Journal of Cleaner Production 184 (2018) 788–797

Contents lists available at ScienceDirect

Journal of Cleaner Production

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

Production weighted water use impact characterisation factors for the global mining industry

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ARTICLE INFO

Article history:
Received 11 August 2017
Received in revised form
26 February 2018
Accepted 28 February 2018
Available online 1 March 2018

Keywords:
Impact characterisation
Mining industry
Mineral and metal production
Water stress index (WSI)
Available water remaining
Life cycle assessment (LCA)

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.

1. 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

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https://doi.org/10.1016/j.jclepro.2018.02.307

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.

2. Background and methods

2.1. 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.

2.1.1. 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.

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

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

Where: P_i 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.

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

2.1.2. 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.

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

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