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Macro-level economic and environmental sustainability of negative emission technologies; Case study of crushed silicate production for enhanced weathering

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ABSTRACT

Enhanced weathering (EW) involves application of crushed silicate rocks on croplands to capture CO_2. Although research on EW is gaining traction, the missing elements in the literature however are the supply chain sustainability impacts associated with large-scale production and deployment of crushed silicates for EW purposes. The need to conduct sustainability assessments for EW systems in addition to validated technical feasibility remains a relevant research gap. In this work, the potential economic and environmental impacts associated with production of crushed silicates is assessed for eight countries, belonging to two separate groups: emerging economies (Brazil, Russia, India, and China) and developed economies (USA, UK, France, and Germany).

A total of six economic and environmental impact categories are included in the assessment; gross domestic product (GDP), gross operating surplus (GOS), imports, greenhouse gas emissions, energy, and material use. The input-output model is used to estimate the economy-wide and macro-level sustainability impacts derived from producing crushed silicates. Findings show developed economies have relatively high levels of positive economic benefits and may experience less negative environmental impacts within their national boundaries by 'leaking' such impacts via imports. Imported consumption for crushed silicate production in developed countries were found to be substantially higher than that of emerging economies. For the emerging economies, imported consumption associated with crushed silicate production constitutes on average, less than 10% whereas for developed economies, imported consumption averages 20%. The UK mining and quarrying sector has the highest imported consumption at approximately 30%. The results of the study provide insightful outlook into the opportunities and challenges surrounding EW sustainability and is important in informing both national and global policy decisions regarding this technique.

1. Introduction

Under the Paris climate agreement, member countries agreed to work collectively toward limiting global warming temperature increase to 1.5 °C, encouraging effort that achieves meaningful positive impact on climate change mitigation (Rogelj et al., 2016). The agreement has motivated research into negative emission technologies (NETs) aimed at CO_2 removal (CDR), including bioenergy with carbon capture and storage (BECCS), direct air capture (DAC), and afforestation (AF) among others. One such NETs gaining significant traction in the literature is enhanced weathering (EW). EW involves the application of crushed silicate rocks such as basalt on croplands (Taylor et al., 2016; Kantola et al., 2017; Beerling et al., 2020).

Although research on rock weathering has an extensive history (Goldich, 1938), its potential and role as a climate change mitigation strategy only gained prominence a little over a decade ago (House, 2007). When rock comes into contact with carbonic acid (from atmospheric CO_2 dissolving in rainwater), the weathered by-products including carbon are washed by surface and groundwater into the ocean where it is trapped for years (Taylor et al., 2016). However, the natural process of weathering is slow. To make significant contributions to CDR, the process is facilitated through the geoengineering process of

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EW where silicate rocks are mined, crushed, and applied to land surfaces (Strefler et al., 2018). Current research suggests that the EW technique has great promise in contributing to climate change mitigation (Jia et al., 2022; Lehmann and Possinger, 2020)

In line with achieving the 1.5 °C target, the unintended sustainability impacts of large scale NETs deployment have been identified as an important research priority (Fuss et al., 2016). It is important that a sustainability assessment of NETs be carried out to ensure that any unintended impacts are addressed. For example, a study by Larkin et al. (2018) enumerates conditions including environmental externalities where the large scale deployment of NETs may fail at scale among the dominant global big emitters. According to the authors, the big emitters are made up of 25 nations including selected countries in this study (Brazil, Russia, India, China, USA, UK, France and Germany) and contribute to 80% of global CO_2 emissions from energy consumption and industrial activity. Based on scenario pathway analysis, it is reported that some NETs, particularly BECCS and carbon capture and storage (CCS) are challenged with scalability when environmental externalities from large scale of deployment are not addressed (Larkin et al., 2018; Smith et al., 2019). In the same way, it is expected that the industrial production of crushed silicates for use in EW will generate economic and environmental externalities that must be critically assessed (Smith et al., 2016a, 2016b; Lefebvre et al., 2019; Beerling et al., 2020; Garcia et al., 2020).

Ideally, an economic activity such as the industrial production of crushed silicates should not lead to increases in environmentally burden, a term referred to as ecological economic decoupling (Wang et al., 2016; Deutch, 2017; Noonan, 2020). The United Nations Environment Program (UNEP, 2011) considers ecological-economic decoupling as vital for a more sustainable form of development. Consequently, it is critical that the potential environmental impact associated with crushed basalt production for EW be assessed in order to put any economic gains from the same process into perspective.

However, there are limited sustainability studies on NETs and in particular, a quantitative sustainability evaluation of EW does not currently exist. In study by Fuss et al. (2016) although the authors acknowledged EW as a NETs, their review of research priority areas focuses primarily on BECCS and afforestation. A more comprehensive study by Smith et al. (2016a, 2016b) looked at greenhouse gas emissions, water and energy use, and land use impact of EW, BECCS, DAC and afforestation. A parallel study by Smith (2016) considered similar impacts for soil carbon sequestration and biochar. In a more recent study by Smith et al. (2019) the contribution of EW to specific Sustainable Development Goals (SDGs) are identified but the study goes no further to measure economic and environmental sustainability related impacts.

The paper focuses on the key processes in EW; mining and crushing of the silicate rocks. These key processes have been identified as energy intensive hotspots in the EW process (Lefebvre et al., 2019). However, beyond energy use, there are other economic and environmentally related impacts such as gross domestic product, greenhouse gas emissions and material use, that are equally critical to the sustainability of EW, and these are considered in the current paper. The aforementioned impacts may also extend to other economic sectors owing to the increased demand for crushed silicates rocks and industry interdependencies within an economy. In the current study, sustainability assessment is focused on quantifying potential economic and environmental impacts of EW from industrial scale production of crushed silicate rock. Specifically, the objective of study is to address research question pertaining to macro-level sustainability impacts associated with crushed silicate production for EW.

1.1. Macro-level sustainability assessment of EW

A review of EW literature shows that although some sustainability assessments have been conducted, focus is primarily on micro-level economic impact assessment with little attention to environmental impacts (Schuiling and Krijgsman, 2006; Hartmann et al., 2013; Smith et al., 2016a, 2016b; Strefler et al., 2018). It is clear that the common approach used for assessment is targeted at the micro- and firm-level such that no indirect impacts from other sectors could be captured with large-scale production of crushed silicates. Although micro level assessments are valuable, it is also critical that macro-level assessments are carried out to inform policy decisions regarding EW implementation.

Economic and environmental sustainability studies on EW are deficient at the macro-level and models are not well determined. For instance, in the study by Strefler et al. (2018) the authors perform a techno-economic assessment using selected economic assessment reports of open-pit mines to assess investment and operation costs from mining and grinding rocks. A recent study by Beerling et al. (2020) also performs a techno-economic forecast of the CO2 removal potential and cost for a number of countries. Another study (Lefebvre et al., 2019) utilizes the lifecycle assessment method (LCA) to estimate seven potential environmental impacts of Enhanced rock weathering with crushed silicate rock. The Lefebvre et al. (2019) study is based on a micro-level process based LCA and therefore is subject to the system boundary problem (Brentrup et al., 2004; Heijungs et al., 2010).

The system boundary challenge arises from the difficult nature of capturing all processes and inputs required to make a product. The resulting boundary defines the constraints on assessing sustainability impacts. The problem with the process based approach is that upstream sustainability impacts may extend beyond the firm level and several suppliers in the extended supply chain may not be captured (Huang et al., 2009; Hayami et al., 2015). Capturing upstream process impacts can help identify supply chain hotspots and assist decision-makers in creating holistic and comprehensive policy. Targeted and informed policy can ensure that solutions are not created by a mere shift of environmental burden to less visible portions in the supply chain (Acquaye et al., 2011). A macro-level sustainability assessment of technologies is implemented to address this challenge.

The need for macro-level analysis of EW stems from the fact that, for any type of production, firms rely on inputs from other firms that may be from different sectors. For instance, to produce the crushed silicates, mining and quarrying (M&Q) firms may require inputs, e.g., machinery, tools, and legal services, from other sectors within an economy. It is important to see what the potential economic and environmental impacts are and how these are spread across sectors in an economy. This is particularly important as the production of crushed silicates mostly takes place in the mining and quarrying sector also requires inputs from other sectors within an economy. Subsequently to enable complete assessment, it is expected that these feedbacks or spillovers are captured in the analysis. Within the mining and quarry (M&Q) sector, the use of direct inputs such as blast explosives for mining and energy for grinding rocks is expected, but it is not reasonable to include and analyse all the direct inputs. In addition, very little is known about indirect inputs and processes beyond a single quarrying firm that produces crushed silicate rocks. Indirect inputs may be created at different supply chain tier levels, creating additional complexity for estimating sustainability impacts at all the tiers, commonly referred to as upstream impacts (Acquaye et al., 2011; Bode and Wagner, 2015).

Upstream impacts may also come from imported products, services, and materials. This implies any environmental damage associated with the imported inputs will be borne by the exporting countries, i.e., the point of origin where production takes place (Oppon et al., 2018). The introduction of the trade dimension in impact assessment acknowledges that for large-scale production, e.g., crushed silicates for EW, impacts may extend beyond national political boundaries. When trade dimension is excluded from economic and environmental impact analysis of supply chains, the extent of sustainability impact remains unknown (Ghertner and Fripp, 2007). Therefore, it is important to quantify the magnitude of these embodied environmental impacts in trade.

An effective way to quantify embodied impacts in supply chains is to model using economic trade data. The economic models that capture trade interactions are commonly referred to as input-output (I-O) developed by Wassily Leontief (1986). I-O models have been used to identify both economic and environmental sustainability impacts in several studies (Gallego and Lenzen, 2005; Wiedmann et al., 2007). The I-O methodology is used in the current study to estimate environmental impacts among the selected countries. The goal of this paper is to introduce macro-level sustainability assessment for the production of crushed silicate rock used in EW.

1.2. Country level sustainability assessment of EW

The implementation of EW will differ from country to country and therefore there is a need for research that provides insights into how countries are impacted economically. When establishing measures to mitigate greenhouse gas emissions at national and even sub-national scales in line with the Paris Agreement, policy makers are informed, either directly or indirectly, by analysis derived from academic research. Therefore it is essential that such analysis evolve from a diverse range of inputs and relationships as well as capture differing national circumstances (Larkin et al., 2018). While countries may differ, there are some similarities that allow for classification. For instance, EW in a developed economy such as United Kingdom or Germany may differ from an emerging economy such as Brazil or China. For this reason, we focus analysis by selecting countries from these two groups of countries that is emerging economies and developed economies.

Overall, eight countries are selected: four from emerging economies (Brazil, Russia, India, and China-BRIC countries), and four from the developed economies (USA, UK, France, and Germany). The rationale for choosing these countries is twofold. First, they are identified as top emitters of GHGs globally (Nejat et al., 2015). Second, there is evidence to suggest that economic growth in these countries have been largely associated with growth in global emissions (Fankhauser and Tol, 2005; Tamazian et al., 2009; Pao and Tsai, 2010; Knight and Schor, 2014). These eight countries are expected to contribute immensely to the fight against climate change by adopting climate change mitigation strategies that may include EW. Wide scale implementation of EW like other climate change mitigation efforts, will have to be reflective of national circumstances (Winkler et al., 2006). Therefore, selecting countries from both emerging and developed economies allows for effective climate policy formulations and decisions for countries with similar national circumstances.

The current study provides a macro analysis to inform country policies on large scale production of crushed silicate rock for the EW carbon capture technology. The dynamics of each mining and quarrying sector differ from country to country and it is anticipated that economic and environmental sustainability concerns from production of crushed silicate will also differ. Assessing the sustainability performance of EW must therefore incorporate these differences to enable a better understanding of economic performance when EW is rolled out on a large scale (Fuss et al., 2016).

Overall, six impact categories are included in the assessment, namely, gross domestic product (GDP), gross operating surplus (GOS), imports, (for the economic impacts) and greenhouse gas emissions, energy, and material use (for the environmental impacts). There are wide range of environmental and economic indicators that can be used in impact assessment but generally these closely align with sustainable development goals (SDGs). The selection of indicators for the study therefore took into account impact categories that reflect SDGS as encouraged in studies carrying out sustainability assessments (Wulf et al., 2018). Specifically, the six indicators selected for inclusion in the study reflects SDG 8 Decent work and economic growth, SDG 12 Responsible consumption and production and SDG 13 Climate action. In addition, the selection of indicators for sustainability assessments must also reflect relevant impacts to the study that are also closely associated with the product's sector (Monteiro et al., 2019). Greenhouse gas emissions, energy use and material use were selected as environmental indicators as these impacts are usually associated with mining and quarrying sector where the production of silicates takes place (Norgate and Haque, 2010; Fugiel et al., 2017; Yang et al., 2022). Economic indicators GDP, GOS and imports were included to highlight the potential link between economic growth and environmental damages. By including these impacts in a national account framework (input-output model), the study aims to show if "*it is possible to decouple economic growth-the production of goods and services- from some aspects of environmental degradation as envisioned by SDG 8*" (Alexander et al., 2018).

The following sections of the paper will describe the national production of crushed silicate rock and associated economic and environmental impacts. First, a detailed discussion on the method and data source is presented. This is followed by presentation and analysis of various results on the economic and environmental impacts associated with crushed silicate production in the selected emerging and developed economies. Finally, suggested future and concluding statements are offered.

2. Method and data

2.1. Input-output (I-O) framework

To capture the complexities of the production and consumption activities of industrial supply chains and related economy-wide impacts (Gallego and Lenzen, 2005; Hayami et al., 2015; Camanzi et al., 2017), the research methodology employed must encapsulate such a framework. For the production of crushed silicates, impacts extend beyond the firm level (e.g., a single quarry business) across the economy due to production interdependencies of firms. When a bottom-up, firm-level data collection approach is used exclusively, very few companies in a supply chain can be fully assessed due to the cut off or boundary problem (Swarr et al., 2011). Also, collecting firm level data throughout a supply chain is a time and cost-prohibitive endeavour (Rebitzer and Hunkeler, 2003). The bottom-up approach offers great detail and in-depth view of impacts at the firm level with available data. However, where the aim of this study is to provide insight into economy-wide impacts from production (mining and crushing) of crushed silicate rocks for EW, the use of a bottom-up approach is inappropriate.

The preferred top-down approach utilizes the input-output (I-O) method, making it possible to capture impacts from extended and interconnected supply chains (Richardson, 1985: Leontief, 1986). The principle of I-O analysis was developed by the Nobel laureate economist Wassily Leontief (Leontief, 1986). The framework is structured on the economic flow of resources (products and services) recorded as monetary transaction usually in US dollars or other national currencies depending on the source of data. The I-O model is centered on the idea of inter-industry transactions. In various studies (Hauknes and Knell, 2009; Guo and Murphy, 2012; Chen et al., 2017), inter-industry transactions are used inter-changeably with inter-sectoral transaction and the same lexicon is inferred in the current study.

Industries use products of other industries to create their own products (McNerney et al., 2013). For example, the mining and quarry industry utilizes fabricated metal products, machinery and equipment, electricity, gas, etc., to produce primary aggregates including crushed silicate rocks. Outputs from one industry therefore become inputs to another. The implication is that when crushed silicates are produced, the demand for metal products, electricity, gas, machinery, etc., are affected. These inter-connections are described in I-O tables, typically compiled using national accounting data compiled by national statistical agencies (McNerney et al., 2013). In the I-O table this inter-industry relationship is known as Intermediate consumption (Z). Other parts of the I-O table show the Final demand (Y) of commodities by households, governments, investment, or exports and the Total Output (X) of a sector.

The relationship between Intermediate consumption (Z), Final demand (Y) and Total output (X) is given by:

$$\mathbf{Z} + \mathbf{Y} = \mathbf{X} \tag{1}$$

Since industries purchase from other industries to produce their own goods / services, the input-output table is therefore used to determine these indirect deliveries (from one industry to another) by deriving a *technology matrix (A), also known as a matrix of direct requirement.* It is the requirement from each of the economic sector needed to produce a unit output.

Technology Matrix (A), is given by:

$$\mathbf{A} = \frac{\textbf{Intermediate consumption}}{\textbf{Total output}} = \frac{\mathbf{Z}}{\mathbf{X}} \tag{2}$$

We know from Eq. (1), that

$$\mathbf{Z} + \mathbf{Y} = \mathbf{X}$$

And from Eq. (2),

$$\mathbf{A} = \frac{\textbf{Intermediate consumption}}{\textbf{Total output}} = \frac{\mathbf{Z}}{\mathbf{X}}$$

$$\therefore \mathrm{Z} = \mathrm{AX}$$

Then $\mathrm{AX} + \mathrm{Y} = \mathrm{X}$

So that $Y = X(1 - A)$.

But A is a matrix, therefore

$Y = X\,(I - A)$

Where I is an identity matrix. Hence

$$X = (\boldsymbol{I} - \boldsymbol{A})^{-1}\boldsymbol{Y} \tag{3}$$

$(\boldsymbol{I} - \boldsymbol{A})^{-1}$ in Eq. (3), is referred to as the **Leontief Inverse Matrix**, named after Wassily Leontief. The **Leontief Inverse Matrix** is the matrix of cumulative (direct and indirect) deliveries needed to produce a product per unit of total output. It is also known as total requirement matrix depicting the direct and indirect input required to produce a unit of output. Emphasis is now placed not just on what goes on within the firm (direct) but on a life cycle wide assessment that traces impact through the entire production and supply chain (upstream).

2.1.1. Extended input-output framework with sustainability indicators

Sustainability impacts of a product or industry have been captured using extended I-O analysis with economic and environmental sustainability indicators (Onat et al., 2014; Kucukvar et al., 2014) as well as social impact metrics (Richter et al., 2019). For this study, three indicators are selected for each economic and environmental impacts, totalling six sustainability indicators. Gross operating surplus (GOS), contribution to Gross domestic product (GDP), and import are selected as key economic indicators measured in millions of dollars ($M). For environmental impact, selected indicators are greenhouse gas (GHG) emissions, energy use, and material use measured in kg CO_2-eq (for GHG) and MJ (for energy and material use), respectively.

Similar indicators have been integrated with the IO framework across various studies (Foran et al., 2005; Acquaye et al., 2011; Egilmez et al., 2013; Kucukvar et al., 2014; Onat et al., 2014; Noori et al., 2015; Ibn-Mohammed et al., 2016) to provide a macro-level sustainability accounting framework using various country I-O tables. In study by Kucukvar et al. (2014), the I-O model is used to estimate GDP contributions of various sectors associated with the final consumption and investment within the US economy. In the study by Noori et al. (2015) the authors integrated economic measures including imports, business profits, and government tax to estimate economic impact of wind energy alternatives.

Carbon hotspots in the biodiesel supply chain were estimated using the I-O framework by Acquaye et al. (2011) and Ibn-Mohammed et al. (2016). Both studies used I-O to estimate the environmental profile of lead based on multiple indicators including energy use, material use, and GHGs. When the I-O framework is extended with environmental outputs, the process is referred to as environmentally extended input-output (EEIO) and has been used extensively for macro-level environmental impact assessment (Lave et al., 1995; Hendrickson et al., 1998; Wiedmann et al., 2007; Kitzes, 2013; Ahi and Searcy, 2015; Yang et al., 2017; Liu et al., 2018). Use of the EEIO model for sustainability assessments assumes that all direct and indirect economic impacts associated with a product supply chain are captured, therefore eliminating the cut-off or system boundary problem associated with firm level micro assessment. I-O data used in the EEIO is available from myriad sources with varying accessibility.

Several I-O databases provide country and sector level data with economic and environmental indicators. These include Global Trade Analysis Project (GTAP), EORA, Exiobase, and World Input Output database (WIOD), among others. Data for the economic and environmental indicators were accessed using the WIOD: the WIOD is an I-O databases that is extensively applied in the literature (Timmer et al., 2012; Ibn-Mohammed et al., 2016; Chen et al., 2017; Lu, 2017). Specifically used in this work are the 2011 single country I-O tables from the 2013 WIOD version that contain series data on 27 EU countries in addition to 13 other major economies. These tables are retrieved from the WIOD environmental satellite account.

Following on from the construction of the Leontief inverse matrix (Eq. (3)), the economic and environmental sustainability indicators (GOS, GDP, Imports, GHG, energy and material use) are introduced into the I-O framework by constructing the ‘direct intensity matrix’ (DiM) which measures sectoral intensities per unit of output produced. The DiM is calculated by dividing the sectoral output of an economic or environmental indicator or metric by the total output of that sector which can then be interpreted for example as the GDP per dollar of output or per kg of output produced. Next, the total intensity matrix (TIM) is constructed as a product of the direct intensity matrix and Leontief matrix refers to the multiplier matrix.

$$\boldsymbol{Total\ Intensity\ Matrix}(\boldsymbol{TiM}) = \boldsymbol{DiM}(\boldsymbol{I} - \boldsymbol{A})^{-1} \tag{4}$$

Hence the lifecycle supply chain economic and environmental impact from a product (crushed silicates in this case) is given by:

$$\boldsymbol{Total\ Intensity\ Matrix} \times \boldsymbol{Final\ Demand} = \boldsymbol{DiM}(\boldsymbol{I} - \boldsymbol{A})^{-1}\boldsymbol{Y} \tag{5}$$

In Eq. (5), the use of the Leontief inverse matrix represented by $(\boldsymbol{I} - \boldsymbol{A})^{-1}$ in the analytical framework ensures complete supply chain visibility of all economic activities as associated impacts.

2.1.2. Assumptions and limitations of the model

There are several assumptions underpinning the I-O model that also pose some inherent limitations in the application of this methodology, specifically for the extended economic and environmental assessment (Hendrickson et al., 1998; Acquaye and Duffy, 2010). One such limitation is the homogeneity hypothesis with an embedded aggregation challenge. The assumption posits that each industry uses identical inputs and processes to produce all the products classified in that industry, e.g., the mining industry uses the same machinery as the automobile industry. The assumption is a true generalization where reality is much more complicated. Each industry may be a represented by many different products or services. Even in the event of products being similar, there might be differences in production technology used.

Another limitation with I-O analysis is the proportionality assumption. Proportionality implies that a linear relationship exists between production inputs, outputs, and subsequent environmental impacts (Baral and Bakshi, 2010). For example, it suggests that industrial production processes utilize strictly fixed proportions of inputs and the ratios are consistent for any expansion or contraction of industrial activity. The assumption of proportionality does not invalidate the use of the I-O model however, since in some cases the linear proportionality gives a reasonable estimate even when non-linear relationship exits (Hendrickson et al., 1998). Tukker and Dietzenbacher (2013) also advocates for the use of the input-output framework especially when a lack of micro-level data available.

The I-O model is used to estimate production related impacts which falls within cradle-to gate system boundary. The crushed rocks need to be transported to application sites which would also generate some associated impacts. Our model excludes such analysis as this is beyond the scope of the research. However, recent research by Tan et al. (2022) and Eufrasio et al. (2022) complements this study as they model enhanced weathering using hypothetical and literature data respectively on different transport modes. Eufrasio et al. (2022), use bottom-up approach (process-based LCA) while in our study we use a top-down approach (Input-Output LCA). The type of lifecycle assessment (LCA) used is determined by the goal of the study. For micro-level impacts, process-based LCA is usually employed (Koornneed and Nieuwlaar, 2009; Lefebvre et al., 2019; Cooper et al., 2022), whereas to highlight macro-level impacts, input output LCA is used. We chose this modelling approach because we wanted to show both the direct and indirect production impacts associated with crushed silicate production as a result of the interconnections between sectors/industries within an economy.

3. Analysis and results

In this section, findings from I-O model application for each economic and environmental impact are presented and discussed. For each impact category, the diagrams shown relate to three key distinct findings explained below:

I. ***Total impact per kg crushed silicate:*** This is the economy-wide impact that is GDP, GOS, import, GHG emissions, energy and material use per kilogram of crushed silicate. The value comprises all impacts arising from the sectoral interactions required within the mining and quarry sectors of the selected countries. The relevance of this finding is that it gives the policy makers a more holistic view of the potential economic and environmental impacts associated with producing crushed silicates for EW and allows for comparison among the two groups of countries.

II. ***Direct and indirect impacts:*** The mining and quarry sector relies on direct inputs from other sectors and is accounted for by the matrix of direct requirements (also referred to as the Technology matrix). The production of direct inputs requires inputs not immediately related to mining and quarrying of crushed silicate and are referred to as indirect inputs. The economy-wide impacts (results from I above) are the total of direct and indirect inputs, and therefore reasonably inferred to result in ***direct impacts*** and ***indirect impacts***. The relevance of this finding is that it indicates whether the crushed crushed silicate production relies more heavily on direct inputs or indirect inputs. It shows where policy should be targeted in addressing sustainability impacts; that is whether at the first-tier supply chain where direct inputs are sourced or at the second-tier, third tier, etc., where indirect inputs are sourced.

III. ***Sectoral Contribution:*** The sectoral contribution to an economic or environmental impact identifies how much each sector contributes to the total impact associated with crushed silicate rock production (results from I above). The relevance of this finding is that it provides industry or sector level insight and identifies the relevant sectoral hotspots (the sectors with relatively high impact). To simplify analysis and comparison of the results, the 35 WIOD sectors are aggregated into nine sectors and a full list is available in the supplementary information.

3.1. Gross domestic product (GDP)

In this section, results are presented for the GDP potential per kg of crushed silicate ($ per kg) produced in both developed and emerging economies. Fig. 2a shows that among emerging economies, China has the highest GDP potential per kg produced that is ~$15 per kg followed by India with approximately $6/kg crushed silicate produced. Russia and Brazil have the lowest GDP potential per kg crushed silicate produced of $4.8/kg and $4.6/kg respectively. Among the developed economies, both USA and France have equally high GDP per kg crushed silicate produced of ~$10/kg followed by Germany with $9 per kg produced. The GDP potential per kg produced in the UK at $7.4/kg is the lowest among the developed countries. Overall, as expected, the results show that developed economies have relatively high GDP per kg crushed silicate produced compared to the emerging economies. (See Fig. 1.)

For the emerging economies, Fig. 2b shows that direct GDP generated per kg unit output in Brazil, Russia, and India are 58%, 60% and 70% respectively. These fractions are higher than indirect GDP except China where the GDP generated indirectly (approximately 58%) is higher than the direct impacts. For the developed economies, direct GDP per unit output for each USA (70%), UK (75%), and Germany (60%) are higher than the indirect GDP generated. France is shown with an equal split of 50% between direct and indirect GDP impacts.

Fig. 2c displays the sectoral contribution of Quarrying, Metals and Minerals to GDP in both the emerging economies and the developed economies is over 60% followed by the Services sector. The exception however is China where both the Machinery and Equipment and the Services sectors have the highest sectoral contributions to GDP at approximately 25% each. The Quarrying, Metals and Minerals in China contributes 10% to GDP.

3.2. Gross operating surplus (GOS)

In this section, results of GOS potential per kg of crushed silicate produced in both the developed and emerging economies are presented. Findings shown in Fig. 3a, indicate that among the emerging economies, China has the highest GOS potential of $6 per kg crushed silicate produced followed by India which has $4 per kg produced. Russia and Brazil have the lowest GOS potential of $2.8 and $2 per kg produced, respectively. Of the developed economies, the USA has the highest GOS potential of $6.5/kg followed by UK which has nearly $4.8 per kg produced. The GOS potential for France and Germany is similar at approximately $4/kg in $4.2/kg, respectively.

In Fig. 3b, results are presented that suggest for emerging economies, direct GOS impacts are generated directly especially in the case of Russia (~70%), India (~80%) and China (~55%). However, for Brazil, the GOS impacts occurring indirectly (~54%) are slightly higher than GOS impact directly occurring (~48%). For developed economies direct GOS impacts are higher in USA (~75%) and UK (80%). In France, there is an equal split of 50% between direct and indirect GOS impacts. For Germany, the direct GOS impacts of 58% are slightly higher than the indirect impacts per unit output (~42%).

Sectoral contribution to GOS is dominated by the Quarrying, Metals and Minerals sector across both emerging and developed economies followed by the Services sector (Fig. 3c). In China, there are some noticeable impact contributions from the Electricity, Gas and Water supply, Machinery and Equipment and the Fossil Fuels Mining sectors totalling up to 10%. In Brazil, the Services sector contribution to GOS, at approximately 25% is relatively higher compared to Russia, India, and China where sectoral contributions from their Services sector are approximately an average of 10%. In the developed economies, the Services sector in both France and Germany has higher sectoral contribution (approximately 25% to GOS) relative to USA and UK. Overall, the results show that sectors including Fossil Fuels Mining, Construction and Non-metallic, Electricity, Gas and Water supply have relatively minimal sectoral contribution to GOS.

3.3. Imports

In this section, results of imports per kg of crushed silicate produced in both the developed and emerging economies are presented. As seen in

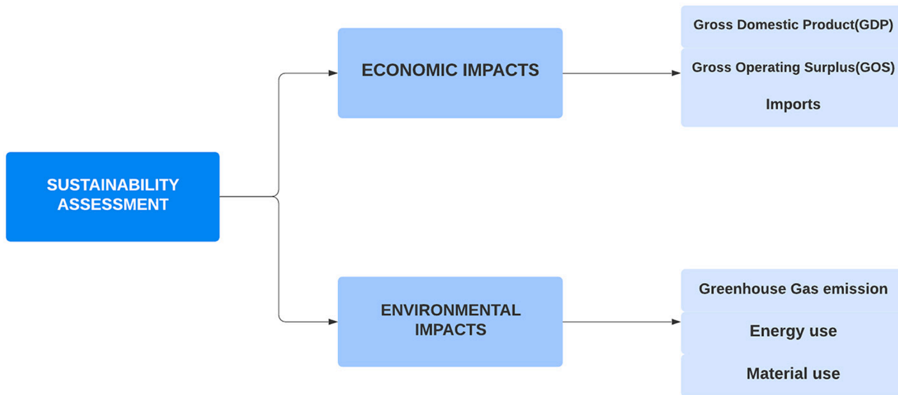

Fig. 1. Overview of economic and environmental impacts.

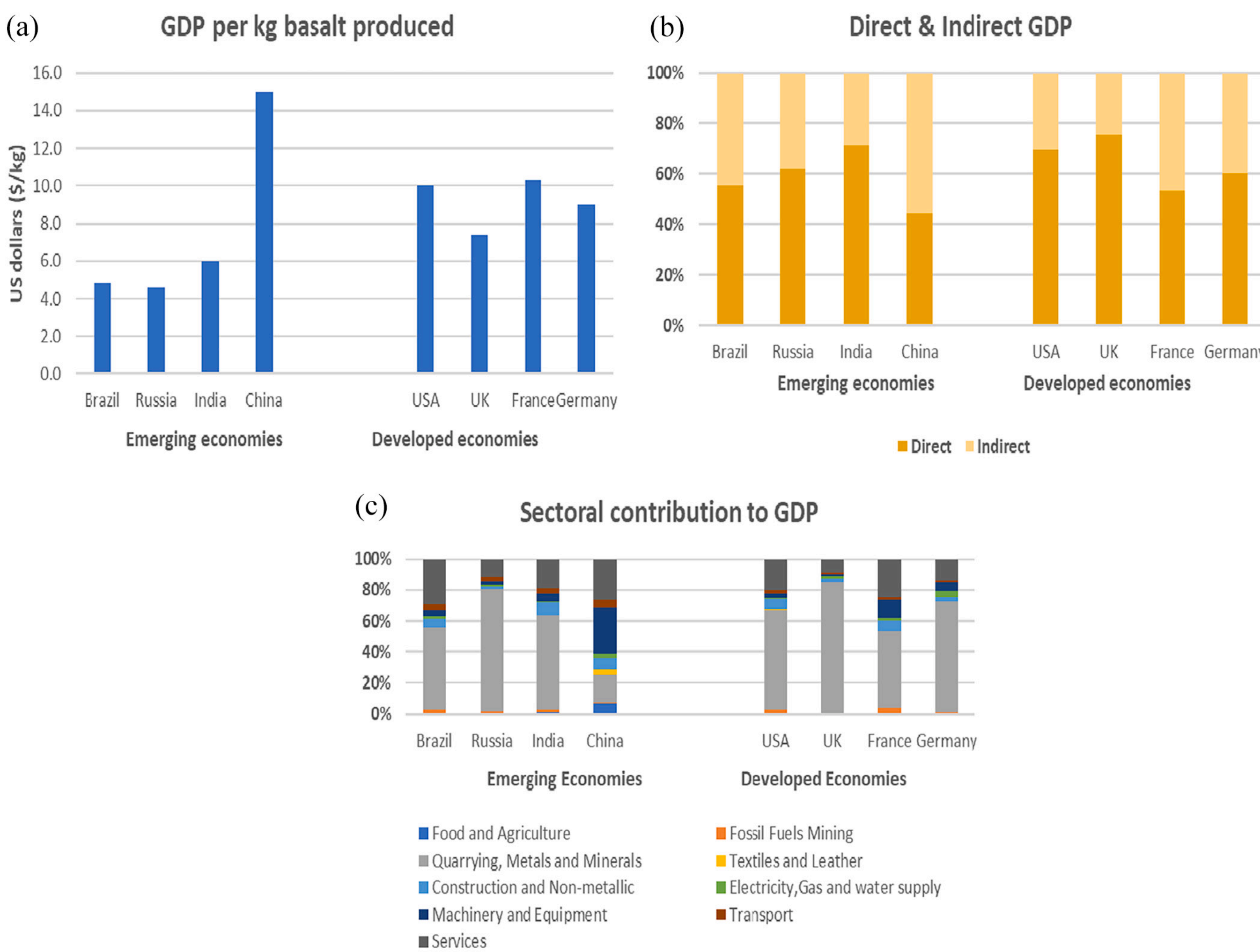

Fig. 2. a) GDP per kg crushed silicate produced, b) Direct & Indirect GDP, and c) Sectoral Contribution to GDP.

Fig. 4a, China and Brazil have the highest potential of approximately \$0.13/kg and \$0.12/kg, respectively compared to Russia and India that each have less than \$0.025/kg. The difference is explained by production of a kilogram of crushed silicate in Brazil and China requires more imported inputs than their counterparts in Russia and India. In the case of the developed economies, imports are significantly higher than the BRIC countries (emerging economies). The UK has the highest import per kg crushed silicate produced of approximately \$0.32/kg. Germany and USA also have high import levels at \$0.20/kg and \$0.16/kg, respectively. However, among the developed economies, France has the lowest imports suggesting that the production of crushed silicates relies more on domestic inputs than imported inputs.

In Fig. 4b, results show that for the emerging economies, specifically Brazil, India, and China, the proportion of direct imports is an average of 80%. Russia is very different however, where direct imports are 45% implying indirect imports are higher than direct imports. For all the

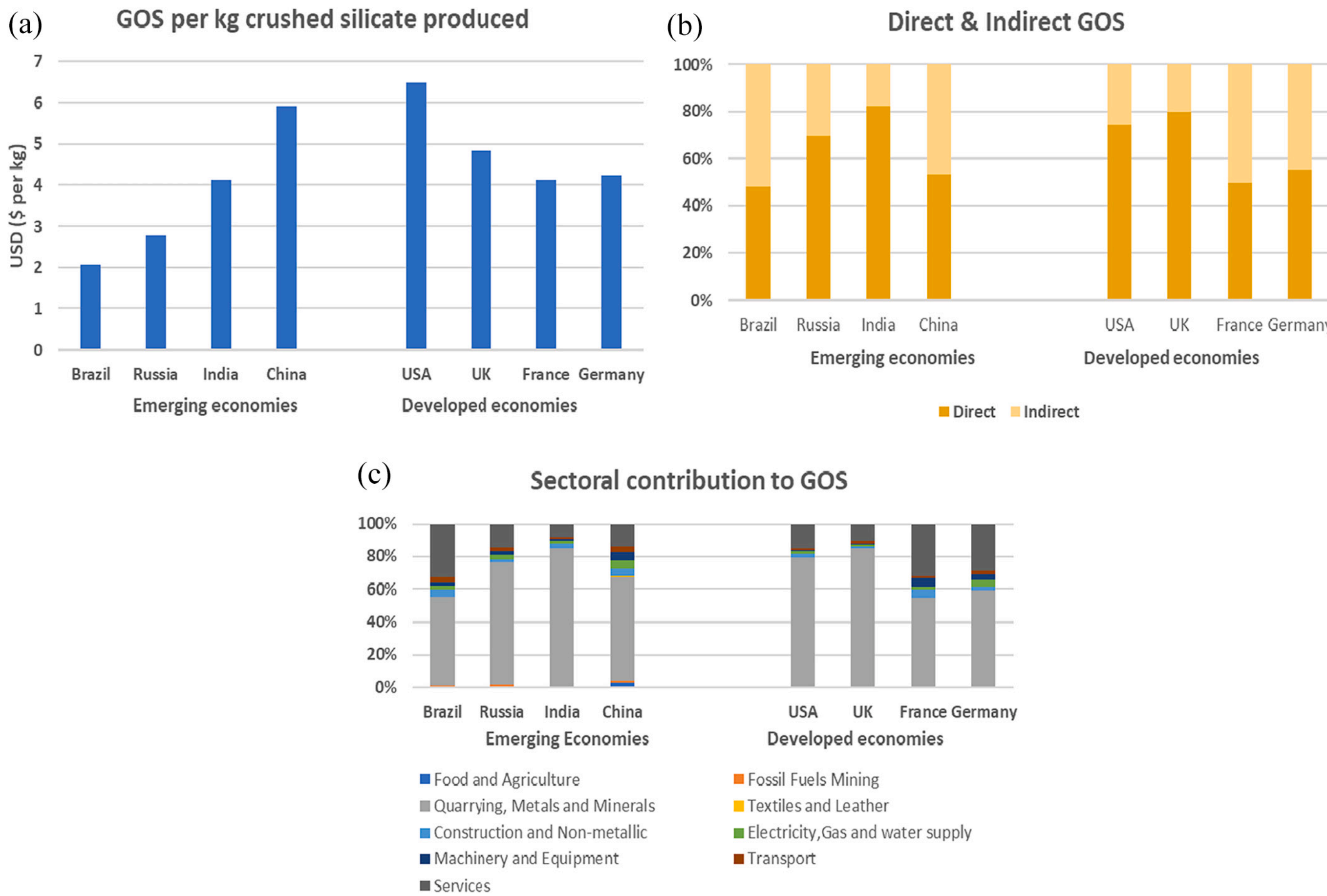

Fig. 3. a) GOS per kg crushed silicate produced, b) Direct & Indirect GOS, and c) Sectoral Contribution to GOS.

developed economies, the proportion of direct imports to indirect imports is an average of 95%.

In terms of sectoral contribution to imports (Fig. 4c), within the developed economies, a relatively significant proportion of the impact contribution is attributed to the Quarrying, Metal and Mineral sector. In the USA, UK, and Germany, an average of 98% contribution to imports comes from Quarrying, Metal and Mineral sector. However, in the case of France, sectoral contribution from the Quarrying, Metal and Mineral sector is slightly lower by comparison, at approximately 88%. For emerging economies, similar results are seen in that the highest sectoral contribution to imports is from the Quarrying, Metal and Mineral sector.

3.4. Greenhouse gas (GHG) emissions

In this section, we present results for energy use associated with crushed silicate production. Greenhouse gas emissions (GHG) among the emerging economies (Fig. 5a) shows that China has the highest impact of 530 kg CO_2-eq per kg crushed silicate produced followed by India and Russia with approximately 450 kg CO_2-eq and 350 kg CO_2-eq per kg produced respectively. Brazil has the lowest GHG potential of ~110 kg CO2-eq per kg produced. For the developed economies USA stands out as having the highest relative GHG of ~260 kg CO_2-eq/kg followed by Germany with a GHG potential of 100 kg CO_2-eq/kg. The GHG of producing crushed silicates in the UK and France are however very low; at less than 10 kg CO_2-eq/kg each.

Results in Fig. 5b shows that direct GHGs are higher than indirect GHGs for all countries in both emerging and developed economies. The results indicate that an average of 80% of GHG impact occurs directly across all four countries in emerging and developed economies each with an average of 20% accounted for as indirect GHG.

The sectoral impact of producing the crushed silicate is concentrated within the mining and quarrying sector in both emerging and developed economies (Fig. 5c). There are some few exceptions where relatively significant impact contributions are observed, e.g., the Services sector (9%) in Brazil, the Transport sector (3%) in Russia, and the Food and Agricultural sector (85%) in France.

3.5. Energy use

In this section, we present results for energy use associated with crushed silicate production. Low energy use is considered better than high energy use. Energy use in producing crushed silicates (Fig. 6a) is highest in India (~280 MJ/kg) and Russia (260 MJ/kg) with China also showing high values (~210 MJ/kg). Brazil is the lowest in terms of energy use among the emerging economies with nearly 70 MJ/kg. Among the developed economies Germany has the highest embedded energy use at 125 MJ/kg which is almost twice that of Brazil. However, generally the results indicate that energy used to produce crushed silicates in developed economies is significantly lower than the emerging economies. In particular, the UK has the lowest energy use of approximately 40 MJ per kg crushed silicate produced. USA and France show similar energy usage of 60 MJ and 50 MJ per kg crushed silicate produced, respectively.

For the emerging economies, excluding India, a large fraction of the impact per unit output from energy use occurs from indirect inputs (Fig. 6b). Indirect energy use in Brazil is approximately 58%, Russia 70%, and China 80%. The exception is India where about 75% of the energy use impacts occur directly within their mining and quarrying sector. In the developed economies, specifically USA, UK and Germany, direct energy use impacts are significantly higher than indirect energy

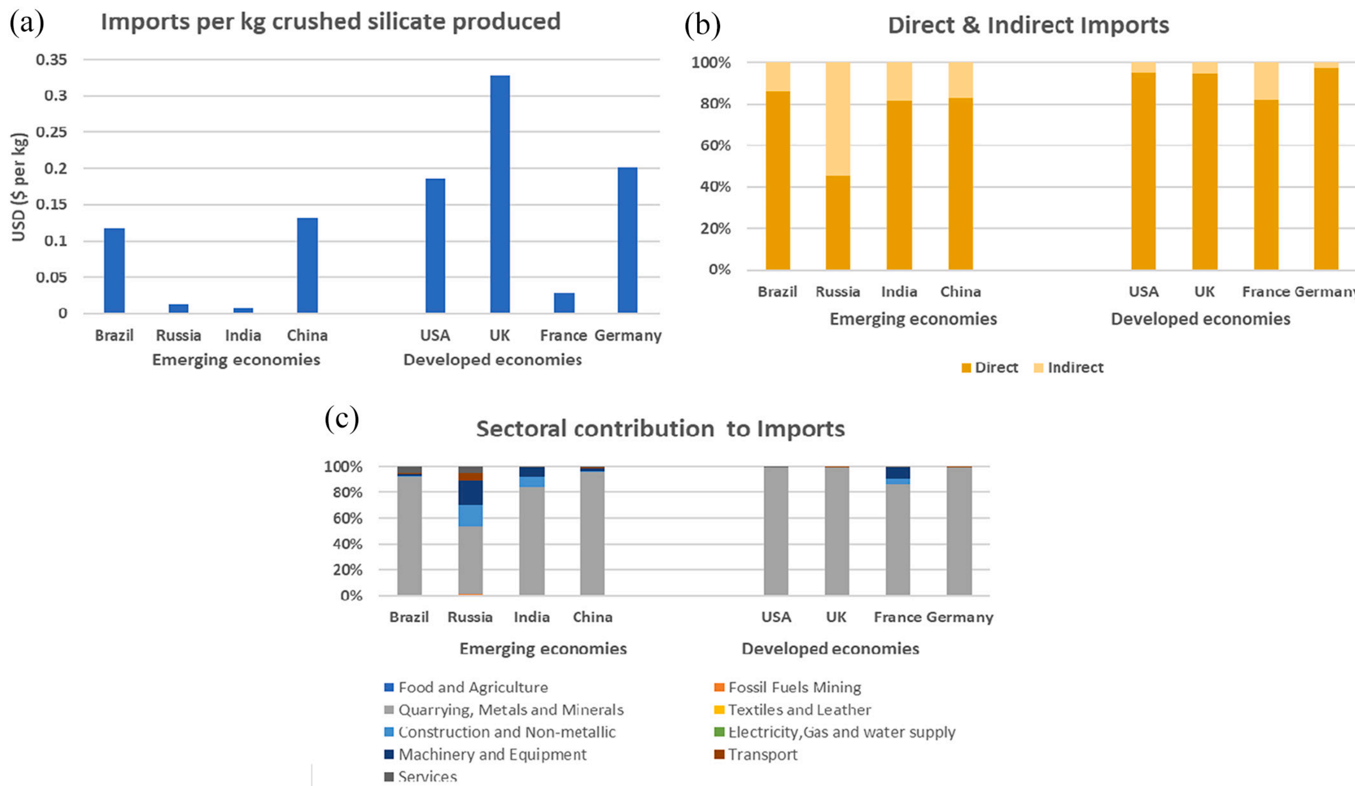

Fig. 4. a) Imports per kg crushed silicate produced, b) Direct & Indirect Imports, and c) Sectoral Contribution to Imports.

use impacts; 60% for USA and 78% for both the UK and Germany, respectively. The situation differs in France where 60% of the energy use impacts are from indirect inputs.

Sectoral distribution of total energy use required to produce crushed silicates is shown in Fig. 6c. In Brazil, the Quarrying, Metals and Minerals sector is the highest contributor to energy use at 60% followed by the Fossil Fuels extraction and mining sector accounting for 25% of total energy use. In Russia, the Quarrying, Metals and Minerals sector is the largest contributor to energy use with 40%, followed by the Electricity, Gas & Water supply sector accounting for about 35%. Energy use impacts in India are attributed to the Quarrying, Metals and Minerals sector, contributing 75%. The Electricity, Gas and Water supply sector in China is responsible for 45% of the total energy use impacts while the Quarrying, Metals and Minerals sector is the second highest with 33% contribution. Energy use in the Quarrying, Metals and Minerals for USA, UK and Germany contribute the highest to energy use impacts at 65%, 80%, and 78% respectively. These contributions are significantly higher compared to energy use impacts in France (45%). Common to all developed countries is the second highest contributor of energy use impacts, the Electricity, Gas and Water supply sector. France observes the highest contribution from this sector at approximately 30%, higher that USA, UK and Germany where the average is ~15%.

3.6. Material use

Total material use associated with production of per kg crushed silicates is shown in Fig. 7a. Among the emerging economies, material use in India is the highest with approximately 900 MJ per kg crushed silicate produced followed by China which requires 400 MJ/kg. However, material use in Germany is the highest of all countries (both emerging and developed) with potential material use of approximately 1200 MJ/kg. The UK has the lowest material use of ~40 MJ per kg crushed silicate produced. Material use in USA (250 MJ/kg) and France(350 MJ/kg) are both higher than Brazil (180MJkg) and Russia (200 MJ/kg).

Across most countries the direct material use is higher than indirect material use. The exception is China, where indirect material use impacts exceed the direct material use. Results in Fig. 7b show that direct material use is higher than indirect impacts in both groups, emerging and developed economies. In Brazil, Russia, and India, direct material use is 94%,95% and 99% respectively and all higher than China at 88%. For the developed countries, direct material use is highest in Germany (98%), second is France (96.5%), with the USA (96%) and the UK (95%) direct material use following closely behind.

Sectoral contribution to material use (Fig. 7c) is mainly accounted for by the Quarrying, Metals and Mineral sector across all the countries in both emerging and developed economies.

4. Discussion

The study provides insight into the economic and environmental sustainability of EW, and more specifically, the impacts from mining and quarrying of crushed silicates. From a sustainability perspective, it is suggested that the economies of developed countries stand to benefit in the large-scale crushed silicate production over emerging economies. In general, developed economies have high levels of positive economic benefits (e.g., GDP, GOS) resulting from the production of crushed silicates. Further, high employee compensation is also observed across supply chains in the developed economies. The results have shown also that most of the developed economies have relatively lower environmental impacts associated with the crushed silicate production compared to the emerging economies. On the other hand, for emerging economies, nearly the opposite relationship was true: environmental impacts were relatively high while fewer economic benefits resulted from the crushed silicate production.

One key observation is the share of impacts attributed to direct and indirect inputs associated with crushed silicate production. Understanding the direct to indirect disparity may be particularly helpful in informing policy makers on strategies to mitigate negative sustainability

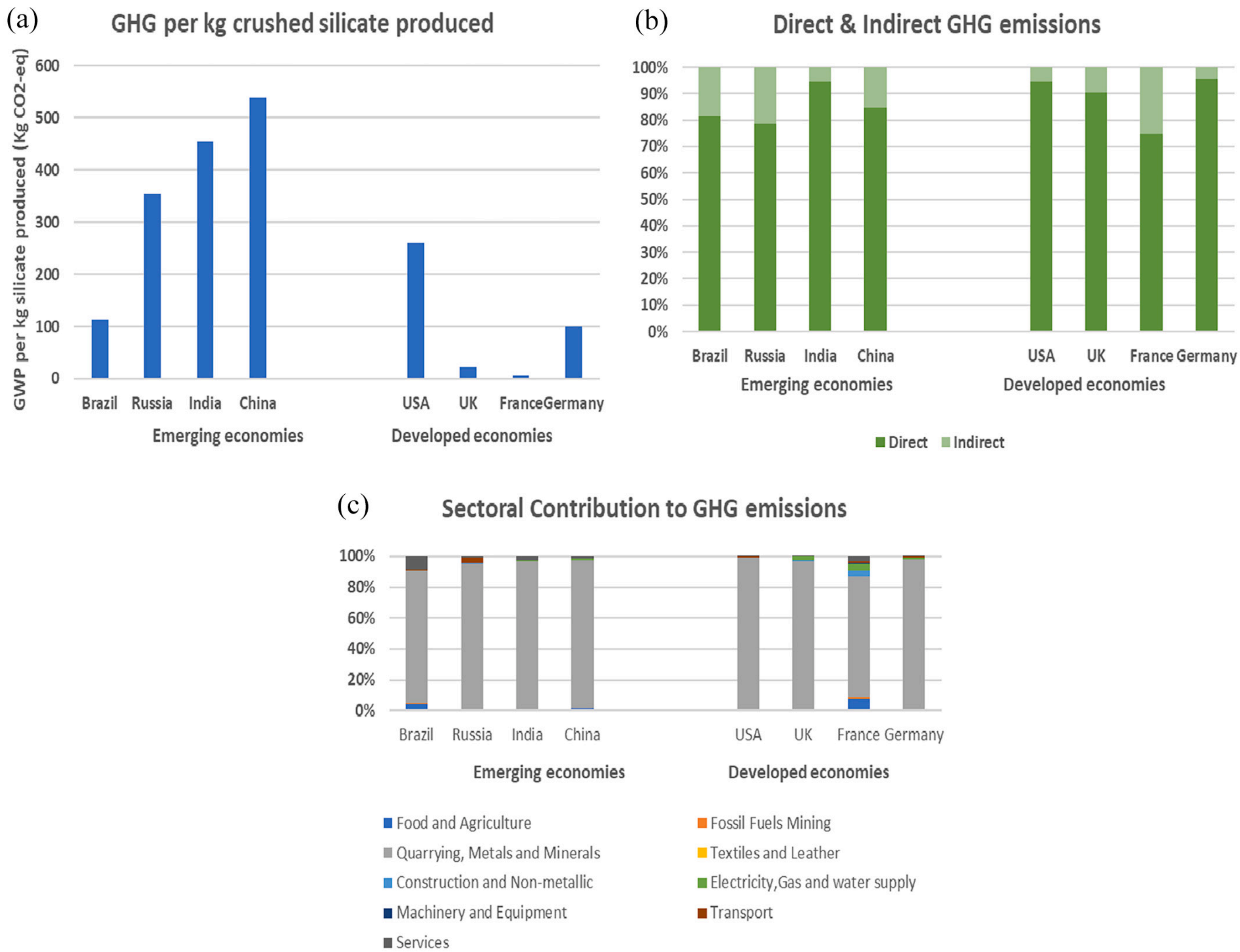

Fig. 5. a) Total Greenhouse gas emissions (GHG) per kg crushed silicate produced, b) Direct & Indirect GHG, and c) Sectoral Contribution of GHG impacts.

impacts. Across most impact categories, the direct inputs were responsible for a larger share of the total impacts. In some cases, impacts from indirect inputs were the highest contributor to total impacts. For instance, over 60% of energy use from crushed silicate production in Russia and China results from indirect inputs required for crushed silicate production.

Thus far, the Quarrying, Metals, and Minerals sector has the highest sectoral contribution to economic and sustainability impacts across the two groups of countries (emerging and developed economies). The Quarrying, Minerals and Mining (QMM) sector is comprised of two sectors; Mining and Quarrying (M&Q) sector and Basic Metals and Minerals sector (see sector aggregation in supplementary information). The highest contribution within the QMM sector comes from the M&Q sector. Therefore, an in-depth analysis of production patterns in the M&Q sector in particular would provide a richer contextual explanation to why potential economic and environmental impacts differ among the emerging and developed economies.

To go deeper, the intermediate consumption of each country input-output table must be considered. Intermediate consumption refers to sector-by-sector consumption of production inputs that take place within domestic boundaries (domestic consumption) or are imported (imported consumption). Total intermediate consumption is therefore the sum of domestic consumption and imported consumption. Fig. 8 shows the percentage share of intermediate consumption that is domestic versus imported from sectors of other countries. The results show that overall, domestic consumption is higher than imported consumption in the M&Q sector for all studied countries. However, imported consumption in developed countries is substantially higher than that of emerging economies. For the emerging economies, imported consumption constitutes on average, less than 10% compared to total consumption where in developed economies, imported consumption averages 20%. The UK mining and quarrying sector has the highest imported consumption at approximately 30%. The result supports the assertion that in some instances, developed economies experience less negative environmental impacts within their national boundaries by ‘leaking’ such impacts via imports that are produced in other countries (Oppon et al., 2018).

Further exploration into intermediate consumption and the M&Q sector can help determine how much a country relies on domestic inputs (within its own Mining and Quarry sector) and imported inputs (from the Mining and Quarry sector of other countries). In Fig. 9, the M&Q-to-M&Q intermediate consumption is presented. This is different from results in Fig. 5.6 which shows interactions between the Mining and Quarrying sector and all other sectors within and outside a country. For the emerging economies, apart from India, a large proportion of inputs for production are sourced domestically from the internal mining and quarrying sector. Only 30% (Brazil), 5% (Russia), and 20% (China) of production inputs are imported from the M&Q sector of other countries. For India however, the model suggests imports approaching 50% from other countries. In the developed economies, inputs for production

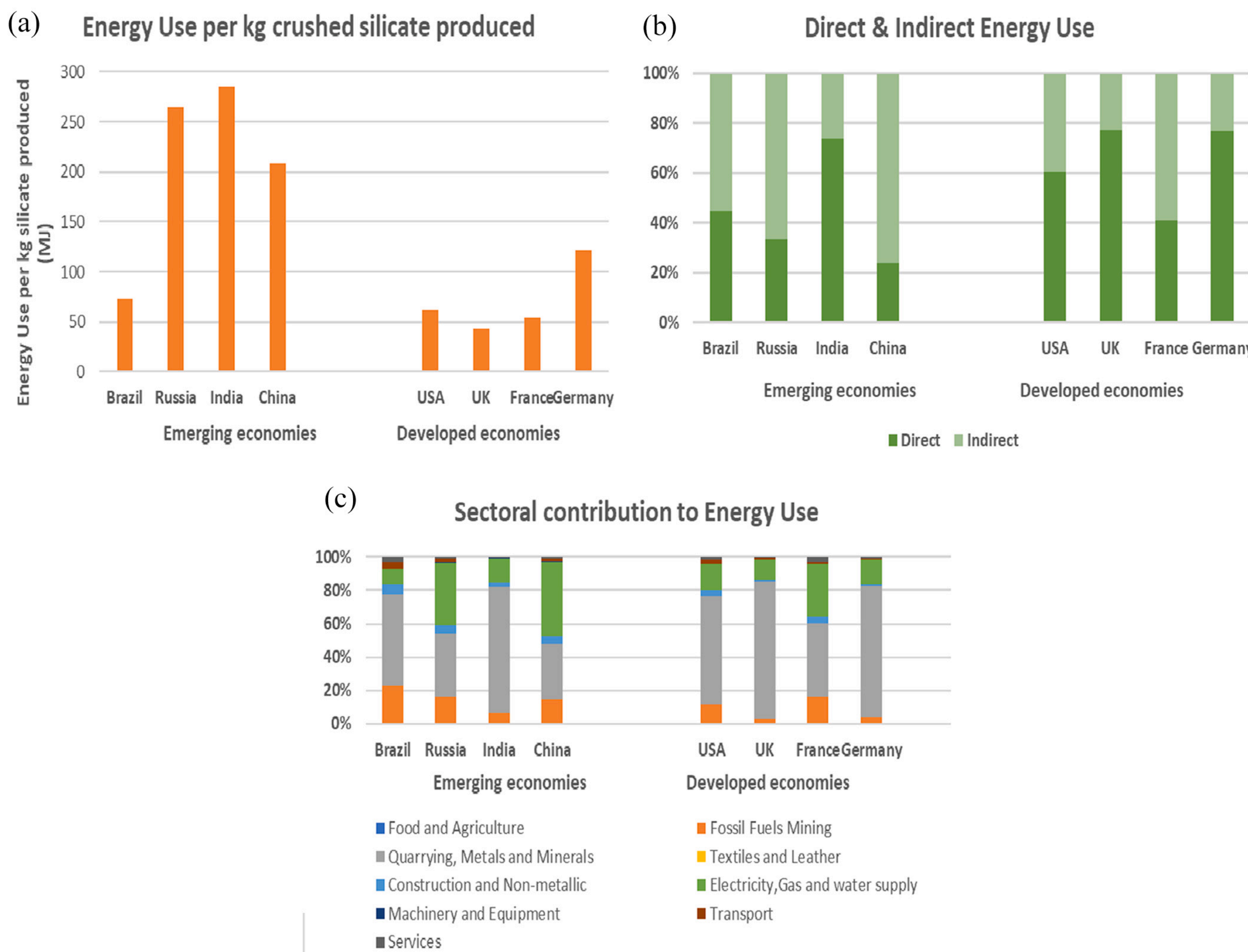

Fig. 6. a) Total energy use per kg crushed silicate produced, b) Direct & Indirect energy use, and c) Sectoral Contribution of energy use impacts.

within the M&Q sector are largely imported from M&Q sectors of other countries. For USA, UK, and Germany, approximately 40%, 50%, and 60% are imported respectively from the M&Q sectors of other countries. The exception is France where only 10% of inputs are imported from the M&Q sectors of other countries.

The implication of this study in relation to the large-scale production of crushed silicate for EW is that whereas developed countries will potentially benefit sustainability wise from crushed silicate production in terms of (GOS, GDP and employee compensation), emerging economies that rely more on indigenous production do not benefit as much as compared to the developed economies although the latter depends so much on imports to meet the demand for crushed silicates. For example, the UK has the highest proportion of imported inputs per kg crushed silicate produced. Although a 2006 British Geological survey report (Brown et al., 2008) concluded that UK's reliance on imported aggregates should be reduced. Brown et al. also recommended an increase in indigenous production. This study suggests that over a decade later, the situation remains the same. The USA shows a similar production pattern with the second highest import per kg crushed silicate produced.

Selection of production inputs for intermediate consumption which comprise both domestic and imported input is paramount in understanding the environmental externalities associated with the crushed silicate production supply chain. Environmental impacts are not captured within the producing country, i.e., exported, and can drastically underestimate the potential impacts within national boundary (Ghertner and Fripp, 2007; Ibn-Mohammed et al., 2016; Oppon et al., 2018). M&Q sectors in the developed economies rely more heavily on imported inputs from the M&Q sectors of other countries to produce crushed silicate and also provides evidence of the industrial outsourcing concept (York et al., 2004). York et al. (2004) explains that through their imports, developed countries can reduce their environmental, material, and energy use impacts. For the environmental impacts assessed in this study, countries like the UK appear to have relatively low impacts in energy use, GHGs, and material use. In line with industrial outsourcing philosophy, low environmental impact in the UK is due to the high imported inputs through intermediate consumption.

The findings are particularly important to inform policy and decision-making to increase crushed silicate production for EW use. When the environmental impact from producing these crushed silicates is found to drive larger environmental damaging impacts elsewhere, perhaps a better solution would be to consider artificial crushed silicates such as coal ash, cement kiln dust, and steel slag (Renforth et al., 2011). Artificial crushed silicates may fit more effectively in the circular economy agenda and present an opportunity to reduce environmental impacts. If countries could tap into these secondary sources, the need for countries to mine and quarry large quantities of crushed silicates could be reduced.

5. Conclusion

The paper presented a multi-country study of macro-level economic and environmental sustainability impacts from the production of

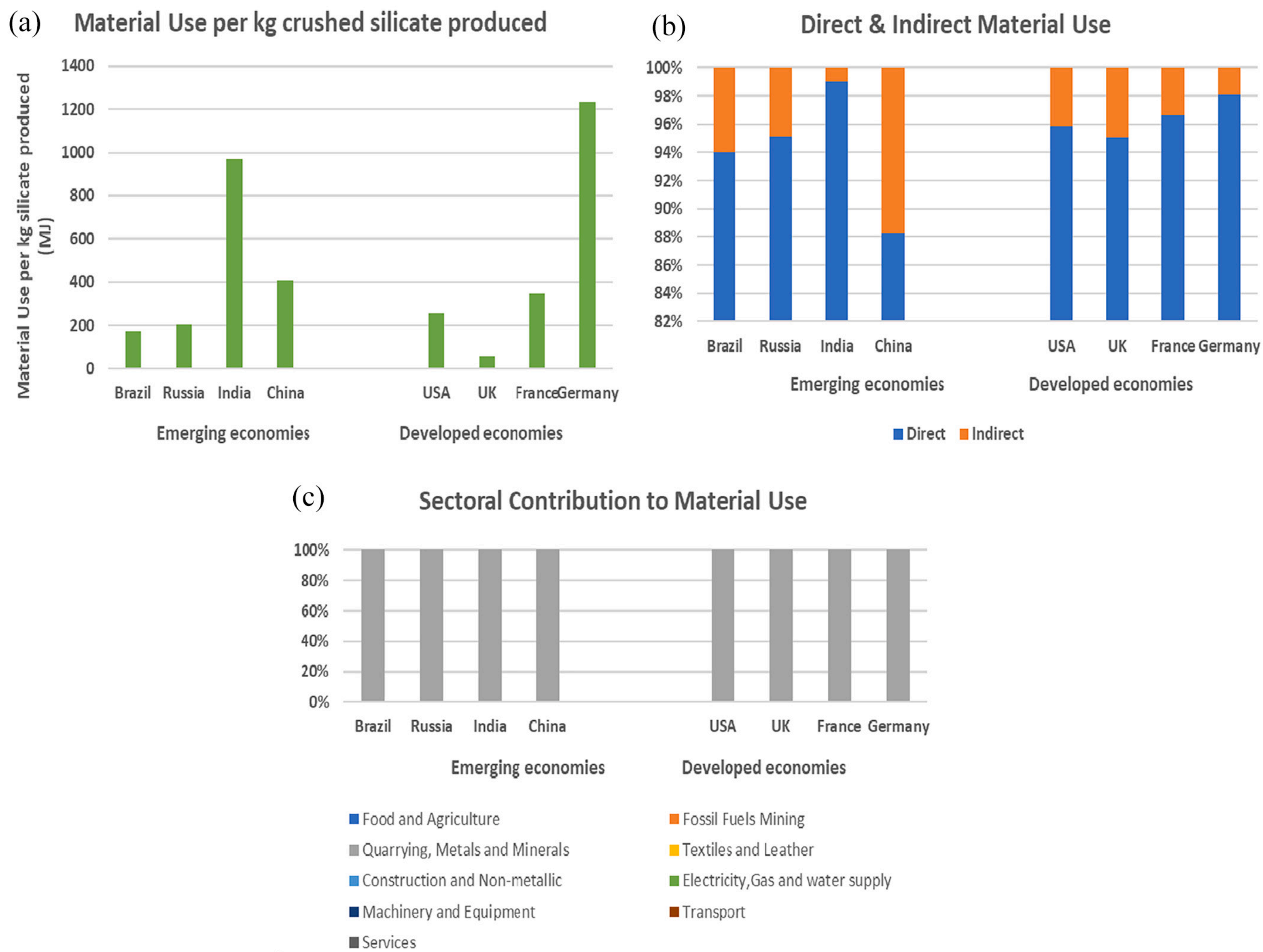

Fig. 7. a) Material use per kg crushed silicate produced, b) Direct & Indirect Material use, and c) Sectoral Contribution of Material use.

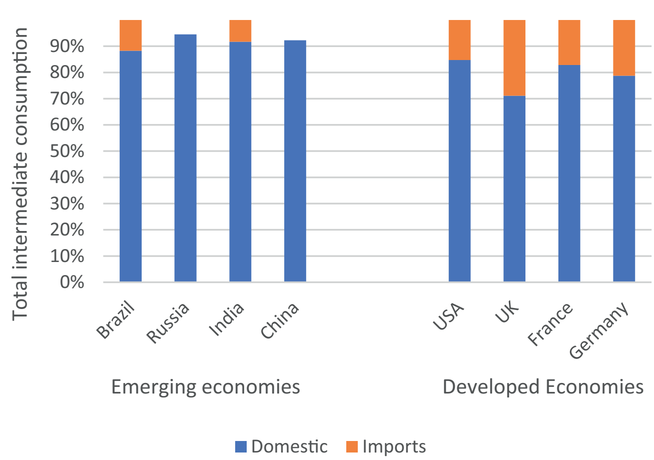

Fig. 8. Comparing intermediate domestic and imported consumption within the M&Q sector.

crushed silicates. Although results are shown based on a functional unit of a kg crushed silicate produced, the fundamental analysis can be scaled up to evaluate how countries are impacted by industrial scale production, e.g., millions of tonnes per annum. The I-O framework is used to estimate potential environmental impacts associated with mining and quarrying of crushed silicates. Six sustainability impact categories were assessed including three economic, GDP, GOS, and imports, with three environmental, energy use, greenhouse gas emissions, and material use. Using the I-O model, total impacts were determined including direct and indirect impacts arising from production inputs required for silicate production. Finally, impacts were also traced to determine the sectors with significant contributions to overall economic and environmental impacts.

In this paper, the result shows evidence of how negative environmental impacts such as GHGs emissions associated with crushed silicate production for EW can be 'leaked' or transferred to other countries through import. This was observed especially in the case of developed countries particularly the UK that relied on relatively high imported inputs in producing the crushed silicates followed by USA and Germany. On the other hand, emerging economies such as Russia and India had relatively low imports which implied, they relied more on domestic inputs in producing crushed silicates. Energy use was also found to be generally higher for the emerging economies (Russia, China, and India) than the developed economies. However, the exception among the developed economies was Germany which had relatively high energy and material use compared to its counterparts like UK and France. In terms of economic impacts, for most of the developed economics especially USA, France and Germany, GDP and GOS potential from crushed silicate production was relatively higher compared to the emerging economies in particular Brazil.

The I-O model was used to trace how much of the impacts estimated

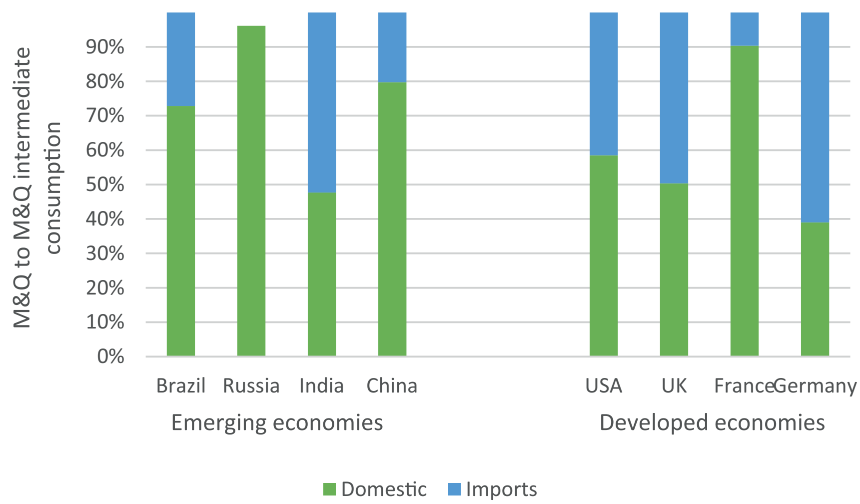

Fig. 9. Comparing M&Q to M&Q intermediate consumption.

was as a result of direct or indirect production inputs. Industry level analysis also highlighted the specific sectors within countries that were hotspots for economic and environmental impacts associated with crushed silicate production. The Mining and Quarrying sector had the highest sectoral contribution to economic and sustainability impacts across the two groups of countries (emerging and developed economies). Knowledge of such information can help guide countries in making decisions to develop targeted interventions or solutions that can address some of the sustainability challenges within that sector. Future research should be directed toward complete triple bottom line sustainability assessment of crushed silicate production in the EW supply chain, specifically by assessing the third sustainability dimension, social.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolecon.2022.107636.

References

Acquaye, A.A, Duffy, A.P, 2010. Input–output analysis of Irish construction sector greenhouse gas emissions. Build. Environ. 45(3, 784–791. https://doi.org/10.1016/j.buildenv.2009.08.022.

Acquaye, A.A., Wiedmann, T., Feng, K., Crawford, R.H., Barrett, J., Kuylenstierna, J., Duffy, A.P., Koh, S.C.L., McQueen-Mason, S., 2011. Identification of 'carbon hot-spots' and quantification of GHG intensities in the biodiesel supply chain using hybrid LCA and structural path analysis. Environ. Sci. Technol. 45 (6), 2471–2478. https://doi.org/10.1021/es103410q.

Ahi, P., Searcy, C., 2015. Assessing sustainability in the supply chain: a triple bottom line approach. Appl. Math. Model. 39 (10–11), 2882–2896. https://doi.org/10.1016/j.apm.2014.10.055.

Alexander, T., Dziobek, M.C.H., Galeza, T., 2018. Sustainable development goals (SDGs) and GDP: what national accounts bring to the table. Int. Monetary Fund. WPIEA2018041, 1–21. http://www.imf.org/external/pubs/cat/longres.aspx?sk=45706.

Baral, A, Bakshi, B.R, 2010. Emergy analysis using US economic input–output models with applications to life cycles of gasoline and corn ethanol. Ecolog. Model. 221 (15) https://doi.org/10.1016/j.ecolmodel.2010.04.010.

Beerling, D.J., Kantzas, E.P., Lomas, M.R., Wade, P., Eufrasio, R.M., Renforth, P., Sarkar, B., Andrews, M.G., James, R.H., Pearce, C.R., 2020. Potential for large-scale CO2 removal via enhanced rock weathering with croplands. Nature 583 (7815), 242–248. https://doi.org/10.1038/s41586-020-2448-9.

Bode, C., Wagner, S.M., 2015. Structural drivers of upstream supply chain complexity and the frequency of supply chain disruptions. J. Oper. Manag. 36, 215–228. https://doi.org/10.1016/j.jom.2014.12.004.

Brentrup, F., Küsters, J., Lammel, J., Barraclough, P., Kuhlmann, H., 2004. Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. Eur. J. Agron. 20 (3), 265–279. https://doi.org/10.1016/S1161-0301(03)00039-X.

Brown, T., McEvoy, F.M., Mankelow, J., Ward, J., Bloomfield, S., Goussarova, T., Shah, N., Souron, L., 2008. The Need for Indigenous Aggregates Production in England. https://nora.nerc.ac.uk/id/eprint/3711.

Camanzi, L., Alikadic, A., Compagnoni, L., Merloni, E., 2017. The impact of greenhouse gas emissions in the EU food chain: a quantitative and economic assessment using an environmentally extended input-output approach. J. Clean. Prod. 157, 168–176. https://doi.org/10.1016/j.jclepro.2017.04.118.

Chen, G., Hadjikakou, M., Wiedmann, T., 2017. Urban carbon transformations: unravelling spatial and inter-sectoral linkages for key city industries based on multi-region input–output analysis. J. Clean. Prod. 163, 224–240. https://doi.org/10.1016/j.jclepro.2016.04.046.

Cooper, J., Dubey, L., Hawkes, A., 2022. The life cycle environmental impacts of negative emission technologies in North America. Sustain. Prod. Consump. 32, 880–894. https://doi.org/10.1016/j.spc.2022.06.010.

Deutch, J, 2017. Decoupling economic growth and carbon emissions. Joule 3–5. https://doi.org/10.1016/j.joule.2017.08.011.

Egilmez, G., Kucukvar, M., Tatari, O., 2013. Sustainability assessment of US manufacturing sectors: an economic input output-based frontier approach. J. Clean. Prod. 53, 91–102. https://doi.org/10.1016/j.jclepro.2013.03.037.

Eufrasio, R.M., Kantzas, E.P., Edwards, N.R., Holden, P.B., Pollitt, H., Mercure, J.F., Koh, S.C., Beerling, D.J., 2022. Environmental and health impacts of atmospheric CO2 removal by enhanced rock weathering depend on nations' energy mix. Commun. Earth Environ. 3 (1), 1–13. https://doi.org/10.1038/s43247-022-00436-3.

Fankhauser, S., Tol, R.S.J., 2005. On climate change and economic growth. Resour. Energy Econ. 27 (1), 1–17. https://doi.org/10.1016/j.reseneeco.2004.03.003.

Foran, B., Lenzen, M., Dey, C., Bilek, M., 2005. Integrating sustainable chain management with triple bottom line accounting. Ecol. Econ. 52 (2), 143–157. https://doi.org/10.1016/j.ecolecon.2004.06.024.

Fugiel, A., Burchart-Korol, D., Czaplicka-Kolarz, K., Smoliński, A., 2017. Environmental impact and damage categories caused by air pollution emissions from mining and quarrying sectors of European countries. J. Clean. Prod. 143, 159–168. https://doi.org/10.1016/j.jclepro.2016.12.136.

Fuss, S., Jones, C.D., Kraxner, F., Peters, G.P., Smith, P., Tavoni, M., van Vuuren, D.P., Canadell, J.G., Jackson, R.B., Milne, J., 2016. Research priorities for negative emissions. Environ. Res. Lett. 11 (11), 115007 https://doi.org/10.1088/1748-9326/11/11/115007.

Gallego, B., Lenzen, M., 2005. A consistent input–output formulation of shared producer and consumer responsibility. Econ. Syst. Res. 17 (4), 365–391. https://doi.org/10.1080/09535310500283492.

Garcia, W.D.O., Amann, T., Hartmann, J., Karstens, K., Popp, A., Boysen, L.R., Smith, P., Goll, D., 2020. Impacts of enhanced weathering on biomass production for negative emission technologies and soil hydrology. Biogeosciences. https://doi.org/10.5194/bg-17-2107-2020.

Ghertner, D.A., Fripp, M., 2007. Trading away damage: quantifying environmental leakage through consumption-based, life-cycle analysis. Ecol. Econ. 63 (2–3), 563–577. https://doi.org/10.1016/j.ecolecon.2006.12.010.

Goldich, S.S, 1938. A study in rock-weathering. J. Geol. 17–58. https://doi.org/10.1086/624619.

Guo, M., Murphy, R.J., 2012. LCA data quality: sensitivity and uncertainty analysis. Sci. Total Environ. 435, 230–243. https://doi.org/10.1016/j.scitotenv.2012.07.006.

Hartmann, J., West, A.J., Renforth, P., Köhler, P., De La Rocha, C.L., Wolf-Gladrow, D.A., Dürr, H.H., Scheffran, J., 2013. Enhanced chemical weathering as a geoengineering strategy to reduce atmospheric CO2, supply nutrients, and mitigate ocean acidification. Rev. Geophys. 51 (2), 113–149. https://doi.org/10.1002/rog.20004.

Hauknes, J., Knell, M., 2009. Embodied knowledge and sectoral linkages: an input–output approach to the interaction of high-and low-tech industries. Res. Policy 38 (3), 459–469. https://doi.org/10.1016/j.respol.2008.10.012.

Hayami, H., Nakamura, M., Nakamura, A.O., 2015. Economic performance and supply chains: the impact of upstream firms' waste output on downstream firms' performance in Japan. Int. J. Prod. Econ. 160, 47–65. https://doi.org/10.1016/j.ijpe.2014.09.012.

Heijungs, R., Huppes, G., Guinée, J.B., 2010. Life cycle assessment and sustainability analysis of products, materials and technologies. Toward a scientific framework for sustainability life cycle analysis. Polym. Degrad. Stab. 95 (3), 422–428. https://doi.org/10.1016/j.polymdegradstab. 2009.11.010.

Hendrickson, C., Horvath, A., Joshi, S., Lave, L., 1998. Peer reviewed: economic input–output models for environmental life-cycle assessment. Environ. Sci. Technol. 32 (7), 184A–191A https://pubs.acs.org/doi/10.1021/es983471i.

Huang, Y.A., Weber, C.L., Matthews, H.S., 2009. Carbon footprinting upstream supply chain for electronics manufacturing and computer services. IEEE. https://doi.org/10.1109/ISSST.2009.5156679.

Ibn-Mohammed, T., Koh, S.C.L., Reaney, I.M., Acquaye, A., Wang, D., Taylor, S., Genovese, A., 2016. Integrated hybrid life cycle assessment and supply chain environmental profile evaluations of lead-based (lead zirconate titanate) versus lead-free (potassium sodium niobate) piezoelectric ceramics. Energy Environ. Sci. 9 (11), 3495–3520. https://doi.org/10.1039/C6EE02429G.

Jia, X., Zhang, Z., Wang, F., Li, Z., Wang, Y., Aviso, K.B., Foo, D.Y., Nair, P.N.S.B., Tan, R. R., Wang, F., 2022. Regional carbon drawdown with enhanced weathering of non-hazardous industrial wastes. Resour. Conserv. Recycl. 176, 105910 https://doi.org/10.1016/j.resconrec.2021.105910.

Kantola, I.B, Masters, M.D, Beerling, D.J, Long, S.P, DeLucia, E.H, 2017. Potential of global croplands and bioenergy crops for climate change mitigation through deployment for enhanced weathering. Biology Lett. 20160714. https://doi.org/10.1098/rsbl.2016.0714.

Kitzes, J., 2013. An introduction to environmentally-extended input-output analysis. Resources 2 (4), 489–503. https://doi.org/10.3390/resources2040489.

Knight, K.W., Schor, J.B., 2014. Economic growth and climate change: a cross-national analysis of territorial and consumption-based carbon emissions in high-income countries. Sustainability 6 (6), 3722–3731. https://doi.org/10.3390/su6063722.

Koornneed, J., Nieuwlaar, E., 2009. *Environmental life cycle assessment of CO2 sequestration through enhanced weathering of olivine*. Working paper, Group Science, Technology and Society, Utrecht University.

Kucukvar, M., Egilmez, G., Tatari, O., 2014. Sustainability assessment of US final consumption and investments: triple-bottom-line input–output analysis. J. Clean. Prod. 81, 234–243. https://doi.org/10.1016/j.jclepro.2014.06.033.

Larkin, A., Kuriakose, J., Sharmina, M., Anderson, K., 2018. What if negative emission technologies fail at scale? Implications of the Paris Agreement for big emitting nations. Clim. Pol. 18 (6), 690–714. https://doi.org/10.1080/14693062.2017.1346498.

Lave, L.B., Cobas-Flores, E., Hendrickson, C.T., McMichael, F.C., 1995. Using input-output analysis to estimate economy-wide discharges. Environ. Sci. Technol. 29 (9), 420A–426A. https://doi.org/10.1021/es00009a748.

Lefebvre, D., Goglio, P., Williams, A., Manning, D.A.C., de Azevedo, A.C., Bergmann, M., Meersmans, J., Smith, P., 2019. Assessing the potential of soil carbonation and enhanced weathering through life cycle assessment: a case study for Sao Paulo State, Brazil. J. Clean. Prod. 233, 468–481. https://doi.org/10.1016/j.jclepro.2019.06.099.

Lehmann, J, Possinger, A, 2020. Removal of atmospheric CO2 by rock weathering holds promise for mitigating climate change. Nature.

Leontief, W., 1986. Input-Output Economics. Oxford University Press.

Liu, L., Huang, G., Baetz, B., Zhang, K., 2018. Environmentally-extended input-output simulation for analyzing production-based and consumption-based industrial greenhouse gas mitigation policies. Appl. Energy 232, 69–78. https://doi.org/10.1016/j.apenergy.2018.09.192.

Lu, Y., 2017. China's electrical equipment manufacturing in the global value chain: a GVC income analysis based on world input-output database (WIOD). Int. Rev. Econ. Financ. 52, 289–301. https://doi.org/10.1016/j.iref.2017.01.015.

McNerney, J., Fath, B.D., Silverberg, G., 2013. Network structure of inter-industry flows. Phys. A: Stat. Mech. Appl. 392 (24), 6427–6441. https://doi.org/10.1016/j.physa.2013.07.063.

Monteiro, N.B.R., da Silva, E.A., Neto, J.M.M., 2019. Sustainable development goals in mining. J. Clean. Prod. 228, 509–520. https://doi.org/10.1016/j.jclepro.2019.04.332.

Nejat, P., Jomehzadeh, F., Taheri, M.M., Gohari, M., Majid, M.Z.A., 2015. A global review of energy consumption, CO2 emissions and policy in the residential sector (with an overview of the top ten CO2 emitting countries). Renew. Sust. Energ. Rev. 43, 843–862. https://doi.org/10.1016/j.rser.2014.11.066.

Noonan, E., 2020. Decoupling Economic Growth from Environmental Harm doi: 20.500.12592/ffsc4x.

Noori, M., Kucukvar, M., Tatari, O., 2015. A macro-level decision analysis of wind power as a solution for sustainable energy in the USA. Int. J. Sustain. Energy 34 (10), 629–644. https://doi.org/10.1080/14786451.2013.854796.

Norgate, T., Haque, N., 2010. Energy and greenhouse gas impacts of mining and mineral processing operations. J. Clean. Prod. 18 (3), 266–274. https://doi.org/10.1016/j.jclepro.2009.09.020.

Onat, N.C., Kucukvar, M., Tatari, O., 2014. Integrating triple bottom line input–output analysis into life cycle sustainability assessment framework: the case for US buildings. Int. J. Life Cycle Assess. 19 (8), 1488–1505. https://doi.org/10.1007/s11367-014-0753-y.

Oppon, E., Acquaye, A., Ibn-Mohammed, T., Koh, L., 2018. Modelling multi-regional ecological exchanges: the case of UK and Africa. Ecol. Econ. 147, 422–435. https://doi.org/10.1016/j.ecolecon.2018.01.030.

Pao, H.-T., Tsai, C.-M., 2010. CO2 emissions, energy consumption and economic growth in BRIC countries. Energy Policy 38 (12), 7850–7860. https://doi.org/10.1016/j.enpol.2010.08.045.

Rebitzer, G., Hunkeler, D., 2003. "life cycle costing in LCM: ambitions, opportunities, and limitations." the. Int. J. Life Cycle Assess. 8 (ARTICLE), 253–256. https://doi.org/10.1007/BF02978913.

Renforth, P., Washbourne, C.L., Taylder, J., Manning, D.A.C., 2011. Silicate Production and Availability for Mineral Carbonation. https://doi.org/10.1021/es103241w.

Richardson, H.W., 1985. Input-output and economic base multipliers: looking backward and forward. J. Reg. Sci. 25 (4), 607–661. https://doi.org/10.1111/j.1467-9787.1985.tb00325.x.

Richter, J.S., Mendis, G.P., Nies, L., Sutherland, J.W., 2019. A method for economic input-output social impact analysis with application to US advanced manufacturing. J. Clean. Prod. 212, 302–312. https://doi.org/10.1016/j.jclepro.2018.12.032.

Rogelj, J, Den Elzen, M, Höhne, N, Fransen, T, Fekete, H, Schaeffer, R, Sha, F, Riahi, K, Meinshausen, M, 2016. Paris Agreement climate proposals need a boost to keep warming well below 2 C. Nature 534 (7609), 631–639. https://doi.org/10.1038/nature18307.

Schuiling, R.D., Krijgsman, P., 2006. Enhanced weathering: an effective and cheap tool to sequester CO 2. Clim. Chang. 74 (1–3), 349–354. https://doi.org/10.1007/s10584-005-3485-y.

Smith, P., 2016. Soil carbon sequestration and biochar as negative emission technologies. Glob. Chang. Biol. 22 (3), 1315–1324. https://doi.org/10.1111/gcb.13178.

Smith, P., Davis, S.J., Creutzig, F., Fuss, S., Minx, J., Gabrielle, B., Kato, E., Jackson, R.B., Cowie, A., Kriegler, E., 2016a. Biophysical and economic limits to negative CO 2 emissions. Nat. Clim. Chang. 6 (1), 42–50. https://doi.org/10.1038/nclimate2870.

Smith, P., Haszeldine, R.S., Smith, S.M., 2016b. Preliminary assessment of the potential for, and limitations to, terrestrial negative emission technologies in the UK. Environ Sci Process Impacts 18 (11), 1400–1405. https://doi.org/10.1039/C6EM00386A.

Smith, P., Adams, J., Beerling, D.J., Beringer, T., Calvin, K.V., Fuss, S., Griscom, B., Hagemann, N., Kammann, C., Kraxner, F., 2019. Impacts of land-based greenhouse gas removal options on ecosystem services and the United Nations Sustainable Development Goals. Annu. Rev. Environ. Resour. 44 https://doi.org/10.1146/annurev-environ-101718-033129.

Strefler, J., Amann, T., Bauer, N., Kriegler, E., Hartmann, J., 2018. Potential and costs of CO2 removal by enhanced weathering of rocks. Environ. Res. Lett. 13 (3), 034010 https://doi.org/10.1088/1748-9326/aaa9c4.

Swarr, T.E, Hunkeler, D., Klöpffer, W., Pesonen, H.L, Ciroth, A, Brent, A.C, Pagan, R, 2011. Environmental life-cycle costing: a code of practice. Internat. J. Lifecycle Assess. 16 (5), 389–391. https://doi.org/10.1007/s11367-011-0287-5.

Tamazian, A., Chousa, J.P., Vadlamannati, K.C., 2009. Does higher economic and financial development lead to environmental degradation: evidence from BRIC countries. Energy Policy 37 (1), 246–253. https://doi.org/10.1016/j.enpol.2008.08.025.

Tan, R.R., Belmonte, B.A., Benjamin, M.F.D., Andiappan, V., Aviso, K.B., 2022. Optimization of enhanced weathering networks with alternative transportation modes. Carbon Res. Conv. https://doi.org/10.1016/j.crcon.2022.04.002.

Taylor, L.L, Quirk, J, Thorley, R, Kharecha, P.A, Hansen, J, Ridgwell, A, Lomas, M.R, Banwart, S.A, Beerling, D.J, 2016. Enhanced weathering strategies for stabilizing climate and averting ocean acidification. Nat. Climate Change 6 (4), 402–406. https://doi.org/10.1038/nclimate2882.

Timmer, M., Erumban, A.A., Gouma, R., Los, B., Temurshoev, U., de Vries, G.J., Arto, I. A., Genty, V.A.A., Neuwahl, F., Francois, J., 2012. The World Input-Output Database (WIOD): Contents, Sources and Methods. Institue for International and Development Economics. https://www.ecb.europa.eu/home/pdf/research/compnet/Timmer_2012.pdf.

Tukker, A., Dietzenbacher, E., 2013. Global multiregional input–output frameworks: an introduction and outlook. Economic Systems Research 25(1), 1–19. https://doi.org/10.1080/09535314.2012.761179.

UNEP, 2011. Decoupling natural resource use and environmental impacts from economic growth. In: Fischer-Kowalski, M., Swilling, M., von Weizsäcker, E.U., Ren, Y., Moriguchi, Y., Crane, W., Krausmann, F., Eisenmenger, N., Giljum, S., Hennicke, P., Romero Lankao, P., Siriban Manalang, A. (Eds.), A Report of the Working Group on Decoupling to the International Resource Panel. http://www.gci.org.uk/Documents/Decoupling_Report_English.pdf.

Wang, Q., Li, R., Liao, H., 2016. Toward Decoupling: Growing GDP without Growing Carbon Emissions. https://doi.org/10.1021/acs.est.6b05150.

Wiedmann, T., Lenzen, M., Turner, K., Barrett, J., 2007. Examining the global environmental impact of regional consumption activities—part 2: review of input–output models for the assessment of environmental impacts embodied in trade. Ecol. Econ. 61 (1), 15–26. https://doi.org/10.1016/j.ecolecon.2006.12.003.

Winkler, H., Brouns, B., Kartha, S., 2006. Future mitigation commitments: differentiating among non-annex I countries. Clim. Pol. 5 (5), 469–486. https://doi.org/10.1080/14693062.2006.9685572.

Wulf, C., Werker, J., Zapp, P., Schreiber, A., Schlör, H., Kuckshinrichs, W., 2018. Sustainable development goals as a guideline for indicator selection in life cycle sustainability assessment. Proc. Cirp 69, 59–65. https://doi.org/10.1016/j.procir.2017.11.144.

Yang, Y., Ingwersen, W.W., Hawkins, T.R., Srocka, M., Meyer, D.E., 2017. USEEIO: a new and transparent United States environmentally-extended input-output model. J. Clean. Prod. 158, 308–318. https://doi.org/10.1016/j.jclepro.2017.04.150.

Yang, X., Hu, M., Zhang, C., Steubing, B., 2022. Urban mining potential to reduce primary material use and carbon emissions in the Dutch residential building sector. Resour. Conserv. Recycl. 180, 106215 https://doi.org/10.1016/j.resconrec.2022.106215.

York, R., Rosa, E.A., Dietz, T., 2004. The ecological footprint intensity of national economies. J. Ind. Ecol. 8 (4), 139–154. https://doi.org/10.1162/1088198043630487.