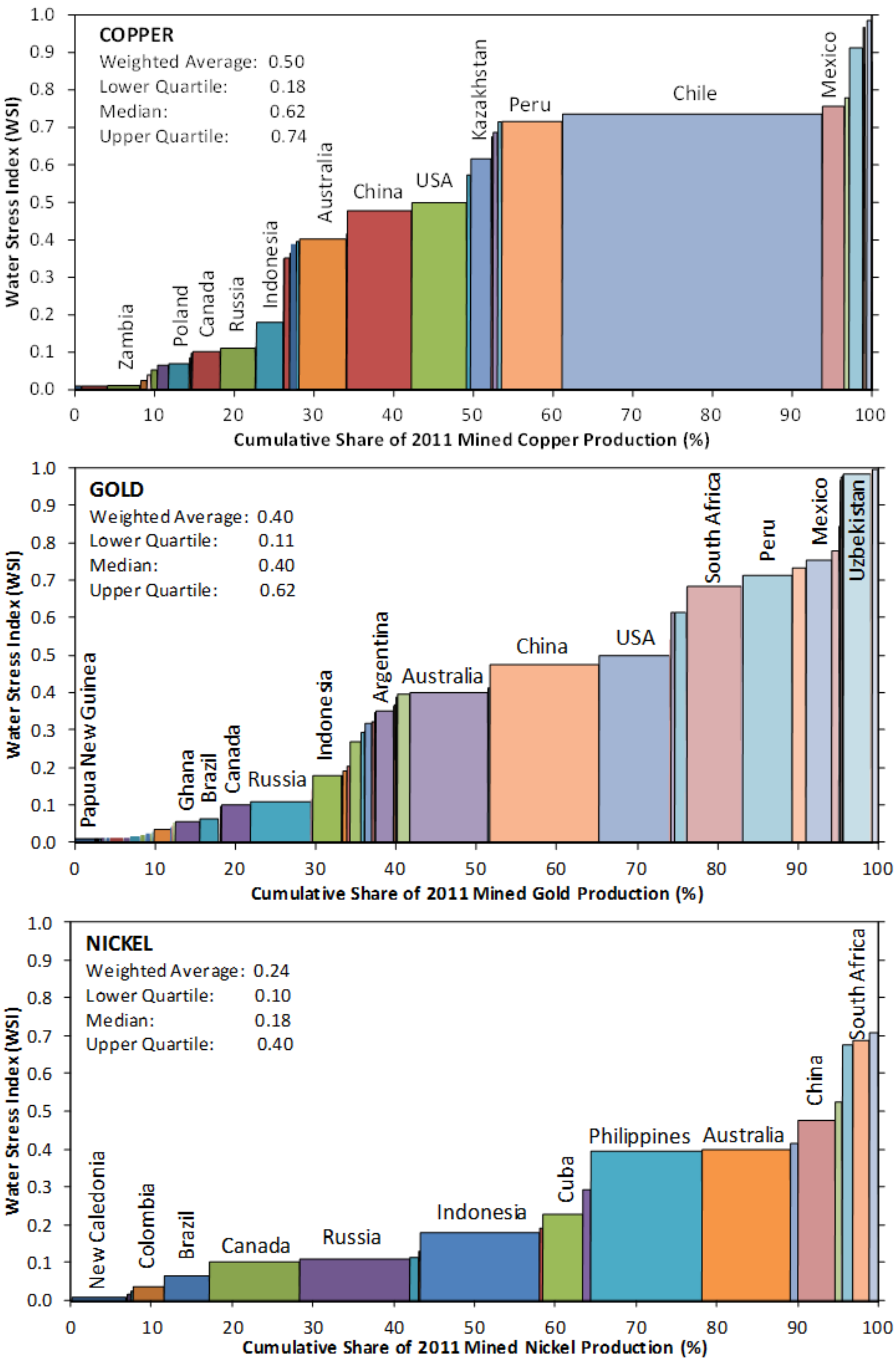

Figure 2.4: Water stress index of nations producing copper, gold and nickel in 2011 (Northey et al., 2014). Data sources: Pfister et al. (2009), USGS (2012).

An observation made by the author whilst conducting these was that the results of life cycle assessment studies focused on assessing embodied energy or carbon footprints of mineral and metal production are fairly reliable. However, the author observed that the life cycle inventory data available for assessing other impact categories, such as water use impacts, was far less representative of the broader mining industry – due to the significant variability in water consumption and water management practices of individual mining and mineral processing operations. Due to this, there is significant uncertainty inherent in the water use data that is present in existing life cycle inventory databases available for mined products. It was also observed that the hydrological models and data sources that underpinned the development of impact characterisation factors such as the

Water Stress Index (Pfister et al., 2009), may lead to development of national factors that are not reflective of conditions where mining occurs in each country. These observations were a major motivation for the pursuit of the doctoral studies and greatly informed the design of the research that comprise this doctoral thesis.

Further life cycle assessment studies that are focused on assessing consumptive water use of the mining industry are described in Chapter 3 (Northey et al., 2016), as part of an assessment of the existing application of 'water footprint' related methods to the mining industry.

2.4. 'Water Footprint' approaches

As briefly described in the introduction, the term 'water footprint' is a relatively recent concept that was first proposed by Hoekstra (2003). The meaning of the term has evolved slightly since this time due to the further development of two competing standards for assessing and communicate a 'water footprint'. Currently a water footprint can refer to either a measure of water consumption if developed according to the standards of the Water Footprint Network (Hoekstra et al., 2009a; 2011) or a measure of water use impact if conducted according to the ISO14046 standard for water footprinting (ISO, 2014). Studies based upon each of these approaches have been conducted for a wide range of industries and regions (Aivazidou et al., 2016), which are described and compared in the following sub-sections.

2.4.1. Water Footprint Network Standards

The Water Footprint Network[1] provided methodological guidance on ways to account for the water consumption associated with products, services, organisations and geographic regions. An initial guidance document – *Water Footprint Manual, State of the Art 2009* (Hoekstra et al., 2009a) – was developed to bring consistency in the terminology and approaches taken by studies. This was followed by a further guidance document – *The Water Footprint Assessment Manual: Setting the Global Standard* (Hoekstra et al., 2011) – that provided further methodological guidance and recommendations.

A Water Footprint Assessment conducted according to these standards will constitute several major stages:

1. **Goal and Scope Setting** to determine the intended purpose and goals of the study, as well as whether the assessment will consider 'blue', 'green' or 'grey' water, defining spatio-temporal boundaries of assessment, whether this will include direct and/or indirect water use, and also the truncation boundaries of indirect / supply chain analysis.

2. **Water Footprint Accounting** to develop a quantitative 'water footprint' estimate that reflects water consumption associated with the system given specific temporal and spatial boundaries.

3. **Water Footprint Sustainability Assessment** to evaluate the 'water footprint' estimate within local hydrological contexts (e.g. a river basin) and/or through consideration of regional water availability, competing users of water, or the broader contribution of the water footprint to water resources pressure (particularly for indirect/supply chain water footprints).

[1] On the 18th August 2017, the Water Footprint Network (WFN) filed for bankruptcy. Following this, the *Water Footprint Research Alliance*, which constitutes the major researchers and proponents of the WFN methodology, made a joint declaration that they would continue to develop, refine, promote and apply the methods developed by the Water Footprint Network (Hoekstra and Chapagain, 2017).

4. **Water Footprint Response Formulation** to determine the implications of the study and to identify potential technical solutions, management options or policy interventions that may be utilised to improve water management outcomes within the system or watershed.

According to the Water Footprint Network's approach to developing a 'water footprint' estimate, a water footprint may be the sum of three categories of consumptive water use, termed: 'blue' water, 'green' water and 'grey' water. The major water flows that contribute to these water use categorise are shown in Figure 2.5 and basic definitions are provided below:

Blue Water represents surface and groundwater flows or stocks of water;
Green Water represents the water contained in surface soil moisture and vegetation; and,
Grey Water represents the flow of water 'consumed' by water quality degradation.

Consumption of *blue, green and grey* water is assessed across a defined system boundary, which may constitute specific geographic and temporal boundaries. The water footprint estimates may consider the *direct* water consumption that occurs within the local boundaries of the product system, or else it may also consider the *indirect* or *virtual* water consumption that occurs within boundaries of supply chains or is embodied within the trade of goods and services.

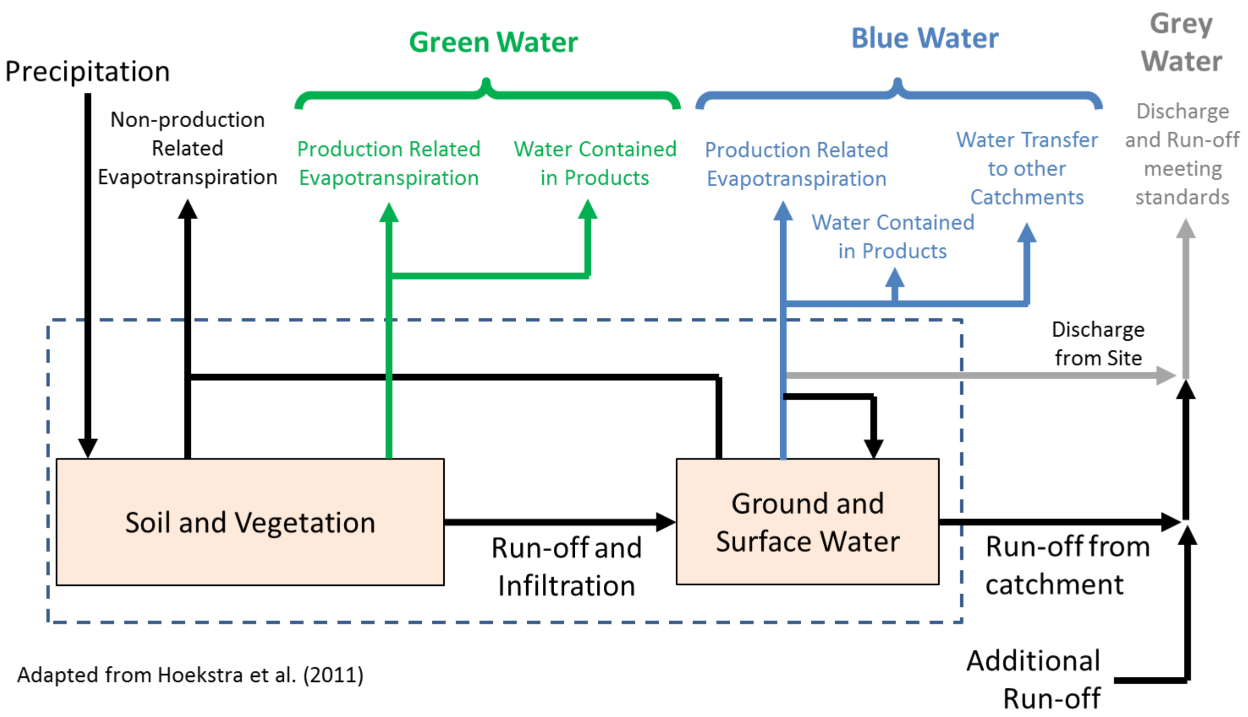

Figure 2.5: The relation of water flows and resources to the Water Footprint Network's definition of blue water, green water and grey water footprints. Adapted from Hoekstra et al. (2011).

2.4.2. ISO14046 – LCA based Water Footprints

An international standard, *ISO14046:2014 Environmental Management -- Water Footprint -- Principles, requirements and guidelines* (ISO, 2014), was developed to ensure consistent approaches and communication are taken to water footprint studies. The standard aligns the assessment of water footprints with the pre-existing international standard for life cycle assessment, ISO14044 (ISO, 2006). The major phases of a water footprint study conducted according to the ISO14046 standard are:

1. A **water footprint inventory analysis** is the compilation and manipulation of the volumetric water use data required for a study. Detailed inventories are developed that describe the water flows into and out of individual processes, and associated information such as the location of water withdrawals, time periods considered, and relevant quality and source parameters. Inventory methods differ based upon how various water flows are to be defined. Some methods require differentiation between individual sources of water (e.g. surface water, renewable groundwater, fossil groundwater) or between various categories of water quality (e.g. potable water, irrigation quality).

2. **Water footprint impact assessment** is then used to convert the inventory data into an assessment of the actual impacts related to water use. The data for water flows and quality developed as part of the inventory analysis are converted into quantitative estimates of impacts using defined *impact characterisation procedures*. Currently, methods most commonly assess user deprivation as a result of consumption of water in regions of various different water scarcity and stress. Alternative methods used in life cycle assessment, such estimates of eutrophication impact or eco-toxicity may be incorporated at this stage depending upon the goals of the study. Where multiple impact categories are assessed, then normalisation and weighting procedures may be used to aid in the interpretation and communication of the results.

3. **Water footprint interpretation** can be considered to be the most important part of the overall assessment. As the strengths and weaknesses of the methodology used in the particular study are described, and the implications for decision making and water resources management are discussed.

A major contribution of the ISO14046 standard was the guidance provided for communication requirements and specific terminology that should be used when describing the results of a study. One requirement is that qualifying statements should be added to the term ‘water footprint’ to ensure that the underlying meaning of the evaluation is better understood. The specific guidelines for terminology defined by ISO14046 that are most relevant to this thesis are that:

- The use of the term ‘Water footprint’ requires that both consumptive and degradative water impacts have been assessed.

- A ‘water scarcity footprint’ is an assessment of just consumptive water use impacts.

The ISO14046 standard provides flexibility in the approaches taken to inventory development and impact assessment, and so a range of methodological approaches could potentially be adopted whilst still complying with the standard. A diverse range of life cycle inventory and impact characterisation approaches have been developed to evaluate water use impacts over the past decade that could be used as part of an ISO14046 based water footprint evaluation (Bayart et al., 2010; Boulay et al., 2015a; 2017; Kounina et al., 2013; Quinteiro et al., 2017).

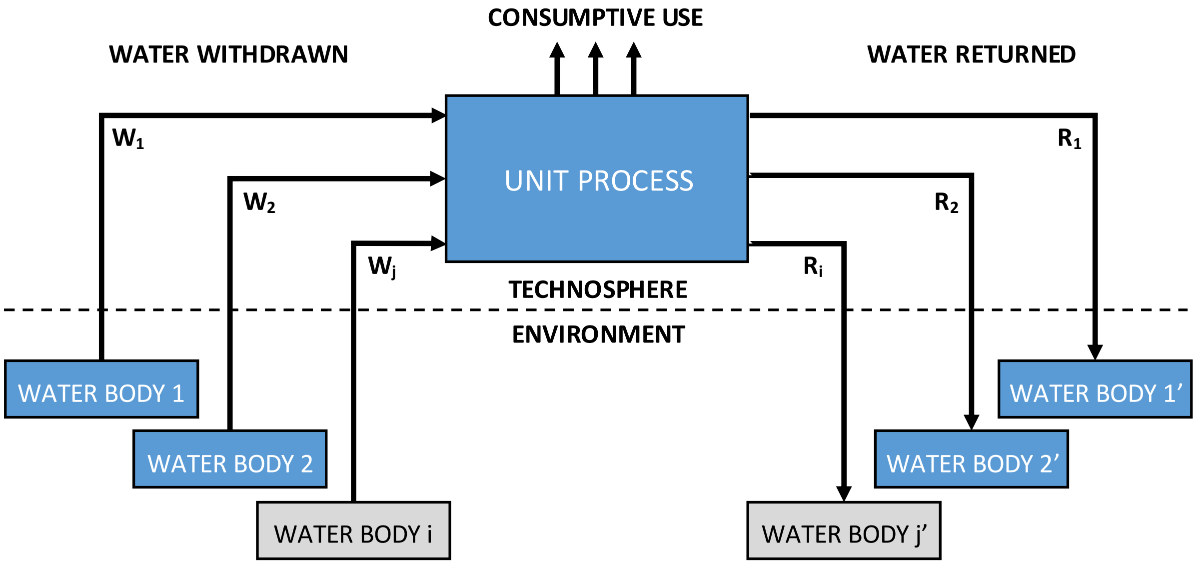

Figure 2.6: A general process diagram for accounting for water flows in life cycle assessment. Redrawn from Bayart et al. (2014).

A simple approach to determining an ISO14046 water scarcity footprint could simply be to take the difference between the water withdrawn and the water withdrawn from individual water bodies (Figure 2.6), adjusted for the local water scarcity or stress of each water body (Equation 1). This can be extended to also include degradative water use by considering changes to water quality also, which is the 'Water Impact Index' proposed by Bayart et al., (2014) that is described by Equation 2.

Equation 1: $WF = \sum_i W_i.S_i - \sum_j R_j.S_j$

Equation 2: Water Impact Index = $\sum_i W_i.Q_{W_i}.S_i - \sum_j R_j.Q_{R_j}.S_j$

Where: W is the quantity of water withdrawn from water body i
R is the quantity of water returned to water body j
S is a water scarcity or stress index (e.g. WSI, AWaRe, etc.)
Q is a water quality index

Another single indicator approach was proposed by Ridoutt and Pfister (2013b) that combines both consumptive and degradative water use (Equation 3), expressed in units of H_2Oequivalent that represents the average burden on water systems from consumptive freshwater use at the global average water stress index. Consumptive water use for each watershed or region is determined by multiplying the water consumption by the ratio of the local water stress index (defined by Pfister et al., 2009) and the global consumption weighted average water stress index (
Equation 4). In this case, degradative water use is defined based upon the outputs of the ReCiPe life cycle impact assessment method (Goedkoop et al., 2009), according to Equation 5.

Equation 3: $WF(H_2Oeq) = CWU + DWU$

Equation 4: $CWU\ (H_2Oeq) = \sum_i \frac{CWU_i.WSI_i}{WSI_{global}}$

Equation 5: $DWU\ (H_2Oeq) = \frac{ReCiPe\ points\ (emissions\ to\ water)}{ReCIPe\ points\ (global\ average\ for\ freshwater\ consumption)}$

Where: CWU_i is consumptive water use in watershed i
WSI_i is the water stress index in watershed i (Pfister et al., 2009)
WSI_{global} is the global average water stress index of 0.602 (Ridoutt and Pfister, 2013)

ReCiPe points (emissions to water) is the sum of end-point impacts for the product system, normalised using European factors and weighted using the 'hierarchist' cultural perspective (Goedkoop et al., 2009)

ReCiPe points (global average for freshwater consumption) is the sum of end-point impacts associated with the consumption of 1L of CWU, normalised using European factors and weighted using the 'hierarchist' cultural perspective (Goedkoop et al., 2009); 1.86 x 10^{-6} ReCiPe points (Ridoutt and Pfister, 2013b).

2.4.3. Comparison of Approaches

The two competing approaches to assessing a 'water footprint' differ in terms of their conceptualisation of what this actually represents (Boulay et al., 2013). The methodological framework for each of these approaches has the same major steps, just with differing terminology being used to describe these. The most obvious difference between the methods is the stage at which a water footprint is defined. The approach taken by the Water Footprint Network defines a water footprint as a volumetric measure of water consumption (Hoekstra et al., 2009a; 2011), whereas the ISO14046 approach defines a water footprint as a measure of the *impacts* of water use (ISO, 2014).

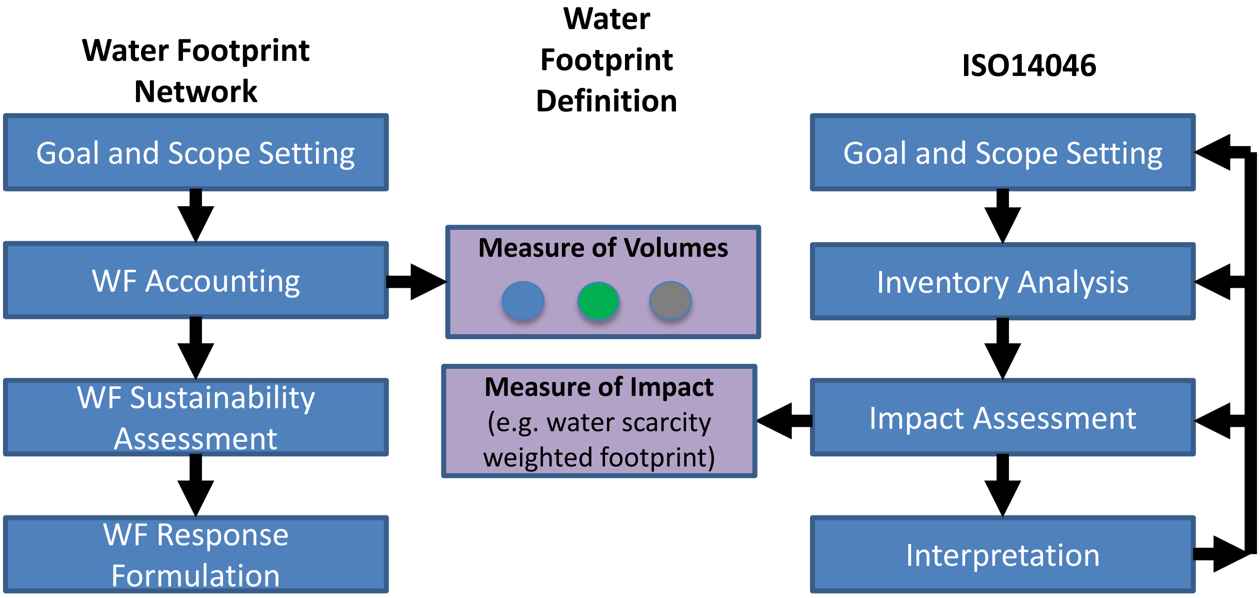

Figure 2.7: Comparison of the definition of a 'Water Footprint' according to the Water Footprint Network's standards (Hoekstra et al., 2009a; 2011) and the ISO14046 standard (ISO, 2014). Adapted from Boulay et al. (2013).

The Water Footprint Network provides a more strictly defined and standardised evaluation process that is potentially more straightforward for consistent interpretation by stakeholders. By comparison, the ISO14046 standard provides greater flexibility in terms of the approach taken to assessing water consumption and impacts. This allows for greater flexibility for scientific studies, while also ensuring consistency and interoperability with the full-range of life cycle impact assessment methods that are continually becoming more sophisticated and scientifically rigorous overtime.

Life cycle assessment researchers has advocated that the definition of a 'water footprint' should not be defined strictly as the volume of water consumption, but rather it should be defined as a measure of impact in the same way as other 'footprints', such as the 'carbon footprint' (Pfister and Hellweg, 2009). As the consideration of only volumes of consumption may lead to perverse or misleading study outcomes if the local conditions of where this water is consumed are not considered (Berger and Finkbeiner, 2013). Alternatively, advocates for the Water Footprint Network approach have argued that life cycle assessment based approaches are not as useful for water resources management as often these studies present aggregated impact indicators or results (Hoekstra et al., 2009b). Or that existing life cycle impact assessment methods are poorly conceived or do not address concepts such as ‘green’ water adequately (Hoekstra, 2016). Or that life cycle assessment studies are purely product focused (Hoekstra, 2017). However, it has been demonstrated that these criticisms are based upon a range misunderstandings of how water use and the potential impacts associated with this are considered within existing life cycle assessment methodology (Pfister et al., 2017).

Increasingly ‘footprint’ based methods are being developed as ways to communicate information on specific environmental issues, such as climate change issues through the carbon footprint, freshwater appropriation through the water footprint, material use through the material footprint, and land productivity requirements through the ecological footprint. (Fang and Heijungs, 2015; Ridoutt and Pfister, 2013a). ‘Footprints’ can be developed to provide as an indicator of either inventory flows or impact characterisation. The most commonly known footprint, the carbon footprint, is sometimes misinterpreted as an inventory based indicator – for instance by Hoekstra (2016) – however it is actually the result of impact characterisation, as the inventory flows of greenhouse gases are multiplied by characterisation factors that represent their expected radiative forcing over a specific time period (typically 100 years), relative to the expected radiative forcing of the equivalent unit of carbon dioxide. In this way, carbon footprint results expressed in terms of kg CO_2-equivalent are not actually a measure of mass, but rather a measure of expected radiative forcing modelled over a defined time horizon. In the same way, the results of ISO14046/LCA based water scarcity footprints are often expressed in units of litres of H_2O-equivalent that may not represent an actual volume of water, but rather the expected impacts of water use expressed relative to the impacts of water use in a particular region or globally. Due to this, sometimes it is suggested in the literature (i.e. Hoektra, 2016) that these lack a physical meaning – when in-fact the underpinning logic is similar to the basic underpinnings of many existing ‘footprinting’ methods (Fang and Heijungs, 2015). Younger (2006) observed that the term itself, ‘water footprint’, implies an aerial impact and so proposed a method for evaluating the water footprint of mining operations in terms of equivalent catchment areas. However, to the author’s knowledge the proposal of Younger (2006) has seen no further adoption.

It is also useful to consider the intended purpose of these approaches. The author’s perspective is that water footprinting and life cycle assessment studies are best suited for the analysis and comparison of complex distributed systems, such as supply chains or commodity production distributed across multiple regions or production facilities. It has been recognised that these approaches have more limited usefulness for assessing the local impacts of individual production facilities, as traditional environmental impact assessment and local catchment management approaches are able to provide a more sophisticated understanding in these cases (Chenoweth et al., 2014). Other approaches such as ‘Water Stewardship Schemes’ may be preferable for managing water use in individual hydrological catchments or basins.

So despite perceived differences between the two approaches, the underlying methodology and principles are quite similar. The meaning differences are simply that the Water Footprint Network approach was developed independently, with slightly different terminology, a more prescribed inventory and (optional) impact assessment requirements. Quinteiro et al. (2017) provides an excellent overview of the similarities and differences of the two approaches. The reader is also referred to the work of McAuliff et al. (2017), which provides a decision tree approach to selecting the appropriate water footprinting methodology for a given study.

3. Water Footprinting and Mining: Where are the limitations and opportunities?

‘Water footprint’ and life cycle assessment based approaches for evaluating consumptive water use impacts have developed significantly over the past decade. Most of this methodological development has been undertaken and applied to the major water consuming industries. Consequently, there is a need to understand the extent that existing methods have been applied to the minerals industry, and whether existing methods and data sources are adequate for comprehensive studies of mining and mineral processing.

An overview of water use at mining and mineral processing operations is provided that highlights the significant variability and diversity of water management practices, risks and potential impacts to water resources from the mining industry. A brief overview of how water consumption is accounted for and evaluated in life cycle assessment and ISO14046 based water footprint assessments was provided, as well as a brief assessment of how previous studies have applied these methods to the mining and mineral processing industry.

Six areas of conceptual or practical limitations are identified that have hindered or prevented the application of these methods to the mining and mineral processing industry. It is explained that these actually represent significant areas for further research to improve our understanding of the mining industry’s water use impacts. Additionally three explicit areas of opportunity are identified for further use of life cycle assessment based water footprinting methodology, in the areas of: improving the communication of water use data, understanding the impacts of technology implementation, and benchmarking the water use efficiency of mining or mineral processing operations.

The contents of this chapter was published in the Journal of Cleaner Production and are presented in the original format of the journal. The conceptual foundations of the article was developed during a three week research stay at the VTT Technical Research Centre of Finland in Espoo and this was then extended upon after returning to Australia. The contributions of Elina Saarivuori and Helena Wessman-Jääskeläinen from VTT are gratefully acknowledged.

Reference:

Northey, S.A., Mudd, G.M., Saarivuori, E., Wessman-Jääskeläinen, H., Haque, N. (2016). Water footprinting and mining: Where are the limitations and opportunities?
Journal of Cleaner Production, 135, pp. 1098-1116.
http://dx.doi.org/10.1016/j.jclepro.2016.07.024

Journal of Cleaner Production 135 (2016) 1098–1116

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

Review

Water footprinting and mining: Where are the limitations and opportunities?

Stephen A. Northey [a, b, *], Gavin M. Mudd [a], Elina Saarivuori [c], Helena Wessman-Jääskeläinen [c], Nawshad Haque [b]

[a] Environmental Engineering, Monash University, Clayton, Victoria, Australia
[b] CSIRO Mineral Resources, Clayton, Victoria, Australia
[c] VTT Technical Research Centre of Finland Ltd, Espoo, Finland

ARTICLE INFO

Article history:
Received 24 May 2016
Received in revised form
4 July 2016
Accepted 4 July 2016
Available online 7 July 2016

Keywords:
Mining
Water footprint
Life cycle assessment
Water scarcity
Water quality
Mine water management

ABSTRACT

The interactions of the mining industry with water resources are highly complex and site specific, with potential impacts to both hydrology and water quality occurring at all stages of a mine's life. A range of water management approaches are employed by the industry to mitigate the risks of adverse water impacts occurring. Consequently, the significant variability within the industry poses a range of challenges when attempting to quantify the water footprint of mining operations and mineral commodities.

Methods for water footprinting have developed significantly over the past decade and have recently become aligned with life cycle assessment approaches. Despite these advances, relatively few studies have focused upon applying these methods within the mining and mineral processing industry. A range of limitations were identified that hinder the ability to conduct these types of studies. These limitations include: the availability of mine site water use data, inventory data for mining supply chains, the uncertainty of post-closure impacts, and the difficulty of accounting for cumulative impacts and extreme events (e.g. flooding, dam failures, etc.). The spatial resolution and data underpinnings of current water footprint impact characterisation factors also limits the ability to interpret results that may be generated. Overcoming these limitations, through methodological development and data collection efforts, represents a significant opportunity to improve our understanding of the mining industry's water use and impacts.

Beyond this, several key opportunities for more widespread use of mine site water footprint assessments were identified, including: to aid the benchmarking of water performance in the mining industry, to improve the quality of cross-sectoral assessments of water use, to assess the indirect impacts of competing technologies, and to provide improved water use disclosures within corporate sustainability reports.

Contents

* Corresponding author. Environmental Engineering, Monash University, Clayton, Victoria, Australia.
E-mail addresses: stephen.northey@monash.edu, stephen.northey@csiro.au, stephen.northey@gmail.com (S.A. Northey).

http://dx.doi.org/10.1016/j.jclepro.2016.07.024

1. Introduction

Approximately 1.7×10^9 people live in regions where groundwater is being overexploited (Gleeson et al., 2012) and an estimated 4×10^9 people live in regions that are exposed to water scarcity for at least 1 month per year (Mekonnen and Hoekstra, 2016). Over the past two decades, these types of pressures have led to the development of approaches to quantify the *water footprint* of regions, products and processes. Earlier approaches had a particular focus on measuring volumes of water consumption (e.g. Chapagain and Hoekstra, 2004); however, more recent methods place greater emphasis on how to relate this water consumption to the potential for impact to end-users and environments. A significant step along this path has been the development of the ISO 14046 standard for water footprinting (ISO, 2014), which has more explicitly aligned water footprinting within the framework of life cycle assessment methodology.

The role of water footprinting differs from individual site based environmental impact assessments, as the methods are not necessarily tailored to understand the *absolute* impact associated with any individual processing facility; rather the methods are tailored to understand the *relative* potential for impact between process facilities and across supply chains. Due to this, results developed using water footprint methods and life cycle assessment may not necessarily be representative of what is actually happening on the ground, particularly when uncertainty related with impact calculation procedures are combined with the current limitations and availability of water use data.

Mining could be considered one of the most diverse industries with respect to how it interacts with water resources (Younger et al., 2002). Mining occurs across the full spectrum of hydrological contexts; from the arid regions of central Australia through the tropics and to the sub-arctic conditions of Canada and Finland. The local climate and hydrology dictates infrastructure requirements at mining operations and has a profound influence on the nature of water related risks faced by mines and nearby communities, ecosystems and industry. Examples of these risks include uncertainty over access to a stable water supply, the potential for flooding of open pits, uncontrolled discharges and catastrophic collapses of waste impoundments. Water quality risks associated with mining can also be viewed quite differently to other industries such as agriculture, as the risks associated with a particular mine are heavily dependent upon a combination of factors, such as: the geochemistry of the ore body, the strategies for managing mine discharge, the types of mining utilised, the processes used to separate valuable minerals from ore, and the approach taken for storage of large mine wastes.

Despite mining being a relatively small consumer of water on a global scale, in the regions where mining does occur it can often represent a major local consumer of water. The impacts of the industry's water consumption, in conjunction with the potential for significant water quality impacts, can lead to social tension with other water user groups such as fisheries (Holley and Mitcham, 2016), agriculture, communities (Ghorbani and Kuan, 2016; Kemp et al., 2010) or tourism (Wessman et al., 2014). As a response, it is increasingly being recognised that mining operations must develop and maintain a social license to operate and, as part of this, that local water quality should be protected at all stages of a mines life (e.g. Caron et al., 2016).

The ability of water footprinting to contribute to our understanding of water usage, impacts and risks across the mining industry will be addressed in this article. The major aims of this article are to:

1. Provide a broad overview of mining's interactions with water resources;
2. Briefly summarise the current state of water footprinting methodology and determine to what extent this has been applied in studies of mined products;
3. Identify the current limitations that need to be overcome to improve water footprints estimates of mining operations and mined products; and,
4. Identify the opportunities for applying water footprint methods in the mining industry.

This article was developed as part of collaboration between the VTT Technical Research Centre of Finland and Australia's Commonwealth Scientific and Industrial Research Organisation (CSIRO) – and so the mine sites and water management approaches highlighted have a particular focus upon these respective countries. However, the issues discussed are broadly applicable to the global mining industry and the hydrological context of different regions.

2. Water use and management at mining and mineral processing operations

2.1. Overview of mine-site water flows and infrastructure

The development of a mining operation can significantly alter the natural flows and quality of water in a region (Younger et al., 2002). Quantifying the exact nature of these impacts is difficult given the significant complexity in how water flows between major mine-site site infrastructure and surrounding environments. However, an understanding of these flows and the underlying variables that affect them is essential to be able to adequately assess the potential water consumption and water quality alterations that may occur as a result of mining. Fig. 1 provides a simplified diagram of water flows between a mining operation and the surrounding environment, as well as some of the common infrastructure at mine sites that influence the magnitude and quality of these flows. To provide a sense of scale for context, Fig. 2 highlights the key infrastructure at the large Cowal gold mine in central New South Wales, Australia.

The processes used to actually mine ore require little water, with only a modest amount being used for activities such as dust suppression, fire control or cooling of equipment. However, the large-scale movement of material and the development of mining voids can result in significant alterations to natural water flow pathways through an area. Infiltration of groundwater into mine voids can be significant and so active dewatering of aquifers is often required to lower the surrounding groundwater table and reduce infiltration rates. Water that does infiltrate into underground mines will need to be pumped to the surface to avoid flooding of mine shafts and voids. For surface mines, the combination of groundwater infiltration, rainfall and runoff can lead to flooding of open pits or the formation of pit lakes. Removal of this water may require active pumping, particularly when a mine is located in a region with low evaporation rates or high rainfall. Diversion channels or barriers may also be constructed to prevent excessive runoff from entering the mine.

Water use within the mineral processing facilities depends greatly upon how the valuable minerals or metals are recovered from ore. These approaches can broadly be categorised into wet or dry processing. Dry processing techniques, such as the use of air cyclones, ore sorting, dry magnetic separation or screening, are usually only used in niche applications (e.g. the mineral sands industry) due to a range of limitations such as dust generation, lower recovery efficiency and typically low-throughput rates (Napier-Munn and Morrison, 2003). Currently wet ore processing strategies are much more widespread, with common techniques being flotation, leaching, gravity separation, electrowinning and solvent extraction. Some of the advantages of wet processing include: higher recovery efficiency and throughput rates, easier transportation of solids between processes, and the ability to utilise the chemical properties of minerals when performing separations. The amount of material being processed and the solids density required for individual processes determine the upper limits of water inputs for process plants. Most of this water ends up in either the product or tailings (residual process waste) streams, and so recovery and recycling of this water via thickening and filtration processes provides one of the main opportunities to reduce the external water required for mineral processing (Gunson et al., 2012). The degree that water is recovered from tailings ultimately depends on the intended approach taken for long-term storage and management of this waste.

The approach taken for management and storage of tailings material will heavily influence the water balance of the operation due to differences in the amount of water lost through evaporation, seepage, discharge, or physical entrainment in the tailings material (Vick, 1990). Tailings may ultimately be stored in dams, backfilled into mines, dry-stacked (Davies, 2011), or in some cases discharged to rivers or marine environments (Ramirez-Llodra et al., 2015). Of these approaches, the storage of tailings in dam structures or natural depressions near the mine site is the most common approach globally. These storage facilities may require wet covers to prevent weathering of tailings material. Alternatively, the wetted area may be actively minimised through tailings placement strategies to improve dam stability or reduce evaporative losses of water.

For some ore bodies, valuable metals may be extracted through the use of leaching processes. This involves stacking of ore into heaps that are then irrigated with a leach solution, which is collected at the base of the heap in containment ponds. Depending upon the target metals and the mineralogy of the ore, chemicals such as sulphuric acid or sodium cyanide may be added to the leach solution. Often the ore will be lightly crushed and agglomerated prior to placement to increase the permeability, stability and porosity of the heap (Bouffard, 2005). Several different irrigation

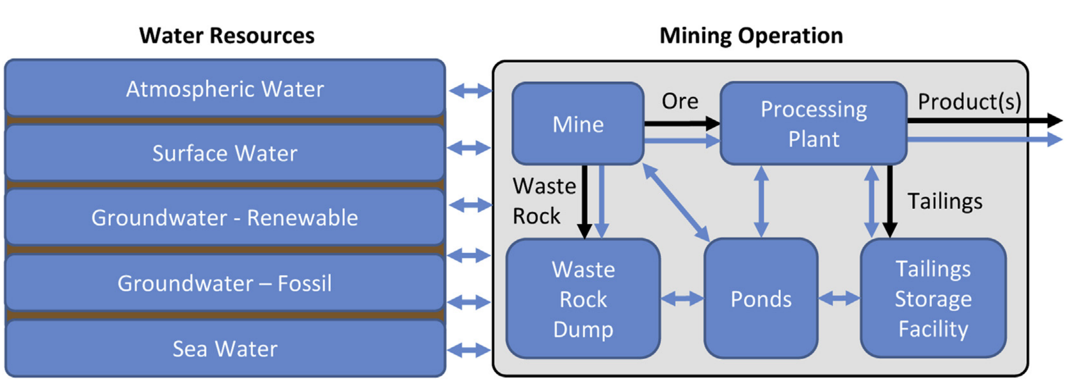

Fig. 1. Hypothetical flows of water on a mine-site. Flow pathways vary from site-to-site and may also be seasonal or change through the life of a mine. Blue arrows indicate flows of water, whereas black arrows indicate flows of solid material. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

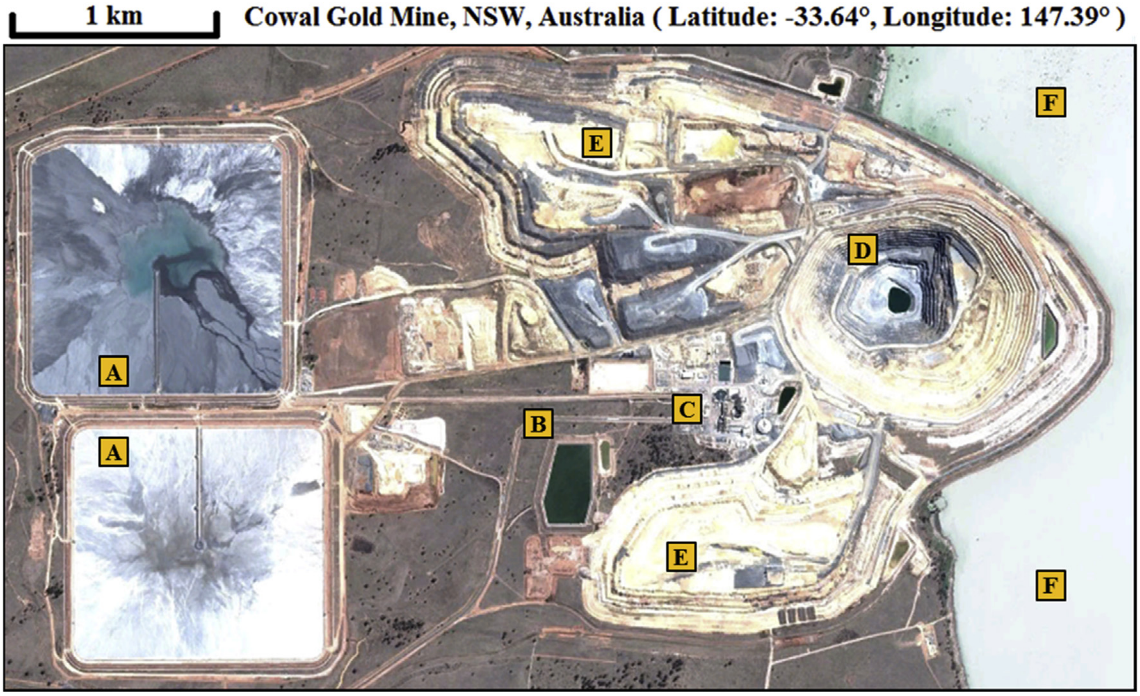

Fig. 2. Satellite image of Cowal Gold Mine located adjacent to Lake Cowal, New South Wales, Australia (Google Earth, 2016). Key points of interest are shown by the letter annotations: A – Tailings Storage Facilities, B – Water Storage, C – Ore Processing Facilities, D – Open Pit Mine, E – Waste Rock Dumps, F - Lake Cowal.

methods may be used (e.g. sub-surface irrigation, sprays, etc.) and this will influence evaporative losses of water. Evaporation can also be increased when the temperature of the heap is elevated due to exothermic reactions occurring during the bacterial or chemical decomposition of the ore. Alternatively, leaching may be performed in-situ (i.e. without excavating the ore) through the use of injection and withdrawal wells. In-situ leaching is only used occasionally within the mining industry, with some examples existing for the production of uranium (Mudd, 2001a, 2001b), copper (Sinclair and Thompson, 2015) and rare earth elements (Yang et al., 2013). The main drawbacks of in-situ leaching include lower product recovery rates and the contamination of groundwater systems.

Mining operations may have dedicated ponds and dams for storing raw water, recycled process water, seepage containment, evaporation ponds, and fire water. The water balance of these stores depend upon the local weather (particularly rates of pan-evaporation and rainfall), the presence of clay or membrane linings to reduce seepage, the use of diversion channels to promote or prevent the capture of run-off, and the transfer of water between ponds and other site processes. When conservation of water is a concern, mines are increasingly seeking to reduce evaporation rates from water stores through the use of covers, shading, or even floating solar panels.

Given the complexity of mining operations, many sites have a poor understanding of how water flows through their operation and the associated impacts to water quality. Even when attempts have been made to map and quantify these flows, there can remain significant uncertainty in the exact magnitude of these flows and how they relate to weather and climatic events. For instance evaporation from underground mines is often poorly quantified and may require long-term baseline data to accurately assess (Rapantova and Grmela, 2000). Water accounting systems have been developed to address these issues by providing guidance on how to account for the quantity, quality, source and uncertainty of water flows between site processes and external water resources (MCA, 2014).

2.2. Major sources of water related risks at mine sites and associated management approaches

Mine-sites need to manage a variety of water related risks throughout the life of the operation. The outcomes of these risks can be manifested internally to the operation, such as through the suspension of production due to insufficient water supply. Alternatively these risks may manifest externally, through the potential for adverse impacts to surrounding ecosystems, communities and industry.

2.2.1. Water balance and hydrological risks

An important consideration when assessing water risks at mine sites is whether the overall water balance of the site is water positive or water negative. A mining operation can be considered water positive during periods of high rainfall and low evaporation, as water will accumulate on-site in dams, ponds and mine voids. Alternatively, a mining operation is water negative when evaporation rates are sufficient to prevent this accumulation of water. Water positive operations will require discharge once on-site water storage has reached capacity, whereas a water negative operation may avoid this.

Variability in local hydrology can mean that a mine site may shift between being water positive or negative. These changes can be seasonal and may be more pronounced during periods of prolonged dryness or wetness (e.g. droughts and floods). Mines that operate in regions with large hydrological variability have to manage their water in a way that accounts for a range of different trade-offs, such as the balance between ensuring sufficient water supplies are available in case of drought conditions, whilst also managing the risk of having too much water stored in times of floods, heavy rainfall, or melting snow – situations that may cause sudden changes in the water balance of a mine site, and require discharge of contaminated or poor quality water to nearby water bodies. These risks may be managed through a variety of approaches that range from investment in additional water storage infrastructure to changes in operating practices of the water pumping and distribution systems. Given the complexity of mine-site water management it has been suggested that systems modelling approaches may be appropriate to develop the optimal management solution for a given context (Kunz and Moran, 2016).

The withdrawal of water from surrounding water resources to supply mineral processing facilities, or as part of dewatering processes, carries a risk of adverse impacts. Excessive water withdrawals may lead to impacts such as the alteration of river flow regimes, drawdown of groundwater aquifer levels, or the reversal of groundwater flow directions. These risks are very site specific but are an important consideration when authorities and regulators assess the location and volume of allowable mine site water withdrawals.

Similar risks also exist when discharging water to surface water or reinjecting water into groundwater systems. Particularly when these discharges occur alongside abnormally high flow conditions, which may be required in some circumstances to dilute pollutants that are present in the discharge waters.

In the context of enhancing groundwater protection and risk management, the Finnish Environment Institute developed a Groundwater Studies Checklist for mining companies, government agencies and consultants (SYKE, 2015; Tuominen et al., 2016). The tool helps in the acquisition and compilation of groundwater data, and to find suitable research methods by listing all the relevant groundwater information needed by mining operations. The checklist includes information on the hydrogeological structure, interactions of groundwater and surface water, management of groundwater flow patterns and transport of contaminants. The Groundwater Studies Checklist can be employed at all stages of a mines life to improve understanding and risk management outcomes (SYKE, 2015; Tuominen et al., 2016).

2.2.2. *Water quality risks*

The water quality risks associated with mining are highly dependent upon the mineralogy and geochemistry of the ore body being mined. The exposure of mined material to oxidation and weathering processes may result in the mobilisation of potential pollutants in the seepage waters of waste rock dumps, tailings storage facilities and the walls of mine voids. The saline, metalliferous and/or acid rock drainage that occurs can be a major source of water pollution. The processes that generate water quality impacts may be transient due to first flush effects, however in some cases they will be long lived and could extend well beyond the closure of the mine (Younger, 1997). The main management strategies include preventing oxidative conditions, reducing rates of seepage flow, intercepting seepage water, buffering with neutralising agents, or releasing to water courses during periods of high flows (Akcil and Koldas, 2006; Johnson and Hallberg, 2005). It has also been suggested that moving from end-of pipe solutions towards preventative measures, such as the selective separation of acid forming minerals may be a preferred approach to managing these types of water quality risks (Edraki et al., 2014).

Management of potential water quality impacts will, almost by definition, be very site specific and need to account for local conditions. For example, acid drainage can often be neutralised using limestone, however this will increase total dissolved solids (TDS) loading of the water, which may have adverse consequences on receiving waters in some contexts (Sarver and Cox, 2013). The generation of impacts associated with high TDS water concentrations can occur at all stages of a mines life (Sarver and Cox, 2013). Approaches to managing TDS risks are varied, but generally rely on long-term monitoring, preventative measures (e.g. selective mining and storage, diversion of surface water), and the mitigation of impacts (e.g. dilution in receiving waters, desalination processes, etc.). An interesting approach to TDS management is the Hunter River Salinity Trading Scheme (HRST), which allows coal mines and power utilities to purchase and trade the right to discharge saline water (Franks et al., 2010). This can be contrasted with other types of *polluter pays* regulation, which may be ineffective when water treatment costs exceed the cost of pollutant discharge fees (Sarver and Cox, 2013).

As the mineralogy and geochemistry of ore deposits is non-homogenous, water quality risks may vary as different sections are mined, processed and disposed of. An example, would be the mining of porphyry copper deposits that contain an oxidised “cap”, which typically has low potential for acid formation. However as this is mined past, the underlying geology would likely be more sulfidic and have larger potential for acid formation when exposed to oxygen and weathering processes.

Discharges of water from mine sites are regulated in most countries and typically there are limits placed on both pollutant concentrations and total loads in discharged waters. Often discharges will preferentially occur during periods of high flows, so that dilution of pollutants in the receiving water is maximised. Depending upon the conditions of the receiving water, sedimentation of pollutants may occur – in essence ‘trapping’ these pollutants. Longer-term there may be potential for remobilisation of these accumulated pollutants if the conditions of the water course change. In addition, cold climates can limit pollutant degradation, as ice cover can lead to oxygen depletion in aquatic systems and subsequently inhibit beneficial chemical reactions (Kauppi, 2013).

2.2.3. *Tailings dam failures and major pollution events*

Globally, tailings dam failures occur with unfortunate regularity. A review of 100 years of tailings dam failures has shown the failure rate to be 1.2%, compared with a failure rate of only 0.01% for more conventional water storage dams (Azam and Li, 2010). Some more recent high profile examples include the Samarco (Brazil) and Mt Polley (Canada) dam failures. Causes and mechanisms for dam failure include: piping, overtopping, foundation issues, slope instability, structural defects and improper management. The method of dam construction will influence the likelihood of these events occurring, with upstream dam raises being more problematic than the typically more expensive downstream method.

Perhaps the only silver linings of mine pollution events is the following scrutiny and motivation of the mining industry to improve operating practices. An example of this is the International Cyanide Management Code, which was developed as part of the response to the accidental release of 100 ML of cyanide containing tailings solutions from the Aural gold mine in Baia Mare, Romania, that occurred in 2000 (Gibbons, 2005). The development of this code is one of the few examples of widespread mobilisation of industry stakeholders around a water quality issue.

2.2.4. *Rehabilitation and post-closure risks*

Many of the water impacts of mining may occur post-closure, following the abandonment or rehabilitation of the mine site. Rehabilitation efforts will vary from site-to-site, but can involve contouring of surfaces, partial back-filling of mine pits and voids, capping of hazardous waste dumps (especially with respect to acidic drainage risks from sulfidic mine wastes), and revegetation of the site. The success of site rehabilitation can vary significantly, especially in the long-term, and some cases may require on-going management in perpetuity (Kempton et al., 2010). Despite rehabilitation efforts, the long-term hydrologic profile of the mine site is likely to be significantly altered from the predevelopment state. The formation of pit-lakes may result in long-term groundwater drawdown due to evaporation (Eary and Watson, 2009); revegetation efforts may alter evapotranspiration rates due to differences in species composition and maturity; acid and metalliferous drainage from waste rock dumps, mine surfaces and tailings storage facilities may migrate through groundwater systems; and also, runoff pathways may be altered due to differences in site topography. Some of these risks are under-recognised by the industry – such as post-closure water consumption, which was essentially disregarded during discussions of the Mining, Minerals and Sustainable Development (MMSD) project (Amezaga et al., 2011).

The long-term risks of mining are intimately linked to the management of large-waste material and mine voids. Franks et al. (2011) defined a series of sustainable development policies for the management of mine wastes. Preventative approaches such as selective mining and separation of acid forming minerals, covering waste piles, diverting water around these and collecting seepage also have a role to play (Edraki et al., 2014). Other approaches to managing this waste, for instance, options to backfill mining voids can lead to a range of impacts on groundwater systems. Development of flowthrough or a terminal groundwater systems may depend upon the formation of pit-lakes and whether evaporation is sufficient to significantly alter groundwater flows (McCullough et al., 2013). Terminal pit lakes may be beneficial in some cases, as they can act as sinks to trap potential groundwater contaminants. However, they can also carry risks to human health if used for recreational purposes in the future (Hinwood et al., 2012). The management of pit-lakes should consider the range of potential end-uses and also the biological processes that may heavily influence long-term water quality (Lund and Blanchette, 2014).

2.2.5. Risk management approaches and guidelines

Considering the issues discussed in the preceding sections as being independent from each other would be a false dichotomy. There is a complex interplay between all the water risk management issues highlighted. Management of one form of impact may have synergistic impacts by also reducing alternative types of risk. Alternatively, there may be an antagonistic relationship when a risk management strategy increases other types of risks.

When considering the appropriate management response, it is important to consider the surrounding environments and regional context. The regulatory requirements for mining can vary significantly between countries and regions, so management approaches that are taken in one location may not be possible in another. The local climate and hydrology may allow or limit different management regimes. As will the presence of downstream communities, industry, or sensitive environmental assets – such as RAMSAR listed wetlands or world heritage areas.

A range of sustainable development initiatives, risk management systems and best practice guidelines have been developed to improve the performance of the mining industry. For instance there have been several best practice guidelines for mine site water management developed within Australia over the years (BPEMM, 1999; LPSDP, 2008). Similar resources have also been developed for related issues such as acid and metalliferous drainage (INAP, 2009), cyanide contaminated water, and tailings management. A recent example of an initiative to improve risk assessment in the mining industry is the Network for Sustainable Mining in Finland, which has approved the use of the Canadian Towards Sustainable Mining (TSM) standard for mines operating in Finland (TSM, 2016). The TSM has been complemented with additional protocols on water management and mine closure. Implementation of the standard will start during 2016.

As an attempt to expand the role of risk assessment in mining, the Finnish Ministry of Environment launched a program to "stress test" Finnish mines in 2013. Triggered by a water discharge incident at the Talvivaara mine in Finland in 2012, the voluntary test attempted to assess the capacity of the mine operators to tackle different types of pressures that lead to negative environmental impacts (Välisalo, 2014). The results of the stress test indicated that dam structure control, rapid dam rupture reparation, identification of harmful emissions, precautions against power failure and sabotage, and communication of incidents were well managed at the Finnish sites. Some areas of improvement were identified, especially related to the management of excess water. However, it is difficult to establish comparable metrics for stress tolerance of mines, due to unique production technologies and environmental issues at each site. According to Wessman et al. (2014), the stress test should promote continuous improvement at mine sites to prevent high risk events in the future.

When framing these mine water management issues, it is important to recognize that mine waters are also a potential resource that can be made available to other users through effective treatment and storage – a potential shared benefit of mining (Schultze, 2012). As an example, water contained in pit-lakes may be useful to supplement water supplies in some developing countries (Soni and Wolkersdorfer, 2016). However, water quality varies considerably between pit-lakes and so careful consideration of the suitability for different end-uses is required (Kumar et al., 2009).

2.3. Examples highlighting the complex diversity of mine-site water interactions

Significant variability exists in the water management approaches and risks that occur at individual mine sites. To illustrate the diversity of situations, some examples that describe different types of interactions between mining, water and surrounding environments are provided in Table 1. Efforts of water footprinting to describe the impacts associated with producing mined products should ideally be able to account for the diversity of expected outcomes. Several useful compilations of water management case studies exist for mines in Europe (Wolkersdorfer and Bowell, 2004, 2005a; 2005b) and elsewhere (ICMM, 2012).

3. Water footprinting methods and data requirements

Due to the evolution of methods and terminology, there is currently some confusion amongst relevant stakeholders regarding what a water footprint actually measures. The methodology originally developed by the Water Footprint Network described a water footprint as a volumetric measure of water consumption according to three different water use categories (blue water, green water, and grey water) (Hoekstra et al., 2009, 2011). However, there are examples where this definition may lead to poor decision making when considering water used in regions of differing water stress or scarcity (Ridoutt and Huang, 2014). The life cycle assessment community has instead argued that a water footprint should measure the actual impact associated with this water use (Berger and Finkbeiner, 2013; Fang and Heijungs, 2015). The equating of green water and grey water volumes to blue water volumes has also been viewed as inconsistent (Pfister and Ridoutt, 2014). Both volumetric and impact based water footprints provide important information that is useful for informing different types of decisions (Boulay et al., 2013). However, defining a water footprint as a measure of impact provides greater alignment with other 'footprints', such as the carbon footprint (Ridoutt and Pfister, 2013a). This perspective has been reflected in the development of the international standard, *ISO 14046 Water footprint – Principles, requirements and guidelines*, which defines a water footprint as a "metric that quantifies the potential environmental impacts related to water" (ISO, 2014).

A water footprint assessment conducted according to the standard is based on a life cycle assessment, is a sum of the water

Table 1
Examples highlighting the complexity of water use, risks and impacts associated with mining operations.

Mine, country	Type	Product(s)	Climate	Areas of complexity	Description of water use, risks or impacts
Bowen Basin, Australia	OP	Coal	Semi-arid	D, F, HV	Widespread flooding of active and abandoned coal mines required unplanned and emergency water releases [1].
Cadia Valley, Australia	OP, UG	Au, Cu	Temperate	SD, HV	Drought conditions threatened site closure, requiring a 5 ML/day temporary withdrawal permit to be granted [2].
Gyama Valley, China	–	Cu, Pb, Zn	Semi-arid Alpine	CI, D, S, WQ	Mines discharging effluents have raised concentrations of Pb, Cr, Mo and Fe above safe drinking guidelines in some parts of the stream. Sedimentation processes have mitigated impacts to some degree, however there is a risk of pollutant remobilisation due to upstream mine development and climate change [3].
Hitura, Finland	OP, UG	Cu, Ni	Sub-arctic	AMD, GW, WQ	Zones of contaminated groundwater have developed, particularly near tailings disposal sites. Alterations to groundwater quality are complex due to naturally high iron concentrations, acid rock drainage and also weathering of the jarosite waste pile. Seepage has transitioned from being acidic in the 1970s to now being neutral [4].
Hope Downs, Australia	OP	Fe	Arid	D, DW	The transition from mining above the groundwater table to now below has required discharge of 220 GL from 2007 to 2013. The ephemeral flow regime of Weeli Wolli Creek has been altered, with continuous flow observed for ~24 km downstream of the discharge point [5].
Las Luces, Chile	UG	Cu	Arid	MW	Seawater is used directly in the concentration plant to treat ore coming from five underground mines [6].
Lihir, Papua New Guinea	OP	Au	Tropical	DW, GW, MTD, S, WQ	Dewatering to prevent seawater intrusion to the open pit have reversed groundwater flow, leading to seawater migration into surrounding groundwater systems [7]. The groundwater that is extracted from the underlying geothermal system may be as hot as 80–100 °C [8]. Also mine wastes are disposed directly to marine waters with observed impacts to coral ecosystems [9].
Mount Lyell, Australia	UG	Au, Cu	Temperate	AMD, IRTD, S, WQ	Extensive acid rock drainage and in-riverine disposal of tailings has significantly degraded the river ecosystem. Acid rain caused by smelting sulphide ores has resulted in prolonged periods of vegetation loss [10].
Mount Polley, Canada	OP, UG	Ag, Au, Cu	Temperate	DF, S, SW, WQ	Failure of tailings dam led to significant sediment transport to downstream river and lake systems [11].
Nifty, Australia	OP	Cu	Arid	AMD, GW, WQ	Scenarios for backfilling of the pit have been evaluated. If fully backfilled, a flow-through groundwater system would develop, acid generation would become a long-term risk and acid seepage from the waste rock dump would no longer flow towards the pit. If partially backfilled, a pit-lake and terminal groundwater sink would develop, resulting in reduced rates of acid generation and little risk of acid water entering surrounding paleo-channels [12].
Ok Tedi, Papua New Guinea	OP	Au, Cu	Tropical	DF, HV, IRTD, SD, S, WQ	Tailings dam wall collapsed during construction. Subsequently, tailings are disposed directly to the Fly River system with extensive impacts to downstream communities and ecology [13]. Recent drought conditions have prevented barges from accessing the site and disrupted production for seven months.
Olympic Dam, Australia	UG	Ag, Au, Cu, U	Arid	DW, GW	Groundwater extraction from the Great Artesian Basin (GAB) is about 35 ML/day, or roughly 7% of the GAB inflow to South Australia, and the 10 m zone of groundwater depression may reach 4400 km^2 by 2055 [14][15][16]. Process water efficiency improved from 1.27 kL/t ore in 2004 to 1.08 kL/t ore in 2014 [15][17]. From 1998 to 2004, BHP funded a program to cap pastoral bores – reducing GAB outflows by 37 ML/day [14].
Pyhäsalmi, Finland	UG	Cu, Zn	Sub-arctic	D, S, SW, WQ	Long-term discharge of treated effluents to the Junttiselkä section of Lake Pyhäjärvi have elevated sediment Cu and Zn concentrations by 9 times and 3 times respectively [18].
Tallering Peak, Australia	OP	Fe	Temperate	AMD, GW, WQ	Pit backfilling scenarios and the impact on groundwater flow have been studied. With no backfilling the pit lake-groundwater system would reach equilibrium after seven years. A partial backfill would reduce oxidation rates and trap contaminants in the pit lake/terminal groundwater sink system. A full backfill would become a groundwater flow-through system, with potential for contaminated groundwater migration to a nearby seasonal stream [12].
Talvivaara, Finland	OP	Ni, Zn	Sub-arctic	HV, SW, WQ	Exceptional amount of rainfall contributed to a situation where excess water accumulated on the mine site. The mine suffered several leaks of metal-contaminated tailings, and the sulfate concentration of the effluent exceeded the permitted levels causing severe pollution in the water course [19].

Type: OP – Open pit mine, UG – Underground mine.
Areas of Complexity: AMD – Acid and/or Metalliferous Drainage, CI – Cumulative Impacts, D – Discharge, DW – Dewatering, DF – Dam Failure, F – Floods, GW – Groundwater, HV – Hydrologic Variability, IRTD – In-Riverine Tailings Disposal, MTD - Marine Tailings Disposal, MW- Marine Waters (seawater), S –Sediments, SD – Supply Disruption, SW – Surface Water, WQ – Water Quality.
References: [1] QFCI, 2012, [2] Newcrest, 2007, [3] Huang et al., 2010, [4] Heikkinen et al., 2002, [5] Dogramaci et al., 2015, [6] Moreno et al., 2011, [7] Vogwill et al., 2009, [8] Williamson and Vogwill, 2001, [9] Haywood et al., 2016, [10] Koehnken, 1997, [11] Chambers, 2016, [12] McCullough et al., 2013, [13] Bolton, 2009, [14] Bekesi et al., 2013, [15] BHPB, 2014, [16] Welsh, 2006, [17] Torrisi and Trotta, 2009, [18] Mäkinen and Lerssi, 2007, [19] Turpeinen and Rainio, 2013.

footprint of different life cycle stages, identifies potential environmental impacts related to water, and includes geographic and temporal dimensions. Water use is considered as use of water by human activity, i.e. the hydrologic water cycle is not included. When interpreting results the positive aspects can be described if relevant. ISO does not recognize the so called green, blue or grey water definitions (ISO, 2014).

There are four main stages when conducting a water footprint or life cycle assessment study (refer to Fig. 3): 1. Goal and scope setting, 2. Inventory development, 3. Impact assessment, 4. Interpretation. Due to differences in individual studies, a range of methods and data sources have been developed for each stage of this process (Kounina et al., 2013).

3.1. Goal and scope setting

The goals and scope of a study will heavily influence data collection decisions, the most appropriate impact assessment methods to use, and ultimately how the results of the study are interpreted. Water footprinting studies may be used to identify the impact hotspots within supply chains or processes, to support comparative assertions of the relative impacts associated with products, to support cross-sectoral analysis of water efficiency and use, or even purely as a marketing exercise. The goals of the study will dictate the process boundaries of the system being considered. Some studies may measure the embodied impacts of a product (i.e. cradle-to-gate), whereas other studies may measure the impacts

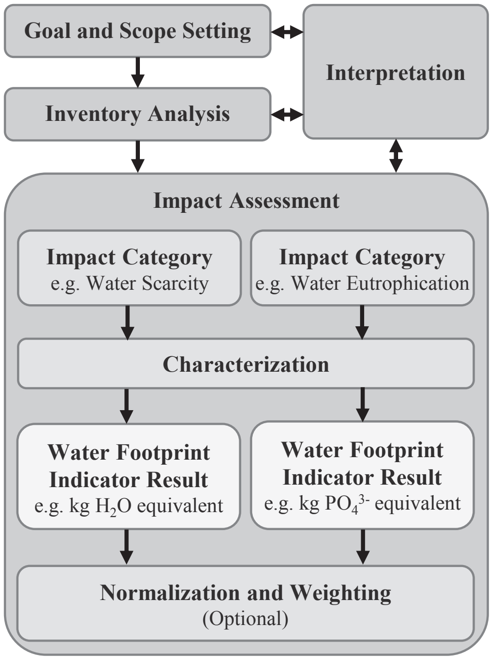

Fig. 3. Main phases of a water footprint assessment. Adapted from the ISO water footprinting standard (ISO, 2014).

across the full life-cycle of a product (i.e. cradle-to-grave; production, use and disposal). Determining appropriate temporal and geographic boundaries for the assessment is also an important consideration.

3.2. Inventory development – accounting for water volumes and quality

Water footprinting and life cycle assessments require the development of an inventory that quantifies the flow of water into and out of the boundary of assessment. These inventories will at a minimum provide an estimate of the magnitude of these water flows. Additional inventory data – such as water quality parameters, the location of flows, the time period that the flows occur, and the underlying data uncertainty – may also be required depending upon the goals of the study and the impact assessment methods used (Berger and Finkbeiner, 2013; Jeswani and Azapagic, 2011).

Inventories may be developed for a range of boundaries such as a processing facility, a geographic region, or the full-life cycle of a product. A principle of water footprinting and life cycle assessment, particularly when developing inventory databases, is that individual process or product inventories should avoid double counting of impacts through consistent selection of process boundaries and the procedures used to allocate impacts to individual products.

There are large overlaps between the data required for a water footprint inventory and that of water accounting methods developed specifically for the mining industry, such as the Minerals Council of Australia's *Water Accounting Framework for the Minerals Industry* (WAFMI) (MCA, 2014). The WAFMI has been shown to be broadly applicable to mine sites, regardless of their on-site water use requirements or the hydrological context that they operate within (Danoucaras et al., 2014). Data reported using the WAFMI could in many cases be utilised directly within a water footprint inventory and an example of this is provided in Table 2.

The approach taken to assigning water quality categories in the WAFMI differs slightly from approaches suggested for use in water

Table 2
Example water footprint inventory of Cadia Valley Operations in New South Wales, Australia. Data is for the financial year July 2012 to June 2013 (Newcrest, 2013, 2014).

Water flows [a]
Used to estimate direct water footprint

Water quality [b]	Cat. 1	Cat. 2	Total
Inputs, ML			
Groundwater	221	1032	1253
Surface Water [c]	6790	2181	8971
Recycled Water [d]	1952	–	1952
Outputs, ML			
Entrainment	–	5365	5365
Evaporation	6952	–	6952
Groundwater	–	576	576
Surface Water	1150	–	1150
Other	772	172	944

Material and Energy Flows
Used to estimate indirect (supply chain) water footprint

Inputs		
Acetylene	89	GJ
Ammonium Nitrate	4347	t
Coolant	1467	GJ
Diesel	865,177	GJ
Electricity – Grid	3,344,844	GJ
Flocculants	501	t
Flotation Frothers	777	t
Grease	7166	GJ
Grinding Balls	22,431	t
Hydraulic Fluid	1228	GJ
LPG	2400	GJ
Oil	23,316	GJ
Oil-based Paint	199	GJ
Packaged Explosives	73	t
Petrol	9	GJ
Quicklime	15,838	T
Scale Inhibitors	197	T
Outputs		
Copper Concentrate	53,913	t Cu
Gold [e]	446,879	oz Au

Notes.
[a] Uncertainty of flow estimates: 34% high confidence, 44% medium confidence, 22% low confidence.
[b] Quality categories as defined by the Water Accounting Framework for the Minerals Industry (MCA, 2014).
[c] Includes rainfall and runoff.
[d] Recycled water from town effluents.
[e] Includes gold doré bars and gold contained in the concentrate.

footprint and life cycle assessments (see Fig. 4). Boulay et al. (2011) provided a method for categorising the quality of surface water and groundwater flows into 8 distinct categories based upon the suitability for use by different end-users (e.g. agriculture, recreation, human consumption, etc.). Water flows are assigned to these categories based upon threshold limits for 136 different water quality parameters. As a comparison the WAFMI only has 3 water quality categories (MCA, 2014): Category 1 water is close to drinking water standards; Category 2 water is usable for some purposes; and, Category 3 water is unusable for most purposes. The WAFMI only states explicit water quality thresholds for pH, total dissolved solids, coliforms and turbidity. For other quality parameters (e.g. heavy metal concentrations) it is recommended to refer to guidelines for safe drinking water.

3.3. Impact assessment methods

The data provided within water footprint inventories is useful for understanding the overall consumption or change in water quality associated with a product or process. However, effective decision making based upon this requires an understanding of how

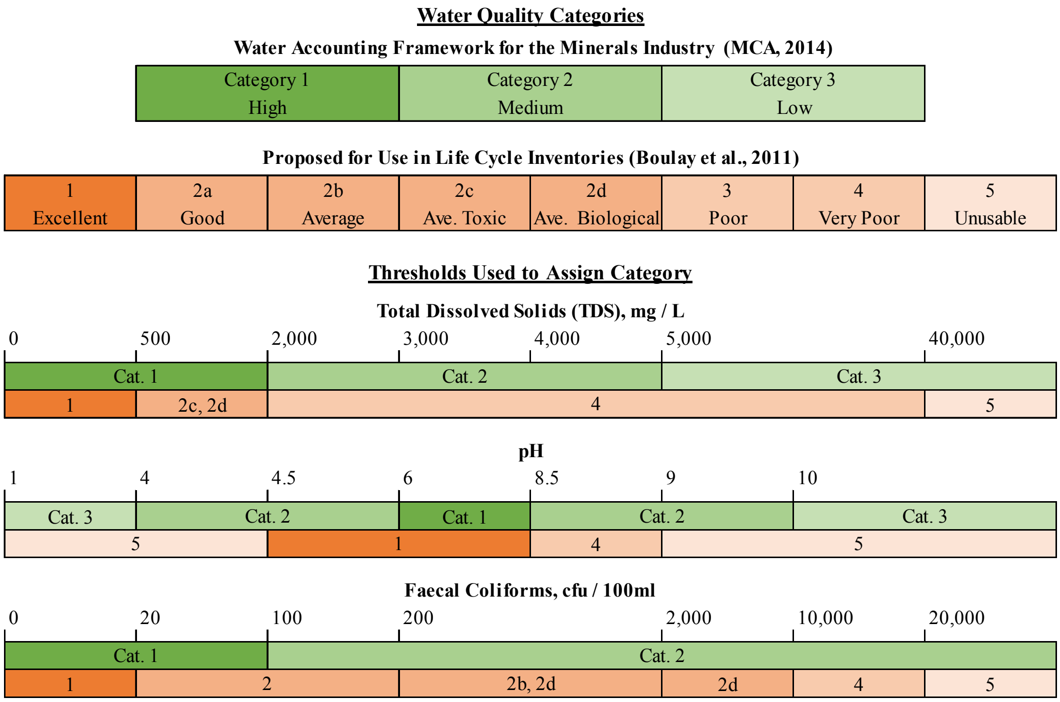

Fig. 4. Examples of the differing quality thresholds proposed for categorising water flows in life cycle inventories (Boulay et al., 2011), with those used by the Water Accounting Framework for the Minerals Industry (MCA, 2014).

this water use will impact surrounding ecosystems, communities and industry (Fang and Heijungs, 2015). To address this, methods have been developed to assess the impacts associated with water consumption or water quality degradation, through the use of defined impact characterisation procedures. Currently a variety of these procedures exist, with each being based upon different underlying assumptions, data sources, modelling choices, and conceptualisations of what actually constitutes an impact of water use (Boulay et al., 2015; Kounina et al., 2013). Impacts may include contributions to regional water scarcity, depriving other users of access to water, reducing the water flows required to maintain ecosystem functions, or degradation of water quality.

The provision of products or services may have adverse consequences for other issues of societal concern, such as climate change. Therefore the results of water footprint assessments may also be complemented by considering more traditional life cycle assessment impact categories, such as eutrophication potential, ecotoxicity or global warming. On this point, the reader is referred to the work of Frischknecht et al. (2016) that outlines the current state of impact assessment methodology and recommended approaches.

3.3.1. *Spatially explicit impact characterisation procedures*

The impacts associated with water use depend heavily on the local context, due to factors such as: the regions hydrologic regime, background water quality, the presence of sensitive ecosystems, and the level of dependence of communities, agriculture and industry on water resources. Consequently, a variety of impact characterisation procedures have been developed that attempt to capture how water use impacts will vary across regions, countries and hydrologic basins. These differ based upon the underlying conceptualisation of what constitutes an impact, and also the specific datasets and models that underpin these. The outputs of these procedures are regional impact characterisation factors that may be available at different spatial scales (e.g. continental, national, watershed, sub-watershed, etc.). Several examples of these are shown in Fig. 5.

Perhaps the most prominent index used in the literature to date is the Water Stress Index (WSI) (Pfister et al., 2009) (Fig. 5). The major variable determining a regions WSI is the ratio of water withdrawals-to-availability, which was determined using the WaterGAP2 model (Alcamo et al., 2003). The WSI also accounts for inter- and intra-annual precipitation variability. Monthly WSI data has also been made available (see Pfister and Bayer, 2014). Despite being the most commonly used method within the literature to date, there has been criticism regarding the underlying conceptualisation and meaningfulness of the method (Hoekstra, 2016).

Alternative methods have been proposed, such as the Water Accounting and Vulnerability Evaluation (WAVE) (Berger et al., 2014). The WAVE method introduces several additional complexities such as modifying inventory flows of evaporation according to factors for local basin internal evaporation recycling (i.e. how much of the water that evaporates is re-precipitated upstream in the same drainage basin). Subsequently, the characterisation of freshwater depletion risk is achieved through the use of the Water Depletion Index (Fig. 5), which provides a measure of the potential for water consumption to permanently decrease water availability in an area.

A working group for Water Use in Life Cycle Assessment (WULCA) was established by the UNEP/SETAC Life Cycle Initiative to provide guidance and develop consensus based methods. WULCA has developed a recommended approach for measuring the potential to deprive other users of water. This has been evaluated by considering the available water remaining (AWaRe) once ecosystem and human consumption demands have been met. The characterisation factors for water deprivation potential developed using the AWaRe method are currently available for download as a beta version (WULCA, 2015) (Fig. 5).

Spatial characterisation factors have also been developed for blue water scarcity (Hoekstra and Mekonnen, 2011; Wada et al., 2011), specific water sources (i.e. precipitation, surface water and groundwater) (Yano et al., 2015), and for some water quality impacts – such as factors for the fate of agricultural phosphorous

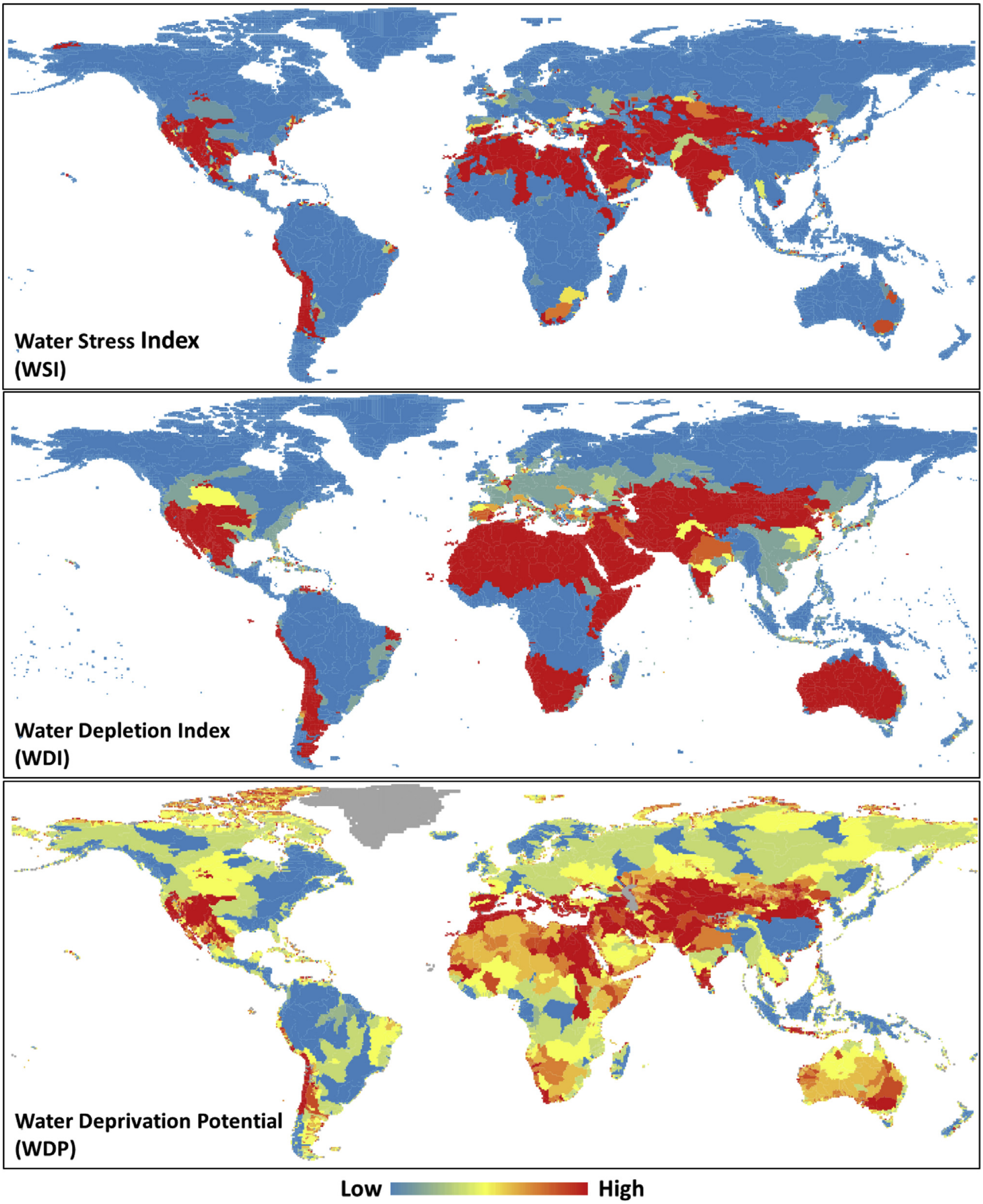

Fig. 5. Examples of spatial impact characterisation factors. The water stress index (Pfister et al., 2009), water depletion index (Berger et al., 2014) and water deprivation potential (WULCA, 2015) are shown and have been redrawn to a common scale.

emissions that can be used in assessments of eutrophication potential (Scherer and Pfister, 2015). Over time we should expect an increase in the availability of spatially explicit characterisation factors for use in water footprinting.

3.3.2. Single-indicator impact assessment methods

Several single-indicator impact methods have been proposed in an attempt to simplify the communication of more complex water footprint results. These methods attempt to combine consumptive water use impacts (e.g. reduction in available water volumes) and degradative water use impacts (water quality changes) into a single indicator.

The Water Impact Index (WII) proposed by Bayart et al. (2014) is an approach that seeks to incorporate both water quality and stress related impacts into a single metric. Regional water stress is incorporated through the use of the WSI (Pfister et al., 2009), however updated water scarcity metrics may be substituted as they become available. Water quality impacts are incorporated through a quality index that is the ratio of pollutant concentrations to ambient water quality standards, which also may vary between regions.

Ridoutt and Pfister (2013b) also proposed a simplified single-indicator method as the sum of consumptive water use and degradative water use. Consumptive water use is estimated using WSI weighted water consumption divided by the global average WSI (Pfister et al., 2009). Whereas degradative water use is quantified using the ReCiPe impact assessment method (Goedkoop et al., 2009) and is estimated as being the sum of water related endpoint impacts divided by the global average impact associated with freshwater use.

Implicit in these methods are impact normalisation and weighting procedures, the use of which have been scrutinised and debated within the life cycle assessment community – as they introduce a degree of subjective value judgement into the results of studies.

3.4. *Water footprint interpretation*

The most important stage of a water footprint assessments is the interpretation of the results. A comprehensive understanding of the underlying validity of the study's methodology and data sources is paramount to ensure that the study can confidently be used as part of decision making processes. The representativeness of data for the system analysed should be considered. Data gaps and relevant cut-off criteria used to determine the data that was included should be clearly identified. A common limiting factor that prevents interpretation of results is the level of aggregation of inventory and impact data (Reap et al., 2008). Over-aggregated data can prevent decision makers from understanding where in the system that impacts actually occur and the underlying causes. The results of water footprint studies will always have a certain level of uncertainty associated with them and efforts should be taken to ensure that these are understood. Consideration may also be required of the trade-offs between the water footprint and other assessments of environmental impact (e.g. carbon footprint), economic performance or social impact.

4. Existing water footprint and withdrawal studies of the mining industry

Water footprinting methodology has seen only limited application within the mining industry. A summary of all mining water footprint studies known to the authors is provided in Table 3. More general life cycle assessment studies that include a “water use” category, but are not focused upon water use, have been excluded from this discussion. Several of the studies have a mine-to-metal boundary and so may include additional data for smelters and refineries.

The majority of the studies to date could only be considered inventory level analysis that have a focus on looking at volumes of water consumption, rather than assessing impacts associated with this water use. However, recent studies are becoming more sophisticated in their methodological approach. Water scarcity impacts, as well as consideration of indirect water use, have routinely been considered in studies occurring since 2014. A large blind spot is the consideration of water quality, which has only been considered by several studies to date.

The advent of sustainability reporting resulted in the disclosure of a significant amount of water withdrawal data by mining companies (Mudd, 2008). Often this data has been communicated as water use intensity (e.g. m^3/kg contained metal produced or m^3/kg ore processed), rather than as an absolute quantification of total water withdrawals at mine sites. Sustainability report data has been compiled and analysed at Monash University for a range of commodities including copper (Mudd, 2008; Northey et al., 2013), platinum group metals (Mudd, 2008; Glaister and Mudd, 2010), bauxite, coal, gold, iron, lead, nickel, uranium and zinc (Mudd, 2008). Gunson (2013) also developed similar datasets for use when estimating the global water withdrawals associated with non-fuel mining, and Wessman et al. (2014) presented some data for mines in Finland. The limitation of these datasets are that they currently only include data for water withdrawals and have limited detail on the sources of water (e.g. surface water, groundwater), the quality of this water, or other important data required for rigorous water footprint estimates (e.g. rainfall, discharges, etc.).

A range of studies have utilised the Water Footprint Network's approach of calculating Blue Water, Green Water or Grey Water footprints. These have been conducted for copper production at the El Teniente mine site (Olivares et al., 2012) and in Northern Chile (Peña and Huijbregts, 2014); also, the South African production of platinum group metals has been assessed (Haggard et al., 2015; Ranchod et al., 2015).

Early studies conducted by Australia's CSIRO focused on comparing the water consumption associated with production technologies for individual metals (Norgate and Lovel, 2004, 2006). Northey and Haque (2014) and Northey et al. (2014) provided updated data for the main copper, gold and nickel production processes - as well as providing an initial assessment of the global average water stress associated with mining and metal production.

5. Limitations to the use of water footprinting in mining

A range of methodological and data limitations hamper the efforts to conduct water footprint studies of mining. Overcoming these will improve the accuracy and representativeness of water footprinting results for the mining industry, as well as improve the general usefulness of studies for decision making.

5.1. *Availability of mine site water data*

The availability of mine-site water use data is a significant impediment to developing water footprint inventories for the industry. A significant amount of water is reported publicly and privately by mine sites as part of statutory reporting obligations, corporate sustainability reporting, and within the broader scientific literature (see Table 4) (Mudd, 2008; Leong et al., 2014). Despite much of the required water use data for many regions being available through these sources, significant effort and ambition would be required to compile and assess the full breadth of data that is available. It is anticipated that in some cases where volumetric water use data is available, the corresponding water quality and seasonal data that is important for some impact assessment methods may not be reported.

Existing life cycle inventory and water footprinting databases do provide some limited data for different stages of the mining value chain, albeit with some data quality issues existing. Due to the significant variability in impacts between individual mining operations, even within the same country, a large degree of industry coverage is required to ensure representativeness. This is

Table 3
List of identified studies that provide water footprint related information for the mining industry. General life cycle assessment studies for mining that include, but are not focused upon, water use have been excluded.

Year	Authors	Region	Commodities	Focus	Accounted for?			Methodological approach
					Indirect water	Scarcity[a]	Quality[b]	
Unpublished. Results	Saarivuori& Wessman-Jääskeläinen	Finland	Au, Cu	Facility	Yes	WSI	Boulay	Water availability footprint method proposed by Boulay et al. (2011).
2016	Buxmann et al.	Global	Al	Commodity	Yes	WSI	–	Water scarcity footprint estimated using individual site data, however presented as an aggregated global value.
2015	Haggard et al.	South Africa	PGM	Facility	No	BWS	GW	Water Footprint Network approach (Hoekstra et al., 2011).
2015	Ranchod et al.	South Africa	PGM	Facility	Yes	–	–	Water Footprint Network approach (Hoekstra et al., 2011).
2014	Northey et al.	Australia	Au, Cu, Ni	Process	Yes	WSI	ReCiPe	Single indicator water footprint method proposed by Ridoutt and Pfister (2013b).
		Global	Al, Ag, Au, Co, Cu, Cr, Fe, Mo, Ni, Sn, Pb, Pt, Pd, Ti, Zn	Commodity	–	WSI	–	Production weighted average WSI (WSI), based upon national scale data. Volumes not considered.
2014	Peña & Huijbregts	Chile	Cu	Process	Yes	WSI	–	Water Footprint Network approach (Hoekstra et al., 2011).
2013	Gunson	Global	Au, Ag, Al, Co, Cr, Cu, Diamonds, K, Fe, Mn, Mo, Ni, Pd, Pb, PO4, Pt, Rh, Sn, Ta, Ti, U, W, Zn	Commodity	No	–	–	Estimate of global water withdrawals associated with non-fuel mining. Extrapolated from m3/t ore and m3/t concentrate data.
2013	Haggard et al.	South Africa	PGM	Facility	No	–	–	Water Footprint Network approach (Hoekstra et al., 2011).
2013	Northey and Haque	Australia	Au, Cu, Ni	Process	Yes	–	–	Direct and indirect water consumption estimated based upon process models.
2013	Northey et al.	Global	Cu	Facility	No	–	–	Sustainability reported water withdrawals allocated to copper product by economic value.
2012	Olivares et al.	Chile	Cu	Facility	No	–	–	Water Footprint Network approach (Hoekstra et al., 2009).
2010	Glaister & Mudd	Global	PGM (Pt, Pd, Rh)	Facility	No	–	–	Sustainability reported water withdrawals.
2008	Mudd	Global	Ag, Al, Au, Coal, Cu, Diamonds, Fe, Ni, Pb, PGM, U, Zn	Commodity	No	–	–	Sustainability reported water withdrawals.
2006	Younger	South America	–	Facility	Yes	Net Rainfall	–	Proposed an areal water footprint method for mining, measured in hectares. Accounted for post-closure impacts.
2006 2004	Norgate & Lovel	Australia	Al, Au, Cu, Fe, Ni, Pb, Ti, Zn	Process	Yes	–	–	Volumetric direct and indirect water use estimates.

Notes.
[a] BWS = Blue Water Scarcity (Hoekstra and Mekonnen, 2011), WSI = Water Stress Index (Pfister et al., 2009), Net Rainfall = (Annual rainfall – Annual Evapotranspiration).
[b] Boulay = Water quality categories for life cycle inventories (Boulay et al., 2011), GW = Grey Water (Hoekstra et al., 2009), ReCiPe = ReCiPe 2008 life cycle impact assessment method (Goedkoop et al., 2009).

Table 4
Potential data sources of relevance for developing mine-site water footprint inventories.

Reporting scheme	Site-specific availability	Volumetric disclosures					Quality disclosures		Periods considered		
		Import	Export	Recycling	Source	Storage	Concentration	Load	Pre-mining	Operating	Post-closure
CDP Water	L	M-H	L-M	L-M	L-M	L	–	–	–	H	–
Environmental Impact Assessments	H	H	H	M	H	H	M	L	H	H	L-H
Feasibility Study	H	L	L	L	M	H	L	L	M	H	L
Life Cycle Inventory Databases	L	H	M	–	M	–	L	H	L	H	L
National Pollution Inventories	H	–	–	–	–	–	–	H	–	H	–
Rehabilitation plans	L	L-H	M-H	L	M-H	H	H	L-M	M	M	H
Scientific literature	L-H	L-H	L-H	L-H	L-H	L-H	L-H	L-H	L-H	L-H	L-H
Statutory Reporting	M	H	H	H	H	H	H	L	L	H	L-M
Sustainability Reporting	M	H	M	M	M	L	L	L	L	H	L

Likelihood: H – High, M – Medium, L – Low; "–" – Not Reported.

particularly the case when considering more recent impact assessment methods that increasingly require the use of more highly localised data.

Inventory databases are slowly being expanded and improved to meet these types of demands. For instance, the Ecoinvent database recently adopted the Ecospold 2 database format and a market structure to facilitate further regionalisation of datasets (Steubing et al., 2016). Importantly, several new water source categories have been added and the consistency of process water balances has been improved (Pfister et al., 2015; Wernet et al., 2016).

The major international metal councils and associations develop some life cycle inventory data for mining, however the level of industry coverage, standardisation of methods, and level of aggregation could be improved. Efforts have been undertaken to attempt to harmonize the methodological approaches taken during collection of inventory data by these organisations (Santero and Hendry, 2016). However there are still inconsistencies in the allocation methods used by different industry sub-sectors (e.g. base metals, precious metals, etc.), which curiously decrease the impacts allocated to the primary product of every sub-sector. The high level of aggregation of these datasets - to avoid disclosure of commercially sensitive site-based data - may make the results of water footprinting and life cycle assessment less meaningful (Ross and Evans, 2002), as it has been argued that "aggregation is the overarching problem in interpretation" of life cycle assessment studies (Reap et al., 2008).

There are major long-term trends within the mining industry, such as: increasing mine size, declining ore grades, and increases in overburden and waste material (Mudd, 2010). These trends place upwards pressure on water use requirements and also the potential for water quality impacts to occur. As a counter-point, water management strategies and technologies are continually improving. Due to these continuous changes in the industry, older data that is contained within inventory databases may no longer be representative of the industry and should be updated periodically. Particularly given the unique characteristics of individual mine sites and their finite nature.

5.2. Availability of data for determining a mine's indirect/supply chain water footprint

Only several of the studies reviewed have included estimates of the indirect water consumption associated with mining, mineral processing or metal production (see Table 3). Consumption rates of energy, materials and process reagents at different mine sites can be considered commercially sensitive and so there is limited public reporting of this information. Occasionally companies will release relevant material and energy consumption data as part of their corporate sustainability reporting (see Table 2) or as part of industry surveys.

As with mining, other industries also have data limitations in the inventory databases available for use in water footprint studies. Many of the chemicals used at mine and mineral processing operations do not have data available within inventory databases (e.g. many flotation reagents). As a result, studies may be required to use generic inventory items (e.g. "inorganic chemicals") as a proxy for missing data.

Water management and handling processes can be significant consumers of energy at both mine sites (Gunson et al., 2010; Sahoo et al., 2014) and the water supply systems that feed these (Ihle, 2014). Therefore efforts to understand the diversity of mine site water management processes has relevance for life cycle assessment more broadly, as it may be a significant contributor to variability in energy requirements between individual mines.

5.3. Temporal nature of the mining industry

The water requirements and impacts change significantly throughout all stages of a mines life (e.g. pre-development, development, operation, rehabilitation, and post-closure). However all of the studies for mining identified in Table 3, with the exception of Younger (2006), only included data for the operational phase of the mines life. The majority also only considered static values for water consumption associated with the various processes, commodities or facilities. However, the water requirements of an individual mine can – and most likely will - change significantly through the operating period of a mine in response to weather events, changing site topography, the creation of reservoirs, increasing mine depths and changing ore processing requirements (see Fig. 6).

The post-closure impacts of mining have rarely been studied in detail using water footprint (or even life cycle assessment) methods, despite post-mining water balances being considerably altered from pre-development states (Eary and Watson, 2009). Younger (2006) provided a case study that showed a mine's annual post-closure water footprint would still be a quarter of the annual operational water footprint due to the formation of a pit-lake. Reid et al. (2009) also performed a life cycle assessment of alternative tailings management options for a Canadian mine and found that the post-closure time horizon considered was important as it may change what is viewed as the best management option. The truncation of time horizons, or the discounting of future impacts that may be used in the assessment of these types of scenarios, introduces an unavoidable value judgement into studies (Reid et al., 2009). Excessive discounting of future impacts may lead to decisions that place increased pollution burdens on future generations.

More complex mine site closure and rehabilitation scenarios may be beyond what could reasonably be evaluated using water footprinting and life cycle assessment. There is uncertainty over the long-term success of mine-site rehabilitation and the eventual uses of the new landscape. For instance, it was recently announced that the two open pits at the historic Kidston gold mine (Queensland, Australia) are to be converted to a pumped hydroelectricity storage scheme (Genex Power, 2016). Hypothetically, this type of scheme may be planned during the initial development of some mines in the future. A theoretical question that arises from this is, should impacts associated with the development of the mine be allocated to both the mined product and also the eventual electricity generated from the site? A more conventional example of this is when a mine has exhausted its' economic mineral resources, closed, and then reopened again later once technology and market conditions enable the profitable reprocessing of tailings – a situation that is not overly uncommon in the industry.

5.4. Accounting for unplanned or abnormal operating conditions and events

Inventory databases currently include no consideration of the impacts associated with abnormal operating conditions, such as during periods of flooding, drought, spillages or tailings dam collapses. This is despite these events potentially being major sources of environmental degradation associated with mining. There seems to be no clear approach for how to account for these types of issues within inventory databases.

An example of this is the widespread flooding of active and abandoned coal mines that occurred during the 2010/2011 floods in Queensland, Australia. In response to these floods, emergency approval procedures were used to allow discharges to local river systems, on the basis that they were necessary to avoid greater pollution risks, such as the risk of dam collapses (QFCI, 2012).

Using the example of a dam collapse, one method may be to take a probabilistic approach and assign a marginal increase in pollution burden to all individual mine-sites. However it would be contentious as miners will always claim that their own dam is safe.

5.5. Cumulative impacts of mining projects

Another difficulty is in accounting for the cumulative impacts of mining. A principle in water footprinting and life cycle assessment is that unit boundaries in inventories should be additive in nature, meaning that the sum of individual unit processes should equal the total impact associated with the system. However, this may not be the case if boundaries are drawn at the individual mine level, particularly when multiple mines are being developed nearby in the same region.

An example of this is the iron ore projects at Cape Preston in the Pilbara region of Australia, where dewatering at individual mines

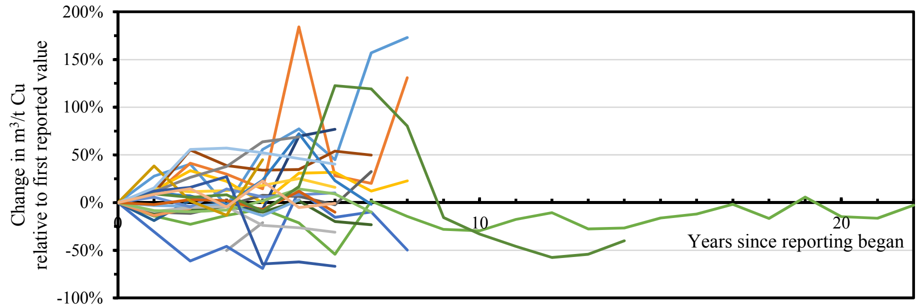

Fig. 6. Unit water withdrawals over time at 28 copper mines (derived and updated from data in Northey et al., 2013). Data is expressed relative to the first reported value.

have been shown to have a synergistic effect by reducing groundwater levels (and hence dewatering requirements) at other nearby mines (Sheppard et al., 2009). Therefore the development of multiple mines in a region may lead to marginal reductions in dewatering requirements when considered on a product basis (e.g. m^3 H_2O/t product).

Overcoming these types of issues for some regions may require the use of a catchment or regional boundary of assessment when developing an inventory, rather than the boundaries of individual mines.

5.6. Resolution and reliability of spatial characterisation factors

Mining occurs in a range of hydrological contexts and in almost all regions of the world. Therefore the results of water footprint studies of mining will be highly sensitive to the conceptual models and data underpinnings used to develop regional impact characterisation factors for water use. Boulay et al. (2015) identified regions where the results of water footprinting are highly sensitive to the impact assessment method and underlying modelling choices related to temporal and geographic resolution.

The regional characterisation factors used by current impact assessment methods are based upon data obtained from global hydrological models, which currently are able to produce results that broadly capture the observed hydrological patterns of large geographic regions. However, the results for individual sub-watersheds or regions may not be an accurate representation of reality. The implications of this for water footprint studies has been shown through the modelling or assessment of water stress at smaller spatial scales in a number of regions, including: the Mississippi Basin (Scherer et al., 2015), Spain (Núñez et al., 2015), Denmark (Hybel et al., 2015), and Great Britain (Hess et al., 2015). This may increase the uncertainty of impact estimates for the mining industry, particularly due to the industry's highly localised water use when compared to more geographically dispersed industries, such as agriculture.

The distribution of water stress and scarcity in the future may be different than it is today due to changes in regional climates and distributions of water use (Alcamo et al., 2007; Kiguchi et al., 2015). Therefore the impact characterisation factors for regions being used currently should not necessarily be considered static values. An implication being that studies of the long-term water impacts of mining may benefit from accounting for the potential changes of characterisation factors.

The accuracy and technical sophistication of these factors is expected to improve over time due to improvements in hydrological data, models and conceptualisations of impact. Additional differentiation of water sources and the incorporation of further data sources, such as regional geochemical data of the type provided by Rapant et al. (2008), may also be possible.

6. Future opportunities for using water footprinting in the mining industry

6.1. Supporting comparative assessments of mining's water use, both internally and cross-sectorally

A key opportunity offered by water footprinting methods is the ability to standardise comparisons of water related impacts associated with different products, production facilities, and industry sectors - both across regions and through supply chains.

Increasingly, estimates of the environmental impacts associated with products are being displayed to end-consumers through the use of eco-labelling schemes. Given the breadth of products that rely on mining in some form (i.e. through the incorporation of metal components into the product or the use of coal-based electricity during manufacture), improvements to the accuracy of mining inventory data and impact assessment will have direct flow on-effects for the accuracy of eco-labelling schemes. This also has implications for decision making processes for material selection and sourcing, which increasingly consider environmental impacts through the use of life cycle assessment and related schemes.

Comparing mineral processing circuits and determining whether an individual operation is *water efficient* is not straightforward. As a comparison, the attempts to benchmark the operational energy efficiency associated with ore crushing and grinding have been required to make adjustments for particle sizes, ore grades, and mineralogical hardness to enable a fair comparison (Ballantyne and Powell, 2014). Similar variables influence the total amount of water a processing circuit requires and the ability to recover water from tailings and mineral concentrates (Mwale et al., 2005; Palaniandy and Powell, 2014). Any scheme developed to fairly benchmark the operational water efficiency of these circuits would also need to account for similar variables to ensure fairness. Water footprint inventories for mine sites may be able to contribute some data towards this type of scheme – however, additional data that is not typically included within inventories would also have to be considered.

6.2. Support comparative technology assessments, the identification of process hotspots and the potential for burden shifting

A major strength of water footprinting and life cycle assessment is the ability to identify the potential for burden shifting, where changes to a process or management practice lead to adverse impacts elsewhere within a supply chain. This may be particularly concerning when impacts are shifted to regions of high water stress or sensitivity to water quality changes. As an example, the direct water consumption of copper heap leaching is relatively low when compared to copper flotation processes (COCHILCO, 2014). However the indirect water consumed to produce sulphuric acid can be

Table 5
Examples of water saving technologies used within the mining industry (Gunson et al., 2012; Moreno et al., 2011; Mwale et al., 2005; Napier-Munn and Morrison, 2003; Wels and Robertson, 2003).

Technology	Brief description
On-line characterisation and ore sorting	Advances in on-line mineral characterisation techniques allow selective separation of high value ore, leading to reductions in the amount of ore processed per unit of product and consequential reductions to water use.
Paste and thickened tailings	Decreases water losses to tailings storage facilities, improves the ease of site rehabilitation, and reduces water related tailings risks.
Seawater flotation	Seawater may be used in some flotation processes as a substitute for freshwater.
Tailings placement strategies	Strategic placement of tailings material within storage facilities can minimise the wetted area and reduce evaporative losses.

considerable and so changing to a leaching process may increase overall supply chain water consumption in some settings (Northey et al., 2014).

A range of water saving technologies have been developed and used within the mining industry (Table 5) (see Gunson et al., 2012). Water footprinting may be used to assess these technologies to identify whether burden shifting will occur and also to identify the major hotspots for process improvement (i.e. the processing steps that contribute disproportionately to overall impacts). Assessments of competing technologies or management options may also be performed, with an example being the comparative life cycle assessment of tailings management options at a Canadian mine site (Reid et al., 2009). Even in water abundant countries the potential of water footprinting has been recognised when benchmarking technologies that reduce water consumption and/or improve water quality.

Increasingly alternatives to freshwater water resources are being considered for use by the mining industry, including seawater (COCHILCO, 2015) and bacterial laden water (Liu et al., 2013). The ability to utilise poorer quality water within mineral processing circuits depends heavily on the mineralogy of the ore and the processing strategy employed. Poor quality water may requirement treatment prior to use in processing plants, and also subsequently prior to discharge to water bodies. A range of technologies are available for these purposes (see Table 6). Improvements in the impact characterisation procedures for different water sources and quality will increasingly enable the effective assessment of these technologies.

6.3. *Support corporate sustainability reporting and sustainability management schemes*

Over the past two decades there has been a substantial uptake of corporate sustainability reporting and management schemes by the mining industry (Fonseca et al., 2014), albeit with different rates of adoption amongst companies (Jenkins and Yakovleva, 2006). As part of this, many companies now voluntarily disclose water related data through schemes such as the Global Reporting Initiative and CDP Water. Information on the water use intensity of individual mine sites is often reported as part of this, however it is difficult to compare the data reported for different operations within a company's portfolio due to a lack of information on the regional context of individual sites. This data could be made more meaningful through improvements to the specification of water sources, water quality, and regionalisation of the data, so that the water context of individual operations could be better understood (Fonseca et al., 2014; Leong et al., 2014). Water footprinting methods that provide data of increased temporal and geographic relevance may be able to address some of these potential areas for improvement. Particularly for large-multinational companies that report aggregated data for multiple sites (e.g. at a regional or divisional level).

The use of water footprinting to communicate on environmental changes in water volume and quality at regional levels, particularly in areas already experiencing water scarcity or quality pressures, may also inform water allocation discussions and debates of a mine's social license to operate.

7. Conclusions

This article has identified a range of opportunities and limitations to the further use of water footprinting assessments of the mining industry. The main opportunities identified for the use of water footprint assessments within the mining industry include:

1. Standardised assessment of the water use impacts associated with mining and mineral processing will enable more fair and meaningful comparisons with other industrial sectors such as agriculture, forestry or manufacturing. Water footprinting methodology may also contribute to any attempt to develop a water efficiency benchmarking scheme for mineral processing operations.
2. Water footprinting and LCA can be used to provide a more holistic assessment of the benefits and drawbacks of technologies being developed and deployed in the mining industry through the consideration of indirect (supply chain) impacts.
3. Improve the usefulness and relevance of water related data disclosures that are presented by corporate sustainability reports. Particularly for companies that have facilities in multiple regions with differing water contexts.

Table 6
Examples of water treatment technologies used within the mining industry (Barakat, 2011; Gupta et al., 2012; Mohan and Pittman, 2007).

Technology	Brief description
Chemical precipitation	Low capital cost and simple operation that requires large dosing of chemicals, with associated high sludge generation and disposal costs. Lime readjustment of pH is often required.
Coagulation – flocculation	Coagulation and flocculation processes result in high sludge production and are relatively simple to operate. Sedimentation or filtration may be required and these processes may produce toxic sludges.
Biological treatment (sulphate removal)	Typically results in low waste production and operating costs, however slow in cold climates. Also a passive method: constructed wetlands or basins as passive bioreactors utilize soil and water borne microbes to remove dissolved metals and sulphates.
Membrane filtration	Small space requirements with high separation selectivity. Capital and operating expenditures can be high due to factors such as membrane fouling.
Adsorption	Adsorption is usually used as a secondary treatment step to avoid rapid usage of adsorbent material. Chemicals may be required for media regeneration. These processes usually have relatively low costs and easy operation, albeit with often low selectivity.

A range of limitations to the more widespread use of water footprinting in the mining industry were also identified. However, these actually represent significant opportunities for further research to improve methodology and data availability. The main limitations identified include:

1. Significant amounts of water-related data for mine sites exist in the public domain, however this is scattered across a range of mandatory and voluntary reporting schemes. Any effort to compile this data would be useful for improving the quality of mine-site inventory data, whilst also contributing to broader efforts to understand the water efficiency and risks of the industry.
2. Currently the ability to estimate indirect (supply chain) water footprints for the mining industry is limited by the availability of material and energy consumption data for individual mines and mineral processors. Even when this consumption data is available, many common mineral processing reagents are not included within life cycle inventory databases.
3. The timescale of many water quality and hydrological impacts associated with mine sites can be measured on decadal or even century timescales, with the expected impacts varying through time. Given that production from mine sites is only temporary, accounting for these post-closure impacts may require standardisation of the time frames considered during inventory development.
4. Unplanned pollution events or abnormal operating conditions, such as tailings dam failures or major flood events, are not currently accounted for within inventories. This is despite being a significant risk to local waterways, ecosystems and communities.
5. The cumulative impacts of mining on regional hydrology are difficult to quantify and may require alternative boundaries to be used when developing inventory data.
6. The spatial impact characterisation factors derived from global hydrological models are broadly accurate over large regions, but may not always be representative for individual basins or watersheds. Given the highly localised nature of mining water impacts compared to other more broad-scale industries (e.g. agriculture), the accuracy of estimated impacts for mining will be more highly sensitive to the underlying accuracy of sub-watershed impact characterisation factors.

Water footprinting and related life cycle assessment methodologies can contribute a unique perspective of the mining industry that is not provided by other types of assessment methods. Supporting these methods, through the further development of fair and consistent approaches to quantify the water related impacts occurring at different mine sites should be pursued.

Acknowledgements

The authors would like to acknowledge the financial support given by the CSIRO Mineral Resources collaboration fund. Discussions with collaborating researchers from the Sustainable Acceptable Mining (SAM) project partner organisations are also gratefully appreciated.

References

Akcil, A., Koldas, S., 2006. Acid Mine Drainage (AMD): causes, treatment and case studies. J. Clean. Prod. 14, 1139–1145. http://dx.doi.org/10.1016/j.jclepro.2004.09.006.

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T., Siebert, S., 2003. Global estimates of water withdrawals and availability under current and future "business-as-usual" conditions. Hydrological Sci. J. 48 (3), 339–348. http://dx.doi.org/10.1623/hysj.48.3.339.45278.

Alcamo, J., Flörke, M., Märker, M., 2007. Future long-term changes in global water resources driven by socio-economic and climatic changes. Hydrological Sci. J. 52 (2), 247–275. http://dx.doi.org/10.1623/hysj.52.2.247.

Amezaga, J.M., Rötting, T.S., Younger, P.L., Nairn, R.W., Noles, A.-J., Oyarzún, R., Quintanilla, J., 2011. A rich vein? Mining and the pursuit of sustainability. Environ. Sci. Technol. 45, 21–26. http://dx.doi.org/10.1021/es101430e.

Azam, S., Li, Q., 2010. Tailings Dam Failures: a review of the last one hundred years. Geotech. News 28 (4), 50–54.

Ballantyne, G.R., Powell, M.S., 2014. Benchmarking comminution energy consumption for the processing of copper and gold ores. Miner. Eng. 65, 109–114. http://dx.doi.org/10.1016/j.mineng.2014.05.017.

Barakat, M.A., 2011. Review Article: new trends in removing heavy metals from industrial wastewater. Arabian J. Chem. 4, 361–377. http://dx.doi.org/10.1016/j.arabjc.2010.07.019.

Bayart, J.-B., Worbe, S., Grimaud, J., Aoustin, E., 2014. The Water Impact Index: a simplified single-indicator approach for water footprinting. Int. J. Life Cycle Assess. 19 (6), 1336–1344. http://dx.doi.org/10.1007/s11367-014-0732-3.

Bekesi, G., Tyler, M., Niejalke, D., Waterhouse, J., 2013. Improved regulation of water supply for the Olympic Dam mine, South Australia. In: 6th International Mine Water Association Conference, Golden, Colorado, USA, pp. 29–34.

Berger, M., Finkbeiner, M., 2013. Methodological challenges in volumetric and impact-oriented water footprints. J. Industrial Ecol. 17 (1), 79–89. http://dx.doi.org/10.1111/j.1530-9290.2012.00495.x.

Berger, M., van der Ent, R., Eisner, S., Bach, V., Finkbeiner, M., 2014. Water accounting and vulnerability evaluation (WAVE): considering atmospheric evaporation recycling and the risk of freshwater depletion in water footprinting. Environ. Sci. Technol. 48, 4521–4528. http://dx.doi.org/10.1021/es404994t.

BPEMM, 1999. Water Management. Best Practice Environmental Management in Mining (BPEMM). Environment Australia, Australian Government, 99pp.

BHPB, 2014. Olympic Dam, Great Artesian Basin Wellfields Report, 1 July 2013-30 June 2014. BHP Billiton (BHPB). Report No. ODENV053. Available via the South Australian Resource Information Geoserver: sarig.pir.sa.gov.au/(accessed 04-07-2016).

Bolton, B.R. (Ed.), 2009. The Fly River, Papua New Guinea: Environmental Studies in an Impacted Tropical River System. Developments in Earth & Environmental Sciences, 9. Elsevier.

Bouffard, S.C., 2005. Review of agglomeration practice and fundamentals in heap leaching. Mineral Process. Extr. Metallurgy Rev. An Int. J. 26 (3–4), 233–294. http://dx.doi.org/10.1080/08827500590944009.

Boulay, A.-M., Bouchard, C., Deschênes, L., Margni, M., 2011. Categorizing water for LCA inventory. Int. J. Life Cycle Assess. 16, 639–651. http://dx.doi.org/10.1007/s11367-011-0300-z.

Boulay, A.-M., Hoekstra, A.Y., Vionnet, S., 2013. Complementarities of water-focused life cycle assessment and water footprint assessment. Environ. Sci. Technol. 47, 11926–11927. http://dx.doi.org/10.1021/es403928f.

Boulay, A.-M., Motoshita, M., Pfister, S., Bulle, C., Muñoz, I., Franceschini, H., Margni, M., 2015. Analysis of water use impact assessment methods (part A): evaluation of modelling choices based on a quantitative comparison of scarcity and human health indicators. Int. J. Life Cycle Assess. 20, 139–160. http://dx.doi.org/10.1007/s11367-014-0814-2.

Buxmann, K., Koehler, A., Thylmann, D., 2016. Water scarcity footprint of primary aluminium. Int. J. Life Cycle Assess. 1–11. http://dx.doi.org/10.1007/s11367-015-0997-1.

Caron, J., Durand, S., Asselin, H., 2016. Principles and criteria of sustainable development for the mineral exploration industry. J. Clean. Prod. 199, 215–222. http://dx.doi.org/10.1016/j.jclepro.2016.01.073.

Chambers, D., 2016. Post-Mount Polley: Tailings Dam Safety in British Columbia. March 2016. Earthworks, Washington, D.C., USA. www.earthworksaction.org/files/publications/Post-MountPolleyTailingsDamSafety.pdf (accessed 04-07-2016).

Chapagain, A.K., Hoekstra, A.Y., 2004. Water Footprint of Nations. UNESCO-IHE Institute for Water Education, The Netherlands.

COCHILCO, 2014. Hydrological Consumption in Copper Mining for 2014. Comisión Chilena del Cobre (COCHILCO), Government of Chile.

COCHILCO, 2015. Water Consumption Projection in Copper Mining through 2026. Comisión Chilena del Cobre (COCHILCO), Government of Chile.

Danoucaras, A.N., Woodley, A.P., Moran, C.J., 2014. The robustness of mine water accounting over a range of operating contexts and commodities. J. Clean. Prod. 84, 727–735. http://dx.doi.org/10.1016/j.jclepro.2014.07.078.

Davies, M., 2011. Filtered dry stacked tailings – The fundamentals. In: Proceedings of Tailings and Mine Waste 2011, Vancouver, Canada, Nov 6-9, 9pp.

Dogramaci, S., Firmani, G., Hedley, P., Skrzypek, G., Grierson, P.F., 2015. Evaluating recharge to an ephemeral dryland stream using a hydraulic model and water, chloride and isotope mass balance. J. Hydrology 521, 520–532. http://dx.doi.org/10.1016/j.jhydrol.2014.12.017.

Eary, L.E., Watson, A., 2009. The post-mining water balance. Min. Technol. 118 (3–4), 212–219. http://dx.doi.org/10.1179/174328610X12682159815064.

Edraki, M., Baumgartl, T., Manlapig, E., Bradshaw, D., Franks, D.M., Moran, C.J., 2014. Designing mine tailings for better environmental, social and economic outcomes: a review of alternative approaches. J. Clean. Prod. 84, 411–420. http://dx.doi.org/10.1016/j.jclepro.2014.04.079.

Fang, K., Heijungs, R., 2015. Investigating the inventory and characterization aspects of footprinting methods: lessons for the classification and integration of footprints. J. Clean. Prod. 108 (A), 1028–1036. http://dx.doi.org/10.1016/

j.jclepro.2015.06.086.
Fonseca, A., McAllister, M.L., Fitzpatrick, P., 2014. Sustainability reporting among mining companies: a constructive critique of the GRI approach. J. Clean. Prod. 84, 70–83. http://dx.doi.org/10.1016/j.jclepro.2012.11.050.
Franks, D.M., Boger, D.V., Côte, C.M., Mulligan, D.R., 2011. Sustainable development principles for the disposal of mining and mineral processing wastes. Resour. Policy 36, 114–122. http://dx.doi.org/10.1016/j.resourpol.2010.12.001.
Franks, D.M., Brereton, D., Moran, C.J., 2010. Managing the cumulative impacts of coal mining on regional communities and environments in Australia. Impact Assess. Proj. Apprais. 28 (4), 299–312. http://dx.doi.org/10.3152/146155110X12838715793129.
Frischknecht, R., Fantke, P., Tshümperlin, L., Niero, M., Antón, A., Bare, J., Boulay, A.-M., Cherubini, F., Hauschild, M.Z., Henderson, A., Levasseur, A., McKone, T.E., Michelsen, O., Milà I Canals, L., Pfister, S., Ridoutt, B., Rosenbaum, R.K., Verones, F., Vigon, B., Jolliet, O., 2016. Global guidance on environmental life cycle impact assessment indicators: progress and case study. Int. J. Life Cycle Assess. 21 (3), 429–442. http://dx.doi.org/10.1007/s11367-015-1025-1.
Genex Power, 2016. The Kidston Hydro Pumped Storage Project. Genex Power Limited, Sydney, Australia. www.genexpower.com.au/the-kidston-hydro-project.html (accessed 04-07-2016).
Ghorbani, Y., Kuan, S.H., 2016. A review of sustainable development in the Chilean mining sector: past, present and future. Int. J. Min. Reclam. Environ. 1–29. http://dx.doi.org/10.1080/17480930.2015.1128799.
Gibbons, T., 2005. International cyanide management code. Dev. Mineral Process. 15, 182–199. http://dx.doi.org/10.1016/S0167-4528(05)15008-X.
Glaister, B.J., Mudd, G.M., 2010. The environmental costs of platinum-PGM mining and sustainability: is the glass half-full or half-empty? Miner. Eng. 23, 438–450. http://dx.doi.org/10.1016/j.mineng.2009.12.007.
Gleeson, T., Wada, Y., Bierkens, M.F.P., van Beek, L.P.H., 2012. Water balance of global aquifers revealed by groundwater footprint. Nature 488, 197–200. http://dx.doi.org/10.1038/nature11295.
Goedkoop, M., Heijungs, R., Huijbregts, M., De Schryver, A., Strujis, J., van Zelm, R., 2009. ReCiPe 2008: a Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the endpoint Level, first ed. Ruimte en Mileeu, Ministerie van Volkshuishvesting, Ruimtelijke Ordening enMilieubeheer, The Netherlands.
Google Earth, 2016. Satellite Image of Cowal Gold Mine, NSW, Australia. Image: 33.64° S, 147.39° E, Eye Altitude 7.26 km, Imagery Date 3/12/2013, DigitalGlobe 2016. Accessed via Google Earth v. 6.0.3.2197.
Gunson, A.J., 2013. Quantifying, Reducing and Improving Mine Water Use. PhD Thesis. The University of British Columbia, Canada. May 2013, 285pp.
Gunson, A.J., Klein, B., Veiga, M., Dunbar, S., 2010. Reducing mine water network energy requirements. J. Clean. Prod. 18, 1328–1338. http://dx.doi.org/10.1016/j.jclepro.2010.04.002.
Gunson, A.J., Klein, B., Veiga, M., Dunbar, S., 2012. Reducing mine water requirements. J. Clean. Prod. 21, 71–82. http://dx.doi.org/10.1016/j.jclepro.2011.08.020.
Gupta, V.K., Ali, I., Saleh, T.A., Nayak, A., Agarwal, S., 2012. Chemical treatment technologies for waste-water recycling – an overview. RSC Adv. 2 (16), 6380–6388. http://dx.doi.org/10.1039/C2RA20340E.
Haggard, E.L., Sheridan, C.M., Harding, K.G., 2013. Water footprint for a South African platinum mine. In: Proceedings of Water in Mining 2013, Brisbane, Australia, Nov 26-28, pp. 15–20.
Haggard, E.L., Sheridan, C.M., Harding, K.G., 2015. Quantification of water usage at a South African platinum processing plant. Water SA 41 (2), 279–286. http://dx.doi.org/10.4314/wsa.v41i2.14.
Haywood, M.D.E., Dennis, D., Thomson, D.P., Pillans, R.D., 2016. Mine waste disposal leads to lower coral cover, reduced species richness and a predominance of simple coral growth forms on a fringing coral reef in Papua New Guinea. Mar. Environ. Res. 115, 36–48. http://dx.doi.org/10.1016/j.marenvres.2016.02.003.
Heikkinen, P.M., Korkka-Niemi, K., Lahti, M., Salonen, V.-P., 2002. Groundwater and surface water contamination in the area of the Hitura nickel mine, Western Finland. Environ. Geol. 42 (4), 313–329. http://dx.doi.org/10.1007/s00254-002-0525-z.
Hess, T.M., Lennard, A.T., Daccache, A., 2015. Comparing local and global water scarcity information in determining the water scarcity footprint of potato cultivation in Great Britain. J. Clean. Prod. 87, 666–674. http://dx.doi.org/10.1016/j.jclepro.2014.10.075.
Hinwood, A.L., Heyworth, J., Tanner, H., McCullough, C., 2012. Recreational use of acidic pit lakes – human health considerations for post closure planning. J. Water Resour. Prot. 4, 1061–1070. http://dx.doi.org/10.4236/jwarp.2012.412122.
Hoekstra, A.Y., 2016. A critique on the water-scarcity weighted water footprint in LCA. Ecol. Indic. 66, 564–573. http://dx.doi.org/10.1016/j.ecolind.2016.02.026.
Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M., 2009. Water Footprint Manual, State of the Art 2009. Water Footprint Network, Earthscan, London, United Kingdom.
Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M., 2011. The Water Footprint Assessment Manual: Setting the Global Standard. Water Footprint Network, Earthscan, London, United Kingdom.
Hoekstra, A.Y., Mekonnen, M.M., 2011. Global Water Scarcity: the Monthly Blue Water Footprint Compared to Blue Water Availability for the World's Major River Basins. Value of Water Research Report Series No. 53. UNESCO-IHE, The Netherlands.
Holley, E.A., Mitcham, C., 2016. The Pebble Mine Dialogue: a case study in public engagement and the social license to operate. Resour. Policy 47, 18–27. http://dx.doi.org/10.1016/j.resourpol.2015.11.002.
Hybel, A.-M., Godskesen, B., Rygaard, M., 2015. Selection of spatial scale for assessing impacts of groundwater-based water supply on freshwater resources. J. Environ. Manag. 160, 90–97. http://dx.doi.org/10.1016/j.jenvman.2015.06.016.
Huang, X., Sillanpää, M., Gjessing, E.T., Peräniemi, S., Vogt, R.D., 2010. Environmental impact of mining on surface water quality in Tibet: Gyama Valley. Sci. Total Environ. 408, 4177–4184. http://dx.doi.org/10.1016/j.scitotenv.2010.05.015.
ICMM, 2012. Water Management in Mining: a Selection of Case Studies. International Council on Mining & Metals (ICMM), London, United Kingdom. May 2012.
Ihle, C.F., 2014. The need to extend the study of greenhouse impacts of mining and mineral processing to hydraulic streams: long distance pipelines count. J. Clean. Prod. 84, 597. http://dx.doi.org/10.1016/j.jclepro.2012.11.013.
INAP, 2009. Global Acid Rock Drainage Guide (GARD Guide). International Network for Acid Prevention (INAP). www.gardguide.com (accessed 04-07-2016).
ISO, 2014. ISO 14046:2014. Environmental Management – Water Footprint – Principles, Requirements and Guidelines. International Organisation for Standardization (ISO), Geneva, Switzerland.
Jenkins, H., Yakovleva, N., 2006. Corporate social responsibility in the mining industry: exploring trends in social and environmental disclosure. J. Clean. Prod. 14, 271–284. http://dx.doi.org/10.1016/j.jclepro.2004.10.004.
Jeswani, H.K., Azapagic, A., 2011. Water footprint: methodologies and a case study for assessing the impacts of water use. J. Clean. Prod. 19, 1288–1299. http://dx.doi.org/10.1016/j.jclepro.2011.04.003.
Johnson, D.B., Hallberg, K.B., 2005. Acid mine drainage remediation options: a review. Sci. Total Environ. 338, 3–14. http://dx.doi.org/10.1016/j.scitotenv.2004.09.002.
Kauppi, S., 2013. Ympäristötietoa kaivoshankkeista – Taustatietoa kaivostoimintaan Liittyvästä Lainsäädännöstä Ja eräiden kaivosten Ympäristötarkkailusta. Suomen ympäristökeskuksen raportteja, Helsinki, p. 10 ([In Finnish]).
Kemp, D., Bond, C.J., Franks, D.M., Cote, C., 2010. Mining, water and human rights: making the connection. J. Clean. Prod. 18, 1553–1562. http://dx.doi.org/10.1016/j.jclepro.2010.06.008.
Kempton, H., Bloomfield, T.A., Hanson, J.L., Limerick, P., 2010. Policy guidance for identifying and effectively managing perpetual environmental impacts from new hardrock mines. Environ. Sci. Policy 13, 558–866. http://dx.doi.org/10.1016/j.envsci.2010.06.001.
Kiguchi, M., Shen, Y., Kanae, S., Oki, T., 2015. Re-evaluation of future water stress due to socio-economic and climate factors under a warming climate. Hydrological Sci. J. 60 (1), 14–29. http://dx.doi.org/10.1080/02626667.2014.888067.
Koehnken, L., 1997. Final Report: Mount Lyell Remediation Research and Demonstration Program. Supervising Scientist Report 126. Department of Environment and Land Management, Tasmania, Australia.
Kounina, A., Margni, M., Bayart, J.-B., Boulay, A.-M., Berger, M., Bulle, C., Frischknecht, R., Koehler, A., Milà I Canals, L., Motoshita, M., Núñez, M., Peters, G., Pfister, S., Ridoutt, B., van Zelm, R., Verones, F., Humbert, S., 2013. Review of methods addressing freshwater use in life cycle inventory and impact assessment. Int. J. Life Cycle Assess. 18, 707–721. http://dx.doi.org/10.1007/s11367-012-0519-3.
Kumar, R.N., McCullough, C.D., Lund, M.A., 2009. Water resources in Australian mine pit lakes. Min. Technol. 118 (3/4), 205–211. http://dx.doi.org/10.1179/174328610X12682159815028.
Kunz, N.C., Moran, C.J., 2016. The utility of a systems approach for managing strategic water risks at a mine site level. Water Resour. Industry 13, 1–6. http://dx.doi.org/10.1016/j.wri.2016.02.001.
Leong, S., Hazelton, J., Taplin, R., Timms, W., Laurence, D., 2014. Mine site-level water reporting in the Macquarie and Lachlan catchments: a study of voluntary and mandatory disclosures and their value for community decision-making. J. Clean. Prod. 84, 94–106. http://dx.doi.org/10.1016/j.jclepro.2014.01.021.
Liu, W., Moran, C.J., Vink, S., 2013. Managing the potential risks of using bacteria-laden water in mineral processing to protect freshwater. Environ. Sci. Technol. 47, 6582–6588. http://dx.doi.org/10.1021/es400671h.
Lund, M.A., Blanchette, M.L., 2014. Are we approaching pit lake closure from the wrong perspective?. In: Proceedings of 12th International Mine Water Association Congress, Xuzhou, China, Aug 18-22, pp. 720–724.
LPSDP, 2008. Water Management. Leading Practice Sustainable Development Program (LPSDP) For the Mining Industry. Australian Government. May 2008. 112pp. Available from: www.industry.gov.au/resource/Programs/LPSD/Pages/LPSDhandbooks.aspx (accessed 04-07-2016).
Mäkinen, J., Lerssi, J., 2007. Characteristics and seasonal variation of sediments in lake Junttiselkä, pyhäsalmi, Finland. Mine Water Environ. 26 (4), 217–228. http://dx.doi.org/10.1007/s10230-007-0015-3.
MCA, 2014. Water Accounting Framework for the Minerals Industry, User Guide, Version 1.3 – January 2014. Minerals Council of Australia (MCA).
McCullough, C.D., Marchand, G., Unseld, J., 2013. Mine closure of pit lakes as terminal sinks: best available practice when options are limited? Mine Water Environ. 32 (4), 302–313. http://dx.doi.org/10.1007/s10230-013-0235-7.
Mekonnen, M.M., Hoekstra, A.Y., 2016. Four billion people facing severe water scarcity. Sci. Adv. 2 (2), e1500323. http://dx.doi.org/10.1126/sciadv.1500323.
Mohan, D., Pittman, C.U., 2007. Arsenic removal from water/wastewater using adsorbents—A critical review. J. Hazard. Mater. 142 (1), 1–53. http://dx.doi.org/10.1016/j.jhazmat.2007.01.006.
Moreno, P.A., Aral, H., Cuevas, J., Monardes, A., Adaro, M., Norgate, T., Bruckard, W., 2011. The use of seawater as process water at Las Luces copper-molybdenum beneficiation plant in Taltal (Chile). Miner. Eng. 24, 852–858. http://

dx.doi.org/10.1016/j.mineng.2011.03.009.
Mudd, G.M., 2001a. Critical review of acid in situ leach uranium mining: 1. USA and Australia. Environ. Geol. 41, 390–403. http://dx.doi.org/10.1007/s002540100406.
Mudd, G.M., 2001b. Critical review of acid in situ leach uranium mining: 2. Soviet Block and Asia. Environ. Geol. 41, 404–416. http://dx.doi.org/10.1007/s002540100405.
Mudd, G.M., 2008. Sustainability reporting and water resources: a preliminary assessment of embodied water and sustainable mining. Mine Water Environ. 27 (3), 136–144. http://dx.doi.org/10.1007/s10230-008-0037-5.
Mudd, G.M., 2010. The environmental sustainability of mining in Australia: key mega-trends and looming constraints. Resour. Policy 35, 98–115. http://dx.doi.org/10.1016/j.resourpol.2009.12.001.
Mwale, A.H., Musonge, P., Fraser, D.M., 2005. The influence of particle size on energy consumption and water recovery in comminution and dewatering systems. Miner. Eng. 18, 915–926. http://dx.doi.org/10.1016/j.mineng.2005.02.014.
Napier-Munn, T.J., Morrison, R.D., 2003. The potential for the dry processing of ores. In: Proceedings of Water in Mining 2003, Brisbane, Qld, 13-15 Oct, pp. 247–250.
Newcrest, 2007. Update on Cadia Water Supply. ASX Market Release. Newcrest Mining Limited, Melbourne, Australia, 15 May 2007.
Newcrest, 2013. Cadia Valley Operations, Annual Environmental Management Report, 2014/2015. Newcrest Mining Limited, Melbourne, Australia.
Newcrest, 2014. Newcrest Mining Limited Sustainability Report 2013, GRI Data Tables. Newcrest Mining Limited, Melbourne, Australia.
Norgate, T.E., Lovel, R., 2004. Water Use in Metal Production: a Life Cycle Perspective. Commonwealth Scientific and Industrial Research Organisation (CSIRO), Australia. Report no. DMR2505.
Norgate, T.E., Lovel, R.R., 2006. Sustainable water use in minerals and metal production. In: Proceedings of Water in Mining Conference 2006, Brisbane, Australia, Nov 14-16, pp. 331–339.
Northey, S., Haque, N., 2013. Life cycle based water footprints of selected mineral and metal processing routes. In: Proceedings of Water in Mining 2013, Brisbane, Australia, Nov 26-28, pp. 35–50.
Northey, S., Haque, N., Mudd, G., 2013. Using sustainability reporting to assess the environmental footprint of copper mining. J. Clean. Prod. 40, 118–128. http://dx.doi.org/10.1016/j.jclepro.2012.09.027.
Northey, S.A., Haque, N., Lovel, R., Cooksey, M.A., 2014. Evaluating the application of water footprint methods to primary metal production systems. Miner. Eng. 69, 65–80. http://dx.doi.org/10.1016/j.mineng.2014.07.006.
Núñez, M., Pfister, S., Vargas, M., Antón, A., 2015. Spatial and temporal specific characterisation factors for water use impact assessment in Spain. Int. J. Life Cycle Assess. 20, 128–138. http://dx.doi.org/10.1007/s11367-014-0803-5.
Olivares, M., Toledo, M., Acuña, A., Garcés, M., 2012. Preliminary estimate of the water footprint of copper concentrate production in central Chile. In: Proceedings of the Third International Congress on Water Management in the Mining Industry, Santiago, Chile, 6–8 June 2012, Gecamin, pp. 390–400.
Palaniandy, S., Powell, M., 2014. Addressing water scarcity in mining through improved comminution practices. AusIMM Bull. 5 (48), 50–52. October 2014.
Peña, C.A., Huijbregts, M.A.J., 2014. The blue water footprint of primary copper production in Northern Chile. J. Industrial Ecol. 18 (1), 49–58. http://dx.doi.org/10.1111/jiec.12036.
Pfister, S., Bayer, P., 2014. Monthly water stress: spatially and temporally explicit consumptive water footprint of global crop production. J. Clean. Prod. 73, 52–62. http://dx.doi.org/10.1016/j.jclepro.2013.11.031.
Pfister, S., Koehler, S., Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Technol. 43, 1098–4104. http://dx.doi.org/10.1021/es802423e.
Pfister, S., Ridoutt, B.G., 2014. Water footprint: pitfalls on common ground. Environ. Sci. Technol. 48 http://dx.doi.org/10.1021/es405340a, pp. 4–4.
Pfister, S., Vionnet, S., Levova, T., Humbert, S., 2015. Ecoinvent 3: assessing water use in LCA and facilitating water footprinting. Int. J. Life Cycle Assess. 12. http://dx.doi.org/10.1007/s11367-015-0937-0.
QFCI, 2012. Chapter 13, mining. In: Final Report of the Queensland Floods Commission of Inquiry (QFCI), March 2012, Australia.
Ramirez-Llodra, E., Trannum, H.C., Evenset, A., Levin, L.A., Andersson, M., Finner, T.E., Hilario, A., Flem, B., Christensen, G., Schaanning, M., Vanreusel, A., 2015. Submarine and deep-sea mine tailings placements: a review of current practices, environmental issues, natural analogs and knowledge gaps in Norway and internationally. Mar. Pollut. Bull. 97, 13–35. http://dx.doi.org/10.1016/j.marpolbul.2015.05.062.
Ranchod, N., Sheridan, C.M., Pint, N., Slatter, K., Harding, K.G., 2015. Assessing the blue-water footprint of an opencast platinum mine in South Africa. Water SA 41 (2), 287–293. http://dx.doi.org/10.4314/wsa.v41i2.15.
Rapantova, N., Grmela, A., 2000. Water balance of coal mines and its determination under the conditions of humid climate of Central Europe. In: 7th International Mine Water Association Congress: Mine Water and the Environment. Ustroň, Poland, September 11-15 2000, pp. 615–623.
Reap, J., Roman, F., Duncan, S., Bras, B., 2008. A survey of unresolved problems in life cycle assessment. Part 2: impact assessment and interpretation. Int. J. Life Cycle Assess. 13, 374–388. http://dx.doi.org/10.1007/s11367-008-0009-9.
Reid, C., Bécaert, V., Aubertin, M., Rosenbaum, R.K., Deschênes, L., 2009. Life cycle assessment of mine tailings management in Canada. J. Clean. Prod. 17, 471–479. http://dx.doi.org/10.1016/j.jclepro.2008.08.014.
Ridoutt, B.G., Huang, J., 2014. Environmental relevance – the key to understanding water footprints. Proc. Natl. Acad. Sci. (PNAS) 109 (22), E1424. http://dx.doi.org/10.1073/pnas.1203809109.
Ridoutt, B.G., Pfister, S., 2013a. Towards an integrated family of footprint indicators. J. Industrial Ecol. 17 (3), 337–339. http://dx.doi.org/10.1111/jiec.12026.
Ridoutt, B.G., Pfister, S., 2013b. A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator. Int. J. Life Cycle Assess. 18, 204–207. http://dx.doi.org/10.1007/s11367-012-0458-z.
Ross, S., Evans, D., 2002. Excluding site-specific data from the LCA inventory: how this affects life cycle impact assessment. Int. J. Life Cycle Assess. 7 (3), 141–150. http://dx.doi.org/10.1007/BF02994048.
Saarivuori, E., Wessman-Jääskeläinen, H., 2016. Water Availability Footprint of a Mine Product: Case Study. VTT Technical Research Centre of Finland. Unpublished Results.
Sahoo, L.J., Bandyopadhyay, S., Banerjee, R., 2014. Water and energy assessment for dewatering in opencast mines. J. Clean. Prod. 84, 736–745. http://dx.doi.org/10.1016/j.jclepro.2014.07.072.
Santero, N., Hendry, J., 2016. Harmonization of LCA methodologies for the metal and mining industry. Int. J. Life Cycle Assess. 11. http://dx.doi.org/10.1007/s11367-015-1022-4.
Sarver, E.A., Cox, A.S., 2013. Salt: an emerging water concern for the global mining and minerals industries. Min. Technol. 122 (3), 145–152. http://dx.doi.org/10.1179/147490013X13639459465655.
Scherer, L., Pfister, S., 2015. Modelling spatially explicit impacts from phosphorus emissions in agriculture. Int. J. Life Cycle Assess. 20 (6), 785–795. http://dx.doi.org/10.1007/s11367-015-0880-0.
Scherer, L., Venkatesh, A., Karuppiah, R., Pfister, S., 2015. Large-scale hydrological modelling for calculating water stress indices: implications of improved spatiotemporal resolution, surface-groundwater differentiation, and uncertainty characterization. Environ. Sci. Technol. 49, 4971–4979. http://dx.doi.org/10.1021/acs.est.5b00429.
Schultze, M., 2012. Maximising use of water produced by mining. In: International Mine Water Association 2012, Bunbury, Australia, pp. 11–19.
Sheppard, G., Gallardo, A., Hall, J., Strizek, M., 2009. Managing the cumulative groundwater impacts of multiple iron ore projects at Cape Preston (or … one plus one plus one does not equal four). Min. Technol. 118 (3/4), 152–161. http://dx.doi.org/10.1179/174328610X12682159814911.
Sinclair, L., Thompson, J., 2015. In situ leaching of copper: challenges and future prospects. Hydrometallurgy 57, 306–324. http://dx.doi.org/10.1016/j.hydromet.2015.08.022.
Soni, A.K., Wolkersdorfer, C., 2016. Mine water: policy perspective for improving water management in the mining environment with respect to developing economies. Int. J. Min. Reclam. Environ. 30 (2), 115–127. http://dx.doi.org/10.1080/17480930.2015.1011372.
Steubing, B., Wernet, G., Reinhard, J., Bauer, C., Moreno-Ruiz, E., 2016. The ecoinvent database version 3 (part II): analyzing LCA results and comparison to version 2. Int. J. Life Cycle Assess. 13. http://dx.doi.org/10.1007/s11367-016-1109-6.
SYKE, 2015. Groundwater Checklist. Suomen Ympäristökeskus (SYKE) – Finnish Environment Institute. Available at: www.syke.fi/groundwaterchecklist (accessed 04-07-2016).
Torrisi, C., Trotta, P., 2009. Sustainable water use at Olympic dam. Min. Technol. 118 (3/4), 193–204. http://dx.doi.org/10.1179/174328610X12682159814984.
TSM, 2016. Towards sustainable mining (TSM) initiative. In: The Mining Association of Canada. mining.Ca/towards-sustainable-mining (accessed 04-07-2016).
Tuominen, S.M., Kauppi, S.H., Nystén, T.H., 2016. Poster, abstract : groundwater Checklist - metadata tool for groundwater protection and interaction. In: Staboulis, S., Karvonen, T., Kujanpää, A. (Eds.), Bulletin of the Geological Society of Finland, Special Volume, Abstracts of the 32nd Nordic Geological Winter Meeting, 13th-15th January 2016. Geological Society of Finland, Helsinki, Finland.
Turpeinen, A., Rainio, R., 2013. Talvivaaraselvitys. Ympäristöministeriön raportteja 2/2013. [In Finnish].
Vick, S.G., 1990. Planning, Design and Analysis of Tailings Dams. BiTech Publishers, Canada.
Vogwill, R.I.J., Williamson, S., Kote, J., 2009. Hydrogeology of the coastal mining area, Lihir goldmine, Papua New Guinea. Aust. J. Earth Sci. 56 (1), 41–52. http://dx.doi.org/10.1080/08120090802541929.
Välisalo, T., 2014. Kaivosten Stressitestit 2013 ([Stress tests for mines in 2013]). Finnish Ministry of Environment Report 2/2014. [In Finnish].
Wada, Y., van Beek, L.P.H., Viviroli, D., Dürr, H.H., Weingartner, R., Bierkens, M.F.P., 2011. Global monthly water stress: 2. Water demand and severity of water stress. Water Resour. Res. 47 (W07518) http://dx.doi.org/10.1029/2010WR009792, 17.
Wels, C., Robertson, A.M., 2003. Conceptual model for estimating water recovery in tailings impoundments. Tailings Mine Wastes Proc. Tenth Int. Conf. 15, 87–94.
Welsh, W.D., 2006. Great Artesian Basin Transient Groundwater Model. Bureau of Rural Sciences, Australian Government.
Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 13. http://dx.doi.org/10.1007/s11367-016-1087-8.
Wessman, H., Salmi, O., Kohl, J., Kinnunen, P., Saarivuori, E., Mroueh, U.-M., 2014. Water and Society: mutual challenges for eco-efficient and socially acceptable mining in Finland. J. Clean. Prod. 84, 289–298. http://dx.doi.org/10.1016/j.jclepro.2014.04.026.
Williamson, S., Vogwill, R.I.J., 2001. Dewatering in the hot groundwater conditions at Lihir Gold. In: International Mine Water Association Symposium, Belo

Horizonte, Brazil, 15pp.
Wolkersdorfer, C., Bowell, R. (Eds.), 2004. Contemporary Reviews of Mine Water Studies in Europe, Part 1. Mine Water and the Environment, 23, pp. 162–182. http://dx.doi.org/10.1007/s10230-004-0060-0.
Wolkersdorfer, C., Bowell, R. (Eds.), 2005a. Contemporary Reviews of Mine Water Studies in Europe, Part 2. Mine Water and the Environment, 24, pp. 2–37. http://dx.doi.org/10.1007/s10230-005-0068-0, 1.
Wolkersdorfer, C., Bowell, R. (Eds.), 2005b. Contemporary Review of Mine Water Studies in Europe, Part 3. Mine Water and the Environment, 24, pp. 58–76. http://dx.doi.org/10.1007/s10230-005-0074-2, 2.
WULCA, 2015. Consensual Method Development: Preliminary Recommendation – AWaRe v1.1. Water Use in LCA (WULCA) working group. Accessible from: www.wulca-waterlca.org (accessed 10-11-2015).
Yang, X.J., Lin, A., Li, X.-L., Wu, Y., Zhou, W., Chen, Z., 2013. China's ion-adsorption rare earth resources, mining consequences and preservation. Environ. Dev. 8, 131–136. http://dx.doi.org/10.1016/j.envdev.2013.03.006.
Yano, S., Hanasaki, N., Itsubo, N., Oki, T., 2015. Water scarcity footprints by considering the differences in water sources. Sustainability 7, 9753–9772. http://dx.doi.org/10.3390/su7089753.
Younger, P.L., 1997. The longevity of minewater pollution: a basis for decision-making. Sci. Total Environ. 194 (195), 457–466. http://dx.doi.org/10.1016/S0048-9697(96) 05383–1.
Younger, P.L., 2006. The water footprint of mining operations in space and time – a new paradigm for sustainability assessments?. In: Proceedings of Water in Mining 2006 (AusIMM), Brisbane, Australia, 14-16 November, pp. 13–22.
Younger, P.L., Banwart, S.A., Hedin, R.S., 2002. Mine Water: Hydrology, Pollution, Remediation. [Book]. Environmental Pollution Series, vol. 5. Springer, The Netherlands. http://dx.doi.org/10.1007/978-94-010-0610-1.

3.1. Other Water Footprint related Studies of the Mining Industry

In the eighteen months since the preceding journal article was written, several additional studies have been published or brought to the authors attention that focus upon the water footprint of mining and metal production systems. Therefore, for completeness sake, an extension to the article's Table 3 is provided below in Table 1.

Table 1: List of additional identified studies that provide water footprint related information for the mining industry. General life cycle assessment studies for mining that include, but are not focused upon, water use have been excluded.

Year	Authors	Region	Commodities	Focus	Accounted for? Indirect Water	Scarcity[a]	Quality[b]	Methodological Approach
2017	Osman et al.	South Africa	Co, Cu, Ni, PGM	Facility	No	-	GW	MCA (2014)'s Water Accounting Framework in combination with the Water Footprint Network approach (Hoekstra et al., 2011)
2016	Echevveri & Restrepo	Colombia	Cement	Facility	Yes	GWT	GW	Water Footprint Network approach (Hoekstra et al., 2011)
2013	TATA	India	Coal, Fe, Lime	Facility	Yes	B	GW	Water Footprint Network approach (Hoekstra et al., 2011)
2013	Zhang & Anadon	China	Coal, Coke + other energy products	Commodity	Yes	L	-	Multi-region input-output model. End-point damage category modelling based upon Pfister et al. (2009).

Notes
[a] B = Basin assessment, GWT = WBSCD Global Water Tool (2015), L = Percentage share of local withdrawal, consumption & discharge.
[b] GW = Grey Water (Hoekstra et al., 2009)

4. The Exposure of Global Base Metal Resources to Water Criticality, Scarcity and Climate Change

Recent life cycle assessment based methods for quantifying consumptive water use impacts require an understanding of the location of production facilities and their local water use context. Despite the potential impact of mining operations on water resources, the dependence of these operations on having stable access to water supplies, as well as the vulnerability of these operations to extreme weather events, there is very limited quantitative understanding currently of how the mining industry as a whole is distributed in relation to local water and climate contexts.

The spatial distribution of global base metal resources was assessed against a range of regional range of spatial water indices that describe local hydrological regimes, water use contexts and climate types. Spatial indicators such as water 'criticality' (Sonderegger et al., 2017) were used to provide insight into how base metal resources intersect with water resource risks in specific regions. The resources were also assessed against several of the spatial characterisation factors that are available for assessing water use impacts in life cycle assessment. Understanding how the mining industry is spatially distributed in relation to these spatial impact characterisation factors will facilitate improved understanding of the relative impacts that are associated with water consumption throughout the industry.

Detailed results and datasets were developed that enable evaluation of the local water and climate contexts facing the base metal mining industry at various levels of detail, including for specific countries, mineral deposit types and also individual mineral deposits. The results of the assessment provide a rich perspective of the relative water risks and contexts of regions containing copper, nickel and lead-zinc resources. A broad ranging discussion is also provided of the conceptual risks associated with water stress, scarcity and climate regimes. As the overall water balance of an individual mine site is heavily influenced by the local climate regime, an improved understanding of how the mining industry is distributed across climate regimes and whether these may change into the future will facilitate more accurate assessment the potential impacts (positive or negative) of climate change on the industry.

The contents of this chapter were published in Global Environmental Change and are presented in the original format of the journal. The electronic supplementary figures or tables referred to in the article have been presented or described further in Appendix A of this thesis.

Reference:

Northey, S.A., Mudd, G.M., Werner, T.T., Jowitt, S.M., Haque, N., Yellishetty, M., Weng, Z. (2017). The exposure of global base metal resources to water criticality, scarcity and climate change. Global Environmental Change, 44, pp. 109-124. http://dx.doi.org/10.1016/j.gloenvcha.2017.04.004

Global Environmental Change 44 (2017) 109–124

Contents lists available at ScienceDirect

Global Environmental Change

journal homepage: www.elsevier.com/locate/gloenvcha

The exposure of global base metal resources to water criticality, scarcity and climate change

Stephen A. Northey[a,b,*], Gavin M. Mudd[a,1], Timothy T. Werner[a], Simon M. Jowitt[c], Nawshad Haque[b], Mohan Yellishetty[a], Zhehan Weng[a]

[a] Faculty of Engineering, Monash University, Clayton, VIC, Australia
[b] CSIRO Mineral Resources, Clayton, VIC, Australia
[c] Department of Geoscience, University of Nevada, Las Vegas, 4540 S Maryland Pkwy, NV 89154-4010, USA

ARTICLE INFO

Keywords:
Base metal mining
Copper
Lead-zinc
Nickel
Water risk
Climate change

ABSTRACT

Mining operations are vital to sustaining our modern way of life and are often located in areas that have limited water supplies or are at an increased risk of the effects of climate change. However, few studies have considered the interactions between the mining industry and water resources on a global scale. These interactions are often complex and site specific, and so an understanding of the local water contexts of individual mining projects is required before associated risks can be adequately assessed. Here, we address this important issue by providing the first quantitative assessment of the contextual water risks facing the global base metal mining industry, focusing on the location of known copper, lead, zinc and nickel resources.

The relative exposure of copper, lead-zinc and nickel resources to water risks were assessed by considering a variety of spatial water indices, with each providing a different perspective of contextual water risks. Provincial data was considered for water criticality (CRIT), supply risk (SR), vulnerability to supply restrictions (VSR) and the environmental implications (EI) of water use. Additionally, watershed or sub-basin scale data for blue water scarcity (BWS), the water stress index (WSI), the available water remaining (AWaRe), basin internal evaporation recycling (BIER) ratios and the water depletion index (WDI) were also considered, as these have particular relevance for life cycle assessment and water footprint studies. All of the indices indicate that global copper resources are more exposed to water risks than lead-zinc or nickel resources, in part due to the large copper endowment of countries such as Chile and Peru that experience high water criticality, stress and scarcity. Copper resources are located in regions where water consumption is more likely to contribute to long-term decreases in water availability and also where evaporation is less likely to re-precipitate in the same drainage basin to cause surface-runoff or groundwater recharge.

The global resource datasets were also assessed against regional Köppen-Geiger climate classifications for the observed period 1951–2000 and changes to 2100 using the Intergovernmental Panel on Climate Change's A1FI, A2, B1 and B2 emission scenarios. The results indicate that regions containing copper resources are also more exposed to likely changes in climate than those containing lead-zinc or nickel resources. Overall, regions containing 27–32% (473–574 Mt Cu) of copper, 17–29% (139–241 Mt Pb + Zn) of lead-zinc and 6–13% (19–39 Mt Ni) of nickel resources may have a major climate re-classification as a result of anthropogenic climate change. A further 15–23% (262–412 Mt) of copper, 23–32% (195–270 Mt) of lead-zinc and 29–32% (84–94 Mt) of nickel are exposed to regional precipitation or temperature sub-classification changes. These climate changes are likely to alter the water balance, water quality and infrastructure risks at mining and mineral processing operations. Effective management of long-term changes to mine site water and climate risks requires the further adoption of anticipatory risk management strategies.

1. Introduction

The mining industry spans all hydrological contexts and climate regions, with these contexts influencing the water risks facing mining operations and the potential for the industry to impact surrounding ecosystems, industries and communities. Access to water is a potential

* Corresponding author at: Faculty of Engineering, Monash University, Clayton, VIC, Australia.
E-mail addresses: stephen.northey@monash.edu, stephen.northey@csiro.au, stephen.northey@gmail.com (S.A. Northey).
[1] Present address: Environmental Engineering, RMIT University, Melbourne, Australia.

http://dx.doi.org/10.1016/j.gloenvcha.2017.04.004
Received 17 October 2016; Received in revised form 22 March 2017; Accepted 10 April 2017
0959-3780/

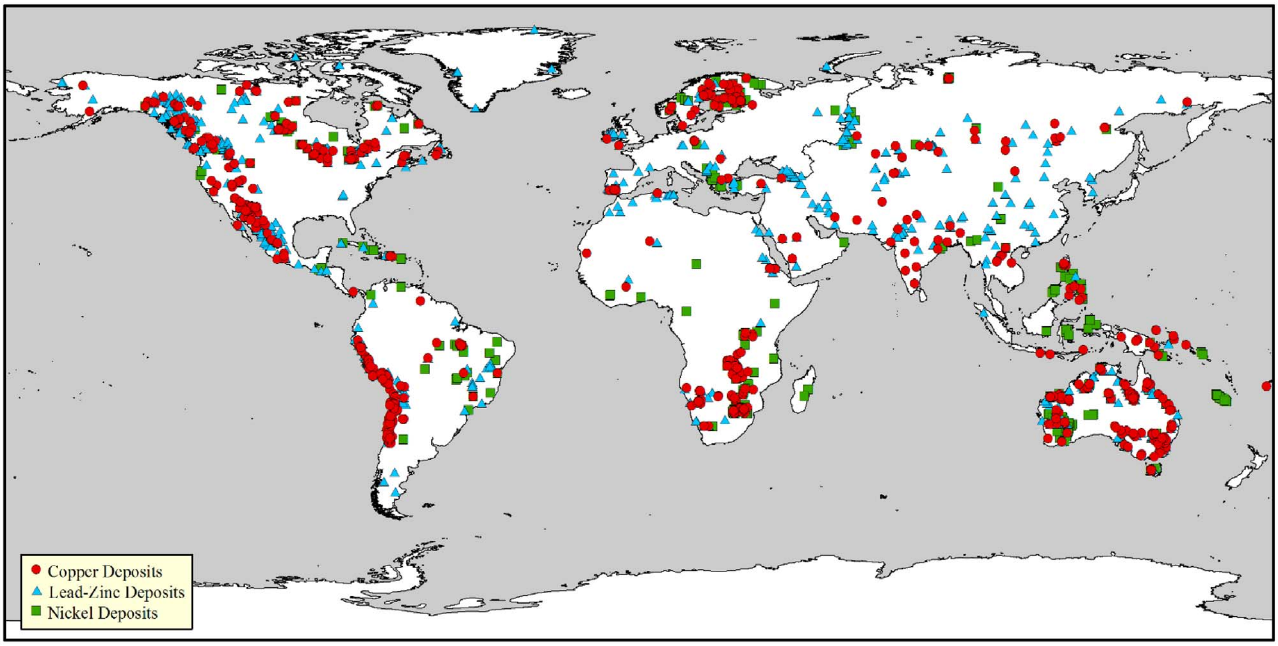

Fig. 1. Location of copper, lead-zinc and nickel resources considered in this study. Maps showing mineral deposit types and operating status are shown in electronic supplementary Figs. S.1–S.5. Raw data provided in supplementary Tables S.24–S.26 and in Google Earth format (.kmz).

constraining factor on mineral resource development, regardless of the climate and absolute water scarcity of a region. The presence of other competing water users, such as agriculture, may limit the ability to allocate water resources to the mining industry (e.g. Shang et al., 2016). Concerns over water may also change community support and reduce a mine's perceived social license to operate (Wessman et al., 2014). Assessing these risks requires the use of systems approaches that can integrate mine site water balances, catchment hydrology and the water use requirements of regions.

Mine sites utilise water in a range of processes, such as mineral processing and dust suppression, and the overall water requirements are highly variable due to factors such as: the local climate, ore mineralogy and grade, the scale of infrastructure and ore processing, and the extent of tailings dewatering and water recycling (Gunson et al., 2012; Mudd, 2008; Northey et al., 2013, 2014a, 2016). The local nature of mine site water use impacts has impeded the ability to produce global scale assessments of the water risks associated with the industry. Previous research has outlined global estimates of the water withdrawals associated with non-fuel mining (Gunson, 2013), however drawing meaningful conclusions requires understanding where this water use occurs. Improving the outcomes of these studies requires knowledge of how the spatial distribution of the mining industry relates to local contexts and environmental pressures. Global assessments have been conducted to assess the distribution of the mining industry in relation to biodiversity and conservation areas (Durán et al., 2013; Murguía et al., 2016). However, to date there have been no quantitative global assessments of the contextual water risks facing the mining industry.

This article presents a detailed assessment of the spatial distribution of known base metal resources in relation to a variety of water risk and impact indices. The assessment focuses on copper, lead-zinc and nickel resources as these metals are vital for modern infrastructure and are expected to have continued or growing demand into the future (Daigo et al., 2014; Elshkaki et al., 2016; Kleijn et al., 2011). The exposure of regions containing these resources to climate change has also been assessed by considering regional data for Köppen-Geiger climate classifications and how these may evolve with climate change (Kottek et al., 2006; Rubel and Kottek, 2010). The information and data provided by this study may form a basis for further assessment of the global base metals industry to understand climate change adaptation requirements, water footprint or life cycle impacts, and expected changes to mine site water balance, water quality and infrastructure risks.

2. Methods and data sources

2.1. Copper, lead-zinc and nickel resource datasets

This study utilises datasets for individual copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017) and nickel (Mudd and Jowitt, 2014) resources that were developed over several years and are primarily based upon the mineral resource reporting of individual exploration and mining companies. Typically these resource disclosures are made as part of a company's statutory or financial reporting obligations. The copper dataset includes resource data for 730 deposits containing 1781 million tonnes (Mt) of copper (363,270 Mt ore @ 0.49% Cu; Mudd et al., 2013). The lead-zinc dataset includes resource data for 852 deposits representing a combined resource of 837 Mt lead-zinc (50,882 Mt ore @ 1.64% Pb + Zn; Mudd et al., 2017). While, the nickel dataset includes data for 476 deposits containing 293 Mt of nickel (61,365 Mt ore @ 0.48% Ni; Mudd and Jowitt, 2014). Individual deposits in these datasets have been classified according to primary and/or dominant mineral deposit types (e.g. Jowitt et al., 2013). Individual resources in the datasets have also been classified as being either an undeveloped deposit or a recently operating mine-site, based upon the status of the deposit for the year the dataset was compiled (Copper: 2010; Lead-Zinc: 2013; Nickel: 2011). The datasets provide minimum estimates of known resources and so the results presented represent the minimum exposed resource to the various water and climate risks.

Coordinate data (latitude and longitude) for individual deposits within the datasets have been added and crosschecked from a range of sources, including government geological organisations (the United States Geological Survey, the British Columbia Geological Survey, Geoscience Australia, Geological Survey of Finland, etc.), online resources (company websites, mindat.org, dmgeode.com, etc.), consultant databases (e.g. SNL database), scholarly literature (journals, conference proceedings, books, etc.), and company technical reports

Table 1
Regional water indices considered by the study.

Abbreviation	Name	Description	Spatial Data Coverage: Contained Cu, Pb + Zn, Ni
CRIT	Water Criticality [1]	Composite water risk indicator that is a function of the SR, VSR and EI indices described below. Measured between 0 and 100.	100%, 96%, 94%
SR	Supply Risk [1]	An index of physical water supply risks, as well as water governance and upstream geopolitical risks. Measured between 0 and 100.	100%, 97%, 95%
VSR	Vulnerability to Supply Restrictions [1]	An index combining the economic importance of water, the ability to compensate for supply restrictions and the general susceptibility of the region. Measured between 0 and 100.	100%, 96%, 94%
EI	Environmental Implications [1]	The potential environmental impacts associated with utilising water in a region – based upon life cycle impact assessment procedures. Measured between 0 and 100.	100%, 96%, 94%
AWaRe	Available Water Remaining [2][3]	The inverse of regional availability minus demand associated with environmental flow requirements and human consumption. AWaRe data specific to non-agricultural water use have been used. Values are normalised between 0.01 and 100, relative to a global average of 1.	100%, 100%, 100%
BWS	Blue water scarcity [4]	The ratio of the domestic consumption of blue water to the availability of blue water within a region.	55%, 68%, 47%
BIER	Basin Internal Evaporation Recycling [5]	The ratio of evaporation that is re-precipitated elsewhere within the same water basin.	100%, 100%, 100%
BIER-h	Hydrologically Effective Basin Internal Evaporation Recycling [5]	The ratio of evaporation that is re-precipitated and causes surface runoff or groundwater recharge elsewhere within the same water basin.	100%, 100%, 100%
WDI	Water Depletion Index [5]	The vulnerability of a basin to freshwater depletion. Accounts for consumption-to-availability ratios, surface and groundwater stocks and the overall aridity of the region. Normalised between 0.01 and 1.	100%, 100%, 100%
WSI	Water Stress Index [6]	A function of a region's water withdrawals, long-term water availability, and inter- and intra-annual precipitation variability. Measured between 0.01 and 1.	100%, 100%, 99%
WTA	Withdrawal-to-Availability ratio [7]	The ratio of water withdrawals in a region to the region's long-term water availability.	100%, 100%, 100%

Data sources: [1] Sonderegger et al. (2015); [2] WULCA (2016); [3] Boulay et al. (2016a, 2016b); [4] Hoekstra et al. (2012); [5] Berger et al. (2014); [6] Pfister et al. (2009); [7] Alcamo et al. (2003).

(Canadian National Instrument 43–101 reporting in the SEDAR database, JORC reporting, etc.). Indian copper deposits were aggregated at the state level during original data compilation and so coordinates for these was assigned based upon the approximate mid-point of each state. Fig. 1 shows the spatial distribution of copper, lead-zinc and nickel deposits considered in this study. More detailed maps showing mineral deposit types and operating status are shown in electronic supplementary Figs. S.1–S.5.

The contribution of copper, lead-zinc and nickel to the total economic value of the individual mineral resources was estimated based upon average metal price data for the period 2006–2010 (USGS, 2013). This also provides a coarse indication of potential metal co-products that may be extracted when developing these resources and could potentially provide a basis for economic allocation within water footprint or life cycle assessment studies.

2.2. Regional water indices

Exposure to water risks was assessed by considering regional data for a range of water risk and impact indices (Table 1 and Fig. 2), with further descriptions of these indices provided in the Supplementary information. Consideration of these indices alongside each other provides a richer perspective of regional water contexts than could be achieved by considering each index in isolation. The reader is encouraged to refer to the associated references for detailed information on the conceptualisation and data underpinnings of each index.

Values for each index were assigned to individual deposits or resources using GIS software. Weighted average index values were then determined for the copper, lead-zinc and nickel resources at several different scales – globally, for individual countries and for primary mineral deposit types. These were calculated according to Eq. (1), where I represents the regional index value and R represents the contained copper, nickel or combined lead-zinc metal tonnage associated with each mineral deposit/resource i. Weighted averages calculated using mineralised ore tonnages in place of contained metal tonnages are presented in the electronic Supplementary information for comparison.

$$ResourceWeightedAverage = \frac{\sum_i I_i \times R_i}{\sum_i R_i} \tag{1}$$

There are some limitations regarding the extent of spatial coverage for several of the indices (see Table 1). Where data has not been available, resources in that region have been excluded from subsequent calculations of statistics for that index.

2.3. Köppen-Geiger climate classifications

The Köppen-Geiger climate classification scheme was first described by Köppen (1900) before being modified by Geiger (1954), and more recently global climate maps based upon modern temperature and precipitation monitoring data have been developed (e.g. Kottek et al., 2006; Peel et al., 2007). Under the Köppen-Geiger climate classification scheme, regions are assigned major classifications based upon temperature ranges, or for the case of arid regions, precipitation levels. Precipitation and temperature sub-classifications are also assigned to provide further detail and seasonal information of the local climate. The scheme utilises five major climate classifications, six precipitation sub-classifications and eight temperature sub-classifications – which are often abbreviated using a short-hand lettering scheme. The temperature and precipitation thresholds used to assign regional climate classifications are shown in Table 2.

The resource datasets were assessed against global maps of historic and future Köppen-Geiger climate classifications. The basis of this analysis is the global climate classification dataset for 1951 to 2000 developed by Kottek et al. (2006) on a 0.5° by 0.5° latitude-longitude grid. The exposure of base metal resources to climate change was assessed for the IPCC emissions scenarios A1FI, A2, B1 and B2 (IPCC, 2000). These emission scenarios correspond to a greater than 66% probability of global temperature increases above pre-industrial levels of 4.1–6.2 °C (A1FI), 3.5–5.2 °C (A2), 2.0–3.2 °C (B1) and 2.6–3.7 °C (B2) respectively by 2100 (Rogelj et al., 2012). Rubel and Kottek (2010) provide spatial data for these scenarios for the periods 2001–2025,

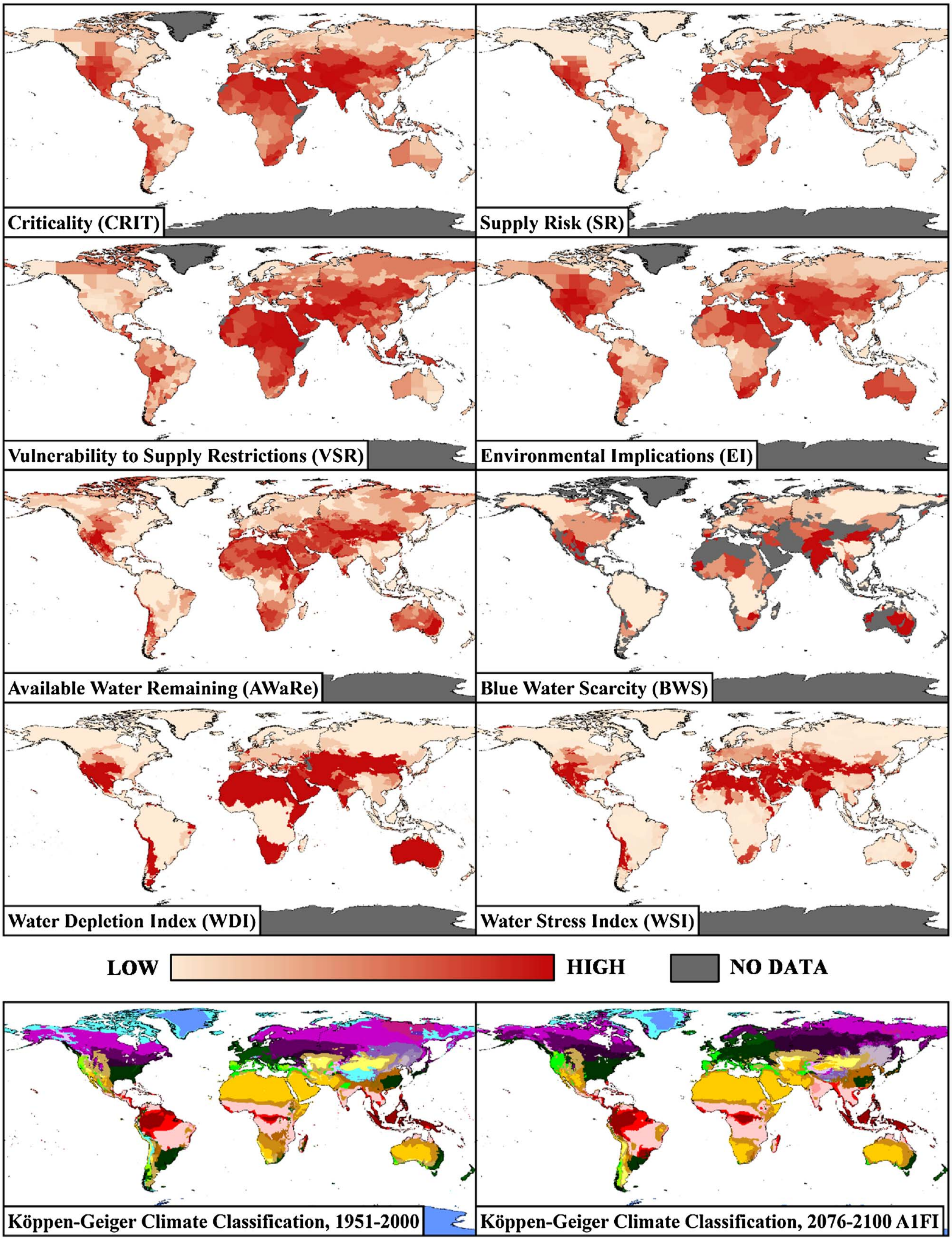

Fig. 2. Water indices and climate classifications used to understand the contextual water risks in regions containing copper, lead-zinc and nickel resources. Data sources: CRIT, SR, VSR, EI (Sonderegger et al., 2015), AWaRe (Boulay et al., 2016a, 2016b; WULCA, 2016), BWS (Hoekstra et al., 2012), WDI (Berger et al., 2014), WSI (Pfister et al., 2009), Köppen-Geiger climate classifications for the observed period 1951–2000 (Rubel and Kottek, 2010) and the IPCC scenario A1FI for the period 2076–2100 (Rubel and Kottek, 2010). BIER and BIER-h are not shown, but are presented in Berger et al. (2014).

Table 2
Temperature and precipitation thresholds for the version of the Köppen-Geiger climate classification scheme utilised by Kottek et al. (2006) and Rubel and Kottek (2010). Regional climates should be assessed against the criteria for E followed by B and then subsequently A, C or D.

Major Climate Classifications					
Precipitation Sub-classifications			Temperature Sub-classifications		
		A: Equatorial ($T_{min} \geq + 18\,^{\circ}C$)			
f	Fully humid	$P_{min} \geq 60$ mm			
m	Monsoon	$P_{ann} \geq 25(100-P_{min})$			
s	Dry summer	$P_{min} < 60$ mm in summer			
w	Dry winter	$P_{min} < 60$ mm in winter			
		B: Arid ($P_{ann} < 10\,P_{th}$)			
S	Steppe	$P_{ann} > 5\,P_{th}$	**h**	Hot	$T_{ann} \geq + 18\,^{\circ}C$
W	Desert	$P_{ann} \leq 5\,P_{th}$	**k**	Cold	$T_{ann} < + 18\,^{\circ}C$
		C: Warm Temperate ($-3\,^{\circ}C < T_{min} < + 18\,^{\circ}C$)			
s	Dry summer	$P_{smin} < P_{wmin}$, $P_{wmax} > 3\,P_{smin}$ and $P_{smin} < 40$ mm	**a**	Hot summer	$T_{max} \geq + 22\,^{\circ}C$
w	Dry winter	$P_{wmin} < P_{smin}$ and $P_{smax} > 10\,P_{wmin}$	**b**	Warm summer	Not (**a**) and at least 4 $T_{mon} \geq + 10\,^{\circ}C$
f	Fully humid	Neither **Cs** nor **Cw**	**c**	Cool summer and cold winter	Not (**b**) and $T_{min} > -38\,^{\circ}C$
		D: Snow ($T_{min} \leq -3\,^{\circ}C$)			
s	Dry summer	$P_{smin} < P_{wmin}$, $P_{wmax} > 3\,P_{smin}$ and $P_{smin} < 40$ mm	**a**	Hot summer	$T_{max} \geq + 22\,^{\circ}C$
w	Drys winter	$P_{wmin} < P_{smin}$ and $P_{smax} > 10\,P_{wmin}$	**b**	Warm summer	Not (**a**) and at least 4 $T_{mon} \geq + 10\,^{\circ}C$
f	Fully humid	Neither **Ds** nor **Dw**	**c**	Cool summer and cold winter	Not (**b**) and $T_{min} > -38\,^{\circ}C$
			d	Extremely continental	Like (**c**) but $T_{min} \leq -38\,^{\circ}C$
		E: Polar ($T_{max} < + 10\,^{\circ}C$)			
			F	Frost	$0\,^{\circ}C \leq T_{max} < +10\,^{\circ}C$
			T	Tundra	$T_{max} < 0\,^{\circ}C$

Nomenclature: T_{ann} – Annual mean near-surface temperature; T_{min} – Coldest month mean temperature; T_{max} – Warmest month mean temperature; T_{mon} – Monthly mean temperature; P_{ann} – Annual Precipitation; P_{min} – Lowest monthly precipitation; P_{smin} – Lowest monthly precipitation in the summer half-year; P_{wmin} – Lowest monthly precipitation in the winter half-year; P_{smax} – Highest monthly precipitation in the summer half-year; P_{wmax} – Highest monthly precipitation in the winter half-year; P_{th} – Dryness threshold (function of annual temperature and precipitation seasonality, see Kottek et al., 2006).

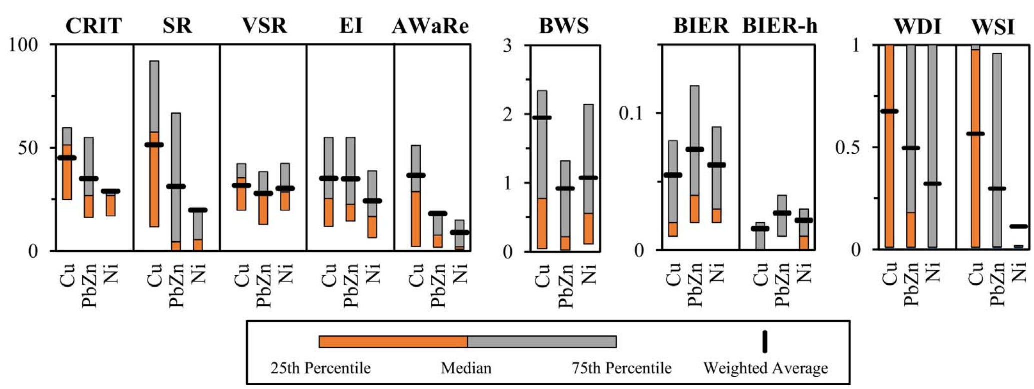

Fig. 3. Weighted average, median and interquartile range for Water Criticality (CRIT), Supply Risk (SR), Vulnerability to Supply Restrictions (VSR), Environmental Implications (EI), Available Water Remaining (AWaRe), Blue Water Scarcity (BWS), Basin Internal Evaporation Recycling (BIER), hydrologically effective Basin Internal Evaporation Recycling (BIER-h), Water Depletion Index (WDI) and the Water Stress Index (WSI). Statistics determined based upon contained metal tonnages of copper, lead-zinc and nickel resources.

2026–2050, 2051–2075, and 2076–2100–based upon averaging the monthly ensemble mean of 5 general circulation models (CGCM2, CSIRO2, HadCM3, PCM, ECHam4) that are available in the TYN SC 2.03 dataset (Mitchell et al., 2004).[1] Changes in the climate classification of regions containing base metal resources for these time periods and scenarios relative to the observed period 1951–2000.

Considerable uncertainty exists when modelling Köppen-Geiger classifications at the global scale, particularly at the level of temperature and precipitation sub-classifications. McMahon et al. (2015) assessed the average accuracy of GCMs to reproduce historical climate classifications (from 1950 to 1999). The proportion of grid cells across 46 GCM runs that were correctly assigned was 77% for a single letter classification, 57% for a two letter classification and 48% for a three letter classification. Additionally, temporal shifts in the boundaries of climate zones are poorly modelled by GCMs (Zhang and Yan, 2016). Therefore considerable care should be taken when interpreting the results presented by this study, particularly the detailed results for individual deposits and countries that are inherently more uncertain than the average results for each commodity.

3. Results

Summary results describing the spatial distribution of copper, lead-zinc and nickel resources in relation to the various water indices and climate classifications are provided in the following sections. Further results, figures and detailed datasets are provided in the electronic Supplementary information.

[1] Mitchell et al. (2004) describes the development of the TYN SC 2.00 dataset that only included four general circulation models (GCM). The ECHam4 GCM was subsequently added in version 2.03 of this dataset.

Table 3
Summary of results for major deposit types and countries with large resource endowments. More detailed results for all countries and weighted averages calculated on an ore tonnage basis are available in the Supplementary information. PGM refers to platinum group metals (Platinum, Palladium, Rhodium and Rhenium).

						Contained Value (%)		Weighted Averages, Metal Basis									
	Copper	**No.**	**Mt Ore**	**% Cu**	**Mt Cu**	**Cu**	**Other metals >5%**	CRIT	SR	VSR	EI	AWaRe	BWS	BIER	BIER-h	WDI	WSI
	Total	730	363,270	0.49	1,781	53	Au, PGM, Ni	45	51	32	35	37	1.95	0.05	0.02	0.68	0.56
	Undeveloped Deposits	468	136,945	0.43	586	48	Au, PGM, Mo, Ni	36	31	29	34	25	0.82	0.08	0.03	0.48	0.35
	Recently Operating	262	226,325	0.53	1,195	55	PGM, Au, Ni	49	61	33	36	42	0.75	0.04	0.01	0.77	0.67
	Deposit Type																
1	Porphyry	200	291,016	0.45	1,317	70	Au, Mo	51	64	32	39	45	2.69	0.04	0.01	0.78	0.72
2	Sediment-hosted Cu	62	10,874	1.52	165	74	Co	27	14	41	11	6	0.19	0.18	0.04	0.08	0.04
3	Iron Oxide Copper Gold	51	17,730	0.71	125	53	Au, U3O8	26	10	19	33	27	1.30	0.03	0.01	0.88	0.15
4	Magmatic Sulfide	133	26,543	0.29	76	9	PGM, Ni, Au	25	14	24	23	6	0.78	0.07	0.03	0.16	0.09
5	Skarn	39	4,901	0.70	34	71	Zn, Mo	39	25	35	50	7	0.23	0.13	0.03	0.07	0.08
6	Volcanogenic Massive Sulfide	144	4,042	0.78	32	50	Zn, Au	30	25	23	30	19	1.26	0.05	0.02	0.47	0.25
7	Other/Miscellaneous	50	2,701	0.59	16	54	Ni, Au	65	81	38	60	43	2.44	0.05	0.02	0.74	0.74
8	Sediment-hosted Pb-Zn	21	2,746	0.39	11	34	Ni, Zn, Co	16	3	12	23	11	0.31	0.05	0.01	0.79	0.03
9	Epithermal	30	2,717	0.18	5	40	Zn, Au, Pb	39	36	29	44	21	0.47	0.07	0.02	0.58	0.57
	Country																
1	Chile	51	122,768	0.54	658	89	Au	61	91	37	34	57	4.00	0.01	0.00	1.00	0.99
2	USA	56	49,427	0.34	170	53	Au, Mo, Ni, PGM	48	51	7	64	57	1.66	0.07	0.01	0.64	0.62
3	Peru	52	35,088	0.48	168	78	Mo, Au	40	25	35	55	38	0.01	0.10	0.02	0.56	0.54
4	Australia	149	20,292	0.63	127	50	Au, U3O8	22	1	15	35	33	2.24	0.02	0.00	0.97	0.04
5	Russia	14	5,516	1.07	59	35	Ni, PGM, Au	21	3	34	12	3	0.25	0.10	0.06	0.01	0.01
	Rest of World	408	130,177	0.46	598	36	PGM, Au, Ni	34	31	35	25	13	0.63	0.09	0.03	0.37	0.26
						Contained Value (%)		Weighted Averages, Metal Basis									
	Lead-Zinc	**No.**	**Mt Ore**	**% Pb+Zn**	**Mt Pb+Zn**	**Pb+Zn**	**Other Metals >5%**	CRIT	SR	VSR	EI	AWaRe	BWS	BIER	BIER-h	WDI	WSI
	Total	852	50,882	1.64	837	30	Mo, Cu, Ag	35	32	28	35	18	0.91	0.07	0.03	0.50	0.30
	Undeveloped Deposits	620	31,467	1.33	419	21	Mo, Cu	33	29	29	30	19	0.70	0.08	0.03	0.44	0.29
	Recently Operating	232	19,415	2.15	418	48	Cu, Ag, Ni	37	34	26	39	18	1.11	0.07	0.02	0.55	0.31
	Deposit Type																
1	Sediment-hosted Pb-Zn	279	7,875	6.22	490	90	Ag	33	28	28	32	17	0.82	0.07	0.03	0.55	0.25
2	Volcanogenic Massive Sulfide	269	4,156	3.15	131	42	Cu, Au, Ag	29	23	27	27	16	1.31	0.05	0.02	0.30	0.23
3	Skarn	92	5,884	1.48	87	36	Cu, Fe, Ag	48	48	33	53	22	0.62	0.07	0.02	0.55	0.50
4	Epithermal	138	4,082	1.70	69	45	Ag, Cu, Au	49	53	30	50	28	1.13	0.11	0.02	0.64	0.51
5	Porphyry	16	14,646	0.18	27	11	Cu, Mo, Ag	51	65	30	44	32	1.59	0.08	0.02	0.41	0.42
6	Sediment-hosted Mixed	9	12,581	0.14	17	1	Mo, Ni	9	1	8	12	1	0.37	0.07	0.03	0.01	0.13
7	Mesothermal Vein	13	100	6.26	6	65	Ag	35	20	21	44	14	0.68	0.10	0.06	0.22	0.19
8	Iron Oxide Copper Gold	6	326	0.61	2	19	Mo, Cu, Ag, Ni	39	52	21	30	65	4.85	0.08	0.01	0.88	0.62
9	Miscellaneous	4	972	0.24	2	14	U3O8	15	0	17	19	0	-	0.01	0.00	0.01	0.01
10	Mine Wastes	14	146	1.55	2	52	Au, Cu, Ag	30	24	27	32	7	0.39	0.07	0.01	0.16	0.10
11	Orogenic Au	8	90	1.71	2	35	Cu, Au, Ag	22	8	14	32	49	1.57	0.04	0.00	0.65	0.43
12	Manto	4	25	4.89	1	68	Ag, Au	22	14	24	19	1	0.08	0.08	0.02	0.12	0.12
	Country																
1	Australia	132	2,427	6.38	155	78	Ag, Cu	17	1	15	25	17	1.10	0.03	0.01	0.88	0.04
2	Canada	230	7,574	1.32	100	58	Cu, V, Ag, Au, Mo	14	0	18	14	2	0.21	0.12	0.07	0.02	0.01
4	Peru	48	7,537	0.85	64	32	Cu, Ag	40	25	35	55	5	0.01	0.15	0.03	0.27	0.29
3	Russia	40	2,562	3.55	91	49	Cu, Fe, Au	22	3	35	13	6	0.35	0.13	0.04	0.02	0.04
5	Mexico	65	4,966	1.22	61	45	Ag, Cu, Au	56	76	24	50	41	2.74	0.05	0.01	0.90	0.75
	Rest of World	337	25,816	1.42	366	19	Mo, Cu	48	56	34	45	24	1.31	0.06	0.02	0.55	0.48
	Nickel					Contained Value (%)		Weighted Averages, Metal Basis									
		No.	**Mt Ore**	**% Ni**	**Mt Ni**	**Ni**	**Other Metals >5%**	CRIT	SR	VSR	EI	AWaRe	BWS	BIER	BIER-h	WDI	WSI
	Total	476	61,365	0.48	292.5	82	Cu, Co	29	20	31	25	9	1.07	0.06	0.02	0.33	0.11
	Laterite	224	15,838	1.12	178.1	91	Co	25	12	32	17	8	0.83	0.06	0.02	0.23	0.05
	Undeveloped Deposits	173	11,955	1.08	129.7	90	Co	26	12	33	18	9	0.99	0.06	0.02	0.26	0.05
	Recently Operating	51	3,883	1.25	48.5	93	Co	21	10	29	14	5	0.11	0.05	0.02	0.16	0.03
	Sulfide	252	45,526	0.25	114.3	71	Cu, PGM	35	31	28	35	10	1.20	0.07	0.02	0.45	0.21
	Undeveloped Deposits	166	27,500	0.20	54.3	75	Cu, PGM	34	31	24	36	12	1.19	0.08	0.03	0.45	0.18
	Recently Operating	86	18,027	0.33	60.0	69	Cu, PGM	35	32	31	33	9	1.21	0.06	0.02	0.44	0.24
	Deposit Type - Laterite																
1	Oxide	112	10,750	1.04	112.3	88	Co	26	15	32	19	9	1.28	0.04	0.01	0.26	0.06
2	Hydrous Mg silicate	74	3,435	1.49	51.2	97		22	6	34	8	4	0.39	0.08	0.03	0.06	0.02
3	Clay silicate	38	1,653	0.88	14.6	90	Co	23	2	25	28	10	0.27	0.08	0.03	0.66	0.02
	Deposit Type - Sulfide																
1	Magmatic Sulfide	224	37,720	0.28	106.9	72	Cu, PGM	37	34	29	36	11	1.26	0.07	0.02	0.48	0.23
2	Hydrothermal Ni	27	6,523	0.09	6.1	61	Ni, Co	8	1	9	9	2	0.40	0.08	0.03	0.03	0.03
3	Fe-Ni alloy	1	1,284	0.11	1.4	100		7	0	5	11	1	0.01	0.12	0.08	0.01	0.01
	Country - Laterite																
1	Indonesia	16	2,071	1.61	33.3	97		26	5	45	1	1	-	0.05	0.03	0.01	0.01
2	Australia	69	4,182	0.75	31.5	88	Co	26	1	24	38	23	4.19	0.02	0.00	0.99	0.06
3	Phillipines	25	1,566	1.15	18.0	94	Co	16	13	21	10	3	1.68	0.02	0.01	0.04	0.02
4	Cuba	10	1,340	1.21	16.2	86	Co	29	31	29	26	5	-	0.01	0.00	0.17	0.05
5	New Caledonia	23	808	1.86	15.0	94	Co	-	-	-	-	1	-	0.00	0.00	0.01	0.01
	Rest of World	81	5,872	1.09	64.1	90	Co	24	15	33	14	8	0.33	0.11	0.04	0.10	0.07
	Country - Sulfide																
1	South Africa	45	18,362	0.18	33.2	89	Cu	66	85	38	63	19	2.08	0.08	0.01	0.91	0.49
2	Canada	54	6,377	0.34	21.9	79	Cu, Co	12	0	11	17	1	0.45	0.09	0.05	0.01	0.02
3	Russia	14	4,493	0.46	20.5	61	Cu, PGM	17	2	28	7	2	0.26	0.05	0.03	0.01	0.01
4	Australia	55	2,034	0.58	11.9	92		26	0	25	38	15	5.95	0.02	0.00	0.97	0.02
5	China	3	435	1.38	6.0	82	Cu	71	87	57	65	11	2.05	0.02	0.00	1.00	1.00
	Rest of World	81	13,826	0.15	20.9	51	Cu, PGM, Co	22	10	23	24	11	0.68	0.11	0.02	0.14	0.06

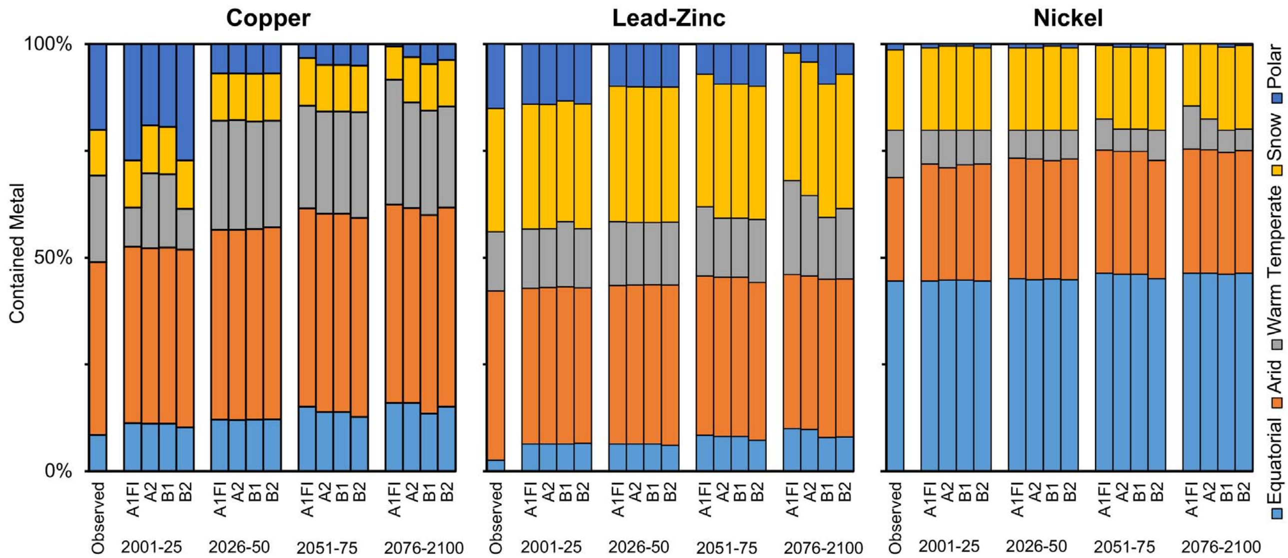

Fig. 4. Proportion of copper, lead-zinc and nickel resources contained in regions with each major Köppen-Geiger climate classification. Data shown for the observed period 1951–2000 (Kottek et al., 2006) and IPCC emissions scenarios A1FI, A2, B1 and B2 until 2100 (Rubel and Kottek, 2010). Resource data sources: copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see Section 2.1).

3.1. Regional water indices

The key results showing the exposure of copper, lead-zinc and nickel resources to the regional water indices are shown in Fig. 3 and Table 3. The results indicate that on average global copper resources are more exposed to these contextual water risks than global lead-zinc or nickel resources across all indices considered (note that a higher BIER or BIER-h may be preferable). Additionally, global lead-zinc resources appear more exposed than nickel resources for all indices barring the BWS, BIER and BIER-h. The aggregate results for the various indices are largely consistent, independent of the individual index. However, the results for specific countries or deposit types are highly variable.

The average water criticality in regions containing copper deposits is higher than those containing lead-zinc or nickel deposits. This is largely due to the exceptionally high SR regions in Chile and southern Peru that contain significant amounts of contained copper in porphyry deposits. The average VSR is similar for all three commodity groups, although some regions containing large lead-zinc and nickel resources, such as China, Kazakhstan or Iran, may be highly susceptible to water supply restrictions. Regions containing sediment-hosted Cu deposits also appear more acutely vulnerable to supply restrictions than regions containing porphyry copper deposits.

The general exposure to water risks can also be assessed by considering the proportion of contained resources that are located in regions experiencing water stress or scarcity above global averages. Some 55% (951 Mt) of current copper, 27% (230 Mt) of current lead-zinc and 4% (11 Mt) of current nickel resources are located in regions with a WSI exceeding 0.602, the global average WSI associated with fresh water consumption (Ridoutt and Pfister, 2013). In comparison, some 57% (1018 Mt) of current copper, 25% (208 Mt) of current lead-zinc and 15% (45 Mt) of current nickel resources are located in regions that have AWaRe values higher than the global average for non-agricultural water use (20; WULCA, 2016). The proportion of contained metal in regions with a BWS exceeding 1–indicating regions where water is being overexploited – is 39% (387 Mt) for copper, 36% (206 Mt) for lead-zinc and 19% (55 Mt) for Ni (note that these values for BWS are highly uncertain due to data limitations – see Table 1).

The BIER data indicate that some 5%, 7%, and 6% of evaporation in regions containing copper, lead-zinc and nickel resources respectively is re-precipitated within the same drainage basin. However only 2%, 3% and 2% respectively of the evaporation will re-precipitate and replenish blue water stores through surface runoff and groundwater recharge (indicated by BIER-h). As with the other indices, there is high variability for the BIER and BIER-h data between individual deposit types and countries. For instance, the weighted average BIER for copper in sediment-hosted Cu deposits and in Skarn deposits is 18% and 13% respectively, considerably higher than the average for all copper deposits of 5%.

The results for recently operating mine sites can also be compared with those for undeveloped deposits. Undeveloped copper resources are located in less water scarce or stressed regions than resources that have been recently mined, primarily as a result of the high proportion of overall copper resources that are contained within the large-scale copper operations in water scarce regions of Chile and Peru. Our data for lead-zinc resources also indicates that undeveloped deposits are generally located in less water stressed regions than recently operated mines, although with less consistency between individual indicators. The averages for CRIT, SR, EI, BWS, WDI, WSI were significantly higher for recently operated lead-zinc resources, whereas VSR, AWaRe, BIER, BIER-h were marginally lower for these resource. There was no significant difference between the overall results for undeveloped and operating nickel sulfide resources. However, undeveloped nickel laterite deposits are located in more water scarce or critical regions than recently operated nickel laterite mines, although it should be noted that the nickel laterite data are more uncertain than data for the other commodities and resource types as a result of the limited spatial indicator data available for regions containing these resources (e.g. New Caledonia; see Tables 1 and 2).

Further detailed results for individual countries or resource types are provided in Supplementary Tables S.1–S.8.

3.2. Köppen-Geiger climate classifications

The proportion of copper, lead-zinc and nickel contained within different climate zones are shown in Fig. 4, with more detailed information for precipitation and temperature sub-classifications given in Table 4.

A large proportion of copper deposits (40%, 721 Mt Cu) are located in regions with arid climates, including 43% (572 Mt Cu) of copper contained in porphyry deposits (largely within southern Peru and Chile) and 85% (106 Mt Cu) of copper in iron oxide copper gold deposits (largely in Australia, Peru and Chile). Of the copper contained within porphyry deposits, 25% (327 Mt Cu) is within regions classified as having a polar climate (mostly in the alpine tundra sections of the

Table 4
Base metal resources contained in Köppen-Geiger climate zones, based upon the observed period 1951–2000 (Kottek et al., 2006). Results for deposit types, other time periods and climate scenarios are available in the electronic Supplementary information. Resource data sources: copper (), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see Section 2.1).

Köppen-Geiger Climate Classification		1951-2000, Observed (Kottek et al., 2006)											
		Copper				Lead-Zinc				Nickel			
		No.	Mt Ore	% Cu	Mt Cu	No.	Mt Ore	%Pb+Zn	Mt Pb+Zn	No.	Mt Ore	% Ni	Mt Ni
	Equatorial	**74**	**31,735**	**0.48**	**151**	**51**	**437**	**4.98**	**22**	**129**	**10,133**	**1.29**	**130**
Af	Fully humid	24	17,297	0.55	95	11	86	5.23	4	47	4,186	1.41	59
Am	Monsoonal	12	9,584	0.38	36	3	49	2.24	1	29	1,639	1.16	19
As	Summer dry	2	9	2.45	0	3	4	2.55	0	4	63	1.96	1
Aw	Winter dry	36	4,845	0.41	20	34	298	5.39	16	49	4,245	1.20	51
	Arid	**255**	**158,801**	**0.45**	**721**	**237**	**13,387**	**2.48**	**331**	**149**	**18,433**	**0.38**	**71**
BSh	Steppe, hot arid	85	16,564	0.42	70	90	4,833	3.73	180	53	13,216	0.24	31
BSk	Steppe, cold arid	49	42,196	0.35	149	60	5,996	0.94	57	22	95	1.50	1
BWh	Desert, hot arid	68	21,940	0.63	137	74	1,887	3.19	60	73	4,690	0.68	32
BWk	Desert, cold arid	53	78,101	0.47	365	13	671	5.09	34	1	432	1.39	6
	Warm Temperate	**169**	**63,455**	**0.57**	**361**	**173**	**4,448**	**2.60**	**116**	**71**	**10,922**	**0.30**	**32**
Cfa	Fully humid, hot summer	22	5,245	0.34	18	20	308	6.61	20	7	577	0.86	5
Cfb	Fully humid, warm summer	37	4,710	1.04	49	70	657	4.28	28	10	382	0.82	3
Cfc	Fully humid, cool summer	1	10	1.17	0	5	39	5.26	2	-	-	-	-
Csa	Summer dry, hot summer	17	4,258	0.36	16	34	2,314	1.76	41	5	411	1.05	4
Csb	Summer dry, warm summer	13	27,377	0.55	151	14	328	1.66	5	7	619	0.77	5
Csc	Summer dry, cool summer	-	-	-	-	-	-	-	-	-	-	-	-
Cwa	Winter dry, hot summer	65	15,813	0.70	111	17	152	7.30	11	32	6,361	0.15	10
Cwb	Winter dry, warm summer	14	6,043	0.28	17	13	650	1.23	8	10	2,573	0.22	6
Cwc	Winter dry, cool summer	-	-	-	-	-	-	-	-	-	-	-	-
	Snow	**171**	**46,466**	**0.41**	**190**	**312**	**25,683**	**0.94**	**241**	**122**	**21,446**	**0.26**	**55**
Dfa	Fully humid, hot summer	-	-	-	-	4	54	1.38	1	1	74	0.86	1
Dfb	Fully humid, warm summer	53	7,865	0.55	43	99	7,839	0.95	74	44	5,638	0.30	17
Dfc	Fully humid, cool summer	101	28,141	0.37	104	165	15,260	0.72	110	71	15,073	0.23	35
Dfd	Fully humid, extremely continental	-	-	-	-	1	18	13.28	2	-	-	-	-
Dsa	Summer dry, hot summer	-	-	-	-	2	32	26.20	8	-	-	-	-
Dsb	Summer dry, warm summer	6	6,221	0.11	7	6	59	5.14	3	-	-	-	-
Dsc	Summer dry, cool summer	3	434	0.24	1	23	180	6.95	13	-	-	-	-
Dsd	Summer dry, extremely continental	-	-	-	-	-	-	-	-	-	-	-	-
Dwa	Desert, hot summer	-	-	-	-	-	-	-	-	-	-	-	-
Dwb	Desert, warm summer	-	-	-	-	2	315	5.00	16	-	-	-	-
Dwc	Desert, cool summer	8	3,804	0.91	35	10	1,924	0.75	14	6	661	0.41	3
Dwd	Desert, extremely continental	-	-	-	-	-	-	-	-	-	-	-	-
	Polar	**61**	**62,813**	**0.57**	**358**	**79**	**6,928**	**1.82**	**126**	**5**	**431**	**0.92**	**4**
EF	Frost	-	-	-	-	-	-	-	-	-	-	-	-
ET	Tundra	61	62,813	0.57	358	79	6,928	1.82	126	5	431	0.92	4

Andean porphyry copper belt through Chile, Peru and Argentina), 7% (86 Mt Cu) is in snow climates, 10% (132 Mt Cu) is in equatorial climates, and 15% (200 Mt Cu) is located in warm temperate climates (mostly within Chile and Argentina). Warm temperate climates also contain 78% (129 Mt Cu) of the copper in sediment-hosted Cu deposits, divided between the fully humid-warm summer climate (Cfb) of Poland (19%, 31 Mt Cu) and the winter dry-hot summer climate (Cwa) of the Central African copper belt (59%, 98 Mt Cu) running through Zambia and the Democratic Republic of Congo. For iron oxide copper gold deposits, 83% (104 Mt Cu) of the contained copper is located within desert regions of Australia, Chile, Peru and Mauritania (BWh or BWk). The majority of copper in magmatic sulfide or skarn deposits are in regions with snow or polar climates.

Lead-zinc deposits are primarily located in the arid regions of Australia, Peru and Mexico as well as in the polar and snow climate regions of Canada and Russia. Some 50% (245 Mt Pb + Zn) of the lead-zinc resources contained in sediment-hosted Pb-Zn deposits are located in arid climate regions – with roughly two-thirds of this (153 Mt Pb + Zn) being in hot arid steppe climate (BSh) regions (predominantly in Australia and India). A further 25% (123 Mt Pb + Zn) of the lead-zinc contained in sediment hosted Pb-Zn deposits are located in snow climate regions (mostly within Russia and Canada). Regions with snow climates also contain 55% (72 Mt Pb + Zn) of the lead-zinc in volcanic massive sulfide deposits.

Nickel laterite deposits are formed through tropical weathering processes and so 71% (127 Mt Ni) of nickel laterite resources are contained in countries with equatorial climates such as New Caledonia, Cuba, Indonesia, and the Philippines. Nickel sulfide resources display a very different geographic distribution, with only 3.5% (4 Mt Ni) being contained in equatorial climate areas. Regions with snow climates (such as parts of Canada, Russia, the USA and Finland) contain 44% (50 Mt Ni) of nickel sulfide resources and a further 38% (43 Mt Ni) is contained in regions with arid climates (primarily within South Africa and Australia).

The proportion of contained resource in each major climate classification for each time period to 2100 under the various IPCC emissions scenarios (Rubel and Kottek, 2010) are shown in Fig. 4. Through time the proportion of contained resource in regions with polar or snow climates is expected to decline due to reclassification to arid or warm temperate climates. Concurrently, many regions with arid or warm temperate climates are likely to be reclassified to equatorial climates (Rubel and Kottek, 2010). Overall, 27–32% (473–574 Mt Cu) of copper, 17–29% (139–241 Mt Pb + Zn) of lead-zinc and 6–13% (19–39 Mt Ni) of nickel is contained in regions that may have a major climate reclassification. A further 15–23% (262–412 Mt Cu) of copper, 23–32% (195–270 Mt Pb + Zn) of lead-zinc and 29–32% (84–94 Mt Ni)

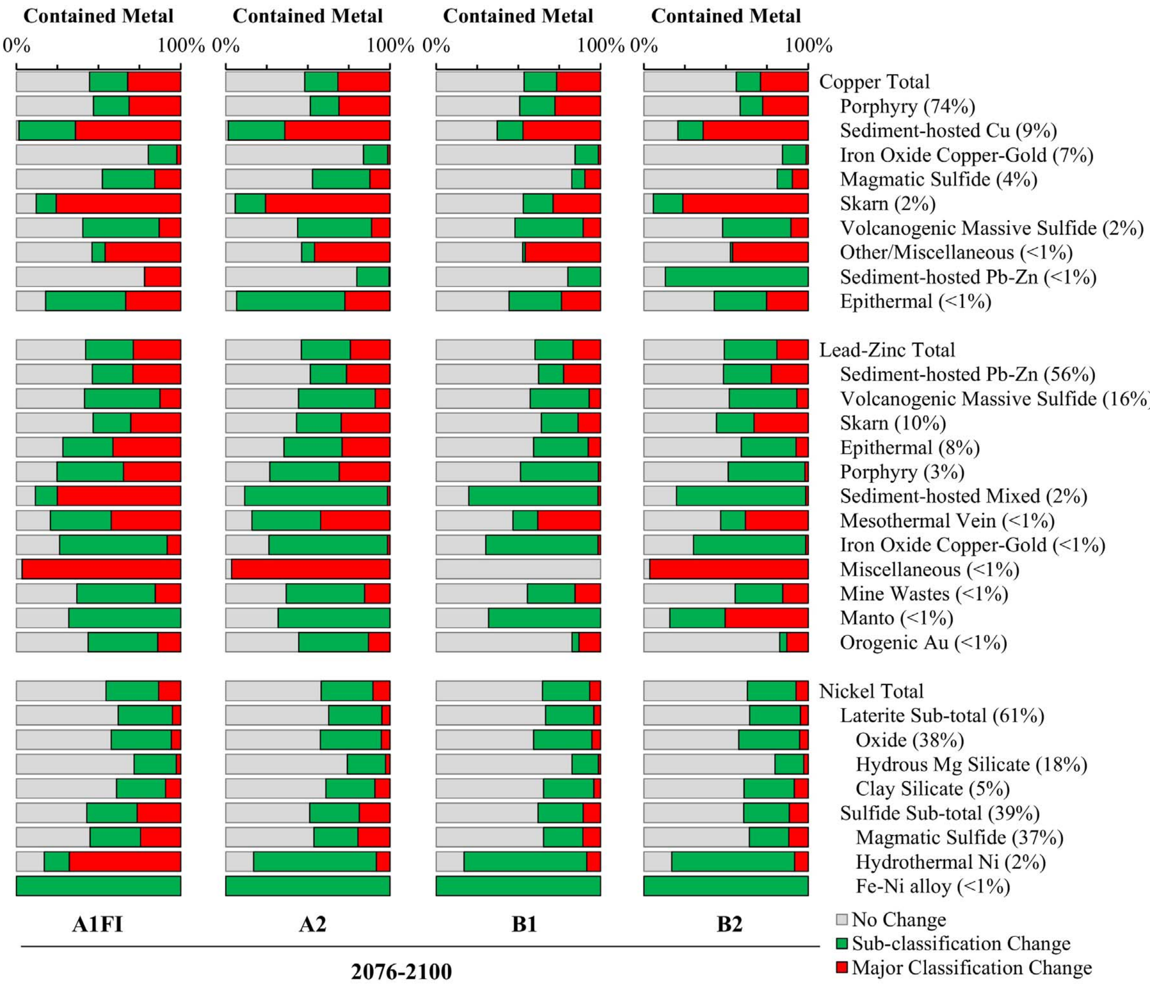

Fig. 5. Proportion of copper, lead-zinc and nickel resources located in regions with a changing Köppen-Geiger climate classification under the A1FI, A2, B1 and B2 IPCC scenarios. Data shown is for the period 2076–2100 (Rubel and Kottek, 2010) and is relative to the observed period 1951–2000 (Kottek et al., 2006). The distribution of copper, lead-zinc and nickel contained in individual deposit types is shown as a percentage in brackets. Figures showing other time periods are available in the electronic Supplementary information. Resource data sources: copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see Section 2.1).

of nickel is contained in regions that may have a precipitation or temperature sub-classification change. The exposure of individual deposit types to major climate classification changes, or precipitation and temperature sub-classification changes, is shown in Fig. 5. In addition, the flow of contained resources between these climate classifications are summarised in Fig. 6 for the IPCC's A1FI and the B1 emission scenarios. Equivalent figures for other climate scenarios or time periods are shown in Supplementary Figs. S.7–S.12.

Regions containing sediment-hosted Cu deposits (large in warm temperature climates) appear to be highly exposed to the impact of future climate change. Regions containing 42–59% (70–98 Mt Cu) of copper in sediment-hosted Cu deposits may be re-classified by the end of the century from having warm temperate climates with dry winters and hot summers (Cwa) to having equatorial climates with dry winters (Aw). A further 19% (31 Mt Cu) of copper in sediment-hosted Cu deposits may also be reclassified from being in fully humid-warm summer temperate climate (Cfb) to being in a fully humid-hot summer climate (Cfa). For porphyry deposits, 73–180 Mt of copper contained in arid climates may experience climate reclassifications, mostly from changes in temperature sub-classification from cold arid to hot arid.

Polar climates contain 15% (126 Mt Pb + Zn) of lead-zinc resources. Of this, 37–52% (47–65 Mt Pb + Zn) and 2–34% (3–43 Mt Pb + Zn) may be reclassified to being in snow and warm temperate climates respectively. Regions with 13–19% (106–161 Mt Pb + Zn) of contained lead-zinc resources are in snow climates that may have temperature or precipitation sub-classification changes by 2100. This manifests as either a shift of precipitation sub-classification from fully-humid to summer dry or desert, or as a change in temperature sub-classification associated with increasing summer temperatures. A further 8–14% (65–121 Mt Pb + Zn) of lead-zinc resources are located in arid regions that may have sub-classification changes.

Regions containing nickel resources are less likely to have major climate reclassifications by 2100, however they are still moderately exposed to climate sub-classification changes. Monsoonal (Am), fully humid (Af) and summer dry (As) equatorial climate sub-classifications are likely to have a net increase in contained nickel laterite resource by 2100, whereas equatorial-winter dry climates (Aw) will likely have a net decrease in contained nickel laterite resource. Nickel laterite resources contained in warm temperate climates may decline by 4–4.5% (7–8 Mt Ni) by 2100, due to reclassification to arid or equatorial climates. Also up to 14% (16 Mt Ni) of nickel contained in sulfide deposits in snow climates may be reclassified to warm temperate climates.

4. Discussion

4.1. Water availability and hydrological variability risks

A mining operation's water balance can be categorised as positive during periods where water accumulates on-site, or as negative during

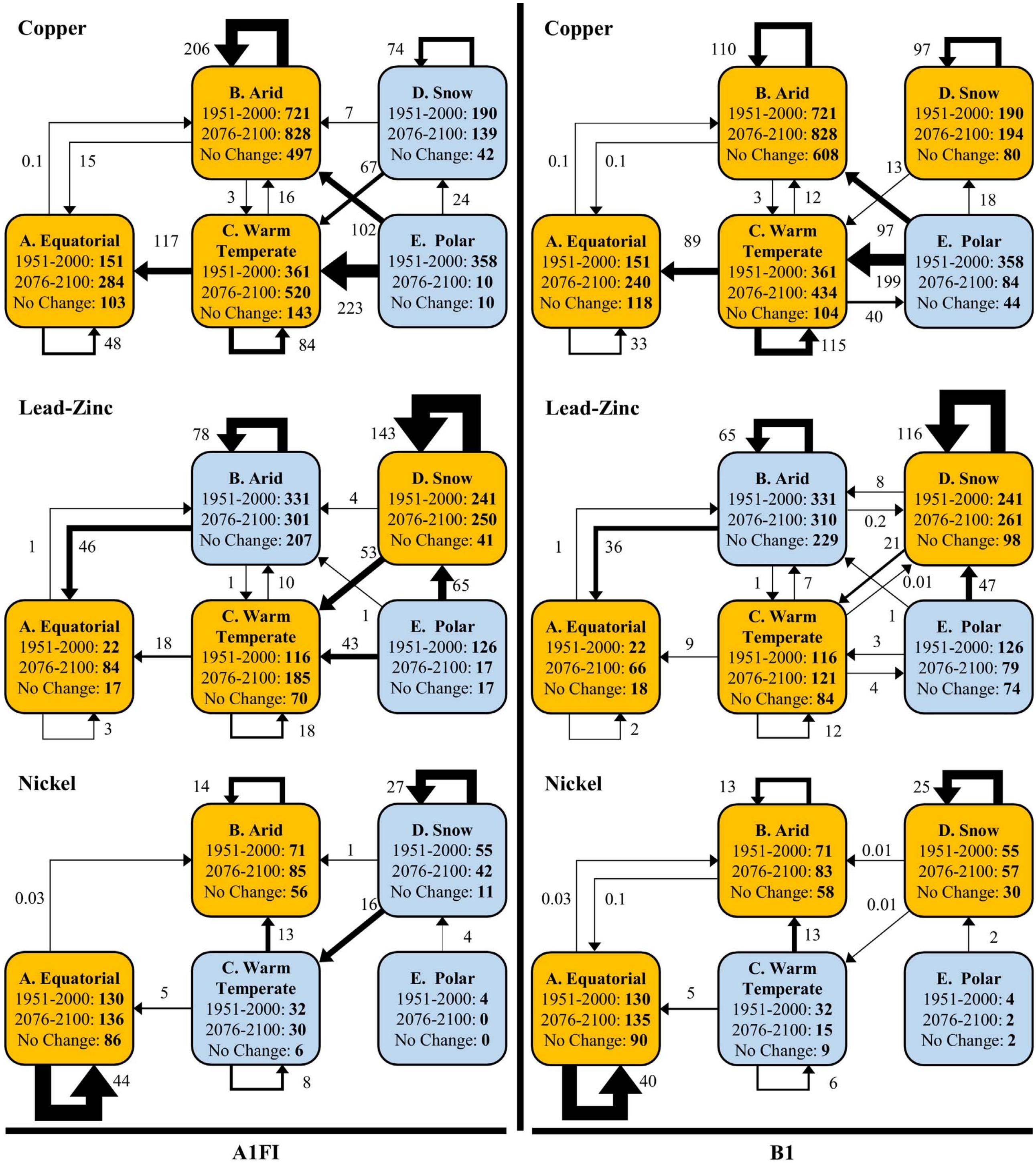

Fig. 6. Changes in the Köppen-Geiger climate classification of regions containing base metals from 1951 to 2000 (Kottek et al., 2006) to 2076–2100 for the IPCC emissions scenario A1FI and B1 (Rubel and Kottek, 2010). Figures for other scenarios are shown in the Supplementary information. Values indicate million tonnes of contained metal. Flow width is in proportion to the global resource for the individual metal(s). Flows returning to the same climate classification represent a change in precipitation or temperature sub-classification. Yellow and blue shading indicates a respective net increase or decrease in contained resource. Resource data sources: copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see Section 2.1). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

periods where a net loss of water occurs. Water positive operations may at some stage require active discharge of water, whereas a water negative operation may require continual imports of water to meet the requirements of ore processing facilities and to achieve other site objectives, such as dust suppression or the maintenance of tailings storage facility wet covers. The data for Köppen-Geiger climate classifications provides an initial basis to assess the average water balance when developing a base metal deposit (e.g. a mine located in an arid region will often have a negative water balance). However, further analysis is required to assess the implications of the various temperature and precipitation sub-classifications on average mine site water balances.

Sourcing sufficient water to meet the requirements of mining operations may be challenging or controversial in some circumstances.

This is particularly the case in regions where access to water resources are also fully allocated for other purposes, such as agriculture, forestry or environmental flows. The SR, BWS, WSI and AWaRe data provide an indication of regions where access to water resources and competition with other users may potentially constrain the development of base metal resources.

Regions containing fully or over-allocated water resources can lead to strong incentives for the mining industry to develop alternative sources of water. As an example, Chile accounts for one third of global mined copper production and also contains the largest known copper resources (Mudd et al., 2013; USGS, 2016). In many regions of Chile, significant water scarcity or overexploitation of water resources is occurring (see BWS and WSI data in Table 3). As a result the Chilean copper industry has achieved considerable improvements in water efficiency. From 2009 to 2014, water use decreased from 0.67 to 0.53 m^3/t ore for copper concentrate producers and from 0.12 to 0.08 m^3/t ore for electrowon copper producers (COCHILCO, 2015a). Despite these efficiency improvements, growth in the Chilean industry has required significant investment in seawater desalination and pipeline capacity (e.g. the US$ 3.4 billion desalination plant to supply the Escondida mine site; BHPB, 2013) to meet water requirements. Desalinated water use by the Chilean copper industry is anticipated to grow by 14% annually between 2015 and 2026 (COCHILCO, 2015b). Raw seawater is also occasionally used in Chilean mineral processing circuits, such as at the Las Luces copper-molybdenum concentrator (Moreno et al., 2011). Further use of seawater in mineral processing circuits is possible, but key issues remain with ore mineralogy, processing constraints, and the limited understanding of flotation chemistry and performance in saline waters (Castro, 2012; Moreno et al., 2011; Wang and Peng, 2014).

The water balance of a mine-site may shift between being water positive and water negative in response to changing seasons, weather events and site water management decisions. This variability can pose operational challenges for mining operations, particularly when regulatory or practical constraints limit the ability to withdraw or discharge water. A variety of strategies are available to mitigate these risks, such as the development of new infrastructure or changes to operating procedures (Kunz and Moran, 2016).

Drought conditions may exacerbate existing water availability and allocation issues for mining operations. The development of additional water storage infrastructure may provide some buffer against these issues. However, generally these conditions will require mines to source additional water from surrounding surface or ground water systems to offset increases in net evaporative water losses (i.e. evaporation minus rainfall). Increases in net evaporative losses from ponds and dams may effectively limit the maximum achievable recycling rate of water at mineral processing operations. Firstly, the physical decrease in water volumes requires external surface or groundwater to be sourced to offset these losses. Additionally, the increased evaporation rates may lead to accelerated accumulation of salts in water storage facilities, requiring additional water treatment or dilution with freshwater to achieve the water quality requirements of ore processing facilities. Drought can also impact mining operations post-closure, through limiting the success of site revegetation and rehabilitation efforts (Halwatura et al., 2014).

Perhaps more concerning for many mining operations are the risks associated with having too much water. Heavy rainfall (or snowfall) can result in sections of mines becoming inaccessible or unsafe to operate, which can lead to supply disruptions. Alternatively periods of excess water may result in unplanned or uncontrolled water discharges to surrounding environments, as happened at the Lady Annie copper mine in Australia in 2009 (Taylor and Little, 2013). The financial implications of flood events in mining regions can be significant. During the 2010/2011 Queensland floods (Australia), 20% of the coal mines in the Bowen Basin became inoperable and a further 60% faced operating restrictions. The associated financial costs were AU$ 5–9 billion (Sharma and Franks, 2013; QRC, 2011) and emergency water discharges were approved to prevent further adverse consequences and infrastructure risks (QFCI, 2012).

Köppen-Geiger climate classifications only provide insight into average seasonal hydrologic variability. However assessing the exposure of the mining industry to drought and flood risks requires more acute measures of hydrologic variability. This could be assessed using data for rainfall frequency and intensity, or the recurrence intervals of drought or flood events. Previously these risks to the mining industry have typically been assessed at the scale of individual mine sites. However some studies are now considering these risks at the scale of major multi-national mining companies (Bonnafous et al., 2016).

4.2. Climate change and infrastructure risks

Mine site infrastructure is typically designed and built based upon historic weather and climate patterns (Pearce et al., 2011). For instance a tailings storage facility may be built to withstand a 1-in–500 year rainfall event. However, as weather patterns and climate change into the future, the assumptions used to develop infrastructure at current and historic mining operations may no longer be appropriate. Thus, risks associated with existing infrastructure in the mining industry may increase or decrease through time depending upon local climate changes. When these climate related risks are relatively small for individual mining operations, a company or institutional investor that has a stake in multiple mining operations may still be exposed to significant technical or financial risks when considered at a portfolio level (see Bonnafous et al., 2016).

The stability of slopes in open pit mines and tailings storage facilities are a potential hazard throughout the world and there are many contributing risk factors, especially in the context of a changing climate. Among the various factors affecting slope stability, water is known to be a major trigger for failure (Azam and Li, 2010). At mine sites, the stress relieving moments are quite commonly observed, and these moments can range anywhere from a few millimetres to a couple of metres, depending on where a particular mine is situated. These are largely influenced by the in-situ stress conditions and the deformation characteristics of the material in which the excavations are made. These movements will increase in-ground strain on the slopes, which may in turn result in localised cracking of the slopes. The rain fed surface water entering these cracks may then result in block instabilities, placing the safety of equipment and personnel at risk.

In many natural mining environments, rocks (ore bearing and overlain) are subjected to cyclic drying-wetting conditions because of repeated water absorption and evaporation. This phenomenon is commonly referred to as 'creep' and prolonged creep leads to 'fatigue' (Özbek, 2014) which in turn causes rocks to undergo weathering, which can lead to the deterioration of its mechanical properties resulting in slope instability (Vergara and Trianafyllidis, 2015). Studies have established that rocks exposed to repeat drying-wetting cycles deteriorate more rapidly when compared to rocks in saturated conditions (Hua et al., 2015). Moreover, expansive and reactive soils (clay) that undergo periodic swelling and shrinkage during the alternate wet and dry environments, can result in severe damage to the slope stability (Erguler and Ulusay, 2009). Therefore alterations to local climates may alter the risk profile of slopes.

Climate change may lead to a thawing of permafrost over time in some regions, which is particularly concerning as thawing of permafrost can affect the stability of existing slopes and dams that were designed under the assumption of being continually frozen (Pearce et al., 2011). Melting permafrost also has implications for other water risks, such as contributing to flood waters or allowing unwanted migration of ground water. Some examples being the Diavik diamond mine that has resorted to actively freezing permafrost to prevent inundation of the site by surrounding lake waters (Haley et al., 2011), and the Red Dog zinc-lead mine in Alaska that has had higher than expected seepage due to heat generation from waste rock oxidation (Clark et al., 2011; Haley et al., 2011).

The potential impacts of climate change on transportation infrastructure, both on-site for mine operations and off-site for product export, have been highlighted as a particular concern for the Canadian mining industry (Ford et al., 2011). Hydrologic variability and climate change may impact transport infrastructure in a variety of ways. Mine wall collapses caused by high rainfall may damage or block access roads. Drought conditions can lower river levels and prevent barges accessing remote mine sites, as occurred at the Ok Tedi copper and gold mine in Papua New Guinea. Ice roads providing access to mines may experience decreases in their operating season, as has been observed at some Canadian mines (Haley et al., 2011). Breaking up of ice sheets may create hazards for shipping of mineral concentrates (Haley et al., 2011). Alternatively there may be some benefits to the industry's transport system from climate changes, such as the opening and development of the north-west passage that may shorten export shipping routes.

The recession of glaciers may make new mineral deposits accessible or easier to develop. Examples of this exist in Canada such as the Red Mountain gold deposit, the Brucejack gold-silver deposit and the Mitchell gold-copper deposit that forms part of the KSM project (the region containing this deposit may transition from being classified as Snow (Dfc) to being Warm Temperate (Cfb) by 2100 under the A1FI and A2 climate scenarios). On balance though the recession of glaciers has potentially significant impacts for mine-sites, due to the significant changes in runoff volumes and frequency that may occur through the life of the mine.

Examples also exist of mining directly reducing the extent of glaciers, such as the Kumtor gold mine in Kyrgyzstan where 39 million m^3 of ice was removed by 2011 (this equates to approximately 5 m^3 per ounce of gold produced; Kronenberg, 2013). In addition, although the Pascua-Lama mine on the border of Chile and Argentina initially intended to remove 0.8 million m^3 of ice during development, the removal was prevented after widespread outcry through the development of government policies to protect endangered glaciers. This led to the sterilisation of some 7% of the deposits contained gold (1.3 million ounces) (Kronenberg, 2013).

Extreme climate events such as cyclones or typhoons carry significant risks for mining operations. The impacts of typhoons and heavy rainfall in the Philippines have led to mine worker deaths, flooding of mines, production halts, transport and processing infrastructure damage, landslides and tailings spillages (Holden, 2015). Overall, 44% of mining projects in the Philippines are located in regions with at least a medium typhoon risk. This is particularly concerning as the expected impacts of climate change − such as sea level rise and increasing sea-surface temperatures − may increase the severity of storm surges, rainfall and winds associated with typhoons (Holden, 2015).

4.3. Water quality risks

The generation of water quality risks from mining operations is heavily dependent upon the local hydrology and climate, the geochemistry of the deposit being mined, techniques used for water and mine waste management, and the chemical and ecological nature of surrounding water bodies. A major source of water quality issues associated with the mining industry is acid mine drainage (AMD). AMD is generated when sulfide minerals − such as pyrite (FeS_2), pyrrhotite (FeS) or chalcopyrite ($CuFeS_2$) – undergo surface weathering and chemical or bacterial oxidation processes (Dold, 2014; Lottermoser, 2010; Nordstrom et al., 2015; Amos et al., 2015). Generation of AMD increases sulfate concentrations, lowers pH and can solubilise salts and metals from surrounding rock, thereby causing adverse consequences when these solutions migrate into surrounding environments. Today, the majority of the world's copper, lead, zinc and a large proportion of nickel are extracted from sulfide ores. Consequently the problem of AMD is relatively widespread throughout the mining industry (Akcil and Koldas, 2006; Benner et al., 2002; Da Rosa et al., 1997). A range of active and passive techniques are available for the prevention, treatment or remediation of AMD (Johnson and Hallberg, 2005; Lottermoser, 2010; Ziemkiewicz et al., 2003). The success or suitability of these techniques depends upon a variety of factors, including the local climate.

Climate change and variability can influence AMD risks through a variety of processes (Anawar, 2013; Lin, 2012; MEND, 2011; Nordstrom, 2009; Phillips, 2016), including:

1. Higher temperatures affecting rates of biologically- or chemically-driven sulfide oxidation, with a probable tendency to increase overall reaction rates,
2. Changes to precipitation altering pollutant migration pathways and rates,
3. Changes to pit lake levels arising from changes to evaporation and rainfall patterns (i.e. changed water balance) altering the direction of groundwater flow towards or away from open pits,
4. Increases in evaporation reducing or eliminating water bodies or moisture-retaining covers on mine waste facilities,
5. Capping layers on waste dumps cracking or degrading (see Section 4.2),
6. Changes to flow rates in receiving bodies altering rates of contaminant dilution (for better or worse),
7. Changes to the length of dry spells altering the magnitude of first flush effects on water quality (e.g. potentially increased solute concentrations due to extended evaporation periods).

Phillips (2016) identified that further work is required to assess the potential impacts of climate change on mining and surrounding regions– and also posed the question of whether there exists a climate threshold that could lead to large-scale potential for AMD generation. As part of our study we have provided some datasets that may contribute to understanding climate change-related AMD risks. The exposure to changes in Köppen-Geiger climate classifications provides some basic understanding of potential changes to temperature and precipitation that will influence a range of risk factors at mining operations. When combined with knowledge of mineral deposit types, the common mineralogy of these deposit types could be used to further understand mine-site water quality risks. As an example, Rayne et al. (2009) provided a conceptual framework to consider potential changes in run-off water quality by considering the rate of geochemical weathering of mine wastes and changes in climate and hydrology. Some other water quality issues such as erosion and transport of sediment can also be significant from surface mining, with potential to impact downstream ecosystems and to exacerbate changes to fluvial sedimentation processes caused by climate change (Phillips, 2016).

At present, it is very difficult to quantify the effects of changes in climate regimes on these aspects of mining and water resource risks. Changes in the climate of regions containing base metal resources (Figs. 5 and 6) can reasonably be assumed to alter water quality risks at mining operations through the processes described in the preceding paragraphs. However it is not possible to generalise whether climate change will lead to an overall increase or reduction in water quality risks across the industry due to the differing geochemistry, waste containment and management practices of individual mine sites. This means that further assessment of these risks at individual mine-site or regional scales is required before any industry-wide conclusions could be reached.

4.4. Climate change adaptation in the mining and minerals industry

The historical climate change focus of the mining industry has been on greenhouse gas emission mitigation efforts and reducing the potential exposure to carbon pricing schemes (ICMM, 2013; Ford et al., 2010, 2011). Comparatively, there has been relatively little attention given to assessing the potential climate change adaptation requirements of the mining industry. The International Council on Mining and Metals provided some guidance to the industry on how to assess the potential impacts, risks and adaptation requirements posed by climate change (ICMM, 2013). Mason et al. (2013) also developed a guide that showed through a range of case studies how the industry can adapt to climate risks and extreme weather events. Despite this the perception of climate change as a risk varies between industry stakeholders and across regions (Loechel et al., 2013; Ford et al., 2010, 2011).

A range of studies by the CSIRO have reported climate change adaptation in the Australian mining industry. This included activities such as assessing the potential climate changes in 11 different mineral producing regions of Australia (Hodgkinson et al., 2010), consultations with industry experts through workshop discussions (Hodgkinson et al., 2010), and surveying the perception of climate risks within the mining industry and surrounding communities (Loechel et al., 2013). Overall, adaptation efforts in the Australian mining industry have generally been reactive rather than anticipatory (Hodgkinson et al., 2014; Sharma and Franks, 2013), which is likely due in part to the need for improved information and understanding of the potential impacts to the industry from climate change (Loechel et al., 2013). Similar findings exist for the Canadian mining industry (Pearce et al., 2011; Ford et al., 2010, 2011).

Any climate change adaptation requirements in the mining industry are likely to be highly site specific. As an example, one adaptation measure may be the post-closure installation of a geo-synthetic cover to mitigate increased acid drainage risks associated with a changed local climate. This hypothetical scenario could result in a decrease of 13.9-14.9% of a copper project's net-present value (MEND, 2011). Further research and evaluation is required to understand the potential financial implications of climate change adaptation measures throughout the industry.

4.5. Implications for water footprint and life cycle assessment studies

Water footprint and life cycle assessment (LCA) methods attempt to quantify the relative impacts associated with production processes. Water footprint and LCA studies of the mining industry are increasingly utilising spatial water indices to assess the relative impacts associated with water use that occurs in different regions (Northey et al., 2016). The AWaRe, WSI, WDI, BIER, BIER-h and BWS indices have all been proposed for or used as part of these impact assessment procedures (Berger et al., 2014; Boulay et al., 2016a, 2016b; Hoekstra et al., 2012; Pfister et al., 2009; WULCA, 2016). There has been some debate over the best index to use for these purposes (Hoekstra, 2016; Pfister et al., 2017). As each index is measuring a different aspect of water use or hydrology, considering multiple indices simultaneously may lead to a greater understanding of water use impacts and trade-offs than when considering a single index in isolation.

Ideally, studies assessing water use impacts should use the highest level of spatial differentiation/resolution possible. However, indices developed based upon global hydrological modelling may not always be accurate when considering an individual region of concern for a particular study. For instance, modelling of the WSI for the Mississippi Basin at a sub-basin scale revealed significant differences when compared to the WSI estimate based upon a global hydrological model (Scherer et al., 2015). Therefore, studies considering water use in specific regions may benefit from recalculating indices based upon hydrologic models tailored specifically to the region under consideration. Boulay et al. (2015) identified regions where results determined using water indices are more sensitive to the spatial resolution, temporal resolution, water source differentiation (e.g. surface or ground water), water quality differentiation, data sources and the conceptualisation of the index.

Berger et al. (2014) proposed that the BIER and BIER-h be used to modify evaporation data in life cycle inventories to more accurately account for the contribution to regional water consumption. The BIER only accounts for evaporation that re-precipitates in the same drainage basin. However, evaporation may re-precipitate and replenish water stores in other water basins that also contain base metal resources. The global average continental evaporation recycling ratio has been shown to be approximately 57% (van der Ent et al., 2010), considerably higher than the 1% global average BIER (Berger et al., 2014). Due to this the perceived impacts associated with evaporation caused by base metal resource development may be very different depending upon if the geographic boundaries of assessment are drawn at local, regional, national, or continental scales.

Northey et al. (2014a) showed how the exposure of regions to water risk (measured using WSI) varies throughout metal supply chains, as different stages of the production process (e.g. mining and mineral processing, smelting, refining, etc.) may be located in different regions. Although based upon national production and WSI data (so not considering where mines are located in those countries), it was shown that on average copper production is occurring in countries with a higher WSI than lead, zinc or nickel production – which is broadly consistent with the results for the location of deposits presented by this study. Understanding how mineral supply chains are spatially distributed will enable more comprehensive assessment of water use impacts as part of LCA.

As mineral exploration continues and individual deposits are mined and depleted, the spatial distribution of identified and remaining mineral resources will change through time. Concurrently, studies have assessed potential changes in the future distribution of water use, stress and scarcity as a result of climate change and socio-economic developments (e.g. Alcamo et al., 2007; Ercin and Hoekstra, 2014; Hejazi et al., 2014; Kiguchi et al., 2015). This means that any assessments of future or long-term resource extraction should ideally account for these changes during inventory development and impact characterisation.

4.6. Implications for long-term resource development

The copper resource dataset has previously been used to assess scenarios for long-term copper supply and demand (Northey et al., 2014b). Incorporating regional data for contextual water risks into this style of assessment and combining this with estimates of water use requirements would provide valuable insights into the potential water constraints and impacts associated with developing mineral resources.

A number of other metals commonly occur within copper, lead-zinc and nickel resources and these are often produced as by-products. Thus to some extent, the inferences we have drawn for these four base metals extend also to their by-products. With approximately one quarter of the periodic table supplied as by-products or co-products of the mining of copper, lead-zinc, and nickel (Nassar et al., 2015), this suggests that the outcomes of our study are potentially quite far reaching. However, a limiting factor is the extent to which each by-product is reliant on each base metal for supply, and the differential spatial distribution of the by-products to the base metals. A full assessment of water-related risks for the by-product metals would therefore require a more detailed understanding of the quantities of by-product metals contained within the deposits identified in this study. This can be problematic to determine as they are often not reported within the mining industry (Mudd et al., 2016). However, steps are being made towards quantifying these metals (Werner et al., 2017), meaning that it may be possible to infer water related risks in detail and the implications for resource development for many more metals in the future.

5. Conclusions

The societal costs and benefits associated with developing mineral resources are not solely related to the nature of individual deposits and prevailing market conditions, but rather they are also a function of the local contexts of the location of these deposits. This study has provided a quantitative assessment of the water and climate contexts associated with global copper, lead-zinc and nickel resources. These resources are located across a diverse array of hydrological and climate contexts. The various indices show that copper resources are, on average, located in regions with more water stress, scarcity and risk than regions containing lead-zinc or nickel resources. In addition, regions containing copper and lead-zinc resources are potentially more exposed to changes in climate classification over the coming century than those containing nickel resources.

The impacts of climate change may be adverse for mines in some regions (e.g. increased evaporation and external water requirements), whereas in other regions these changes could be beneficial for managing various water risks at mine sites (e.g. potential decreases in AMD risks). Further work is required to understand the full extent of these risks and the likely impacts for mines in individual regions. However, we emphasise that reactive approaches to mine site water management, such as assuming a continuation of historic climate conditions and responding to weather conditions as they occur, should be avoided. Rather, mining operations should further embrace anticipatory risk management strategies and plan to be resilient to − or potentially even benefit from – the expected changes in regional climates. These plans should be developed well before mining commences.

Acknowledgements

Financial support for this study was primarily provided by CSIRO Mineral Resources. Several co-authors also receive funding as part of the CSIRO ‘Wealth from Waste’ cluster, a collaborative program between the Australian Commonwealth Scientific Industrial Research Organisation (CSIRO); the University of Queensland, the University of Technology, Sydney; Monash University, Swinburne University of Technology and Yale University.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.gloenvcha.2017.04.004.

References

Özbek, A., 2014. Investigation of the effects of wetting-drying and freezing-thawing cycles on some physical and mechanical properties of selected ignimbrites. Bull. Eng. Geol. Environ. 73 (2), 595–609. http://dx.doi.org/10.1007/s10064-013-0519-y.

Akcil, A., Koldas, S., 2006. Acid Mine Drainage (AMD): causes, treatment and case studies. J. Clean. Prod. 14, 1139–1145. http://dx.doi.org/10.1016/j.jclepro.2004.09.006.

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T., Siebert, S., 2003. Global estimates of water withdrawals and availability under current and future business-as-usual conditions. Hydrol. Sci. J. 48 (3), 339–348. http://dx.doi.org/10.1623/hysj.48.3.339.45278.

Alcamo, J., Flörke, M., Märker, M., 2007. Future long-term changes in global water resources driven by socio-economic and climatic changes. Hydrol. Sci. J. 52 (2), 247–275. http://dx.doi.org/10.1623/hysj.52.2.247.

Amos, R.T., Blowes, D.W., Bailey, B.L., Sego, D.C., Smith, L., Ritchie, A.I.M., 2015. Waste-rock hydrogeology and geochemistry. Appl. Geochem. 57, 140–156. http://dx.doi.org/10.1016/j.apgeochem.2014.06.020.

Anawar, H.M., 2013. Impact of climate change on acid mine drainage generation and contaminant transport in water ecosystems of semi-arid and arid mining areas. Phys. Chem. Earth. 58–60 (Parts A/B/C), 13–21. http://dx.doi.org/10.1016/j.pce.2013.04.002.

Azam, S., Li, Q., 2010. Tailings Dam Failures: a review of the last one hundred years. Geotechnical News 28 (4), 50–54.

BHPB, 2013. BHP billiton approves escondida water supply project. BHP Billiton (BHPB), News Release to the Australian Stock Exchange (ASX), 25 July.

Benner, S.G., Blowes, D.W., Ptacek, C.J., Mayer, K.U., 2002. Rates of sulfate reduction and metal sulfide precipitation in a permeable reactive barrier. Appl. Geochem. 17 (3), 301–320. http://dx.doi.org/10.1016/S0883-2927(01)00084-1.

Berger, M., van der Ent, R., Eisner, S., Bach, V., Finkbeiner, M., 2014. Water accounting and vulnerability evaluation (WAVE): considering atmospheric evaporation recycling and the risk of freshwater depletion in water footprinting. Environ. Sci. Technol. 48, 4521–4528. http://dx.doi.org/10.1021/es404994t.

Bonnafous, L., Lall, U., Siegel, J., 2016. A water risk index for portfolio exposure to climatic extremes: conceptualization and an application to the mining industry. Hydrol. Earth Syst. Sci. Discuss.. http://dx.doi.org/10.5194/hess-2016-515. Under Review.

Boulay, A.-M., Motoshita, M., Pfister, S., Bulle, C., Muñoz, I., Franceschini, H., Margni, M., 2015. Analysis of water use impact assessment methods (part A): evaluation of modelling choices based on a quantitative comparison of scarcity and human health indicators. Int. J. Life Cycle Assess. 20, 139–160. http://dx.doi.org/10.1007/s11367-014-0814-2.

Boulay, A.-M., Bare, J., Benini, L., Berger, M., Manzardo, A., Margni, M., Motoshita, M., Núñez, M., Lathuillière, M., Pastor, A.V., Ridoutt, B., Worbe, S., Pfister, S., 2016a. Consensus-based water scarcity footprint method from WULCA: the AWARE model. Int. J. Life Cycle Assess Under review.

Boulay, A.-M., Pfister, S., Motoshita, M., Schenker, U., Benini, L., Gheewala, S.H., de Figueiredo, M.C.B., Harding, K., 2016b. Chapter 5. water use related impacts: water scarcity and human health effects, part 1: water scarcity. In: Frischknecht, R., Jolliet, O. (Eds.), UNEP-SETAC Life Cycle Initiative, Global Guidance for Life Cycle Impact Assessment Indicators 1. pp. 100–115.

COCHILCO, 2015a. Hydrological Consumption in Copper Mining for 2014. Comisión Chilena del Cobre (COCHILCO), Government of Chile 02/2015.

COCHILCO, 2015b. Water Consumption Projection in Copper Mining Through 2026. Comisión Chilena del Cobre (COCHILCO), Government of Chile DEPP 16/2015.

Castro, S., et al., 2012. Challenges in flotation of Cu-Mo sulfide ores in sea water. Drelich (Ed.), Water in Mineral Processing – Proceedings of the First International Symposium, Society for Mining, Metallurgy, and Exploration 29–40.

Clark, J.L., Christensen, D., Lum, B., 2011. Acid Rock Drainage Source Control Opportunities at the Red Dog Zinc and Lead Mine. USA. Mine Closure 2011, Alberta, Canada, pp. 341–348.

Da Rosa, C.D., Lyon, J.S., Hocker, P.M., 1997. Golden Dreams, Poisoned Streams: How Reckless Mining Pollutes America's Waters, and How We can Stop It. Mineral Policy Center, Washington, D.C., United States 269p.

Daigo, I., Osako, S., Adachi, Y., Matsuno, Y., 2014. Time-series analysis of global zinc demand associated with steel. Resour. Conserv. Recycl. 82, 35–40. http://dx.doi.org/10.1016/j.resconrec.2013.10.013.

Dold, B., 2014. Evolution of acid mine drainage formation in sulphidic mine tailings. Minerals 4, 621–641. http://dx.doi.org/10.3390/min4030621.

Durán, A.P., Rauch, J., Gaston, K.J., 2013. Global spatial coincidence between protected areas and metal mining activities. Biol. Conserv. 160, 272–278. http://dx.doi.org/10.1016/j.biocon.2013.02.003.

Elshkaki, A., Graedel, T.E., Ciacci, L., Reck, B., 2016. Copper demand, supply, and associated energy use to 2050. Global Environ. Change 39, 305–315. http://dx.doi.org/10.1016/j.gloenvcha.2016.06.006.

Ercin, A.E., Hoekstra, A.Y., 2014. Water footprint scenarios for 2050: a global analysis. Environ. Int. 64, 71–82. http://dx.doi.org/10.1016/j.envint.2013.11.019.

Erguler, Z.A., Ulusay, R., 2009. Water-induced variations in mechanical properties of clay-bearing rocks. Int. J. Rock Mech. Min. Sci. 46 (2), 355–370. http://dx.doi.org/10.1016/j.ijrmms.2008.07.002.

Ford, J.D., Pearce, T., Prno, J., Duerden, F., Berrang Ford, L., Beaumier, M., Smith, T., 2010. Perceptions of climate change risks in primary resource use industries: a survey of the Canadian mining sector. Reg. Environ. Change 10, 65–81. http://dx.doi.org/10.1007/s10113-009-0094-8.

Ford, J.D., Pearce, T., Prno, J., Duerden, F., Berrang Ford, L., Smith, T.R., Beaumier, M., 2011. Canary in a coal mine: perceptions of climate change risks and response options among Canadian mine operations. Clim. Change 109, 399–415. http://dx.doi.org/10.1007/s10584-011-0029-5.

Geiger, R., 1954. Klassifikation Der Klimate Nach W. Köppen.–Landolt-Börnstein: Zahlenwerte Und Funktionen Aus Physik, Chemie, Astronomie, Geophysik Und Technik, (alte Serie), Band III (Astronomie Und Geophysik). Springer), Berlin, pp. 603–607.

Gunson, A.J., Klein, B., Veiga, M., Dunbar, S., 2012. Reducing mine water requirements. J. Clean. Prod. 21, 71–82. http://dx.doi.org/10.1016/j.jclepro.2011.08.020.

Gunson, A.J., 2013. Quantifying, reducing and improving mine water use. PhD Thesis. The University of British Columbia, Canada, pp. 2013 May 285pp.

Haley, S., Klick, M., Szymoniak, N., Crow, A., 2011. Observing trends and assessing data for Arctic mining. Polar Geogr. 34 (1–2), 37–61. http://dx.doi.org/10.1080/1088937X.2011.584449.

Halwatura, D., Lechner, A.M., Arnold, S., 2014. Design droughts as planning tool for ecosystems establishment in post-mining landscapes. Hydrol. Earth Syst. Sci. Discuss. 11 (5), 4809–4849. http://dx.doi.org/10.5194/hessd-11-4809-2014.

Hejazi, M., Edmonds, J., Clarke, L., Kyle, P., Davies, E., Chaturvedi, V., Wise, M., Patel, P., Eom, J., Calvin, K., Moss, R., Kim, S., 2014. Long-term global water projections using six socioeconomic scenarios in an integrated assessment modeling framework. Technol. Forecasting Social Change 81, 205–226. http://dx.doi.org/10.1016/j.techfore.2013.05.006.

Hodgkinson, J.H., Loechel, B., Moffat, K., Howden, M., Littleboy, A.K., Crimp, S., 2010. Climate risks to the Australian mining industry – a preliminary review of vulnerabilities. In: Sustainable Mining Conference, 17–19 August 2010. Kalgoorlie, Western Australia. pp. 341–350.

Hodgkinson, J.H., Hobday, A.J., Pinkard, E.A., 2014. Climate adaptation in Australia’s resource-extraction industries: ready or not? Reg. Environ. Change 14 (4),

1663–1678. http://dx.doi.org/10.1007/s10113-014-0618-8.
Hoekstra, A.Y., Mekonnen, M.M., Chapagain, A.K., Mathews, R.E., Richter, B.D., 2012. Global monthly water scarcity: blue water footprints versus blue water availability. PLoS One 7 (2), e32688. http://dx.doi.org/10.1371/journal.pone.0032688.
Hoekstra, A.Y., 2016. A critique on the water-scarcity weighted water footprint in LCA. Ecol. Indic. 66, 564–573. http://dx.doi.org/10.1016/j.ecolind.2016.02.026.
Holden, W.N., 2015. Mining amid typhoons: large-scale mining and typhoon vulnerability in the Philippines. Extr. Ind. Soc. 2, 445–461. http://dx.doi.org/10.1016/j.exis.2015.04.009.
Hua, W., Dong, S., Li, Y., Xu, J., Wang, Q., 2015. The influence of cyclic wetting and drying on the fracture toughness of sandstone. Int. J. Rock Mech. Min. Sci. 78, 331–335. http://dx.doi.org/10.1016/j.ijrmms.2015.06.010.
ICMM, 2013. Adapting to a Changing Climate: Implications for the Mining and Metals Industry. International Council on Mining & Metals, London, UK March.
IPCC, 2000. Special report on emissions scenarios. In: Nakicenovik, Swart (Ed.), Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK July 612 pages.
Johnson, D.B., Hallberg, K.B., 2005. Acid mine drainage remediation options: a review. Sci. Total Environ. 338, 3–14. http://dx.doi.org/10.1016/j.scitotenv.2004.09.002.
Jowitt, S.M., Mudd, G.M., Weng, Z., 2013. Hidden mineral deposits in Cu-dominated porphyry-skarn systems: how resource reporting can occlude important mineralization types within mining camps. Econ. Geol. 108 (5), 1185–1193. http://dx.doi.org/10.2113/econgeo.108.5.1185.
Köppen, W., 1900. Versuch einer Klassifikation der Klimate, vorzugsweise nach ihren Beziehungen zur Pflanzenwelt. Geographische Zeitschrift 6, 593–611.
Kiguchi, M., Shen, Y., Kanae, S., Oki, T., 2015. Re-evaluation of future water stress due to socio-economic and climate factors under a warming climate. Hydrol. Sci. J. 60 (1), 14–29. http://dx.doi.org/10.1080/02626667.2014.888067.
Kleijn, R., van der Voet, E., Kramer, G.J., van Oers, L., van der Giesen, C., 2011. Metal requirements of low-carbon power generation. Energy 36, 5640–5648. http://dx.doi.org/10.1016/j.energy.2011.07.003.
Kottek, M., Grieser, J., Beck, C., Rudolf, B., Rubel, F., 2006. World Map of the Köppen-Geiger climate classification updated. Meteorol. Z. 15 (3), 259–263. http://dx.doi.org/10.1127/0941-2948/2006/0130.
Kronenberg, J., 2013. Linking ecological economics and political ecology to study mining, glaciers and global warming. Environ. Policy Governance 23, 75–90. http://dx.doi.org/10.1002/eet.1605.
Kunz, N.C., Moran, C.J., 2016. The utility of a systems approach to managing strategic water risks at a mine site level. Water Resour. Ind. 13, 1–6. http://dx.doi.org/10.1016/j.wri.2016.02.001.
Lin, C., 2012. Climate change adaptation in acid sulfate landscapes. Am. J. Environ. Sci. 8 (4), 433–442. http://dx.doi.org/10.3844/ajessp.2012.433.442.
Loechel, B., Hodgkinson, J., Moffat, K., 2013. Climate change adaptation in Australian mining communities: comparing mining company and local government views and activities. Clim. Change 119 (2), 465–477. http://dx.doi.org/10.1007/s10584-013-0721-8.
Lottermoser, B., 2010. Mine Wastes: Characterization, Treatment and Environmental Impacts. Springer Science & Business Media.
MEND, 2011. Climate Change and Acid Rock Drainage –Risks for the Canadian Mining Sector. Mine Environment Neutral Drainage (MEND), Mining Association of Canada MEND Report 1.61.7.
Mason, L., Unger, C., Lederwasch, A., Razian, H., Wynne, L., Giurco, D., 2013. Adapting to Climate Risks and Extreme Weather: A Guide for Mining and Minerals Industry Professionals. National Climate Change Adaptation Research Facility, Gold Coast, Australia 76p.
McMahon, T.A., Peel, M.C., Karoly, D.J., 2015. Assessment of precipitation and temperature data from CMIP3 global climate models for hydrologic simulation. Hydrol. Earth Syst. Sci. 19, 361–377. http://dx.doi.org/10.5194/hess-19-361-2015.
Mitchell, T.D., Carter, T.R., Jones, P.D., Hulme, M., New, M., 2004. A comprehensive set of high-resolution grids of monthly climate for Europe and the globe: the observed records (1901–2000) and 16 scenarios (2001–2100). Working Paper 55. Tyndall Centre of Climate Change Research.
Moreno, P.A., Aral, H., Cuevas, J., Monardes, A., Adaro, M., Norgate, T., Bruckard, W., 2011. The use of seawater as process water at Las Luces copper-molybdenum beneficiation plant in Taltal (Chile). Miner. Eng. 24, 852–858. http://dx.doi.org/10.1016/j.mineng.2011.03.009.
Mudd, G.M., Jowitt, S.M., 2014. A detailed assessment of global nickel resource trends and endowments. Econ. Geol. 109, 1813–1841. http://dx.doi.org/10.2113/econgeo.109.7.1813.
Mudd, G.M., Weng, Z., Jowitt, S.M., 2013. A detailed assessment of global Cu resource trends and endowments. Econ. Geol. 108, 1163–1183. http://dx.doi.org/10.2113/econgeo.108.5.1163.
Mudd, G.M., Jowitt, S.M., Werner, T.T., 2016. The World's by-product and critical metal resources part I: uncertainties, current reporting practices, implications and ground for optimism. Ore Geol. Rev.. http://dx.doi.org/10.1016/j.oregeorev.2016.05.001. In press.
Mudd, G.M., Jowitt, S.M., Werner, T.T., 2017. The world's lead-zinc mineral resources: scarcity, data, issues and opportunities. Ore Geol. Rev. 80, 1160–1190. http://dx.doi.org/10.1016/j.oregeorev.2016.08.010.
Mudd, G.M., 2008. Sustainability reporting and water resources: a preliminary assessment of embodied water and sustainable mining. Mine Water Environ. 27, 136–144. http://dx.doi.org/10.1007/s10230-008-0037-5.
Murguía, D.I., Bringezu, S., Schaldach, R., 2016. Global direct pressures on biodiversity by large-scale metal mining: spatial distribution and implications for conservation. J. Environ. Manage. 180, 409–420. http://dx.doi.org/10.1016/j.jenvman.2016.05.040.
Nassar, N.T., Graedel, T.E., Harper, E.M., 2015. By-product metals are technologically essential but have problematic supply. Sci. Adv. 1, 1–10. http://dx.doi.org/10.1126/sciadv.1400180. e1400180.
Nordstrom, D.K., Blowes, D.W., Ptacek, C.J., 2015. Hydrogeochemistry and microbiology of mine drainage: an update. Appl. Geochem. 57, 3–16. http://dx.doi.org/10.1016/j.apgeochem.2015.02.008.
Nordstrom, D.K., 2009. Acid rock drainage and climate change. J. Geochem Explor. 100, 97–104. http://dx.doi.org/10.1016/j.gexplo.2008.08.002.
Northey, S., Haque, N., Mudd, G., 2013. Using sustainability reporting to assess the environmental footprint of copper mining. J. Clean. Prod. 40, 118–128. http://dx.doi.org/10.1016/j.jclepro.2012.09.027.
Northey, S.A., Haque, N., Lovel, R., Cooksey, M.A., 2014a. Evaluating the application of water footprint methods to primary metal production systems. Miner. Eng. 69, 65–80. http://dx.doi.org/10.1016/j.mineng.2014.07.006.
Northey, S., Mohr, S., Mudd, G.M., Weng, Z., Giurco, D., 2014b. Modelling future copper ore grade decline based on a detailed assessment of copper resources and mining. Resour. Conserv. Recycl. 83, 190–201. http://dx.doi.org/10.1016/j.resconrec.2013.10.005.
Northey, S.A., Mudd, G.M., Saarivuori, E., Wessman-Jääskeläinen, H., Haque, N., 2016. Water footprinting and mining: where are the limitations and opportunities? J. Clean. Prod. 135, 1098–1116. http://dx.doi.org/10.1016/j.jclepro.2016.07.024.
Pearce, T.D., Ford, J.D., Prno, J., Duerden, F., Pittman, J., Beaumier, M., Berrang-Ford, L., Smit, B., 2011. Climate change and mining in Canada. Mitig. Adaptation Strategies Global Change 16, 347–368. http://dx.doi.org/10.1007/s11027-010-9269-3.
Peel, M.C., Finlayson, B.L., McMahon, T.A., 2007. Updated world map of the Köppen-Geiger climate classification. Hydrology and Earth System Sciences Discussions 4. pp. 439–473.
Pfister, S., Koehler, A., Hellweg, S., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environ. Sci. Technol. 43, 4098–4104. http://dx.doi.org/10.1021/es802423e.
Pfister, S., Boulay, A.-M., Berger, M., Hadjikakou, M., Motoshita, M., Hess, T., Ridoutt, R., Weinzettel, J., Scherer, L., Döll, P., Manzardo, A., Núñez, M., Verones, F., Humbert, S., Harding, K., Benini, L., Oki, T., Finkbeiner, M., Henderson, A., 2017. Understanding the LCA and ISO water footprint: a response to Hoekstra (2016) A critique on the water-scarcity weighted water footprint in LCA. Ecol. Indic. 72, 352–359. http://dx.doi.org/10.1016/j.ecolind.2016.07.051.
Phillips, J., 2016. Climate change and surface mining: a review of environment-human interactions & their spatial dynamics. Appl. Geogr. 74, 95–108. http://dx.doi.org/10.1016/j.apgeog.2016.07.001.
QFCI, 2012. Chapter 13, mining. Final Report of the Queensland Floods Commission of Inquiry. QFCI), Australia March.
QRC, 2011. QRC Submission: Queensland Floods Commission of Inquiry. Queensland Resources Council (QRC), Australia March.
Rayne, S., Forest, K., Friesen, K.J., 2009. Analytical framework for a risk-based estimation of climate change effects on mine site runoff water quality. Mine Water Environ. 28, 124–135. http://dx.doi.org/10.1007/s10230-009-0070-z.
Ridoutt, B.G., Pfister, S., 2013. A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator. Int. J. Life Cycle Assess. 18, 204–207. http://dx.doi.org/10.1007/s11367-012-0458-z.
Rogelj, J., Meinshausen, M., Knutti, R., 2012. Global warming under old and new scenarios using IPCC climate sensitivity range estimates. Nat. Clim. Change 2, 248–253. http://dx.doi.org/10.1038/NCLIMATE1385.
Rubel, F., Kottek, M., 2010. Observed and projected climate shifts 1901–2100 depicted by world maps of the Köppen-Geiger climate classification. Meteorol. Z. 19 (2), 135–141. http://dx.doi.org/10.1127/0941-2948/2010/0430.
Scherer, L., Venkatesh, A., Karuppiah, R., Pfister, S., 2015. Large-scale hydrological modeling for calculating water stress indices: implications of improved spatiotemporal resolution, surface-Groundwater differentiation, and uncertainty characterization. Environ. Sci. Technol. 49, 4971–4979. http://dx.doi.org/10.1021/acs.est.5b00429.
Shang, Y., Lu, S., Li, X., Hei, P., Lei, X., Gong, J., Liu, J., Zhai, J., Wang, H., 2016. Balancing development of major coal bases with available water resources in China through 2020. Appl. Energ.. http://dx.doi.org/10.1016/j.apenergy.2016.07.002. In press 16p.
Sharma, V., Franks, D.M., 2013. In situ adaptation to climatic change: mineral industry responses to extreme flooding events in Queensland, Australia. Soc. Nat. Resour. 26 (11), 1252–1267. http://dx.doi.org/10.1080/08941920.2013.797528.
Sonderegger, T., Pfister, S., Hellweg, S., 2015. Criticality of water: aligning water and mineral resources assessment. Environ. Sci. Technol. 49, 12315–12323. http://dx.doi.org/10.1021/acs.est.5b02982.
Taylor, M.P., Little, J.A., 2013. Environmental impact of a major copper mine spill on a river and floodplain system. Anthropocene 3, 36–50. http://dx.doi.org/10.1016/j.ancene.2014.02.004.
USGS, 2013. Metal Prices in the United States Through 2010: U.S. Geological Survey (USGS) Scientific Investigations Report 2012–5188. . 204p. http://pubs.usgs.gov/sir/2012/5188 (accessed 14/07/2016).
USGS, 2016. Copper. Mineral Commodity Summaries, January 2016. U.S. Geological Survey (USGS)pp. 54–55.
van der Ent, R.J., Savenije, H.H.G., Schaefli, B., Steele-Dunne, S.C., 2010. Origin and fate of atmospheric moisture over continents. Water Resour. Res. 46, W09525. http://dx.doi.org/10.1029/2010WR009127.
Vergara, M.R., Trianafyllidis, T., 2015. Swelling behavior of volcanic rocks under cyclic wetting and drying. Int. J. Rock Mech. Min. Sci. 80, 231–240. http://dx.doi.org/10.1016/j.ijrmms.2015.08.021.
WULCA, 2016. Consensual Method Development: Preliminary Recommendation –AWaRe v1.2. Water Use in LCA (WULCA) working group. Accessible from: www.wulca-

waterlca.org (accessed 08-05-2016).
Wang, B., Peng, Y., 2014. The effect of saline water on mineral flotation – a critical review. Miner. Eng. 66–68, 13–24. http://dx.doi.org/10.1016/j.mineng.2014.04.017.
Werner, T.T., Mudd, G.M., Jowitt, S.M., 2017. The World's by-product and critical metal resources part II: a method for quantifying the resources of rarely reported metals. Ore Geol. Rev. 80, 658–675. http://dx.doi.org/10.1016/j.oregeorev.2016.08.008.
Wessman, H., Salmi, O., Kohl, J., Kinnunen, P., Saarivuori, E., Mroueh, U.-M., 2014. Water and society: mutual challenges for eco-efficient and socially acceptable mining in Finland. J. Clean. Prod. 84, 289–298. http://dx.doi.org/10.1016/j.jclepro.2014.04.026.
Zhang, X., Yan, X., 2016. Deficiencies in the simulation of the geographic distribution of climate types by global climate models. Clim. Dyn. 46, 2749–2757. http://dx.doi.org/10.1007/s00382-015-2727-6.
Ziemkiewicz, P.F., Skousen, J.G., Simmons, J., 2003. Long-term performance of passive acid mine drainage treatment systems. Mine Water Environ. 22, 118–129. http://dx.doi.org/10.1007/s10230-003-0012-0.

5. Production Weighted Water Use Impact Characterisation Factors for the Global Mining Industry

Life cycle assessment impact characterisation for water use have largely been developed in the context of the agricultural industry. Currently spatial water use impact characterisation factors are developed using the outputs of global hydrological and water use models. In order to develop characterisation factors that match the geographic boundaries of national, continental or global life cycle inventories, it is necessary for finer resolution characterisation factors to be weighted according to the spatial and temporal distribution of water use across all (sub-) watersheds within these boundaries. There is limited understanding of how the results of life cycle impact assessment will differ according to the spatial boundaries of assessment, particularly for industries such as mining that are generally considered relatively minor consumers of water and so are less likely to be spatially correlated with overall water use in a region. To address this we evaluated the representativeness of existing impact characterisation factors for use when assessing water use impacts of the mining industry.

In order to provide recommendations for the appropriate spatial boundaries for life cycle assessment of the mining industry, we have developed alternative aggregations of water use impact characterisation factors to assess the potential deviation arising from impact evaluation at different spatial scales. The deviation between the Water Stress Index (WSI) (Pfister et al., 2009) and the Available Water Remaining (AWaRe) method (Boulay et al., 2017) characterisation factors were assessed for 25 mined commodities at operational, national and global boundaries of assessment. As global datasets of mining operation water use are unavailable, characterisation factors were weighted according to commodity production from individual mining operations and nations.

The results of the study highlighted that life cycle impact assessment based upon the existing national average characterisation factors would display a bias to overestimate water use impacts for the mining industry, compared to results of assessments performed for individual operations and watersheds. Several conceptual limitations with existing characterisation methods were also identified, such as the current unavailability of an impact method well suited for assessing fossil groundwater consumption. The study also allowed an assessment of the relative exposure of mineral commodity production to water stress or scarcity, and highlighted how spatial boundaries may change our perspective of which commodities are exposed to these issues.

This study has been submitted to the Journal of Cleaner Production and is currently under review. Further supplementary tables and figures are presented in Appendix B of this thesis.

Reference:

Northey, S.A., Madrid Lopez, C., Haque, N., Mudd, G.M., Yellishetty, M. Production weighted water use impact characterisation factors for the global mining industry: A comparison of watershed and national average WSI and AWaRe factors. Submitted to the Journal of Cleaner Production, under review.

5.1. Abstract

Methods for quantifying the impacts of water use within life cycle assessment have developed significantly over the past decade. These methods account for local differences in hydrology and water use contexts through the use of regionally specific impact characterisation factors. However, few studies have applied these methods to the mining industry and so there is limited understanding regarding how spatial boundaries may affect assessments of the mining industry's consumptive water use impacts. To address this, we developed production weighted characterisation factors for 25 mineral and metal commodities based upon the spatial distribution of global mine production across watersheds and nations. Our results indicate that impact characterisation using the national average 'Water Stress Index' (WSI) would overestimate the water use impacts for 67% of mining operations when compared to assessments using watershed WSI values. Comparatively, national average 'Available Water Remaining' (AWaRe) factors would overestimate impacts for 60% of mining operations compared to assessments using watershed factors. In the absence of watershed scale inventory data, assessments may benefit from developing alternative characterisation factors reflecting the spatial distribution of commodity production across watersheds. The results also provide an indication of the commodities being mined in highly water stressed or scarce regions.

5.2. Introduction

The global mining industry is situated across a wide range of regional hydrological contexts that can result in complex water related risks that must be managed by mining and mineral processing operations (CDP, 2013; Northey et al., 2017). In order to mitigate or manage these risks, mining operations will tailor their management practices and process design to address the specific hydrological conditions affecting the site (Kunz and Moran, 2016), As a result of this, it has been observed that there is significant variability in rates of water consumption and efficiency between mining operations (Mudd, 2008; Gunson, 2013; Northey et al., 2013). Given the myriad of drivers that influence water consumption throughout the mining industry, methodological approaches are required that enable the fair comparison of water consumption and efficiency of mine sites located across geographic regions, which may have significantly different climate, hydrological and water use contexts. To address this, recent studies that evaluate water consumption in the mining industry are increasingly utilising spatially explicit life cycle impact characterisation factors to account for differences in the local water scarcity or stress of mines located in different regions (Northey et al., 2016).

Life cycle impact assessment aims to quantify the environmental burdens associated with the provision of products or services. A variety of methods have been proposed over the last decade for characterising the relative water use impacts of production systems as part of life cycle assessment studies (Boulay et al., 2015a; Kounina et al., 2013). These methods differ based upon the underlying conceptualisation of what constitutes a water use impact, their data underpinnings and the approach taken to calculating and normalising impact characterisation factors. These impact characterisation factors are typically modelled for (sub-) watersheds based on the outputs of global hydrological and water use models. Characterisation factors for different spatial scales (e.g. regional, national, continental) may be determined via weighting watershed factors based on the distribution of withdrawals or consumption across the region. Consequently, these national average factors are largely representative of the conditions where major water users, such as the agricultural industry, are situated. Although mining can occasionally be a large local consumer of water within an individual watershed, at national scales other industries – such as agriculture – typically consume at least an order of magnitude more water (Gunson, 2013; Hejazi et al., 2014). As the spatial distribution of mineral resources may not be correlated with the spatial distribution of overall water use or availability within a region, we hypothesise that assessments of the mining industry's water consumption may produce substantially different results depending on whether watershed or national average impact characterisation factors are used.

This paper tests the above hypothesis by developing production weighted average characterisation factors based on the spatial distribution of mine site production across watersheds and countries. Region specific weighted average factors are developed for twenty five mined commodities and compared with national average factors to understand the influence that spatial scale and watershed aggregation procedures would have on the accuracy of impact assessment of mined products. The results of the study also provide an indication of the relative exposure of global mining industry sub-sectors to water stress and scarcity related risks.

5.3. Background and Methods

5.3.1. Water Use Impact Characterisation Factors

A variety of methods have been proposed over the last decade for characterising the relative water use impacts of production systems as part of life cycle assessment studies (Boulay et al., 2015a; Kounina et al., 2013). Our assessment focuses upon the widely used 'Water Stress Index'(WSI)(Pfister et al., 2009) and the recently developed 'Available Water Remaining' (AWaRe) methods (Boulay et al., 2016, 2017; WULCA, 2017). The potential influence that characterisation factors produced at different spatial scales would have on water use impact estimates for the mining industry is assessed by considering the spatial distribution of mine site production.

5.3.1.1. Water Stress Index (WSI)

The WSI was developed by Pfister et al. (2009) as a mid-point indicator to measure the potential for water use to lead to user deprivation. The basic data underpinning the WSI is the ratio of water withdrawals to long-term water availability (WTA) within a watershed. These WTA ratios are modified by a variation factor to account for the degree of precipitation variability and the regulation of flows within the watershed (defined by Nilsson et al., 2005), according to Equation 1 and Equation 2. The modified WTA is then scaled between 0.01 and 1 using a logistic function shown in Equation 3 to produce the WSI. This logistic function is calibrated so that a WSI of 0.5 corresponds to a WTA of 0.4 (assuming the median watershed variation factor), which is commonly considered the threshold between moderate and severe water scarcity. Pfister et al. (2009) provided annual WSI data on a watershed basis, using data from the WaterGAP 2 global hydrological and water use model (Alcamo et al., 2003), as well as national averages developed by weighting watershed data according to the spatial distribution of withdrawals. Although there has been criticism and debate over the conceptualisation of the WSI (Hoekstra, 2016; Pfister et al., 2017), it is perhaps the most widely used approach to assessing consumptive water use impacts within life cycle assessment studies to date. Other conceptualisations of the WSI with alternative normalisation methods have been proposed to account for differences when assessing marginal and consequential water use impacts in life cycle assessment (Pfister and Bayer, 2014). However, in this assessment we focus on the original WSI presented by Pfister et al. (2009) because it is the most extensively used water use impact characterisation factor.

Equation 1: $$WTA^* = \begin{cases} \sqrt{VF} \times WTA & for\ non-strongly\ regulated\ flows \\ VF \times WTA & for\ strongly\ regulated\ flows \end{cases}$$

Equation 2: $$VF = \frac{1}{\sum P_i} \sum_{i=1}^{n} e^{\sqrt{\ln(S_{month})^2 + ln(S_{year})^2}}$$

Where: Pi is the mean annual precipitation in each grid cell i within a watershed, and S_{month} and S_{year} represent the standard deviation of monthly and annual precipitation respectively.

Equation 3: $$WSI = \frac{1}{1 + e^{-6.4 \cdot WTA^*} (\frac{1}{0.01} - 1)}$$

5.3.1.2. Available Water Remaining (AWaRe)

The international working group for Water Use in Life Cycle Assessment (WULCA) developed the Available Water Remaining (AWaRe) method as a consensus based approach for assessing the potential for water use to deprive other users of water (Boulay et al., 2015b, 2016, 2017; WULCA, 2017). The basic underpinning of the AWaRe method is the inverse of water availability minus demand (AMD) from environmental water requirements (EWR) and human water consumption (HWC) per unit area (equation 4), which can be interpreted as the surface-time equivalent (STE) required to produce the excess water availability in a region ($m^2.month.m^{-3}$). The AWaRe characterisation factors are determined from sub-watershed AMD values that have been normalised according to Equation 5, so that a value of 1 is equivalent to the global consumption weighted average AMD (0.0136 $m^3.m^{-2}.month^{-1}$). Therefore an AWaRe value of 20 represents a region where there is 20 times less excess water available per unit area than the global average.

Equation 4: $$\frac{1}{AMD_i} = \frac{Area}{A-D} = \frac{Area}{A-HWC-EWR} = STE_i$$

Equation 5: $$AWaRe_{ws,month} = \begin{cases} AMD_{world\ ave.}/AMD_i & for\ D_i < A_i \\ 0.1 & for\ AMD_i > 10 \cdot AMD_{world\ average} \\ 100 & for\ D_i \geq A_i\ or\ AMD_i < AMD_{world\ average}/100 \end{cases}$$

WULCA (2017) has provided AWaRe factors for a variety of temporal and spatial scales based upon monthly sub-watershed data from WaterGAP 2.2, which is underpinned by a global hydrological model (Müller Schmied et al., 2014) and water use data (Flörke et al., 2013). Annual AWaRe factors at the watershed scale are produced via weighting monthly watershed factors by monthly water consumption in the watershed, according to Equation 6. National AWaRe factors are produced by first spatially averaging monthly watershed factors by the monthly water consumption occurring in each watershed in the country according to equation 7. The resulting monthly, national AWaRe factors are then weighted according to the total water consumption occurring in the country in each month, according to Equation 8, to produce the annual factor. In order to more accurately assess the impacts of different industries, AWaRe factors are available that have been weighted based upon the temporal and spatial distribution of agricultural, non-agricultural and total water consumption (for assessing unknown water use). The annual watershed and national average AWaRe factors weighted by the spatial and temporal distribution of non-agricultural water consumption are used in this study.

Equation 6: $$AWaRe_{ws,annual} = \frac{1}{C_{ws,annual}} \sum_{month=1}^{12} AWaRe_{ws,month} \cdot C_{ws,month}$$

Equation 7: $$AWaRe_{national,month} = \frac{1}{C_{national,month}} \sum_{ws=1}^{n} AWaRe_{ws,month} \cdot C_{ws,month}$$

Equation 8: $$AWaRe_{national,annual} = \frac{1}{C_{national,annual}} \sum_{month=1}^{12} AWaRe_{national,month} \cdot C_{national,month}$$

5.3.2. Mine Production Weighted Averages and Boundaries of Assessment

To determine the potential influence of spatial aggregation of life cycle inventories and impact characterisation factors on assessments of mining industry water use, the deviation between watershed and national average factors was assessed at several different spatial boundaries - global, national and individual mining operations. Ideally this assessment would be based upon the spatial distribution of mining industry water consumption between watersheds and nations within these boundaries. However, there is currently no global inventory of the water use requirements of individual mining operations at a level of detail that would facilitate this. Fortunately, commodity production from individual mining operations is available for a large proportion of the mining industry

(Table 2) and this has been used as a proxy for mine-site water use. Therefore weighted average WSI and AWaRe factors were estimated based upon the spatial distribution of annual commodity production amongst watersheds and nations (equation 9).

Equation 9: $$Production\ Weighted\ Average = \frac{\sum_i CF_i \times Production_i}{\sum_i Production_i}$$

Where: CF_i and $Production_i$ represent the AWaRe or WSI value and annual commodity production respectively of each watershed or nation "i" within the boundary of assessment.

Production data for the year 2014 was obtained for 25 mined commodities: antimony, bauxite, chromite, coal, cobalt, copper, diamonds, gold, iron ore, lead, lithium, manganese, molybdenum, nickel, palladium, phosphate, platinum, potash, rutile, silver, tin, tungsten, uranium, zinc and zircon (Table 2). National mine production data was sourced from the British Geological Survey (2016), whereas production and location data for individual mining operations was sourced from the SNL Mining & Metals database (SNL, 2017). Operations that did not intersect with both the spatial WSI and AWaRe datasets were excluded from the analysis. The operation production data covers approximately 50-90% of the national production for most commodities considered. The spatial distribution of the mining operations in relation to the WSI and AWaRe factors is shown in Figure 5.1 and a summary of the production data is provided in Table 2 for individual commodities. Data for individual countries is provided in the Appendix Table A. 1 to Table A. 30. For several commodities there was inconsistency in the reporting basis (i.e. contained metal and gross mineral) used for the available national and operation production data. For uranium production, the national data was converted from a U to a U_3O_8 basis to ensure consistency with the operation data. For potash it was not possible to express the national and operational scale production data in consistent units. The national data represents production on a K_2O equivalent basis, whereas the operation data represents gross potash minerals (it was not possible to determine the mineralogy of each operation's products). However, our judgement is that the overall production coverage of potash is likely high, hence in the figures presented in subsequent sections, potash has been labelled as having greater than 75% production coverage.

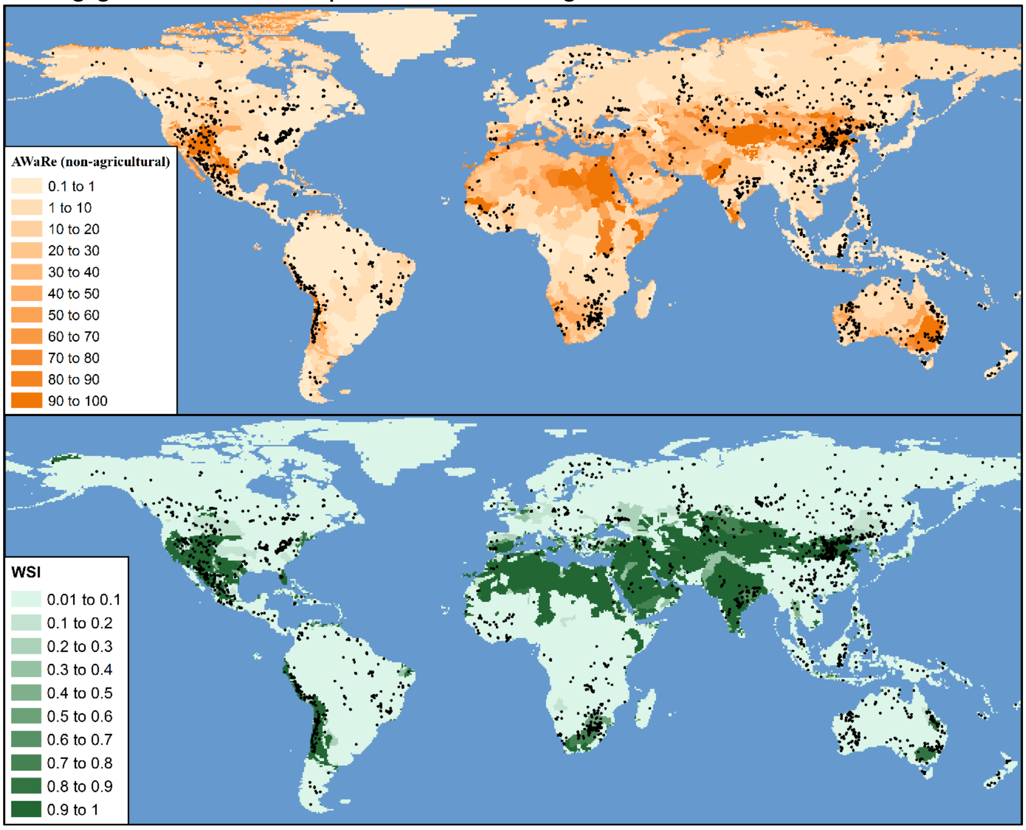

Figure 5.1: Location of mining operations (SNL, 2017) considered in this study in relation to annual AWaRe factors for non-agricultural water use (Boulay et al., 2016, 2017; WULCA, 2017) and the WSI (Pfister et al., 2009).

Table 2: Summary of 2014 production data and key results for the 25 mined commodities considered by this study.

	National Production [e]		Operation Production [f]				Global Production Weighted Average						% of Production	
	Countries	Production	Operations	Countries	Production	Coverage	National Average (NA) [e]		Watershed (W) [f]		Ratio (W/NA)		Overestimated (W>NA)	
Commodity	No.	kg	No.	No.	kg	% of national	AWaRe	WSI	AWaRe	WSI	AWaRe	WSI	AWaRe	WSI
Antimony [a]	17	1.57×10^{8}	2	2	6.02×10^{6}	4	26.5	0.49	61.7	0.71	2.33	1.44	0	40
Bauxite [b]	31	2.60×10^{11}	47	15	1.91×10^{11}	74	19.9	0.37	6.2	0.11	0.31	0.31	88	90
Chromite [b]	18	3.00×10^{10}	31	7	1.45×10^{10}	48	20.6	0.67	18.7	0.48	0.9	0.72	47	92
Coal [b]	66	8.09×10^{12}	1138	30	4.93×10^{12}	61	20.2	0.46	25.2	0.45	1.25	0.98	53	60
Cobalt [a]	18	1.29×10^{8}	29	15	6.40×10^{7}	50	10.1	0.11	4.5	0.07	0.44	0.67	92	90
Copper [a]	56	1.84×10^{10}	270	44	1.60×10^{10}	87	23.1	0.49	40.9	0.55	1.77	1.13	36	44
Diamonds [b]	23	2.51×10^{4}	39	11	1.96×10^{4}	78	14.3	0.27	17.4	0.07	1.21	0.27	47	96
Gold [a]	88	3.02×10^{6}	661	75	2.30×10^{6}	76	18.9	0.38	21.7	0.31	1.15	0.83	56	64
Iron Ore [b]	48	3.38×10^{12}	280	35	1.77×10^{12}	52	21.8	0.42	22.5	0.15	1.03	0.36	38	88
Lead [a]	43	5.37×10^{9}	103	23	1.96×10^{9}	36	22.3	0.48	15	0.27	0.67	0.56	77	74
Lithium [b]	9	6.54×10^{8}	2	2	9.90×10^{7}	15	25	0.43	35.1	0.31	1.41	0.73	70	70
Manganese [b]	25	5.47×10^{10}	26	11	3.37×10^{10}	68	20.5	0.48	23.4	0.11	1.14	0.23	67	84
Molybdenum [a]	14	2.95×10^{8}	42	13	2.00×10^{8}	68	23.7	0.55	46.5	0.7	1.96	1.28	32	35
Nickel [a]	31	2.06×10^{9}	77	21	1.50×10^{9}	73	9.4	0.24	8.6	0.12	0.91	0.52	85	89
Palladium [a]	11	1.84×10^{5}	31	6	1.84×10^{5}	100	9.8	0.33	8.3	0.18	0.84	0.53	63	100
Phosphate [b]	40	2.45×10^{11}	22	8	4.71×10^{10}	19	29.7	0.56	9.3	0.51	0.31	0.91	87	52
Platinum [a]	10	1.45×10^{5}	40	7	1.61×10^{5}	111	14.9	0.51	15.8	0.39	1.06	0.77	29	99
Potash [c]	12	3.91×10^{10}	22	6	6.86×10^{10}	[c]	11.1	0.26	11.3	0.15	1.02	0.6	71	86
Rutile [b]	12	7.40×10^{8}	9	5	4.46×10^{8}	60	17.8	0.38	37.6	0.29	2.11	0.74	60	66
Silver [a]	65	2.74×10^{7}	257	40	1.77×10^{7}	64	17.9	0.51	21.5	0.46	1.21	0.9	63	60
Tin [a]	23	3.55×10^{8}	7	5	7.02×10^{7}	20	18.1	0.34	1.6	0.35	0.09	1.03	100	65
Tungsten [a]	20	8.53×10^{7}	4	4	1.28×10^{7}	15	25.1	0.42	2.6	0.07	0.11	0.16	100	100
Uranium [d]	18	6.59×10^{7}	55	15	6.40×10^{7}	97	21.8	0.42	27.7	0.44	1.27	1.04	56	58
Zinc [a]	49	1.37×10^{10}	140	33	8.08×10^{9}	59	21.2	0.5	15.4	0.35	0.73	0.7	73	68
Zircon [b]	17	1.57×10^{8}	11	7	8.48×10^{8}	72	19.3	0.47	31	0.14	1.6	0.29	71	85

Reporting Basis

[a] Produced or contained metal. [b] Gross mineral. [c] National scale data is expressed on a K_2O equivalent basis. Operation data expressed on a gross potash mineral basis. [d] U_3O_8 equivalent.

Production Data Basis

[e] National production in 2014 (British Geological Survey, 2016). [f] Operation production in 2014 (SNL, 2017).

5.4. Results and Discussion

Production weighted WSI and AWaRe factors were developed according to equation 9 for all 25 mined commodities at global, national and operational boundaries of assessment. The results provide a basis for evaluating the influence that the use of watershed or national average factors would have upon impact assessments of the mining industry, as well as improving understanding of the relative exposure of mining supply chains to water stress and scarcity related risks. Key results for all commodities are provided in Table 2, with more detailed results for commodity production from individual countries being presented in Appendix Figure A. to Figure A. 7 and Table A. 1 to Table A. 30 (see Appendix B).

5.4.1. Deviation between watershed and national average factors

The deviation between watershed and national average factors for estimating the water use impacts of mined commodities was assessed at several different spatial aggregation boundaries, shown in Figure 5.2 for all commodities. For the boundary of individual mining operations, the local watershed factor is compared with the national average factor. At a national boundary, production weighted watershed values are compared with the national average factors. At the global boundary, watershed factors weighted according to operation production are compared with national average factors weighted production according to national production for each commodity assessed.

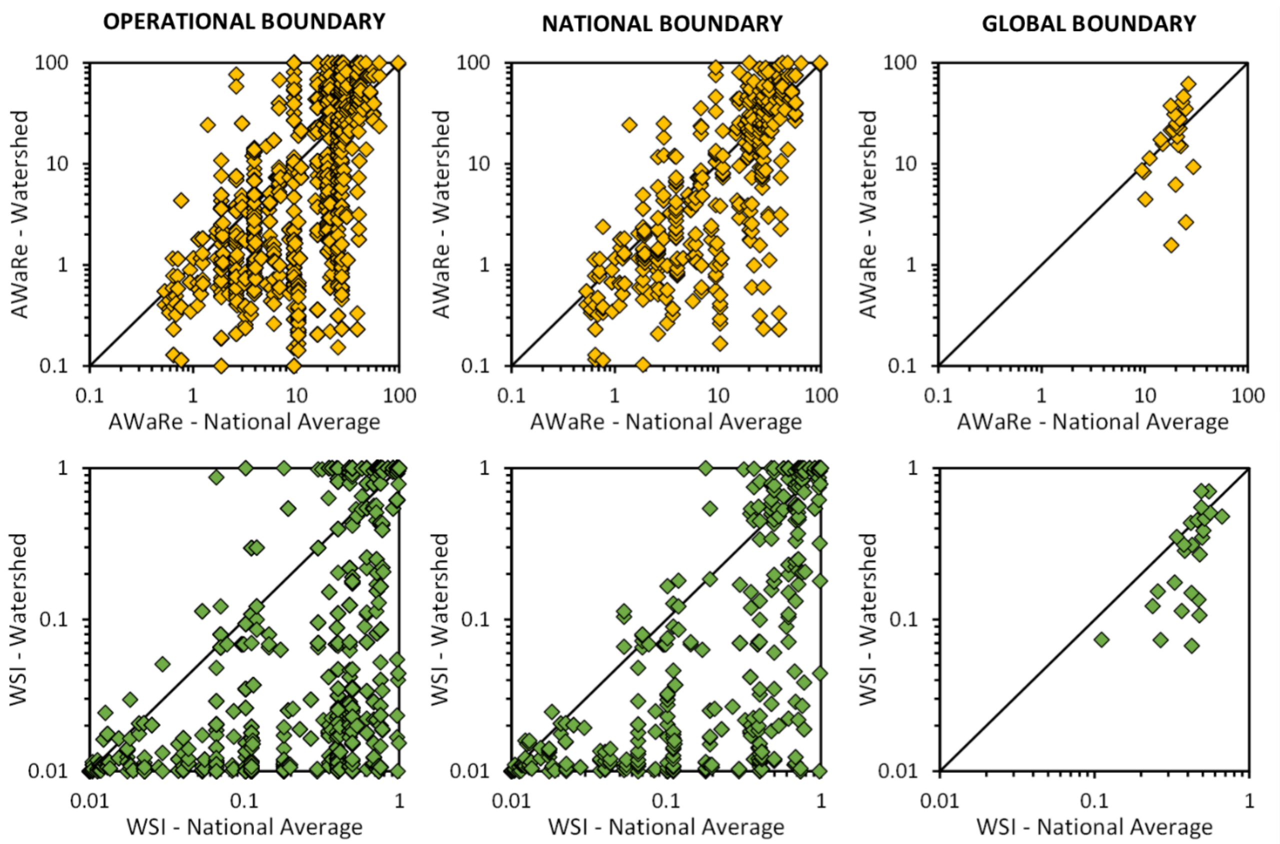

Figure 5.2: A comparison of production weighted average AWaRe and WSI factors for 25 mined commodities determined at operational, national and global system boundaries. Watershed characterisation factors were weighted based on production from individual mining operations in 2014 (SNL, 2017), whereas national average characterisation factors were weighted based upon national production in 2014 (British Geological Survey, 2016).

The ratio of watershed based factors to the national average based factors provides a measure of the potential error that may be introduced when assessing the impacts of the mining industry's water use. The ratio between factors can range up to several orders of magnitude for individual operations or countries (Figure 5.3). Deviations are slightly reduced at higher levels of spatial aggregation (i.e. global). Figure 5.2 and Figure 5.3 support our hypothesis that the use of national average water use characterisation factors would be likely to overestimate impacts for the mining industry, when compared to results generated from watershed based assessments. Figure 5.3 shows the magnitude of deviation and the proportion of mining operations whose water use impacts would be over or under-estimated by the use of national average characterisation factors, when compared to the use of watershed specific factors. Across the 25 mined commodities, the use of national average WSI is likely to overestimate impacts for 67% of mining operations and 72% of impact estimates at the national boundary. The use of non-agricultural AWaRe factors also leads to a similar tendency to overestimate water use impacts, albeit for only 60% operations and 60% of national estimates. It is clear that the use of national average factors will lead to a systemic bias to overestimate water use impacts of the mining industry, compared to if watershed scale data was used. This bias is observable at all scales of spatial aggregation, although the average deviation is reduced at higher levels of spatial aggregation.

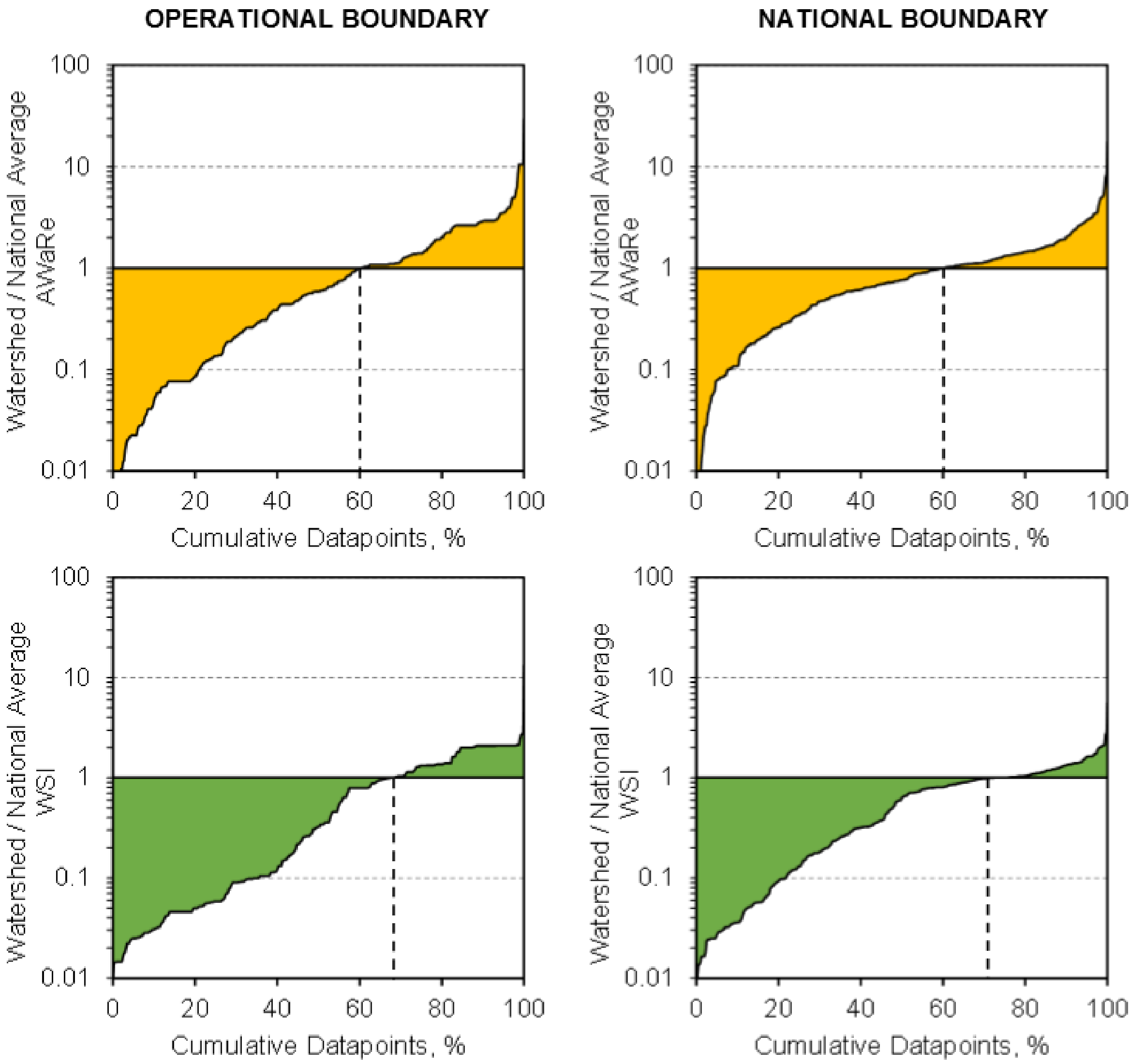

Figure 5.3: Ratio between watershed and national average AWaRe and WSI factors for all commodities at the operational and national boundary. The proportion of datapoints overestimated by national average factors when compared to watershed factors is indicated by the vertical dotted lines.

Assuming water use is uniform across production, global assessments based on national average factors would overestimate impacts for 20 (out of 25) commodities using the WSI and for 10 commodities using AWaRe factors for non-agricultural water use (see Figure 5.4, Table 2 and Appendix Figure A. 7). However, even when the overall results for a given commodity are under- or overestimated, it is important to recognise that the individual regions where this commodity is produced may display significant divergence between watershed and national average factors. The cumulative production distribution of individual commodities in relation to the national average and watershed AWaRe and WSI is shown in Table 2 and Appendix Figure A. to Figure A. 4. These distributions highlight that, while there is a tendency for the national average factors to overestimate the impacts across the mining industry, this can be highly variable between individual mined commodities – with a large proportion of individual commodity production being over- or underestimated. Therefore the reliability of life cycle assessment studies that seek to redistribute mined commodity production away from water stressed or scarce regions may be severely limited by the use of national average characterisation factors.

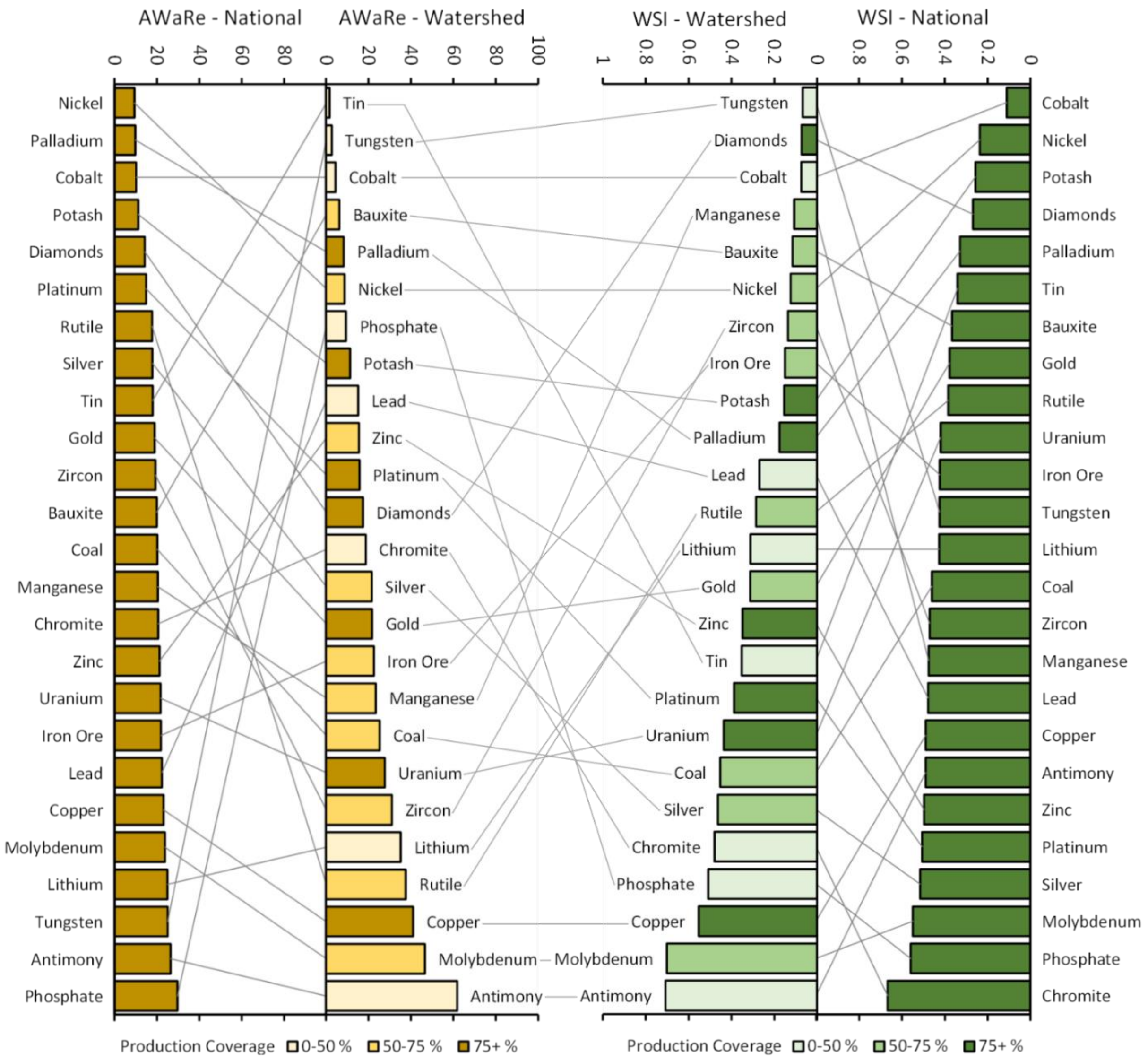

Figure 5.4: Global production weighted average WSI and non-agricultural AWaRe factors for each commodity based upon watershed and national average water indices. Grey lines show the change in relative ranking of each commodity depending upon the scale and characterisation factor used. Watershed factors were weighted based upon production from individual mining operations in 2014 (SNL, 2017), whereas national average factors were weighted based upon national production in 2014 (British Geological Survey, 2016).

Given that non-agricultural products are less likely to be spatially correlated with overall water use, it is reasonable to expect that the results of this study for the WSI may be similar when assessing other minor water consuming industries (e.g. manufacturing). During the development of the AWaRe factors, the provision of factors specific to agricultural, non-agricultural and unknown water use partially addresses this issue. However our results (Figure 5.2 and Figure 5.3) show that the non-agricultural factors still show a bias to overestimate impacts of the mining industry. Therefore assessments of the consumptive water use impacts of mining should be conducted at watershed scales whenever possible.

To provide further evidence of this judgement, Appendix Figure A. 7 shows that the use of national average factors will result in commodity production weighted factors that display a strong reversion to the global average WSI of 0.602 (Ridoutt and Pfister, 2013b) and the global consumption weighted average AWaRe factor for non-agricultural water use of 20 (Boulay et al., 2016), when compared to the watershed based assessment. Therefore finer levels of spatial resolution provide improved discriminatory power when assessing water use impacts, particularly for assessments of production systems that are less likely to be spatially correlated with the distribution of water use across watersheds.

5.4.2. Relative exposure of commodities to water stress and scarcity

The WSI and AWaRe characterisation factors can also be interpreted as indicators of contextual water risk for the mining industry (Northey et al., 2014a, 2017a). Local scarcity or overexploitation of water resources in a region can impact mining operations in a range of ways, from making water sourcing more difficult to increasing tension with competing water users such as agriculture and/or other consumptive uses.

The relative exposure of commodity production to these issues may be ranked using the global production weighted average factors as shown in Figure 5.4. Commodities being mined in highly water stressed or scarce regions include phosphate, molybdenum and copper. Other commodities such as nickel, cobalt and potash are predominantly mined in less water stressed regions. There are some differences in the relative ‘ranking’ of commodity risk depending on whether WSI or AWaRe factors are used. For instance, chromite is one of the most exposed commodities when using the WSI, however the AWaRe factors suggest that chromite production is only moderately exposed to these issues when compared with the other mined commodities. A major reason for these differences is that each of the indicators, WSI and AWaRe, are fundamentally measuring a different aspect of local hydrology due to the differences in their formulation. The WSI is a normalised measure of the ratio of withdrawals to long-term water availability modified by inter- and intra-annual hydrologic variability, which may be interpreted as a relative measure of the intensity of withdrawals or competition for water use between regions. Whereas the AWaRe index is an indicator based upon the absolute availability of water beyond current demands and so to a greater degree also reflects other hydrological factors – beyond the intensity of available water use – such as the general aridity of the region. Therefore, any comparison of the relative ‘ranking’ of the exposure to water risks of an individual commodity’s production based upon these indicators should carefully consider the underlying formulation of each index, as well as the hydrological context of the countries producing each commodity (the distribution of commodity production amongst countries is provided in the electronic supplementary tables).

Differences in the relative ranking of commodities also occurs depending on if watershed or national average factors are used. Compared to the national average factors, watershed based assessment will result in the relative exposure ranking increasing for 11 and 15 commodities for the WSI and AWaRe factors respectively. However, it is important to emphasise that the uncertainty of the relative results based upon watershed assessment is closely related to the degree of operation production data available for each commodity (refer to Table 2 or the shading in Figure 5.4). The operation production data covers less than 50% of global production for 8 of the 25 commodities (antimony, cobalt, chromite, lead, phosphate, lithium, tin, tungsten), and so the watershed based results are more uncertain for these commodities.

Previously, Northey et al. (2014a) identified that different processing stages of mineral and metal supply chains (e.g. mining, mineral concentrating, smelting and refining) are not always co-located and may be located in regions experiencing substantially different water stress. Therefore, further assessment of the relative exposure of commodity production to water stress risks may benefit from considering the spatial distribution of the downstream production processes following mining – as this may alter the overall risk profile for an individual commodity.

The WSI and AWaRe factors were also used as part of an assessment of the global spatial distribution of copper, lead-zinc and nickel resources in relation to regional climate zones and water risks (see Chapter 4; Northey et al., 2017a). The study used watershed scale data for AWaRe and the WSI, as well as several other indicators such as water criticality (Sonderegger et al., 2015), blue water scarcity (Hoekstra et al., 2012), and the water depletion index (Berger et al., 2014). Weighting of these indices was conducted based upon remaining resources rather than production levels, providing an indication of how future supply and life-of-mine production may be distributed in relation to water stress and scarcity. The key findings were that copper resources are located in regions with higher water stress than either lead-zinc or nickel resources (see Chapter 4; Northey et al., 2017a), which is broadly consistent with our assessment of the current production of these commodities.

5.4.3. Suitability of production weighted average characterisation factors

A valid question is whether the spatial distribution of mining production is a reasonable proxy for the spatial distribution of mining industry water consumption. Studies have shown that there is considerable variability in the water use requirements of mining operations when expressed on a cubic metre per tonne of product basis, even for operations producing the same commodity (Gunson, 2013; Mudd, 2008; Northey et al., 2013). There are a range of causal reasons for this, including differences in: mineral deposits (e.g. ore grades and grain sizes), processing methods, site infrastructure, local climate and site water management practices. The water balance of individual mining operations can be quite dynamic and an individual operation may face risks associated with both shortfalls and excesses of water at different periods of time (Gao et al., 2017; Kunz and Moran, 2016; Northey et al., 2016, 2017). Detailed water balance modelling exists for individual mining operations, however currently there is very limited statistical understanding of how mine site water consumption varies across regions in response to local climates and hydrological settings

Conceptually, mining operations in dry climates will have lower surface runoff into on-site dams and greater evaporative losses, resulting in a greater dependence on surface and/or groundwater withdrawals. Conversely, mining operations in wet climates are likely to accumulate more water on-site and require active discharge through time. Given the role of local climates in governing regional water availability that underpins WSI and AWaRe factor estimates, there may be a correlation between mine water use intensity (i.e. cubic metre consumed per tonne of product) and the WSI or AWaRe factors. If this correlation was moderate, it would invalidate our assumption that mining production is a reasonable proxy for water use in this assessment. However, the absence of a comprehensive global mine water use dataset currently precludes our ability to test this assumption. Therefore, we must assume that mine production could be a reasonable proxy for estimation of mine water use across regions, whilst recognising that this assumption introduces uncertainty to the results of our assessment.

Although the input water requirements for mineral processing are relatively constant through the year, the overall water balance of a mining operation can vary substantially throughout the year and an individual operation will display seasonality in its water withdrawals from surrounding water sources or stores – as well as when water discharges will occur. All aspects including evaporation, flow into pits or mine workings, and runoff into water storage facilities may vary seasonally and in response to local catchment rainfall events (Northey et al., 2016). The temporal distribution of the mining industry's water use may differ from the temporal distribution considered during weighting of monthly factors when developing annualised characterisation factors.

Due to seasonal variations of water consumption, it has previously been determined that the annual WSI will, on average, underestimate impacts when assessing the agricultural industry – hence the use of sub-annual (e.g. monthly) characterisation factors is encouraged (Pfister and Bayer, 2014; Scherer et al., 2015; Scherer and Pfister, 2016). Another limitation of the WSI is that in some regions where water is physically scarce (e.g. central Australia), the region may not be considered water 'stressed' due to only limited water withdrawals occurring (possibly due to the regional water scarcity). Further compounding this is the tendency of the WaterGAP model to overestimate river discharge in arid regions (Scherer and Pfister, 2016). In contrast, the formulation of the AWaRe factors being based upon an absolute measure of excess water availability (Equation 4) overcomes the inherent limitations associated with the WSI being based upon the ratio of water withdrawals to availability (Equation 1). Therefore, we suggest that the use of AWaRe factors may be preferable to the WSI when assessing the water use impacts of mining operations located in arid regions, particularly when there are limited water withdrawals from other user groups.

Previous analysis of water use impact characterisation factors has found a substantial deviation when factors are developed at different spatial and temporal boundaries (Ansorge and Beránkova, 2017; Boulay et al., 2015a; Núñez et al., 2015; Scherer et al., 2015; Quinteiro et al., 2017). As the national average characterisation factors may not reflect the water use context of specific industries and commodities within a country, uncertainty data is available for the national average AWaRe factors reflecting the spatial and temporal differences in water use and availability across watersheds within a country (Boulay et al., 2017; WULCA, 2017). Appendix Figure A. 5 shows that 81% of mining operations and 90% of national production weighted averages fall within 1 standard deviation of the spatial uncertainty associated with the national average.

5.4.4. Limitations in the impact assessment of groundwater use

Many mining operations consume water from confined and unconfined aquifer systems – to meet the requirements of ore processing, dust suppression, waste management practices, and aquifer depressurisation to prevent groundwater seepage into mine voids or to alleviate slope stability issues (Northey et al., 2016). The potential impacts of mining operations on groundwater systems are highly complex, uncertain and site specific (Currell et al., 2017). Existing life cycle assessment based water use impact methods are not tailored to assess the potential impacts of water use on groundwater systems. Current approaches typically utilise estimates of water availability derived from global hydrological models, such as WaterGAP, which has been calibrated to estimate discharge from major river systems (Alcamo et al., 2003; Müller Schmied et al., 2014). Water availability in the determination of characterisation factors therefore largely reflects what can be considered 'flow' water resources (Madrid et al., 2013), however many mining operations extract groundwater from what may be considered 'fund' (i.e. rechargeable aquifers) or 'stock' (i.e. fossil aquifers) groundwater resources – which may require assessment using different water use impact pathways (Kounina et al., 2013; Milà I Canals et al., 2009). Approaches for assessing the impacts of fund and stock groundwater depletion are still underdeveloped within life cycle assessment and require further conceptualisation. This topic has been a focus of discussion within the recently formed WULCA sub-committee that is developing water use impact characterisation pathways to the natural resources area-of-protection within life cycle assessment.

5.4.5. Implications for mine water use disclosures and reporting

Over the past two decades there has been increasing transparency and reporting of water use data as part of corporate sustainability and environmental management reporting in the mining industry (Leong et al., 2014; Mudd, 2008; Northey et al., 2013). There is growing recognition of the need for local water scarcity or stress information to be reported alongside mine-site water use to facilitate meaningful interpretation of data (Northey et al., 2014, 2016).

The International Council on Mining & Metals (ICMM) recently released reporting guidelines to improve the quality and consistency of the industry's water use and risk disclosures (ICMM, 2017). ICMM member companies are expected to implement these standards by November 2018. The standard was heavily based upon the previous Water Accounting Framework for the Minerals Industry that was developed for the Minerals Council of Australia (2014), which has been shown to be broadly applicable for mine sites regardless of the local hydrological context (Danoucaras et al., 2014). Beyond clearly outlining water accounting procedures, the ICMM's standard also recommends that companies report on the local water stress of regions that they operate within by using tools, such as: the WRI Aqueduct Water Risk Atlas (2013), the GEMI Local Water Tool (2016), the WBCSD Global Water Tool (2015), or the WWF Water Risk Filter (2012). Northey et al. (2017) demonstrated that a more meaningful understanding of the mining industry's water use contexts could be achieved by considering multiple water risk indices simultaneously. Therefore, alternative watershed risk indices such as Water Criticality (Sonderegger et al., 2015), the WSI (Pfister et al., 2009) or AWaRe factors (Boulay et al., 2016, 2017; WULCA, 2017) may add further insight to the industry's water use reporting, and also facilitate greater data interoperability with LCA and water footprint assessments.

Often mining companies will aggregate the water withdrawal or consumption estimates of multiple mining operations into divisional, national or corporate totals when reporting the companies water use (Mudd, 2008; Northey et al., 2016). In these cases we emphasise that any water scarcity or stress information provided alongside this data should be sourced from watershed scale data, rather than national data that may not be representative of the local water use context of the company's individual operations.

5.4.6. Implications for life cycle inventory development

The results of a recent life cycle assessment methodology harmonisation project for metal associations recommended that water scarcity impacts should not be reported as part of life cycle assessment studies of metal supply, due to limitations with existing inventory data (e.g. high levels of spatial aggregation) and the need for further methodological development (Santero and Hendry, 2016). Existing inventory data for mined commodities are highly spatially aggregated (i.e. global, continental or national boundaries – often with poor production coverage within the region). We recommend that future mine site water use inventory data be developed and reported at the scale of individual operations. If aggregation of site inventory data is required to protect commercially sensitive data, then aggregation to the scale of watersheds rather than national boundaries would facilitate more accurate assessment of water use impacts. Where this is not possible, then inventory developers could also develop weighted average impact factors that reflect the underlying spatial and temporal distribution of the inventory's water use data.

5.5. Conclusions

Life cycle assessment and water footprint studies of the mining industry are increasingly utilising spatially explicit characterisation factors when assessing consumptive water use impacts (Northey et al., 2016). This study has shown that the use of existing national average impact factors may lead to a bias to overestimate the consumptive water use impacts of the mining industry. . Despite this observed biased at an industry wide scale, there is high variability in results for commodity production in specific countries and so for some regions national average factors may actually underestimate impacts for a particular commodity. Due to these disparities, it is encouraged that future assessments of the mining industries consumptive water use utilise watershed specific inventory data and impact factors. In the absence of watershed specific water use inventories for the mining industry, weighting watershed factors based upon the spatial distribution of commodity production may improve the estimation of the industry's relative water use impacts. Overall, there are significant opportunities for continued development of life cycle inventory datasets and impact characterisation procedures to improve the assessment of the mining industry's water use impacts.

6. Intersection of Water Resources and Mineral Resource Development

The local context surrounding a mineral resource influences the development of a mining project. From the social and economic settings to infrastructure development and, of relevance to this thesis, water use contexts. An understanding of the importance of these factors to the development of mining projects is not just a recent phenomenon, but rather they have been recognised for centuries as outlined in the 16th century by Georgius Agricola in his seminal book *De Re Metallica*[2]:

"Now a miner, before he begins to mine the veins, must consider seven things, namely: the situation, the conditions, the water, the roads, the climate, the right of ownership, and the neighbours."

De Re Metallica (Agricola, 1556, Book II. 30)

In this chapter we explore how local water contexts can influence the development of mineral resource projects through a variety of ways. A mineral resource project will interact with water resources at all stages of the development's life: from exploration through construction, operation, closure and rehabilitation of the mining operation. Mining projects require a stable supply of water to meet a range of changing on-site demands. Therefore, securing stable supplies of water is of utmost importance for the development of mineral projects. However, water itself can also be a significant risk factor for operations, particularly in regions that display significant hydrologic variability that must be managed to avoid shortfalls or excesses of water on-site. Compounding this is the potential impacts of mining on surface water catchments, groundwater systems and water quality.

Closely related to these water issues are the broader and complex range of social and economic impacts associated with the development of mineral resource projects, which heavily influence the perspective and actions of industry stakeholders (Bebbington et al., 2008). From an economic perspective, mining operations provide the raw materials necessary to sustain a large proportion of global economic activity. However, the distribution of this wealth creation may not always be viewed as fair by all supply chain participants or stakeholders. Concern also exits that regional economies may become over-dependent on mineral resource extraction, to their long-term economic detriment.

[2] *De Re Metallica* (Agricola, 1556) provides a detailed overview of the state of geological understanding, mining and ore processing techniques, and the broader context of the mining industry in Europe in the 16th Century. When viewed in context, the book highlights how many of the basic principles and practices of the mining industry have remained unchanged for centuries. The observations and conceptual theories of geology and metallurgical properties presented are remarkably astute given – what modern society perceives as – the significant advances in natural philosophy, scientific investigation and engineering practices that has occurred since this time. Unfortunately, Georgius Agricola never lived to see the full influence of his work as the book was published posthumously one year after his passing due to delays in the publishing process. The book was translated from Latin into English in the early 20th Century by Herbert C. Hoover, a civil and mining engineer who later went on to become the 31st President of the United States, and his wife Lou H. Hoover. The translation itself can be viewed as a significant work of scholar, as it meticulously details the historical context, prevailing natural philosophy and influences that informed the writing of *De Re Metallica*.

6.1. Sourcing Water

"The miner should next consider the locality, as to whether it has a perpetual supply of running water, or whether it is always devoid of water except when a torrent supplied by rains flows down from the summits of the mountains. The place that Nature has provided with a river or stream can be made serviceable for many things […] Yet on the other hand, to convey a constant supply of water by artificial means to mines where Nature has denied it access, or to convey the ore to the stream, increases the expense greatly, in proportion to the distance the mines are away from the river."

De Re Metallica (Agricola, 1556, Book II. 31-32)

Mining and mineral processing operations will typically require water continuously through the life of the mine to meet the various needs of site processes. Therefore, these operations require a stable supply of water over-time and across seasons to prevent interruptions to production and also to achieve other site objectives, such as dust suppression activities to comply with environmental permitting requirements. The water sourcing requirements may vary through the life of a mineral project. During geological exploration, typically only minor volumes of water are required for use by site personnel for drinking or hygiene, and some other water may be required for dust suppression or to support drilling operations. Generally once the operation transitions to the active mining phase then substantially more water withdrawals are required due to aquifer dewatering, ore processing, as well other uses. Elevated water consumption associated with the project can also extend beyond the life of the mine into closure, rehabilitation and then perpetuity due to the need for site revegetation and the alterations to local hydrology that may be caused by changed site topography, tailings storage facilities and the formation of pit lakes (Eary and Watson, 2009).

Depending upon the source of water – such as shallow aquifers, river systems, lakes, or pipeline water – there may be differing restrictions or regulations placed upon the operation in terms of allowable withdrawals of water through the year and any prices paid. These factors are heavily influenced by local regulatory environments, and the presence of established water markets can also result in negotiation with other water rights holders being required. When establishing allowable extraction limits for mining operations, consideration often has to be given to the maintenance off environmental flows that are required to protect sensitive ecosystems downstream of the mine site, as well as the needs of other stakeholders such as the agricultural industry or local communities (Barret, 2009). Given that an individual mine is a transient operation, there may be tension between the short-term water requirements of the mining operation and the long-term security of water resources for communities and industries.

More advanced regulatory processes for the granting of water rights will often require detailed baseline monitoring and modelling of local hydrology. The extent and accuracy of these assessments is heavily dictated by the extent of available information, such as long-term rainfall gauging in the region. Certain aspects of this process, such as the development of hydrogeological models to understand groundwater flows, can be highly uncertain due to limitations in the conceptual models and data available for the system. These uncertainties can lead to substantial difficulty for attempts to understand how the local hydrogeological system will respond to groundwater and surface water extractions – and this can represent a barrier to a mining operation securing the right to withdraw water from local rivers, lakes or aquifers.

The regulations that dictate the ability of mining operations to source water for their mines may vary substantially across jurisdictions. The outcomes of processes to allocate water resources to mining operations can be highly contentious, and this can raise concerns with the rigour with which regulatory schemes are applied to mining developments (Currell et al., 2017). Examples exist of water resource allocation decisions being challenged through court processes due to insufficient baseline studies, or improper consideration of potentially affected communities or environmental assets. As a result there is increasing industry awareness of the potential difficulties associated with water sourcing and so concerted efforts are now often being made to proactively address these issues early in the development cycle of mineral resource projects.

There are a variety of water sourcing options available to support mineral resource projects. When the local climate, site topography and infrastructure design allows it then water can be sourced on-site through the capture and interception of rainfall or surface runoff. Alternatively, projects may be able to acquire the rights to extract water from local rivers, lakes or groundwater aquifers. However, in cases where local water supplies are not able to be secured then a mining operation may have to resort to transporting water long distances. Commonly this may be done through the use of tankers (typically only for exploration or smaller scale projects), or alternatively through the development of long-distance pipelines. Depending upon the length and altitude differences, pipeline projects can be costly ways of sourcing water – particularly when combined with technologies such as desalination. However, in some regions such as Chile, the industry is increasingly turning to desalination and pipeline projects in order to meet the water demands of the expanding mining industry. Capital expenditure requirements of water supply systems for mines in Chile are regularly in the hundreds of millions of dollars, equating to roughly 5 to 30 USD per m^3 of annual capacity, with further ongoing operating costs of 1 to 5 USD per m^3 (Soruco and Philippe, 2012). For many of the projects in Chile, the cost of water pipeline and conveyance systems exceeds the cost of the associated seawater desalination processes – both on a capital and an operating cost basis – due to the long-pipeline lengths and the need to pump to high altitudes of several thousand metres above sea-level. The energy requirements of these long-distance pipeline projects can be substantial, and can approach levels that are of similar magnitude to other important life cycle inventory items that contribute to the embodied energy and carbon footprint of mined products (Ihle, 2014). Although the desalination component of this is partially offset through the increased use of seawater directly in the Chilean industry, which is expected to grow further in the coming decade (Cisternas and Gálvez, 2017; COCHILCO, 2015) – although there are processing constraints that may limit this.

6.2. Water as an Operational Risk Factor

"Some of these evils, as well as certain other things, are the reason why pits are occasionally abandoned […] The second cause is the quantity of water which flows in; sometimes the miners can neither divert this water into the tunnels, since tunnels cannot be driven so far into the mountains, or they cannot draw it out with machines because the shafts are too deep ; or if they could draw it out with machines, they do not use them, the reason undoubtedly being that the expenditure is greater than the profits of a moderately poor vein."

De Re Metallica (Agricola, 1556, Book VI. 217)

Mining projects can be exposed to a range of water related risks throughout their operation that can affect investment returns through the interruption of production, changes to operating costs, or through alterations to the timing and/or magnitude of capital investments.

Historically, a major operational risk was the flooding of underground mines as a result of infiltration of groundwater. Although mechanical systems have been available to haul water out of mines for centuries, these were costly and troublesome endeavours (Agricola, 1556). By contrast, modern mining and pumping equipment are much more effective at removing water from mine workings and so these considerations are typically no longer viewed as a limiting factor on mine development. However, some water and climate contexts present inherent risks for the development of modern mining operations.

For instance, large intra-annual (seasonal) or inter-annual (year-to-year) hydrologic variability in a region can require careful management by mining operations. Therefore in some regions, mines may have to balance between having too much water during wet periods, whereas they may run the risk of running out of water during dry periods. Adapting to this variability may require the combination of infrastructure, operation and management responses (Kunz and Moran, 2016). Management of hydrologic variability in the mining industry requires careful consideration of the competing needs and objectives of individual mining operations. A key objective of mining operations is the maintenance of a stable supply of water that is necessary to ensure that the requirements of the mining operation – such as ore processing, wash-down of equipment, and dust suppression – are able to be met. This requires sufficient storages of water to be maintained in case of drought conditions, an objective that needs to be balanced against the desire to have excess dam capacity available to provide buffering capacity in case of drought conditions.

Excess water in tailings and water storage dams can pose an operational risk for mining operations, particularly for those mine sites that have restrictions in their capacity to discharge water for either practical or environmental reasons. The response of mine site management may be considerably different depending upon whether the accumulation of water is gradual due to an average site water balance that is positive, or if dam capacity is filled suddenly due to extreme weather events. When water is building up gradually on-site, the mining operation has greater flexibility in managing this accumulation of water. Options may include diverting excess water to evaporation ponds, increasing water storage capacity through the raising of tailings dam walls or constructing new dams/ponds, pumping water to unused sections of the mine workings, or discharging water to rivers, lakes or the ocean in accordance with the mines environmental permitting and statutory discharge limits.

Mine site management responses may be considerably different when a mine site is inundated with water as a result of short-term extreme weather events or flooding. The consequences of these extreme weather events may include: mine workings becoming inaccessible due to flooding, slope instability and the collapse of rock faces, flood damage to site infrastructure and equipment, roads being damaged and in extreme cases the failure or collapse of dams and waste containment facilities. Managing and mitigating these risks and hazards may require the mining operations to take on additional short-term costs, such as the installation of additional pumping, construction of flood barriers and diversion channels or the breaching discharge licenses, in order to avoid the risks of severe or catastrophic consequences to infrastructure and the higher long-term costs associated with this.

Regional approaches can also potentially be taken to mitigate these risks, such as the development of pipeline projects to and/or between mining operations so that water can be traded or shared, so that less water is required to be stored on-site, thereby reducing excess accumulation of water on-site and the potential risks to infrastructure or of discharges in breaches of environmental permitting or licenses (Gao et al., 2016).

Flooding can be a perpetual seasonal risk, such as flood risk due to inundation due with summer snow or glacial melts, or alternatively due to extreme rainfall in wet or monsoonal seasons. Many examples exist of extreme weather impacting mining operations through flooding or the failing of infrastructure. Some examples include:

- Flooding of the Yallourn coal mine due to the collapse of an embankment along the Morwell River in Victoria, Australia (Mason et al., 2013).

- Flooding of coal mines in the Bowen Basin due to extreme weather and flood events that significantly impacted regions in Queensland, Australia (Sharma and Franks, 2013).

There are also examples reduced rainfall or drought conditions impacting mining operations, such as:

- The Ok Tedi mining operation in Papua New Guinea being impacted by drought events that reduced water levels in the Fly River system, thereby making the operation inaccessible by barges resulting in interruptions to production.

- Cadia Valley Operations in New South Wales, Australia being impact by drought conditions that threatened water supply for ore processing and required negotiation with the local council to obtain a 5 ML/day temporal withdrawal permit (Newcrest, 2007).

- Extremely low rainfall at the Lihir gold mine in Papua New Guinea that reduced the freshwater available for the processing circuits, resulting in 40,000 oz Au lower production for the period (Newcrest, 2011a).

When drought conditions occur, mining operations may be constrained depending upon whether they can utilise poorer quality water sources, such as hypersaline groundwater aquifers. Resilience in these instances is therefore dictated by the availability of on-site water storage capacity to provide a buffer against hydrologic variability. This buffering capacity is also useful for the mitigation of potential flood events (Gao et al., 2017).

6.3. Water and the Social License to Operate

For a mining company to develop an identified mineral resource into an active mining operation will typically require the acceptance and support of local governments, communities and stakeholders. If this support is absent then the mining company may face great difficulty in progressing the project through the various development stages, from exploration through mining, construction operation and then final closure and rehabilitation. This support is now commonly referred to as a mine's 'social license to operate' – a phrase that emerged in the 1990s and has been attributed to Jim Cooney, a Canadian Mining executive (Prno, 2013). However, despite the recent emergence of this terminology to describe social concerns and the level of opposition or support for mining projects, debates surrounding the development of mining projects are not a new phenomenon. Once again, this is evidenced by an excerpt from *De Re Metallica* (Agricola, 1556), which formed part of a longer discussion on the prevailing perceptions of the benefits and detriments of mining – although it should be noted that in contrast to this excerpt, Georgius Agricola's final conclusion was that the negative impacts of mining are outweighed by the overall benefits to society:

"Further, when the ores are washed, the water which has been used poisons the brooks and streams, and either destroys the fish or drives them away. Therefore the inhabitants of these regions, on account of the devastation of their fields, woods, groves, brooks and rivers, find great difficulty in procuring the necessaries of life, and by reason of the destruction of the timber they are forced to greater expense in erecting buildings. Thus it is said, it is clear to all that there is greater detriment from mining than the value of the metals which the mining produces."

De Re Metallica (Agricola, 1556, Book I. 8)

An assessment by Davis and Franks (2014) discovered that management of social license to operate issues can results in significant costs to mining companies. These costs can manifest in several ways, including: lost productivity due to shutdowns or delays, lost opportunity costs due to prevention of future expansions, projects or sales; as well as the often hidden cost associated with time required by staff – often senior management – to manage social issues, or through the loss of reputation that may affect the company's ability to recruit the best talent.

Many social license issues are related to the potential impacts of mining operations on water resources (Bebbington and Williams, 2008; Bebbington et al., 2009; Kemp et al., 2009; Holley and Mitcham, 2016; Wessman et al., 2014). In some contexts, the presence of different water user groups can lead to social conflict when the potential impacts of water consumption or quality degradation are not managed, or expected to be managed, appropriately. There has been limited formalisation through international treaties and human rights declarations of whether access to drinking water is an inalienable right for humans, although it is an obvious necessity that is often assumed to be an implied human right. Due to this lack of formalisation, different stakeholders may have opposing opinions on the roles and responsibilities of mining companies when it comes to maintaining or providing access to safe drinking water for communities. These issues can become particularly acute in regions with poor natural resource governance, limited economic development, or that lack appropriate water supply, storage and treatment infrastructure.

To facilitate better engagement with local stakeholders, mining companies will often invest in community and infrastructure development projects to ensure that the local communities benefit from the mining operation. The types of community development project that companies invest in can vary considerably depending upon the local context and should, in best practice cases, be developed through effective engagement with the community to understand and address their concerns and needs. In regions with shortages or poor access to high quality water, mining companies often work with local communities, businesses and governments to implement water treatment and supply projects. Some examples of mining company investments specifically in water related community development projects are provided in Table 3.

Table 3: Examples of mining company investment in water related community development and assistance.

Description	Value
Newcrest's Response to the 2010/2011 Queensland Floods (Newcrest, 2011b)	
Donation to Queensland Premiers Disaster Relief Appeal.	AU$150,000
Donation to Australian Red Cross Victorian Floods Appeal.	AU$100,000
Installation of a potable water treatment plant to assist community.	AU$250,000
Hidden Valley Project, Papua New Guinea (Newcrest, 2011b)	
Water supply projects in 20 local communities to provide safe drinking water for over 5000 people.	PGK 1,800,000 (AUD $704,415)
Sepon Mine, Laos (MMG, 2011)	
Installation of water filtration, supply and tap systems for 12 local villages.	US$800,000
Contribution to UN Habitat's urban water supply project.	US$250,000

Although investment in community projects could now be considered a standard part of mining industry attempts to gain and maintain a social license to operate, there can be a risk of community development project's being viewed cynically by local stakeholders if they feel their concerns or requirements are not being met. When the investment and benefits being provided by the company are not perceived to be directed in an equitable way across the community then this can lead to resentment and hostility towards the mining company. Therefore, investment in community projects by itself does not offset the need to address – or at least be perceived to address – the potential negative impacts of the mining project (Martinez and Franks, 2014).

It has been proposed that 'trust' amongst stakeholders is a major factor in determining whether a project will be able to develop and maintain a social license to operate (Moffat and Zhang, 2014). Trust amongst stakeholders, information providers and the overall engagement processes may be more important for maintaining a social license than the magnitude of expected or experienced impacts to individuals or the community. Therefore, a company can increase their chances of obtaining a social license to operate through proactive engagement with communities early in the project life cycle, through clear communication of the various government regulatory processes relevant to the project and the company's commitment to community engagement processes (Zhang et al., 2018).

6.4. Providing context to long-term mineral resource development and scarcity research

There is substantial interest in the sustainable development oriented research community on the evolution of mineral supply chains and whether there may be growing economic scarcity of mineral resources or the potential for depletion of these resources over the long-term (Ali et al., 2017). The majority of these studies take a relatively top-down approach, using national or global resource datasets for the magnitude of identified mineral reserves or resources. By comparison, few studies consider the factors surrounding individual mineral deposits and the full-dynamics of how resource exploration, mine development and mineral processing will evolve in relation to local economic, social or environmental conditions. Due to this, local contexts that may influence the development of mineral resource projects, such as water and climate conditions, are typically not accounted for in any way within these broader studies that look at the industry as a whole.

Improving the quality of studies that take a forward looking perspective of the minerals industry requires the development of a more in-depth understanding of the basic principles of economic geology, how local contexts relate to the progression of resource projects and also the potential technology development that may occur over time. As part of this, the studies that constitute this thesis can potentially be used to support discussions and analysis of how long-term mineral resource development relate to local water and climate contexts. The spatial analysis of hydrological contexts that was presented previously in Chapters 4 and 5 of this thesis provides an example for how other 'risk' factors surrounding the industry – such as governance, social license, regulatory or economic conditions – may be used assessed using spatial indices.

The contents of this sub-chapter were published in Natural Resources Research in a special issue on 'Resourcing Future Generations' and are presented with in the original format of the journal. The article extends upon the content of an invited presentation at the 35th International Geological Congress in Cape Town, South Africa in August 2016 (Northey and Mudd, 2016).

Reference:

Northey, S.A., Mudd, G.M., Werner, T.T. (2017). Unresolved Complexity in Assessments of Mineral Resource Depletion and Availability. Natural Resources Research, volume unassigned, 15p. http://dx.doi.org/10.1007/s11053-017-9352-5

Natural Resources Research (© 2017)
DOI: 10.1007/s11053-017-9352-5

Review Paper

Unresolved Complexity in Assessments of Mineral Resource Depletion and Availability

Stephen A. Northey,[1,3] Gavin M. Mudd,[2] and T. T. Werner[1]

Received 3 February 2017; accepted 21 July 2017

Considerations of mineral resource availability and depletion form part of a diverse array of sustainable development-oriented studies, across domains such as resource criticality, life cycle assessment and material flow analysis. Given the multidisciplinary nature of these studies, it is important that a common understanding of the complexity and nuances of mineral supply chains be developed. In this paper, we provide a brief overview of these assessment approaches and expand on several areas that are conceptually difficult to account for in these studies. These include the dynamic nature of relationships between reserves, resources, cut-off grades and ore grades; the ability to account for local economic, social and environmental factors when performing global assessments; and the role that technology improvements play in increasing the availability of economically extractable mineral resources. Advancing knowledge in these areas may further enhance the sophistication and interpretation of studies that assess mineral resource depletion or availability.

KEY WORDS: Resource availability, Mineral resource depletion, Ore grades, Life cycle assessment, Criticality assessment, Material flow analysis.

INTRODUCTION

Concerns over the depletion of natural resources have been an ever-present part of the modern sustainability dialogue. The nature of mineral and metallic resource depletion differs from the depletion of other resources such as food or energy, in that metals are non-renewable resources that are generally not physically consumed upon use, but rather tend to remain in society and are often available for longer-term reuse and recycling (albeit dissipative end uses do exist). Resource depletion can be conceptualized based upon two competing viewpoints—the fixed stock paradigm and the opportunity cost paradigm. The fixed stock paradigm is based on the principle that only a finite geologic resource is available to meet the long-term demands of society, whereas the opportunity cost paradigm suggests that there is no fixed limit to the geologic resource available for use, as once cheaper to develop resources are depleted then any unmet demand will result in an increase in market prices, thereby making marginally higher-cost resources still economic to develop and extract. There has been healthy debate among the proponents of each of these perspectives (see Gordon et al. 2006; Tilton and Lagos 2007). The combination of these perspectives recognizes that while there exists a fixed geologic resource present in Earth's upper crust, the fraction of this geologic resource that is available for use by society is ultimately a function of technology, economic relationships and ever-changing market

[1]Faculty of Engineering, Monash University, Clayton, VIC, Australia.
[2]Environmental Engineering Group, RMIT, Melbourne, VIC, Australia.
[3]To whom correspondence should be addressed; e-mail: stephen.northey@monash.edu; stephen.northey@gmail.com

Published online: 27 July 2017

conditions. Due to this, there is large uncertainty present in estimates of future mineral resource availability (Crowson 2011; Graedel 2017). With substantially increasing demand for minerals as a result of population growth and economic development, there exists some concern that we will quickly exhaust high-quality mineral deposits leading to progressively poorer-quality mineral resources being available to meet sustainable development goals and the needs of future generations (Harmsen et al. 2013; Elshkaki et al. 2016; Ali et al. 2017; Nickless 2017). General mining industry trends such as increasing overburden and waste rock ratios, and declining mined ore grades provide support to this notion (Mudd 2010; Crowson 2012). However, it is important to recognize that these trends are not strictly indicators of mineral resource availability or depletion per se, but rather they are also a function of evolving market conditions and technology development (West 2011).

Our understanding of mineral resource availability and depletion issues is therefore intimately linked with questions pertaining to society's long-term mineral demand, the economic relationships within mineral supply chains, and the nature and distribution of geologic resources. Due to this, discussions of mineral resource availability and depletion occur across a broad range of disciplines, albeit with varying levels of sophistication. Interpreting the results of these studies requires a nuanced understanding of the various feedbacks, relationships and data sources available for understanding mineral resources and the mining industry.

This article provides a brief review of how mineral resources were evaluated as part of a major Australian research effort and also within life cycle assessment, material flow analysis, and resource criticality assessments. From this, several areas of conceptual or practical complexity that hinder these assessments are discussed, with an aim to advance dialogue and develop a common understanding of these issues.

BACKGROUND AND BRIEF OVERVIEW OF LCA, MFA, AND RESOURCE CRITICALITY

A variety of disciplines and assessment methods seek to facilitate an understanding of the nature of mineral resource availability or depletion and how this may impact future society.

Australian Mineral Futures Collaboration Cluster

A recent major initiative was the Australian Mineral Futures Collaboration Cluster research programme, a collaborative effort between six Australian universities and the CSIRO that ran from 2009 to 2013. As a part of this, a series of studies that explored the concept of ''peak minerals'' were completed to better understand the nature of how mineral extraction and resource depletion can affect regions and society (Mason et al. 2011; Giurco and Cooper 2012; Giurco et al. 2012a, b; May et al. 2012; Mohr et al. 2012, 2015; Prior et al. 2012; Northey et al. 2014a). A variety of approaches were taken to address the nature of resource depletion and peak minerals, ranging from the use of ''Hubbert curves'' through to qualitative and social assessment, to more sophisticated modeling approaches. As part of this, detailed data on identified mineral resources were compiled for a range of commodities, including copper (Mudd et al. 2013a), nickel (Mudd and Jowitt 2014), and cobalt (Mudd et al. 2013b). A major outcome of the research programme was quantitative studies to assess mineral resource supply and depletion that were undertaken using the Geologic Resources Supply–Demand Model (GeRS-DeMo) developed by Mohr (2010). Scenarios were developed using GeRS-DeMo to explore aspects of the future supply of iron ore (Mohr et al. 2015), lithium (Mohr et al. 2012), gold (Mudd and Mohr 2010), and copper (Northey et al. 2014a). Some related work has also been conducted using GeRS-DeMo to assist the strategic planning of regional governments. For instance, Mudd and Mohr (2010) undertook an assessment of future production and depletion of gold, nickel, copper, lead, and zinc in the Goldfields Esperance Development Commission region of Western Australia.

As an example of these research outputs, Figure 1 shows the key model and data components of the global assessment of primary copper supply, which also attempted to provide insight on potential rates of copper ore grade decline (Northey et al. 2014a, b). The aim of the study was to develop a scenario for long-term copper supply so that the potential magnitude of annual copper supply that could be achieved based upon currently known mineral deposits and the potential for long-term ore grade decline could be better understood. The study has generated renewed discussion in these areas, such as Kerr (2014)'s exploratory article on long-term copper availability. In recent years, other au-

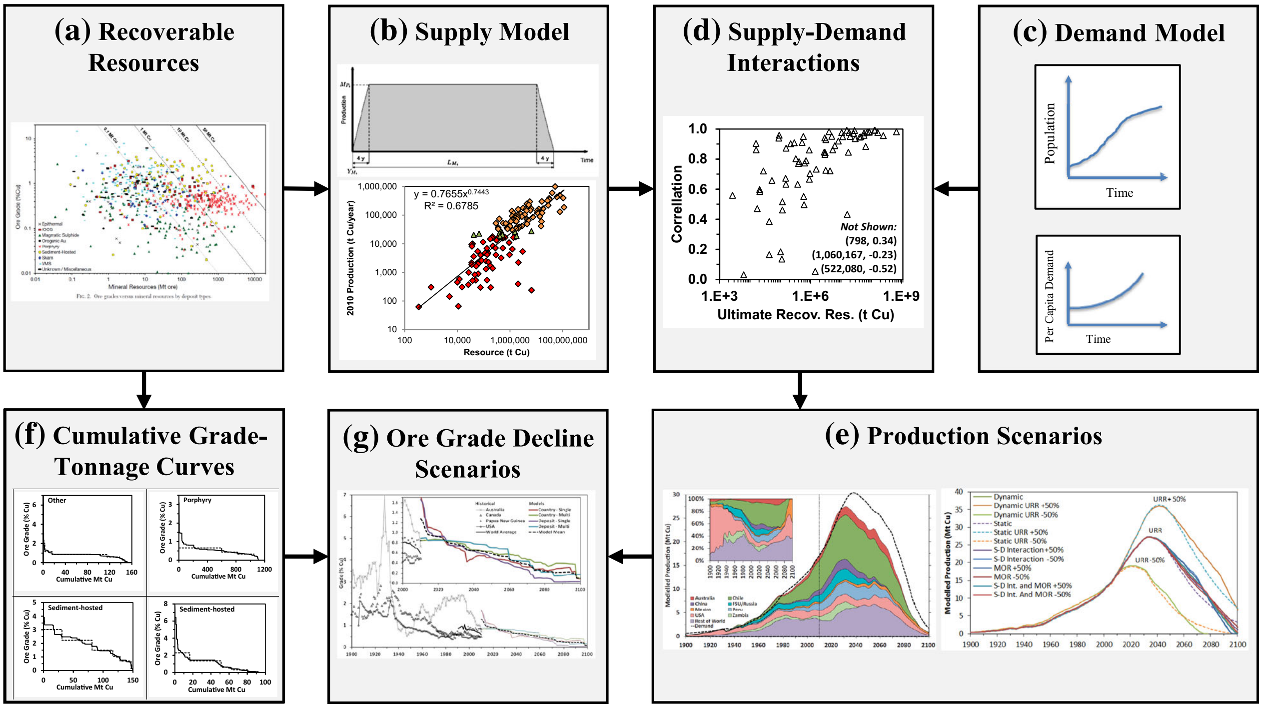

Figure 1. Key model elements, data, and linkages underpinning the primary global copper supply scenario model developed by Northey et al. (2014a).

thors have also published detailed assessments of available copper resources, supply, and demand (Sverdrup et al. 2014; Elshkaki et al. 2016; Meinert et al. 2016; Arndt et al. 2017; Singer 2017). A general conclusion of these studies is that there is still capacity for growth in primary copper supply; however, resource quality may decline over time, requiring further exploration and increases in secondary production (i.e., recycling) to ensure supply is maintained long term.

Resource Criticality Assessments

Resource criticality assessments attempt to develop our understanding of the risks associated with resource supply chains to economies and regions. These assessments often broadly address aspects of supply security, resource availability, geopolitics, environmental considerations, and regional vulnerability or adaptability to supply disruptions (Graedel et al. 2012; Graedel and Reck 2015). From this the relative "criticality" of individual mineral commodities can be assessed to determine the supply chains that warrant additional attention from government and corporate actors. Assessments of mineral and metal resource criticality now cover the majority of the periodic table (Nassar et al. 2012, 2015a, b; EU Commission 2014; Nuss et al. 2014; Harper et al. 2015; Panousi et al. 2016). There are also examples of the criticality methodology being applied to the assessment of other resource categories such as water (Sonderegger et al. 2015).

A variety of metals, such as the rare earth elements, are now commonly considered critical (Nassar et al. 2015a). The vast majority of both supply and demand occurs internally within China, and so some concern exists within foreign companies and nations regarding their ability to maintain stable access to rare earth products. These concerns are somewhat warranted given the sensitivity of the international rare earth market to the trade regulations and policies of China—and this was highlighted by the 2010 rare earth crisis that resulted in a sharp, albeit temporary, spike in foreign rare earth prices and a dispute at the World Trade Organisation (Biedermann 2014; Sprecher et al. 2015). However, the size of the known geologic rare earth resource is large relative to current levels of demand (Weng et al. 2015) and so additional foresight within Western economies may have prevented the extreme concentration of the rare earth market in China (Tukker 2014). From this it is important to recognize that the perceived criticality of resources, such as rare earths, can shift through time due to the evolution of economies and supply chains. Therefore, the time horizon considered by a study may influence our understanding of the relative criticality of different resources (Nassar et al. 2015a).

A critique of criticality assessments is that they have represented a single snapshot in time and have failed to reflect the dynamic nature of supply and demand. For example, indium resources in Pb–Zn deposits in Australia could be substantial (see Werner et al. 2017); however, given that Australia currently has no refining capacity for indium, it is considered to have a high import dependence and thus at greater risk of a supply disruption. More recent ‘‘dynamic criticality’’ assessments (e.g., Ciacci et al. 2016 and references cited therein), while permitting the assessment of criticality across multiple years, have not yet determined a way to look at the adaptive capacity of a country's supply chain to respond to supply disruptions or an altered security of resource supply. Indeed, the criticality assessments presented in Ciacci et al. (2016) show indium to rate as highly critical, even though it relied on the results of an MFA study of indium in Australia which showed indium to be quite abundant in Australia's Pb–Zn deposits. Australia might be considered more adaptable than other countries that are also dependent for imports of refined indium but do not possess the same resource endowment; however, this is not captured in great detail at present. More light is likely to be shed in studies looking at supply chain dynamics, to see how different actors respond in the event of a supply disruption (see Sprecher et al. 2015). Other criticisms of criticality studies also exist, such as a current lack of theoretical grounding in classical risk theory (Frenzel et al. 2017).

Material Flow Analysis (“Stocks and Flows”)

Material flow analysis (MFA) seeks to quantify the movement, or flows, of materials through their life cycles. Along various stages of these life cycles, materials may spend significant periods as a stock, unmoving for many years. Stocks are viewed as reservoirs of materials that exist within the economy, which might be recoverable in the future. Examples of stocks might be the quantities of unrecovered material in landfills or still in use within buildings. Often calculations of stocks and flows are viewed as a way of measuring the material efficiency of a system. In doing so, new policies and procedures around material usage can be developed (Graedel et al. 2015; Clift and Druckman 2016).

Studies of material flows may be distinguished as those that examine a single-year snapshot of metal within a given system boundary (i.e., static MFAs), or those that look across multiple years (dynamic MFAs). Given that resource depletion must be studied across a specific timescale, the dynamic MFAs are of most interest here. Among the dynamic studies conducted (see Müller et al. 2014), it is common to see actual mining activity documented as flows from the lithosphere; however, often the remaining mineral resource remains unspecified or unchanged. In this sense, MFA may be a useful tool in measuring resource depletion as a change in stock over time, but thus far this opportunity has been largely missed. It is more common for MFA studies to cite an estimate of present-day mineral resources and to project future demand against this value as an indicator of future scarcity (e.g., Elshkaki and Graedel 2013), rather than to model economic resource availability as a dynamic value itself. This may be because a number of metals that have been the subject of MFA studies have not been the subject of detailed bottom-up global resource assessments, whereas these are increasingly being developed for major metals such as copper (e.g., Mudd et al. 2013a). In addition, relatively few MFA studies have attempted to model long-term mineral supply using bottom-up approaches to model the exploitation of individual deposits. This is possible, however, as deposit and national-scale production data are available in some countries to indicate the amount of metal that has been mined/milled on an annual basis. If recent mineral resource estimates are available alongside metal deportment data, it is possible to balance the amount milled over time such that the present-day resource estimates are achieved. This permits historical resource depletion to be modeled in good detail using MFA methodologies, which may at least indicate trends for future resource depletion, as well as the accumulation of valuable metals in tailings. At the deposit level, this is shown in Figure 2 (Werner et al. 2015). This indication of resource depletion disaggregated into the deposit or country scales could

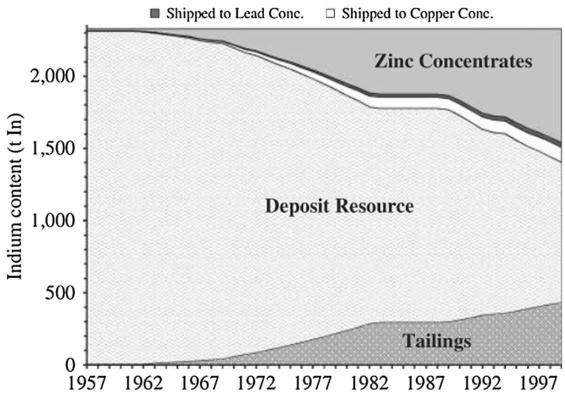

Figure 2. Historical material balance of indium resources and mining for the Heath Steele deposit, Brunswick, Canada, from 1957 to 1999. Figure reproduced from Werner et al. (2015).

further reveal more about the relative depletion of resources between locations and hence inform future ratings of metal criticality.

Life Cycle Assessment

Life cycle assessment (LCA) studies provide an evaluation of the environmental impacts associated with the provision, use, and disposal of products or services. Life cycle impact characterization procedures are used to relate inventory flows of materials, energy, natural resources, and emissions to defined impacts on the environment or society (e.g., contributions to global warming, ozone depletion, or freshwater eutrophication).

Often in LCA, impact assessment methods will attempt to measure damage to three so-called areas of protection—ecosystems, human health, and natural resources. The LCA community has struggle to conceptualize what should be measured by the natural resources area of protection, and there have been calls for further subdivision of this category due to the varied roles of resources in society and ecosystems (Steen 2006; Klinglmair et al. 2014; Dewulf et al. 2015; Sonderegger et al. 2017). Dewulf et al. (2015) identified a range of perspectives of what could be protected by the natural resource category, which were:

- Natural resources as an asset for future society;
- The provisioning capacity of natural resources;
- The role of natural resources in global functions;
- The role of natural resources to support supply chains; and
- The role of natural resources to provide for human welfare.

Due to these varied perspectives, the existing methods and approaches to quantifying the impacts associated with mineral resource depletion take a variety of approaches (Yellishetty et al. 2009; Klinglmair et al. 2014; Swart and Dewulf 2013; Drielsma et al. 2016; Sonderegger et al. 2017). Some of the approaches to assessing impacts associated with the consumption of mineral resources include:

- Measuring the relative decline in known mineral reserves, resources (Schneider et al. 2011), or crustal abundance (van Oers et al. 2002; van Oers and Guinée 2016);
- Measuring the relative decline in combined mineral resource and anthropogenic stocks (Schneider et al. 2011);
- Measuring the marginal decline of ore grades, assuming grade–tonnage relationships and that higher-grade resources are preferentially mined (Vieira et al. 2012);
- Measuring rates of exergy depletion (i.e., depletion of available work) associated with extracting a mineral resource (Bösch et al. 2007) or exergy replacement costs (Valero and Valero 2012).

There have also been some attempts to incorporate aspects of criticality assessment into LCA. Combining these two types of assessment would enable combined assessment of both the environmental impacts of supply chains and also the potential geopolitical supply risks that are present. For instance, Gemechu et al. (2015) proposed a method for incorporating aspects of criticality analysis into LCA via accounting for the degree of import dependency of nations for different materials. Schneider et al. (2014) also proposed assessment of the "economic scarcity potential" associated with resource use, which constituted a range of impact categories related to the concentration of supply, the presence of trade barriers, resource availability (measured as depletion time; reserves divided by production), national governance and socioeconomic stability, demand growth, relative companion metal fractions in ore bodies, and the degree of recycling of the resource.

Given the uncertainty in defining and measuring what constitutes an impact of mineral resource use within LCA, the UNEP-SETAC Life Cycle Initiative recently initiated a taskforce to provide recommendations on how to address the natural resources area of protection within LCA.

MAJOR AREAS OF COMPLEXITY

Each of the assessment types discussed in the preceding sections attempts to measure different aspects of mineral supply chains. Interpreting and understanding the results of these studies is greatly aided by developing our understanding of the inherent complexity that is present in mineral supply chains. To begin to provide a common understanding, several of these complexities are explored in the following sections.

Accounting for the Dynamic Nature of Resources, Reserves, Grade, and Production

A common assumption of some LCA impact characterization methods (e.g., Vieira et al. 2012) and studies of future mineral supply (Harmsen et al. 2013; Northey et al. 2014a, b) is that there will be a continuation of declining mined ore grades, a trend that has been observed within the industry (Mudd 2010; Crowson 2012). However, the common narrative that the decline of average ore grades is due to depletion of high-grade resources is only telling one part of a more complex story (see West 2011), as mined ore grades are as much a reflection of technology and market conditions, as they are about the nature of the individual mineral deposits being mined. Given this, it is worthwhile to highlight the various forms of ore grade data that may be reported by the minerals industry and how these relate to reserve or resource definitions and mining.

The term ''ore grade'' can be used in a variety of contexts within the mining industry. Depending upon the context of use, the nuances and relationship between ''ore grade'' at various uses or scales deserve a more detailed examination to inform discussions of resource depletion. Resource cut-off grades are used to define the extent of a geologic resource above that grade (i.e., the quantity of mineralized rock of potential economic interest); resource ore grades are typically expressed as the proportion of contained metal per unit of mineralized ore for a given resource cut-off grade; reserve cut-off grades are used to define the extent of geologic resource that is currently considered economic to mine; run-of-mine (ROM) grades are the concentration of metals in the ore actually mined; milled grades are the concentration of metals in the ore that is actually processed; subsets of grades are ''recoverable grades'' that can also be defined at any stage along this process. In reality, recoverable grades are as much a function of economics, technology, and engineering relationships, as the underlying mineralogy of the ore being processed. When there are multiple metals of economic interest within an ore body, these may be reported as the equivalent grade of a single metal (e.g., gold contributing to a copper equivalent grade based upon relative economic value and assuming typical process and market conditions). As a result of these various definitions, typically:

- Cut-off grades will be lower than the associated ore grade;
- Resource grades will be lower than reserve grades;
- Recoverable grades will be lower than the ore grade; and
- ROM and milled grades will often be higher than reserve grades due to selective mining practices.

The relationship between grades at various steps along the resource definition and extraction value chain is not straightforward and is influenced by market conditions, geologic conditions, mining methods, and expected process conditions. Various approaches are taken to mine planning to maximize the economic value generated by a mining operation, and these need to consider a variety of short- and long-term trade-offs. The type of trade-offs that a mine site must make is varied. Production rates and the efficiencies of scale must be balanced against the expected mine life when making capital investment decisions. During periods of high commodity prices, a mineral processing operation may choose to process ore at a higher throughput to maximize short-term production, with the trade-off being lower recovery rates and a reduced production over the mine's lifetime (Yap et al. 2013). Alternatively, during periods of low commodity prices an operation may choose to selectively mine and process higher-grade ore to minimize potential losses or cash flow risk (Bao et al. 2013)—which can lead to the lower-grade ore becoming uneconomic to mine and process in the future (e.g., this occurred in the Rudny Altai field in Russia; Koslov 2015). Mines may also initially target higher-grade ore to quickly pay off capital financing costs (Crowson 2012). The consideration of these trade-offs may be assessed using traditional block model and cut-off grade approaches combined with detailed mine planning and financial models. More sophisticated methods such as real options analysis may also be adopted to better address uncertainty in markets, geology, or mine planning (Dimitrakopoulos and Sabour 2007; Azimi et al. 2013; Bao et al. 2013).

The relationship between cut-off grades and resource grades is particularly important to understand when making assessments of mineral resource availability. Figure 3 provides an example of this for the Malku Khota project in Bolivia (Armitage et al. 2011). The Malku Khota resource is subdivided into several distinct zones, each exhibiting different geology and distributions of potentially recoverable metals (namely silver, indium, gallium, copper, lead, and zinc). As cut-off grades decline the size of the resource increases, however the rate of this increase is not uniform for all sections of the mineral resource. The level of uncertainty associated with the resource estimate is also not uniform for the whole resource as denoted in Figure 3 by ''measured'', ''indicated'', and ''inferred''. Therefore, there is a great deal of financial and technical risk associated with defining the economic cut-off grades for profitable extraction of the target mineral or metal.

As cut-off grade selection of individual deposits is partially an economic decision, there are implications for studies that have attempted to derive global resource estimates based upon cumulative grade–tonnage curves. Typically cumulative grade–tonnage curves will be constructed for specific mineral deposit types using the cut-off grade that is (or would be) adopted for mining at each deposit (e.g., volcanic massive sulphide deposits; Mosier et al. 2009; porphyry copper deposits, Singer et al. 2008). In order to assess potential development scenarios, such as smaller scale with higher grades versus larger scale with lower grade (i.e., lower capital and operating costs with lower revenues versus higher capital and operating costs with higher revenues), a mineral deposit will be modeled at various cut-off grades to give tonnages and grades for these scenarios, which can then be used in detailed mine planning and financial assessments to optimize the economic return on a given project. Based on a company's preferred investment strategy, a particular development scenario will be chosen and this locks in the cut-off grade and effective resource size for mining. Given these trade-offs, it is very difficult to compare cut-off grades between projects and generalize due to the variations in costs, companies, deposits, project configurations, and so on.

In addition to the long-term trend of declining mined ore grades, there is also a long-term trend towards declining cut-off grades used for resource estimation (Schodde 2011). This is mainly a reflection of the greater economies of scales associated with larger projects (e.g., bulk mining of porphyry Cu projects), as well as the significant increase in deposit size as lower cut-off grades are applied and the technology used to extract the minerals or metals has changed (e.g., cheaper heap leach technology). This means that for many mining projects over time the total mineral resource has grown considerably simply due to changing economics, technology, and geology leading to lower cut-off grades—which is also often augmented by additional exploration increasing the zones or areas of mineralization (see examples in Jowitt et al. 2013). The response of long-term cut-off grade decisions to technology improvements or structural changes in supply–demand profiles in mineral markets is highly uncertain and may not be uniform between different classes and sizes of deposit. As the full cumulative grade–tonnage relationship of each individual deposit is generally not reflected in databases of mineral resources (typically only the lowest cut-off grade assumed for mining), we hypothesize that the overall ''shape'' of the estimated global cumulative grade–tonnage curves developed by studies (e.g., Gerst 2008) may be significantly altered were there to be widespread revisions of the mineral resource estimates of individual deposits due to revised cut-off grade assumptions. The widespread revision of cut-off grades could potentially occur as a result of major technology improvements, long-term structural changes to supply–demand balances, changes in the capital efficiency trade-offs involved in determining mine site production capacities, or a myriad of other long-term economic factors (e.g., substantially altered fuel and labour costs). The changes to aggregated cumulative grade–tonnage curves would alter the perceived size of the global resources as well as expectations regarding long-term ore grade decline. Additional uncertainty is

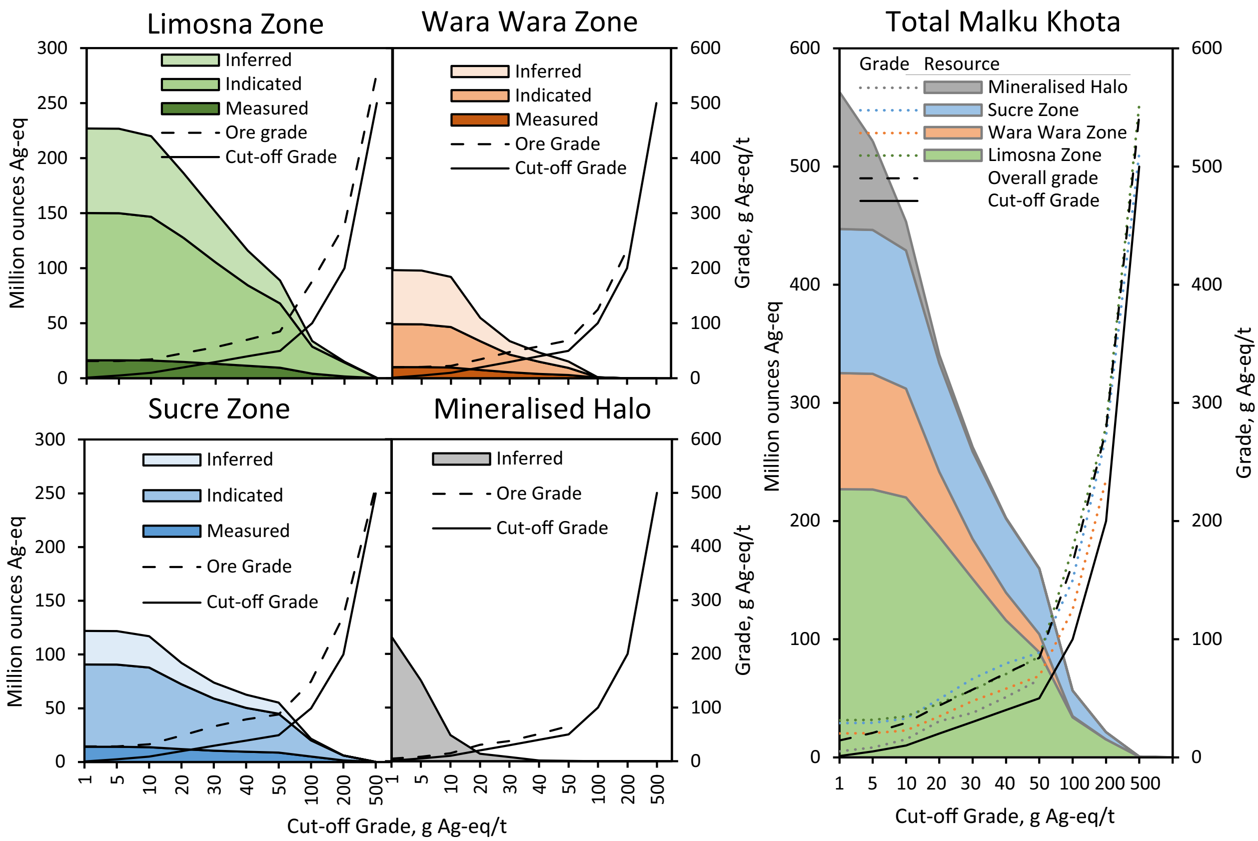

Figure 3. Influence of cut-off grade on the ore grade and contained resource size for the Malku Khota project in Bolivia (Armitage et al. 2011). Values are expressed in terms of silver equivalent (Ag-eq), which accounts for the additional contained value of indium, gallium, lead, copper, and zinc.

added to the global cumulative grade–tonnage curves for specific resource types due to uncertainty associated with the classification and segmentation of co-occurring mineral formation processes within deposits (see Jowitt et al. 2013).

Overall there are a range of complex economic and engineering considerations that determine our perception of resource size and the proportion of this that will ultimately be recovered, with many of these relationships manifesting through changes in mined ore grades. Sterilization of some resources because of mining at current economic grades is possible, whereas in other cases the decline in economic ore grades over time enables resources at historic or abandoned mine sites, or even tailings dumps, to be re-evaluated with potential increases in resource sizes. This can enable further mining of previously uneconomic ore or the profitable retreatment of historic waste and tailings material (Lèbre et al. 2017).

Resource Quality

In addition to ore grades, resource quality more generally is an important consideration when assessing long-term resource availability. Aspects of resource quality may include the depth of deposits, mineral grain complexity and size distributions (e.g., fine versus coarse), the presence of gangue minerals, penalty elements (e.g., arsenic deporting to a copper concentrate or interfering with milling and smelting/refining), ore hardness, or even the remoteness of the deposit. For some bulk commodities such as bauxite and iron ore, there may not be a substantial expectation of declining grades of the primary target metal. However, the quality of resources may still be perceived to be declining over time due to changes in mineralogy or impurity concentrations. In the case of iron ore, one manifestation of this may be increasing phosphorous, silica, and/or alumina content over time. Declining ore grades or resource quality typically results in the need to mine (often from greater depths) and process greater amounts of ore in order to produce the same quantity of mineral product, and so can result in significant increases in the energy, material, and water requirements associated with mineral production (Mudd 2010; Norgate and Jahanshahi 2010; Northey et al. 2013; Koppelaar and Koppelaar 2016).

Differences in mineralogy are particularly important to understand as it directly affects the approaches available for extracting valuable minerals or metals from the ore (La Brooy et al. 1994). Mineralogy is typically heterogeneous across individual ore deposits. As an example, porphyry copper systems often have oxidized "caps" that can have elevated grades compared to the underlying sulphide resources, due to supergene enrichment processes. Due to this the initial mining of these oxidized resources may be at elevated copper ore grades; however, as the oxidized material is mined past and the more sulphidic underlying porphyry system is reached, there may be a decline in copper grades (albeit with a typically much larger resource size). The shift from oxide to sulphide copper mineralogy will usually require differing approaches to ore processing and copper recovery, such as a transition from heap leaching to flotation processes. These structural changes in the processing requirements from porphyry copper deposits are reflected in scenarios developed by the Chilean Copper Commission for the industry's water and energy consumption, which indicate a potential decrease in the proportion of Chilean copper produced via hydrometallurgy over the next ten years (Comisión Chilena del Cobre 2014).

Accounting for Local Context

The development of mineral resource projects is impacted by a range of contextual social, economic, and environmental factors. Constraining factors may slow or prevent the development of a mineral resource project, whereas enabling factors may allow or accelerate the development of mineral projects. Therefore, understanding the local contexts surrounding mineral resources has implications for our broader understanding of mineral resource depletion or accessibility.

Mining operations have the potential to cause a significant degradation to land, water, and air environments. Due to this, most developed jurisdictions around the world require mineral resource projects to comply with a range of environmental regulations and processes before mining can commence. These permitting processes may place restrictions on how a mine site can be operated to minimize the risk of adverse environmental consequences (e.g., only allowing water discharges during specific periods of the year), and in some circumstances these requirements render an entire mineral resource inaccessible for mining. The types of issues that are commonly considered by these processes may include:

- Water sourcing, discharge, and quality issues;
- Dust and air pollution;
- Noise pollution;
- Large-scale mine waste management (i.e., tailings and waste rock);
- Acid and/or metalliferous drainage risks;
- Presence of nearby or downstream sensitive or unique habitat and ecology;
- Land clearing and topsoil management;
- Post-mining landform and rehabilitation planning.

The failure of a mining project to adequately address these environmental issues may result in significant ecosystem and public health risks. Due to this, environmental concerns can lead to significant community activism and opposition against the development of mining and mineral processing operations. When community concern or outrage reaches a tipping point, the project is said to have lost its "social licence to operate". In this circumstance, the development of the project may be slowed or be prevented entirely due to an inability to obtain the necessary government approvals or financing that would allow the project to go ahead. This can occur even in situations where there are strong economic incentives for the development of the resource. A good example is provided by the situation surrounding the proposed Pebble mine in Alaska, which would mine one of the largest undeveloped resources of gold, copper, and molybdenum in the world. The project faces a significant opposition from indigenous communities due to the potential impacts to local waterways and salmon fisheries. A community engagement process was orchestrated by the mining companies in an attempt to gain a social licence to operate; however, this process was unsuccessful and the project is now on hold indefinitely (Holley and Mitcham 2016).

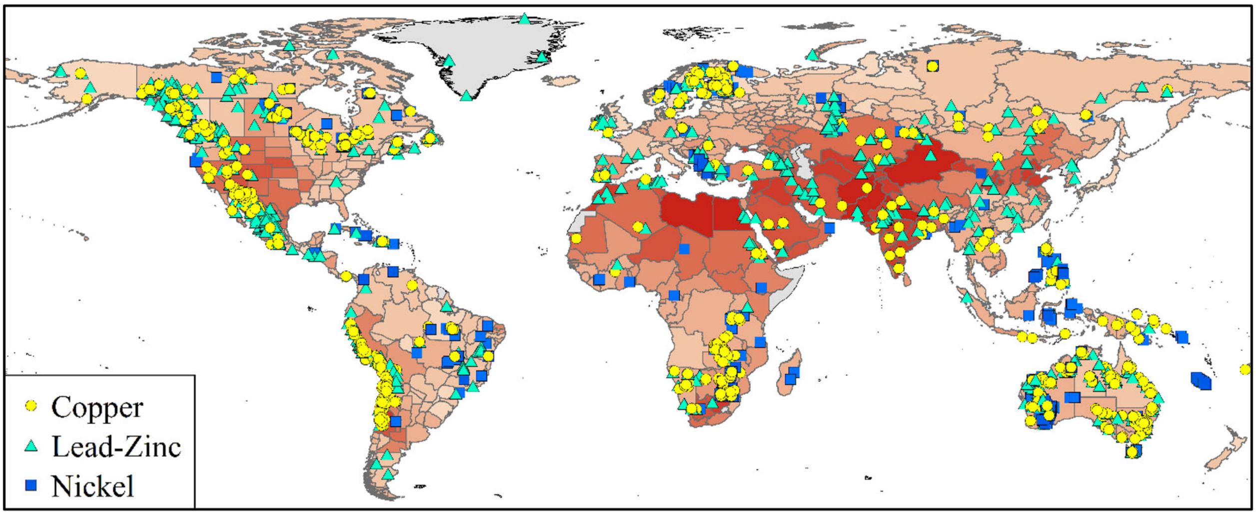

Figure 4. Global distribution of copper (Mudd et al. 2013a), lead–zinc (Mudd et al. 2017), and nickel resources (Mudd and Jowitt 2014) in relation to water criticality (Sonderegger et al. 2017). Darker shading indicates greater regional "water criticality". For further information see Northey et al. (2017).

The ability for an operation to achieve a social licence to operate may be heavily dictated by local social factors, such as:

- Community trust in information provided by companies, regulatory authorities, and decision-makers;
- The effectiveness of community consultation processes;
- Employment opportunities;
- Indigenous and cultural values towards the affected landscape; and
- Community cohesion concerns (e.g., in relation to fly-in fly-out workforces);

Beyond environmental and social concerns, various economic factors also directly influence the ease of developing a mineral resource. These may include:

- The presence of stable markets and distribution systems for fuels, materials, and process reagents;
- The availability of skilled labour;
- The security of site personnel and infrastructure;
- Access to ports and transport infrastructure;
- Governance issues (e.g., the potential for resource nationalization);
- Mineral royalty and taxation arrangements.

These environmental, social, and economic factors all play a role in determining the accessibility, rate of development, and ultimately the costs associated with developing a mineral resource. Incorporating these factors into assessments of global mineral resource availability and depletion is a conceptually difficult task, particularly when attempting to do so on a quantitative basis.

As an example, the absence or over-allocation of water in a region may prevent the development of a mineral resource due to difficulties in securing stable water supply and the perceived lack of water being a flashpoint for community opposition and social licence to operate risks. However, once water supplies have been secured, then water is generally a minor cost item for mining operations—and excesses of water may become more of a concern (e.g., due to flood risks or discharge restrictions). Northey et al. (2017) assessed the spatial distribution of copper, lead–zinc, and nickel resources in relation to a range of regional water indices such as water criticality (Fig. 4)—a regional indicator of water supply security, vulnerability to supply restrictions, and the environmental implications of water use (Sonderegger et al. 2015). One finding was that undeveloped copper resources are on average located in less water scarce or stressed regions than copper resources currently being mined and exploited (Northey et al. 2017), indicating that there may be potential for the copper industry to become less exposed to these risks in the future. Analysis based upon national-scale data also suggests that the hydrometallurgical copper supply chain is located in regions more exposed to water stress than the pyrometallurgical copper supply chain (Northey et al. 2014b). Comparisons can be made between resource types, commodity groups, and regions in this way to understand the relative exposure of mineral production to water scarcity issues. These forms of assessment would be greatly aided by having estimates of the water requirements of each individual mineral project. Coarse estimates of the total water withdrawals associated with mining exist (e.g., Gunson 2013); however, these contain a significant uncertainty and the data are generally not spatially disaggregated. Several conceptual and data limitations prevent more widespread evaluation of water consumption throughout the mining industry (Northey et al. 2016). However, several studies addressing these data availability issues are currently being prepared by the authors to better understand the magnitude and spatial distribution of water consumption in the mining industry. This will facilitate a more comprehensive assessment of the exposure of mineral resource supply chains to regional water scarcity and stress issues that may hinder resource development.

Accounting for Technology

Many of the basic strategies employed in mining, mineral processing, and metal production have been in use for centuries (Agricola 1556). However, there have been significant refinement and improvement of these approaches through time that have been aided by modern science, instrumentation, and engineering design and practice. The increasing improvement and sophistication of technology has led to sharp reductions in the capital, labour, material, and energy costs associated with mining and ore processing over time. These technology improvements have facilitated an expansion of economically available mineral resources, as lower production costs enable lower-quality mineral resources to be profitably mined and processed (Yaksic and Tilton 2009).

Therefore, assessments of mineral resource availability and depletion may benefit by considering the potential rates of technology improvement in mining and mineral/metal production processes. However, a potential complication distinguishes between incremental and disruptive technology improvements. The rate of incremental technology change, such as energy efficiency improvements in established unit processes, could potentially be estimated using performance data for mineral production operations over time. However, disruptive technology changes are by their nature irregular and their historic rates of introduction into the industry may not be predictive of the future—particularly in light of declining research budgets within the industry and the general outsourcing of this to equipment manufacturers that focus upon incremental improvements (Bartos 2007).

In order to understand trends in technology, it is important to understand the current state of technology deployment in the industry at discrete points in time. A variety of studies have surveyed the technologies being used to produce specific commodities. For instance, Marsden (2006) provided an overview of the processing technologies used to produce gold in 2004. Ramachandran et al. (2003) conducted a global survey of processing conditions at copper smelters. JOM published a summary of that survey (Kapusta 2004) as well as detailed smelter surveys for the production of platinum group metals (Jones 2004), nickel sulphides (Warner et al. 2006), and nickel laterites (Warner et al. 2007). The USGS has also published a detailed survey of material flows at global copper smelters (Goonan 2004), which was developed in a way that enables relatively straightforward benchmarking of operations. The Coalition for Eco-efficient Comminution (i.e., crushing and grinding) has also compiled and benchmarked comminution energy efficiency data for 167 mine sites that represent 1850 million tonnes of annual rock throughput (CEEC 2017), extending the work of Ballantyne and Powell (2014).

Some sources also exist that provide detailed technology information for specific regions. For instance, the Australasian Institute of Mining and Metallurgy (AusIMM) has published a series of monographs that outline operating practices and technology in use at Australian mining and mineral processing operations (Woodcock 1980; Woodcock and Hamilton 1993; Rankin 2013). These include detailed data for individual mine sites, such as the number and type of mining and ore processing equipment, as well as ore geology, typical operating conditions, and material, reagent, and energy consumption. Another example is the long-term average energy efficiency and greenhouse gas emission data that are available for Chile's copper industry (Comisión Chilena del Cobre 2014).

The copper industry provides some clear historical examples of technology changes transforming our understanding of available resources. Fine grinding and flotation methods were introduced into the copper industry in the early twentieth century and facilitated a large reduction in copper production costs (and incidentally copper grades; Radetzki 2009). Following this, the introduction of solvent extraction and electrowinning (SX-EW) processes into the copper industry in the latter half of the century led to another step change in production costs (although primarily for treating the oxidized portion of copper ore bodies). Both of these technology changes, in combination with efficiencies of scale associated with mass mining techniques (Crowson 2003), have facilitated a large expansion in copper production from low-grade ores that has occurred without noticeable long-term impacts on real copper prices (although there have been significant fluctuations at certain periods) (Radetzki 2009). Due to this, the magnitude of what could be considered economically recoverable copper resources has increased substantially over the past century, despite a substantial increase in the rate of physical depletion. Similar observations can be made for other mineral commodities.

CONCLUSIONS

The depletion of mineral resources potentially has large consequences for society over the long term. Understanding the extent and potential impacts of mineral resource depletion requires both a detailed understanding of geologic resources and the societal, economic, and environmental costs of extracting these resources. Therefore, the use of multi-disciplinary approaches is required to ensure that the conceptual underpinnings of studies are sound. Resource criticality assessments, material flow analysis, and life cycle assessment are three assessment frameworks that attempt to quantify various impacts or risks associated with mineral resource use. The conceptualization and quantification of mineral resource availability and depletion within these studies may benefit by reconsidering the fundamental relationships of economic geology (e.g., relationships between cut-off grades, resource grades, and mined grades), further accounting for the contextual factors influencing the development of mining operations, and incorporating potential rates of technology change. Therefore, the development of multi-disciplinary research teams and projects should be encouraged to enable more nuanced studies of mineral resource depletion and availability. Addressing these issues more comprehensively in studies will facilitate more informed decision-making, policy recommendations, and sustainable development outcomes.

ACKNOWLEDGEMENTS

We would like to thank the Natural Resources Research editorial team and the anonymous reviewers who provided detailed feedback that has greatly improved the quality of the paper. Discussions over the years with Dr. Steve Mohr, Dr. Damien Giurco, Dr. Simon Jowitt, and Dr. Mohan Yellishetty have informed our perspectives and are gratefully appreciated. Financial support for this study was provided by CSIRO Mineral Resources, as well as the CSIRO ''Wealth from Waste'' cluster—a collaborative programme between the Australian Commonwealth Scientific and Industrial Research Organisation (CSIRO); the University of Queensland, the University of Technology, Sydney; Monash University, Swinburne University of Technology, and Yale University.

REFERENCES

Agricola, G. (1556). *De Re Metallica* (H. C. Hoover & L. H. Hoover, Trans.). North Chelmsford: Courier Corporation (1950).

Ali, S. H., Giurco, D., Arndt, N., Nickless, E., Brown, G., Demetriades, A., et al. (2017). Mineral supply for sustainable development requires resource governance. *Nature, 543,* 367–372.

Armitage, A., Desautels, P., Zurowski, G., Pennstrom, W., Malbran, F., & Fitch, R. (2011). Preliminary Economic Assessment Update—Technical Report for the Malku Khota Project, Department of Potosi, Bolivia. Prepared for South American Silver Corp.

Arndt, N. T., Hedenquist, J. W., Kesler, S. E., Thompson, J. F. H., Arndt, N. T., & Wood, D. G. (2017). Future global mineral resources. *Geochemical Perspectives*. doi:10.7185/geochempersp.6.1.

Azimi, Y., Osanloo, M., & Esfahanipour, A. (2013). An uncertainty based multi-criteria ranking system for open pit mining cut-off grade strategy selection. *Resources Policy, 38,* 212–223.

Ballantyne, G. R., & Powell, M. S. (2014). Benchmarking comminution energy consumption for the processing of copper and gold ores. *Minerals Engineering, 65,* 109–114.
Bao, C., Mortazavi-Naeini, M., Northey, S., Tarnopolskaya, T., Monch, A., & Zhu, Z. (2013). Valuing flexible operating strategies in nickel production under uncertainty. In *Proceedings 20th international congress on modelling and simulation*, *Adelaide, Australia*, 1-6 December 2013, pp. 1426–1432.
Bartos, P. J. (2007). Is mining a high-tech industry? Investigations into innovation and productivity advance. *Resources Policy, 32*(4), 149–158.
Biedermann, R. P. (2014). China's rare earth sector—between domestic consolidation and global hegemony. *International Journal of Emerging Markets, 9*(2), 276–293.
Bösch, M. E., Hellweg, S., Huijbregts, M. A. J., & Frischknecth, R. (2007). Applying cumulative exergy demand (CExD) indicators to the ecoinvent database. *International Journal of Life Cycle Assessment, 12*(3), 181–190.
CEEC (2017). Energy Curve Program. Coalition for Eco-Efficient Comminution (CEEC). https://www.ceecthefuture.org/energy-curve-program/. Accessed 22 May 2017.
Ciacci, L., Nuss, P., Reck, B., Werner, T., & Graedel, T. (2016). Metal criticality determination for Australia, the US, and the Planet—Comparing 2008 and 2012 Results. *Resources, 5*(4), 29.
Clift, R., & Druckman, A. (Eds.). (2016). *Taking stock of industrial ecology*. Dordrecth: Springer.
Comisión Chilena del Cobre (2014). Análisis de Variables Clav es para la Sustentabilidad de la Minería en Chile, Cochilco, Government of Chile, Santiago. **(in Spanish)**.
Crowson, P. (2003). Mine size and the structure of costs. *Resources Policy, 29,* 15–36.
Crowson, P. C. F. (2011). Mineral reserves and future minerals availability. *Mineral Economics, 24,* 1–6.
Crowson, P. (2012). Some observations on copper yields and ore grades. *Resources Policy, 37,* 59–72.
Dewulf, J., Benini, L., Mancini, L., Sala, S., Blengini, G. A., Ardente, F., et al. (2015). Rethinking the area of protection ''Natural Resources'' in life cycle assessment. *Environmental Science and Technology, 49,* 5310–5317.
Dimitrakopoulos, R. G., & Sabour, S. A. A. (2007). Evaluating mine plans under uncertainty: Can the real options make a difference? *Resources Policy, 32,* 116–125.
Drielsma, J. A., Russell-Vaccari, A. J., Drnek, T., Brady, T., Weihed, P., Mistry, M., et al. (2016). Mineral resources in life cycle impact assessment—defining the path forward. *International Journal of Life Cycle Assessment, 21,* 85–105.
Elshkaki, A., & Graedel, T. E. (2013). Dynamic analysis of the global metals flows and stocks in electricity generation technologies. *Journal of Cleaner Production, 59,* 260–273.
Elshkaki, A., Graedel, T. E., Ciacci, L., & Reck, B. (2016). Copper demand, supply, and associated energy use to 2050. *Global Environmental Change, 39,* 305–315.
EU Commission (2014). Report on Critical Raw Materials for the EU, Report of the Ad hoc Working Group on Defining Critical Raw Materials, May 2014. http://ec.europa.eu/DocsRoom/documents/10010/attachments/1/translations/en/renditions/pdf.
Frenzel, M., Kullik, J., Reuter, M. A., & Gutzmer, J. (2017). Raw material 'criticality'—sense or nonsense? *Journal of Physics D: Applied Physics, 50*(12), 18.
Gemechu, E. D., Helbig, C., Sonnemann, G., Thorenz, A., & Tuma, A. (2015). Import-based Indicator for the geopolitical supply risk of raw materials in life cycle sustainability assessments. *Journal of Industrial Ecology, 201*(1), 154–165.
Gerst, M. D. (2008). Revisiting the cumulative grade-tonnage relationship for major copper ore types. *Economic Geology, 103,* 615–628.
Giurco, D., & Cooper, C. (2012). Mining and sustainability: asking the right questions. *Minerals Engineering, 29,* 3–12.
Giurco, D., Mohr, S., Mudd, G., Mason, L., & Prior, T. (2012a). Resource criticality and commodity production projections. *Resources, 1,* 23–33.
Giurco, D., Prior, T., Mason, L., Mohr, S., & Mudd, G. (2012b). Life-of-resource sustainability considerations for mining. *Australian Journal of Civil Engineering, 10*(1), 47–56.
Goonan, T.G. (2004). Flows of selected materials associated with world copper smelting. U.S. Geological Survey, Open-file report 2004-1395.
Gordon, R. B., Bertram, M., & Graedel, T. E. (2006). Metal stocks and sustainability. *Proceedings of the National Academy of Sciences (PNAS), 103*(5), 1209–1214.
Graedel, T. E. (2017). Grand challenges in metal life cycles. *Natural Resources Research*. doi:10.1007/s11053-017-9333-8.
Graedel, T. E., Barr, R., Chandler, C., Chase, T., Choi, J., Christoffersen, L., et al. (2012). Methodology of metal criticality determination. *Environmental Science and Technology, 46,* 1063–1070.
Graedel, T. E., Harper, E. M., Nassar, N. T., & Reck, B. K. (2015). On the materials basis of modern society. *Proceedings of the National Academy of Sciences, 112,* 6295–6300.
Graedel, T. E., & Reck, B. K. (2015). Six years of criticality assessments. What have we learned so far? *Journal of Industrial Ecology, 20*(4), 692–699.
Gunson, A.J. (2013). Quantifying, reducing and improving mine water use. PhD Thesis, The University of British Columbia.
Harmsen, J. H. M., Roes, A. L., & Patel, M. K. (2013). The impact of copper scarcity on the efficiency of 2050 global renewable energy scenarios. *Energy, 50,* 62–73.
Harper, E. M., Kavlak, G., Burmeister, L., Eckelman, M. J., Erbis, S., Espinooza, V. S., et al. (2015). Criticality of the geological zinc, tin, and lead family. *Journal of Industrial Ecology, 19*(4), 628–644.
Holley, E. A., & Mitcham, C. (2016). The pebble mine dialogue: A case study in public engagement and the social license to operate. *Resources Policy, 47,* 18–27.
Jones, R. T. (2004). JOM world nonferrous smelter survey. Part II: Platinum group metals. *JOM Journal of the Minerals Metals and Materials Society, 56*(12), 59–63.
Jowitt, S. M., Mudd, G. M., & Weng, Z. (2013). Hidden mineral deposits in Cu-dominated porphyry-skarn systems: How resource reporting can occlude important mineralization types within mining camps. *Economic Geology, 108,* 1185–1193.
Kapusta, J. P. T. (2004). JOM world nonferrous smelters survey, part I: Copper. *JOM Journal of the Minerals Metals and Materials Society, 56*(7), 21–27.
Kerr, R. (2014). The coming copper peak. *Science, 343,* 722–724.
Klinglmair, M., Sala, S., & Brandão, M. (2014). Assessing resource depletion in LCA: A review of methods and methodological issues. *International Journal of Life Cycle Assessment, 19*(3), 580–592.
Koppelaar, R. H. E. M., & Koppelaar, H. (2016). The ore grade and depth influence on copper energy inputs. *Biophysical Economics and Resource Quality*. doi:10.1007/s41247-016-0012-x.
Koslov, M. S. (2015). Formation conditions of the Rudny Altai metallogenic province. *Geology of Ore Deposits, 57*(4), 266–291.
La Brooy, S. R., Linge, H. G., & Walker, G. S. (1994). Review of gold extraction from ores. *Minerals Engineering, 7*(10), 1213–1241.
Lèbre, E., Corder, G., & Golev, A. (2017). The role of the mining industry in a circular economy, a framework for resource management at the mine site level. *Journal of Industrial Ecology, 21*(3), 662–672.
Marsden, J. O. (2006). Overview of gold processing techniques around the world. *Minerals & Metallurgical Processing, 23*(3), 121–125.

Mason, L., Prior, T., Mudd, G., & Giurco, D. (2011). Availability, addiction and alternatives: Three criteria for assessing the impact of peak minerals on society. *Journal of Cleaner Production, 19,* 958–966.
May, D., Prior, T., Cordell, D., & Giurco, D. (2012). Peak minerals: Theoretical foundations and practical application. *Natural Resources Research, 21*(1), 43–60.
Meinert, L. D., Robinson G. R. Jr., & Nassar, D. T. (2016). Mineral resources: Reserves, peak production and the future. *Resources, 5*, 14. doi:10.3390/resources5010014.
Mohr, S.H. (2010). Projection of world fossil fuel production with supply and demand interactions. PhD Thesis, The University of Newcastle.
Mohr, S., Giurco, D., Yelishetty, M., Ward, J., & Mudd, G. (2015). Projections of iron ore production. *Natural Resources Research, 24*(3), 317–327.
Mohr, S. H., Mudd, G. M., & Giurco, D. (2012). Lithium resources and production: Critical assessment and global projections. *Minerals, 2,* 65–84.
Mosier, D.L., Berger, V.I., & Singer, D.A. (2009). Volcanogenic Massive Sulfide Deposits of the World: Database and Grade and Tonnage Models, U.S. Geological Survey, Reston, Virginia, USA, Open File Report 2009-1034.
Mudd, G. M. (2010). The environmental sustainability of mining in Australia: Key mega-trends and looming constraints. *Resources Policy, 35,* 98–115.
Mudd, G. M., & Jowitt, S. M. (2014). A detailed assessment of global nickel resource trends and endowments. *Economic Geology, 109,* 1813–1841.
Mudd, G. M., Jowitt, S. M., & Werner, T. T. (2017). The world's lead-zinc mineral resources: Scarcity, data, issues and opportunities. *Ore Geology Reviews, 80,* 1160–1190.
Mudd, G. M., & Mohr, S. (2010). Mineral Resources and Production in the Goldfields-Esperance Region: Projecting Future Scenarios. Monash University. Prepared for the Goldfields Esperance Development Commission. http://www.gedc.wa.gov.au/getattachment/News-and-publications/Publications/Mineral-Resourcesand-Production_Goldfields-Esperance-Dec-2010.pdf.aspx.
Mudd, G. M., Weng, Z., & Jowitt, S. M. (2013a). A detailed assessment of global Cu resource trends and endowments. *Economic Geology, 108,* 1163–1183.
Mudd, G. M., Weng, Z., Jowitt, S. M., Turnbull, I. D., & Graedel, T. E. (2013b). Quantifying the recoverable resources of by-product metals: The case of cobalt. *Ore Geology Reviews, 55,* 87–98.
Müller, E., Hilty, L. M., Widmer, R., Schluep, M., & Faulstich, M. (2014). Modeling metal stocks and flows: A review of dynamic material flow analysis methods. *Environmental Science and Technology, 48*(4), 2102–2113.
Nassar, N. T., Barr, R., Browning, M., Diao, Z., Friedlander, E., Harper, E. M., et al. (2012). Criticality of the geological copper family. *Environmental Science and Technology, 46,* 1071–1078.
Nassar, N. T., Du, X., & Graedel, T. E. (2015a). Criticality of the rare earth elements. *Journal of Industrial Ecology, 19*(6), 1044–1054.
Nassar, N. T., Graedel, T. E., & Harper, E. M. (2015b). By-product metals are technologically essential but have problematic supply. *Science Advances, 1*(3), e1400180. doi:10.1126/sciadv.1400180.
Nickless, E. (2017). Resourcing future generations: A contribution by the Earth Science Community. *Natural Resources Research*. doi:10.1007/s11053-017-9331-x.
Norgate, T., & Jahanshahi, S. (2010). Low grade ores—smelt, leach or concentrate? *Minerals Engineering, 23,* 65–73.
Northey, S. A., Haque, N., Lovel, R., & Cooksey, M. A. (2014a). Evaluating the application of water footprint methods to primary metal production systems. *Minerals Engineering, 26,* 65–80.
Northey, S., Haque, N., & Mudd, G. (2013). Using sustainability reporting to assess the environmental footprint of copper mining. *Journal of Cleaner Production, 40,* 118–128.
Northey, S. A., Mohr, S., Mudd, G. M., Weng, Z., & Giurco, D. (2014b). Modelling future copper ore grade decline based on a detailed assessment of copper resources and mining. *Resources, Conservation and Recycling, 83,* 190–201.
Northey, S. A., Mudd, G. M., Saarivuori, E., Wessman-Jääskeläinen, H., & Haque, N. (2016). Water footprinting and mining: Where are the limitations and opportunities? *Journal of Cleaner Production, 135,* 1098–1116.
Northey, S. A., Mudd, G. M., Werner, T. T., Jowitt, S. M., Haque, N., Yellishetty, M., et al. (2017). The exposure of global base metal resources to water criticality, scarcity and climate change. *Global Environmental Change, 44,* 109–124.
Nuss, P., Harper, E. M., Nassar, N. T., Reck, B. K., & Graedel, T. E. (2014). Criticality of iron and its principal alloying elements. *Environmental Science and Technology, 48*(7), 4171–4177.
Panousi, S., Harper, E. M., Nuss, P., Eckelman, M. J., Hakimian, A., & Graedel, T. E. (2016). Criticality of seven specialty metals. *Journal of Industrial Ecology, 20*(4), 837–853.
Prior, T., Giurco, D., Mudd, G., Mason, L., & Behrisch, J. (2012). Resource depletion, peak minerals and the implications for sustainable resource management. *Global Environmental Change, 22*(3), 577–587.
Radetzki, M. (2009). Seven thousand years in the service of humanity—the history of copper, the red metal. *Resources Policy, 34*, 176–184.
Ramachandran, V., Diaz, C., Eltringham, T., Jiang, C. Y., Lehner, T., Mackey, P. J., et al. (2003). Primary copper production - a survey of operating world copper smelters. In *Copper 03-Cobre03 International conference, Vol. 4 Pyrometallurgy of Copper, Book 1: Smelting Operations, Ancillary Operations and Furnace Integrity,* pp. 3–16.
Rankin, W.J. (2013). *Australasian mining and metallurgical operating practices*. The Sir Maurice Mawby Memorial Volume (3rd ed.). Australasian Institute of Mining and Metallurgy, Monograph 28.
Schneider, L., Berger, M., & Finkbeiner, M. (2011). The anthropogenic stock extended abiotic depletion potential (AADP) as a new parameterization to model the depletion of abiotic resources. *International Journal of Life Cycle Assessment, 16*, 929–936.
Schneider, L., Berger, M., Schüler-Hainsch, E., Knöfel, S., Ruhland, K., Mosig, J., et al. (2014). The economic resource scarcity potential (ESP) for evaluating resource use based on life cycle assessment. *International Journal of Life Cycle Assessment, 119*(3), 601–610.
Schodde, R. (2011). Recent trends in mineral exploration—are we finding enough? In: *Proceedings of RMG 8th annual mining and exploration investment conference, Stockholm, Sweden*, 15–16 November 2011.
Singer, D. A. (2017). Future copper resources. *Ore Geology Reviews, 86,* 271–279.
Singer, D.A., Berger, V.I., & Moring, B.C. (2008). Porphyry copper deposits of the world—database and grade and tonnage models. U.S. Geological Survey (USGS), Reston, Virginia, USA, Open-File Report 2008-1155.
Sonderegger, T., Dewulf, J., Fantke, P., de Souza, D. M., Pfister, S., Stoessel, F., et al. (2017). Towards harmonizing natural resources as an area of protection in life cycle impact assessment. *International Journal of Life Cycle Assessment*. doi:10.1007/s11367-017-1297-8.
Sonderegger, T., Pfister, S., & Hellweg, S. (2015). Criticality of water: Aligning water and mineral resource assessment. *Environmental Science and Technology, 49,* 12315–12323.

Sprecher, B., Daigo, I., Murakami, S., Kleijn, R., Vos, M., & Kramer, G. J. (2015). Framework for resilience in material supply chains, with a case study from the 2010 Rare Earth Crisis. *Environmental Science and Technology, 49*(11), 6740–6750.

Steen, B. A. (2006). Abiotic resource depletion, different perceptions of the problem with mineral deposits. *International Journal of Life Cycle Assessment, 11*(1), 49–54.

Sverdrup, H. U., Ragnarsdottir, K. V., & Koca, D. (2014). On modelling the global copper mining rates, market supply, copper price and the end of copper reserves. *Resources, Conservation and Recycling, 87,* 158–174.

Swart, P., & Dewulf, J. (2013). Quantifying the impacts of primary metal resource use in life cycle assessment based on recent mining data. *Resources, Conservation and Recycling, 73,* 180–187.

Tilton, J. E., & Lagos, G. (2007). Assessing the long-run availability of copper. *Resources Policy, 32,* 19–23.

Tukker, A. (2014). Rare earth elements supply restrictions: Market failures, not scarcity, hamper their current use in high-tech applications. *Environmental Science and Technology, 48,* 9973–9974.

Valero, A., & Valero, A. (2012). From grave to cradle. *Journal of Industrial Ecology, 17*(1), 43–52.

Van Oers, L., de Koning, A., Guinée, J.B., & Huppes, G. (2002). Abiotic resource depletion in LCA, Road and Hydraulic Engineering Institute, Ministry of Water and Transport, Amsterdam, 25 June 2002. http://media.leidenuniv.nl/legacy/report%20abiotic%20resource%20depletion.pdf.

Van Oers, L., & Guinée, J. (2016). The abiotic depletion potential: Background, updates, and future. *Resources, 5*(1), 16. doi:10.3390/resources5010016.

Vieira, M. D. M., Goedkoop, M. J., Storm, P., & Huijbregt, M. A. J. (2012). Ore grade decrease as life cycle impact indicator for metal scarcity: The case of copper. *Environmental Science and Technology, 46,* 12772–12778.

Warner, A. E. M., Díaz, C. M., Dalvi, A. D., Mackey, P. J., & Tarasov, A. V. (2006). JOM world nonferrous smelter survey. Part III: Nickel: Laterite. *JOM Journal of the Minerals Metals and Materials Society, 58*(4), 11–20.

Warner, A. E. M., Díaz, C. M., Dalvi, A. D., Mackey, P. J., Tarasov, A. V., & Jones, R. T. (2007). JOM world nonferrous smelter survey Part IV: Nickel: Sulfide. *JOM Journal of the Minerals Metals and Materials Society, 59*(4), 58–72.

Weng, Z., Jowitt, S. M., Mudd, G. M., & Haque, N. (2015). A detailed assessment of global rare earth resources: Opportunities and challenges. *Economic Geology, 110,* 1925–1952.

Werner, T. T., Mudd, G. M., & Jowitt, S. M. (2015). Indium: key issues in assessing mineral resources and long-term supply from recycling. *Applied Earth Science (Transactions of the Institute of Minerals and Metallurgy B), 124,* 213–226.

Werner, T. T., Mudd, G. M., & Jowitt, S. M. (2017). The world's by-product and critical metal resources part III: A global assessment of indium. *Ore Geology Reviews, 86*, 939–956.

West, J. (2011). Decreasing metal ore grades, are they really being driven by the depletion of high-grade deposits? *Journal of Industrial Ecology, 15*(2), 165–168.

Woodcock, J. T. (1980). *Mining and metallurgical practices in Australasia*. The Australasian Institute of Mining and Metallurgy.

Woodcock, J. T., & Hamilton, J. K. (1993). *Australasian mining and metallurgy*, vol. 1 & 2. The Australian Institute of Mining and Metallurgy, Victoria.

Yaksic, A., & Tilton, J. E. (2009). Using the cumulative availability curve to assess the threat of mineral depletion: The case of lithium. *Resources Policy, 34,* 185–194.

Yap, A., Saconi, F., Nehring, M., Arteaga, F., Pinto, P., Asad, M. W. A., et al. (2013). Exploiting metallurgical throughput-recovery relationship to optimise resource value as part of the production scheduling process. *Minerals Engineering, 53,* 74–83.

Yellishetty, M., Ranjith, P. G., Tharumarajah, A., & Bhosale, S. (2009). Life cycle assessment in the minerals and metals sector: A critical review of selected issues and challenges. *International Journal of Life Cycle Assessment, 14,* 257–267.

7. Water Use Reporting and Data for the Mining Industry

In this chapter, a brief overview is provided of the corporate sustainability reporting and water accounting standards that are being used and developed for the mining industry. Following this, it is shown how the public reporting of mining companies can be used to develop detailed databases of the water use, consumption and discharges of individual mining operations. A conceptual approach to developing more rigorous water footprint inventory data for the mining industry based upon these datasets is then discussed.

7.1. Water Use Reporting

Over the past two decades the mining industry has increasingly made disclosures of water use as part of environmental management and corporate sustainability reporting (Perez and Sanchez, 2009). These disclosures may include mandatory reporting, such as environmental compliance reporting to regulatory authorities that may be made public in some jurisdictions. In other cases, mining companies are voluntarily disclosing water use data through initiatives such as corporate sustainability reporting and market disclosures (Leong et al., 2014). Most existing studies that have assessed these activities have focused upon the degree of compliance with the various reporting standards that guide these disclosures, such as the Global Reporting Initiative (Fonseca et al., 2014; Jenkins and Yakovleva, 2006). However, there has been more limited analysis of the actual data being communicated within these reports and how this can be used to develop a more rigorous understanding interactions of the mining industry with the environment and society.

The Global Reporting Initiative (GRI) provides a framework for organisation's to regularly publish ‘Sustainability Reports’ that describe performance of the company on social, economic and environmental grounds (GRI, 2013a) . Despite being voluntary, there has been strong uptake of GRI based sustainability reporting by major mining companies (Perez and Sanchez, 2009; Fonseca et al., 2014). The GRI requires organisations to report against a variety of societal, environmental and economic performance indicators. The main indicators that could potentially provide useful data be reported under by the most recent reporting standard GRI4 include (GRI, 2013a):

- G4-EN8 – Total water withdrawal by source.
- G4-EN9 – Water sources significantly affected by withdrawal of water.
- G4-EN10 – Percentage and total volume of water recycled and reused.
- G4-EN22 – Total water discharge by quality and destination.
- G4-EN26 – Identity, size, protected status, and biodiversity value of water bodies and related habitats significantly affected by the organisations discharges of water and runoff.

The GRI has evolved over time to meet the needs of stakeholders and to improve the meaningfulness or requirements of reporting indicators. Additional reporting supplements specifically for the mining industry have been made available to improve the quality of disclosures being made by the sector (GRI, 2013b).

The Carbon Disclosure Project (CDP) began as a scheme that was focused upon querying companies on their exposure to climate change risks and the actions they were taking to mitigate or adapt to these risks (CDP, 2017). Following the success of the CDP, a derivative scheme, CDP Water, was established to assess how companies are exposed to water risks and the actions they are taking in manage or address these. CDP Water is structured as a questionnaire that is sent to companies and the focus is on understanding an individual company's exposure to water related risks. Most major mining companies now regularly report to the CDP and CDP Water schemes as part of their voluntary reporting practices. From this considerable insights are able to be reached regarding the water related risks that the mining industry faces, and their responses to these risks (CDP, 2013). CDP Water reports are accessible through an online database and readers are encouraged to explore these, as they provide unique insights into how the mining industry views water risks (GRI, 2017).

Despite the widespread reporting of mining companies using schemes such as the GRI and CDP Water, there has been substantial inconsistency in how mining companies have been reporting water use data to these schemes (Cote et al., 2012; Mudd, 2008; Leong et al., 2014). Thefore the mining industry has developed its' own water accounting standards to facilitate the more consistent communication and reporting of water use information by mining companies. As an example of this, the International Council on Mining & Metals recently released a water reporting framework for the industry (ICMM, 2017) that was heavily developed based upon the Water Accounting Framework for the Minerals Industry that was developed by the Sustainable Minerals Institute for the Minerals Council of Australia (MCA, 2014).

The *Water Accounting Framework for the Minerals Industry* (WAFMI) was developed by the Sustainable Minerals Institute (University of Queensland) for the Minerals Council of Australia (MCA) (Cote et al., 2012; MCA, 2014). The framework provides a consistent way of accounting for water flows through and within a mine-site, to provide consistency in the accounting and reporting of this information. The WAFMI provides a systematic way of recording the inputs, outputs, diversion and storage of water at a site level. Water quality thresholds are used by the WAFMI to account for water inputs and outputs across three water quality categories

The WAFMI provides data in two ways. An input-output table is the main outcome of adopting the WAF. This provides a measure of all the inflows to the site, such as: rainfall, mine water infiltration, ground and surface water withdrawals, and moisture entrainment in ores. Outputs include parameters such as: seepage, evaporation, discharges and tailings entrainment. A statement of task usages is also included and contains flows into individual processes, such as: concentrators, mine site equipment, etc. Analysis as shown that the WAFMI is flexible enough to be broadly applicable to mining operations, regardless of the local climate or hydrological context of the mining operation (Danoucaras et al., 2014).

The International Council on Mining & Metals has in recent years begun to provide guidance on water management in the mining industry through the release of a range of publications listed below:

- Water management in mining: a selection of case studies (ICMM, 2012)
- Adapting to a changing climate: implications for the mining and metals industry' (ICMM, 2013)
- Water stewardship framework (ICMM, 2014)
- A practical guide to catchment-based water management for the mining and metals industry (ICMM, 2015)
- A practical guide to consistent water reporting (ICMM, 2017)

In March 2017, the ICMM released the 'Practical Guide to Consistent Water Reporting' (ICMM, 2017). The ICMM's water reporting guide was heavily modelled upon the WAFMI and so the two accounting standards share the same basic underpinnings. However, a major improvement over the WAFMI is the greater treatment given to providing guidance to describe the local water context surrounding mining operations and more broadly the communication of water risk related information.

7.2. Water Use Database Development

Several authors have previously compiled datasets of mine water use statistics, based most commonly on the corporate sustainability reporting of mining companies. The early assessment of direct water use for various metal production routes by Norgate and Lovel (2004; 2006) was based directly on data compiled from the corporate sustainability reporting of mining, mineral processing and metal producing companies – although this is not typically recognised. Considerable work was also undertaken by Mudd (2008) to compile a dataset of mine water use intensity (e.g. m^3/t product, m^3/t ore) that, despite being the most comprehensive assessment at the time. Mudd considered this as a preliminary effort only, as he was aware of considerably more reporting by companies that was not captured by his data compilation efforts. Prior to the present doctoral studies, the author compiled a dataset of the water use intensity of copper mining operations (Northey et al., 2013). Gunson (2013) also compiled a detailed dataset of mine water use intensity for 19 mined commodities, which he used to estimate the global water withdrawals associated with non-fuel mining for the years 2005-2008 (Table 4). A range of limitations and shortcomings exist with these studies, including:

- They have tended to focus compiling either water 'use', 'consumption' and 'withdrawals' data for mining operations, with very limited definition or differentiation of these terms.
- Water sources (e.g. surface, groundwater, rainfall, etc.) has not been specified.
- Water discharge data has typically not been compiled.
- The water quality of withdrawals and discharges has also not been specified.
- Limited assessment how the data relates to local climate and water use contexts.

These short-comings are in large part due to the highly variable and inconsistent water use reporting practices of individual companies, which would have prevented such analysis.

Table 4: Gunson's (2013) estimate of global water withdrawals associated with non-fuel mining, based upon his ore production method.

	Withdrawals, Mm^3 H_2O			
	2006	**2007**	**2008**	**2009**
Total	**6,870**	**7,766**	**7,489**	**7,518**
Phosphate	3,046	3,258	3,187	3,052
Copper	1,337	1,233	1,301	1,363
Gold	657	942	997	1,141
Iron	589	737	555	883
Diamonds	546	713	206	305
Nickel	20	71	147	165
Zinc	195	243	82	95
Platinum	50	59	97	94
Potash	62	61	66	80
Bauxite	67	65	76	79
Molybdenum	129	102	86	58
Silver	22	21	28	40
Chromite			256	32
Lead	44	118	30	29
Tungsten	65	76	35	27
Uranium	9	12	18	25
Cobalt	3	16	26	19
Rhodium	21	31	45	16
Palladium	8	8	11	14

The development of industry water reporting guidance the such as those developed by the Minerals Council of Australia (MCA, 2014) and the ICMM (2017) is expected to lead to increasing consistency and sophistication of mine-site water use by companies. Therefore, there is an expectation of improved availability and quality of water use data for the mining industry will improve going forward, and that this will provide new sources of information that can be used to evaluate the water consumption and performance of mining operations.

Currently, the author is involved in ongoing work to develop a detailed compilation of publically disclosed water use data for mining companies, divisions or individual mining operations. As part of this, instances of water use disclosures in corporate sustainability and environmental compliance reporting are being reviewed and retrospectively assigned to one of 55 data categories (Figure 7.1), which have been developed based upon the reporting categories of the MCA's (2014) and ICMM's (2017) water reporting frameworks. The water quality categories defined by these frameworks have also been applied to these data points when sufficient information is available.

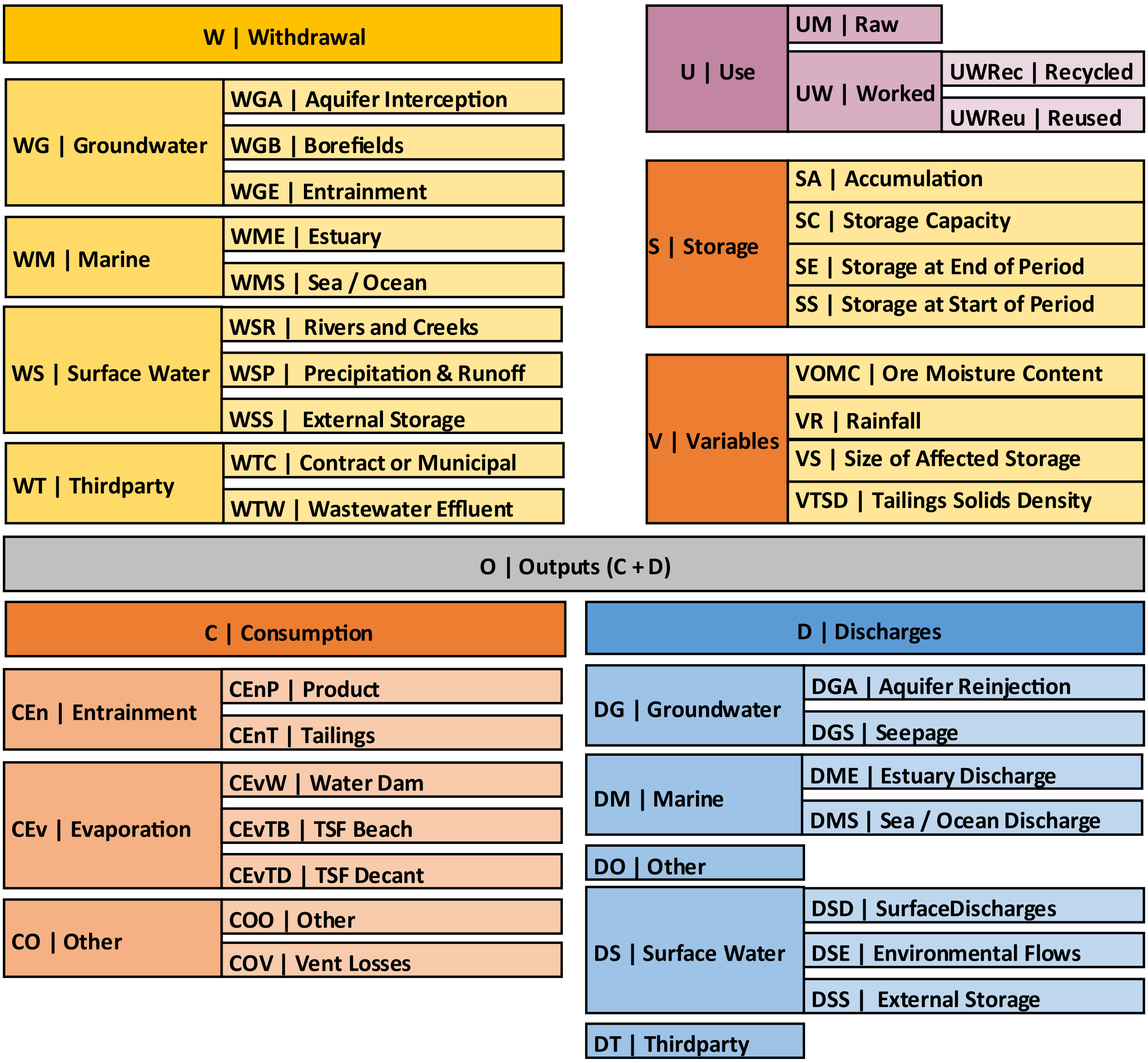

Figure 7.1: Key data categories included in the mine water reporting database, which was largely based upon the reporting categories outlined by MCA (2014) and ICMM (2017).

As the development of this database will extend beyond the duration of doctoral studies considered by this thesis, only a brief review of the dataset is provided here. Currently the database contains reported data for 225 mining operations located across 29 countries (Table 5). Additionally the database also contains aggregated reporting for entire companies or company divisions. In total, the database currently contains 8,346 data points that have been identified within 359 separate corporate sustainability or environmental management reports, and these data points have each been assigned a data category. Table 6 shows the extent of identified reporting against each of these data categories. The most commonly reported water use metrics by the industry are water withdrawal data, often specifying the source of withdrawals, and also raw water and worked water (reused or recycled) requirements of site processes. Discharge data also exists for many mining operations. However, reporting is less common for data categories related to the on-site storage of water and also specific modes of water consumption (e.g. evaporation, tailings entrainment, etc.).

Table 5: List of countries containing a mining operation or division that reported data has been compiled for.

Countries Containing A Mining Operation or Division with Reported Data			
Argentina	Fiji	Mozambique	Suriname
Australia	France	Namibia	Tanzania
Brazil	Germany	Papua New Guinea	United Kingdom
Canada	India	Peru	United States
Chile	Indonesia	Philippine	Zambia
Cote d'Ivoire	Ireland	Saudi Arabia	
Dem. Rep. of Congo	Laos	South Africa	
Dominican Republic	Mongolia	Spain	

Table 6: Number of data points compiled for each data category.

Code	Description	No.	Code	Description	No.
W	Withdrawals	552	C	Consumption	43
WG	Groundwater	333	CEn	Entrainment	39
WGA	Aquifer Interception	133	CEnP	Product	4
WGB	Borefields	486	CEnT	Tailings Entrainment	4
WGE	Ore Entrainment	24	CEv	Evaporation	58
WM	Marine	54	CEvTB	Tailings Beach Evaporation	5
WMS	Seawater	16	CEvTD	Tailings Decant Evaporation	4
WS	Surface Water	275	CEvW	Water Dam Evaporation	10
WSP	Precipitation and Runoff	109	CO	Other	15
WSR	Rivers and Creeks	79	COD	Dust Suppression	11
WSS	External Storage	33	COO	Other	16
WT	Third-party	151	COV	Vent Losses	10
WTC	Contract or Municipal	124		**Sub-total**	**219**
WTW	Wastewater Effluent	40	D	Discharges	610
	Sub-total	**2,409**	DG	Groundwater	48
U	Use	151	DGS	Seepage	27
UM	Makeup/Raw	2,547	DM	Marine	10
UW	Worked	1,239	DME	Estuary	11
UWRec	Recycled (treated)	18	DMS	Sea/Ocean	28
UWReu	Reused (untreated)	20	DO	Other	46
	Sub-total	**3,975**	DS	Surface Water	124
Div	Diversions	7	DSD	Surface Discharges	23
SA	Accumulation	16	DSE	Environmental Flows	16
SC	Storage Capacity	9	DSS	External Storage	10
SE	Storage at End of Period	16	DT	Third-party	61
SES	Storage at Start of Period	16		**Sub-total**	**1014**
VOMC	Ore Moisture Content	5	O	Outputs (C+D)	150
VR	Rainfall	411			
VP	Pan Evaporation	59			
VS	Size of Affected Water Source	10			
VTSD	Tailings Solids Density	30			
	Sub-total	**579**		**Grand Total**	**8,346**

Given the significant breadth of data currently being reported for the mining industry, it is possible to develop detailed understanding of the overall water balance of many mining operations. For instance, Table 7 presents a summary of key water use statistics for 35 copper mining operations currently included within the database. From this data it is clear that there is substantial variability in how mining operations are utilising and interacting with water. Some mines sites are heavily dependent upon withdrawing water from groundwater, whereas others are more dependent upon surface water systems (including rainfall runoff) or occasionally third-party sources (e.g. municipalities). There is also substantial variability in the intensity of raw water use and also the contribution of worked water (reused or recycled) to total water use. The discharge data is also highly variable and it is assumed that many of the mining operations are operated as zero discharge sites.

Understanding the factors that drive the variability in the observed withdrawal, use and discharge data is not a straightforward task. Copper mining operations are highly variable in terms of their processing configurations, although from a water perspective this can roughly be generalised into two main processing archetypes – heap leaching or flotation separation of sulphide ores – that could form the basis for further assessment (see for instance Northey et al., 2013). There is also a temporal aspect to consider, as multiple decades of data is available for some mines (e.g. Olympic Dam). Therefore, process and technology improvements through time may also be a relevant consideration, as for instance there is some evidence that Chilean copper mines are becoming more water efficient over time (Lagos et al., 2017). Finally, the water balance of a mining operation is heavily influenced by variability in weather and hydrological conditions and so evaluating the dataset to identify the influence of these factors is also another potential avenue of future research.

Table 7: Summary of water withdrawals, use and discharges for 25 copper mines. Absence of a data value does not imply that the flow does not exist, rather only no public reporting was identified. Data key: Arithmetic average ± standard deviation (years of data). 'Worked' water is presented relative to total water use (raw + worked water).

Operation	Withdrawals					Use		Discharges
	Groundwater kL/t ore	Surface kL/t ore	Marine kL/t ore	Third-party kL/t ore	Total kL/t ore	Raw kL/t ore	Worked %	kL/t Ore
Antamina	0.07±0.00(4)	0.43±0.06(4)	-	-	0.53±0.08(2)	-	-	0.90±0.06(4)
Alumbrera	-	-	-	-	-	0.59±0.03(2)	-	-
Andina	0.20±0.14(7)	0.83±0.20(7)	-	0.00±0.00(7)	1.01±0.07(10)	1.06±0.16(11)	44.8±6.9(13)	0.80±0.52(11)
Bingham Canyon	-	-	-	-	-	1.13±0.18(6)	49.8±3.9(6)	0.32±0.00(1)
Cadia Valley Operations	0.05±0.01(9)	0.51±0.44(9)	-	0.14±0.04(9)	0.69±0.42(9)	0.40±0.14(11)	74.4±11.4(10)	0.17±0.22(8)
Chuquicamata	0.31±0.52(3)	0.89±0.59(4)	-	0.00±0.00(4)	1.12±0.14(4)	1.04±0.12(4)	87.1±1.4(5)	-
Cobar-CSA	0.29±0.00(2)	21.24±0.00(1)	-	-	-	1.04±0.20(16)	45.5±8.7(3)	-
Codelco Norte	0.10±0.16(3)	0.30±0.18(2)	-	0.00±0.00(1)	0.45±0.02(7)	0.45±0.02(7)	84.8±1.6(8)	0.06±0.14(9)
Collahuasi	0.64±0.04(10)	-	-	0.05±0.00(1)	0.64±0.04(10)	0.61±0.04(13)	77.4±1.5(10)	0.02±0.00(5)
El Teniente	0.10±0.11(5)	1.07±0.28(6)	-	0.00±0.00(3)	1.24±0.24(9)	1.24±0.26(11)	57.5±3.9(12)	1.03±0.40(11)
Ernest Henry	0.62±0.10(5)	0.49±0.13(7)	-	-	-	0.48±0.13(10)	-	-
Escondida	0.58±0.10(6)	-	0.02±0.02(3)	0.00±0.00(1)	0.53±0.06(2)	0.64±0.03(2)	28.3±2.6(2)	0.02±0.01(4)
Gabriela Mistral	0.11±0.03(6)	0.00±0.00(4)	-	0.00±0.00(4)	0.12±0.02(4)	0.12±0.02(3)	51.7±44.2(4)	-
Golden Grove	1.57±0.00(1)	-	-	-	1.51±0.19(5)	-	75.0±0.0(1)	0.37±0.23(3)
Kidd Mine	-	-	-	-	-	-	85.0±0.0(1)	-
Kinsevere	-	-	-	-	4.27±4.10(4)	-	-	-
Las Bambas	-	-	-	-	0.18±0.00(1)	-	15.2±3.7(3)	-
Lomas Bayas	0.06±0.04(8)	0.06±0.05(8)	-	-	0.11±0.02(8)	0.12±0.02(4)	24.4±0.2(2)	-
Lumwana	-	-	-	-	-	0.15±0.06(5)	88.0±2.6(4)	0.82±0.59(4)
Ministro Hales	0.58±0.39(2)	0.00±0.00(2)	-	0.01±0.00(2)	0.58±0.40(2)	0.31±0.00(1)	20.2±1.7(2)	-
Mount Isa - Copper	-	0.89±0.29(4)	-	-	-	0.49±0.28(9)	66.1±6.7(5)	-
Mount Lyell	1.60±0.00(1)	2.37±0.40(3)	-	-	2.61±1.22(4)	2.03±0.60(2)	14.3±0.0(1)	9.36±4.39(7)
Northparkes	0.43±0.17(9)	0.18±0.10(8)	-	-	0.64±0.17(8)	1.53±2.39(14)	49.7±21.6(10)	-
Oyu Tolgoi	0.56±0.00(1)	-	-	-	-	0.48±0.06(4)	85.0±1.2(4)	-
Ok Tedi	-	0.57±0.21(12)	-	-	-	-	-	-
Olympic Dam	1.49±0.51(28)	-	-	-	-	1.15±0.10(10)	-	-
Palabora	0.34±0.16(5)	0.24±0.11(5)	-	0.50±0.08(3)	1.00±0.35(3)	0.89±0.39(18)	72.1±10.1(12)	0.09±0.00(1)
Prominent Hill	0.65±0.08(6)	0.00±0.00(2)	-	0.00±0.00(2)	0.64±0.14(3)	-	19.5±3.5(2)	-
Radomiro Tomic	0.08±0.02(3)	0.02±0.00(3)	-	-	0.10±0.02(3)	0.10±0.02(4)	88.2±2.7(5)	-
Rosebery	-	-	-	-	19.08±12.27(3)	-	-	7.86±6.58(3)
Salvador	0.58±0.46(6)	1.20±0.54(7)	-	0.00±0.00(6)	1.64±0.16(11)	1.59±0.19(12)	37.9±8.5(12)	0.31±0.45(12)
Sepon	0.05±0.02(2)	1.93±0.86(3)	-	-	3.34±1.52(8)	-	-	3.72±3.51(7)
Spence	-	-	-	0.32±0.04(2)	-	0.32±0.04(2)	96.0±0.0(1)	-
Telfer	0.91±0.00(1)	-	-	-	-	0.87±0.34(7)	12.9±8.2(6)	0.04±0.01(2)
Zaldivar	-	-	-	-	-	0.15±0.01(6)	93.0±1.4(2)	0.00±0.00(6)

7.3. A Conceptual Framework for Mine Water Inventory Development

Given the widespread availability of reported water use data for the mining industry, the compilation of this data presents an opportunity to substantially improve estimates of the ‘direct’ (on-site) water footprint of mined products. From the author’s perspective, an ideal inventory dataset would have a few key features. Any aggregation of inventory data would be avoided to enable attribution of resource use or impacts to specific producers and regions. All unit processes would be assigned locations and geographic boundaries to enable interoperability with future impact characterisation approaches, which are increasingly becoming spatially specific. The database would be structured correctly to allow full traceability of supply chains and flexibility in the use of allocation methods, utilising many of the concepts outlined by Majeau-Bettez et al. (2014). References would be provided for individual data points and the overall representativeness, accuracy and potential sampling bias of each data point would be assessed (Freschknecht et al., 2004; Koffler et al., 2017), particularly when using data from one facility to act as a proxy for missing data at other facilities.

With those points in mind, a conceptual framework for developing rigorous water footprint estimates of the global mining industry is presented in Figure 7.2.

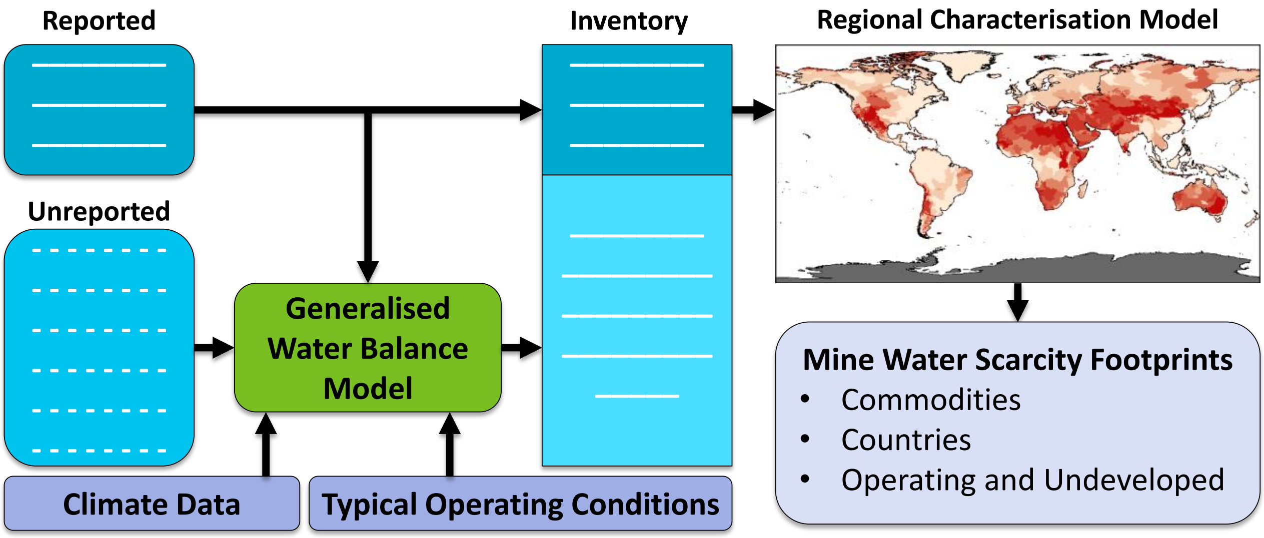

Figure 7.2: Conceptual framework for developing rigorous water scarcity footprint estimates of the mining industry.

Each component of the conceptual framework is briefly described below:

‘Reported’ – A database comprising the reported production and water use statistics of individual mining operations.

‘Unreported’ - A database comprising partial production, resource and/or water use statistics of individual mining operations.

‘Generalised Water Balance Model’ – A stochastic or deterministic model used to estimate the potential water footprint inventory for mining operations that report no, or only partial, water use data. The model is calibrated using the reported dataset, as well as information on local climate and the expected operating conditions of the mine, mineral separation and tailings management processes. Depending upon data quality and availability, regression based models may be sufficient to estimate expected water balances based upon the available data. Alternatively, (semi-) empirical models could also be developed for particular mine and mineral processing configurations, based upon systems modelling approaches. Systems modelling can provide an understanding of how mine site infrastructure, processes and water management practices may respond to weather events and changes in the hydrological balance of the mining operation and key infrastructure (Gao et al., 2014; 2016; 2017). The processes used to separate valuable minerals and metals from mined ore are varied (Figure 7.3), however studies have identified the main factors that determine the water requirements of mineral processing circuits. These include: the ore grade and the degree of pre-concentration prior to the addition the water required for ore processing, the optimum solids density required for unit processes, the ability to thicken and/or filter the mineral concentrate or waste products (i.e. tailings) to recover water, and also the ability to return water from tailings storage facilities or other mine site infrastructure (Bleiwas, 2012a; 2012b; Gao et al., 2014; 2016; 2017; Gunson et al., 2012; Mwale et al., 2005; Northey et al., 2014a). Therefore with sufficient data and the classification of processes at individual mining operations, a generalised water balance model could potentially be developed using either regression or systems modelling approaches.

‘Typical Operating Conditions’ – A database comprising expected relationships between mining operation parameters, such as: the relationship between resource size and the surface area of mine site infrastructure (e.g. open pits, tailings impoundment, etc.), the typical tailings thickener solids underflow density, the relationship between local climate and discharge requirements, etc.

‘Inventory’ – A synthetic water footprint inventory that combines the reported dataset, as well as the estimated inventory data generated by the generalised water balance model for all other known mining operations.

Regional Characterisation Model – A life cycle impact characterisation model used to evaluate the relative impacts associated with water consumption in different regions.

Mine Water Scarcity Footprints – The inventory data developed would provide flexibility to evaluate the water footprint of mined commodities from specific geographic regions to support material sourcing decisions and industry benchmarking. The modelling approach would also enable assessment of the likely water footprint of exploiting currently undeveloped mineral deposits. Therefore this modelling approach could be integrated with mineral production scenarios to understand how the industry’s consumptive water use impacts may change through time. Effectively, this would provide greater sophistication to studies such as Elshkaki et al. (2017).

Figure 7.3: Examples of mineral separation and waste treatment processes. (Left) A 150m^3 flotation tank used in the initial ‘rougher’ separation of a copper concentrate from gangue mineralogy. Photo taken in March 2013 by the author (S.A. Northey). (Right) Aeration and settling tanks used for the treatment of acid mine drainage. Photo taken in August 2016 by the author (S.A. Northey).

7.3.1. Water Quality Considerations

The mining industry can significantly impact water quality in surrounding surface, marine and groundwater systems through both diffuse and point sources of emissions such as dust generation, wastewater discharge and acid mine drainage. A land-scape affected by acid mine drainage is shown in Figure 7.4 to provide the reader with a sense of the potential land degradation that can occur in mining regions. Given the potentially large impacts of mining on downstream water quality, water footprint and life cycle assessment studies should aim to accurately account for these impacts.

Figure 7.4: A golf course impacted by acid mine drainage in the West Rand mining region near Johannesburg, South Africa. The soil's yellow colouring is indicative of the presence of elemental sulphur and the lake on the right has been contaminated with radionucleides. Photo taken in August 2016 by the author (S.A. Northey).

The conceptual framework outlined earlier in Figure 7.5 may be used to develop a more representative inventory that can be used to assess consumptive water use impacts. However, such an approach would have more limited usefulness for assessing the potential degradative water use impacts of the mining industry – which are highly site specific due to differences in geochemistry, hydrology, mining techniques, processing, and ore handling and waste management practices. Therefore, alternative approaches are required to improve assessment of the water quality impacts associated with mining.

Many existing life cycle impact assessment methods simply require data of total pollutant loads when for characterising the potential for freshwater eutrophication or eco-toxicity impacts (e.g. Goedkoop et al., 2009). In many regions, mining operations are regularly required to provide national or regional pollutant inventories with estimates of total emissions to land, air and water that exceed reporting thresholds. Inventories and databases exist for most major economies such as Australia (NPI, 2017), Canada (NPRI, 2017), Chile, Europe (EEA, 2017), Japan, Mexico, United States (US EPA, 2017) and the OECD (2017). Through the author's discussions with mine-site environmental managers and scientists, the quality of data that is reported by mining operations to these schemes is often poor due to difficulties with sampling and modelling emission sources. Despite this, the various pollutant inventory schemes provide a wealth of data on total pollutant loads that would otherwise be unavailable for incorporation into life cycle inventories and impact assessment.

Some life cycle inventory and impact assessment methods specific to water require an understanding of the water quality associated with flows on a concentration, rather than total load, basis (Boulay et al., 2011; Bayart et al., 2014). For many mining operations, information on water quality and pollutant concentrations is included within public environmental compliance reporting. However, this data often represents the water quality at monitoring stations and defined sampling points, rather than as the quality parameters associated with a specific flow from the mining operation. As the water quality impacts are more diffuse than some other industries such as manufacturing, which will often have point discharges of wastewater, there can often be significant difficulty in attributing water quality changes to specific mining operations. Particularly, for regions where there is significant variability in background concentrations of water quality parameters and also other industries or land-users.

The water quality parameters of input water to site processes often well quantified by the industry as it can affect ore processing and metal recovery. As a result, there have been some successful surveys of the quality of water used by the mining industry for processing. For instance, Table 8 show the results of a survey into the operating practices of medium-sized gold mines in Australia, which captured data on processing techniques, reagent and power consumption, raw water consumption and quality as well as tailings management (Sparrow and Woodcock, 1993). Given the level of detail captured by this survey across many mining operations and companies, it suggests that it may be possible to conduct successful surveys for other sectors of the mining industry to develop more rigorous life cycle inventory datasets. Such a research effort would be a significant undertaking in terms of time and effort, however the results would be highly useful for both life cycle assessment practitioners and also the mining industry more generally.

Table 8: Raw water source and quality parameters for medium Australian gold mines in the year 1990. Adapted from Sparrow and Woodcock (1993).

	Raw water		pH	TDS	Na	K	Mg	Ca	Cl	SO4	Other
Mine	**kL/t**	**Source**	**pH**	**ppm**	**ppm**	**ppm**	**ppm**	**ppm**	**ppm**	**ppm**	
Bamboo Creek	-	Bore	-	-	-	-	-	-	-	-	-
Bannockburn	1	-	7.7	900	250	12	75	45	320	170	350ppm HCO_3
Bluebird	-	Potable	-	-	-	-	-	-	-	-	-
Browns Creek	-	Fresh (from pit)	-	-	-	-	-	-	-	-	-
Comet	-	Bore	7.8	3000	300	25	150	100	500	100	20 ppm CO_3
Copperhead	1.32	Bore	7.9	112000	36500	470	3200	2250	66500	2800	90 ppm NO_3
Darlot	0.8	Bore	7.6	1080	280	45	39	57	315	216	5 ppm CO_3 70 ppm NO_3
Davyhurst	1.3	Bore	6.3	43900	12400	195	1760	5404	22500	3500	<1 ppm CO_3 7 ppm NO_3
Enterprise	-	Fresh and bore	6.6	-	48	2	26	19	7	5	225 ppm HCO_3
Fortnum	1	Fresh bore	7	650	100	11	25	20	150	80	<1 ppm CO_3 60ppm NO_3
Fraser	1.5	Saline bore	7.4	116000	25550	-	3400	870	52500	5100	-
Gidgee	1.27	Bore and pits	7.2	1500	-	-	-	-	-	-	-
Golden Crown	-	Underground discharge	7.6	58100	15350	195	1850	1130	30390	1855	<0.3 ppm CO_3
Golden Kilometre	0.85	Saline bore	6.5	100000	55000	-	2000	2000	36000	4000	-
Golden Kilometre	1	Saline bore	6.5	100000	55000	-	2000	2000	36000	4000	-
Goodall	1.32	Dam	3.3	-	10	3	87	23	4	610	-

	Raw water		pH	TDS	Na	K	Mg	Ca	Cl	SO4	Other
Mine	kL/t	Source	pH	ppm	ppm	ppm	ppm	ppm	ppm	ppm	
		Dewater. bores	3.2		11	6	140	29	16	1100	
Greenfields	1.5	W. Ford decline	8.1	30200	8650	61	1500	240	13500	4260	<1 ppm CO_3
Harbour Lights	1	Bore	7.5	2500	400	20	60	60	600	250	60 ppm NO_3
Higginsville	1	Saline bore	5.3	200000	60000	800	7000	500	110000	10500	<500 ppm CO_3
Jubilee	-	Saline bore	3.3	50700	15400	320	2130	170	28200	4360	<2 ppm CO_3 105 ppm SiO_2
Kaltails	-	Saline bore	3.9	150000	65000	230	8700	830	109000	11000	-
Kanowna	-	Bore	6.8	40000- 120000	8000	-	4000	-	-	-	-
Karonie	0.5	Bore	5	181000	52700	295	6470	690	83400	11900	<1 ppm CO_3 8 ppm Fe
Kundana	1	Saline bore	7	238000	85500	215	8000	2300	150000	5000	-
Labouchere	-	Bore	7.4	1200	200	11	55	75	300	200	<1 ppm CO_3 <1 ppm NO_3 1450 µS/cm
Lady Bountiful	1.3	-	6.8	64000	19400	29	2990	820	34000	6600	<1 ppm NO_3
Laverton operations	1.2	-	8.1	10000	2000	8550	1000	2000	70	450	-
Lawlers	1.6	Fresh bore and saline pit	8.3	600	100	8	70	10	140	120	-
Lucky Draw	-	Fresh	-	-	-	-	-	-	-	-	-
Magdala, Wonga	0.35	Mine	7.9	7400	1850	42	340	350	2400	1000	-
Maldon	1.5	Mine	7.1	2000	500	15	200	40	1000	160	450 ppm HCO_3
Marvel Loch	1.79	Saline bore	6.5	23000	6400	160	1000	275	12100	1800	110 ppm HCO_3
Matilda	1.2	Bore	7.8	3500	-	-	-	-	-	-	-
Moline	0.95	Bore and dam return	7	720	100	5	68	82	39	500	-
Mount Gibson	1.6	Bore Pit	7.5 6.4	26000 209000	7450 64000	240 2000	1100 7500	250 750	13500 120000	2000 14300	<5 ppm CO_3 90 ppm SiO_2
Nevoria	-	Saline bore	6.8	68000	11000	200	2500	1000	25000	4000	80 ppm CO_3
Nobles Nob, White Devil	-	Bore and fresh	6.6	8300	1700	85	500	570	3700	1660	400 ppm CO_3 25 ppm NO_3
Ora Banda	-	Saline bore	7.5	44000	11000	-	2600	245	20800	4350	68 ppm SiO_2
Pajingo	1.5	Bore and pits	7	3500	800	13	150	100	1150	300	700 ppm CO_3
Parkes	0.75	Sewage plant Old quarries	6.9 8.2	900 3600	160 600	18 6	19 288	36 159	216 1800	46 300	355 ppm CO_3 275 ppm CO_3
Peak Hill Resources	-	Bore	7.2	850	250	10	40	35	190	100	1 ppm CO_3 550 ppm HCO_3
Polaris	2	Bore and pits	7.3	70000	300000	300	4000	600	55000	6000	40 ppm CO_3
Ravenswood	-	Open cut rainwater	7.5	1000	47	9	57	250	7	730	-
Reedy	1.1	Bore, dam return	9.2	2900	860	8	23	25	120	350	100 ppm CO_3
Sheahan-Grants	1.3	Belubula R. Dam return	8.5 8.5	-	32 1470	6 31	22 23	34 610	39 13550	15 2200	218 ppm CO_3 2000 ppm SCN
Sons of Gwalia	-	Saline bore	7.5	54000	14500	300	1500	1100	24000	3800	-
Tanami	1.43	Bore and pits	7.8	1130	167	48	64	53	165	85	289 ppm CO_3 36 ppm NO_3
Tarmoola	1.2	-	7	1200	340	50	38	70	750	240	1 ppm CO_3
Temora	1.3	Fresh dam & potable bore	6.6	-	10	5	3	2	10	2	30 ppm CO_3
Three Mile Hill	1	Saline bore	6.5	57000	17060	100	2750	140	30400	6800	128 ppm HCO_3
Tower Hill	1.3	Saline bore	7.5- 8.0	80000	10000	4000	-	-	20000	7200	120 ppm HCO_3
Transvaal	1.8	-	6.7	113200	25500	630	3400	870	72500	5100	50 ppm CO_3
Tuckabianna	1.4	Fresh and bore	7.8	675	110	7	20	28	150	90	60 ppm CO_3
White Range	-	Bore	-	-	-	-	-	-	-	-	-
Youanmi	1.22	Fresh bore and saline pit	7.3	1100	275	10	47	54	400	120	-
Zoroastrian, Davyhurst	1.5	Saline bore	6.9	89100	25000	190	5000	515	47600	5200	-

7.3.2. Life-of-Mine Considerations

The hydrology at a mine-site can be significantly altered throughout the life of the project, when compared to the hydrology of the landscape in its predevelopment state. Figure 7.5 shows a basic conceptualisation of how the components of a landscapes hydrology may evolve with the development of an open-cut mining operation.

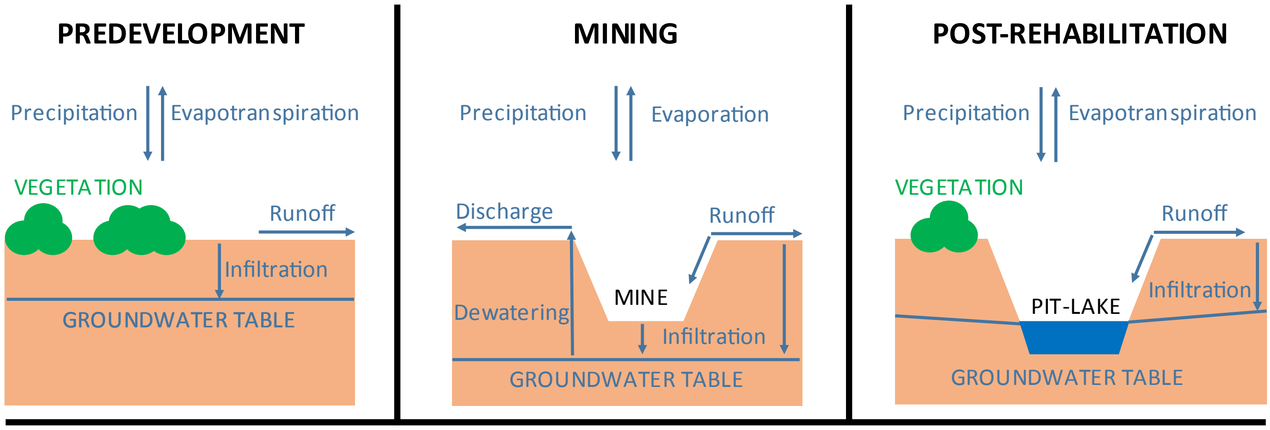

Figure 7.5: Hypothetical evolution of water flows through the life of an open-cut mining operation.

When hydrological contexts allow, pit-lakes may form following the closure of the mine site. Particularly in cases where active dewatering of the groundwater table and aquifers is not continued once mining is ceased. In these cases, the formation and surface height of a pit-lake may be governed by groundwater flows into and/or out of pit, surface runoff into the pit, and the balance of precipitation and evaporation on the pit-lake surface. When evaporation exceeds the rainfall and run-off into a pit, then the pit-lake may act as a long-term sink that depresses local groundwater levels. This may also alter local groundwater flow towards the pit, leading to the long-term accumulation of dissolved solutes in the lake. Alternatively, in cases where the rainfall and surface runoff into the pit exceed evaporation, then the pit-lake may become a long-term source of groundwater recharge that elevates surrounding groundwater levels. McCullough et al. (2013) provides a good overview of possible scenarios for arid regions and the potential influence of mine backfilling practices. As has been reported by Eary and Watson (2009) and Younger (2006), the post-closure water consumption of mining operations from pit-lakes, dams and waste enclosures can represent significant sources of long-term water consumption.

Due to the clearing of vegetation and the compaction of soils by site machinery, there may also be significant alterations in rainfall-runoff coefficients across the site – with associated changes to the fraction of rainfall that infiltrates to groundwater systems or is returned to the atmospheric water cycle through evapotranspiration. Expected evapotranspiration in revegetated areas following site closure may also be altered due to the differences in species composition and maturity

These alterations of a land-scape by mining operations can result in permanent changes to local hydrology that should ideally be considered during the development of life cycle inventory data (Northey et al., 2016). However, currently these issues are not addressed by existing life cycle inventories due to conceptual difficulties and the lack of appropriate modelling and data sources. Further research into the magnitude of post-closure water consumption and the variability between individual mine sites is required before these aspects could adequately be incorporated into life cycle inventory data. This also necessitates further theoretical framing regarding the appropriate post-closure time horizons for assessing mining operations, which may differ depending upon if an attributional life cycle assessment or a consequential life cycle assessment study is performed.

- - - - - PAGE LEFT INTENTIONALLY BLANK - - - - -

8. Conclusions

The use of life cycle assessment based approaches to evaluate the magnitude and impacts of water consumption may provide for a more holistic understanding of how the mining industry interacts with water resources. Over the longer term, this understanding may lead to more informed decision making, improved water resource management outcomes, and a greater contribution by the industry to meeting society's sustainable development objectives. Several opportunities have been identified for the further application of water footprint and life cycle assessment methods in the mining industry (see Chapter 3; Northey et al., 2016). These include:

- Improving understanding and associated decision making processes, through the use of standardised assessment approaches that enable regional and cross-sectoral comparisons of contributions to consumptive and degradative water use impacts.

- Providing avenues for the future development of water efficiency benchmarking schemes for the mining and mineral processing industry, which are able to incorporate data pertaining to local contexts to enable fair and meaningful comparisons of operational performance.

- Developing a more holistic understanding of the benefits and detriments of the various technology and resource management options available for mining and mineral processing operations, particularly through the consideration of indirect supply chain impacts.

- Improving the communication of water related data, through the adoption of standardised approaches that can account for local contexts and improve the quality of cross-sectoral or facility comparisons. This may facilitate improved stakeholder engagement, particularly for companies with production facilities across multiple regions, as the data presented becomes more meaningful and easier to compare.

Throughout these research activities, especially the preparation of Chapter 3 (Northey et al., 2016), a range of methodological and data limitations were identified that have hindered further application of water footprint approaches to the mining industry. Overcoming these limitations would enable a substantial improvement in our collective understanding of how the mining industry as a whole interacts with water resources. The limitations identified include:

- The limitations of existing water use data for the mining industry, which could be improved through the compilation and aggregation of the significant amount of disaggregated data that is available for individual mining operations through various reporting mechanisms (e.g. statutory environmental compliance reporting, corporate sustainability reporting, etc.).

- The limitations of existing life cycle inventory data describing the required energy and material requirements of individual mining operations, which prevents more robust assessment of the 'indirect' or supply chain water footprint associated with the mining industry. This could be improved through improved industry reporting and also the presentation of fully disaggregated datasets by inventory developers, such as the international metal associations.

- Conceptual and practical difficulties in evaluating hydrological impacts throughout the full life cycle of a mine, from the pre-development stages through the active mining phase and then the final rehabilitated or post-closure state of the site. Incorporating these aspects into water footprint or life cycle inventory databases would require standardisation of the time horizons of inventory assessment. Also significant research efforts would be required to translate the detailed outputs of hydrological models of mining and mineral processing operations, which by their nature are highly site specific, into data formats that meet the requirements of inventory datasets.

- A current lack of inclusion of abnormal mine operating conditions, infrastructure failings (e.g. tailings dam collapses) or unexpected pollution events within inventory datasets. This is despite these types of events being commonly considered as some of the major risks to water resources associated with the mining industry. Inclusion of these within inventory datasets would necessitate taking a probabilistic approach to assigning water resource and pollution burdens to individual operations, which would likely be a highly contentious approach for some mining industry stakeholders.

- Conceptual difficulty regarding how inventory datasets could incorporate the possible cumulative hydrological impacts that may occur between mining operations in some regions, such as when increased aquifer dewatering at one mining operation would alter aquifer dewatering requirements at other mining operations. For regions where cumulative impacts may be expected, then inventory development at regional rather than operational boundaries may be a suitable approach to overcoming this conceptual difficulty.

- Existing life cycle impact characterisation procedures and factors may not always be suited for assessing the consumptive water use impacts of the mining industry (see Chapter 5). The accuracy and representativeness of spatial impact characterisation procedures may vary across spatial resolutions and geographical regions. Also, existing characterisation procedures and the hydrological models underpinning these are largely focused on assessing impacts related to the water availability of surface water and inter-connected shallow groundwater systems. However many mining operations physically intersect or abstract water from deep or ‘fossil’ groundwater aquifer systems, for which there is a current lack of suitable impact characterisation methods. Therefore, further methodological development of impact characterisation approaches is required to enable deep or ‘fossil’ groundwater consumption to be evaluated alongside consumption of surface or shallow groundwater resources. Furthermore, advances in the modelling of the global hydrological system and the spatiotemporal distributions of water use should also be pursued to enable more accurate impact characterisation factors to be developed.

Understanding how the mining industry relates to and interacts with water resources is greatly aided by considering the local water and climate contexts in regions containing mineral resources or production facilities. Developing a quantitative understanding of these contexts will enable more accurate assessment of the water consumption and water use impacts associated with particular mine sites or production facilities. Additionally, a quantitative understanding of local contexts may also provide greater understanding of the local risk factors that may influence the development trajectory of mineral resource projects and the mining industry in particular regions.

The spatial distribution of global copper, lead-zinc and nickel resources were evaluated in relation to local water and climate contexts, including spatial water use impact characterisation factors that are available for use in life cycle assessment (Chapter 4; Northey et al., 2017a). Beyond directly providing data to improve future life cycle impact characterisation studies for these metals, the analysis also revealed how knowledge of local water and climate contexts can be used to inform an understanding of industry risks. The end result being that greater quantitative rigour can now be provided to justify some of the common narratives in the mining industry regarding levels of exposure to water stress and scarcity issues. From this study a few insights were revealed:

- Copper resources are, on average, located in regions of higher water stress, scarcity or risk than either nickel or lead-zinc resources. These results were broadly consistent across multiple spatial indices that describe various aspects of local water contexts.

- Operating copper mines are located in regions with higher water stress than undeveloped copper deposits, which indicates that future copper supply may be less likely to place excessive burden on water resources than the current copper supply chain.

- Regions containing copper resources are also more likely to be reclassified to different Köppen-Geiger climate classifications as a result of climate changes to 2100, when compared with regions containing lead-zinc or nickel resources. Although it is emphasised that the results for changing Köppen-Geiger climate classifications presented in chapter 4 (Northey et al., 2017a) are highly uncertain due to the limitations of global climatic models when producing results for individual regions.

- There are a range of plausible mechanisms through which climatic changes can either exacerbate or reduce risks at individual mining operations. Some mining regions and projects have already experienced the impacts of changing climates. Therefore, it is encouraged that individual mining operations and companies proactively plan for expected climatic changes in their region when undertaking site hydrological studies, environmental impact assessment, or risk management and mitigation studies. Given the hydrologic variability that some mining operations are already exposed to, there is strong capacity within the industry to evaluate climate change adaptation requirements and to implement effective solutions to reduce risk.

Additionally, the spatial distribution of production for 25 mined commodities were evaluated in relation two water use impact characterisation factors: the Water Stress Index (WSI; Pfister et al., 2009), and the Available Water Remaining (AWaRe; Boulay et al., 2017) for non-agricultural water use. Production weighted averages developed using watershed AWaRe and WSI factors were compared with the existing national average WSI and AWaRe factors to determine the potential influence of spatial aggregation on the results of water use impact assessment. Several interesting conclusions were reached from the study (Chapter 5):

- The use of the existing national average WSI and AWaRe factors are likely to, on average, overestimate the water use impacts of the mining industry when compared to impact assessment using watershed factors. It was found that there was a bias of national average AWaRe factors for non-agricultural water use to lead to an overestimation of impacts for 60% of mining operations, when compared to watershed based assessment. The use of national average WSI displayed a stronger bias and would lead to overestimation of impacts for 67% of mining operations, when compared to watershed based assessment.

- The bias for national average factors to overestimate impacts of the mining industry relative to assessment using watershed factors is due to differences in the spatial distribution of mine production and water consumption. The existing national average WSI and AWaRe factors are developed by weighting (sub-) watershed factors according to the spatial distribution of water withdrawals or consumption in the region. Also the derivation of the watershed WSI and AWaRe factors is also based upon the water availability and the withdrawals or demand that occurs in the region. Given that the mining industry is only a minor consumer of water when compared to other industries such as agriculture, the national average factors may not always reflect the conditions where mines are located within the country.

- Therefore, the development of life cycle inventory datasets for the mining industry that are spatially aggregated to watershed rather than national boundaries is encouraged to enable the use of watershed impact characterisation factors. Where this is not possible due to confidentiality or data limitations, then impact characterisation factors specific to the inventory datasets could be recalculated by weighting watershed values according to the spatial distribution of mine production or water consumption. National factors recalculated to reflect the spatial production distribution for 25 mined commodities are provided in appendix Table A. 6 to Table A. 30.

It was also shown that there is substantial reporting of water use data by the mining industry (Chapter 7). The communication of this data is increasingly becoming standardised overtime, due to the development of water accounting and reporting guidelines by organisations such as the Minerals Council of Australia (MCA, 2014) and the International Council for Metals and Mining (ICMM, 2017). Although this water use data exists in the public domain, considerable time and effort is required to compile and aggregate this data into useful datasets. However, the development of these forms of data sources have the potential to significantly improve the quality and representativeness of data available for use in water footprint and life cycle inventories. With further methodological development, these data sources may also provide for a fair and meaningful way of comparing the water use efficiency of mining operations. Therefore further research efforts to compile and assess the mining industries reported water use data is strongly encouraged.

Finally, I would like to thank the reader for making it this far (or skipping straight to the conclusions). If any of the topics or studies presented in this thesis piqued your interest then please feel free to reach out to me for further discussion. If you're reading this in the near future, then I may be contactable through my current institutional email addresses: stephen.northey@monash.edu or stephen.northey@csiro.au. If my career has moved on, then you could try my personal email address: stephen.northey@gmail.com. And for anyone reading in the (hopefully) far distant future when I am no longer be around, I hope that society has worked through the various challenges facing us in the early 21st century (climate change, water resource degradation, etc.) and that approaches that holistically consider the environmental trade-offs of decision making, such as life cycle assessment, are commonplace and routine.

References

Agricola, G. (1556). De Re Metallica. Translated by H.C. Hoover & L.H. Hoover (1950), Dover Publications Inc., New York. ISBN: 0486600068.

Aivazidou, E., Tsolakis, N., Iokovou, E., Vlachos, D. (2016). The emerging role of water footprint in supply chain management: A critical literature synthesis and a hierarchical decision-making framework. Journal of Cleaner Production, 137, pp. 1018-1037. http://dx.doi.org/10.1016/j.jclepro.2016.07.210

Akcil, A., Koldas, S. (2006). Acid Mine Drainage (AMD): causes, treatment and case studies. Journal of Cleaner Production, 14, pp. 1139-1145. http://dx.doi.org/10.1016/j.jclepro.2004.09.006

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T., Siebert, S. (2003a). Development and testing of the WaterGAP 2 global model of water use and availability. Hydrological Sciences Journal, 48(3), pp. 317-337. http://dx.doi.org/10.1623/hysj.48.3.317.45290

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T., Siebert, S. (2003b). Global estimates of water withdrawals and availability under current and future "business-as-usual" conditions. Hydrological Sciences Journal, 48(3), pp. 339-348. http://dx.doi.org/10.1623/hysj.48.3.339.45278

Alcamo, J., Flörke, M., Märker, M. (2007). Future long-term changes in global water resources driven by socio-economic and climatic changes. Hydrological Sciences Journal, 52(2), pp. 247-275. http://dx.doi.org/10.1623/hysj.52.2.247

Allen, T. (1998). Watersheds and problemsheds: Explaining the absence of armed conflict over water in the Middle East. Middle East Review of International Affairs, 2(1), pp. 49-51.

Ali, S.H., Giurco, D., Arndt, N., Nickless, E., Brown, G., Deemetriades, A., Durrheim, R., Enriquez, M.A., Kinnaird, J., Littleboy, A., Meinert, L.D., Oberhänsli, R., Salem, J., Schodde, R., Schneider, G., Vidal, O., Yakovleva, N. (2017). Mineral supply for sustainable development requires resource governance. Nature, 543(7645), pp. 367-372. http://dx.doi.org/10.1038/nature21359

Althaus, H.-J., Classen, M. (2005). Life Cycle Inventories of Metals and Methodological Aspects of Inventorying Material Resources in ecoinvent (7pp). International Journal of Life Cycle Assessment, 10(1), pp. 43-49. http://dx.doi.org/10.1065/lca2004.11.181.5

Amezaga, J.M., Rötting, T.S., Younger, P.L., Nairn, R.W., Noles, A.-J., Oyarzún, R., Quintanilla, J. (2011). A rich vein? Mining and the pursuit of sustainability. Environmental Science & Technology, 45, pp. 21-26. http://dx.doi.org/10.1021/es101430e

Amos, R.T., Blowes, D.W., Bailey, B.L., Sego, D.C., Smith, L., Ritchie, A.I.M. (2015). Waste-rock hydrogeology and geochemistry. Applied Geochemistry, 57, pp. 140-156. http://dx.doi.org/10.1016/j.apgeochem.2014.06.020

Anawar, H.M. (2013) Impact of climate change on acid mine drainage generation and contaminant transport in water ecosystems of semi-arid and arid mining areas. Physics and Chemistry of the Earth, Parts A/B/C 58–60, pp. 13-21. http://dx.doi.org/10.1016/j.pce.2013.04.002

Ansorge, L., Beránkova, T. (2017). LCA Water Footprint AWARE Characterization Factor Based on Local Specific Conditions. European Journal of Sustainable Development, 6(4), pp. 13-20. http://dx.doi.org/10.14207/esjd.2017.v6n4p13

Armitage, A., Desautels, P., Zurowski, G., Pennstrom, W., Malbran, F., Fitch, R. (2011). Preliminary Economic Assessment Update – Technical Report for the Malku Khota Project. Department of Potosi, Bolivia. Prepared for South American Silver Corp., 10 May 2011, 181p.

Arndt, N.T., Fontboté, Hedenquist, J.W., Kesler, S.E., Thompson, J.F.H., Wood, D.G. (2017). Future Global Mineral Resources. Geochemical Perspectives, 6(1), April 2017. http://dx.doi.org/10.7185/geochempersp.6.1

Askham, T.M., Van der Poll, H.M. (2017). Water Sustainability of Selected Mining Companies in South Africa. Sustainability, 9, 957, 19p. http://dx.doi.org/10.3390/su9060957

Awuah-Offei, K., Adekpedjou, A. (2011). Application of life cycle assessment in the mining industry. International Journal of Life Cycle Assessment, 16, pp. 82-89. http://dx.doi.org/10.1007/s11367-010-0246-6

Azam, S., Li, Q. (2010). Tailings Dam Failures: A review of the last one hundred years. Geotechnical News, 28(4), pp. 50-54.

Azimi, Y., Osanloo, M., Esfahanipour, A. (2013). An uncertainty based multi-criteria ranking system for open pit mining cut-off grade strategy selection. Resources Policy, 38, pp. 212-223. http://dx.doi.org/10.1016/j.resourpol.2013.01.004

Bao, C., Mortazavi-Naeini, M., Northey, S., Tarnopolskaya, T., Monch, A., Zhu, Z. (2013). Valuing flexible operating strategies in nickel production under uncertainty. 20th International Congress on Modelling and Simulation, Adelaide, Australia, 1-6 December 2013, pp. 1426-1432.

Balanay, R.M., Halog, A. (2016). Promoting life cycle thinking for sustainability in the mining sector of the Philippines. International Journal of Life Cycle Assessment, 11p. https://doi.org/10.1007/s11367-016-1105-x

Ballantyne, G.R., Powell, M.S. (2014). Benchmarking comminution energy consumption for the processing of copper and gold ores. Minerals Engineering, 65, pp. 109-114. http://dx.doi.org/10.1016/j.mineng.2014.05.017

Barakat, M.A. (2011)-. Review Article: New trends in removing heavy metals from industrial wastewater. Arabian Journal of Chemistry, 4, pp. 361–377. http://dx.doi.org/10.1016/j.arabjc.2010.07.019

Bartos, P.J. (2007). Is mining a high-tech industry?: Investigations into innovation and productivity advance. Resources Policy, 32(4), pp. 149-158. http://dx.doi.org/10.1016/j.resourpol.2007.07.001

Bayart, J.-B., Bulle, C., Deschênes, L., Margni, M., Pfister, S., Vince, F., Koehler, A. (2010). A framework for assessing off-stream freshwater use in LCA. International Journal of Life Cycle Assessment, 15, pp. 439-453. http://dx.doi.org/10.1007/s11367-010-0172-7

Bayart, J.-B., Worbe, S., Grimaud, J., Aoustin, E. (2014). The Water Impact Index: a simplified single-indicator approach for water footprinting. International Journal of Life Cycle Assessment, 19(6), pp. 1336-1344. http://dx.doi.org/10.1007/s11367-014-0732-3

Bebbington, A., Williams, M. (2008). Water and Mining Conflicts in Peru. Mountain Research and Development, 28(3), pp. 190-195. http://dx.doi.org/10.1659/mrd.1039

Bebbington, A., Hinojosa, L., Bebbington, D.H., Burneo, M.L., Warnaars, X. (2008). Contention and ambiguity: Mining and the possibilities of development. Development and Change, 39(6), pp. 887-914. http://dx.doi.org/10.1111/j.1467-7660.2008.00517.x

Bekesi, G., Tyler, M., Niejalke, D., Waterhouse, J. (2013). Improved regulation of water supply for the Olympic Dam mine, South Australia. 6th International Mine Water Association Conference, Golden, Colorado, USA, pp. 29-34.

Benner, S.G., Blowes, D.W., Ptacek, C.J., Mayer, K.U. (2002). Rates of sulfate reduction and metal sulfide precipitation in a permeable reactive barrier. Applied Geochemistry, 17(3), pp. 301-320. http://dx.doi.org/10.1016/S0883-2927(01)00084-1

Berger, M., Finkbeiner, M. (2013). Methodological challenges in volumetric and impact-oriented water footprints. Journal of Industrial Ecology, 17(1), pp. 79-89. http://dx.doi.org/10.1111/j.1530-9290.2012.00495.x

Berger, M., van der Ent, R., Eisner, S., Bach, S., Finkbeiner, M. (2014). Water Accounting and Vulnerability Evaluation (WAVE): Considering Atmospheric Evaporation Recycling and the Risk of Freshwater Depletion in Water Footprinting. Environmental Science & Technology, 48, pp. 4521-45428. http://dx.doi.org/10.1021/es404994t

BHPB. (2013). BHP Billiton Approves Escondida Water Supply Project. BHP Billiton (BHPB), News Release to the Australian Stock Exchange (ASX), 25 July 2013.

BHPB. (2014). Olympic Dam, Great Artesian Basin Wellfields Report, 1 July 2013-30 June 2014. BHP Billiton (BHPB), Report No. ODENV053. Available via the South Australian Resource Information Geoserver: http://sarig.pir.sa.gov.au/ (accessed 04-07-2016)

Biedermann, R.P. (2014). China’s rare earth sector – between domestic consolidation and global hegemony. International Journal of Emerging Markets, 9(2), pp. 276-293. http://dx.doi.org/10.1108/IJoEM-05-2013-0080

Bleiwas, D.I. (2012a). Estimated Water Requirements for the Conventional Flotation of Copper Ores. U.S. Geological Survey, Open-File Report 2012-1089, 13p. https://pubs.usgs.gov/of/2012/1089/

Bleiwas, D.I. (2012b). Estimated Water Requirements for Gold Heap-Leach Operations. U.S. Geological Survey, Open-File Report 2012-1085, 17p. https://pubs.usgs.gov/of/2012/1085/

Bolton, B.R. (Editor). (2009). The Fly River, Papua New Guinea: Environmental Studies in an Impacted Tropical River System. Developments in Earth & Environmental Sciences, 9, Elsevier.

Bonnafous, L., Lall, U., Siegel, J. (2016; 2017). A water risk index for portfolio exposure to climatic extremes: conceptualization and an application to the mining industry. Hydrology and Earth System Sciences Discussions, 21, pp. 2075-2106. https://doi.org/10.5194/hess-21-2075-2017

Bösch, M.E., Hellweg, S., Huijbregts, M.A.J., Frischknecth, R. (2007). Applying cumulative exergy demand (CExD) indicators to the ecoinvent database. International Journal of Life Cycle Assessment, 12(3), pp. 181-190. http://dx.doi.org/10.1065/lca2006.11.282

Bouffard, S.C. (2005). Review of agglomeration practice and fundamentals in heap leaching. Mineral Processing and Extractive Metallurgy Review: An International Journal, 26(3-4), pp. 233-294. http://dx.doi.org/10.1080/08827500590944009

Boulay, A.-M., Bouchard, C., Deschênes, L., Margni, M. (2011). Categorizing water for LCA inventory. International Journal of Life Cycle Assessment, 16, pp. 639-651. http://dx.doi.org/10.1007/s11367-011-0300-z

Boulay, A.-M., Hoekstra, A.Y., Vionnet, S. (2013). Complementarities of Water-Focused Life Cycle Assessment and Water Footprint Assessment. Environmental Science & Technology, 47, pp. 11926-11927. http://dx.doi.org/10.1021/es403928f

Boulay, A.-M., Motoshita, M., Pfister, S., Bulle, C., Muñoz, I., Franceschini, H., Margni, M. (2015a). Analysis of water use impact assessment methods (part A): evaluation of modelling choices based on a quantitative comparison of scarcity and human health indicators. International Journal of Life Cycle Assessment, 20, pp. 139-160. http://dx.doi.org/10.1007/s11367-014-0814-2

Boulay, A-M., Bare, J., De Camillis, C., Döll, P., Gassert, F., Gerten, D., Humbert, S., Inaba, A., Itsubo, N., Lemoine, Y., Margni, M., Motoshita, M., Núñez, M., Pastor, A. V., Ridoutt, B., Schencker, U., Shirakawa, N., Vionnet, S., Worbe, S., Yoshikawa, S., Pfister, S. (2015b). Consensus building on the development of a stress-based indicator for LCA-based impact assessment of water consumption: outcome of the expert workshops. International Journal of Life Cycle Assessment, 20(5), pp. 577-583. http://dx.doi.org/10.1007/s11367-015-0869-8

Boulay, A.-M., Pfister, S., Motoshita, M., Schenker, U., Benini, L., Gheewala, S.H., de Figueirêdo, M.C.B., Harding, K. (2016). Chapter 5a: Water scarcity. In Global Guidance for Life Cycle Impact Assessment Indicators; UNEP SETAC Life Cycle Initiative, 2016. http://www.lifecycleinitiative.org/applying-lca/lcia-cf/

Boulay, A-M., Bare, J., Benini, L., Berger, M., Lathuillière, M., Manzardo, A., Margni, M., Motoshita, M., Núñez, M., Pastor, A.V., Ridoutt, B., Oki, T., Worbe, S., Pfister, S. (2017). The WULCA consensus characterization model for water scarcity footprints: assessing impacts of water consumption based on available water remaining (AWARE). International Journal of Life Cycle Assessment, 20, pp. 139-160. http://dx.doi.org/10.1007/s11367-017-1333-8

BPEMM. (1999). Water Management, Best Practice Environmental Management in Mining (BPEMM). Environment Australia, Australian Government. 99p.

British Geological Survey. (2016). World Mineral Production 2010-2014. British Geological Survey, Keyworth, Nottingham, United Kingdom. http://www.bgs.ac.uk/mineralsuk/statistics/wms.cfc?method=searchWMS (Accessed 16 March 2017)

Buxmann, K., Koehler, A., Thylmann, D. (2016). Water scarcity footprint of primary aluminium. International Journal of Life Cycle Assessment, 21(11), pp. 1605-1615. http://dx.doi.org/10.1007/s11367-015-0997-1

Caron, J., Durand, S., Asselin, H. (2016). Principles and criteria of sustainable development for the mineral exploration industry. Journal of Cleaner Production, 199, pp. 215-222. http://dx.doi.org/10.1016/j.jclepro.2016.01.073

Castro, S. (2012). Challenges in Flotation of Cu-Mo Sulfide Ores in Sea Water. Water in Mineral Processing – Proceedings of the First International Symposium (Eds. Drelich et al.), Society for Mining, Metallurgy, and Exploration, pp. 29-40.

CDP. (2013). Metals & Mining: a sector under water pressure. Analysis for institutional investors of critical issues facing the industry. Carbon Disclosure Project, United Kingdom, July 2013, 20p.

CDP. (2017). Carbon Disclosure Project (CDP), United Kingdom. https://www.cdp.net (last accessed 8-11-2017)

Chambers, D. (2016). Post-Mount Polley: Tailings Dam Safety in British Columbia. March 2016. Earthworks, Washington, D.C., USA. www.earthworksaction.org/files/publications/Post-MountPolleyTailingsDamSafety.pdf (accessed 04-07-2016)

Chapagain, A.K., Hoekstra, A.Y. (2004). Water Footprint of Nations. UNESCO-IHE Institute for Water Education, The Netherlands.

Chenoweth, J., Hadjikakou, M., Zoumides, C. (2014). Quantifying the human impact on water resources: a critical review of the water footprint concept. Hydrology and Earth System Sciences, 18, pp. 2325-2342. http://dx.doi.org/10.5194/hess-18-2325-2014

Ciacci, L., Nuss, P., Reck, B., Werner, T., Graedel, T. (2016). Metal Criticality Determination for Australia, the US, and the Planet—Comparing 2008 and 2012 Results. Resources, 5(4), 29. http://dx.doi.org/10.3390/resources5040029

Cisternas, L.A., Gálvez, E.D. (2017). The use of seawater in mining. Mineral Processing and Extractive Metallurgy Review, *volume unassigned*, 16p. http://dx.doi.org/10.1080/08827508.2017.1389729

Clark, J.L., Christensen, D., Lum, B. (2011). Acid rock drainage source control opportunities at the Red Dog Zinc and Lead Mine, USA. Mine Closure 2011, Alberta, Canada, pp. 341-348.

Classen, M., Althaus, H.-J., Blaser, S., Tuchschmid, M., Jungbluth, N., Doka, G., Faist Emmenegger, M., Scarnhorst, W. (2007). Life Cycle Inventories of Metals. Final report ecoinvent data v2.0, No 10. EMPA Dübendorf, Swiss Centre for Life Cycle Inventories, Dübendorf, CH.

Clift, R., Druckman, A. (Editors). (2016). Taking stock of industrial ecology. Springer, 362p. http://dx.doi.org/10.1007/978-3-319-20571-7

Coalition for Eco-Efficient Comminution (CEEC). (2017). Energy Curve Program. https://www.ceecthefuture.org/energy-curve-program/ (last accessed 22 May 2017)

COCHILCO. (2014a). Hydrological consumption in copper mining for 2014. Comisión Chilena del Cobre (COCHILCO), Government of Chile, 02/2015.

COCHILCO. (2014b). Comisión Chilena del Cobre, Análisis de Variables Clav es para la Sustentabilidad de la Minería en Chile. Comisión Chilena del Cobre (COCHILCO), Government of Chile, 300p. [in Spanish].

COCHILCO. (2015). Water consumption projection in copper mining through 2026. Comisión Chilena del Cobre (COCHILCO), Government of Chile.

Cote, C.M., Cummings, J., Moran, C.J., Ringwood, D. (2012). Water accounting in mining and minerals processing. In: Godfrey, J.M., Chalmers, K. (Eds.). Water Accounting – International Approaches to Policy and Decision-making. Edward Elgar, Cheltenham, UK, pp. 91-105.

Crowson, P. (2003). Mine size and the structure of costs. Resources Policy, 29, pp. 15-36. http://dx.doi.org/10.1016/j.resourpol.2004.04.002

Crowson, P.C.F. (2011). Mineral reserves and future minerals availability. Mineral Economics, 24, pp. 1-6. http://dx.doi.org/10.1007/s13563-011-0002-9

Crowson, P. (2012). Some observations on copper yields and ore grades. Resources Policy, 37, pp. 59-72. http://dx.doi.org/10.1016/j.resourpol.2011.12.004

Currell, M.J., Werner, A.D., McGrath, C., Webb, J.A., Berkman, M. (2017). Problems with the application of hydrogeological science to regulation of Australian mining projects: Carmichael Mine and Doongmabulla Springs. Journal of Hydrology, 548, pp. 674-682. http://dx.doi.org/10.1016/j.jhydrol.2017.03.031

Daigo, I., Osako, S., Adachi, Y., Matsuno, Y. (2014). Time-series analysis of global zinc demand associated with steel. Resources, Conservation and Recycling, 82, pp. 35-40. http://dx.doi.org/10.1016/j.resconrec.2013.10.013

Danoucaras, A.N., Woodley, A.P., Moran, C.J. (2014). The robustness of mine water accounting over a range of operating contexts and commodities. Journal of Cleaner Production, 84, pp. 727-735. http://dx.doi.org/10.1016/j.jclepro.2014.07.078

Davies, M., 2011. Filtered dry stacked tailings – The fundamentals. Proceedings of Tailings and Mine Waste 2011, Vancouver, Canada, Nov 6-9, 9p.

Davis, R., Franks, D.M. (2014). Costs of Company-Community Conflict in the Extractive Sector. Corporate Social Responsibility Initiative Report No. 66, Harvard Kennedy School, Cambridge, MA, United States.

Da Rosa, C.D., Lyon, J.S., Hocker, P.M. (1997). Golden Dreams, Poisoned Streams: How reckless mining pollutes America's waters, and how we can stop it. Mineral Policy Center, Washington, D.C., United States, 269p.

Dewulf, J., Benini, L., Mancini, L., Sala, S., Blengini, G.A., Ardente, F., Recchioni, M., Maes, J., Pant, R., Pennington, D. (2015). Rethinking the Area of Protection “Natural Resources” in Life Cycle Assessment. Environmental Science & Technology, 49, pp. 5310-5317. http://dx.doi.org/10.1021/acs.est.5b00734

Dimitrakopoulos, R.G., Sabour, S.A.A. (2007). Evaluating mine plans under uncertainty: Can the real options make a difference? Resources Policy, 32, pp. 116-125. http://dx.doi.org/10.1016/j.resourpol.2007.06.003

Dogramaci, S., Firmani, G., Hedley, P., Skrzypek, G., Grierson, P.F. (2015). Evaluating recharge to an ephemeral dryland stream using a hydraulic model and water, chloride and isotope mass balance. Journal of Hydrology, 521, pp. 520-532. http://dx.doi.org/10.1016/j.jhydrol.2014.12.017

Dold, B. (2014). Evolution of Acid Mine Drainage Formation in Sulphidic Mine Tailings. Minerals, 4, pp. 621-641. http://dx.doi.org/10.3390/min4030621

Drielsma, J.A., Russell-Vaccari, A.J., Drnek, T., Brady, T., Weihed, P., Mistry, M., Simbor, L.P. (2016). Mineral resources in life cycle impact assessment – defining the path forward. International Journal of Life Cycle Assessment, 21, pp. 85-105. http://dx.doi.org/10.1007/s11367-015-0991-7

Durán, A.P., Rauch, J., Gaston, K.J. (2013). Global spatial coincidence between protected areas and metal mining activities. Biological Conservation, 160, 272-278. http://dx.doi.org/10.1016/j.biocon.2013.02.003

Eary, L.E., Watson, A. (2009). The post-mining water balance. Mining Technology, 118(3/4), pp. 212-219. http://dx.doi.org/10.1179/174328610X12682159815064

Echeverri, X., Restrepo, O. (2016). Estimated Water Footprint in the Limestone Quarry Opencast – Rioclaro Cement Plant, Cementos Argos – Colombia. Water in Mining 2016 (Eds. Wiertz and McIntyre), Proceedings of the 5th International Congress on Water Management in Mining, Gecamin, 18-20 May 2016, Santiago, Chile, pp. 244-256.

Edraki, M., Baumgartl, T., Manlapig, E., Bradshaw, D., Franks, D.M., Moran, C.J. (2014). Designing mine tailings for better environmental, social and economic outcomes: a review of alternative approaches. Journal of Cleaner Production, 84, pp. 411-420. http://dx.doi.org/10.1016/j.jclepro.2014.04.079

E-PRTR. (2017). European Pollutant Release and Transfer Register (E-PRTR). European Environment Agency, European Union. http://prtr.ec.europa.eu

Elshkaki, A., Graedel, T.E. (2013). Dynamic analysis of the global metals flows and stocks in electricity generation technologies. Journal of Cleaner Production, 59, pp. 260-273. http://dx.doi.org/10.1016/j.jclepro.2013.07.003

Elshkaki, A., Graedel, T.E., Ciacci, L., Reck, B. (2016). Copper demand, supply, and associated energy use to 2050. Global Environmental Change, 39, pp. 305-315. http://dx.doi.org/10.1016/j.gloenvcha.2016.06.006

Elshkaki, A., Reck, B.K., Graedel, T.E. (2017). Anthropogenic nickel supply, demand, and associated energy and water use. Resources, Conservation & Recycling, 125, pp. 300-307. http://dx.doi.org/10.1016/j.resconrec.2017.07.002

Ercin, A.E., Hoekstra, A.Y. (2014). Water footprint scenarios for 2050: A global analysis. Environment International, 64, pp. 71-82. http://dx.doi.org/10.1016/j.envint.2013.11.019

Erguler Z.A., Ulusay, R. (2009). Water-induced variations in mechanical properties of clay-bearing rocks. International Journal of Rock Mechanics and Mining Sciences, 46(2), pp. 355-370. http://dx.doi.org/10.1016/j.ijrmms.2008.07.002

EU Commission. (2014). Report on Critical Raw Materials for the EU, Report of the Ad hoc Working Group on defining critical raw materials, EU Commission, May 2014, 41p.

Fang, K., Heijungs, R. (2015). Investigating the inventory and characterization aspects of footprinting methods: lessons for the classification and integration of footprints. Journal of Cleaner Production, 108, pp. 1028-1036. http://dx.doi.org/10.1016/j.jclepro.2015.06.086

Flörke, M., Kynast, E., Bärlund, I., Eisner, S., Wimmer, F., Alcamo, J. (2013). Domestic and industrial water uses of the past 60 years as a mirror of socio-economic development: A global simulation study. Global Environmental Change, 23(1), pp. 144-156. http://dx.doi.org/10.1016/j.gloenvcha.2012.10.018

Fonseca, A., McAllister, M.L., Fitzpatrick, P. (2013). Measuring what? A comparative anatomy of five mining sustainability frameworks. Minerals Engineering, 46-47, pp. 180-186. http://dx.doi.org/10.1016/j.mineng.2013.04.008

Fonseca, A., McAllister, M.L., Fitzpatrick, P. (2014). Sustainability reporting among mining corporations: a constructive critique of the GRI approach. Journal of Cleaner Production, 84, pp. 70-83. http://dx.doi.org/10.1016/j.jclepro.2012.11.050

Ford, J.D., Pearce, T., Prno, J., Duerden, F., Berrang Ford, L., Beaumier, M., Smith, T. (2010). Perceptions of climate change risks in primary resource use industries: a survey of the Canadian mining sector. Regional Environmental Change, 10, pp. 65-81. http://dx.doi.org/10.1007/s10113-009-0094-8

Ford, J.D., Pearce, T., Prno, J., Duerden, F., Berrang Ford, L., Smith, T.R., Beaumier, M. (2011). Canary in a coal mine: perceptions of climate change risks and response options among

Canadian mine operations. Climatic Change, 109, pp. 399-415. http://dx.doi.org/10.1007/s10584-011-0029-5

Franks, D.M., Brereton, D., Moran, C.J. (2010). Managing the cumulative impacts of coal mining on regional communities and environments in Australia. Impact Assessment and Project Appraisal, 28(4), pp. 299-312. http://dx.doi.org/10.3152/146155110X12838715793129

Franks, D.M., Boger, D.V., Côte, C.M., Mulligan, D.R. (2011). Sustainable development principles for the disposal of mining and mineral processing wastes. Resources Policy, 36, pp. 114-122. http://dx.doi.org/10.1016/j.resourpol.2010.12.001

Franks, D.M., Brereton, D., Moran, C.J. (2013). The cumulative dimensions of impact in resource regions. Resources Policy, 38(4), pp. 640-647. https://doi.org/10.1016/j.resourpol.2013.07.002

Frenzel, M., Kullik, J., Reuter, M.A., Gutzmer, J. (2017). Raw material 'criticality' – sense or nonsense? Journal of Physics D: Applied Physics, 50(12), 18p. http://dx.doi.org/10.1088/1361-6463/aa5b64

Frischknecht, R., Jungbluth, N., Althaus, H.-J., Doga, G., Dones, R., Hischier, R., Hellweg, S., Nemecek, T., Rebitzer, G., Spielmann, M. (2004). Overview and Methodology. Final report ecoinvent 2000, No. 1. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.

Frischknecht, R., Fantke, P., Tschümperlin, L., Niero, M., Antón, A., Bare, J., Boulay, A.-M., Cherubini, F., Hauschild, M.Z., Henderson, A., Levasseur, A., McKone, T.E., Michelsen, O., Milà I Canals, L., Pfister, S., Ridoutt, B., Rosenbaum, R.K., Verones, F., Vigon, B., Jolliet, O. (2016). Global guidance on environmental life cycle impact assessment indicators: progress and case study. International Journal of Life Cycle Assessment, 21, pp. 429-442. http://dx.doi.org/10.1007/s11367-015-1025-1

Gao, L., Barrett, D., Chen, Y., Zhou, M., Cuddy, S., Paydar, Z., Renzullo, L. (2014). A systems model combining process-based simulation and multi-objective optimisation for strategic management of mine water. Environmental Modelling & Software, 60, pp. 250-264. http://dx.doi.org/10.1016/j.envsoft.2014.06.020

Gao, L., Hou, C., Chen, Y., Barrett, D., Mallants, D., Li, W., Liu, R. (2016). Potential for mine water sharing to reduce unregulated discharge. Journal of Cleaner Production, 131, pp. 133-144. http://dx.doi.org/10.1016/j.jclepro.2016.05.061

Gao, L., Bryan, B.A., Liu, J. Li, W., Chen, Y., Liu, R., Barrett, D. (2017). Managing too little and too much water: Robust mine-water management strategies under variable climate and mine conditions. Journal of Cleaner Production, 162, pp. 1009-1020. https://doi.org/10.1016/j.jclepro.2017.06.101

Geiger, R. (1954). Klassifikation der Klimate nach W. Köppen.–Landolt-Börnstein: Zahlenwerte und Funktionen aus Physik, Chemie, Astronomie, Geophysik und Technik,(alte Serie), Band III (Astronomie und Geophysik), pp. 603-607. Berlin (Springer).

Gemechu, E.D., Helbig, C., Sonnemann, G., Thorenz, A., Tuma, A. (2015). Import-based Indicator for the Geopolitical Supply Risk of Raw Materials in Life Cycle Sustainability Assessments. Journal of Industrial Ecology, 201(1), pp. 154-165. http://dx.doi.org/10.1111/jiec.12279

GEMI Local Water Tool. (2016); http://gemi.org/localwatertool

Genex Power. (2016). The Kidston Hydro Pumped Storage Project. Genex Power Limited, Sydney, Australia. www.genexpower.com.au/the-kidston-hydro-project.html (accessed 04-07-2016)

Gerst, M.D. (2008). Revisiting the Cumulative Grade-Tonnage Relationship for Major Copper Ore Types. Economic Geology, 103, pp. 615-628. http://dx.doi.org/10.2113/gsecongeo.103.3.615

Ghorbani, Y., Kuan, S.H. (2016). A review of sustainable development in the Chilean mining sector: past, present and future. International Journal of Mining, Reclamation and Environment. http://dx.doi.org/10.1080/17480930.2015.1128799

Gibbons, T. (2005). International cyanide management code. Developments in Mineral Processing, 15, pp. 182-199. http://dx.doi.org/10.1016/S0167-4528(05)15008-X

Giurco, D., Cooper, C. (2012). Mining and sustainability: asking the right questions. Minerals Engineering, 29, pp. 3-12. http://dx.doi.org/10.1016/j.mineng.2012.01.006

Giurco, D., Mohr, S., Mudd, G., Mason, L., Prior, T. (2012a). Resource Criticality and Commodity Production Projections. Resources, 1, pp. 23-33. http://dx.doi.org/10.3390/resources1010023

Giurco, D., Prior, T., Mason, L., Mohr, S., Mudd, G. (2012b). Life-of-resource sustainability considerations for mining. Australian Journal of Civil Engineering, 10(1), pp. 47-56. http://dx.doi.org/10.7158/C11-687.2012.10.1

Glaister, B.J., Mudd, G.M. (2010). The environmental costs of platinum-PGM mining and sustainability: Is the glass half-full or half-empty? Minerals Engineering, 23, pp. 438-450. http://dx.doi.org/10.1016/j.mineng.2009.12.007

Gleeson, T., Wada, Y., Bierkens, M.F.P., van Beek, L.P.H. (2012). Water balance of global aquifers revealed by groundwater footprint. Nature, 488, pp. 197-200. http://dx.doi.org/10.1038/nature11295

Goedkoop, M., Heijungs, R., Huijbregts, M., De Schryver, A., Strujis, J., van Zelm, R. (2009). ReCiPe 2008: a Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the endpoint Level, first ed. Ruimte en Mileeu, Ministerie van Volkshuishvesting, Ruimtelijke Ordening en Milieubeheer, The Netherlands.

Google Earth. (2016). Satellite image of Cowal Gold Mine, NSW, Australia. Image: 33.64° S, 147.39° E, Eye Altitude 7.26 km, Imagery Date 3/12/2013, DigitalGlobe 2016. Accessed via Google Earth v. 6.0.3.2197.

Goonan, T.G. (2004). Flows of selected materials associated with world copper smelting. U.S. Geological Survey, Open-file report 2004-1395.

Gordon, R.B., Bertram, M., Graedel, T.E. (2006). Metal stocks and sustainability. Proceedings of the National Academy of Sciences (PNAS), 103(5), pp. 1209-1214. http://dx.doi.org/10.1073/pnas.0509498103

Graedel, T.E. (2017). Grand Challenges in Metal Life Cycles. Natural Resources Research, in press. http://dx.doi.org/10.1007/s11053-017-9333-8

Graedel, T.E., Reck, B.K. (2015). Six Years of Criticality Assessments, What Have We Learned So Far? Journal of Industrial Ecology, 20(4), pp. 692-699. http://dx.doi.org/10.1111/jiec.12305

Graedel, T.E., Barr, R., Chandler, C., Chase, T., Choi, J., Christoffersen, L., Friedlander, E., Henly, C., Jun, C., Nassar, N.T., Schechner, D., Warren, S., Yang, M.-Y., Zhu, C. (2012). Methodology of Metal Criticality Determination. Environmental Science & Technology, 46, pp. 1063-1070. http://dx.doi.org/10.1021/es203534z

Graedel, T.E., Harper, E.M., Nassar, N.T., Reck, B.K. (2015). On the materials basis of modern society. Proceedings of the National Academy of Sciences (PNAS), 112, pp. 6295-6300. http://dx.doi.org/10.1073/pnas.1312752110

GRI. (2013a). G4 Sustainability Reporting Guidelines: Reporting Principles and Standards Disclosures. Global Reporting Initiative (GRI), Amsterdam, Netherlands.

GRI. (2013b). G4 Sector Disclosures - Mining and Metals Sector Disclosures. Global Reporting Initiative (GRI), Amsterdam, Netherlands.

Gunson, A.J. (2013). Quantifying, reducing and improving mine water use. PhD Thesis, The University of British Columbia, Canada, May 2013, 285p.

Gunson, A.J., Klein, B., Veiga, M., Dunbar, S. (2010). Reducing mine network energy requirements. Journal of Cleaner Production, 18, pp. 1328-1338. http://dx.doi.org/10.1016/j.jclepro.2010.04.002

Gunson, A.J., Klein, B., Veiga, M., Dunbar, S. (2012). Reducing mine water requirements. Journal of Cleaner Production, 21, 71-82. http://dx.doi.org/10.1016/j.jclepro.2011.08.020

Gupta, V.K., Ali, I., Saleh, T.A., Nayak, A., Agarwal, S. (2012). Chemical treatment technologies for waste-water recycling – an overview. RSC Advances, 2(16), pp. 6380–6388. http://dx.doi.org/10.1039/C2RA20340E

Haggard, E.L., Sheridan, C.M., Harding, K.G. (2013). Water footprint for a South African platinum mine. Proceedings of Water in Mining 2013, Brisbane, QLD, Australia, 26-28 November, pp. 15-20.

Haggard, E.L., Sheridan, C.M., Harding, K.G. (2015). Quantification of water usage at a South African platinum processing plant. Water SA 41(2), pp. 279-286. http://dx.doi.org/10.4314/wsa.v41i2.14

Haley, S., Klick, M., Szymoniak, N., Crow, A. (2011). Observing trends and assessing data for Arctic mining. Polar Geography, 34(1-2), pp. 37-61. http://dx.doi.org/10.1080/1088937X.2011.584449

Halwatura, D., Lechner, A.M., Arnold, S. (2014). Design droughts as planning tool for ecosystems establishment in post-mining landscapes. Hydrology and Earth System Sciences Discussion, 11(5), pp. 4809-4849. http://dx.doi.org/10.5194/hessd-11-4809-2014

Harmsen, J.H.M., Roes, A.L., Patel, M.K. (2013). The impact of copper scarcity on the efficiency of 2050 global renewable energy scenarios. Energy, 50, pp. 62-73. http://dx.doi.org/10.1016/j.energy.2012.12.006

Harper, E.M., Kavlak, G., Burmeister, L., Eckelman, M.J., Erbis, S., Espinooza, V.S., Nuss, P., Graedel, T.E. (2015). Criticality of the Geological Zinc, Tin, and Lead Family. Journal of Industrial Ecology, 19(4), pp. 628-644. http://dx.doi.org/10.1111/jiec.12213

Haque, N., Norgate, T. (2013). Estimation of greenhouse gas emissions from ferroalloy production using life cycle assessment with particular reference to Australia. Journal of Cleaner Production, 39, pp. 220-230. http://dx.doi.org/10.1016/j.jclepro.2012.08.010

Haque, N., Norgate, T. (2014). The greenhouse gas footprint of in-situ leaching of uranium, gold and copper in Australia. Journal of Cleaner Production, 84, pp. 382-390. http://dx.doi.org/10.1016/j.jclepro.2013.09.033

Haywood, M.D.E., Dennis, D., Thomson, D.P., Pillans, R.D., 2016. Mine waste disposal leads to lower coral cover, reduced species richness and a predominance of simple coral growth forms on a fringing coral reef in Papua New Guinea. Marine Environmental Research, 115, pp. 36-48. http://dx.doi.org/10.1016/j.marenvres.2016.02.003

Heikkinen, P.M., Korkka-Niemi, K., Lahti, M., Salonen, V.-P. (2002). Groundwater and surface water contamination in the area of the Hitura nickel mine, Western Finland. Environmental Geology, 42(4), pp. 313-329. http://dx.doi.org/10.1007/s00254-002-0525-z

Hejazi, M., Edmonds, J., Clarke, L., Kyle, P., Davies, E., Chaturvedi, V., Wise, M., Patel, P., Eom, J., Calvin, K., Moss, R., Kim, S. (2014). Long-term global water projections using six socioeconomic scenarios in an integrated assessment modelling framework. Technological Forecasting & Social Change, 81, pp. 205-226. http://dx.doi.org/10.1016/j.techfore.2013.05.006

Hess, T.M., Lennard, A.T., Daccache, A. (2015). Comparing local and global water scarcity information in determining the water scarcity footprint of potato cultivation in Great Britain. Journal of Cleaner Production, 87, pp. 666-674. http://dx.doi.org/10.1016/j.jclepro.2014.10.075

Hinwood, A.L., Heyworth, J., Tanner, H., McCullough, C. (2012). Recreational use of acidic pit lakes – Human health considerations for post closure planning. Journal of Water Resource and Protection, 4, pp. 1061-1070. http://dx.doi.org/10.4236/jwarp.2012.412122

Hodgkinson, J.H., Loechel, B., Moffat, K., Howden, M., Littleboy, A.K., Crimp, S. (2010). Climate Risks to the Australia Mining Industry – A Preliminary Review of Vulnerabilities. Sustainable Mining Conference, 17-19 August 2010, Kalgoorlie, Western Australia, pp. 341-350.

Hodgkinson, J.H., Hobday, A.J., Pinkard, E.A. (2014). Climate adaptation in Australia's resource-extraction industries: ready or not? Regional Environmental Change, 14(4), pp. 1663-1678. http://dx.doi.org/10.1007/s10113-014-0618-8

Hoekstra, A.Y. (Editor). (2003). Virtual Water Trade: Proceedings of the International Expert Meeting on Virtual Water Trade. Delft, The Netherlands, 12-13 December 2002. Value of Water Research Report Series No. 12. IHE, Delft, the Netherlands.

Hoekstra, A.Y. (2016). A critique on the water-scarcity weighted water footprint in LCA. Ecological Indicators, 66, pp. 564-573. http://dx.doi.org/10.1016/j.ecolind.2016.02.026

Hoekstra, A., Chapagain, A. (2017). Water Footprint Research Alliance will intensify its efforts. Water Footprint Research Alliance, 25 August 2017, 35 co-signatories. Available from: http://waterfootprint.org/media/downloads/WFRA_statement_25082017.pdf (accessed October 2017)

Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M. (2009a). Water Footprint Manual, State of the Art 2009. Water Footprint Network, Earthscan, London, United Kingdom.

Hoekstra, A.Y., Gerbens-Leenes, W., van der Meer, T.H. (2009b). Reply to Pfister and Hellweg: Water footprint accounting, impact assessment, and life-cycle assessment. Proceedings of the National Academy of Sciences (PNAS), 106(40), pp. E114. http://dx.doi.org/10.1073/pnas.0909948106

Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M. (2011). The Water Footprint Assessment Manual: Setting the Global Standard. Water Footprint Network, Earthscan, London, United Kingdom.

Hoekstra, A.Y., Mekonnen, M.M., Chapagain, A.K., Mathews, R.E., Richter, B.D. (2012). Global monthly water scarcity: blue water footprints versus blue water availability. PLoS One, 7(2), e32688. http://dx.doi.org/10.1371/journal.pone.0032688

Hoekstra, A.Y., Mekonnen, M.M. (2011). Global water scarcity: the monthly blue water footprint compared to blue water availability for the world's major river basins. Value of Water Research Report Series No. 53, UNESCO-IHE, The Netherlands.

Holden, W.N. (2015). Mining amid typhoons: Large-scale mining and typhoon vulnerability in the Philippines. The Extractive Industries and Society, 2, pp. 445-461. http://dx.doi.org/10.1016/j.exis.2015.04.009

Holley, E.A., Mitcham, C. (2016). The Pebble Mine Dialogue: A case study in public engagement and the social license to operate. Resources Policy, 47, pp. 18-27. http://dx.doi.org/10.1016/j.resourpol.2015.11.002

Hybel, A.-M., Godskesen, B., Rygaard, M. (2015). Selection of spatial scale for assessing impacts of groundwater-based water supply on freshwater resources. Journal of Environmental Management, 160, pp. 90-97. http://dx.doi.org/10.1016/j.jenvman.2015.06.016

Hua, W., Dong, S., Li, Y., Xu, J., Wang, Q. (2015). The influence of cyclic wetting and drying on the fracture toughness of sandstone. International Journal of Rock Mechanics and Mining Sciences, 78, pp. 331-335. http://dx.doi.org/10.1016/j.ijrmms.2015.06.010

Huang, X., Sillanpää, M., Gjessing, E.T., Peräniemi, S., Vogt, R.D., 2010. Environmental impact of mining on surface water quality in Tibet: Gyama Valley. Science of the Total Environmental, 408, pp. 4177-4184. http://dx.doi.org/10.1016/j.scitotenv.2010.05.015

ICMM. (2012). Water management in mining: a selection of case studies. International Council on Mining & Metals (ICMM), London, UK, May 2012, 32p.

ICMM. (2013). Adapting to a changing climate: implications for the mining and metals industry. International Council on Mining & Metals, London, UK, March 2013.

ICMM. (2014). Water stewardship framework. International Council on Mining & Metals (ICMM), London, UK, April 2014, 4p.

ICMM. (2015). A practical guide to catchment-based water management for the mining and metals industry. International Council on Mining & Metals (ICMM), London, UK, 59p.

ICMM. (2017). A practical guide to consistent water reporting. International Council on Mining & Metals (ICMM), London, UK, March 2017, 72p.

Ihle, C.F. (2014). The need to extend the study of greenhouse impacts of mining and mineral processing to hydraulic streams: long distance pipelines count. Journal of Cleaner Production, 84, pp. 597.

IIED. (2012). MMSD+10: Reflecting on a decade of mining and sustainable development. International Institute for Environment and Development (IIED), London, United Kingdom. http://pubs.iied.org/16041IIED.html

INAP, 2009. Global Acid Rock Drainage Guide (GARD Guide). International Network for Acid Prevention (INAP). http://www.gardguide.com (accessed 04-07-2016)

IPCC, 2000. Special report on emissions scenarios. In: Nakicenovik, Swart (Editor), Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK, July 2000, 612p.

ISO. (2006). “ISO 14044:2006. Environmental Management – Life Cycle Assessment – Requirements and Guidelines”. International Organisation for Standardization (ISO), Geneva, Switzerland.

ISO. (2014). “ISO 14046:2014. Environmental Management -- Water Footprint -- Principles, requirements and guidelines". International Organisation for Standardization (ISO), Geneva, Switzerland.

Jahanshahi, S., Mathieson, J.G., Somerville, M.A., Haque, N., Norgate, T.E., Deev, A., Pan, Y., Xie, D., Ridgeway, P., Zulli, P. (2015). Development of Low-Emission Integrated Steelmaking Process. Journal of Sustainable Metallurgy, 1(1), pp. 94-114. https://doi.org/10.1007/s40831-015-0008-6

Jenkins, H., Yakovleva, N. (2006). Corporate social responsibility in the mining industry: Exploring trends in social and environmental disclosure. Journal of Cleaner Production, 14, pp. 271-284. http://dx.doi.org/10.1016/j.jclepro.2004.10.004

Jeswani, H.K., Azapagic, A. (2011). Water footprint: methodologies and a case study for assessing the impacts of water use. Journal of Cleaner Production, 19, pp. 1288-1299. http://dx.doi.org/10.1016/j.jclepro.2011.04.003

Johnson, D.B., Hallberg, K.B. (2005). Acid mine drainage remediation options: a review. Science of the Total Environment, 338, pp. 3-14. http://dx.doi.org/10.1016/j.scitotenv.2004.09.002

Jones, R.T. (2004). JOM world nonferrous smelter survey, Part II: Platinum group metals. JOM, 56(12), pp. 59-63. http://dx.doi.org/10.1007/s11837-004-0238-8

Jowitt, S.M., Mudd, G.M., Weng, Z. (2013). Hidden mineral deposits in Cu-dominated porphyry-skarn systems: How resource reporting can occlude important mineralization types within mining camps. Economic Geology, 108(5), 1185-1193. http://dx.doi.org/10.2113/econgeo.108.5.1185

Kapusta, J.P.T. (2004). JOM world nonferrous smelters survey, part I: Copper. JOM, 56(7), pp. 21-27. http://dx.doi.org/10.1007/s11837-004-0086-6

Kauppi, S. (2013). Ympäristötietoa kaivoshankkeista – taustatietoa kaivostoimintaan liittyvästä lainsäädännöstä ja eräiden kaivosten ympäristötarkkailusta. Suomen ympäristökeskuksen raportteja 10, Helsinki 2013. [In Finnish].

Kemp, D., Bond, C.J., Franks, D.M., Cote, C. (2010). Mining, water and human rights: making the connection. Journal of Cleaner Production, 18, pp. 1553-1562. http://dx.doi.org/10.1016/j.jclepro.2010.06.008

Kempton, H., Bloomfield, T.A., Hanson, J.L., Limerick, P. (2010). Policy guidance for identifying and effectively managing perpetual environmental impacts from new hardrock mines. Environmental Science and Policy, 13, pp. 558-866. http://dx.doi.org/10.1016/j.envsci.2010.06.001

Kerr, R. (2014). The Coming Copper Peak. Science, 343(6712), 14 February 2014, pp. 722-724. http://dx.doi.org/10.1126/science.343.6172.722

Kiguchi, M., Shen, Y., Kanae, S., Oki, T. (2015). Re-evaluation of future water stress due to socio-economic and climate factors under a warming climate. Hydrological Sciences Journal, 60(1), pp. 14-29. http://dx.doi.org/10.1080/02626667.2014.888067

Kleijn, R., van der Voet, E., Kramer, G.J., van Oers, L., van der Giesen, C. (2011). Metal requirements of low-carbon power generation. Energy, 36, pp. 5640-5648. http://dx.doi.org/10.1016/j.energy.2011.07.003

Klinglmair, M., Sala, S., Brandão, M. (2014). Assessing resource depletion in LCA: a review of methods and methodological issues. International Journal of Life Cycle Assessment, 19(3), pp. 580-592. http://dx.doi.org/10.1007/s11367-013-0650-9

Koehnken, L. (1997). Final report: Mount Lyell remediation research and demonstration program. Supervising Scientist Report 126, Department of Environment and Land Management, Tasmania, Australia.

Koffler, C., Shonfield, P., Vickers, J. (2017). Beyond pedigree – optimizing and measuring representativeness in large-scale LCAs. International Journal of Life Cycle Assessment, 22(7), pp. 1065-1077. http://dx.doi.org/10.1007/s11367-016-1223-5

Koppelaar, R.H.E.M., Koppelaar, H. (2016). The Ore Grade and Depth Influence on Copper Energy Inputs. Biophysical Economics and Resource Quality, 1(2), Article 11, 16p. http://dx.doi.org/10.1007/s41247-016-0012-x

Köppen, W. (1900). Versuch einer Klassifikation der Klimate, vorzugsweise nach ihren Beziehungen zur Pflanzenwelt. Geographische Zeitschrift, 6, pp. 593-611.

Kottek, M., Grieser, J., Beck, C., Rudolf, B., Rubel, F. (2006). World Map of the Köppen-Geiger climate classification updated. Meteorologische Zeitschrift, 15(3), pp. 259-263. http://dx.doi.org/10.1127/0941-2948/2006/0130

Kounina, A., Margni, M., Bayart, J.-B., Boulay, A.-M., Berger, M., Bulle, C., Frischknecht, R., Koehler, A., Milà I Canals, L., Motoshita, M., Núñez, M., Peters, G., Pfister, S., Ridoutt, B., van Zelm, R., Verones, F., Humbert, S. (2013). Review of methods addressing freshwater use in life cycle inventory and impact assessment. International Journal of Life Cycle Assessment, 18, pp. 707-721. http://dx.doi.org/10.1007/s11367-012-0519-3

Kronenberg, J. (2013). Linking Ecological Economics and Political Ecology to Study Mining, Glaciers and Global Warming. Environmental Policy and Governance, 23, pp. 75-90. http://dx.doi.org/10.1002/eet.1605

Kumar, R.N., McCullough, C.D., Lund, M.A. (2009). Water resources in Australian mine pit lakes. Mining Technology, 118(3/4), pp. 205-211. http://dx.doi.org/10.1179/174328610X12682159815028

Kunz, N.C., Moran, C.J. (2016). The utility of a systems approach for managing strategic water risks at a mine site level. Water Resources and Industry, 13, pp. 1-6. http://dx.doi.org/10.1016/j.wri.2016.02.001

La Brooy, S.R., Linge, H.G., Walker, G.S. (1994). Review of Gold Extraction From Ores. Minerals Engineering, 7(10), pp. 1213-1241. http://dx.doi.org/10.1016/0892-6875(94)90114-7

Lagos, G., Peters, D., Videla, A., Jara, J.J. (2017). The effect of mine aging on the evolution of environmental footprint indicators in the Chilean copper mining industry 2001-2015. Journal of Cleaner Production, in press. http://dx.doi.org/10.1016/j.jclepro.2017.10.290

Lèbre, E., Corder, G., Golev, A. (2017). The Role of the Mining Industry in a Circular Economy: A Framework for Resource Management at the Mine Site Level. Journal of Industrial Ecology, 21(3), pp. 662-672. http://dx.doi.org/10.1111/jiec.12596

Leong, S., Hazelton, J., Taplin, R., Timms, W., Laurence, D. (2014). Mine site-level water reporting in the Macquarie and Lachlan catchments: a study of voluntary and mandatory disclosures and their value for community decision-making. Journal of Cleaner Production, 84, pp. 94-106. http://dx.doi.org/10.1016/j.jclepro.2014.01.021

Lin, C. (2012). Climate change adaptation in acid sulfate landscapes. American Journal of Environmental Sciences, 8(4), pp. 433-442. http://dx.doi.org/10.3844/ajessp.2012.433.442

Liu, W., Moran, C.J., Vink, S. (2013). Managing the potential risks of using bacteria-laden water in mineral processing to protect freshwater. Environmental Science & Technology, 47, pp. 6582-6588. http://dx.doi.org/10.1021/es400671h

Loechel, B., Hodgkinson, J., Moffat, K. (2013). Climate change adaptation in Australian mining communities: comparing mining company and local government views and activities. Climatic Change, 119(2), pp. 465-477. http://dx.doi.org/10.1007/s10584-013-0721-8

LPSDP. (2008). Water Management, Leading Practice Sustainable Development Program (LPSDP) For the Mining Industry. Australian Government, May 2008. 112p. Available from: www.industry.gov.au/resource/Programs/LPSD/Pages/LPSDhandbooks.aspx (accessed 04-07-2016)

Lottermoser, B. (2010). Mine Wastes: Characterization, Treatment and Environmental Impacts. Springer Science & Business Media.

Lund, M.A., Blanchette, M.L. (2014). Are we approaching pit lake closure from the wrong perspective? Proceedings of 12th International Mine Water Association Congress, Xuzhou, China, Aug 18-22, pp. 720-724.

Madrid, C., Cabello, V., Giampietro, M. (2013). Water-Use Sustainability in Socioecological Systems: A Multiscale Integrated Approach. BioScience, 63, pp. 14–24. http://dx.doi.org/10.1525/bio.2013.63.1.6

Majeau-Bettez, G., Wood, R., Strømman, A.H. (2014). Unified Theory of Allocations and Constructs in Life Cycle Assessment and Input-Output Analysis. Journal of industrial Ecology, 18, pp. 747-770. http://dx.doi.org/10.1111/jiec.12142

Mäkinen, J. Lerssi, J. (2007). Characteristics and Seasonal Variation of Sediments in Lake Junttiselkä, Pyhäsalmi, Finland. Mine Water and the Environment, 26(4), pp. 217-228. http://dx.doi.org/217-228. 10.1007/s10230-007-0015-3

Marsden, J.O. (2006). Overview of gold processing techniques around the world. Minerals & Metallurgical Processing, 23(3), pp. 121-125.

Martinez, C., Franks, D.M. (2014). Does mining company-sponsored community development influence social license to operate? Evidence from private and state-owned companies in Chile. Impact Assessment and Project Appraisal, 32(4), pp. 294-303. http://dx.doi.org/10.1080/14615517.2014.929783

Mason, L., Prior, T., Mudd, G., Giurco, D. (2011). Availability, addiction and alternatives: three criteria for assessing the impact of peak minerals on society. Journal of Cleaner Production, 19, pp. 958-966. http://dx.doi.org/10.1016/j.jclepro.2010.12.006

Mason, L., Unger, C., Lederwasch, A., Razian, H., Wynne, L., Guirco, D. (2013). Adapting to climate risks and extreme weather: A guide for mining and minerals industry professionals. National Climate Change Adaptation Research Facility, Gold Coast, Australia, 77p.

May, D., Prior, T., Cordell, D., Giurco, D. (2012). Peak Minerals: Theoretical Foundations and Practical Application. Natural Resources Research, 21(1), pp. 43-60. http://dx.doi.org/10.1007/s11053-011-9163-z

MCA. (2014). Water Accounting Framework for the Minerals Industry, User Guide, Version 1.3, January 2014. Prepared by the Sustainable Minerals Institute, University of Queensland for the Minerals Council of Australia (MCA).

McAuliff, K., Sadiq, R., Hewage, K. (2017). Water use in unconventional oil and gas development: an assessment on water use metric evaluation and selection. Clean Technologies and Environmental Policy, volume unassigned, 13p. http://dx.doi.org/10.1007/s10098-017-1431-0

McCullough, C.D., Marchand, G., Unseld, J. (2013). Mine Closure of Pit Lakes as Terminal Sinks: Best Available Practice When Options are Limited? Mine Water and the Environment, 32(4), pp. 302-313. http://dx.doi.org/10.1007/s10230-013-0235-7

McMahon, T.A., Peel, M.C., Karoly, D.J. (2015). Assessment of precipitation and temperature data from CMIP3 global climate models for hydrologic simulations. Hydrology and Earth System Sciences, 19, pp. 361-377. http://dx.doi.org/10.5194/hess-19-361-2015

Mekonnen, M.M., Hoekstra, A.Y. (2016). Four billion people facing severe water scarcity. Science Advances, 2(2), e1500323. http://dx.doi.org/10.1126/sciadv.1500323

Memary, R., Giurco, D., Mudd, G., Mason, L. (2012). Life cycle assessment: a time-series analysis of copper. Journal of Cleaner Production, 33, pp. 97-108. http://dx.doi.org/10.1016/j.jclepro.2012.04.025

MEND. (2011). Climate Change and Acid Rock Drainage – Risks for the Canadian Mining Sector. Mine Environment Neutral Drainage (MEND), Mining Association of Canada, MEND Report 1.61.7.

Milà I Canals, L., Chenoweth, J., Chapagain, A., Orr, S., Antón, A., Clift, R. (2009). Assessing freshwater use impacts in LCA: Part I–inventory modelling and characterisation factors for the main impact pathways International Journal of Life Cycle Assessment, 14, pp. 28-42. http://dx.doi.org/10.1007/s11367-008-0030-z

Mitchell, T.D., Carter, T.R., Jones, P.D., Hulme, M., New, M. (2004). A comprehensive set of high-resolution grids of monthly climate for Europe and the globe: the observed records (1901-2000) and 16 scenarios (2001-2100). Work Paper 55. Tyndall Centre of Climate Change Research.

Moffat, K., Zhang, A. (2014). The paths to social licence to operate: An integrative model explaining community acceptance of mining. Resources Policy, 39, pp. 61-70. http://dx.doi.org/10.1016/j.resourpol.2013.11.003

Mohan, D., Pittman, C.U. (2007). Arsenic removal from water/wastewater using adsorbents—A critical review. Journal of Hazardous Materials, 142(1), pp. 1–53. http://dx.doi.org/10.1016/j.jhazmat.2007.01.006

Mohr, S.H. (2010). Projection of world fossil fuel production with supply and demand interactions. PhD Thesis, The University of Newcastle, 2010.

Mohr, S.H., Mudd, G.M., Giurco, D. (2012). Lithium Resources and Production: Critical Assessment and Global Projections. Minerals, 2, pp. 65-84. http://dx.doi.org/10.3390/min2010065

Mohr, S., Giurco, D., Yelishetty, M., Ward, J., Mudd, G. (2015). Projections of Iron Ore Production. Natural Resources Research, 24(3), pp. 317-327. http://dx.doi.org/10.1007/s11053-014-9256-6

Moran, C.J., Franks, D.M., Sonter, L.J. (2013). Using the multiple capitals framework to connect indicators of regional cumulative impacts of mining and pastoralism in the Murray Darling Basin, Australia. Resources Policy, 38(4), pp. 733-744. https://doi.org/10.1016/j.resourpol.2013.01.002

Moran, C.J., Lodhia, S., Kunz, N.C., Huisingh, D. (2014). Sustainability as it pertains to minerals and energy supply and demand: a new interpretative perspective for assessing progress. Journal of Cleaner Production, 84, pp. 16-26. http://dx.doi.org/10.1016/j.jclepro.2014.09.016

Moreno, P.A., Aral, H., Cuevas, J., Monardes, A., Adaro, M., Norgate, T., Bruckard, W. (2011). The use of seawater as process water at Las Luces copper-molybdenum beneficiation plant in Taltal (Chile). Minerals Engineering, 24, pp. 852-858. http://dx.doi.org/10.1016/j.mineng.2011.03.009

Mosier, D.L., Berger, V.I., Singer, D.A. (2009). Volcanogenic Massive Sulfide Deposits of the World: Database and Grade and Tonnage Models. U.S. Geological Survey, Reston, Virginia, USA, Open File Report 2009-1034, 50p. http://pubs.usgs.gov/of/2009/1034/

Mudd, G.M. (2001a). Critical review of acid in situ leach uranium mining: 1. USA and Australia. Environmental Geology, 41, pp. 390-403. http://dx.doi.org/10.1007/s002540100406

Mudd, G.M. (2001b). Critical review of acid in situ leach uranium mining: 2. Soviet Block and Asia. Environmental Geology, 41, pp. 404-416. http://dx.doi.org/10.1007/s002540100405

Mudd, G.M. (2008). Sustainability Reporting and Water Resources: a Preliminary Assessment of Embodied Water and Sustainable Mining. Mine Water and the Environment, 27, pp. 136-144. http://dx.doi.org/10.1007/s10230-008-0037-5

Mudd, G.M. (2010). The environmental sustainability of mining in Australia: key megra-trends and looming constraints. Resources Policy, 35, pp. 98-115. http://dx.doi.org/10.1016/j.resourpol.2009.12.001

Mudd, G.M., Jowitt, S.M. (2014). A Detailed Assessment of Global Nickel Resource Trends and Endowments. Economic Geology, 109, pp. 1813-1841. http://dx.doi.org/10.2113/econgeo.109.7.1813

Mudd, G.M., Mohr, S. (2010). Mineral Resources and Production in the Goldfields-Esperance Region: Projecting Future Scenarios. Monash University, December 2010, Prepared for the Goldfields Esperance Development Commission.

Mudd, G.M., Weng, Z., Jowitt, S.M. (2013a). A Detailed Assessment of Global Cu Resource Trends and Endowments. Economic Geology, 108, pp. 1163-1183. http://dx.doi.org/10.2113/econgeo.108.5.1163

Mudd, G.M., Weng, Z., Jowitt, S.M., Turnbull, I.D., Graedel, T.E. (2013b). Quantifying the recoverable resources of by-product metals: The case of cobalt. Ore Geology Reviews, 55, pp. 87-98. http://dx.doi.org/10.1016/j.oregeorev.2013.04.010

Mudd, G.M., Jowitt, S.M., Werner, T.T. (2016). The World's by-product and critical metal resources part I: Uncertainties, current reporting practices, implications and ground for optimism. Ore Geology Reviews, In Press. http://dx.doi.org/10.1016/j.oregeorev.2016.05.001

Mudd, G.M., Jowitt, S.M., Werner, T.T. (2017). The World's Lead-Zinc Mineral Resources: Scarcity, Data, Issues and Opportunities. Ore Geology Reviews, 80, pp. 1160-1190. http://dx.doi.org/10.1016/j.oregeorev.2016.08.010

Mudd, G.M., Roche, C.P., Northey, S.A., Jowitt, S.M. (unpublished). Mining in Papua New Guinea: Linking Key Trends and Complex Environmental-Social Issues. Submitted to Science of the Total Environment.

Müller, E., Hilty, L.M., Widmer, R., Schluep, M., Faulstich, M. (2014). Modeling Metal Stocks and Flows: A Review of Dynamic Material Flow Analysis Methods. Environmental Science & Technology, 48(4), pp. 2102-2113. http://dx.doi.org/10.1021/es403506a

Müller Schmied, H., Eisner, S., Franz, D., Wattenbach, M., Portmann, F.T., Flörke, M., Döll, P. (2014). Sensitivity of simulated global-scale freshwater fluxes and storages to input data, hydrological model structure, human water use and calibration. Hydrology and Earth System Science, 18, pp. 3511-3538. http://dx.doi.org/10.5194/hess-18-3511-2014

Murguía, D. I., Bringezu, S., Schaldach, R. (2016). Global direct pressures on biodiversity by large-scale metal mining: Spatial distribution and implications for conservation. Journal of Environmental Management, 180, pp. 409-420. http://dx.doi.org/10.1016/j.jenvman.2016.05.040

Mwale, A.H., Musonge, P., Fraser, D.M. (2005). The influence of particle size on energy consumption and water recovery in comminution and dewatering systems. Minerals Engineering, 18, pp. 915-926. http://dx.doi.org/10.1016/j.mineng.2005.02.014

Napier-Munn, T.J., Morrison, R.D. (2003). The potential for the dry processing of ores. Proceedings of Water in Mining 2003, Brisbane, Qld, 13-15 Oct, pp. 247-250.

Nassar, N.T., Barr, R., Browning, M., Diao, Z., Friedlander, E., Harper, E.M., Henly, C., Kavlak, G., Kwatra, S., Jun, C., Warren, S., Yang, M.-Y., Graedel, T.E. (2012). Criticality of the Geological Copper Family. Environmental Science & Technology, 46, pp. 1071-1078. http://dx.doi.org/10.1021/es203535w

Nassar, N.T., Du, X., Graedel, T.E. (2015a). Criticality of the Rare Earth Elements. Journal of Industrial Ecology, 19(6), pp. 1044-1054. http://dx.doi.org/10.1111/jiec.12237

Nassar, N.T., Graedel, T.E., Harper, E.M. (2015). By-product metals are technologically essential but have problematic supply. Science Advances, 1, e1400180. http://dx.doi.org/10.1126/sciadv.1400180

Newcrest, 2007. Update on Cadia Water Supply. ASX market release, Newcrest Mining Limited, 15 May 2007, Melbourne, Australia.

Newcrest, (2011a). Quarterly Report for the Three Months Ending 31 March 2011. Newcrest Mining Ltd, Melbourne, VIC, 10 p.

Newcrest. (2011b). Sustainability Report 2011. Newcrest Mining Limited, Melbourne, Australia.

Newcrest. (2013). Cadia Valley Operations, Annual Environmental Management Report, 2014/2015. Newcrest Mining Limited, Melbourne, Australia.

Newcrest. (2014). Newcrest Mining Limited Sustainability Report 2013, GRI Data Tables. Newcrest Mining Limited, Melbourne, Australia.

Nickless, E. (2017). Resourcing Future Generations: A Contribution by the Earth Science Community. Natural Resources Research, in press, 16p. http://dx.doi.org/10.1007/s11053-017-9331-x

Nilsson, C., Reidy, C.A., Dynesius, M., Revenga, C. (2005). Fragmentation and Flow Regulation of the World's Large River Systems. Science, 308(5720), pp. 405-408. http://dx.doi.org/10.1126/science.1107887

Nordstrom, D.K. (2009). Acid rock drainage and climate change. Journal of Geochemical Exploration, 100, pp. 97-104. http://dx.doi.org/10.1016/j.gexplo.2008.08.002

Nordstrom, D.K., Blowes, D.W., Ptacek, C.J. (2015). Hydrogeochemistry and microbiology of mine drainage: An update. Applied Geochemistry, 57, pp. 3-16. http://dx.doi.org/10.1016/j.apgeochem.2015.02.008

Norgate, T.E., Lovel, R.R. (2004). Water Use in Metal Production: a Life Cycle Perspective. Commonwealth Scientific and Industrial Research Organisation (CSIRO), Australia. Report no. DMR2505.

Norgate, T.E., Lovel, R.R. (2006). Sustainable water use in minerals and metal production. Proceedings of Water in Mining Conference 2006, Brisbane, QLD, Australia, pp. 331-339.

Norgate, T., Haque, N. (2010). Energy and greenhouse gas impacts of mining and mineral processing operations. Journal of Cleaner Production, 18, pp. 266-274. http://dx.doi.org/10.1016/j.jclepro.2009.09.020

Norgate, T., Haque, N. (2012). Using life cycle assessment to evaluate some environmental impacts of gold production. Journal of Cleaner Production, 29-30, pp. 53-63. http://dx.doi.org/10.1016/j.jclepro.2012.01.042

Norgate, T., Haque, N. (2013). The greenhouse gas impact of IPCC and ore-sorting technologies. Minerals Engineering, 42, pp. 13-21. http://dx.doi.org/10.1016/j.mineng.2012.11.012

Norgate, T., Jahanshahi, S. (2010). Low grade ores – Smelt, leach or concentrate? Minerals Engineering, 23, pp. 65-73. http://dx.doi.org/10.1016/j.mineng.2009.10.002

Norgate, T., Jahanshahi, S. (2011). Assessing the energy and greenhouse gas footprints of nickel laterite processing. Minerals Engineering, 24, pp. 698-707. http://dx.doi.org/10.1016/j.mineng.2010.10.002

Norgate, T.E., Jahanshahi, S., Rankin, W.J. (2007). Assessing the environmental impact of metal production processes. Journal of Cleaner Production, 15, pp. 838-848. http://dx.doi.org/10.1016/j.jclepro.2006.06.018

Northey, S., Haque, N. (2013). Life cycle based water footprints of selected mineral and metal processing routes. Water in Mining 2013 Conference, Brisbane QLD, Australia, 26-28 November 2013, pp. 35-50.

Northey, S.A., Mudd, G.M. (2016). Resource Depletion Scenarios – How should we address the limitations? 35th International Geological Congress, Cape Town, South Africa, paper 2731, 2p.

Northey, S., Haque, N., Mudd, G. (2013). Using sustainability reporting to assess the environmental footprint of copper mining. Journal of Cleaner Production, 40, pp. 118-128. http://dx.doi.org/10.1016/j.jclepro.2012.09.027

Northey, S.A., Haque, N., Lovel, R., Cooksey, M.A. (2014a). Evaluating the application of water footprint methods to primary metal production systems. Minerals Engineering, 69, pp. 65-80. http://dx.doi.org/10.1016/j.mineng.2014.07.006

Northey, S.A., Mohr, S., Mudd, G.M., Weng, Z., Giurco, D. (2014b). Modelling future copper ore grade decline based on a detailed assessment of copper resources and mining. Resources, Conservation and Recycling, 83, pp. 190-201. http://dx.doi.org/10.1016/j.resconrec.2013.10.005

Northey, S.A., Mudd, G.M., Saairvuori, E., Wessman-Jääskeläinen, H., Haque, N. (2016). Water footprinting and mining: Where are the limitations and opportunities? Journal of Cleaner Production, 135, pp. 1098-1116. http://dx.doi.org/10.1016/j.jclepro.2016.07.024

Northey, S.A., Mudd, G.M., Werner, T.T., Jowitt, S.M., Haque, N., Yellishetty, M., Weng, Z. (2017a). The exposure of global base metal resources to water criticality, scarcity and climate change. Global Environmental Change, 44, pp. 109-124. http://dx.doi.org/10.1016/j.gloenvcha.2017.04.004

Northey, S.A., Mudd, G.M., Werner, T.T. (2017b). Unresolved Complexity in Assessments of Mineral Resource Depletion and Availability. Natural Resources Research, volume unassigned, 15p. http://dx.doi.org/10.1007/s11053-017-9352-5

Northey, S.A., Madrid Lopez, C., Haque, N., Mudd, G.M., Yellishetty, M. (unpublished). Production weighted water use impact characterisation factors for the global mining industry: A comparison of watershed and national average WSI and AWaRe factors. Submitted to the Journal of Cleaner Production, under review.

NPI. (2017). National Pollutant Inventory (NPI). Australian Government. Accessible via: http://www.npi.gov.au (last accessed 9-11-2017).

NPRI. (2017). National Pollutant Release Inventory (NPRI). Government of Canada. https://www.canada.ca/en/services/environment/pollution-waste-management/national-pollutant-release-inventory.html (last accessed 9-11-2017)

Núñez, M., Pfister, S., Vargas, M., Antón, A. (2015). Spatial and temporal specific characterisation factors for water use impacts in Spain. International Journal of Life Cycle Assessment, 20, pp. 128-138. http://dx.doi.org/10.1007/s11367-014-0803-5

Nuss, P., Eckelman, M.J. (2014). Life Cycle Assessment of Metals: A Scientific Synthesis. PLoS ONE, 9(7), e101298. http://dx.doi.org/10.1371/journal.pone.0101298

Nuss, P., Harper, E.M., Nassar, N.T., Reck, B.K., Graedel, T.E. (2014). Criticality of Iron and Its Principal Alloying Elements. Environmental Science & Technology, 48(7), PP. 4171-4177. http://dx.doi.org/10.1021/es405044w

OECD. (2017). Centre for PRTR Data. Organisation for Economic Co-Operation and Development (OECD). http://www.oecd.org/env_prtr_data/ (last accessed 9-11-2017)

Olivares, M., Toledo, M. Acuña, A., Garcés, M. (2012). Preliminary estimate of the water footprint of copper concentrate production in central Chile. Proceedings of the Third International Congress on Water Management in the Mining Industry, Santiago, Chile, 6-8 June 2012, Gecamin, pp. 390-400.

Osman, A., Crundwell, F., Harding, K.G., Sheridan, C.M. (2017). Application of the water footprinting method and water accounting framework to a base metal refining process. Water SA, 43(4), pp. 722-729. http://dx.doi.org/10.4314/wsa.v43i4.18

Özbek, A. (2014). Investigation of the effects of wetting-drying and freezing-thawing cycles on some physical and mechanical properties of selected ignimbrites. Bulletin of Engineering Geology and the Environment, 73(2), pp. 595-609. http://dx.doi.org/10.1007/s10064-013-0519-y

Palaniandy, S., Powell, M. (2014). Addressing water scarcity in mining through improved comminution practices. AusIMM Bulletin October 2014, 5(48), pp. 50-52.

Pearce, T.D., Ford, J.D., Prno, J., Duerden, F., Pittman, J., Beaumier, M., Berrang-Ford, L., Smit, B. (2011). Climate change and mining in Canada. Mitigation and Adaptation Strategies for Global Change, 16, pp. 347-368. http://dx.doi.org/10.1007/s11027-010-9269-3

Peel, M.C., Finlayson, B.L., McMahon, T.A. (2007). Updated world map of the Köppen-Geiger climate classification. Hydrology and earth system sciences discussions, 4, pp. 439-473.

Peña, C.A., Huijbregts, M.A.J. (2014). The blue water footprint of primary copper production in Northern Chile. Journal of Industrial Ecology, 18(1), pp. 49-58. http://dx.doi.org/10.1111/jiec.12036

Perez, F., Sanchez, L.E. (2009). Assessing the Evolution of Sustainability Reporting in the Mining Sector. Environmental Management, 43, pp. 949-961. https://doi.org/10.1007/s00267-008-9269-1

Pfister, S., Bayer, P. (2014). Monthly water stress: spatially and temporally explicit consumptive water footprint of global crop production. Journal of Cleaner Production, 73, pp. 52-62. http://dx.doi.org/10.1016/j.jclepro.2013.11.031

Pfister, S., Hellweg, S. (2009). The water “shoesize” vs. footprint of bioenergy. PNAS, 106(35), pp. E93-E94. http://dx.doi.org/10.1073/pnas.0908069106

Pfister, S., Ridoutt, B.G. (2014). Water footprint: pitfalls on common ground. Environmental Science and Technology, 48, pp. 4-4. http://dx.doi.org/10.1021/es405340a

Pfister, S., Koehler, A., Hellweg, S. (2009). Assessing the environmental impacts of freshwater consumption in LCA. Environmental Science & Technology, 43, pp. 4098-4104. http://dx.doi.org/10.1021/es802423e

Pfister, S., Bayer, P., Koehler, A., Hellweg, S. (2011). Environmental impacts of water use in global crop production: hotspots and trade-offs with land use. Environmental Science & Technology, 45(13), pp. 5761-5768. http://dx.doi.org/10.1021/es1041755

Pfister, S., Vionnet, S., Levova, T., Humbert, S. (2015). Ecoinvent 3: assessing water use in LCA and facilitating water footprinting. International Journal of Life Cycle Assessment, 12p. http://dx.doi.org/10.1007/s11367-015-0937-0

Pfister, S., Boulay, A.-M., Berger, M., Hadjikakou, M., Motoshita, M., Hess, T., Ridoutt, B., Weinzettel, J., Scherer, L., Döll, P., Manzardo, A., Núñez, M., Verones, F., Humbert, S., Buxmann, K., Harding, K., Benini, L., Oki, T., Finkbeiner, M., Henderson, A. (2017). Understanding the LCA and ISO water footprint: A response to Hoekstra (2016) “A critique on the water-scarcity weighted water footprint in LCA”. Ecological Indicators, 72, pp. 352-359. http://dx.doi.org/10.1016/j.ecolind.2016.07.051

Phillips, J. (2016). Climate change and surface mining: A review of environment-human interactions & their spatial dynamics. Applied Geography, 74, pp. 95-108. http://dx.doi.org/10.1016/j.apgeog.2016.07.001

Prior, T., Giurco, D., Mudd, G., Mason, L., Behrisch, J. (2012). Resource depletion, peak minerals and the implications for sustainable resource management. Global Environmental Change, 22(3), pp. 577-587. http://dx.doi.org/10.1016/j.gloenvcha.2011.08.009

Prno, J. (2013). An analysis of factors leading to the establishment of a social license to operate in the mining industry. Resources Policy, 38, pp. 577-590. http://dx.doi.org/10.1016/j.resourpol.2013.09.010

QFCI. (2012). Chapter 13, Mining. In: Final Report of the Queensland Floods Commission of Inquiry (QFCI), March 2012, Australia.

QRC. (2011). QRC submission: Queensland Floods Commission of Inquiry. Queensland Resources Council (QRC), Australia, March 2011.

Qi, C., Ye, L., Ma, X., Yang, D., Hong, J. (2017). Life cycle assessment of the hydrometallurgical zinc production chain in China. Journal of Cleaner Production, 156, pp. 451-458. http://dx.doi.org/10.1016/j.jclepro.2017.04.084

Quinteiro, P., Ridoutt, B.G., Arroja, L., Dias, A.C. (2017). Identification of methodological challenges remaining in the assessment of a water scarcity footprint. International Journal of Life Cycle Assessment, volume unassigned, 17p. http://dx.doi.org/10.1007/s11367-017-1304-0

Ramirez-Llodra, E., Trannum, H.C., Evenset, A., Levin, L.A., Andersson, M., Finner, T.E., Hilario, A., Flem, B., Christensen, G., Schaanning, M., Vanreusel, A. (2015). Submarine and deep-sea mine tailings placements: A review of current practices, environmental issues, natural analogs and knowledge gaps in Norway and internationally. Marine Pollution Bulletin, 97, pp. 13-35. http://dx.doi.org/10.1016/j.marpolbul.2015.05.062

Ranchod, N., Sheridan, C.M., Pint, N., Slatter, K., Harding, K.G. (2015). Assessing the blue-water footprint of an opencast platinum mine in South Africa. Water SA, 41(2), pp. 287-293. http://dx.doi.org/10.4314/wsa.v41i2.15

Rankin, W.J. (2013). Australasian mining and metallurgical operating practices: the Sir Maurice Mawby memorial volume, Third Edition. Australasian Institute of Mining and Metallurgy (AusIMM), Monograph 28.

Rapantova, N., Grmela, A. (2000). Water balance of coal mines and its determination under the conditions of humid climate of Central Europe. 7th International Mine Water Association Congress: Mine water and the environment. Ustroň, Poland, September 11-15 2000, pp. 615-623.

Rayne, S., Forest, K., Friesen, K.J. (2009). Analytical Framework for a Risk-based Estimation of Climate Change Effects on Mine Site Runoff Water Quality. Mine Water and the Environment, 28, pp. 124-135. http://dx.doi.org/10.1007/s10230-009-0070-z

Reap, J., Roman, F., Duncan, S., Bras, B. (2008). A survey of unresolved problems in life cycle assessment. Part 2: impact assessment and interpretation. International Journal of Life Cycle Assessment, 13, pp. 374-388. http://dx.doi.org/10.1007/s11367-008-0009-9

Reid, C., Bécaert, V., Aubertin, M., Rosenbaum, R.K., Deschênes, L. (2009). Life cycle assessment of mine tailings management in Canada. Journal of Cleaner Production, 17, pp. 471-479. http://dx.doi.org/10.1016/j.jclepro.2008.08.014

Ridoutt, B.G., Huang, J. (2014). Environmental relevance – the key to understanding water footprints. Proceedings of the National Academy of Science (PNAS), 109(22), E1424. http://dx.doi.org/10.1073/pnas.1203809109

Ridoutt, B.G., Pfister, S. (2013a). Towards an integrated family of footprint indicators. Journal of Industrial Ecology, 17(3), pp. 337-339. http://dx.doi.org/10.1111/jiec.12026

Ridoutt, B.G., Pfister, S. (2013b). A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator. International Journal of Life Cycle Assessment, 18(1), pp. 204-207. http://dx.doi.org/10.1007/s11367-012-0458-z

Rogelj, J., Meinshausen, M., Knutti, R. (2012). Global warming under old and new scenarios using IPCC climate sensitivity range estimates. Nature Climate Change, 2, pp. 248-253. http://dx.doi.org/10.1038/NCLIMATE1385

Ross, S., Evans, D. (2002). Excluding site-specific data from the LCA inventory: How this affects life cycle impact assessment. International Journal of Life Cycle Assessment, 7(3), pp. 141-150. http://dx.doi.org/10.1007/BF02994048

Rubel, F., Kottek, M. (2010). Observed and projected climate shifts 1901–2100 depicted by world maps of the Köppen-Geiger climate classification. Meteorologische Zeitschrift, 19(2), pp. 135-141. http://dx.doi.org/10.1127/0941-2948/2010/0430

Saarivuori, E., Wessman-Jääskeläinen, H., Unpublished Results. Water availability footprint of a mine product: case study. VTT Technical Research Centre of Finland.

Sahoo, L. J., Bandyopadhyay, S., Banerjee, R. (2014). Water and energy assessment for dewatering in opencast mines. Journal of Cleaner Production, 84, pp. 736-745. http://dx.doi.org/10.1016/j.jclepro.2014.07.072

Santero, N., Hendry, J. (2016). Harmonization of LCA methodologies for the metal and mining industry. International Journal of Life Cycle Assessment, 21(11), pp. 1543-1553. http://dx.doi.org/10.1007/s11367-015-1022-4

Sarver, E.A., Cox, A.S. (2013). Salt: an emerging water concern for the global mining and minerals industries. Mining Technology, 122(3), pp. 145-152. http://dx.doi.org/10.1179/147490013X13639459465655

Scherer, L., Pfister, S. (2015). Modelling spatially explicit impacts from phosphorus emissions in agriculture. International Journal of Life Cycle Assessment, 20(6), pp. 785-795. http://dx.doi.org/10.1007/s11367-015-0880-0

Scherer, L., Pfister, S. (2016). Dealing with uncertainty in water scarcity footprints. Environmental Research Letters, 11, 054008. http://dx.doi.org/10.1088/1748-9326/11/5/054008

Scherer, L., Venkatesh, A., Karuppiah, R., Pfister, S. (2015). Large-Scale Modeling for Calculating Water Stress Indices: Implications of Improved Spatiotemporal Resolution, Surface-Groundwater Differentiation, and Uncertainty Characterization. Environmental Science & Technology, 49, pp. 4971-4979. http://dx.doi.org/10.1021/acs.est.5b00429

Schultze, M. (2012). Maximising use of water produced by mining. International Mine Water Association 2012, Bunbury, Australia, pp. 11-19.

Schneider, L., Berger, M., Schüler-Hainsch, E., Knöfel, S., Ruhland, K., Mosig, J., Bach, V., Finkbeiner, M. (2014). The economic resource scarcity potential (ESP) for evaluating resource use based on life cycle assessment. International Journal of Life Cycle Assessment, 119(3), pp. 601-610. http://dx.doi.org/10.1007/s11367-013-0666-1

Schodde, R. (2011). Recent Trends in Mineral Exploration – Are we Finding Enough? Proceedings of RMG 8th Annual Mining and Exploration Investment Conference, Stockholm, Sweden, 15-16 November 2011, 39p.

Shang, Y., Lu, S., Li, X., Hei, P., Lei, X., Gong, J., Liu, J., Zhai, J., Wang, H. (2017). Balancing development of major coal bases with available water resources in China through 2020. Applied Energy, 194, pp. 735-750. http://dx.doi.org/10.1016/j.apenergy.2016.07.002

Sharma, V., Franks, D.M. (2013). In Situ Adaptation to Climatic Change: Mineral Industry Responses to Extreme Flooding Events in Queensland, Australia. Society & Natural Resources 26(11), pp. 1252-1267. http://dx.doi.org/10.1080/08941920.2013.797528

Sheppard, G., Gallardo, A., Hall, J., Strizek, M. (2009). Managing the cumulative groundwater impacts of multiple iron ore projects at Cape Preston (or … one plus one plus one plus one does not equal four). Mining Technology, 118(3/4), pp. 152-161. http://dx.doi.org/10.1179/174328610X12682159814911

Sinclair, L., Thompson, J. (2015). In situ leaching of copper: Challenges and future prospects. Hydrometallurgy, 57, pp. 306-324. http://dx.doi.org/10.1016/j.hydromet.2015.08.022

Singer, D.A. (2017). Future copper resources. Ore Geology Reviews, 86, pp. 271-279. http://dx.doi.org/10.1016/j.oregeorev.2017.02.022

Singer, D.A., Berger, V.I., Moring, B.C. (2008). Porphyry Copper Deposits of the World – Database and Grade and Tonnage Models. U.S. Geological Survey (USGS), Reston, Virginia, USA, Open-File Report 2008-1155.

SNL, 2017. SNL Metals & Mining Database. S&P Global Market Intelligence.

Sonderegger, T., Pfister, S., Hellweg, S. (2015). Criticality of Water: Aligning Water and Mineral Resources Assessment. Environmental Science & Technology, 49, pp. 12315-12323. http://dx.doi.org/10.1021/acs.est.5b02982

Sonderegger, T., Dewulf, J., Fantke, P., de Souza, D.M., Pfister, S., Stoessel, F., Verones, F., Vieira, M., Weidema, B., Hellweg, S. (2017). Towards harmonizing natural resources as an area of protection in life cycle impact assessment. International Journal of Life Cycle Assessment, in press, 16p. http://dx.doi.org/10.1007/s11367-017-1297-8

Song, X., Petterson, J.B., Pedersen, K.B., Røberg, S. (2017). Comparative life cycle assessment of tailings management and energy scenarios for a copper ore mine: A case study in Northern Norway. Journal of Cleaner Production, 164, pp. 892-904. http://dx.doi.org/10.1016/j.jclepro.2017.07.021

Soni, A.K., Wolkersdorfer, C. (2016). Mine water: policy perspective for improving water management in the mining environment with respect to developing economies. International Journal of Mining, Reclamation and Environment, 30(2), pp. 115-127. http://dx.doi.org/10.1080/17480930.2015.1011372

Soruco, L., Philippe, R. (2012). Upcoming trends in water supply costs for copper mining in arid regions. Water in Mining 2012, 3rd International Congress on Water Management in the Mining Industry, 6-8 June, Santiago, Chile, pp. 18-27.

Sparrow, G.J., Woodcock, J.T. Gold ore treatment at medium-size Australian gold plants. In: Australasian mining and metallurgy (Eds. Woodcock, J.T., Hamilton, J.K.), Volume 1 & 2, The Australian Institute of Mining and Metallurgy, Victoria (1993). pp. 1035-1059.

Sprecher, B., Daigo, I., Murakami, S., Kleijn, R., Vos, M., Kramer, G.J. (2015). Framework for Resilience in Material Supply Chains, With a Case Study from the 2010 Rare Earth Crisis. Environmental Science & Technology, 49(11), pp. 6740-6750. http://dx.doi.org/10.1021/acs.est.5b00206

Steen, B.A. (2006). Abiotic resource depletion, different perceptions of the problem with mineral deposits. International Journal of Life Cycle Assessment, 11(1), pp. 49-54. http://dx.doi.org/10.1065/lca2006.04.011

Steubing, B., Wernet, G., Reinhard, J., Bauer, C., Moreno-Ruiz, E. (2016). The ecoinvent database version 3 (part II): analyzing LCA results and comparison to version 2. International Journal of Life Cycle Assessment, 13p. http://dx.doi.org/10.1007/s11367-016-1109-6

Sverdrup, H.U., Ragnarsdottir, K.V., Koca, D. (2014). On modelling the global copper mining rates, market supply, copper price and the end of copper reserves. Resources, Conservation and Recycling, 87, pp. 158-174. http://dx.doi.org/10.1016/j.resconrec.2014.03.007

Swart, P., Dewulf, J. (2013). Quantifying the impacts of primary metal resource use in life cycle assessment based on recent mining data. Resources, Conservation and Recycling, 73, pp. 180-187. http://dx.doi.org/10.1016/j.resconrec.2013.02.007

SYKE. (2015). Groundwater Checklist. Suomen Ympäristökeskus (SYKE) – Finnish Environment Institute. Available at: www.syke.fi/groundwaterchecklist (accessed 04-07-2016)

TATA. (2013). Tata Industrial Water Footprint Assessment: Results and Learning. June 2013. Report prepared through collaboration between Tata Chemicals, Tata Motors, Tata Power, Tata Steel, Tata Quality Management Services, International Finance Corporate (IFC) and the Water Footprint Network.

Taylor, M.P., Little, J.A. (2013). Environmental impact of a major copper mine spill on a river and floodplain system. Anthropocene, 3, pp. 36-50. http://dx.doi.org/10.1016/j.ancene.2014.02.004

Tilton, J.E., Lagos, G. (2007). Assessing the long-run availability of copper. Resources Policy, 32, pp. 19-23. http://dx.doi.org/10.1016/j.resourpol.2007.04.001

Torrisi, C., Trotta, P. (2009). Sustainable water use at Olympic Dam. Mining Technology, 118(3/4), pp. 193-204. http://dx.doi.org/10.1179/174328610X12682159814984

TSM. (2016). Towards Sustainable Mining (TSM) initiative. The Mining Association of Canada. http://mining.ca/towards-sustainable-mining (accessed 04-07-2016)

Tukker, A. (2014). Rare Earth Elements Supply Restrictions: Market Failures, Not Scarcity, Hamper Their Current Use in High-Tech Applications. Environmental Science & Technology, 48, pp. 9973-9974. http://dx.doi.org/10.1021/es503548f

Tuominen, S.M., Kauppi, S.H., Nystén T.H. (2016). Poster, abstract : Groundwater Checklist - Metadata tool for groundwater protection and interaction. Eds: Staboulis S., Karvonen T & Kujanpää A. Bulletin of the Geological Society of Finland, Special Volume, Abstracts of The 32nd Nordic Geological Winter Meeting, 13th-15th January 2016, Helsinki, Finland. Geological Society of Finland.

Turpeinen, A., Rainio, R. (2013). Talvivaaraselvitys. Ympäristöministeriön raportteja 2/2013. [In Finnish]

UNEP. (2013). Environmental Risks and Challenges of Anthropogenic Metals Flows and Cycles, A Report of the Working Group on the Global Metal Flows to the International Resource Panel. Lead Editor: van der Voet, E. Chapter Editors: Salinen, R., Eckelman, M., Mudd, G., Norgate, T., Hischier, R. United Nations Environment Programme, 234p. ISBN: 978-92-807-3266-5

US EPA. (2017). Toxic Release Inventory (TRI) Program. United State Environmental Protection Agency. https://www.epa.gov/toxics-release-inventory-tri-program (last accessed 9-11-2017)

USGS. (2012). Minerals Yearbook 2011, Volume 1, Metals and Minerals. U.S. Geological Survey (USGS), Virginia, USA, 2012.

USGS. (2013). Metal prices in the United States through 2010: U.S. Geological Survey (USGS) Scientific Investigations Report 2012–5188, 204p. http://pubs.usgs.gov/sir/2012/5188 (accessed 14/07/2016)

USGS. (2016). Copper. Mineral Commodity Summaries, January 2016, U.S. Geological Survey (USGS), 54-55.

van der Ent, R.J., Savenije, H.H.G., Schaefli, B., Steele-Dunne, S.C. (2010). Origin and fate of atmospheric moisture over continents. Water Resources Research, 46, W09525. http://dx.doi.org/10.1029/2010WR009127

van Oers, L., de Koning, A., Guinée, J.B., Huppes, G. (2002). Abiotic resource depletion in LCA. Road and Hydraulic Engineering Institute, Ministry of Water and Transport, Amsterdam, 25 June 2002, 75p.

van Oers, L., Guinée, J. (2016). The Abiotic Depletion Potential: Background, Updates, and Future. Resources, Volume 5(1), 16. http://dx.doi.org/10.3390/resources5010016

Valero, A., Valero, A. (2012). From Grave to Cradle. Journal of Industrial Ecology, 17(1), pp. 43-52. http://dx.doi.org/10.1111/j.1530-9290.2012.00529.x

Välisalo, T. (2014). Kaivosten stressitestit 2013 [Stress tests for mines in 2013]. Finnish Ministry of Environment Report 2/2014. [In Finnish]

Vergara, M.R., Trianafyllidis, T. (2015). Swelling behavior of volcanic rocks under cyclic wetting and drying. International Journal of Rock Mechanics and Mining Sciences, 80, pp. 231-240. http://dx.doi.org/10.1016/j.ijrmms.2015.08.021

Vick, S.G. (1990). Planning, Design and Analysis of Tailings Dams. BiTech Publishers, Canada.

Vieira, M.D.M., Goedkoop, M.J., Storm, P., Huijbregt, M.A.J. (2012). Ore grade decrease as life cycle impact indicator for metal scarcity: The case of copper. Environmental Science & Technology, 46, pp. 12772-12778. http://dx.doi.org/10.1021/es302721t

Vogwill, R.I.J., Williamson, S., Kote, J. (2009). Hydrogeology of the coastal mining area, Lihir goldmine, Papua New Guinea. Australian Journal of Earth Sciences, 56(1), pp. 41-52. http://dx.doi.org/10.1080/08120090802541929

Wada, Y., van Beek, L.P.H., Viviroli, D., Dürr, H.H., Weingartner, R., Bierkens, M.F.P. (2011). Global monthly water stress: 2. Water demand and severity of water stress. Water Resources Research, 47, W07518, 17p. http://dx.doi.org/10.1029/2010WR009792

Wang, B., Peng, Y. (2014). The effect of saline water on mineral flotation – A critical review. Minerals Engineering, 66-68, pp. 13-24. http://dx.doi.org/10.1016/j.mineng.2014.04.017

Warner, A.E.M., Díaz, C.M., Dalvi, A.D., Mackey, P.J., Tarasov, A.V. (2006). JOM World Nonferrous Smelter Survey, Part III: Nickel: Laterite. JOM, 58(4), pp. 11-20. http://dx.doi.org/10.1007/s11837-006-0209-3

Warner, A.E.M., Díaz, C.M., Dalvi, A.D., Mackey, P.J., Tarasov, A.V., Jones, R.T. (2007). JOM world nonferrous smelter survey Part IV: Nickel: Sulfide. JOM, 59(4), pp. 58-72. http://dx.doi.org/10.1007/s11837-007-0056-x

WBCSD Global Water Tool. (2015). http://www.wbcsd.org/work-program/sector-projects/water/globalwater-tool.aspx

Wels, C., Robertson, A.M. (2003). Conceptual model for estimating water recovery in tailings impoundments. Tailings and Mine Wastes: Proceedings of the Tenth International Conference, 15, 87-94.

Welsh, W.D. (2006). Great Artesian Basin transient groundwater model. Bureau of Rural Sciences, Australian Government.

Weng, Z., Jowitt, S.M., Mudd, G.M., Haque, N. (2015). A detailed assessment of global rare earth resources: Opportunities and challenges. Economic Geology, 110, pp. 1925-1952. http://dx.doi.org/10.2113/econgeo.110.8.1925

Weng, Z., Haque, N., Mudd, G.M., Jowitt, S.M. (2016). Assessing the energy requirements and global warming potential of the production of rare earth elements. Journal of Cleaner Production, 139, pp. 1282-1297. http://dx.doi.org/10.1016/j.jclepro.2016.08.132

Werner, T.T., Mudd, G.M., Jowitt, S.M. (2015). Indium: key issues in assessing mineral resources and long-term supply from recycling. Applied Earth Science (Transactions of the Institute of Minerals and Metallurgy B), 124, pp. 213-226. http://dx.doi.org/10.1179/1743275815Y.0000000007

Werner, T.T., Mudd, G.M., Jowitt, S.M. (2017). The World's by-product and critical metal resources part II: A method for quantifying the resources of rarely reported metals. Ore Geology Reviews, 80, 658-675. http://dx.doi.org/10.1016/j.oregeorev.2016.08.008

Werner, T.T., Ciacci, L., Mudd, G., Reck, B., Northey, S. (2018). Looking Down Under for a circular economy of indium. Environmental Science & Technology, in press. http://dx.doi.org/10.1021/acs.est.7b05022

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B. (2016). The ecoinvent database version 3 (part I): overview and methodology. International Journal of Life Cycle Assessment, 21(9), 1218-1230. http://dx.doi.org/10.1007/s11367-016-1087-8

Wessman, H., Salmi, O., Kohl, J., Kinnunen, P., Saarivuori, E., Mroueh, U.-M. (2014). Water and Society: mutual challenges for eco-efficient and socially acceptable mining in Finland. Journal of Cleaner Production, 84, pp. 289-298. http://dx.doi.org/10.1016/j.jclepro.2014.04.026

West, J. (2011). Decreasing Metal Ore Grades, Are they really being driven by the depletion of high-grade deposits? Journal of Industrial Ecology, 15(2), pp. 165-168. http://dx.doi.org/10.1111/j.1530-9290.2011.00334.x

Williamson, S., Vogwill, R.I.J. (2001). Dewatering in the hot groundwater conditions at Lihir Gold. International Mine Water Association Symposium, Belo Horizonte, Brazil. 15p.

Wolkersdorfer, C., Bowell, R. (Editors). (2004). Contemporary Reviews of Mine Water Studies in Europe, Part 1. Mine Water and the Environment, 23, pp. 162-182. http://dx.doi.org/10.1007/s10230-004-0060-0

Wolkersdorfer, C., Bowell, R. (Editors). (2005a). Contemporary Reviews of Mine Water Studies in Europe, Part 2. Mine Water and the Environment, 24(1), pp. 2-37. http://dx.doi.org/10.1007/s10230-005-0068-0

Wolkersdorfer, C., Bowell, R. (Editors). (2005b). Contemporary Review of Mine Water Studies in Europe, Part 3. Mine Water and the Environment, 24(2), 58-76. http://dx.doi.org/10.1007/s10230-005-0074-2

Woodcock, J.T. (1980). Mining and Metallurgical Practices in Australasia, The Australasian Institute of Mining and Metallurgy, Victoria, 947p.

Woodcock, J.T., Hamilton, J.K. (1993). Australasian mining and metallurgy, Volume 1 & 2, The Australian Institute of Mining and Metallurgy, Victoria.

WRI Aqueduct Water Risk Atlas. (2013). World Resources Institute. http://www.wri.org/our-work/project/aqueduct

WWF-DEG Water Risk Filter. (2012). http://waterriskfilter.panda.org

WULCA. (2015). Consensual Method Development: Preliminary recommendation – AWaRe v1.1. Water Use in LCA (WULCA) working group.

WULCA. (2017). AWaRe version 1.2. Working Group for Water Use in Life Cycle Assessment (WULCA). Available from: http://www.wulca-waterlca.org (Last accessed 20/06/2017)

Yang, X.J., Lin, A., Li, X.-L., Wu, Y., Zhou, W., Chen, Z. (2013). China's ion-adsorption rare earth resources, mining consequences and preservation. Environmental Development, 8, pp. 131-136. http://dx.doi.org/10.1016/j.envdev.2013.03.006

Yaksic, A., Tilton, J.E. (2009). Using the cumulative availability curve to assess the threat of mineral depletion: The case of lithium. Resources Policy, 34, pp. 185-194. http://dx.doi.org/10.1016/j.resourpol.2009.05.002

Yano, S., Hanasaki, N., Itsubo, N., Oki, T. (2015). Water Scarcity Footprints by Considering the Differences in Water Sources. Sustainability, 7, pp. 9753-9772. http://dx.doi.org/10.3390/su7089753

Yap, A., Saconi, F., Nehring, M., Arteaga, F., Pinto, P., Asad, M.W.A., Oozhigin, S., Ozhigina, S., Mozer, D., Nagibin, A., Pul, E., Shvedov, D., Ozhigin, D., Gorokhov, D., Karatayeva, V. (2013). Exploiting metallurgical throughput-recovery relationship to optimise resource value as part of the production scheduling process. Minerals Engineering, 53, pp. 74-83. http://dx.doi.org/10.1016/j.mineng.2013.06.005

Yellishetty, M., Ranjith, P.G., Tharumarajah, A., Bhosale, S. (2009). Life cycle assessment in the minerals and metals sector: a critical review of selected issues and challenges. International Journal of Life Cycle Assessment, 14, pp. 257-267. http://dx.doi.org/10.1007/s11367-009-0060-1

Younger, P.L. (1997). The longevity of minewater pollution: a basis for decision-making. The Science of the Total Environment, 194/195, pp. 457-466. http://dx.doi.org/10.1016/S0048-9697(96)05383-1

Younger, P.L. (2006). The water footprint of mining operations in space and time – a new paradigm for sustainability assessments? Proceedings of Water in Mining 2006 (AusIMM), Brisbane, Australia, 14-16 November, pp. 13-22.

Younger, P.L., Banwart, S.A., Hedin, R.S. (2002). Mine Water: Hydrology, Pollution, Remediation. [Book]. Environmental Pollution series, vol. 5, Springer, The Netherlands. http://dx.doi.org/10.1007/978-94-010-0610-1

Ziemkiewicz, P.F., Skousen, J.G., Simmons, J. (2003). Long-term Performance of Passive Acid Mine Drainage Treatment Systems. Mine Water and the Environment, 22, pp. 118-129. http://dx.doi.org/10.1007/s10230-003-0012-0

Zhang, C., Anadon, L.D. (2013). Life Cycle Water Use of Energy Production and Its Environmental Impacts in China. Environmental Science & Technology, 47, pp. 14459-14467. http://dx.doi.org/10.1021/es402556x

Zhang, X., Yan, X. (2016). Deficiencies in the simulation of the geographic distribution of climate types by global climate models. Climate Dynamics, 46(9-10), pp. 2749-2757.

Zhang, A., Measham, T.G., Moffat, K. (2018). Preconditions for social licence: The importance of information in initial engagement. Journal of Cleaner Production, 172, pp. 1559-1566. http://dx.doi.org/10.1016/j.jclepro.2017.10.323

Appendices

A. Chapter 4 Supplementary Information

This appendix provides the supplementary information to the article presented in Chapter 4:

Northey, S.A., Mudd, G.M., Werner, T.T., Jowitt, S.M., Haque, N., Yellishetty, M., Weng, Z. (2017). The exposure of global base metal resources to water criticality, scarcity and climate change. Global Environmental Change, 44, pp. 109-124. http://dx.doi.org/10.1016/j.gloenvcha.2017.04.004

A.1. Description of Water Risk and Impact Indices

A.1.1. Water criticality and component indices

Water Criticality (CRIT) is a composite indicator that is a function of the water supply risk (SR), vulnerability to supply restrictions (VSR) and the environmental implications (EI) of water use within a region (Sonderegger et al., 2015). Each of these indices is based upon the weighting of several sub-metrics. SR is determined based upon geopolitical considerations (political stability, upstream water stress, supply concentration) and hydrology (using the water stress index described below), whereas VSR accounts for the importance of water to the local economy, the ability to compensate for supply shortfalls (compensation potential, water stress ratio, dependency ratio, environmental impact ratio) and the susceptibility of the region to supply shortfalls (adaptive capacity, dam capacity and supply dependency). EI accounts for the human health and ecosystem impacts associated with regional water use, determined using data provided by Pfister et al. (2011) based upon the ReCiPe life cycle impact assessment method (Goedkoop et al., 2009). An advantage of these indices is the recognition of the importance of infrastructure, institutions and governance when assessing water risks. Spatial data for CRIT, SR, VSR and EI are available on a scale from 0 to 100 at a province/state level for most countries (Sonderegger et al., 2015), although data are unavailable for some countries with base metal resources, particularly island nations such as New Caledonia (refer to Table 1 and Figure 2 in the main article [Chapter 4] for data coverage).

A.1.2. Water stress, scarcity, depletion and evaporation recycling indices

Water stress, scarcity, depletion and evaporation recycling indices provide an understanding of the hydrology and water use occurring in different regions and have been used for a range of purposes, including to assess the potential freshwater use impacts within water footprint and life cycle assessment studies. The UNEP-SETAC Life Cycle Initiative working group for water use in life cycle assessment developed the Available Water Remaining (AWaRe) method as a consensus based indicator for assessing water use impacts in life cycle assessment (Boulay et al., 2016; 2017; WULCA, 2017). This method measures the inverse of water availability minus water demand associated with environmental water requirements and human water consumption. The AWaRe indicator can be interpreted as being a proxy for the potential of water consumption to deprive other users of water. Values are normalised against the global average and limited to between 0.1 and 100, and so an AWaRe value of 20 indicates that there is 20 times less unused water per unit area than the world average. Spatial datasets for AWaRe are provided by WULCA (2017) for individual sub-watersheds, separated into agricultural and non-agricultural water. This study uses the factors specific to non-agricultural water use.

Blue water scarcity (BWS) was assessed by Hoekstra et al. (2012) as the ratio of domestic blue water consumption to availability within a region, where blue water was defined as groundwater and surface water resources, excluding rainfall and surface soil moisture. Blue water availability was estimated as being 20% of the natural surface runoff to account for environmental flow requirements. A BWS greater than 1 indicates that water resources in a region are being overexploited. BWS is available on a five by five arc minute mesh for major river basins only (Hoekstra et al., 2012) and so the dataset does not cover all regions containing base metal resources (see Table 1 and Figure 2 in the main article; Chapter 4).

The Water Accounting and Vulnerability Evaluation (WAVE) method developed by Berger et al. (2014) addresses a variety of shortcomings associated with other indices and assessments of water use. Three main spatial indices underpin the WAVE methodology, namely the basin-internal evaporation recycling (BIER) ratio, the hydrologically effective basin-internal evaporation recycling (BIER-h) ratio, and the Water Depletion Index (WDI). The BIER ratio provides an estimate of the proportion of evaporation that will re-precipitate within the same drainage basin or catchment. BIER-h is an extension of this that represents the proportion of evaporation re-precipitation that results in surface runoff or groundwater recharge in the same basin, and so excludes the fraction that re-evapotranspirates. Water that re-precipitates in this way is used but not ‘consumed’ as it will still become available in a reasonable time horizon to meet water use requirements within the basin. The WDI provides an estimate of the relative exposure of a region to freshwater depletion. The determination of the WDI is based upon water consumption-to-availability ratios, surface and groundwater stores, and the general aridity of the basin. A high WDI indicates that water consumption will effectively lead to long-term depletion of water resources within the catchment, whereas low WDI values indicate that the impact of water consumption (at least from a hydrological perspective) will be relatively short-lived.

The water stress index (WSI) was developed by Pfister et al. (2009) to assess the potential for water deprivation as part of life cycle assessment studies. The WSI is based upon regional withdrawal-to-availability (WTA) ratios that have been multiplied by a variation factor that accounts for inter-and intra- annual precipitation variability, which are then normalised between 0.01 and 1. Annual WSI data for individual watersheds was used as part of this study (Pfister et al., 2009).

The ratio of regional water withdrawals-to-availability (WTA) is a conceptually simple indicator to determine the relative water scarcity of regions. WTA ratios have been used in the calculation of several of the indicators used by this study (i.e. the WSI and as a consequence SR and CRIT). The WTA ratios that form the basis of the WSI were taken from the results of the WaterGAP 2 global hydrological model (Alcamo et al., 2003b). The WTA ratio of regions containing individual mineral deposits are shown in the electronic Supplementary Tables S.24 to S.26 [available from http://dx.doi.org/. However, no aggregated WTA data is presented, as the presence of extremely high WTA ratios (i.e. 1011) for several watersheds (due to very low water availability) leads to highly skewed weighted averages. For this reason, this study only presents weighted averages for indices that are bounded or normalised.

A.2. Electronic Supplementary Figures S.1 to S.12

List of Figures

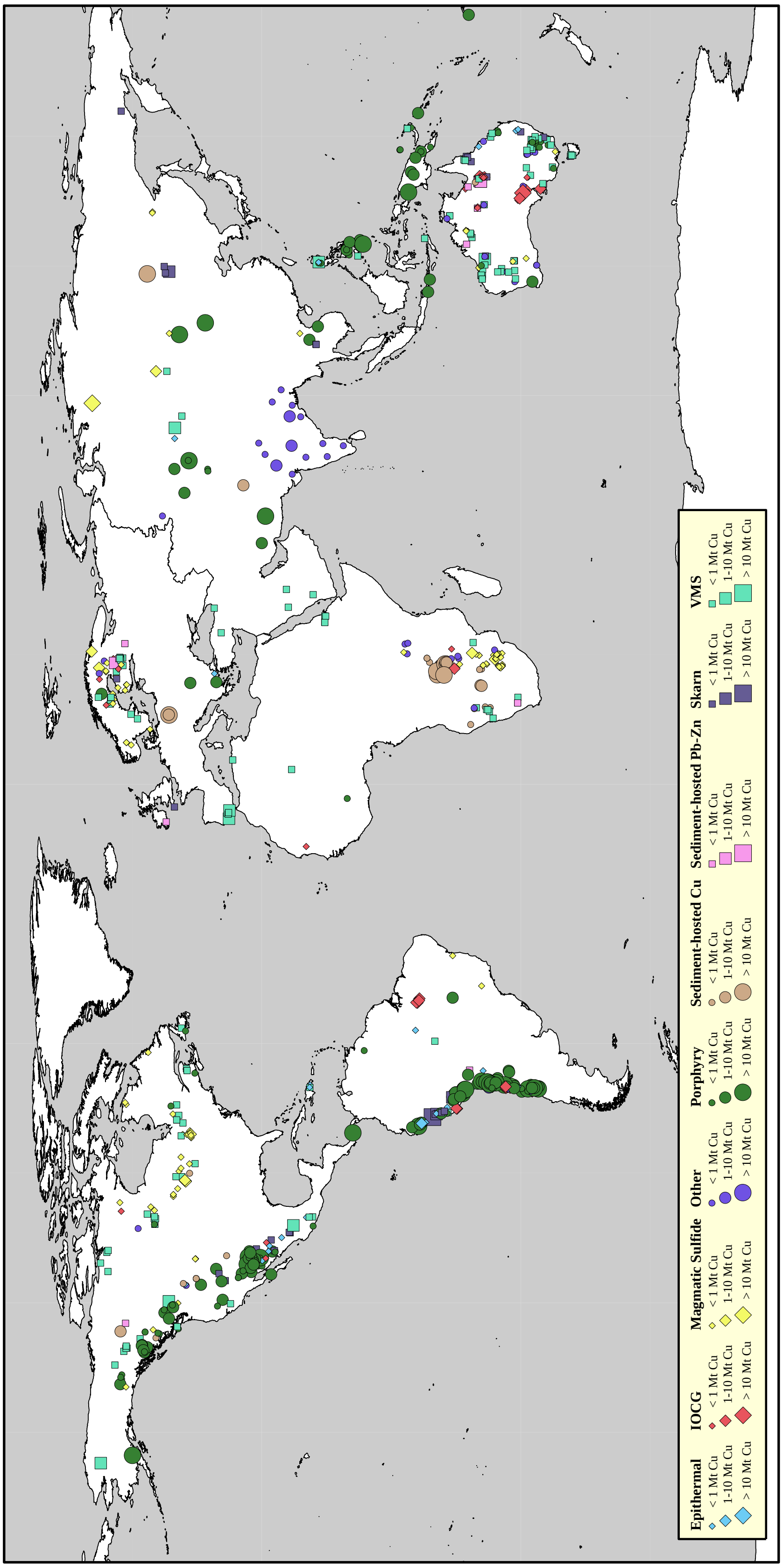

Figure S. 1: Copper deposits classified according to deposit type and resource size (Mudd et al., 2013). Location data sources described in section 2.1 of the main article [Chapter 4]. Abbreviations: IOCG (iron oxide copper gold), VMS (volcanogenic massive sulfide).

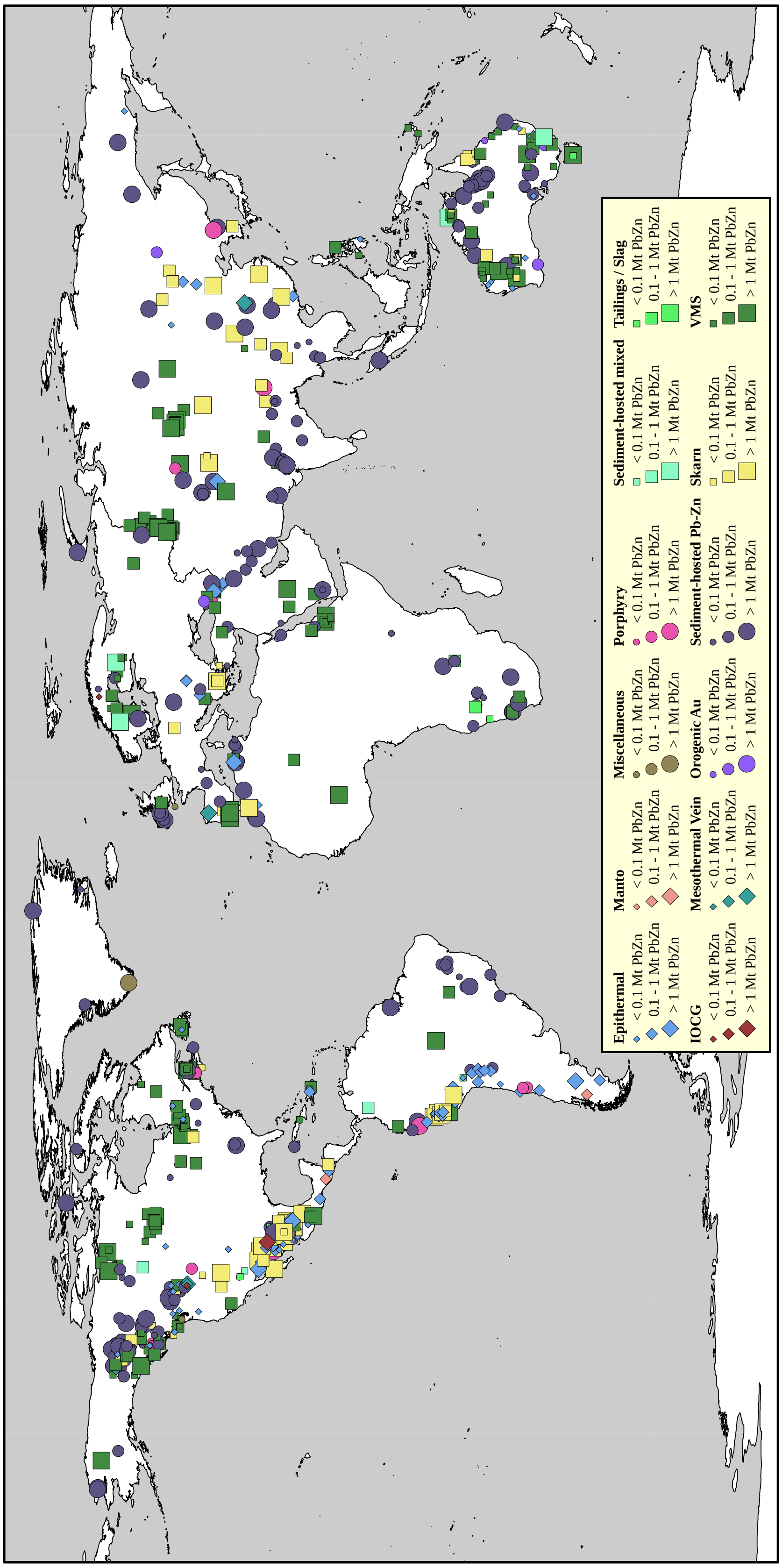

Figure S. 2: Lead-Zinc deposits classified according to deposit type and resource size (Mudd et al., 2017). Location data sources described in section 2.1 of the main article [Chapter 4]. Abbreviations: IOCG (iron oxide copper gold), VMS (volcanogenic massive sulfide).

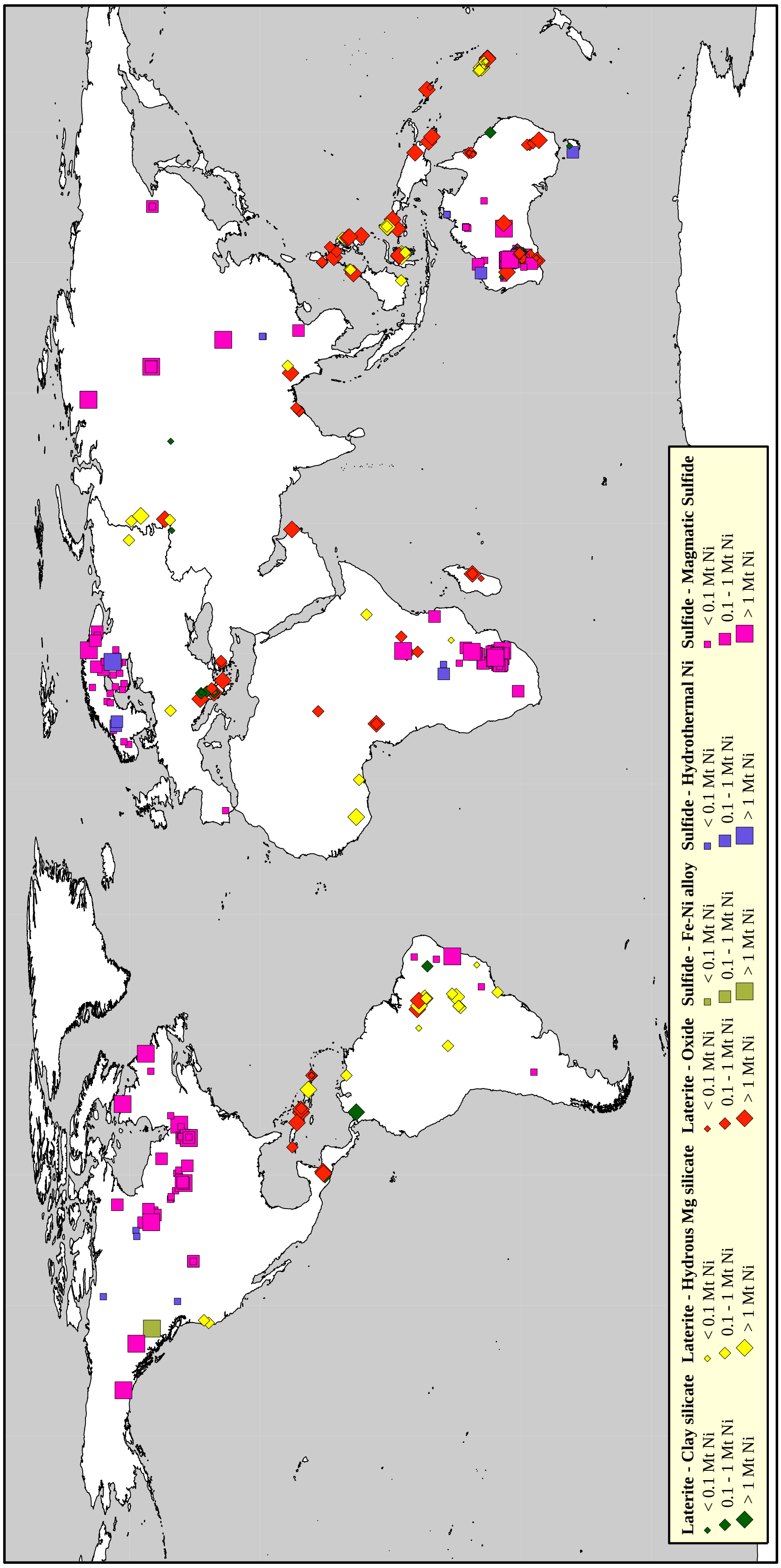

Figure S. 3: Nickel deposits classified according to deposit type and contained copper (Mudd and Jowitt, 2014). Location data sources described in section 2.1 of the main article [Chapter 4].

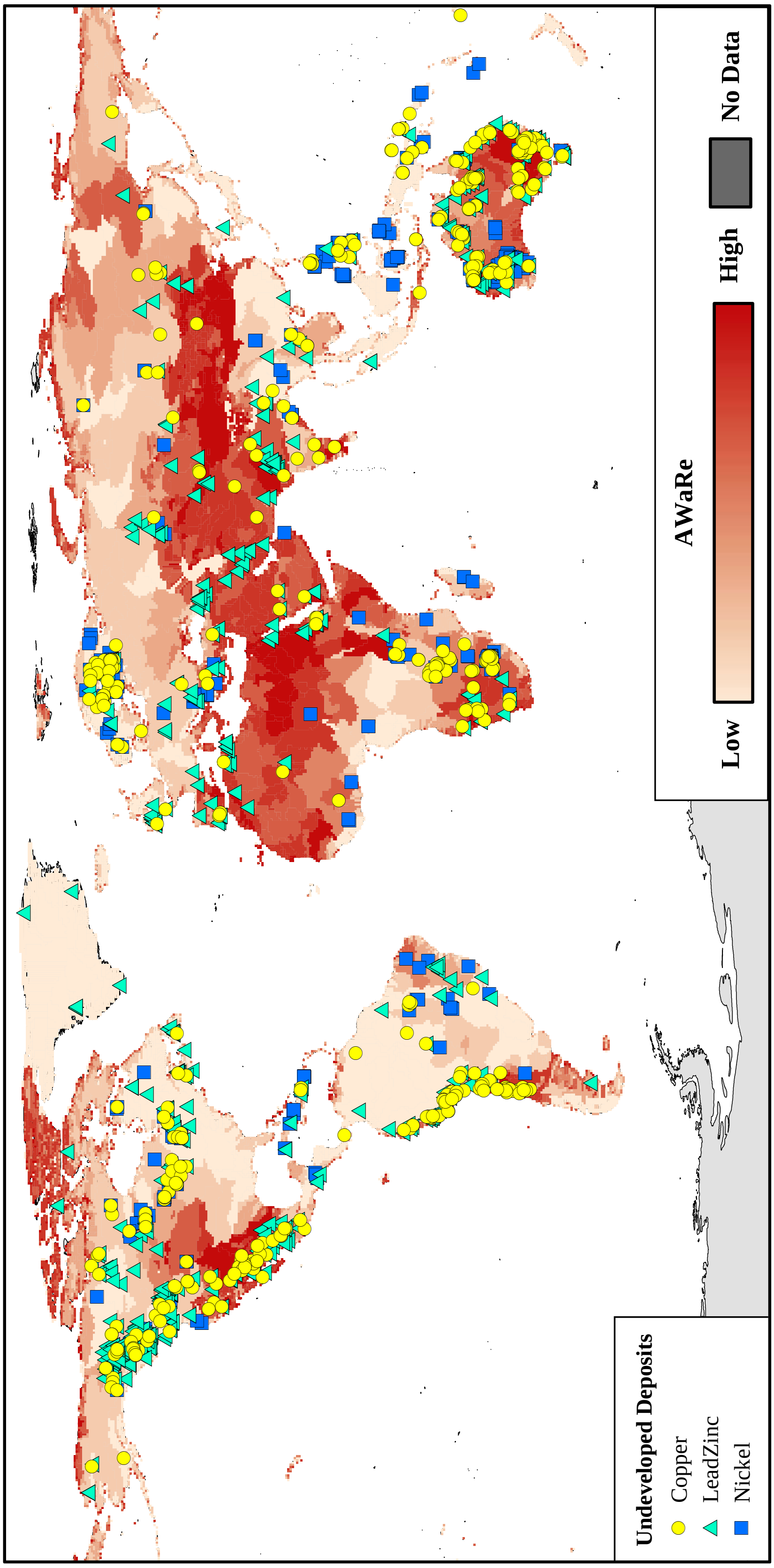

Figure S. 4: The location of undeveloped copper, lead-zinc and nickel deposits in relation to the AWaRe index. Resource data sources: copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see section 2.1 of the article [Chapter 4]).

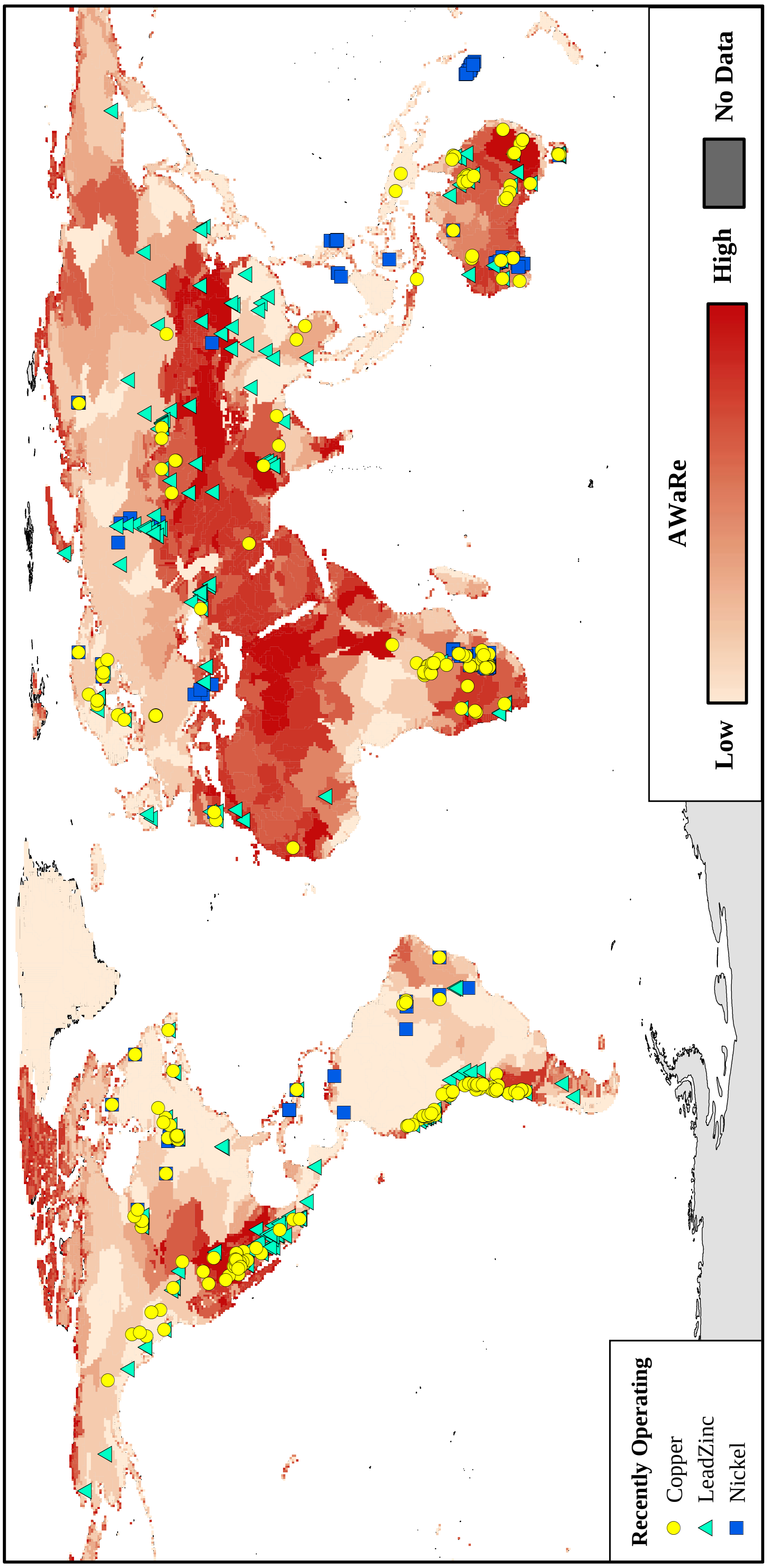

Figure S. 5: The location of recently operating copper, lead-zinc and nickel projects in relation to the AWaRe index. Resource data sources: copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see section 2.1 of the article [Chapter 4]).

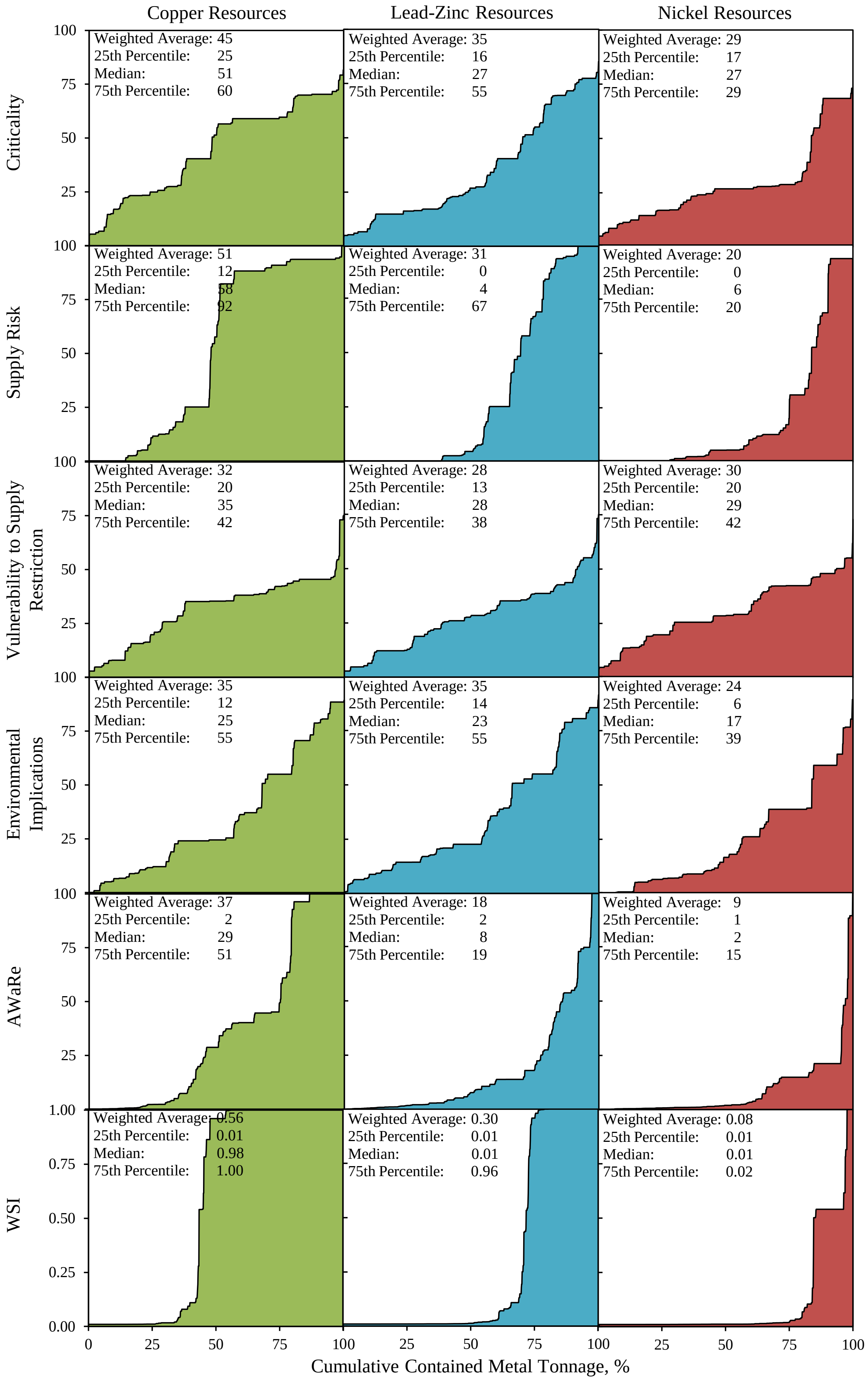

Figure S. 6: Water indices plotted against the cumulative contained metal tonnage of individual copper, lead-zinc and nickel mineral resources.

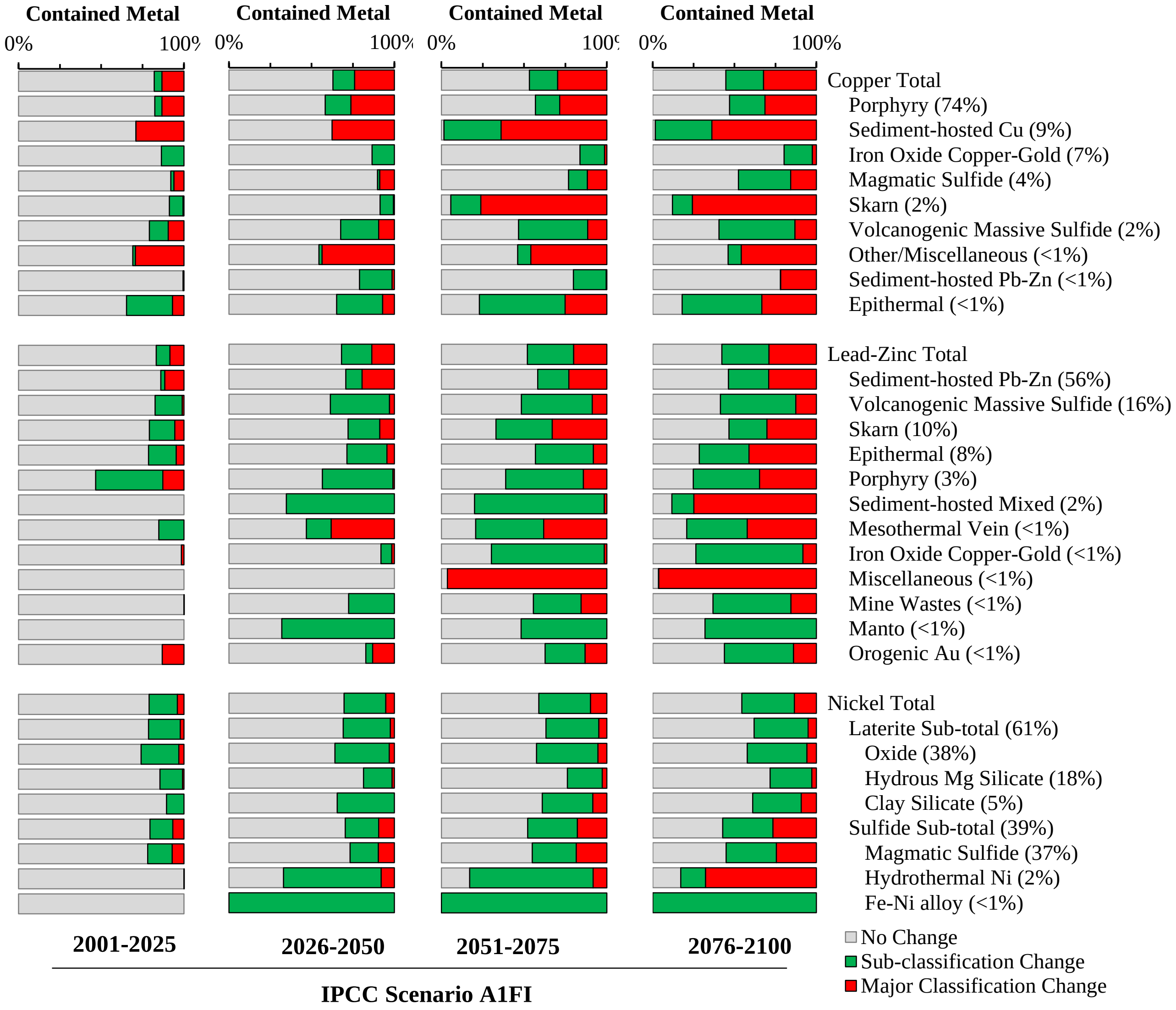

Figure S. 7: Proportion of global copper, lead-zinc and nickel resources in regions with a changing Köppen-Geiger climate classification under the IPCC Scenario A1FI (Rubel and Kottek, 2010). Expressed relative to the observed period 1951-2000 (Kottek et al., 2006). The distribution of copper, lead-zinc and nickel contained in individual deposit types is shown as a percentage in brackets. Resource data sources: copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see section 2.1 of the article [Chapter 4]).

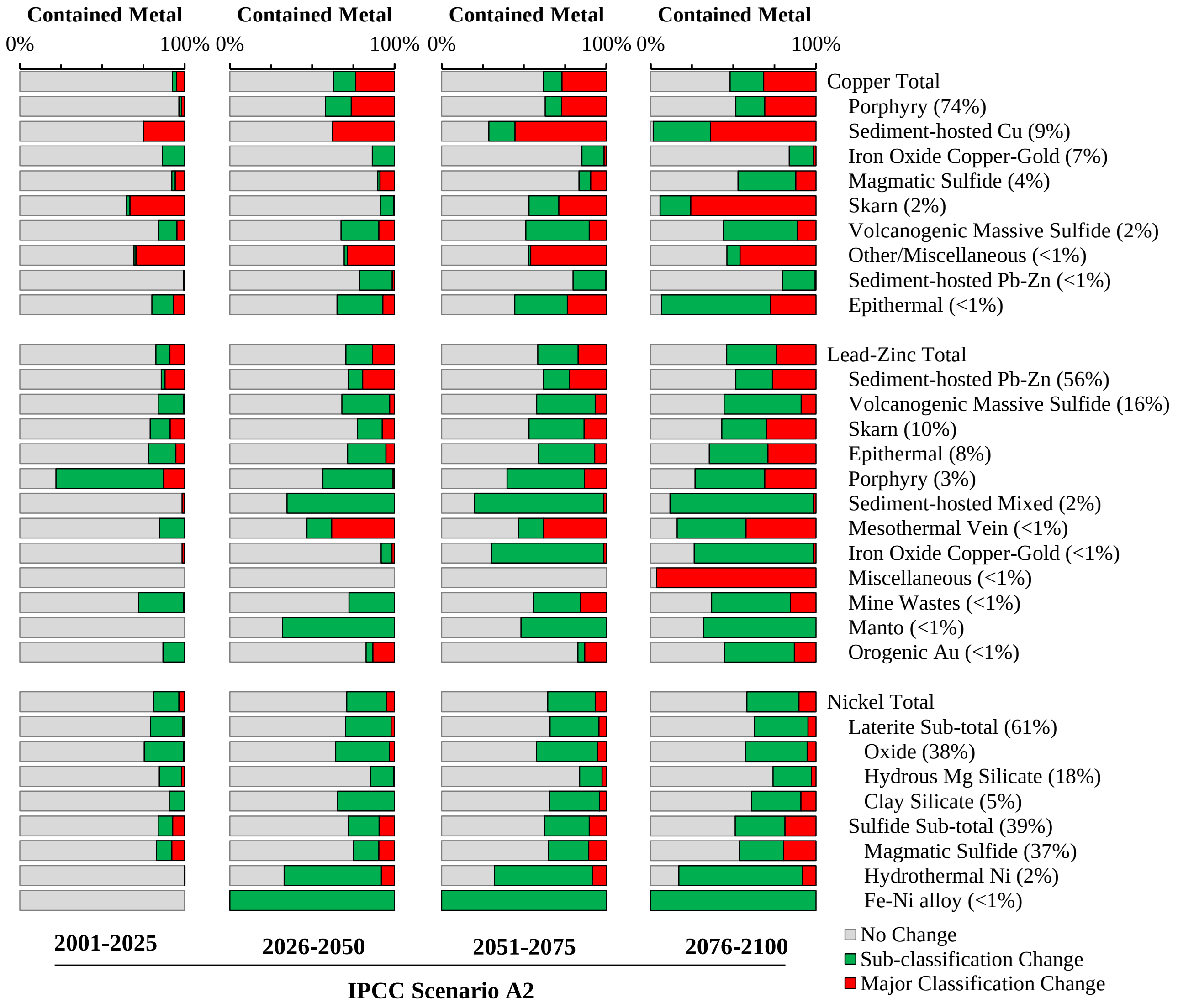

Figure S. 8: Proportion of global copper, lead-zinc and nickel resources in regions with a changing Köppen-Geiger climate classification under the IPCC Scenario A2 (Rubel and Kottek, 2010). Expressed relative to the observed period 1951-2000 (Kottek et al., 2006). The distribution of copper, lead-zinc and nickel contained in individual deposit types is shown as a percentage in brackets. Resource data sources: copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see section 2.1 of the article [Chapter 4]).

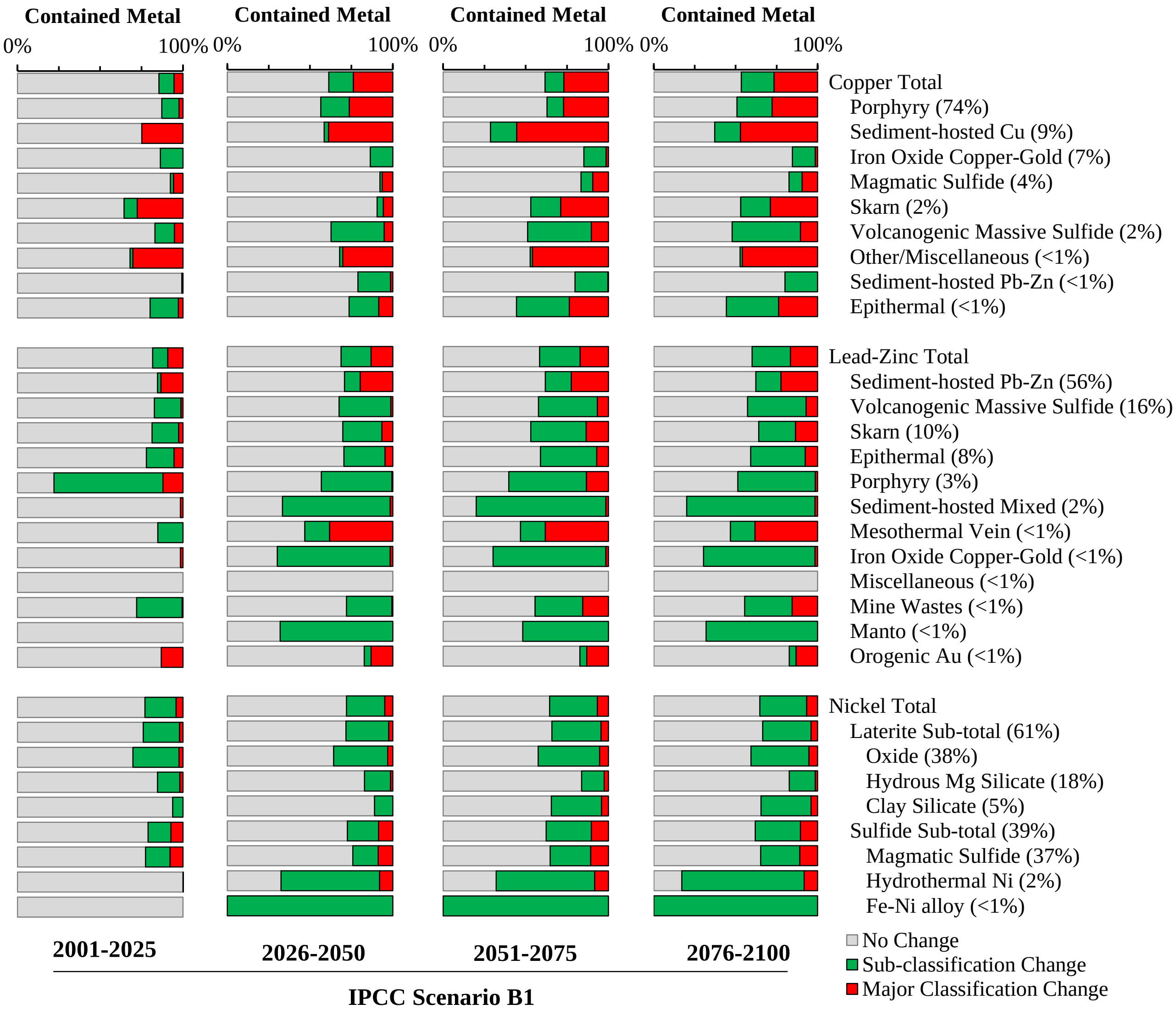

Figure S. 9: Proportion of global copper, lead-zinc and nickel resources in regions with a changing Köppen-Geiger climate classification under the IPCC Scenario B1 (Rubel and Kottek, 2010). Expressed relative to the observed period 1951-2000 (Kottek et al., 2006). The distribution of copper, lead-zinc and nickel contained in individual deposit types is shown as a percentage in brackets. Resource data sources: copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see section 2.1 of the article [Chapter 4]).

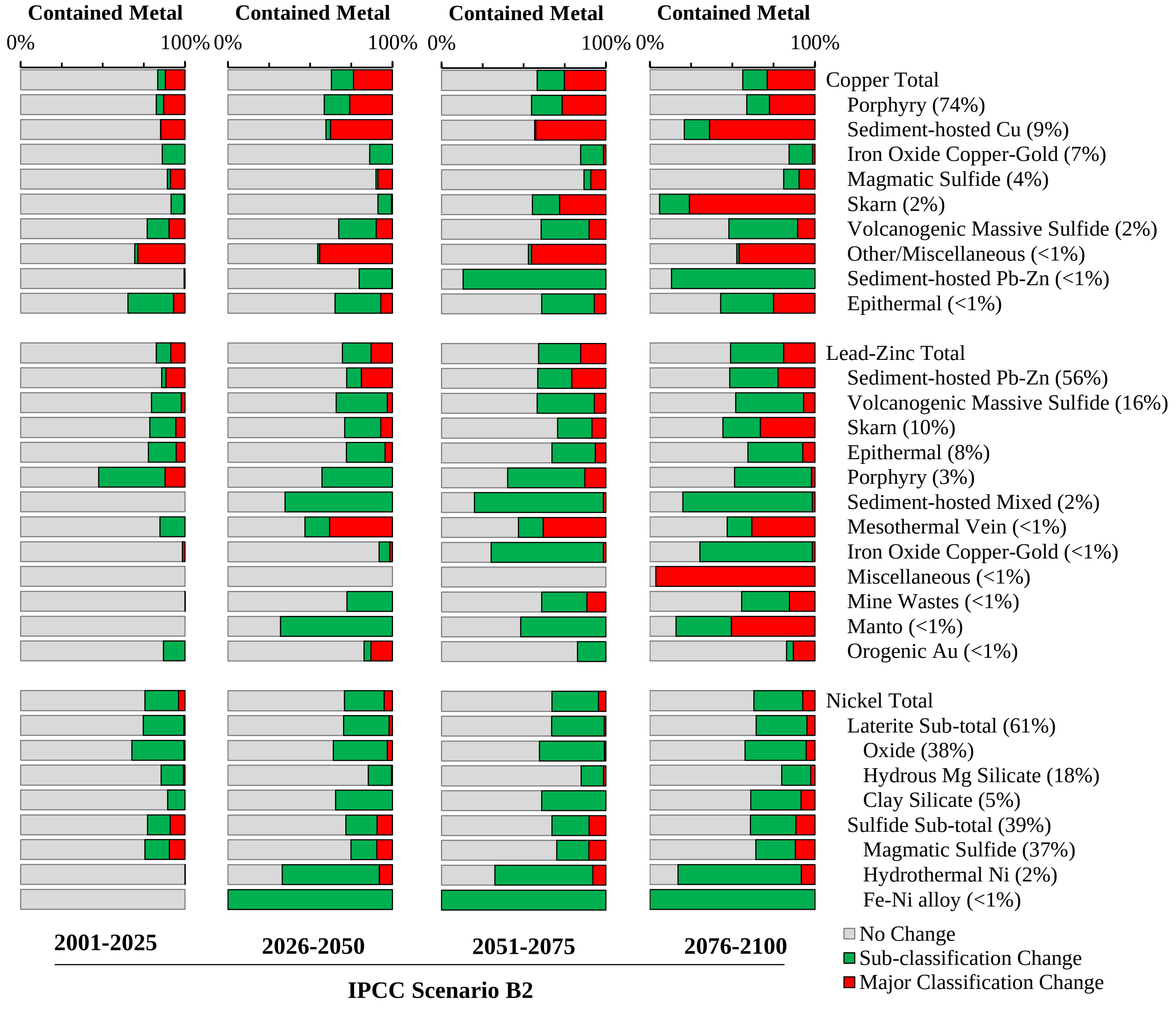

Figure S. 10: Proportion of global copper, lead-zinc and nickel resources in regions with a changing Köppen-Geiger climate classification under the IPCC Scenario B2 (Rubel and Kottek, 2010). Expressed relative to the observed period 1951-2000 (Kottek et al., 2006). The distribution of copper, lead-zinc and nickel contained in individual deposit types is shown as a percentage in brackets. Resource data sources: copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see section 2.1 of the article [Chapter 4]).

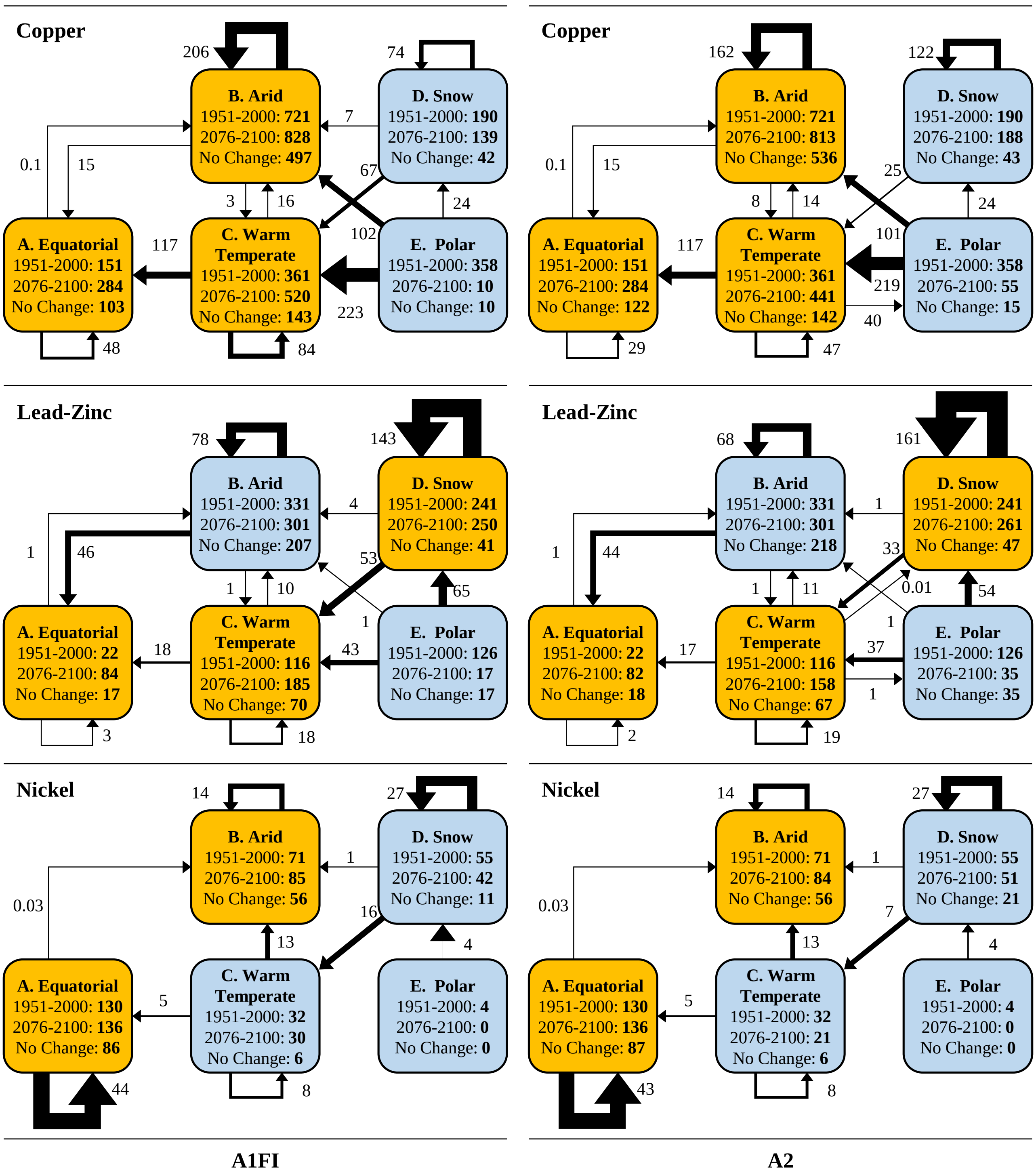

Figure S. 11: Changes in the Köppen-Geiger climate classification of regions containing base metals from 1951-2000 (Kottek et al., 2006) to 2076-2100 for the IPCC emissions scenario A1FI and A2 (Rubel and Kottek, 2010). Values indicate million tonnes of contained metal. Flow width is in proportion to the global resource for the individual metal(s). Flows returning to the same climate classification represent a change in precipitation or temperature sub-classification. Yellow and blue shading indicates a respective net increase or decrease in contained resource. Resource data sources: copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see section 2.1 of the main article [Chapter 4]).

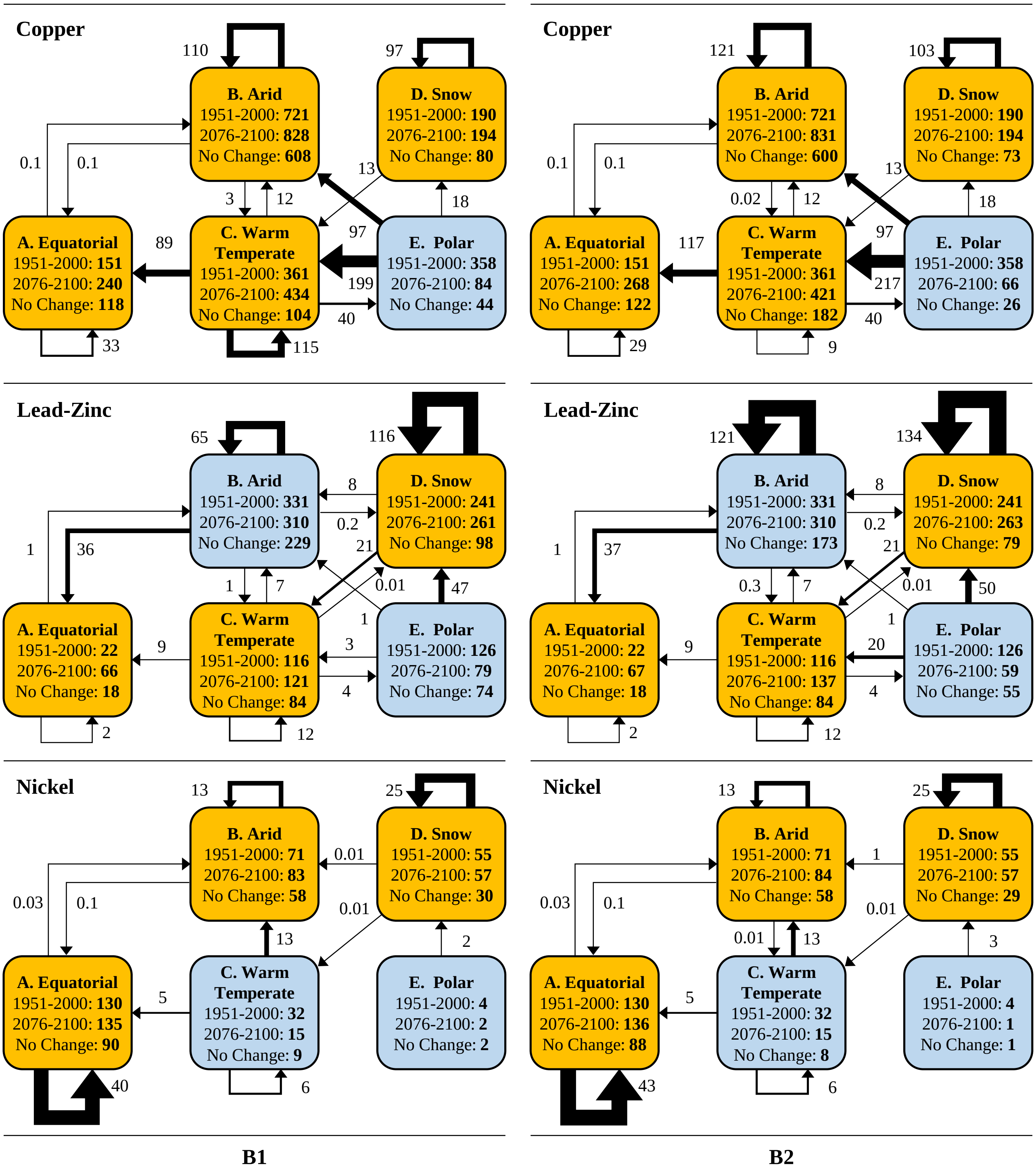

Figure S. 12: Changes in the Köppen-Geiger climate classification of regions containing base metals from 1951-2000 (Kottek et al., 2006) to 2076-2100 for the IPCC emissions scenario B1 and B2 (Rubel and Kottek, 2010). Values indicate million tonnes of contained metal. Flow width is in proportion to the global resource for the individual metal(s). Flows returning to the same climate classification represent a change in precipitation or temperature sub-classification. Yellow and blue shading indicates a respective net increase or decrease in contained resource. Resource data sources: copper (Mudd et al., 2013), lead-zinc (Mudd et al., 2017), nickel (Mudd and Jowitt, 2014), resource locations (see section 2.1 of the main article [Chapter 4]).

A.3. Electronic Supplementary Tables S.1 to S.26

Due to the extensive nature of supplementary tables S.1 to S.26 associated with the article (Chapter 4; Northey et al., 2017a), reformatting these tables for inclusion in this thesis would have resulted in an additional 600-800 pages of tabulated data. Instead, the readers are strongly encouraged to refer to the online version of the article to explore the full breadth and detail of the available datasets. Access to the online version is available via: **http://dx.doi.org/10.1016/j.gloenvcha.2017.04.004**

If you or your institution does not have subscription access to Global Environmental Change, then please contact the author directly for a copy of these datasets.

This dataset is provided as a Microsoft Excel workbook (.xlsx) and contains the following tables:

Table S.1: Summary of water indices results for major deposit types and countries with large resource endowments.

Table S.2: Weighted average water indices for copper resources by country.

Table S.3: Weighted average water indices for lead-zinc resources by country.

Table S.4: Weighted average water indices for nickel resources by country.

Table S.5: Weighted average water indices for copper resources by country and deposit type.

Table S.6: Weighted average water indices for lead-zinc resources by country and deposit type.

Table S.7: Weighted average water indices for nickel resources by country and deposit type.

Table S.8: Base metal resources contained in each Köppen-Geiger climate classification.

Table S.9: Copper resource contained in each Köppen-Geiger climate classification by deposit type.

Table S.10: Lead-zinc resource contained in each Köppen-Geiger climate classification by deposit type.

Table S.11: Nickel resource contained in each Köppen-Geiger climate classification by deposit type.

Table S.12: Copper resource contained in each major Köppen-Geiger climate classification by country.

Table S.13: Lead-zinc resource contained in each major Köppen-Geiger climate classification by country.

Table S.14: Nickel resource contained in each major Köppen-Geiger climate classification by country.

Table S.15: Base metal resources in regions with changing major and minor Köppen-Geiger climate classifications.

Table S.16: Changes to major Köppen-Geiger climate classification of regions containing base metal resources.

Table S.17: Changes to major Köppen-Geiger climate classification of regions containing copper resources by deposit type.

Table S.18: Changes to major Köppen-Geiger climate classification of regions containing lead-zinc resources by deposit type.

Table S.19: Changes to major Köppen-Geiger climate classification of regions containing nickel resources by deposit type.

Table S.20: Changes to Köppen-Geiger climate sub-classification of regions containing base metal resources.

Table S.21: Changes to Köppen-Geiger climate sub-classification of regions containing copper resources by deposit type.

Table S.22: Changes to Köppen-Geiger climate sub-classification of regions containing lead-zinc resources by deposit type.

Table S.23: Changes to Köppen-Geiger climate sub-classification of regions containing nickel resources by deposit type.

Table S.24: Copper resource dataset, showing intersection with water indices and climate classifications.

Table S.25: Lead-zinc resource dataset, showing intersection with water indices and climate classifications.

Table S.26: Nickel resource dataset, showing intersection with water indices and climate classifications.

B. Chapter 5 Appendix Figures and Tables

Contents:

Figure A. : The cumulative production distribution of 25 mined commodities in relation to watershed and national average AWaRe factors for non-agricultural water use (Boulay et al., 2016, 2017; WULCA, 2017). Watershed factors are shown for the production distribution of individual mining operations (SNL, 2017), whereas national average factors are shown for the production distribution of individual countries (British Geological Survey, 2016).

Figure A. 2: The cumulative production distribution of 25 mined commodities in relation to watershed and national average water stress index (WSI)(Pfister et al., 2009). Watershed factors are shown for the production distribution of individual mining operations (SNL, 2017), whereas national average factors are shown for the production distribution of individual countries (British Geological Survey, 2016).

Figure A. 3: Cumulative operation production distribution for 25 mined commodities (SNL, 2017) of the ratio between watershed and national average AWaRe factors for non-agricultural water use (Boulay et al., 2016, 2017; WULCA, 2017).

Figure A. 4: Cumulative operation production distribution for 25 mined commodities4 of the ratio between watershed and national average water stress index (WSI)(Pfister et al., 2009).

Figure A. 6: Cumulative distribution of operation (SNL, 2017) and national (British Geological Survey, 2016) datapoints falling within 1 standard deviation of the spatial uncertainty associated with national average AWaRe factors (Boulay et al., 2016, 2017; WULCA, 2017).

Figure A. 5: Cumulative operation production distribution for 25 mined commodities (SNL, 2017) in relation to watershed AWaRe factors for non-agricultural water use (Boulay et al., 2016, 2017; WULCA, 2017), measured relative to the national average factor and expressed in terms of the standard deviation of spatial uncertainty associated with each national average.

Figure A. 7: Commodity production weighted averages determined using national average and watershed WSI (Pfister et al., 2009) and AWaRe factors for non-agricultural water use (Boulay et al., 2016, 2017; WULCA, 2017). Commodities are sorted based upon watershed production averages. Shading represents the degree of production coverage of the operation data (SNL, 2017) in relation to the sum of national production data (British Geological Survey, 2016).

Table A. 1 to Table A. 5: National average WSI (Pfister et al., 2009), AWaRe factors for non-agricultural water use (Boulay et al., 2016; 2017; WULCA, 2017) and national mined commodity production data for 2014 (BGS, 2016).

Table A. 6 to Table A. 30: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted.

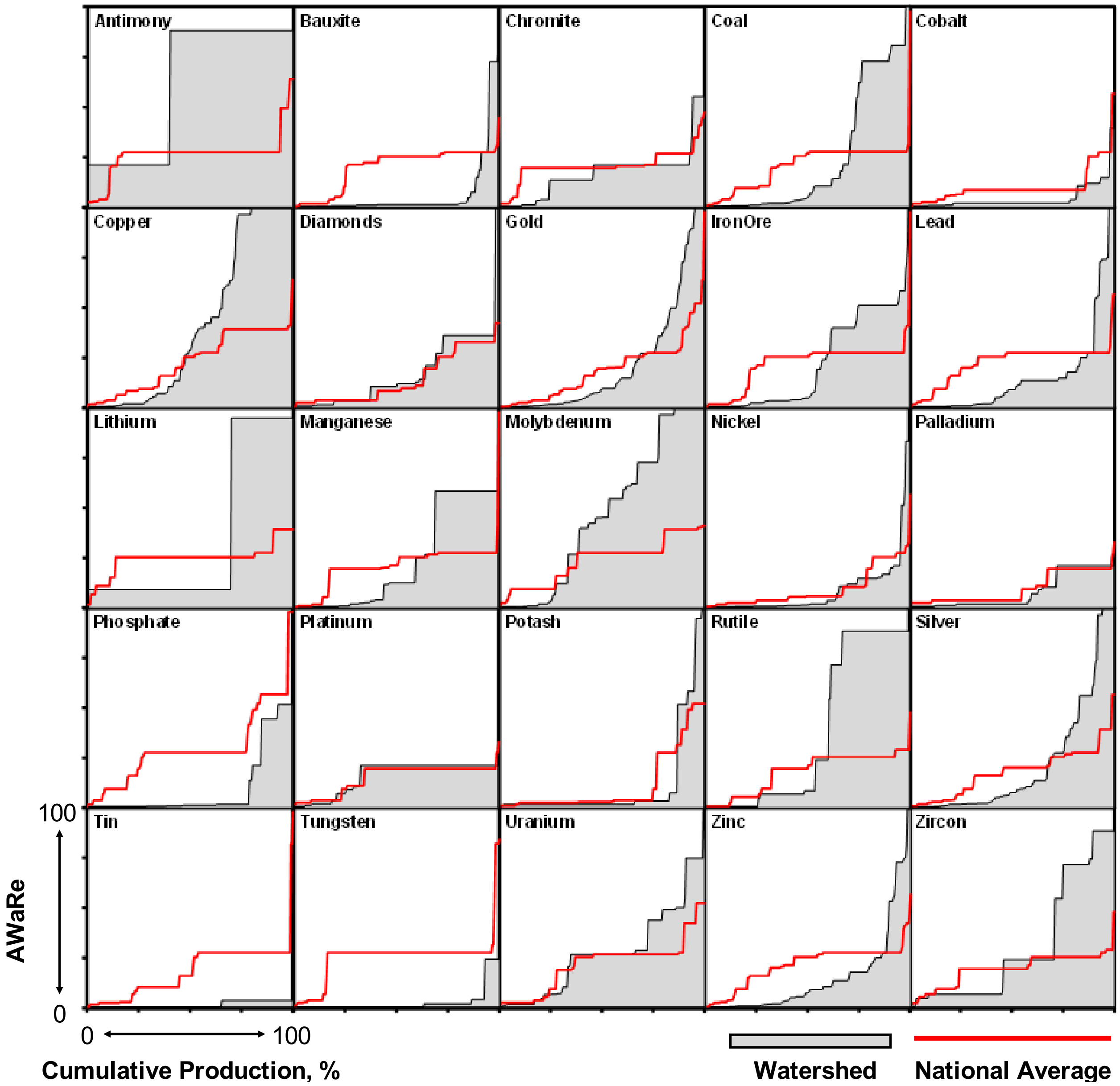

Figure A. 1: The cumulative production distribution of 25 mined commodities in relation to watershed and national average AWaRe factors for non-agricultural water use (Boulay et al., 2016, 2017; WULCA, 2017). Watershed factors are shown for the production distribution of individual mining operations (SNL, 2017), whereas national average factors are shown for the production distribution of individual countries (British Geological Survey, 2016).

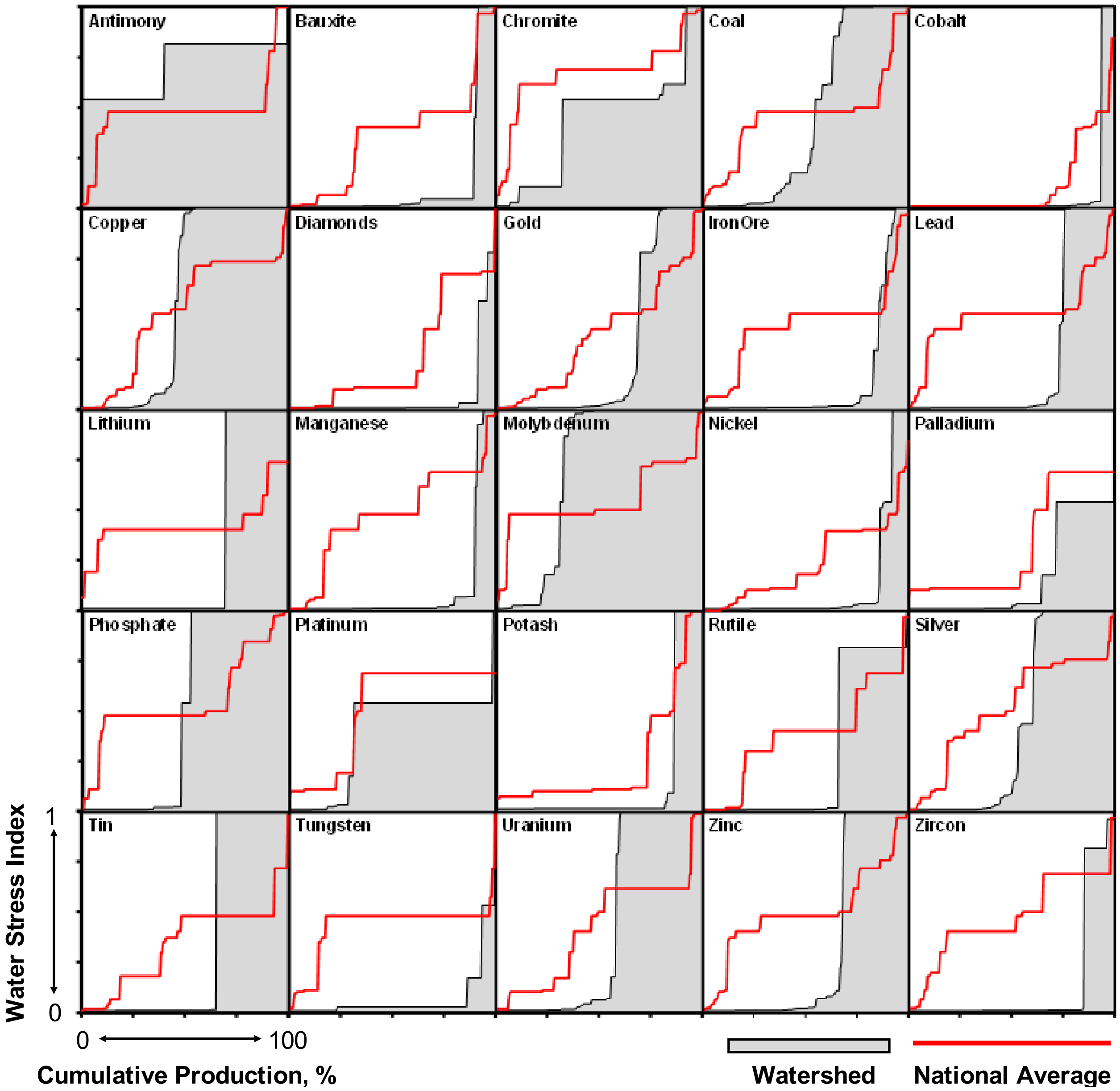

Figure A. 2: The cumulative production distribution of 25 mined commodities in relation to watershed and national average water stress index (WSI)(Pfister et al., 2009). Watershed factors are shown for the production distribution of individual mining operations (SNL, 2017), whereas national average factors are shown for the production distribution of individual countries (British Geological Survey, 2016).

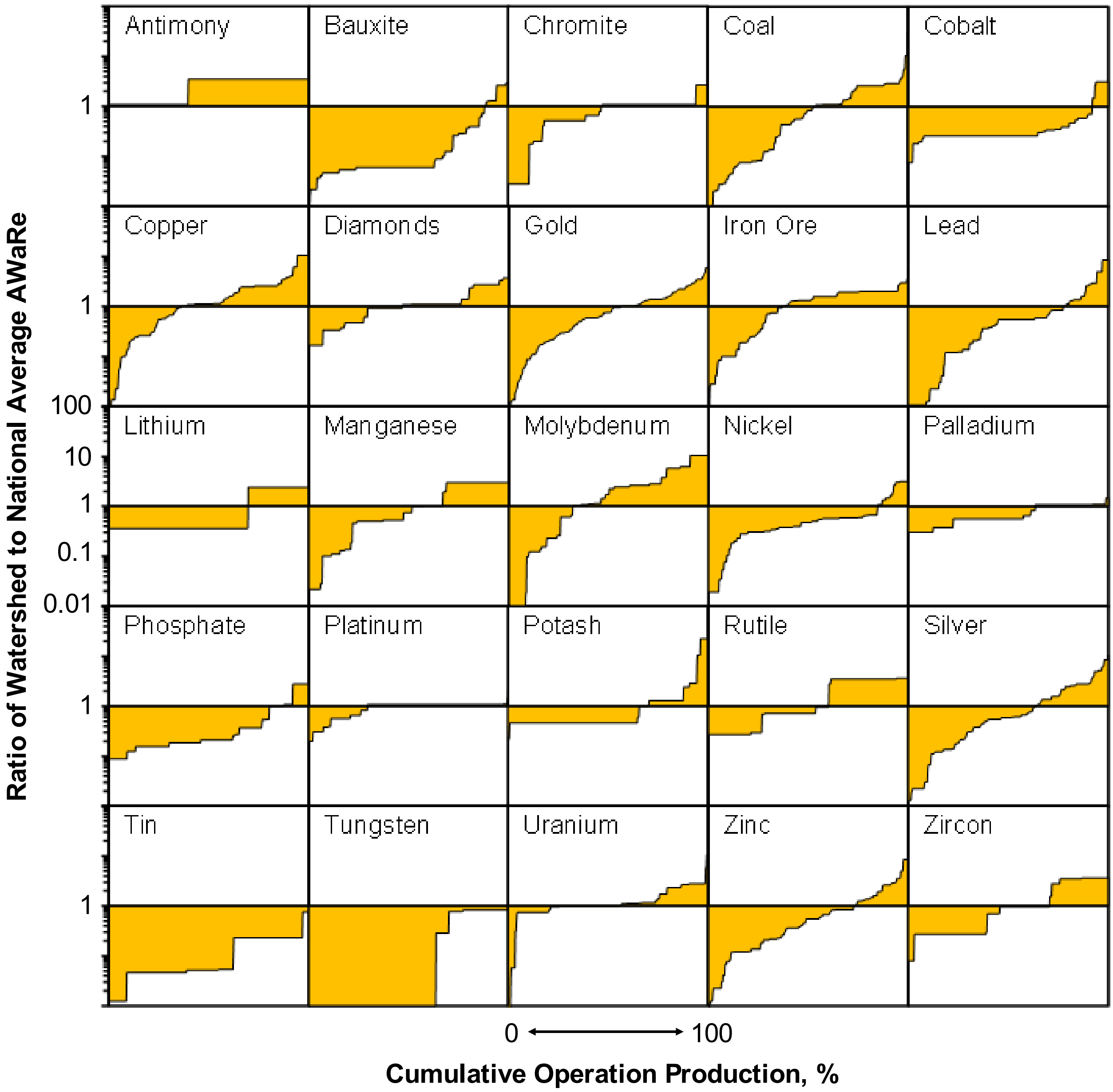

Figure A. 3: Cumulative operation production distribution for 25 mined commodities (SNL, 2017) in relation to the ratio between watershed and national average AWaRe factors for non-agricultural water use (Boulay et al., 2016, 2017; WULCA, 2017).

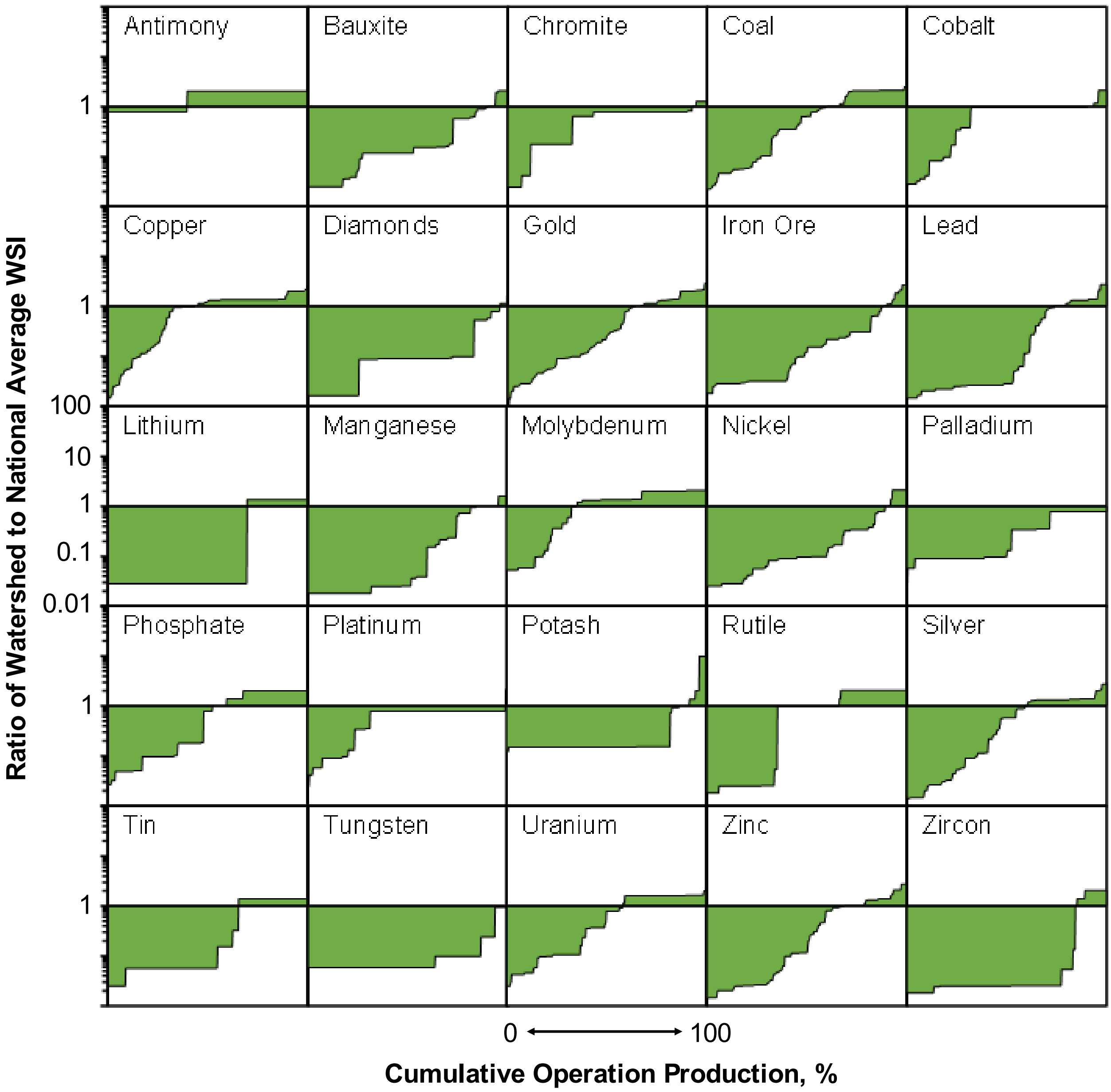

Figure A. 4: Cumulative operation production distribution for 25 mined commodities (SNL, 2017) in relation to the ratio between watershed and national average water stress index (WSI)(Pfister et al., 2009).

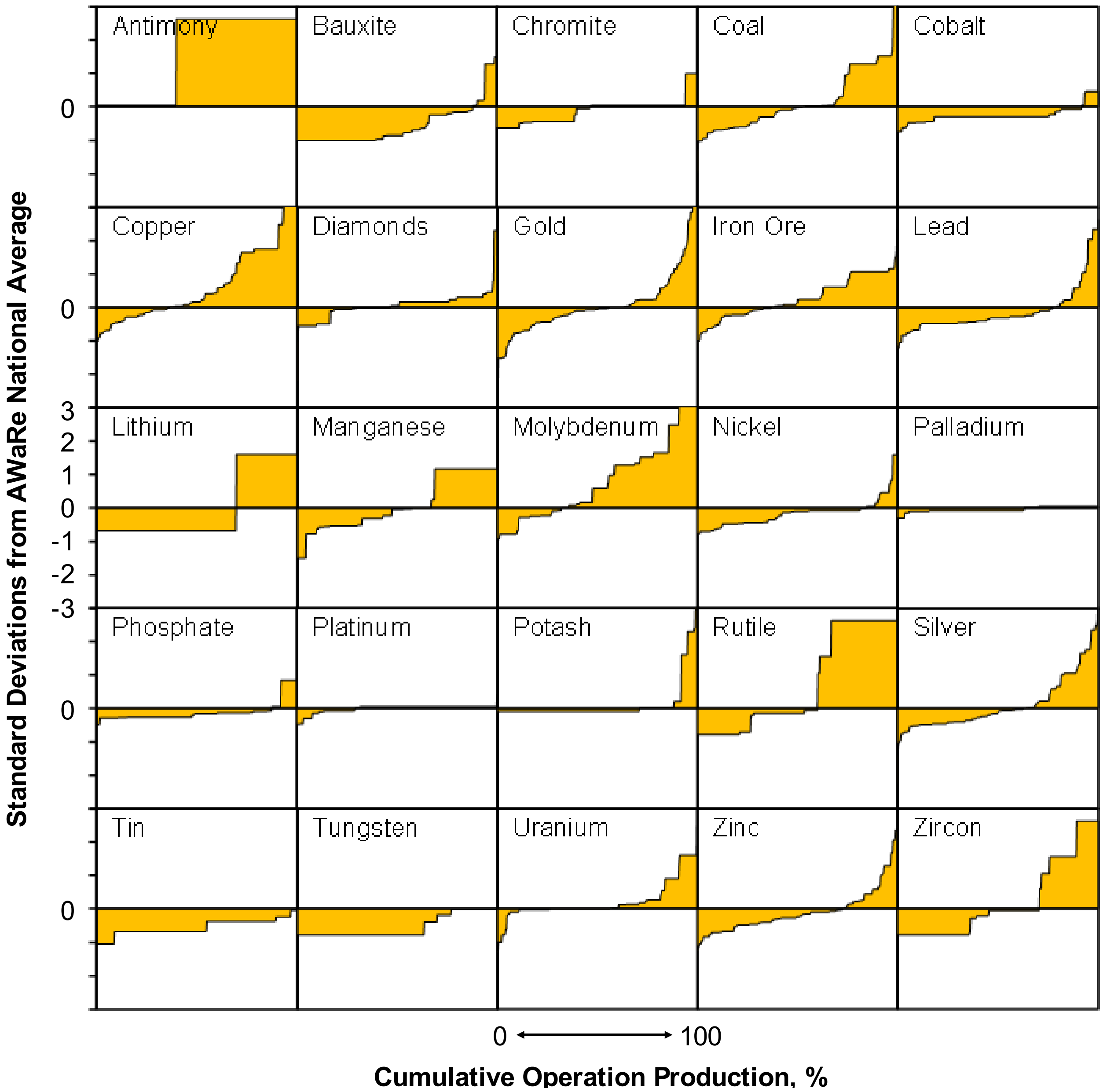

Figure A. 5: Cumulative operation production distribution for 25 mined commodities (SNL, 2017) in relation to watershed AWaRe factors for non-agricultural water use (Boulay et al., 2016, 2017; WULCA, 2017), measured relative to the national average factor and expressed in terms of the standard deviation of spatial uncertainty associated with each national average.

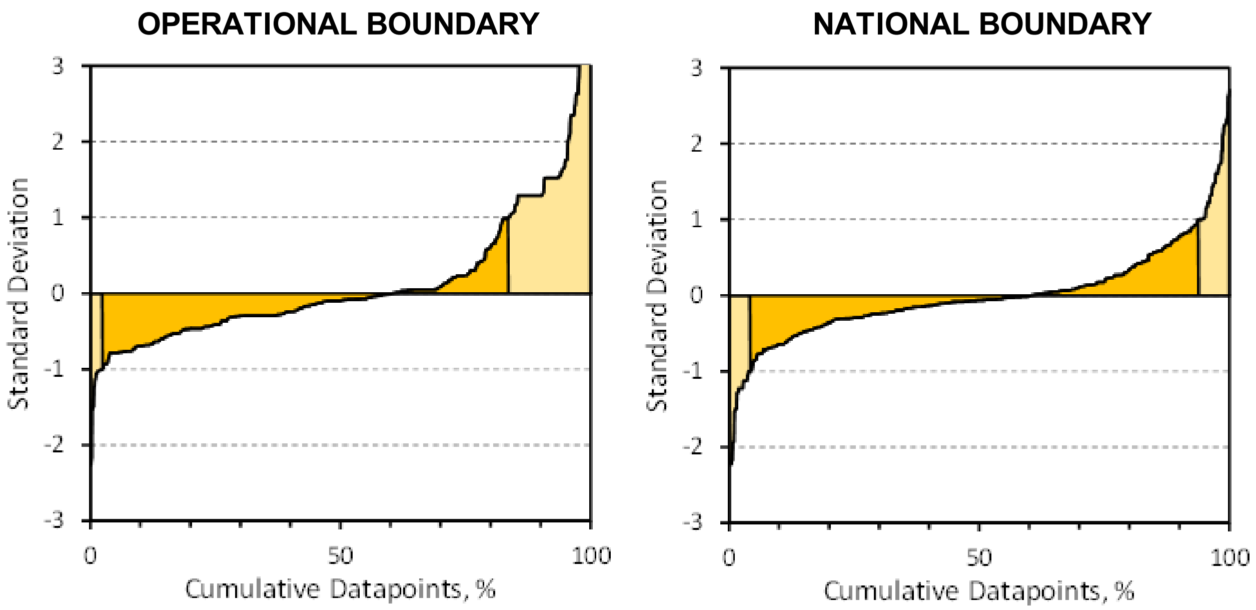

Figure A. 6: Cumulative distribution of operation (SNL, 2017) and national (British Geological Survey, 2016) datapoints falling within 1 standard deviation of the spatial uncertainty associated with national average AWaRe factors (Boulay et al., 2016, 2017; WULCA, 2017).

a

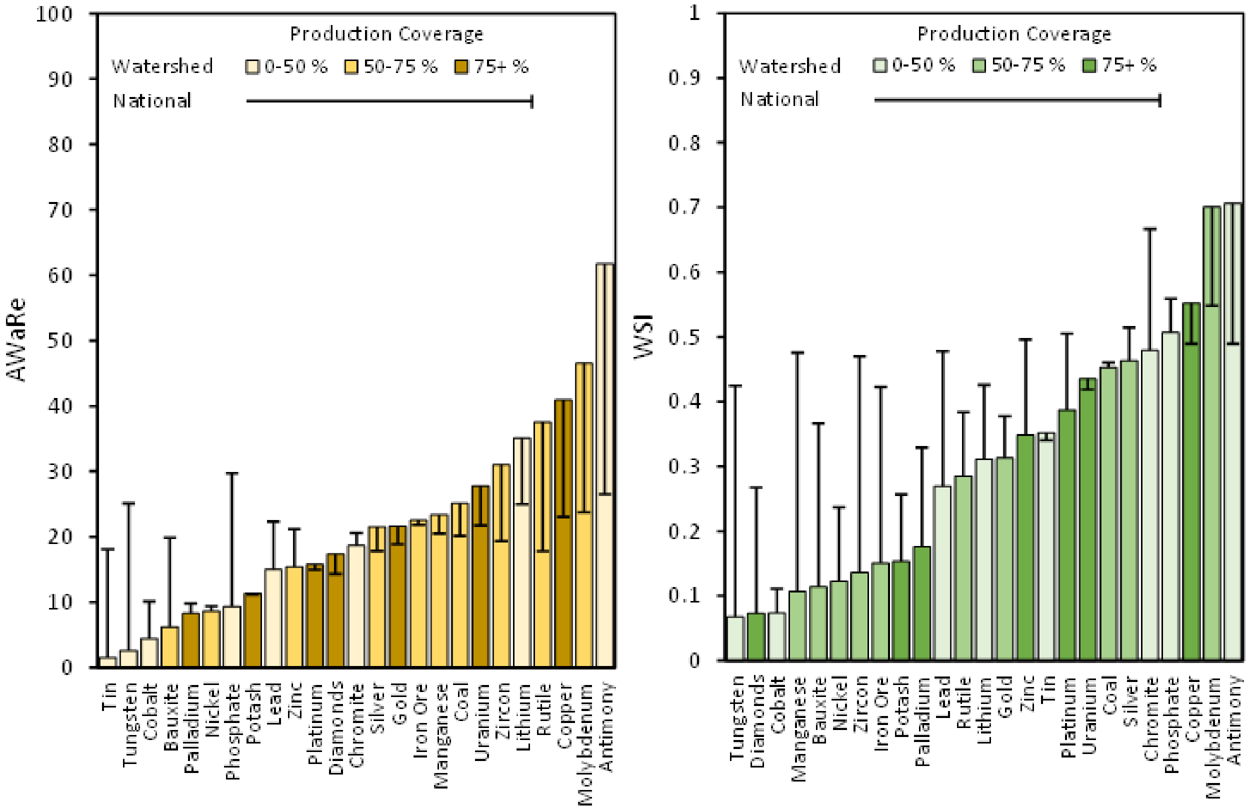

Figure A. 7: Commodity production weighted averages determined using national average and watershed WSI (Pfister et al., 2009) and AWaRe factors for non-agricultural water use (Boulay et al., 2016, 2017; WULCA, 2017). Commodities are sorted based upon watershed production averages. Shading represents the degree of production coverage of the operation data (SNL, 2017) in relation to the sum of national production data (British Geological Survey, 2016).

Table A. 1: National average WSI (Pfister et al., 2009), AWaRe factors for non-agricultural water use (Boulay et al., 2016; 2017; WULCA, 2017) and national mined commodity production data for 2014 (BGS, 2016). Part 1 – Antimony to Cobalt.

	National Average		Antimony	Bauxite	Chromite	Coal	Cobalt
Country	AWaRe	WSI	kg	kg	kg	kg	Kg
Afghanistan	32.2	0.97	-	-	4.50E+06	1.80E+07	-
Albania	5.0	0.13	-	-	6.79E+08	-	-
Algeria	36.2	0.79	-	-	-	-	-
Angola	9.8	0.02	-	-	-	-	-
Argentina	6.8	0.35	-	-	-	1.50E+08	-
Armenia	40.3	0.98	-	-	-	-	-
Australia	25.5	0.40	3.83E+06	7.86E+10	-	4.92E+11	6.25E+06
Austria	1.2	0.10	-	-	-	-	-
Azerbaijan	51.0	0.90	-	-	-	-	-
Bahrain	n.d.	n.d.	-	-	-	-	-
Bangladesh	2.9	0.50	-	-	-	9.43E+08	-
Belarus	2.9	0.08	-	-	-	-	-
Bhutan	0.8	0.02	-	-	-	1.22E+08	-
Bolivia	3.0	0.37	4.19E+06	-	-	-	-
Bosnia & Herzegovina	1.0	0.08	-	6.05E+08	-	1.17E+10	-
Botswana	32.9	0.68	-	-	-	1.71E+09	1.96E+05
Brazil	1.9	0.07	-	3.54E+10	4.56E+08	7.70E+09	3.50E+06
Bulgaria	10.8	0.39	-	-	-	3.13E+10	-
Burkina Faso	18.4	0.02	-	-	-	-	-
Burundi	66.4	0.01	-	-	-	-	-
Cameroon	4.9	0.01	-	-	-	-	-
Canada	2.6	0.10	5.00E+03	-	-	6.90E+10	6.57E+06
Chile	39.2	0.74	-	-	-	4.15E+09	-
China	27.7	0.48	1.20E+08	6.50E+10	2.20E+08	3.87E+12	8.50E+06
Colombia	0.8	0.04	-	-	-	8.86E+10	-
Cote d'Ivoire	6.8	0.01	-	-	-	-	-
Croatia	3.2	0.05	-	8.00E+06	-	-	-
Cuba	3.8	0.23	-	-	-	-	3.21E+06
Cyprus	60.9	0.88	-	-	-	-	-
Czech Republic	1.7	0.14	-	-	-	4.67E+10	-
Dem. Rep. Congo	8.6	0.01	-	-	-	1.32E+08	7.65E+07
Dominican Republic	6.2	0.11	-	1.45E+09	-	-	-
Ecuador	1.4	0.18	-	-	-	-	-
Egypt	97.8	0.98	-	-	-	3.00E+08	-
Eritrea	37.5	0.61	-	-	-	-	-
Ethiopia	30.0	0.21	-	-	-	2.00E+07	-
Fiji	1.1	0.01	-	4.89E+08	-	-	-
Finland	2.0	0.42	-	-	1.03E+09	-	2.10E+06
France	2.3	0.18	-	7.10E+07	-	-	-
French Guiana	0.7	0.01	-	-	-	-	-
Gabon	1.2	0.01	-	-	-	-	-
Georgia	22.9	0.68	-	-	-	3.74E+08	-
Germany	1.2	0.12	-	-	-	1.87E+11	-
Ghana	20.3	0.06	-	9.40E+08	-	-	-
Greece	30.7	0.71	-	1.88E+09	-	5.04E+10	-
Guatemala	1.1	0.01	-	-	-	-	-
Guinea	22.4	0.02	-	1.87E+10	-	-	-

Country	National Average AWaRe	WSI	Antimony kg	Bauxite kg	Chromite kg	Coal kg	Cobalt Kg
Guyana	0.7	0.01	-	1.56E+09	-	-	-
Honduras	1.2	0.01	-	-	-	-	-
Hungary	1.2	0.10	-	1.40E+07	-	9.55E+09	-
India	21.3	0.97	-	2.02E+10	1.68E+09	5.85E+11	-
Indonesia	10.5	0.18	-	2.56E+09	-	4.03E+11	3.29E+05
Iran	41.0	0.91	-	9.00E+08	3.50E+08	2.80E+09	-
Iraq	35.0	0.97	-	-	-	-	-
Ireland	0.7	0.02	-	-	-	-	-
Israel	52.2	1.00	-	-	-	-	-
Jamaica	5.2	0.01	-	9.68E+09	-	-	-
Japan	0.9	0.32	-	-	-	-	-
Jordan	49.1	0.97	-	-	-	-	-
Kazakhstan	26.9	0.62	8.00E+05	4.52E+09	5.41E+09	1.15E+11	-
Kenya	29.0	0.02	-	-	-	-	-
Kosovo [a]	4.6	0.10	-	-	1.27E+07	7.20E+09	-
Kyrgyzstan	64.0	1.00	2.45E+06	-	-	1.78E+09	-
Laos	3.4	0.03	6.20E+05	-	-	5.10E+08	-
Lesotho	22.1	0.99	-	-	-	-	-
Liberia	0.6	0.01	-	-	-	-	-
Macedonia	15.0	0.53	-	-	-	6.47E+09	-
Madagascar	2.6	0.03	-	-	5.37E+07	-	2.92E+06
Malawi	6.4	0.01	-	-	-	7.10E+07	-
Malaysia	0.6	0.04	-	9.63E+08	-	2.67E+09	-
Mali	28.6	0.27	-	-	-	-	-
Mauritania	56.1	0.09	-	-	-	-	-
Mexico	20.2	0.76	2.71E+05	-	-	1.61E+10	-
Mongolia	30.8	0.05	-	-	-	2.92E+10	-
Montenegro [b]	3.6	0.10	-	1.55E+08	-	1.66E+09	-
Morocco	56.7	0.84	1.00E+06	-	-	-	1.39E+06
Mozambique	7.1	0.20	-	2.91E+06	-	8.50E+09	-
Myanmar	2.6	0.02	3.30E+06	-	-	3.87E+08	-
Namibia	42.4	0.02	-	-	-	-	-
Nauru	n.d.	n.d.	-	-	-	-	-
Nepal	17.8	1.00	-	-	-	-	-
Netherlands	1.0	0.31	-	-	-	-	-
New Caledonia	5.7	0.00	-	-	-	8.15E+06	3.00E+06
New Zealand	3.2	0.02	-	-	-	3.99E+09	-
Nicaragua	2.7	0.03	-	-	-	-	-
Niger	19.1	0.17	-	-	-	2.56E+08	-
Nigeria	10.4	0.30	-	-	-	3.10E+07	-
North Korea	3.0	0.37	-	-	-	4.20E+10	-
Norway	0.6	0.08	-	-	-	1.70E+09	-
Oman	34.9	0.98	-	-	7.51E+08	-	-
Pakistan	44.7	0.97	1.27E+05	3.34E+07	3.50E+08	3.14E+09	-
Panama	1.7	0.01	-	-	-	-	-
Papua New Guinea	0.6	0.01	-	-	-	-	1.80E+06
Peru	16.1	0.72	-	-	-	2.29E+08	-
Philippines	6.1	0.40	-	-	4.71E+07	7.35E+09	4.09E+06
Poland	1.9	0.07	-	-	-	1.30E+11	-
Portugal	15.3	0.57	-	-	-	-	-
Rep. Congo	0.7	0.01	-	-	-	-	-

Country	National Average AWaRe	WSI	Antimony kg	Bauxite kg	Chromite kg	Coal kg	Cobalt Kg
Romania	2.5	0.10	-	-	-	2.48E+10	-
Russia	3.9	0.11	6.50E+06	5.59E+09	3.60E+08	3.56E+11	2.30E+06
Rwanda	81.7	0.02	-	-	-	-	-
Saudi Arabia	29.0	1.00	-	1.96E+09	-	-	-
Senegal	47.9	0.11	-	-	-	-	-
Serbia	4.6	0.10	-	-	-	2.98E+10	-
Sierra Leone	0.9	0.01	-	1.16E+09	-	-	-
Slovakia	1.3	0.09	-	-	-	2.05E+09	-
Slovenia	1.0	0.10	-	-	-	3.11E+09	-
Solomon Islands	1.1	0.01	-	-	-	-	-
South Africa	19.6	0.69	8.16E+05	-	1.40E+10	2.61E+11	1.33E+06
South Korea	2.2	0.60	-	-	-	1.75E+09	-
Spain	31.5	0.72	-	-	-	4.54E+09	-
Sri Lanka	9.7	0.61	-	-	-	-	-
Sudan	47.4	0.32	-	-	6.00E+07	-	-
Suriname	0.5	0.01	-	2.71E+09	-	-	-
Swaziland	2.8	0.02	-	-	-	1.78E+08	-
Sweden	4.0	0.04	-	-	-	-	-
Syria	48.3	1.00	-	-	-	-	-
Tajikistan	49.5	1.00	6.50E+06	-	-	8.78E+08	-
Tanzania	40.4	0.01	-	2.56E+07	-	2.46E+08	-
Thailand	5.4	0.53	7.06E+05	-	-	1.80E+10	-
Togo	14.3	0.01	-	-	-	-	-
Tunisia	41.7	0.91	-	-	-	-	-
Turkey	20.7	0.78	4.50E+06	8.00E+08	4.10E+09	8.09E+10	-
Uganda	84.0	0.02	-	-	-	-	-
Ukraine	5.6	0.30	-	-	-	4.54E+10	-
United Kingdom	3.1	0.40	-	-	-	1.16E+10	-
Uruguay	0.5	0.01	-	-	-	-	-
USA	9.5	0.50	-	1.28E+08	-	9.49E+11	-
Uzbekistan	52.2	0.99	-	-	-	4.03E+09	-
Venezuela	3.6	0.30	-	2.32E+09	-	6.88E+08	-
Vietnam	6.9	0.35	1.07E+06	1.09E+09	-	4.17E+10	-
Zambia	6.5	0.01	-	-	-	-	4.32E+06
Zimbabwe	11.1	0.19	-	-	4.08E+08	5.78E+09	3.59E+05
Global	-	-	**1.57E+08**	**2.60E+11**	**3.00E+10**	**8.09E+12**	**1.29E+08**

Notes

[a] For Kosovo assumed Serbia AWaRe and Serbia and Montenegro WSI

[b] For Montenegro assumed Serbia and Montenegro WSI

Table A. 2: National average WSI (Pfister et al., 2009), AWaRe factors for non-agricultural water use (Boulay et al., 2016; 2017; WULCA, 2017) and national mined commodity production data for 2014 (BGS, 2016). Part 2 – Copper to Lead.

Country	National Average		Copper	Diamonds	Gold	IronOre	Lead
	AWaRe	WSI	kt	kg	kg	kg	kg
Afghanistan	32.2	0.97	-	-	-	-	-
Albania	5.0	0.13	3.50E+06	-	-	-	-
Algeria	36.2	0.79	-	-	8.00E+01	9.11E+08	-
Angola	9.8	0.02	-	1.76E+03	-	-	-
Argentina	6.8	0.35	1.03E+08	-	7.18E+04	-	2.90E+07
Armenia	40.3	0.98	5.10E+07	-	2.81E+03	-	-
Australia	25.5	0.40	9.70E+08	1.86E+03	2.74E+05	7.46E+11	7.28E+08
Austria	1.2	0.10	-	-	-	2.44E+09	-
Azerbaijan	51.0	0.90	7.84E+05	-	1.88E+03	9.14E+07	-
Bahrain	n.d.	n.d.	-	-	-	-	-
Bangladesh	2.9	0.50	-	-	-	-	-
Belarus	2.9	0.08	-	-	-	-	-
Bhutan	0.8	0.02	-	-	-	1.90E+07	-
Bolivia	3.0	0.37	1.07E+07	-	3.92E+04	-	7.56E+07
Bosnia & Herzegovina	1.0	0.08	-	-	-	2.13E+09	4.00E+06
Botswana	32.9	0.68	4.77E+07	4.93E+03	9.58E+02	-	-
Brazil	1.9	0.07	2.40E+08	1.14E+01	8.00E+04	3.46E+11	1.00E+07
Bulgaria	10.8	0.39	1.08E+08	-	7.90E+03	-	1.56E+07
Burkina Faso	18.4	0.02	-	-	3.72E+04	-	4.00E+06
Burundi	66.4	0.01	-	-	1.00E+03	-	-
Cameroon	4.9	0.01	-	1.20E+00	6.00E+02	-	-
Canada	2.6	0.10	6.96E+08	2.42E+03	1.52E+05	4.42E+10	3.62E+06
Chile	39.2	0.74	5.75E+09	-	4.60E+04	1.89E+10	2.68E+06
China	27.7	0.48	1.64E+09	2.20E+02	4.52E+05	1.51E+12	2.70E+09
Colombia	0.8	0.04	3.99E+06	-	5.70E+04	6.76E+08	-
Cote d'Ivoire	6.8	0.01	-	2.15E-01	1.70E+04	-	-
Croatia	3.2	0.05	-	-	-	-	-
Cuba	3.8	0.23	-	-	-	-	-
Cyprus	60.9	0.88	3.09E+06	-	-	-	-
Czech Republic	1.7	0.14	-	-	-	-	-
Dem. Rep. Congo	8.6	0.01	1.07E+09	2.93E+03	3.60E+04	-	-
Dominican Republic	6.2	0.11	9.26E+06	-	3.51E+04	-	-
Ecuador	1.4	0.18	3.20E+06	-	7.32E+03	4.08E+05	-
Egypt	97.8	0.98	-	-	1.17E+04	1.50E+09	-
Eritrea	37.5	0.61	8.89E+07	-	8.40E+02	-	-
Ethiopia	30.0	0.21	-	-	1.03E+04	-	-
Fiji	1.1	0.01	-	-	1.20E+03	-	-
Finland	2.0	0.42	4.28E+07	-	8.09E+03	-	-
France	2.3	0.18	-	-	-	-	-
French Guiana	0.7	0.01	-	-	2.00E+03	-	-
Gabon	1.2	0.01	-	-	1.01E+03	-	-
Georgia	22.9	0.68	6.80E+06	-	2.30E+03	-	-
Germany	1.2	0.12	-	-	-	4.56E+08	-
Ghana	20.3	0.06	-	4.82E+01	9.85E+04	-	-
Greece	30.7	0.71	-	-	5.52E+02	-	1.67E+07
Guatemala	1.1	0.01	-	-	6.14E+03	2.04E+06	1.04E+07
Guinea	22.4	0.02	-	3.28E+01	1.57E+04	-	-

Country	National Average AWaRe	WSI	Copper kt	Diamonds kg	Gold kg	IronOre kg	Lead kg
Guyana	0.7	0.01	-	2.00E+01	1.21E+04	-	-
Honduras	1.2	0.01	-	-	2.76E+03	-	1.55E+07
Hungary	1.2	0.10	-	-	-	-	-
India	21.3	0.97	2.67E+07	7.22E+00	1.32E+03	1.29E+11	1.01E+08
Indonesia	10.5	0.18	3.77E+08	-	6.90E+04	3.00E+09	-
Iran	41.0	0.91	2.17E+08	-	1.76E+03	5.00E+10	3.60E+07
Iraq	35.0	0.97	-	-	-	-	-
Ireland	0.7	0.02	-	-	-	-	4.05E+07
Israel	52.2	1.00	-	-	-	-	-
Jamaica	5.2	0.01	-	-	-	-	-
Japan	0.9	0.32	-	-	7.11E+03	-	-
Jordan	49.1	0.97	-	-	-	-	-
Kazakhstan	26.9	0.62	4.72E+08	-	4.20E+04	5.15E+10	3.78E+07
Kenya	29.0	0.02	-	-	0.00E+00	-	-
Kosovo [a]	4.6	0.10	-	-	-	-	7.68E+06
Kyrgyzstan	64.0	1.00	7.00E+06	-	1.78E+04	-	-
Laos	3.4	0.03	1.60E+08	-	5.27E+03	-	1.00E+05
Lesotho	22.1	0.99	-	6.92E+01	-	-	-
Liberia	0.6	0.01	-	1.59E+01	5.35E+02	4.81E+09	-
Macedonia	15.0	0.53	8.19E+06	-	-	-	4.38E+07
Madagascar	2.6	0.03	-	-	-	-	-
Malawi	6.4	0.01	-	-	-	-	-
Malaysia	0.6	0.04	-	-	4.31E+03	9.62E+09	-
Mali	28.6	0.27	-	-	4.54E+04	1.00E+08	-
Mauritania	56.1	0.09	3.31E+07	-	9.63E+03	1.33E+10	-
Mexico	20.2	0.76	5.15E+08	-	1.18E+05	2.52E+10	2.50E+08
Mongolia	30.8	0.05	2.52E+08	-	2.00E+04	6.00E+09	-
Montenegro [b]	3.6	0.10	-	-	-	-	2.76E+06
Morocco	56.7	0.84	1.66E+07	-	3.77E+02	2.29E+07	2.75E+07
Mozambique	7.1	0.20	-	-	1.97E+02	-	-
Myanmar	2.6	0.02	2.70E+07	-	9.00E+02	-	1.80E+07
Namibia	42.4	0.02	5.25E+06	3.88E+02	2.14E+03	-	1.01E+07
Nauru	n.d.	n.d.	-	-	-	-	-
Nepal	17.8	1.00	-	-	-	-	-
Netherlands	1.0	0.31	-	-	-	-	-
New Caledonia	5.7	0.00	-	-	-	-	-
New Zealand	3.2	0.02	-	-	1.20E+04	3.25E+09	-
Nicaragua	2.7	0.03	-	-	8.65E+03	-	-
Niger	19.1	0.17	-	-	7.32E+02	-	-
Nigeria	10.4	0.30	-	-	4.00E+03	7.00E+07	1.14E+07
North Korea	3.0	0.37	4.60E+06	-	-	2.80E+09	5.30E+07
Norway	0.6	0.08	-	-	-	1.05E+10	-
Oman	34.9	0.98	1.37E+07	-	-	-	-
Pakistan	44.7	0.97	1.77E+07	-	-	1.93E+08	2.00E+05
Panama	1.7	0.01	-	-	0.00E+00	-	-
Papua New Guinea	0.6	0.01	7.59E+07	-	5.29E+04	-	-
Peru	16.1	0.72	1.38E+09	-	1.40E+05	7.19E+09	2.78E+08
Philippines	6.1	0.40	9.19E+07	-	1.84E+04	1.06E+09	-
Poland	1.9	0.07	4.22E+08	-	2.26E+02	-	8.32E+07
Portugal	15.3	0.57	7.54E+07	-	-	-	3.19E+06
Rep. Congo	0.7	0.01	-	1.06E+01	-	-	-

Country	National Average		Copper	Diamonds	Gold	IronOre	Lead
	AWaRe	WSI	kt	kg	kg	kg	kg
Romania	2.5	0.10	7.20E+06	-	5.00E+02	-	-
Russia	3.9	0.11	7.20E+08	7.66E+03	2.49E+05	1.02E+11	1.94E+08
Rwanda	81.7	0.02	-	-	-	-	-
Saudi Arabia	29.0	1.00	9.00E+06	-	4.79E+03	-	-
Senegal	47.9	0.11	-	-	6.59E+03	-	-
Serbia	4.6	0.10	3.37E+07	-	1.20E+03	-	3.70E+06
Sierra Leone	0.9	0.01	-	1.03E+02	3.30E+01	1.45E+10	-
Slovakia	1.3	0.09	4.60E+04	-	5.82E+02	-	1.62E+05
Slovenia	1.0	0.10	-	-	-	-	-
Solomon Islands	1.1	0.01	-	-	6.03E+02	-	-
South Africa	19.6	0.69	7.87E+07	1.61E+03	1.52E+05	8.08E+10	2.93E+07
South Korea	2.2	0.60	-	-	2.84E+02	6.93E+08	2.76E+06
Spain	31.5	0.72	1.04E+08	-	1.80E+03	-	1.20E+06
Sri Lanka	9.7	0.61	-	-	-	-	-
Sudan	47.4	0.32	-	-	7.00E+04	4.00E+08	-
Suriname	0.5	0.01	-	-	1.06E+04	-	-
Swaziland	2.8	0.02	-	-	-	6.03E+08	-
Sweden	4.0	0.04	7.97E+07	-	6.85E+03	3.58E+10	7.08E+07
Syria	48.3	1.00	-	-	-	-	-
Tajikistan	49.5	1.00	-	-	3.40E+03	-	2.80E+07
Tanzania	40.4	0.01	6.36E+06	5.06E+01	4.06E+04	-	-
Thailand	5.4	0.53	-	-	4.58E+03	3.48E+08	-
Togo	14.3	0.01	-	4.60E-03	2.06E+04	-	-
Tunisia	41.7	0.91	-	-	-	3.08E+08	-
Turkey	20.7	0.78	1.22E+08	-	3.10E+04	7.70E+09	6.22E+07
Uganda	84.0	0.02	-	-	-	4.20E+07	-
Ukraine	5.6	0.30	-	-	-	6.83E+10	-
United Kingdom	3.1	0.40	-	-	0.00E+00	-	1.00E+05
Uruguay	0.5	0.01	-	-	1.72E+03	-	-
USA	9.5	0.50	1.37E+09	-	2.10E+05	5.75E+10	3.55E+08
Uzbekistan	52.2	0.99	8.00E+07	-	1.02E+05	-	-
Venezuela	3.6	0.30	-	-	1.10E+03	1.14E+10	-
Vietnam	6.9	0.35	1.65E+07	-	1.64E+02	2.31E+09	1.40E+06
Zambia	6.5	0.01	7.11E+08	-	4.80E+03	-	-
Zimbabwe	11.1	0.19	8.26E+06	9.54E+02	1.54E+04	-	-
Global	-	-	**1.84E+10**	**2.51E+04**	**3.02E+06**	**3.38E+12**	**5.37E+09**

Notes

[a] For Kosovo assumed Serbia AWaRe and Serbia and Montenegro WSI

[b] For Montenegro assumed Serbia and Montenegro WSI

Table A. 3: National average WSI (Pfister et al., 2009), AWaRe factors for non-agricultural water use (Boulay et al., 2016; 2017; WULCA, 2017) and national mined commodity production data for 2014 (BGS, 2016). Part 3 – Lithium to Palladium.

Country	National Average		Lithium	Manganese	Molybdenum	Nickel	Palladium
	AWaRe	WSI	kg	kg	kg	kg	kg
Afghanistan	32.2	0.97	-	-	-	-	-
Albania	5.0	0.13	-	-	-	4.79E+06	-
Algeria	36.2	0.79	-	-	-	-	-
Angola	9.8	0.02	-	-	-	-	-
Argentina	6.8	0.35	1.60E+07	-	1.91E+06	-	-
Armenia	40.3	0.98	-	-	6.02E+06	-	-
Australia	25.5	0.40	4.44E+08	7.59E+09	-	2.45E+08	7.66E+02
Austria	1.2	0.10	-	-	-	-	-
Azerbaijan	51.0	0.90	-	-	-	-	-
Bahrain	n.d.	n.d.	-	-	-	-	-
Bangladesh	2.9	0.50	-	-	-	-	-
Belarus	2.9	0.08	-	-	-	-	-
Bhutan	0.8	0.02	-	-	-	-	-
Bolivia	3.0	0.37	-	-	-	-	-
Bosnia & Herzegovina	1.0	0.08	-	-	-	-	-
Botswana	32.9	0.68	-	-	-	1.50E+07	5.60E+02
Brazil	1.9	0.07	8.00E+06	2.50E+09	2.00E+05	8.56E+07	-
Bulgaria	10.8	0.39	-	7.50E+07	-	-	-
Burkina Faso	18.4	0.02	-	-	-	-	-
Burundi	66.4	0.01	-	-	-	-	-
Cameroon	4.9	0.01	-	-	-	-	-
Canada	2.6	0.10	1.08E+05	-	9.05E+06	2.35E+08	1.87E+04
Chile	39.2	0.74	6.23E+07	-	4.88E+07	-	-
China	27.7	0.48	6.10E+07	1.60E+10	1.23E+08	9.00E+07	7.00E+02
Colombia	0.8	0.04	-	-	-	4.12E+07	-
Cote d'Ivoire	6.8	0.01	-	1.00E+08	-	-	-
Croatia	3.2	0.05	-	-	-	-	-
Cuba	3.8	0.23	-	-	-	5.00E+07	-
Cyprus	60.9	0.88	-	-	-	-	-
Czech Republic	1.7	0.14	-	-	-	-	-
Dem. Rep. Congo	8.6	0.01	-	-	-	-	-
Dominican Republic	6.2	0.11	-	-	-	0.00E+00	-
Ecuador	1.4	0.18	-	-	-	-	-
Egypt	97.8	0.98	-	3.00E+07	-	-	-
Eritrea	37.5	0.61	-	-	-	-	-
Ethiopia	30.0	0.21	-	-	-	-	-
Fiji	1.1	0.01	-	-	-	-	-
Finland	2.0	0.42	-	-	-	1.97E+07	8.08E+02
France	2.3	0.18	-	-	-	-	-
French Guiana	0.7	0.01	-	-	-	-	-
Gabon	1.2	0.01	-	4.00E+09	-	-	-
Georgia	22.9	0.68	-	3.70E+08	-	-	-
Germany	1.2	0.12	-	-	-	-	-
Ghana	20.3	0.06	-	1.35E+09	-	-	-
Greece	30.7	0.71	-	-	-	2.14E+07	-
Guatemala	1.1	0.01	-	-	-	4.84E+07	-
Guinea	22.4	0.02	-	-	-	-	-

Country	National Average AWaRe	National Average WSI	Lithium kg	Manganese kg	Molybdenum kg	Nickel kg	Palladium kg
Guyana	0.7	0.01	-	-	-	-	-
Honduras	1.2	0.01	-	-	-	-	-
Hungary	1.2	0.10	-	5.10E+07	-	-	-
India	21.3	0.97	-	2.17E+09	-	-	-
Indonesia	10.5	0.18	-	-	-	2.16E+08	-
Iran	41.0	0.91	-	1.65E+08	3.50E+06	-	-
Iraq	35.0	0.97	-	-	-	-	-
Ireland	0.7	0.02	-	-	-	-	-
Israel	52.2	1.00	-	-	-	-	-
Jamaica	5.2	0.01	-	-	-	-	-
Japan	0.9	0.32	-	-	-	-	-
Jordan	49.1	0.97	-	-	-	-	-
Kazakhstan	26.9	0.62	-	2.61E+09	-	-	-
Kenya	29.0	0.02	-	-	-	-	-
Kosovo [a]	4.6	0.10	-	-	-	6.72E+06	-
Kyrgyzstan	64.0	1.00	-	-	-	-	-
Laos	3.4	0.03	-	-	-	-	-
Lesotho	22.1	0.99	-	-	-	-	-
Liberia	0.6	0.01	-	-	-	-	-
Macedonia	15.0	0.53	-	-	-	0.00E+00	-
Madagascar	2.6	0.03	-	-	-	3.71E+07	-
Malawi	6.4	0.01	-	-	-	-	-
Malaysia	0.6	0.04	-	8.35E+08	-	-	-
Mali	28.6	0.27	-	-	-	-	-
Mauritania	56.1	0.09	-	-	-	-	-
Mexico	20.2	0.76	-	6.52E+08	1.44E+07	-	-
Mongolia	30.8	0.05	-	-	1.80E+06	-	-
Montenegro [b]	3.6	0.10	-	-	-	-	-
Morocco	56.7	0.84	-	9.13E+07	-	2.00E+05	-
Mozambique	7.1	0.20	-	-	-	-	-
Myanmar	2.6	0.02	-	2.42E+08	-	2.10E+07	-
Namibia	42.4	0.02	-	1.01E+08	-	-	-
Nauru	n.d.	n.d.	-	-	-	-	-
Nepal	17.8	1.00	-	-	-	-	-
Netherlands	1.0	0.31	-	-	-	-	-
New Caledonia	5.7	0.00	-	-	-	1.78E+08	-
New Zealand	3.2	0.02	-	-	-	-	-
Nicaragua	2.7	0.03	-	-	-	-	-
Niger	19.1	0.17	-	-	-	-	-
Nigeria	10.4	0.30	-	-	-	-	-
North Korea	3.0	0.37	-	-	-	-	-
Norway	0.6	0.08	-	-	2.00E+03	2.90E+05	-
Oman	34.9	0.98	-	-	-	-	-
Pakistan	44.7	0.97	-	-	-	-	-
Panama	1.7	0.01	-	-	-	-	-
Papua New Guinea	0.6	0.01	-	-	-	1.77E+07	-
Peru	16.1	0.72	-	-	1.70E+07	-	-
Philippines	6.1	0.40	-	-	-	3.63E+08	-
Poland	1.9	0.07	-	-	-	7.80E+05	3.00E+01
Portugal	15.3	0.57	1.75E+07	-	-	-	-
Rep. Congo	0.7	0.01	-	-	-	-	-

Country	National Average AWaRe	WSI	Lithium kg	Manganese kg	Molybdenum kg	Nickel kg	Palladium kg
Romania	2.5	0.10	-	1.27E+07	-	-	-
Russia	3.9	0.11	-	-	3.60E+06	2.64E+08	8.13E+04
Rwanda	81.7	0.02	-	-	-	-	-
Saudi Arabia	29.0	1.00	-	-	-	-	-
Senegal	47.9	0.11	-	-	-	-	-
Serbia	4.6	0.10	-	-	-	-	2.00E+01
Sierra Leone	0.9	0.01	-	-	-	-	-
Slovakia	1.3	0.09	-	-	-	-	-
Slovenia	1.0	0.10	-	-	-	-	-
Solomon Islands	1.1	0.01	-	-	-	-	-
South Africa	19.6	0.69	-	1.39E+10	-	5.50E+07	5.84E+04
South Korea	2.2	0.60	-	-	-	-	-
Spain	31.5	0.72	-	-	-	8.63E+06	-
Sri Lanka	9.7	0.61	-	-	-	-	-
Sudan	47.4	0.32	-	3.50E+06	-	-	-
Suriname	0.5	0.01	-	-	-	-	-
Swaziland	2.8	0.02	-	-	-	-	-
Sweden	4.0	0.04	-	-	-	-	-
Syria	48.3	1.00	-	-	-	-	-
Tajikistan	49.5	1.00	-	-	-	-	-
Tanzania	40.4	0.01	-	-	-	-	-
Thailand	5.4	0.53	-	1.43E+07	-	-	-
Togo	14.3	0.01	-	-	-	-	-
Tunisia	41.7	0.91	-	-	-	-	-
Turkey	20.7	0.78	-	3.20E+08	1.00E+03	2.40E+06	-
Uganda	84.0	0.02	-	-	-	-	-
Ukraine	5.6	0.30	-	1.53E+09	-	-	-
United Kingdom	3.1	0.40	-	-	-	-	-
Uruguay	0.5	0.01	-	-	-	-	-
USA	9.5	0.50	9.00E+05	-	6.55E+07	4.09E+06	1.22E+04
Uzbekistan	52.2	0.99	-	-	-	-	-
Venezuela	3.6	0.30	-	-	-	2.50E+06	-
Vietnam	6.9	0.35	-	8.00E+05	-	6.85E+06	-
Zambia	6.5	0.01	-	-	-	-	-
Zimbabwe	11.1	0.19	4.40E+07	-	-	1.66E+07	1.01E+04
Global	-	-	**6.54E+08**	**5.47E+10**	**2.95E+08**	**2.06E+09**	**1.84E+05**

Notes

[a] For Kosovo assumed Serbia AWaRe and Serbia and Montenegro WSI

[b] For Montenegro assumed Serbia and Montenegro WSI

Table A. 4: National average WSI (Pfister et al., 2009), AWaRe factors for non-agricultural water use (Boulay et al., 2016; 2017; WULCA, 2017) and national mined commodity production data for 2014 (BGS, 2016). Part 4 – Phosphate to Silver.

Country	National Average		Phosphate	Platinum	Potash	Rutile	Silver
	AWaRe	WSI	kg	kg	kg	kg	kg
Afghanistan	32.2	0.97	-	-	-	-	-
Albania	5.0	0.13	-	-	-	-	-
Algeria	36.2	0.79	1.13E+09	-	-	-	1.60E+01
Angola	9.8	0.02	-	-	-	-	-
Argentina	6.8	0.35	-	-	-	-	9.05E+05
Armenia	40.3	0.98	-	-	-	-	1.33E+04
Australia	25.5	0.40	2.53E+09	-	-	3.00E+08	1.85E+06
Austria	1.2	0.10	-	-	-	-	-
Azerbaijan	51.0	0.90	-	-	-	-	9.70E+02
Bahrain	n.d.	n.d.	-	-	-	-	-
Bangladesh	2.9	0.50	-	-	-	-	-
Belarus	2.9	0.08	-	-	6.34E+09	-	-
Bhutan	0.8	0.02	-	-	-	-	-
Bolivia	3.0	0.37	-	-	-	-	1.35E+06
Bosnia & Herzegovina	1.0	0.08	-	-	-	-	-
Botswana	32.9	0.68	-	9.30E+01	-	-	2.23E+04
Brazil	1.9	0.07	6.00E+09	-	4.42E+08	2.00E+06	2.20E+04
Bulgaria	10.8	0.39	-	-	-	-	5.50E+04
Burkina Faso	18.4	0.02	1.83E+06	-	-	-	6.00E+04
Burundi	66.4	0.01	-	-	-	-	-
Cameroon	4.9	0.01	-	-	-	-	-
Canada	2.6	0.10	-	1.07E+04	1.13E+10	-	4.93E+05
Chile	39.2	0.74	3.43E+07	-	1.11E+09	-	1.57E+06
China	27.7	0.48	1.20E+11	1.40E+03	3.60E+09	-	3.57E+06
Colombia	0.8	0.04	3.04E+07	-	-	-	1.15E+04
Cote d'Ivoire	6.8	0.01	-	-	-	-	5.85E+02
Croatia	3.2	0.05	-	-	-	-	-
Cuba	3.8	0.23	1.30E+06	-	-	-	-
Cyprus	60.9	0.88	-	-	-	-	-
Czech Republic	1.7	0.14	-	-	-	-	-
Dem. Rep. Congo	8.6	0.01	-	-	-	-	6.49E+03
Dominican Republic	6.2	0.11	-	-	-	-	1.28E+05
Ecuador	1.4	0.18	-	-	-	-	5.77E+02
Egypt	97.8	0.98	6.00E+09	-	-	-	-
Eritrea	37.5	0.61	-	-	-	-	4.74E+04
Ethiopia	30.0	0.21	-	-	-	-	1.00E+03
Fiji	1.1	0.01	-	-	-	-	3.61E+02
Finland	2.0	0.42	9.46E+08	1.06E+03	-	-	1.28E+04
France	2.3	0.18	-	-	-	-	-
French Guiana	0.7	0.01	-	-	-	-	-
Gabon	1.2	0.01	-	-	-	-	-
Georgia	22.9	0.68	-	-	-	-	-
Germany	1.2	0.12	-	-	3.13E+09	-	-
Ghana	20.3	0.06	-	-	-	-	3.90E+03
Greece	30.7	0.71	-	-	-	-	3.58E+04
Guatemala	1.1	0.01	-	-	-	-	8.58E+05
Guinea	22.4	0.02	-	-	-	-	-

Country	National Average AWaRe	WSI	Phosphate kg	Platinum kg	Potash kg	Rutile kg	Silver kg
Guyana	0.7	0.01	-	-	-	-	-
Honduras	1.2	0.01	-	-	-	-	5.88E+04
Hungary	1.2	0.10	-	-	-	-	-
India	21.3	0.97	1.01E+09	-	-	1.60E+07	3.40E+05
Indonesia	10.5	0.18	-	-	-	-	1.19E+05
Iran	41.0	0.91	4.00E+07	-	-	-	3.49E+04
Iraq	35.0	0.97	1.40E+08	-	-	-	-
Ireland	0.7	0.02	-	-	-	-	6.44E+03
Israel	52.2	1.00	2.78E+09	-	2.21E+09	-	-
Jamaica	5.2	0.01	-	-	-	-	-
Japan	0.9	0.32	-	-	-	-	3.54E+03
Jordan	49.1	0.97	6.00E+09	-	1.10E+09	-	-
Kazakhstan	26.9	0.62	1.83E+09	-	-	-	9.89E+05
Kenya	29.0	0.02	-	-	-	5.25E+07	-
Kosovo [a]	4.6	0.10	-	-	-	-	-
Kyrgyzstan	64.0	1.00	-	-	-	-	-
Laos	3.4	0.03	-	-	-	-	3.98E+04
Lesotho	22.1	0.99	-	-	-	-	-
Liberia	0.6	0.01	-	-	-	-	-
Macedonia	15.0	0.53	-	-	-	-	3.30E+04
Madagascar	2.6	0.03	-	-	-	7.00E+06	-
Malawi	6.4	0.01	1.20E+07	-	-	-	-
Malaysia	0.6	0.04	-	-	-	3.07E+06	5.33E+02
Mali	28.6	0.27	2.00E+07	-	-	-	-
Mauritania	56.1	0.09	-	-	-	-	-
Mexico	20.2	0.76	1.66E+09	-	-	-	5.77E+06
Mongolia	30.8	0.05	-	-	-	-	3.92E+04
Montenegro [b]	3.6	0.10	-	-	-	-	-
Morocco	56.7	0.84	3.22E+10	-	-	-	2.77E+05
Mozambique	7.1	0.20	-	-	-	6.10E+06	-
Myanmar	2.6	0.02	-	-	-	-	-
Namibia	42.4	0.02	-	-	-	-	2.17E+03
Nauru	n.d.	n.d.	1.53E+08	-	-	-	-
Nepal	17.8	1.00	-	-	-	-	-
Netherlands	1.0	0.31	-	-	-	-	-
New Caledonia	5.7	0.00	-	-	-	-	-
New Zealand	3.2	0.02	-	-	-	-	1.58E+04
Nicaragua	2.7	0.03	-	-	-	-	1.36E+04
Niger	19.1	0.17	-	-	-	-	6.70E+01
Nigeria	10.4	0.30	-	-	-	-	-
North Korea	3.0	0.37	1.00E+08	-	-	-	5.00E+04
Norway	0.6	0.08	-	-	-	-	-
Oman	34.9	0.98	-	-	-	-	-
Pakistan	44.7	0.97	8.78E+07	-	-	-	-
Panama	1.7	0.01	-	-	-	-	-
Papua New Guinea	0.6	0.01	-	-	-	-	8.43E+04
Peru	16.1	0.72	1.09E+10	-	-	-	3.78E+06
Philippines	6.1	0.40	4.31E+06	-	-	-	2.30E+04
Poland	1.9	0.07	-	6.00E+01	-	-	1.38E+06
Portugal	15.3	0.57	-	-	-	-	4.32E+04
Rep. Congo	0.7	0.01	-	-	-	-	-

Country	National Average AWaRe	WSI	Phosphate kg	Platinum kg	Potash kg	Rutile kg	Silver kg
Romania	2.5	0.10	-	-	-	-	1.80E+04
Russia	3.9	0.11	1.05E+10	2.20E+04	7.40E+09	-	1.33E+06
Rwanda	81.7	0.02	-	-	-	-	-
Saudi Arabia	29.0	1.00	1.91E+09	-	-	-	4.89E+03
Senegal	47.9	0.11	8.06E+08	-	-	2.62E+05	-
Serbia	4.6	0.10	-	3.00E+00	-	-	8.40E+03
Sierra Leone	0.9	0.01	-	-	-	8.31E+07	-
Slovakia	1.3	0.09	-	-	-	-	4.37E+02
Slovenia	1.0	0.10	-	-	-	-	-
Solomon Islands	1.1	0.01	-	-	-	-	2.80E+02
South Africa	19.6	0.69	2.01E+09	9.40E+04	-	1.33E+08	3.73E+04
South Korea	2.2	0.60	-	-	-	-	3.29E+03
Spain	31.5	0.72	-	-	9.72E+08	-	3.10E+04
Sri Lanka	9.7	0.61	5.10E+07	-	-	3.71E+07	-
Sudan	47.4	0.32	-	-	-	-	1.00E+03
Suriname	0.5	0.01	-	-	-	-	-
Swaziland	2.8	0.02	-	-	-	-	-
Sweden	4.0	0.04	-	-	-	-	3.83E+05
Syria	48.3	1.00	4.00E+08	-	-	-	-
Tajikistan	49.5	1.00	-	-	-	-	1.70E+03
Tanzania	40.4	0.01	2.30E+07	-	-	-	1.45E+04
Thailand	5.4	0.53	5.00E+05	-	-	-	2.92E+04
Togo	14.3	0.01	1.09E+09	-	-	-	-
Tunisia	41.7	0.91	3.78E+09	-	-	-	-
Turkey	20.7	0.78	-	-	-	-	2.05E+05
Uganda	84.0	0.02	-	-	-	-	-
Ukraine	5.6	0.30	-	-	-	1.00E+08	-
United Kingdom	3.1	0.40	-	-	6.00E+08	-	-
Uruguay	0.5	0.01	-	-	-	-	-
USA	9.5	0.50	2.71E+10	3.65E+03	8.50E+08	-	1.18E+06
Uzbekistan	52.2	0.99	7.00E+08	-	-	-	6.00E+04
Venezuela	3.6	0.30	3.58E+07	-	-	-	-
Vietnam	6.9	0.35	2.47E+09	-	-	-	-
Zambia	6.5	0.01	-	-	-	-	-
Zimbabwe	11.1	0.19	1.00E+07	1.25E+04	-	-	-
Global	-	-	**2.45E+11**	**1.45E+05**	**3.91E+10**	**7.40E+08**	**2.74E+07**

Notes

[a] For Kosovo assumed Serbia AWaRe and Serbia and Montenegro WSI

[b] For Montenegro assumed Serbia and Montenegro WSI

Table A. 5: National average WSI (Pfister et al., 2009), AWaRe factors for non-agricultural water use (Boulay et al., 2016; 2017; WULCA, 2017) and national mined commodity production data for 2014 (BGS, 2016). Part 5 – Tin to Zircon.

Country	National Average AWaRe	WSI	Tin kg	Tungsten kg	Uranium kg	Zinc kg	Zircon kg
Afghanistan	32.2	0.97	-	-	-	-	-

Country	National Average AWaRe	WSI	Tin kg	Tungsten kg	Uranium kg	Zinc kg	Zircon kg
Albania	5.0	0.13	-	-	-	-	-
Algeria	36.2	0.79	-	-	-	-	-
Angola	9.8	0.02	-	-	-	-	-
Argentina	6.8	0.35	-	-	-	5.76E+07	-
Armenia	40.3	0.98	-	-	-	7.65E+06	-
Australia	25.5	0.40	7.21E+06	1.20E+04	5.90E+06	1.56E+09	4.00E+08
Austria	1.2	0.10	-	8.62E+05	-	-	-
Azerbaijan	51.0	0.90	-	-	-	-	-
Bahrain	n.d.	n.d.	-	-	-	-	-
Bangladesh	2.9	0.50	-	-	-	-	-
Belarus	2.9	0.08	-	-	-	-	-
Bhutan	0.8	0.02	-	-	-	-	-
Bolivia	3.0	0.37	1.98E+07	1.58E+06	-	4.49E+08	-
Bosnia & Herzegovina	1.0	0.08	-	-	-	8.10E+06	-
Botswana	32.9	0.68	-	-	-	-	-
Brazil	1.9	0.07	1.70E+07	5.00E+05	2.72E+05	1.59E+08	2.00E+07
Bulgaria	10.8	0.39	-	-	-	1.19E+07	-
Burkina Faso	18.4	0.02	-	-	-	6.78E+07	-
Burundi	66.4	0.01	1.06E+05	5.00E+04	-	-	-
Cameroon	4.9	0.01	-	-	-	-	-
Canada	2.6	0.10	-	2.69E+06	1.03E+07	3.53E+08	-
Chile	39.2	0.74	-	-	-	4.51E+07	-
China	27.7	0.48	1.60E+08	6.80E+07	1.77E+06	5.20E+09	3.35E+07
Colombia	0.8	0.04	-	-	-	-	-
Cote d'Ivoire	6.8	0.01	-	-	-	-	-
Croatia	3.2	0.05	-	-	-	-	-
Cuba	3.8	0.23	-	-	-	-	-
Cyprus	60.9	0.88	-	-	-	-	-
Czech Republic	1.7	0.14	-	-	1.95E+05	-	-
Dem. Rep. Congo	8.6	0.01	4.61E+06	8.00E+03	-	6.37E+06	-
Dominican Republic	6.2	0.11	-	-	-	-	-
Ecuador	1.4	0.18	-	-	-	-	-
Egypt	97.8	0.98	8.50E+04	-	-	-	-
Eritrea	37.5	0.61	-	-	-	-	-
Ethiopia	30.0	0.21	-	-	-	-	-
Fiji	1.1	0.01	-	-	-	-	-
Finland	2.0	0.42	-	-	-	4.61E+07	-
France	2.3	0.18	-	-	-	-	-
French Guiana	0.7	0.01	-	-	-	-	-
Gabon	1.2	0.01	-	-	-	-	-
Georgia	22.9	0.68	-	-	-	-	-
Germany	1.2	0.12	-	-	3.89E+04	-	-
Ghana	20.3	0.06	-	-	-	-	-
Greece	30.7	0.71	-	-	-	2.31E+07	-
Guatemala	1.1	0.01	-	-	-	1.34E+07	-
Guinea	22.4	0.02	-	-	-	-	-
Guyana	0.7	0.01	-	-	-	-	-
Honduras	1.2	0.01	-	-	-	3.00E+07	-
Hungary	1.2	0.10	-	-	-	-	-
India	21.3	0.97	2.50E+04	-	4.54E+05	6.71E+08	2.00E+07

Country	National Average AWaRe	WSI	Tin kg	Tungsten kg	Uranium kg	Zinc kg	Zircon kg
Indonesia	10.5	0.18	7.02E+07	-	-	-	2.10E+07
Iran	41.0	0.91	-	-	-	1.50E+08	-
Iraq	35.0	0.97	-	-	-	-	-
Ireland	0.7	0.02	-	-	-	2.83E+08	-
Israel	52.2	1.00	-	-	-	-	-
Jamaica	5.2	0.01	-	-	-	-	-
Japan	0.9	0.32	-	-	-	-	-
Jordan	49.1	0.97	-	-	-	-	-
Kazakhstan	26.9	0.62	-	-	2.73E+07	3.86E+08	-
Kenya	29.0	0.02	-	-	-	-	4.01E+07
Kosovo [a]	4.6	0.10	-	-	-	5.51E+06	-
Kyrgyzstan	64.0	1.00	-	1.00E+05	-	-	-
Laos	3.4	0.03	8.68E+05	-	-	2.00E+06	-
Lesotho	22.1	0.99	-	-	-	-	-
Liberia	0.6	0.01	-	-	-	-	-
Macedonia	15.0	0.53	-	-	-	3.16E+07	-
Madagascar	2.6	0.03	-	-	-	-	2.70E+07
Malawi	6.4	0.01	-	-	4.35E+05	-	-
Malaysia	0.6	0.04	3.78E+06	-	-	-	6.77E+05
Mali	28.6	0.27	-	-	-	-	-
Mauritania	56.1	0.09	-	-	-	-	-
Mexico	20.2	0.76	-	-	-	6.60E+08	-
Mongolia	30.8	0.05	6.00E+04	-	-	4.66E+07	-
Montenegro [b]	3.6	0.10	-	-	-	5.51E+06	-
Morocco	56.7	0.84	-	-	-	4.54E+07	-
Mozambique	7.1	0.20	-	-	-	-	5.08E+07
Myanmar	2.6	0.02	3.50E+07	1.40E+05	-	6.10E+06	-
Namibia	42.4	0.02	-	-	3.84E+06	1.78E+08	-
Nauru	n.d.	n.d.	-	-	-	-	-
Nepal	17.8	1.00	-	-	-	-	-
Netherlands	1.0	0.31	-	-	-	-	-
New Caledonia	5.7	0.00	-	-	-	-	-
New Zealand	3.2	0.02	-	-	-	-	-
Nicaragua	2.7	0.03	-	-	-	-	-
Niger	19.1	0.17	5.00E+03	-	4.90E+06	-	-
Nigeria	10.4	0.30	2.49E+06	-	-	7.00E+06	-
North Korea	3.0	0.37	-	7.00E+04	-	3.20E+07	-
Norway	0.6	0.08	-	-	-	-	-
Oman	34.9	0.98	-	-	-	-	-
Pakistan	44.7	0.97	-	-	5.31E+04	-	-
Panama	1.7	0.01	-	-	-	-	-
Papua New Guinea	0.6	0.01	-	-	-	-	-
Peru	16.1	0.72	2.31E+07	7.70E+04	-	1.32E+09	-
Philippines	6.1	0.40	-	-	-	-	-
Poland	1.9	0.07	-	-	-	5.60E+07	-
Portugal	15.3	0.57	7.50E+04	6.71E+05	-	6.74E+07	-
Rep. Congo	0.7	0.01	-	-	-	-	-
Romania	2.5	0.10	-	-	9.08E+04	3.30E+06	-
Russia	3.9	0.11	1.50E+05	6.30E+06	3.53E+06	2.17E+08	8.50E+06
Rwanda	81.7	0.02	4.45E+06	1.70E+06	-	-	-
Saudi Arabia	29.0	1.00	-	-	-	4.18E+07	-

Country	National Average AWaRe	WSI	Tin kg	Tungsten kg	Uranium kg	Zinc kg	Zircon kg
Senegal	47.9	0.11	-	-	-	-	8.28E+06
Serbia	4.6	0.10	-	-	-	7.10E+06	-
Sierra Leone	0.9	0.01	-	-	-	-	1.87E+06
Slovakia	1.3	0.09	-	-	-	1.76E+05	-
Slovenia	1.0	0.10	-	-	-	-	-
Solomon Islands	1.1	0.01	-	-	-	-	-
South Africa	19.6	0.69	-	-	6.67E+05	2.61E+07	3.93E+08
South Korea	2.2	0.60	-	-	-	1.92E+06	-
Spain	31.5	0.72	-	1.04E+06	-	2.68E+07	-
Sri Lanka	9.7	0.61	-	-	-	-	3.25E+05
Sudan	47.4	0.32	-	-	-	-	-
Suriname	0.5	0.01	-	-	-	-	-
Swaziland	2.8	0.02	-	-	-	-	-
Sweden	4.0	0.04	-	-	-	2.22E+08	-
Syria	48.3	1.00	-	-	-	-	-
Tajikistan	49.5	1.00	-	-	-	5.43E+07	-
Tanzania	40.4	0.01	5.90E+04	-	-	-	-
Thailand	5.4	0.53	1.56E+05	1.39E+05	-	3.40E+07	-
Togo	14.3	0.01	-	-	-	-	-
Tunisia	41.7	0.91	-	-	-	7.40E+06	-
Turkey	20.7	0.78	-	-	-	2.10E+08	-
Uganda	84.0	0.02	3.30E+04	6.00E+04	-	-	-
Ukraine	5.6	0.30	-	-	1.13E+06	-	3.50E+07
United Kingdom	3.1	0.40	-	-	-	-	-
Uruguay	0.5	0.01	-	-	-	-	-
USA	9.5	0.50	-	-	2.22E+06	8.20E+08	1.15E+08
Uzbekistan	52.2	0.99	-	3.00E+05	2.83E+06	-	-
Venezuela	3.6	0.30	-	-	-	-	-
Vietnam	6.9	0.35	5.40E+06	1.00E+06	-	1.70E+07	7.00E+06
Zambia	6.5	0.01	-	-	-	-	-
Zimbabwe	11.1	0.19	-	-	-	-	-
Global	-	-	**3.55E+08**	**8.53E+07**	**6.59E+07**	**1.37E+10**	**1.18E+09**

Notes

[a] For Kosovo assumed Serbia AWaRe and Serbia and Montenegro WSI

[b] For Montenegro assumed Serbia and Montenegro WSI

Table A. 6: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 1 – Antimony.

Country	Antimony Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Australia	1	3.64E+06	95	88.22	0.82
Bolivia	-	-	0	-	-
Canada	-	-	0	-	-
China	-	-	0	-	-
Kazakhstan	-	-	0	-	-

Country	Antimony				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Kyrgyzstan	-	-	0	-	-
Laos	-	-	0	-	-
Mexico	-	-	0	-	-
Morocco	-	-	0	-	-
Myanmar	-	-	0	-	-
Pakistan	-	-	0	-	-
Russia	-	-	0	-	-
South Africa	1	2.38E+06	292	21.20	0.54
Tajikistan	-	-	0	-	-
Thailand	-	-	0	-	-
Turkey	-	-	0	-	-
Vietnam	-	-	0	-	-
Global	**2**	**6.02E+06**	**4**	**61.73**	**0.71**

Table A. 7: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 2 – Bauxite.

Country	Bauxite				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Australia	4	8.16E+10	104	1.57	0.03
Bosnia & Herzegovina	-	-	0	-	-
Brazil	4	3.37E+10	95	0.45	0.01
China	17	1.79E+10	27	38.42	0.53
Croatia	-	-	0	-	-
Dominican Republic	-	-	0	-	-
Fiji	-	-	0	-	-
France	-	-	0	-	-
Ghana	1	5.00E+08	53	1.77	0.01
Greece	2	2.63E+09	140	22.79	0.37
Guinea	2	1.92E+10	102	0.99	0.01
Guyana	2	1.74E+09	111	0.47	0.01
Hungary	-	-	0	-	-
India	4	8.06E+09	40	22.01	0.96
Indonesia	2	7.67E+08	30	0.17	0.01
Iran	-	-	0	-	-
Jamaica	3	9.90E+09	102	4.97	0.02
Kazakhstan	1	5.00E+09	111	7.63	0.02
Malaysia	-	-	0	-	-
Montenegro	-	-	0	-	-
Mozambique	-	-	0	-	-
Pakistan	-	-	0	-	-
Russia	2	5.59E+09	100	3.03	0.01
Saudi Arabia	-	-	0	-	-
Sierra Leone	1	1.16E+09	100	0.67	0.01
Suriname	1	1.50E+09	55	0.55	0.01
Tanzania	-	-	0	-	-

Country	Bauxite Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Turkey	-	-	0	-	-
USA	-	-	0	-	-
Venezuela	1	2.00E+09	86	0.37	0.01
Vietnam	-	-	0	-	-
Global	**47**	**1.91E+11**	**74**	**6.25**	**0.11**

Table A. 8: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 3 – Chromite.

Country	Chromite Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Afghanistan	-	-	0	-	-
Albania	-	-	0	-	-
Brazil	-	-	0	-	-
China	-	-	0	-	-
Finland	1	1.00E+09	97	1.29	0.01
India	5	1.55E+09	92	0.60	0.62
Iran	-	-	0	-	-
Kazakhstan	1	3.00E+09	55	13.81	0.11
Kosovo	-	-	0	-	-
Madagascar	-	-	0	-	-
Oman	3	3.10E+08	41	24.74	1.00
Pakistan	-	-	0	-	-
Philippines	1	1.00E+07	21	1.26	0.01
Russia	-	-	0	-	-
South Africa	17	7.78E+09	55	22.20	0.58
Sudan	-	-	0	-	-
Turkey	3	8.86E+08	22	53.17	0.98
Zimbabwe	-	-	0	-	-
Global	**31**	**1.45E+10**	**48**	**18.66**	**0.48**

Table A. 9: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 4 – Coal.

Country	Coal Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Afghanistan	-	-	0	-	-
Argentina	-	-	0	-	-
Australia	78	4.42E+11	90	15.28	0.34
Bangladesh	-	-	0	-	-
Bhutan	-	-	0	-	-
Bosnia & Herzegovina	-	-	0	-	-
Botswana	-	-	0	-	-

Country	Coal				
	Operations	2014 Production	Production Coverage	Prod. Weighted Ave.	
	No.	kg	% of national data	AWaRe	WSI
Brazil	-	-	0	-	-
Bulgaria	-	-	0	-	-
Canada	17	7.84E+10	114	5.90	0.08
Chile	1	2.50E+09	60	0.33	0.01
China	372	1.51E+12	39	59.78	0.78
Colombia	5	7.57E+10	85	2.37	0.01
Czech Republic	4	8.25E+09	18	2.04	0.07
Dem. Rep. Congo	-	-	0	-	-
Egypt	-	-	0	-	-
Ethiopia	-	-	0	-	-
Georgia	-	-	0	-	-
Germany	7	1.57E+11	84	1.27	0.18
Greece	1	1.50E+10	30	3.20	0.99
Hungary	-	-	0	-	-
India	44	5.55E+11	95	7.30	0.82
Indonesia	49	3.23E+11	80	0.38	0.01
Iran	-	-	0	-	-
Kazakhstan	5	7.87E+10	69	43.70	0.85
Kosovo	-	-	0	-	-
Kyrgyzstan	-	-	0	-	-
Laos	-	-	0	-	-
Macedonia	-	-	0	-	-
Malawi	1	2.75E+07	39	7.30	0.01
Malaysia	-	-	0	-	-
Mexico	2	1.23E+10	77	100.00	1.00
Mongolia	5	9.36E+09	32	41.53	0.01
Montenegro	-	-	0	-	-
Mozambique	2	6.59E+09	78	7.30	0.01
Myanmar	-	-	0	-	-
New Caledonia	-	-	0	-	-
New Zealand	9	1.97E+10	492	0.27	0.01
Niger	-	-	0	-	-
Nigeria	-	-	0	-	-
North Korea	-	-	0	-	-
Norway	-	-	0	-	-
Pakistan	-	-	0	-	-
Peru	-	-	0	-	-
Philippines	1	8.08E+09	110	1.60	0.01
Poland	21	1.33E+11	102	2.08	0.08
Portugal	-	-	-	-	-
Rep. Congo	-	-	-	-	-
Romania	1	2.00E+09	8	1.16	0.07
Russia	137	3.49E+11	98	2.28	0.02
Serbia	1	4.00E+09	13	1.16	0.07
Slovakia	-	-	0	-	-
Slovenia	1	3.50E+09	113	1.16	0.07
South Africa	39	1.78E+11	68	21.66	0.56
South Korea	-	-	0	-	-
Spain	-	-	0	-	-

Country	Coal				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Swaziland	-	-	0	-	-
Tajikistan	-	-	0	-	-
Tanzania	1	2.03E+08	83	7.30	0.01
Thailand	1	1.60E+10	89	3.49	0.48
Turkey	-	-	0	-	-
Ukraine	30	5.02E+10	111	4.41	0.17
United Kingdom	2	1.32E+09	11	0.47	0.02
USA	299	8.86E+11	93	12.76	0.20
Uzbekistan	-	-	0	-	-
Venezuela	-	-	0	-	-
Vietnam	3	6.00E+09	14	23.10	0.50
Zimbabwe	1	1.33E+09	23	7.30	0.01
Global	**1138**	**4.93E+12**	**61**	**25.17**	**0.45**

Table A. 10: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 5 – Cobalt.

Country	Cobalt				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Australia	5	3.40E+06	54	14.59	0.01
Botswana	1	1.00E+05	51	21.20	0.54
Brazil	2	1.62E+06	46	1.44	0.01
Canada	5	3.93E+06	60	1.45	0.02
China	1	2.54E+06	30	10.88	1.00
Cuba	2	4.14E+06	129	11.97	0.02
Dem. Rep. Congo	5	3.64E+07	48	2.24	0.01
Finland	1	9.42E+05	45	1.06	0.01
Indonesia	-	-	0	-	-
Madagascar	1	2.92E+06	100	0.88	0.01
Morocco	1	1.39E+06	100	39.48	1.00
New Caledonia	1	1.38E+06	46	0.44	0.01
Papua New Guinea	1	2.13E+06	119	0.23	0.01
Philippines	1	1.85E+06	45	1.10	0.01
Russia	-	-	0	-	-
South Africa	1	1.13E+06	85	3.87	0.03
Zambia	-	-	0	-	-
Zimbabwe	1	8.00E+04	22	9.23	0.03
Global	**29**	**6.40E+07**	**50**	**4.46**	**0.07**

Table A. 11: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 6 – Copper.

	Copper				
				Prod. Weighted Ave.	
Country	Operations No.	2014 Production kg	Production Coverage % of national data	AWaRe	WSI
Albania	-	-	0	-	-
Argentina	1	1.03E+08	100	6.11	0.15
Armenia	2	3.50E+07	69	44.95	0.96
Australia	29	9.90E+08	102	34.26	0.07
Azerbaijan	2	4.78E+06	610	44.95	0.96
Bolivia	1	7.00E+06	65	1.82	0.01
Botswana	3	3.07E+07	64	35.13	0.05
Brazil	5	2.90E+08	121	0.60	0.01
Bulgaria	2	7.01E+07	65	1.16	0.07
Canada	22	6.57E+08	95	1.30	0.03
Chile	32	5.54E+09	96	70.55	0.98
China	12	3.22E+08	20	9.04	0.16
Colombia	1	4.12E+06	103	0.11	0.01
Cyprus	-	-	0	-	-
Dem. Rep. Congo	12	7.66E+08	72	2.44	0.01
Dominican Republic	1	1.10E+07	119	1.07	0.04
Ecuador	-	-	0	-	-
Eritrea	1	8.89E+07	100	33.82	1.00
Finland	3	4.21E+07	98	1.15	0.01
Georgia	-	-	0	-	-
India	2	3.57E+07	134	30.05	1.00
Indonesia	3	3.67E+08	97	3.82	0.01
Iran	1	2.00E+08	92	59.34	0.54
Kazakhstan	7	3.37E+08	72	24.31	0.23
Kyrgyzstan	-	-	0	-	-
Laos	2	1.60E+08	100	3.26	0.02
Macedonia	-	-	0	-	-
Mauritania	1	3.31E+07	100	31.30	0.02
Mexico	21	4.89E+08	95	15.00	0.96
Mongolia	2	2.73E+08	108	21.74	0.07
Morocco	-	-	0	-	-
Myanmar	1	2.00E+07	74	0.76	0.01
Namibia	1	5.09E+06	97	58.45	0.01
North Korea	-	-	0	-	-
Oman	-	-	0	-	-
Pakistan	1	1.31E+07	74	34.47	1.00
Papua New Guinea	1	7.59E+07	100	0.13	0.01
Peru	22	1.28E+09	93	39.80	0.56
Philippines	4	8.90E+07	97	3.97	0.56
Poland	1	5.06E+08	120	2.17	0.08
Portugal	1	5.14E+07	68	22.64	0.99
Romania	-	-	0	-	-
Russia	9	6.54E+08	91	6.57	0.05
Saudi Arabia	-	-	0	-	-
Serbia	1	5.00E+06	15	1.16	0.07
Slovakia	-	-	0	-	-

Country	Copper				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
South Africa	15	1.02E+08	129	18.82	0.45
Spain	4	1.06E+08	102	52.75	0.80
Sweden	4	7.93E+07	100	0.82	0.01
Tanzania	2	6.38E+06	100	56.41	0.02
Turkey	1	2.94E+07	24	1.74	0.02
USA	20	1.38E+09	100	75.76	0.94
Uzbekistan	1	6.50E+07	81	74.65	1.00
Vietnam	1	3.44E+06	21	0.99	0.04
Zambia	10	6.77E+08	95	7.30	0.01
Zimbabwe	2	4.89E+06	59	9.23	0.03
Global	**270**	**1.60E+10**	**87**	**40.94**	**0.55**

Table A. 12: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 7 – Diamonds.

Country	Diamonds				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Angola	2	1.31E+03	74	1.64	0.01
Australia	2	1.86E+03	100	12.02	0.03
Botswana	5	4.93E+03	100	36.10	0.01
Brazil	-	-	0	-	-
Cameroon	-	-	0	-	-
Canada	4	2.15E+03	89	1.00	0.01
China	-	-	0	-	-
Cote d'Ivoire	-	-	0	-	-
Dem. Rep. Congo	-	-	0	-	-
Ghana	-	-	0	-	-
Guinea	-	-	0	-	-
Guyana	-	-	0	-	-
India	1	7.42E+00	103	17.92	1.00
Lesotho	1	2.17E+01	31	27.33	0.78
Liberia	-	-	0	-	-
Namibia	1	3.77E+02	97	100.00	0.01
Rep. Congo	-	-	0	-	-
Russia	7	7.24E+03	95	7.33	0.01
Sierra Leone	-	-	0	-	-
South Africa	14	1.53E+03	95	27.08	0.74
Tanzania	1	3.77E+01	75	3.15	0.01
Togo	-	-	0	-	-
Zimbabwe	1	8.84E+01	9	21.20	0.54
Global	**39**	**1.96E+04**	**78**	**17.38**	**0.07**

Table A. 13: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 8 – Gold.

	Gold				
				Prod. Weighted Ave.	
Country	Operations No.	2014 Production kg	Production Coverage % of national data	AWaRe	WSI
Algeria	-	-	0	-	-
Argentina	9	58,917	82	23.71	0.54
Armenia	2	2,144	76	28.18	0.32
Australia	68	276,119	101	24.65	0.10
Azerbaijan	1	1,875	100	44.95	0.96
Bolivia	1	657	2	1.82	0.01
Botswana	1	958	100	21.20	0.54
Brazil	21	68,752	86	2.33	0.02
Bulgaria	2	5,486	69	1.16	0.07
Burkina Faso	7	36,199	97	16.25	0.01
Burundi	-	-	0	-	-
Cameroon	-	-	0	-	-
Canada	52	153,477	101	1.38	0.02
Chile	16	43,111	94	74.21	0.96
China	70	137,907	31	42.03	0.53
Colombia	6	7,252	13	0.37	0.01
Cote d'Ivoire	4	17,089	101	6.38	0.01
Dem. Rep. Congo	3	20,002	56	1.76	0.01
Dominican Republic	3	35,951	102	1.07	0.04
Ecuador	1	805	11	24.30	1.00
Egypt	1	11,734	100	96.24	0.01
Eritrea	1	840	100	33.82	1.00
Ethiopia	1	4,230	41	17.83	0.03
Fiji	1	1,188	99	0.66	0.01
Finland	8	7,578	94	1.24	0.01
French Guiana	1	505	25	0.77	0.01
Gabon	1	1,012	100	0.65	0.01
Georgia	1	1,493	65	44.95	0.96
Ghana	11	90,607	92	2.19	0.01
Greece	1	552	100	26.94	0.57
Guatemala	2	6,140	100	0.64	0.02
Guinea	2	16,986	108	65.31	0.02
Guyana	1	8	0	0.32	0.01
Honduras	1	2,762	100	0.51	0.01
India	1	1,555	118	4.03	1.00
Indonesia	8	66,332	96	1.02	0.01
Iran	1	529	30	59.34	0.54
Japan	1	7,000	98	0.35	0.02
Kazakhstan	12	30,508	73	7.02	0.10
Kenya	1	36	N/A	89.32	0.02
Kyrgyzstan	3	18,067	101	99.30	1.00
Laos	3	5,260	100	3.26	0.02
Liberia	-	-	0	-	-
Malaysia	5	4,220	98	0.36	0.01
Mali	7	39,989	88	79.50	0.03
Mauritania	2	9,624	100	30.63	0.02
Mexico	59	108,279	92	20.84	0.83

Country	Gold				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	Prod. Weighted Ave. WSI
Mongolia	3	20,119	101	34.42	0.10
Morocco	-	-	0	-	-
Mozambique	-	-	0	-	-
Myanmar	-	-	0	-	-
Namibia	2	1,249	58	58.05	0.01
New Zealand	4	10,395	87	0.98	0.01
Nicaragua	3	8,172	94	0.58	0.02
Niger	-	-	0	-	-
Nigeria	-	-	0	-	-
North Korea	1	8,709	-	4.35	0.02
Panama	1	10	N/A	0.60	0.01
Papua New Guinea	6	54,097	102	0.12	0.01
Peru	35	111,626	80	25.57	0.75
Philippines	9	19,665	107	4.61	0.07
Poland	1	2,706	1197	2.17	0.08
Portugal	1	31	-	43.59	0.65
Romania	-	-	0	-	-
Russia	50	181,257	73	4.18	0.01
Saudi Arabia	5	5,856	122	22.04	1.00
Senegal	1	6,588	100	13.88	0.02
Serbia	1	778	65	1.16	0.07
Sierra Leone	-	-	0	-	-
Slovakia	-	-	0	-	-
Solomon Islands	1	1,403	233	0.40	0.01
South Africa	51	153,271	101	26.64	0.75
South Korea	-	-	0	-	-
Spain	1	1,958	109	1.11	0.02
Sudan	1	2,239	3	100.00	0.98
Suriname	1	10,639	100	0.55	0.01
Sweden	5	6,403	93	0.85	0.01
Tajikistan	2	2,311	68	43.31	1.00
Tanzania	6	40,984	101	65.64	0.02
Thailand	1	4,185	91	3.49	0.48
Togo	-	-	0	-	-
Turkey	8	29,862	96	42.70	0.69
United Kingdom	-	-	N/A	-	-
Uruguay	1	1,875	109	0.40	0.01
USA	36	206,545	98	13.43	0.77
Uzbekistan	2	84,602	83	53.50	0.18
Venezuela	2	635	58	0.32	0.01
Vietnam	2	350	213	0.54	0.02
Zambia	1	4,803	100	7.30	0.01
Zimbabwe	15	9,719	63	12.12	0.18
Global	**661**	**2,296,777**	**76**	**21.66**	**0.31**

Table A. 14: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 9 – Iron Ore.

	IronOre				
				Prod. Weighted Ave.	
	Operations	2014 Production	Production Coverage		
Country	No.	kg	% of national data	AWaRe	WSI
Algeria	1	5.00E+08	55	43.11	0.95
Argentina	1	3.28E+08	-	8.59	0.02
Australia	40	6.29E+11	84	42.67	0.01
Austria	1	2.30E+09	95	1.16	0.07
Azerbaijan	-	-	0	-	-
Bahrain	1	9.80E+09	-	7.30	1.00
Bhutan	-	-	0	-	-
Bosnia & Herzegovina	1	2.10E+09	99	1.16	0.07
Brazil	29	4.23E+11	122	2.36	0.02
Canada	6	4.70E+10	106	0.61	0.01
Chile	5	1.24E+10	66	64.84	1.00
China	65	9.71E+10	6	32.70	0.62
Colombia	1	6.49E+08	96	0.38	0.01
Ecuador	-	-	0	-	-
Egypt	1	1.94E+09	129	100.00	0.98
Germany	1	4.00E+08	88	1.30	0.09
Guatemala	-	-	0	-	-
India	34	9.03E+10	70	2.15	0.75
Indonesia	2	8.12E+08	27	0.41	0.01
Iran	9	4.11E+10	82	48.14	1.00
Kazakhstan	4	1.86E+10	36	7.43	0.15
Liberia	1	4.90E+09	102	0.78	0.01
Malaysia	2	2.34E+09	24	0.37	0.01
Mali	-	-	0	-	-
Mauritania	3	1.29E+10	97	26.73	0.01
Mexico	10	1.53E+10	61	14.51	0.21
Mongolia	-	-	0	-	-
Morocco	-	-	0	-	-
New Zealand	2	1.89E+09	58	0.56	0.01
Nigeria	-	-	0	-	-
North Korea	1	9.40E+08	34	4.89	0.02
Norway	2	2.45E+09	23	1.13	0.01
Pakistan	-	-	0	-	-
Peru	1	7.19E+09	100	43.86	1.00
Philippines	-	-	0	-	-
Russia	21	1.06E+11	103	3.63	0.13
Sierra Leone	1	5.15E+09	35	0.89	0.01
South Africa	8	7.39E+10	91	46.87	0.24
South Korea	-	-	0	-	-
Sudan	-	-	0	-	-
Swaziland	-	-	0	-	-
Sweden	4	2.75E+10	77	0.78	0.01
Thailand	-	-	0	-	-
Tunisia	1	1.86E+08	60	43.11	0.95
Turkey	1	2.58E+09	33	55.19	1.00
Uganda	-	-	0	-	-

Country	IronOre				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Ukraine	8	6.32E+10	92	4.12	0.07
USA	10	5.51E+10	96	3.92	0.06
Venezuela	1	1.12E+10	98	0.37	0.01
Vietnam	1	2.46E+08	11	0.60	0.02
Global	**280**	**1.77E+12**	**52**	**22.51**	**0.15**

Table A. 15: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 10 – Lead.

Country	Lead				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Argentina	-	-	0	-	-
Australia	10	6.32E+08	87	17.17	0.01
Bolivia	2	6.21E+07	82	24.88	0.99
Bosnia & Herzegovina	-	-	0	-	-
Brazil	1	1.20E+07	120	3.59	0.02
Bulgaria	-	-	0	-	-
Burkina Faso	1	4.42E+05	11	24.44	0.01
Canada	1	1.50E+06	41	0.21	0.01
Chile	1	1.80E+06	67	0.23	0.01
China	9	5.65E+07	2	24.77	0.34
Greece	-	-	0	-	-
Guatemala	1	1.04E+07	100	1.79	0.01
Honduras	1	1.55E+07	100	0.51	0.01
India	5	1.10E+08	109	23.48	1.00
Iran	-	-	0	-	-
Ireland	2	4.37E+07	108	0.87	0.02
Kazakhstan	2	3.12E+07	83	8.45	0.20
Kosovo	-	-	0	-	-
Laos	-	-	0	-	-
Macedonia	-	-	0	-	-
Mexico	28	2.29E+08	91	39.51	0.84
Montenegro	-	-	0	-	-
Morocco	1	1.90E+07	69	74.85	1.00
Myanmar	-	-	0	-	-
Namibia	1	1.23E+07	122	53.32	0.01
Nigeria	-	-	0	-	-
North Korea	-	-	0	-	-
Pakistan	-	-	0	-	-
Peru	24	2.05E+08	73	2.93	0.48
Poland	2	5.71E+07	69	2.05	0.07
Portugal	1	3.19E+06	100	22.64	0.99
Russia	1	1.20E+07	6	2.74	0.01
Serbia	-	-	0	-	-
Slovakia	-	-	0	-	-
South Africa	1	3.76E+07	128	11.52	0.01

Country	Lead				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
South Korea	-	-	0	-	-
Spain	1	8.28E+05	69	34.53	0.25
Sweden	3	7.09E+07	100	1.15	0.01
Tajikistan	-	-	0	-	-
Turkey	-	-	0	-	-
United Kingdom	-	-	0	-	-
USA	5	3.33E+08	94	0.56	0.01
Vietnam	-	-	0	-	-
Global	**103**	**1.96E+09**	**36**	**15.04**	**0.27**

Table A. 16: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 11 – Lithium.

Country	Lithium				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Argentina	-	-	0	-	-
Australia	1	6.90E+07	16	9.25	0.01
Brazil	-	-	0	-	-
Canada	-	-	0	-	-
Chile	1	3.00E+07	48	94.67	1.00
China	-	-	0	-	-
Portugal	-	-	0	-	-
USA	-	-	0	-	-
Zimbabwe	-	-	0	-	-
Global	**2**	**9.90E+07**	**15**	**35.13**	**0.31**

Table A. 17: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 12 – Manganese.

Country	Manganese				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Australia	3	7.36E+09	97	15.42	0.01
Austria	1	2.20E+09	-	1.16	0.07
Brazil	4	2.40E+09	96	1.21	0.01
Bulgaria	-	-	0	-	-
China	2	2.06E+09	13	0.60	0.02
Cote d'Ivoire	-	-	0	-	-
Egypt	-	-	0	-	-
Gabon	2	3.81E+09	95	0.65	0.01
Georgia	-	-	0	-	-
Ghana	1	1.50E+09	111	2.22	0.01
Hungary	-	-	0	-	-
India	3	1.78E+09	82	2.55	0.87

Country	Manganese				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Iran	-	-	0	-	-
Kazakhstan	4	1.80E+09	69	26.75	1.00
Malaysia	-	-	0	-	-
Mexico	1	9.44E+08	145	2.62	0.03
Morocco	-	-	0	-	-
Myanmar	-	-	0	-	-
Namibia	-	-	0	-	-
Romania	-	-	0	-	-
South Africa	4	1.16E+10	84	58.45	0.01
Sudan	-	-	0	-	-
Thailand	-	-	0	-	-
Turkey	-	-	0	-	-
Ukraine	1	1.53E+09	100	4.12	0.07
Vietnam	-	-	0	-	-
Global	**26**	**3.70E+10**	**68**	**23.39**	**0.11**

Table A. 18: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 13 – Molybdenum.

Country	Molybdenum				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Argentina	-	-	0	-	-
Armenia	1	5.80E+06	96	44.95	0.96
Brazil	-	-	0	-	-
Bulgaria	1	1.44E+05	-	1.16	0.07
Canada	3	8.79E+06	97	0.51	0.01
Chile	7	3.88E+07	80	64.40	1.00
China	8	4.39E+07	36	40.65	0.49
Iran	1	3.67E+06	105	59.34	0.54
Mexico	1	1.08E+07	75	12.31	1.00
Mongolia	1	2.19E+06	122	2.99	0.01
Norway	-	-	0	-	-
Peru	4	1.65E+07	97	80.59	0.88
Russia	1	3.15E+06	88	6.78	0.01
South Korea	1	3.65E+05	-	0.52	0.09
Turkey	-	-	0	-	-
USA	12	6.56E+07	100	46.09	0.71
Uzbekistan	1	4.20E+05	-	74.65	1.00
Global	**42**	**2.00E+08**	**68**	**46.53**	**0.70**

Table A. 19: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 14 – Nickel.

	Nickel				
				Prod. Weighted Ave.	
	Operations	2014 Production	Production Coverage		
Country	No.	kg	% of national data	AWaRe	WSI
Albania	-	-	0	-	-
Australia	11	2.51E+08	103	15.40	0.01
Botswana	2	2.14E+07	143	21.20	0.54
Brazil	6	1.02E+08	119	1.41	0.01
Canada	7	2.34E+08	99	1.41	0.03
China	6	1.36E+08	151	29.40	0.79
Colombia	1	4.43E+07	107	0.38	0.01
Cuba	2	6.29E+07	126	11.97	0.02
Finland	2	2.14E+07	109	1.16	0.01
Greece	1	1.80E+07	84	51.36	0.51
Guatemala	-	-	0	-	-
Indonesia	2	9.57E+07	44	0.26	0.01
Kosovo	-	-	0	-	-
Macedonia	1	1.80E+07	N/A	20.00	0.54
Madagascar	1	3.71E+07	100	0.88	0.01
Morocco	-	-	0	-	-
Myanmar	-	-	0	-	-
New Caledonia	3	8.63E+07	48	1.20	0.01
Norway	-	-	0	-	-
Papua New Guinea	1	2.10E+07	119	0.23	0.01
Philippines	1	2.10E+07	6	1.10	0.01
Poland	-	-	0	-	-
Russia	3	2.40E+08	91	1.88	0.01
South Africa	20	5.57E+07	101	14.09	0.33
Spain	1	8.63E+06	100	60.23	1.00
Turkey	-	-	0	-	-
USA	1	4.18E+06	102	0.80	0.03
Venezuela	-	-	0	-	-
Vietnam	1	6.85E+06	100	0.99	0.04
Zimbabwe	4	1.66E+07	100	7.85	0.02
Global	**77**	**1.50E+09**	**73**	**8.62**	**0.12**

Table A. 20: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 15 – Palladium.

	Palladium				
				Prod. Weighted Ave.	
	Operations	2014 Production	Production Coverage		
Country	No.	kg	% of national data	AWaRe	WSI
Australia	-	-	0	-	-
Botswana	-	-	0	-	-
Canada	5	25,139	134	1.12	0.03
China	-	-	0	-	-
Finland	1	808	100	1.29	0.01
Poland	-	-	0	-	-

Country	Palladium				
	Operations	2014 Production	Production Coverage	Prod. Weighted Ave.	
	No.	kg	% of national data	AWaRe	WSI
Russia	2	82,735	102	2.05	0.01
Serbia	-	-	0	-	-
South Africa	18	52,234	89	21.20	0.54
USA	2	12,448	102	10.45	0.18
Zimbabwe	3	10,281	101	8.07	0.02
Global	**31**	**183,645**	**100**	**8.27**	**0.18**

Table A. 21: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 16 – Phosphate.

Country	Phosphate				
	Operations	2014 Production	Production Coverage	Prod. Weighted Ave.	
	No.	kg	% of national data	AWaRe	WSI
Algeria	-	-	0	-	-
Australia	1	7.72E+08	31	13.83	0.01
Brazil	7	6.14E+09	102	0.71	0.01
Burkina Faso	-	-	0	-	-
Chile	-	-	0	-	-
China	-	-	0	-	-
Colombia	-	-	0	-	-
Cuba	-	-	0	-	-
Egypt	-	-	0	-	-
Finland	1	9.46E+08	100	1.06	0.01
India	-	-	0	-	-
Iran	-	-	0	-	-
Iraq	-	-	0	-	-
Israel	1	3.36E+09	121	51.67	1.00
Jordan	-	-	0	-	-
Kazakhstan	-	-	0	-	-
Malawi	-	-	0	-	-
Mali	-	-	0	-	-
Mexico	-	-	0	-	-
Morocco	-	-	0	-	-
Nauru	-	-	0	-	-
North Korea	-	-	0	-	-
Pakistan	-	-	0	-	-
Peru	1	3.80E+09	35	44.55	1.00
Philippines	-	-	0	-	-
Russia	2	8.39E+09	80	0.86	0.01
Saudi Arabia	-	-	0	-	-
Senegal	-	-	0	-	-
South Africa	1	2.08E+09	103	21.20	0.54
Sri Lanka	-	-	0	-	-
Syria	-	-	0	-	-
Tanzania	-	-	0	-	-
Thailand	-	-	0	-	-
Togo	-	-	0	-	-

Country	Phosphate				
	Operations	2014 Production	Production Coverage	Prod. Weighted Ave.	
	No.	kg	% of national data	AWaRe	WSI
Tunisia	-	-	0	-	-
USA	8	2.17E+10	80	1.41	0.71
Uzbekistan	-	-	0	-	-
Venezuela	-	-	0	-	-
Vietnam	-	-	0	-	-
Zimbabwe	-	-	0	-	-
Global	**22**	**4.71E+10**	**19**	**9.35**	**0.51**

Table A. 22: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 17 – Platinum.

Country	Platinum				
	Operations	2014 Production	Production Coverage	Prod. Weighted Ave.	
	No.	kg	% of national data	AWaRe	WSI
Botswana	-	-	0	-	-
Canada	5	9,745	91	1.12	0.03
China	1	2,000	143	10.88	1.00
Finland	1	1,060	100	1.29	0.01
Poland	-	-	0	-	-
Russia	3	19,965	91	2.32	0.01
Serbia	-	-	0	-	-
South Africa	25	111,692	119	20.75	0.53
USA	2	3,655	100	10.45	0.18
Zimbabwe	3	12,757	102	8.10	0.02
Global	**40**	**160,875**	**111**	**15.79**	**0.39**

Table A. 23: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 18 – Potash.

Country	Potash				
	Operations	2014 Production	Production Coverage	Prod. Weighted Ave.	
	No.	kg	% of national data	AWaRe	WSI
Belarus	-	-	0	-	-
Brazil	1	4.92E+08	[a]	1.13	0.05
Canada	10	1.74E+10	[a]	11.70	0.17
Chile	1	1.99E+09	[a]	94.67	1.00
China	-	-	0	-	-
Germany	-	-	0	-	-
Israel	1	3.50E+09	[a]	51.67	1.00
Jordan	-	-	0	-	-
Russia	3	4.38E+10	[a]	1.84	0.02
Spain	-	-	0	-	-
United Kingdom	-	-	0	-	-
USA	6	1.38E+09	[a]	90.06	1.00
Global	**22**	**6.86E+10**	[a]	**11.35**	**0.15**

Country	Potash				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Note: [a] Production coverage unable to be calculated due to difference in reporting basis.					

Table A. 24: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 19 – Rutile.

Country	Rutile				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Australia	4	2.72E+08	91	52.89	0.45
Brazil	-	-	0	-	-
India	-	-	0	-	-
Kenya	1	2.42E+07	46	8.63	0.02
Madagascar	-	-	0	-	-
Malaysia	-	-	0	-	-
Mozambique	1	6.10E+06	100	4.82	0.01
Senegal	-	-	0	-	-
Sierra Leone	1	1.14E+08	137	0.67	0.01
South Africa	2	2.98E+07	22	69.22	0.13
Sri Lanka	-	-	0	-	-
Ukraine	-	-	0	-	-
Global	**9**	**4.46E+08**	**60**	**37.56**	**0.29**

Table A. 25: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 20 – Silver.

Country	Silver				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Algeria	-	-	0	-	-
Argentina	6	7.57E+05	84	19.86	0.12
Armenia	1	1.33E+04	100	44.95	0.96
Australia	29	1.51E+06	82	20.12	0.02
Azerbaijan	1	9.70E+02	100	44.95	0.96
Bolivia	5	7.70E+05	57	12.16	0.45
Botswana	1	1.96E+04	88	36.10	0.01
Brazil	1	9.24E+03	42	0.90	0.01
Bulgaria	1	7.34E+03	13	1.16	0.07
Burkina Faso	1	7.51E+03	13	24.44	0.01
Canada	16	3.10E+05	63	1.28	0.02
Chile	14	1.38E+06	88	69.38	0.92
China	4	1.66E+05	5	56.34	0.77
Colombia	3	4.71E+03	41	0.34	0.01
Cote d'Ivoire	1	5.85E+02	100	7.37	0.01
Dem. Rep. Congo	1	6.49E+03	100	2.24	0.01
Dominican Republic	3	1.37E+05	108	1.07	0.04

Country	Silver				
	Operations	2014 Production	Production Coverage	Prod. Weighted Ave.	
Country	No.	kg	% of national data	AWaRe	WSI
Ecuador	-	-	0	-	-
Eritrea	1	4.74E+04	100	33.82	1.00
Ethiopia	-	-	0	-	-
Fiji	-	-	0	-	-
Finland	1	2.18E+02	2	1.29	0.01
Ghana	-	-	0	-	-
Greece	-	-	0	-	-
Guatemala	2	8.58E+05	100	1.47	0.01
Honduras	1	5.68E+04	97	0.51	0.01
India	-	-	0	-	-
Indonesia	5	1.78E+05	150	0.29	0.01
Iran	-	-	0	-	-
Ireland	1	2.43E+03	38	1.14	0.02
Japan	-	-	0	-	-
Kazakhstan	5	2.55E+05	26	5.81	0.07
Laos	2	3.98E+04	100	3.26	0.02
Macedonia	-	-	0	-	-
Malaysia	1	6.50E+02	122	0.34	0.01
Mexico	60	5.05E+06	88	30.71	0.76
Mongolia	1	2.78E+04	71	37.53	0.11
Morocco	2	1.89E+05	68	40.28	1.00
Namibia	1	1.87E+04	862	53.32	0.01
New Zealand	-	-	0	-	-
Nicaragua	-	-	0	-	-
Niger	-	-	0	-	-
North Korea	-	-	0	-	-
Papua New Guinea	4	8.00E+04	95	0.40	0.01
Peru	43	3.26E+06	86	9.55	0.46
Philippines	3	1.54E+04	67	2.01	0.02
Poland	-	-	0	-	-
Portugal	1	4.32E+04	100	22.64	0.99
Romania	-	-	0	-	-
Russia	10	9.89E+05	74	3.42	0.02
Saudi Arabia	-	-	0	-	-
Serbia	-	-	0	-	-
Slovakia	-	-	0	-	-
Solomon Islands	-	-	0	-	-
South Africa	-	-	0	-	-
South Korea	-	-	0	-	-
Spain	2	2.82E+04	91	28.74	0.21
Sudan	-	-	0	-	-
Sweden	4	3.96E+05	104	0.93	0.01
Tajikistan	-	-	0	-	-
Tanzania	1	3.15E+03	22	2.27	0.01
Thailand	1	3.09E+04	106	3.49	0.48
Turkey	5	2.38E+05	116	3.39	0.04
USA	14	7.52E+05	63	12.37	0.46
Uzbekistan	-	-	0	-	-
Global	**257**	**1.77E+07**	**64**	**21.55**	**0.46**

Table A. 26: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 21 – Tin.

	Tin				
				Prod. Weighted Ave.	
Country	Operations No.	2014 Production kg	Production Coverage % of national data	AWaRe	WSI
Australia	1	6.22E+06	86	0.31	0.01
Bolivia	-	-	0	-	-
Brazil	2	5.18E+06	30	0.10	0.01
Burundi	-	-	0	-	-
China	-	-	0	-	-
Dem. Rep. Congo	-	-	0	-	-
Egypt	-	-	0	-	-
India	-	-	0	-	-
Indonesia	2	3.23E+07	46	0.50	0.01
Laos	-	-	0	-	-
Malaysia	1	2.24E+06	59	0.44	0.01
Mongolia	-	-	0	-	-
Myanmar	-	-	0	-	-
Niger	-	-	0	-	-
Nigeria	-	-	0	-	-
Peru	1	2.42E+07	105	3.74	1.00
Portugal	-	-	0	-	-
Russia	-	-	0	-	-
Rwanda	-	-	0	-	-
Tanzania	-	-	0	-	-
Thailand	-	-	0	-	-
Uganda	-	-	0	-	-
Vietnam	-	-	0	-	-
Global	**7**	**7.02E+07**	**20**	**1.57**	**0.35**

Table A. 27: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 22 – Tungsten.

	Tungsten				
				Prod. Weighted Ave.	
Country	Operations No.	2014 Production kg	Production Coverage % of national data	AWaRe	WSI
Australia	-	-	0	-	-
Austria	-	-	0	-	-
Bolivia	-	-	0	-	-
Brazil	-	-	0	-	-
Burundi	-	-	0	-	-
Canada	1	2.96E+06	110	2.15	0.01
China	1	8.11E+06	12	0.23	0.03
Dem. Rep. Congo	-	-	0	-	-
Kyrgyzstan	-	-	0	-	-
Myanmar	-	-	0	-	-
North Korea	-	-	0	-	-
Peru	-	-	0	-	-

Country	Tungsten				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Portugal	1	8.46E+05	126	4.33	0.53
Russia	-	-	0	-	-
Rwanda	-	-	0	-	-
Spain	1	9.03E+05	87	24.39	0.17
Thailand	-	-	0	-	-
Uganda	-	-	0	-	-
Uzbekistan	-	-	0	-	-
Vietnam	-	-	0	-	-
Global	**4**	**1.28E+07**	**15**	**2.65**	**0.07**

Table A. 28: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 23 – Uranium.

Country	Uranium				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Australia	4	5.54E+06	94	22.49	0.02
Brazil	1	2.73E+05	100	4.92	0.03
Canada	4	1.07E+07	104	2.91	0.01
China	7	1.94E+06	110	13.25	0.50
Czech Republic	1	2.28E+05	117	1.16	0.07
Germany	1	3.90E+04	100	1.83	0.12
India	-	-	0	-	-
Kazakhstan	17	2.50E+07	92	36.65	0.98
Malawi	1	1.07E+06	245	7.30	0.01
Namibia	2	4.18E+06	109	48.95	0.01
Niger	2	4.52E+06	92	43.83	0.06
Pakistan	-	-	0	-	-
Romania	-	-	0	-	-
Russia	3	3.53E+06	100	6.03	0.01
South Africa	2	6.71E+05	101	27.33	0.78
Ukraine	1	1.09E+06	96	4.12	0.07
USA	8	2.29E+06	103	32.97	0.43
Uzbekistan	1	2.84E+06	100	50.00	0.04
Global	**55**	**6.40E+07**	**97**	**27.71**	**0.44**

Table A. 29: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 24 – Zinc.

Country	Zinc				
	Operations No.	2014 Production kg	Production Coverage % of national data	Prod. Weighted Ave. AWaRe	WSI
Algeria	1	2.00E+07	-	68.26	0.21
Argentina	1	1.36E+07	24	35.55	0.02
Armenia	1	5.47E+06	71	44.95	0.96

Country	Zinc				
	Operations	2014 Production	Production Coverage	Prod. Weighted Ave.	
	No.	kg	% of national data	AWaRe	WSI
Australia	9	1.44E+09	92	13.50	0.01
Bolivia	4	3.01E+08	67	18.27	0.75
Bosnia & Herzegovina	-	-	0	-	-
Brazil	1	1.50E+08	94	3.59	0.02
Bulgaria	-	-	0	-	-
Burkina Faso	1	7.07E+07	104	24.44	0.01
Canada	8	3.52E+08	100	2.24	0.03
Chile	1	3.70E+07	82	0.23	0.01
China	17	5.77E+08	11	47.42	0.58
Dem. Rep. Congo	-	-	0	-	-
Finland	3	4.36E+07	95	1.06	0.01
Greece	-	-	0	-	-
Guatemala	1	1.34E+07	100	1.79	0.01
Honduras	1	3.00E+07	100	0.51	0.01
India	5	7.70E+08	115	19.78	1.00
Iran	1	9.00E+07	60	10.66	1.00
Ireland	2	3.01E+08	106	0.86	0.02
Kazakhstan	5	4.00E+08	104	9.30	0.23
Kosovo	-	-	0	-	-
Laos	-	-	0	-	-
Macedonia	-	-	0	-	-
Mexico	28	6.29E+08	95	30.43	0.92
Mongolia	1	5.00E+07	107	80.97	0.01
Montenegro	-	-	0	-	-
Morocco	1	2.20E+06	5	74.85	1.00
Myanmar	-	-	0	-	-
Namibia	2	1.82E+08	102	53.32	0.01
Nigeria	-	-	0	-	-
North Korea	1	3.60E+07	113	4.35	0.02
Peru	25	1.09E+09	83	3.99	0.47
Poland	1	6.00E+07	107	1.95	0.07
Portugal	1	6.74E+07	100	22.64	0.99
Romania	-	-	0	-	-
Russia	5	1.60E+08	74	11.77	0.09
Saudi Arabia	-	-	0	-	-
Serbia	-	-	0	-	-
Slovakia	-	-	0	-	-
South Africa	1	2.90E+07	111	11.52	0.01
South Korea	-	-	0	-	-
Spain	1	2.84E+07	106	34.53	0.25
Sweden	3	2.22E+08	100	1.09	0.01
Tajikistan	-	-	0	-	-
Thailand	1	3.00E+07	88	1.60	0.02
Tunisia	-	-	0	-	-
Turkey	1	3.62E+07	17	1.74	0.02
USA	6	7.96E+08	97	0.99	0.01
Uzbekistan	1	4.50E+07	-	74.65	1.00
Vietnam	-	-	0	-	-
Global	**140**	**8.08E+09**	**59**	**15.42**	**0.35**

Table A. 30: Production weighted average factors at the national boundary, based upon operation production data (SNL, 2017) and watershed scale AWaRe and WSI. Countries with no known production are omitted. Part 25 – Zircon

	Zircon				
				Prod. Weighted Ave.	
Country	Operations No.	2014 Production kg	Production Coverage % of national data	AWaRe	WSI
Australia	4	6.09E+08	152	25.30	0.13
Brazil	-	-	0	-	-
China	-	-	0	-	-
India	-	-	0	-	-
Indonesia	-	-	0	-	-
Kenya	1	4.49E+06	11	8.63	0.02
Madagascar	-	-	0	-	-
Malaysia	-	-	0	-	-
Mozambique	1	5.08E+07	100	4.82	0.01
Russia	-	-	0	-	-
Senegal	1	9.04E+06	109	74.23	0.02
Sierra Leone	1	2.67E+06	143	0.67	0.01
South Africa	2	1.47E+08	37	67.60	0.22
Sri Lanka	-	-	0	-	-
Ukraine	-	-	0	-	-
USA	1	2.51E+07	22	0.76	0.01
Vietnam	-	-	0	-	-
Global	**11**	**8.48E+08**	**72**	**31.03**	**0.14**

C. Conference Abstracts and Papers

During the period of candidature, a variety of conference abstracts and presentations were prepared. The following abstracts or papers were presented by the author, and are shown in the following pages:

1. **Life-cycle based water footprinting methodology in the production of metal commodities**
   Northey, S., Haque, N., Mudd, G. (2015).
   LCM Australia 2015, Melbourne, VIC, 23-27 November 2015, 2p.

2. **The Challenges in Estimating the Water Footprint of Mined Commodities**
   Northey, S.A., Mudd, G.M., Haque, N. (2015).
   Seng 2015 national conference, Adelaide, VIC, 9-10 September 2015, paper 15, 4p.

3. **Resource Depletion Scenarios – How should we address the limitations?**
   Northey, S.A., Mudd, G.M. (2016).
   35th International Geological Congress, Cape Town, South Africa, 28 August to 5 September 2016, paper 2731, 2p.

4. **Water Footprinting — Communicating Mine Site Water Performance in a Circular Economy**
   Northey, S., Upton, M., Williamson, P., Hoekstra, D. (2017).
   Engineering Solutions for Sustainability: Materials and Resources 3 symposium, Denver, CO, United States, 18-19 February 2017, 1p.

Life-cycle based water footprinting methodology in the production of metal commodities

Stephen Northey[1,2], Nawshad Haque[2], Gavin Mudd[1]
[1] Monash University
[2] CSIRO Mineral Resources
Email: stephen.northey@monash.edu

ABSTRACT

The production of metal commodities is often dependent upon access to water resources, particularly during the initial mining and mineral processing stages of production where the majority of water is typically used. Given that different metals are sourced from mines operating in different regions, the relative water supply risk facing individual metals will differ.

Recent advances in water footprinting methodology have resulted in a variety of spatial indicators for water scarcity and stress being available. Different indicators assess these issues from different perspectives, such as assessing the level of competition for water resources in a given area or relating the impact of water consumption to the long-term availability of water in a given area. Mapping the location of metal ore mining and processing against these various indicators provides some insight into which metal commodities are potentially at risk from water supply related issues.

Keywords: water footprint, mining, metals, water scarcity

1. INTRODUCTION

The concept of a water footprint has been developed, partially independent of developments in the assessment of water use within life cycle assessment. The recent international standard for water footprinting, ISO 14046:2014 [1], advocates taking a life cycle approach when conducting a water footprinting. This provides a framework for communicating estimates of the consumptive and degradative water use impact associated with products and processes, in a way that is complementary to life cycle assessment studies that consider a broader range of environmental impacts.

Several of the major environmental impacts associated with metal production are the impacts to water resources, particularly at the initial mining and mineral processing stage of production. In the particular regions where mines operate they can be large local consumers of water and are often also perceived as being major contributors to water quality risks. Due to this the development of any new mines is often predicated upon the availability of water resources.

The interactions of mining operations with water resources is very site specific, depending heavily on: local climate, processing configurations, and site water management strategies. Assessments of mine site water withdrawals have highlighted significant variability existing between sites when expressed on a product basis (e.g. m^3 / t metal) [2-3]. However this type of data by itself is not particularly informative and would lead to perverse outcomes if used as the basis of decision making for sourcing of mined materials. This is because the unit water consumption associated with mined products is often lower in regions experiencing water scarcity, due to increased optimisation to reduce the water requirements of processing. Benchmarking the water consumption associated with mined products produced from different regions therefore requires methods to account for these regional differences in water scarcity.

2. CONSUMPTIVE WATER USE IMPACT CHARACTERISATION METHODS

The impacts associated with water use vary heavily due to differences in the local availability, quality and demand for water in different regions. A variety of spatially explicit impact characterisation factors have been developed to assess the consumption of water that occurs in different regions. Examples of mid-point characterisation factors include the water stress index [4], which provides a measure of competition for water resources in a region, and the water depletion index [5], which indicates the potential for water consumption in an area to lead to long-term decreases in water availability. A range of other characterisation methods also exist at the mid-point and end-point level, with each providing a different perspective on the nature of impacts associated with water use [6]. The main differences between the methods include the conceptual basis for estimating relative water scarcity of regions, how to account for water quality, and the nature of how water deprivation impacts various end-users.

3. QUANTIFYING THE EXPOSURE OF THE MINING INDUSTRY TO WATER STRESS

Characterisation factors, such as the water stress index, can also be used to begin to quantify the overall exposure of an industry to water stress related issues. In highly water stressed regions, companies may face increased difficulty in gaining access to water rights and entitlements due to greater competition for water resources. Additionally more scrutiny may be placed on water discharges due to the proximity of other water users that could potentially be adversely affected.

The water stress index has been evaluated against national production data for mining, mineral processing and metal production on a global scale [7]. Metals such as chromium, platinum and copper are found to be mined and processed in countries with relatively high water stress. Whereas other metals such as lead, molybdenum, titanium, tin, nickel and cobalt are produced from countries that experience relatively less water stress. There is also differences in the exposure of individual stages of a metals supply chain to water stress. For instance, copper smelting and refining is occurring in countries with lower water stress than the countries where copper ores are being mined and concentrated.

Limitations exist when using national scale production and water stress data to make these types of assessments. Countries such as Australia are hydrologically diverse and display an uneven distribution of water use. Therefore the water stress index of individual watersheds in a country can be substantially different to the national average. As an example, Fig. 1 shows the water stress index associated with global copper production. The production weighted average water stress using country scale data is 0.55, compared to 0.50 when using individual watershed and mine data. The watershed scale data is also more binary in determining whether a production facility is located in a water stressed area. Based upon this, sub-national information pertaining to the location of mine sites and other industrial production facilities should ideally be included in LCI databases to enable accurate estimation of water related impacts.

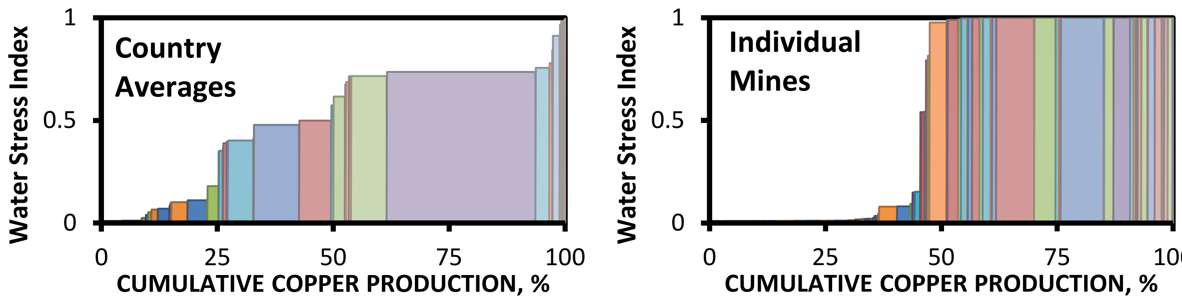

Fig 1. Water stress index associated with global mine copper production in 2012, based upon country scale data (left) and individual mine/watershed data (right).

4. ACKNOWLEDGEMENTS

The financial support provided by the CSIRO Mineral Resources collaboration fund is also gratefully acknowledged.

5. REFERENCES

1 International Standards Organisation *ISO 14046 Environmental Management Standard - Water Footprint – Principles, requirements and guidelines.* 2014

2 Northey, Haque, Mudd, *Using sustainability reporting to assess the environmental footprint of copper mining*, Journal of Cleaner Production 40, pp. 118-128, 2013. http://dx.doi.org/10.1016/j.jclepro.2012.09.027

3 Gunson, *Quantifying, Reducing and Improving Mine Water Use*, PhD Thesis, University of British Columbia, 2013

4 Pfister, Koehler, Hellweg, *Assessing the Environmental Impacts of Freshwater Consumption in LCA*, Environmental Science & Technology 43, pp. 4098-4104, 2009. /http://dx.doi.org/10.1021/es802423e

5 Berger, van der Ent, Eisner, Bach, Finkbeiner, *Water Accounting and Vulnerability Evaluation (WAVE): Considering Atmospheric Evaporation Recycling and the Risk of Freshwater Depletion in Water Footprinting.* Environmental Science & Technology 48, pp. 4521-4528, 2014. http://dx.doi.org/10.1021/es404994t

6 Boulay, Motoshita, Pfister, Bulle, Munoz, Franceschini, Margni, *Analysis of water use impact assessment methods (part A): evaluation of modelling choices based on a quantitative comparison of scarcity and human health indicators*, Int. J. of Life Cycle Assessment 20, pp. 139-160, 2015, http://dx.doi.org/10.1007/s11367-014-0814-2

7 Northey, Haque, Lovel, Cooksey, *Evaluating the application of water footprint methods to primary metal production systems,* Minerals Engineering 69, pp. 65-80, 2014. http://dx.doi.org/10.1016/j.mineng.2014.07.006

The Challenges in Estimating the Water Footprint of Mined Commodities

Stephen A. Northey[1*], Gavin M. Mudd[1] and Nawshad Haque[2]

[1] Environmental Engineering, Monash University, Clayton VIC
[2] Mineral Resources Flagship, CSIRO, Clayton VIC

[*] Corresponding author. Email: stephen.northey@monash.edu

ABSTRACT: *The concept of a 'water footprint' has gradually developed over the past decade as an extension to the 'virtual' or 'embodied water' concept. A 'water footprint' estimate attempts to quantify the environmental impacts that arise from the use of water during the manufacture, use or disposal of a product or service. Methodologies for estimating water footprints have been evolving to account for factors such as changes to water quality and the relative scarcity of water in different regions. Recently an international standard for water footprinting (ISO 14046) was developed to provide an over-arching framework for how studies should be conducted and presented.*

Despite this progress there are still challenges to address to improve the methodology underpinning water footprinting studies, particularly when applied to mined products. As an example, mines are often transient in nature. The production only lasts a decade or few decades before the closure, rehabilitation or abandonment of the mine occurs. Following open cut mining, pit lakes sometimes form, leading to permanent drawdown of the surrounding groundwater levels. Current methodology provides little guidance on how to account for long-term hydrological and water quality impacts that occur after mine closure, when assessing the water footprint of a mined product.

Addressing these types of methodological issues will enable competing mineral processing technologies, individual mines and commodities to be fairly and consistently benchmarked against each other on the basis of their impact to water resources. Key areas that need to be improved for future water footprint estimates of mined commodities include: the spatial resolution of water consumption and availability data, understanding how to model and incorporate long-term changes in hydrology and water quality, and developing consistent geographical and temporal boundaries of assessments.

KEYWORDS: mining, water footprint, water scarcity, water stress, water deprivation potential

1 Introduction

Recently there has been significant research focus placed upon the best way to account for water use impacts during environmental assessments of products and processes. Concepts such as the 'virtual' or 'embodied' water required to produce a product are being extended using life cycle assessment based methodology to provide more sophisticated estimates of environmental impacts associated with water use. From this a variety of approaches have arisen to produce stand-alone 'water footprints' of products and services, in a way that is analogous to a carbon footprint.

Despite the large environmental impacts associated with the mining industry, there have been relatively few attempts to quantify water related impacts from the industry using these methods.

Within Australia, CSIRO has developed estimates of the embodied water use (including supply chain water use) for various metal commodities and production technologies [1] [2]. More recently CSIRO has begun to consider the use of impact assessment methods that account for relative differences in regional water scarcity and stress [3]. Monash University has also conducted a range of related assessments of 'water use intensity' (excludes supply chain water use) using corporate sustainability reporting data [4] [5].

Internationally, we are aware of only several other groups that are involved in quantifying the water footprint associated with mined products. Notably studies have been conducted for several mines and mineral processing operations in Chile [6] and South Africa [7] [8].

2 Water Footprinting Methodology

An international standard, "ISO 14046:2014 Environmental Management - Water Footprint - Principles, requirements and guidelines" [9], has recently been developed to provide more consistency to the way that assessments of water footprints are conducted and presented. The approach advocated by the water footprinting standard is similar to the related standard for life cycle assessment, ISO 14044 [10], in that it describes four distinct phases of a water footprint assessment. These are:

1. Goal and Scope Definition
2. Water Footprint Inventory Analysis
3. Water Footprint Impact Assessment
4. Results interpretation.

The ISO standard emphasises the need to take a life cycle perspective when quantifying a water footprint. A key aspect of this approach is that estimating water use on a purely volumetric basis is insufficient to improve water management outcomes. Rather decision making should be based on fair and consistent estimates of the impacts that occur as a result of water use. At a simplistic level, water use impacts can be categorised into those associated with the physical consumption of water and those associated with the degradation of water quality.

2.1 Consumptive Water Use Impacts

Consumptive water use impacts are those that are related to changes in the volume of water in a catchment that is available for use by different end-users. The impacts associated with consuming a given volume of water will be very different depending upon whether water in a region is very scarce or highly available. Due to this there has been a variety of indices proposed to account for the relative water scarcity or stress of different regions. Several of these indices are shown in Figure 1. Each of these indices is based upon a different perspective of water use:

The Water Stress Index (WSI) [11] provides a measure of competition for water resources in an area. If there is only limited withdrawals of water in an area (e.g. central Australia), then the WSI will be low despite the relatively low physical availability of water.

The Water Depletion Index (WDI) [12] provides an indication of the risk that consumption of water will reduce the long-term availability of water in an area.

The Water Deprivation Potential (WDP) [13] provides an indication of the potential of water consumption to deprive other users of water. This index is Water Use in LCA (WULCA) working group's preliminary recommendation for quantifying water scarcity footprints.

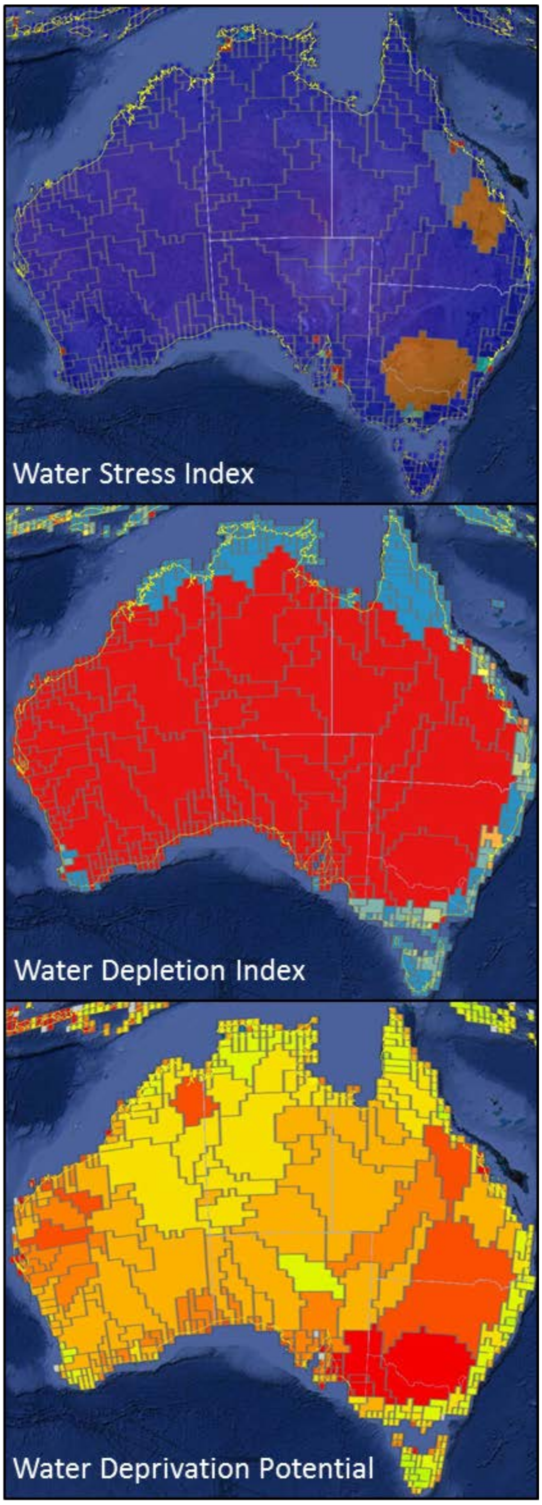

Figure 1: Several indices are available to evaluate the relative impacts of water use occurring in different regions [11] [12] [13].

2.2 Degradative Water Use Impacts

Degradative water use impacts are those that arise from changes to water quality. The types of impacts that can occur are varied, but may include: aquatic acidification, eutrophication, eco-toxicity (freshwater or marine), human toxicity, thermal pollution, etc. Standardised approaches to quantifying these impacts are available through the use of life cycle assessment impact characterisation methods. However due to the site specific nature of these impacts, estimates usually involve large uncertainties that make interpretation of the results difficult, particularly when considering more than one impact category at once.

2.3 Combining Consumptive and Degradative Water Use Impacts

The use of the different indicators available to assess water use impacts may lead to conflicting recommendations on how to reduce water use impacts. For instance, should a process alteration that reduces water consumption be adopted if it leads to increased water quality degradation? In order to handle these types of questions, there have been several methods proposed that attempt to combine aspects of water consumption and water degradation impacts into a single indicator. These have been based upon the notion that water quality degradation is equivalent to water consumption as it can deprive end-users of water suitable for their purposes. An example of this approach is the Water Impact Index (WII) [14], which is described in equation (1) below.

$$WII = \sum_i W_i . Q_{W_i} . WSI_i - \sum_j R_j . Q_{R_j} . WSI_j \quad (1)$$

Where,

- W is the quantity of water withdrawn from water body *i.*
- R is the quantity of water returned to water body *j*.
- Q is a water quality index.
- WSI is the Water Stress Index of the water body.

Although single-indicator approaches lose valuable information of the type of water related impacts that may occur, their relative simplicity may lead to easier interpretation of results by decision makers.

3 Challenges for quantifying the water footprint of mined products

The development of water footprints of mined products is heavily dependent upon rigorously quantified estimates of the flows of water into and out of production processes, and the quality of water associated with these flows. The Minerals Council of Australia and the University of Queensland recently developed the ‘Water Accounting Framework for the Minerals Industry’ that provides a method for individual mining companies to consistently record and report water flow, quality and storage data for their individual operations [15]. Overtime the increased adoption of this framework should lead to improvements in the quality and availability of data that can be used in water footprint assessments. However, due to the types of interactions that mining has with local water resources, additional data may be required to develop rigorous water footprint estimates for mined products.

3.1 Temporal scales

The data that is available for mined products within process inventory databases generally assume ‘steady state’ conditions, where all the flows into and out of the process are for a fixed period of production. This data is suitable for providing estimates of the short-term, ‘instantaneous’ impacts associated with a mined product; however it may be not be suitable for estimating the true longer-term impacts.

Whereas the agricultural industry can be assumed to produce food products from a given location indefinitely (and so water related impacts will always occur at the same time as production), the mining industry is relatively transient in nature. The exploration, development, operation, closure and rehabilitation of an individual mine may take place over a period of just a decade or two. Unfortunately the impacts to water resources associated with a mine often occur long after a mine has ceased production, due to changes in topography, hydrology and the mobilisation of pollutants. Therefore it may make sense to incorporate these long-term impacts into the water footprint estimate of a mine’s product.

3.2 Long-term impacts

There are many different types of long-term hydrological and water quality impacts that can occur from mining and mineral processing operations. The types of impacts that will occur from an operation are highly site specific and depend on a variety of factors, such as: local climate, site topography, groundwater levels, mine type and depth, soil and waste rock chemistry, and the overall success of site rehabilitation measures.

As an example, an open cut mine could have a range of different impacts upon groundwater in an area. When the mine intersects an aquifer, a pit lake may form and the evaporation from this could lead to permanent drawdown of groundwater levels. The pit lake would likely also accumulate salt due to this process and become hypersaline overtime. However, if the mine was located in a region of high rainfall, then the mine could act as a recharge zone and permanently increase surrounding groundwater levels.

Given the range of long-term impacts that can occur, further methodology development is required to provide guidance on how to account for these impacts fairly and consistently between individual mine sites.

Conclusions

Improvements to society's interactions with water resources are essential if we are to meet the challenges of the 21st century. Recent developments in water footprinting methodology provide new opportunities to quantify and reduce the impacts associated with individual products, services and processing technologies. The use of these methods enable us consistently track the progress of our process improvements, identify more efficient ways to source materials and reduce our overall impact on the environment.

Acknowledgements

The authors would like to thank the financial support of CSIRO Mineral Resources Flagship's collaboration fund. Discussions with Mark Cooksey, Roy Lovel and Terry Norgate have also greatly informed our thinking.

References

1. T.E. Norgate, R. Lovel, 2004, "Water use in metal production: a life cycle perspective", Commonwealth Scientific and Industrial Research Organisation (CSIRO), Report no. DMR2505.

2. S. Northey, N. Haque, 2013, "Life cycle based water footprints of selected mineral and metal processing routes", *Proceedings of Water in Mining 2013*, Brisbane, Australia, Nov 26-28, pp. 35-50.

3. S.A. Northey, N. Haque, R. Lovel and M.A. Cooksey, 2014, "Evaluating the application of water footprint methods to primary metal production systems", *Minerals Engineering*, vol. 69, pp. 65-80.

4. G.M. Mudd, 2008, "Sustainability Reporting and Water Resources: a Preliminary Assessment of Embodied Water and Sustainable Mining", *Mine Water and the Environment*, vol. 27, no. 3, pp. 136-144.

5. S. Northey, N. Haque, G. Mudd, 2013, "Using sustainability reporting to assess the environmental footprint of copper mining", *Journal of Cleaner Production*, vol. 40, pp. 118-128.

6. C.A. Peña, M.A.J. Huijbregts, 2014, "The Blue Water Footprint of Primary Copper Production in Northern Chile", Journal of Industrial Ecology, vol. 18, no. 1, pp. 49-58.

7. N. Ranchod, C.M. Sheridan, N. Pint, K. Slatter, K.G. Harding, 2015, "Assessing the blue-water footprint of an opencast platinum mine in South Africa", *Water SA*, vol. 41, no. 2, pp. 287-293.

8. E.L. Haggard, C.M. Sheridan, K.G. Harding, 2015, "Quantification of water usage at a South African platinum processing plant", *Water SA*, vol. 41, no. 2, pp. 279-286.

9. "ISO 14046:2014. Environmental Management -- Water Footprint -- Principles, requirements and guidelines", *International Organisation for Standardization* (ISO), Geneva, Switzerland.

10. "ISO 14044:2006. Environmental Management -- Life cycle assessment -- Requirements and guidelines", *International Organisation for Standardization* (ISO), Geneva, Switzerland.

11. S. Pfister, A. Koehler, S. Hellweg, 2009, "Assessing the Environmental Impacts of Freshwater Consumption in LCA", *Environmental Science and Technology*, 2009, vol. 43, pp. 4098-4104.

12. M. Berger, R. van der Ent, S. Eisner, V. Bach, M. Finkbeiner, 2014, "Water Accounting and Vulnerability Evaluation (WAVE): Considering Atmospheric Evaporation Recycling and the Risk of Freshwater Depletion in Water Footprinting", *Environmental Science and Technology*, vol. 48, pp. 4521-4528.

13. "Consensual Method Development: Preliminary recommendation – AwaRe v1.0", Water Use in LCA (WULCA) working group, 2015.,

14. J.-B. Bayart, S. Worbe, J. Grimaud, E. Aoustin, 2014, "The Water Impact Index: a simplified single-indicator approach for water footprinting", *International Journal for Life Cycle Assessment*, vol. 19, no. 6, pp. 1336-1344.

15. Minerals Council of Australia (MCA), 2014, "Water Accounting Framework for the Minerals Industry, User Guide, Version 1.3 - January 2014".

Water-related Data Requirements for Improved Life Cycle Assessment of Mining, Mineral Processing and Tailings Management

S A Northey[1], N Haque[2] and G Mudd[3]

INTRODUCTION

Life cycle assessment (LCA) provides an internationally standardised framework for estimating the environmental impacts associated with products or services (International Organisation for Standardisation, 2006). A particular strength of LCA is the ability to consider indirect impacts that occur at different stages of a product's life (eg raw material acquisition, manufacture, use, disposal or recycling) and through its supply chain (Figure 1). A wide variety of impact categories are able to be considered by LCA, such as the potential contribution to global warming, ozone depletion, marine eco-toxicity or freshwater eutrophication.

CSIRO has a long history of using LCA to compare the environmental impacts associated with alternative mining, mineral processing and metal production technologies (Haque and Norgate, 2014; Norgate and Haque, 2012, 2013). These 'mine-to-metal' assessments have primarily focused upon estimating embodied energy and greenhouse gas impacts associated with these production processes; however, more recently there has been a refocus to also consider consumptive water-use impacts in more detail (Northey *et al*, 2014).

The reliability of impact estimates developed using LCA are dependent upon two main factors:

1. the accuracy and representativeness of industrial input and output flows contained in life cycle inventory databases
2. the accuracy of characterisation factors used to quantify impacts based upon the inventory data.

The technical sophistication and regionalisation of characterisation factors for water-use impacts have improved substantially over the past several years, particularly as a result of increased interest in the 'water footprint' of products due to increasing concerns over global water security. However, the overall representativeness of water-use data contained in the major life cycle inventory databases is at times poor, particularly for industries such as mining that have very diverse interactions with water resources.

Water use at mining operations varies considerably due to a range of factors such as: the local climate, the particular ore processing technologies utilised, the approach taken to tailings management, varying water quality needs and risks and the ability of local environments to support water withdrawals and discharges. Analysis of data contained

1. GAusIMM, Doctoral Candidate, Monash University, Clayton Vic 3800. Email: stephen.northey@monash.edu

2. MAusIMM, Senior Scientist and Team Leader, CSIRO, Clayton Vic 3168. Email: nawshad.haque@csiro.au

3. MAusIMM, Senior Lecturer, Monash University, Clayton Vic 3800. Email: gavin.mudd@monash.edu

within the corporate sustainability reporting of mining companies has highlighted the scale of this variability (Mudd, 2008; Northey, Haque and Mudd, 2013).

Mining operations are increasingly collecting and reporting water-use data according to the Water Accounting Framework for the Minerals Industry (WAFMI) (MCA, 2014). However there are some differences in the data developed using the WAFMI and the data required by LCA methods, particularly as it relates to water quality. Some LCA methods, such as the ReCiPe impact assessment method (Goedkoop *et al*, 2009), simply require estimation of total pollutant loads to water – something that is routinely reported by mining operations to national and regional pollutant inventories. More recent LCA methods require classification of an operation's water inputs and outputs into eight distinct water quality categories, which have been defined using thresholds for 136 water quality parameters (Boulay *et al*, 2011). The WAFMI defines three water quality categories (MCA, 2014) that could potentially provide simplified data to these LCA methods. However, the influence of differing water quality thresholds used by the WAFMI and LCA water quality categories needs to be carefully assessed.

LCA studies typically only include detailed data for the operational phase of mining (see Figure 1), with post-closure impacts being accounted for very coarsely or in many cases ignored entirely. A range of impacts can occur post-closure following the rehabilitation or abandonment of a mine site. Hypersaline pit-lakes may form and become a permanent source of drawdown for surrounding groundwater systems. Post-closure impacts also occur due to generation of acid and/or metalliferous drainage (AMD) from waste rock, tailings material and the walls of mine voids. The impacts of AMD on local water quality may be transient due to 'first flush' effects, or in other cases may actively occur over decades or centuries – representing a perpetual impact to surrounding ecosystems and communities. There are several case studies of post-closure impacts included within the inventory databases used by LCA practitioners to assess the impacts of the industry.

Improvements to the quality and availability of life cycle inventory data for mine sites represents a significant opportunity to increase the reliability of results generated using LCA. Further research is required to provide guidance on accounting for life-of-mine

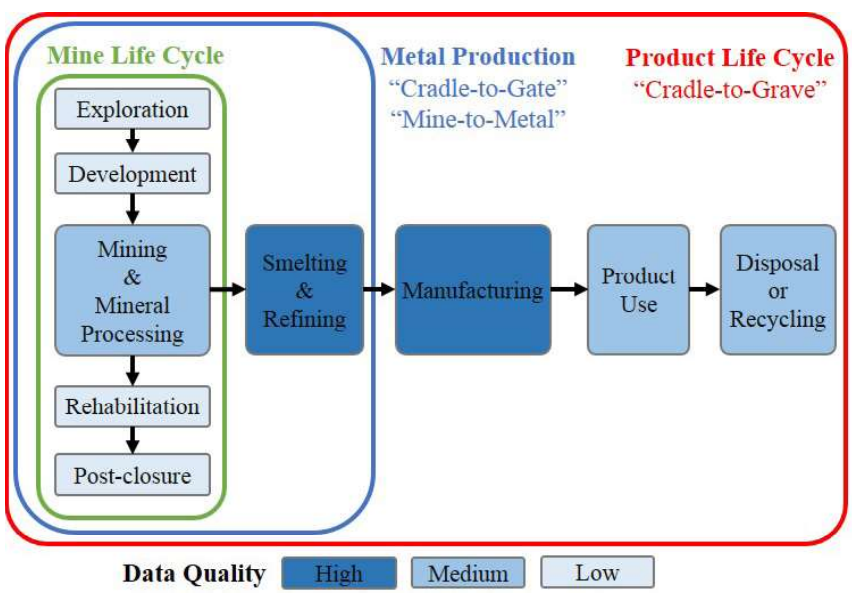

FIG 1 – Processes associated with a mine, metal production and a product. The perceived quality of data currently available for life cycle assessment studies is shown.

water consumption and quality impacts consistently within life cycle inventory data sets, as well as the most appropriate temporal and geographic boundaries of assessments. Addressing these issues will enable more sophisticated inventory data sets for mining to be developed, while also improving estimates of the embodied impacts of mined products (eg carbon footprint, water footprint etc).

ACKNOWLEDGEMENTS

The authors would like to thank CSIRO Mineral Resources for providing financial support for this work.

REFERENCES

Boulay, A-M, Bouchard, C, Deschênes, L and Margni, M, 2011. Categorising water for LCA inventory, *International Journal of Life Cycle Assessment*, 16:639–651.

Goedkoop, M J, Heijungs, R, Huijbregts, M, De Schryver, A, Strujis, J and van Zelm, R, 2009. *ReCiPe 2008: A Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the Endpoint Level, First Edition* (Ministry of Housing, Spatial Planning and the Environment: Netherlands).

Haque, N and Norgate, T, 2014. The greenhouse gas footprint of *in situ* leaching of uranium, gold and copper in Australia, *Journal of Cleaner Production*, 84:382–390.

International Organisation for Standardisation (ISO), 2006. ISO 14044:2006 – Environmental management – life cycle assessment – requirements and guidelines.

Minerals Council of Australia (MCA), 2014. Water accounting framework for the minerals industry, user guide, version 1.3 – January 2014.

Mudd, G M, 2008. Sustainability reporting and water resources: a preliminary assessment of embodied water and sustainable mining, *Mine Water and the Environment*, 27:136–144.

Norgate, T and Haque, N, 2012. Using life cycle assessment to evaluate some environmental impacts of gold production, *Journal of Cleaner Production*, 29–30:53–63.

Norgate, T and Haque, N, 2013. The greenhouse gas impact of IPCC and ore-sorting technologies, *Minerals Engineering*, 42:13–21.

Northey, S A, Haque, N, Lovel, R and Cooksey, M, 2014. Evaluating the application of water footprint methods to primary metal production systems, *Minerals Engineering*, 69:65–80.

Northey, S, Haque, N and Mudd, G, 2013. Using sustainability reporting to assess the environmental footprint of copper mining, *Journal of Cleaner Production*, 40:118–128.

Paper Number: 2731

Resource Depletion Scenarios – How should we address the limitations?

Northey, S.A.[1] and Mudd, G.M.[1]

[1]Environmental Engineering, Monash University, Clayton, VIC, Australia, stephen.northey@monash.edu

---

Society is dependent upon access to geologic resources to maintain our current economic system and societal wellbeing. However, the past several decades have seen a range of studies that have raised concerns that the long-term material needs of society may exceed what is potentially recoverable from presently known geologic resources.

A series of studies that explored the concept of 'peak minerals' were completed as part of the Mineral Futures Collaboration Cluster, a collaborative research programme between six Australian universities and the CSIRO that ran from 2009 to 2013. Scenarios were developed for primary copper [1], steel [2] and lithium [3] production, using the Geologic Resource Supply-Demand Model (GeRS-DeMo) developed by Mohr [4]. The models were calibrated using datasets of known mineral resources, historic mine production levels and estimates of future demand. The scenarios indicated that: copper production could grow for at least the next twenty years, with further exploration success required to sustain production levels beyond that [1]; growth in iron ore production may slow by the end of the decade, with production possibly plateauing until 2100 [2]; and, lithium resources appear sufficient to supply batteries for a high market penetration of electric vehicles [3]. Due to the exhaustion of available resources, all scenarios displayed a peak in total production at some stage in the next 20 to 100 years. Increases in assumed resources through exploration efforts would only delay these production peaks.

Using cumulative grade-tonnage curves combined with GeRS-DeMo results, the potential rate of ore grade decline was assessed for copper production [1]. From this, only two conclusions could be reached: 1) there is still considerable copper present in mineral deposits at grades above the current global average mined grade, and 2) the rate of decline of mined ore grades may slow as the average resource grade of large-scale porphyry copper systems is approached. Although global Cu grades are in terminal decline [6], this is the combination of major technology improvements (e.g. flotation, haul trucks), economies of scale, growing markets as well as geologic factors. For many Cu deposits, especially porphyry systems, as ore grades decline the total size of the deposit increases markedly, offsetting the decline in ore grade and leading to considerably more Cu available. Thus declining ore grades are not a sign of growing scarcity per se, but a reflection of the ongoing changes in the mining industry – which is often misunderstood in the sustainability literature. Overall, although Cu ore grades can be expected to decline very gradually, when applying 'peak minerals' thinking, it is clear that future constraints are environmental, social and economic in nature.

It is important to note that when metals are used they remain in anthropogenic stock – unlike coal, oil or gas, which are consumed in the process. Whilst recycling of these metals is attractive in concept, the ability to recycle a metal depends on its use. Metals used in pure form or alloys (e.g. iron, copper, gold) are relatively easy to recycle; whereas metals with diffuse or dissipative uses (e.g. nickel, rare earths) are comparatively harder to recycle. Overall, as more mining occurs, the total stock builds up in society and may eventually be comparable to the scale of remaining geologic resources. At present, GeRS-DeMo and peak models cannot reflect the complex interplay between a geologic resource, its mining and use, economics, environmental issues and social expectations such as recycling. Some thoughts on how best to address this complexity will be presented.

References:

[1] Northey S et al. (2014) Resources, Conservation & Recycling 83:190-201

[2] Mohr S et al. (2015) Natural Resources Research 24(3):317-327

[3] Mohr S et al. (2012) Minerals 2:65-84

[4] Mohr S (2010) PhD Thesis, University of Newcastle.

[5] Mudd G et al. (2013) Economic Geology 108(5):1163-1183