Environmental impacts and environmental justice implications of supplementary cementitious materials for use in concrete

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Abstract
As the second most used material after water and the producer of 8%–9% of anthropogenic greenhouse gas (GHG) emissions, concrete is a key target for environmental sustainability efforts. Of these efforts, a main focus has been the use of industrial byproducts as supplementary cementitious materials (SCMs) to replace some of the cement binder, the source of most of the GHG emissions from concrete production. As byproducts, these SCMs are frequently assumed to have limited or no emissions from production. Our goal is to see if this assumption should continue to drive mitigation efforts and to arrive at a clearer understanding of the contribution of SCMs to the environmental impacts of concrete. Needing further examination are: (1) how environmentally beneficial SCMs are if some of the primary process impacts are attributed to them rather than considering them waste products; (2) whether transporting SCMs creates greater environmental impacts than the materials they are replacing; and (3) whether location of primary processes that result in SCMs as well as location of concrete production creates particular burdens on lower income and minority communities. This work focuses on three of the most common industrial byproduct SCMs, namely silica fume, fly ash (FA), and ground granulated blast furnace slag (BFS), exploring both GHG and particulate matter emissions. We show that allocation of impacts from primary processes dramatically increases emissions attributed to SCMs. High levels of transportation of FA and BFS typically do not result in these SCMs having higher GHG emissions than a 95% clinker-content Portland cement. We find that SCMs may be produced in areas with low income or minority populations then used to lower GHG emissions concrete in another location. As such, beyond common environmental impact assessment methods, the role of environmental justice should be incorporated into impact assessments.

1. Introduction
Since 1990, global production of cement (the hydraulic binder in concrete) has increased by a factor of 4, growing faster than the production of any other construction material [1, 2]. Concrete is now the second-most consumed material after water [2]. Cement production is leading to significant greenhouse gas (GHG) emissions, both from the energy resources used in its production and from process-derived emissions from calcination (in which limestone is converted to lime and CO₂). The highest emissions for cement production occur during the manufacture of clinker, a kilned and quenched material that is interground with other minerals to form Portland cement (PC). Currently, these calcination emissions are estimated to contribute to about 4% of global fossil-derived CO₂ emissions [3]. When energy-derived emissions are included, emissions from the cement industry are estimated to be about 7% of the global CO₂ emissions annually [4]. Including other constituents and processes, concrete production results in approximately 8%–9% of global anthropogenic GHG emissions [5]. Lowering the clinker content of PC or partial replacement of PC with supplementary cementitious materials (SCMs) is a commonly proposed method for decreasing GHG emissions from cement. SCMs are widely used today to enhance concrete performance, such as contributing to lower heat of hydration, improved control of alkali–silica reaction,
increased long-term strength, and improvements to long-term durability \[6, 7\]. It has been claimed that it is possible to have a 50% replacement of conventional PC with SCMs while maintaining or improving performance, depending on SCMs used \[8\]. If the improved performance from SCMs can contribute to increased concrete longevity, there can be notable reductions in GHG emissions if new cement production is offset \[9\].

While SCMs can originate from natural mineral deposits, such as natural pozzolans and limestone \[10\], many common SCMs are byproducts of other industries, with among the three most prevalent industrial byproduct SCMs being fly ash (FA) from coal combustion, ground granulated blast furnace slag (BFS) from iron production, and silica fume (SF) from silicon manufacturing. These byproduct SCMs are typically considered to have low or negligible environmental impacts. Yet byproduct SCMs are inherently dependent on other industries. Some have argued that the allocation of emissions of a byproduct’s main producer is not necessary \[11, 12\]. However, more recent literature has noted that the strong influence of allocation to these byproducts on the cumulative impacts for concrete mixtures and their value to the concrete industry warrants analysis \[13\]. Both mass and economic methods of allocation have been examined, and both methods have been shown to increase the environmental impacts of the SCMs, with mass allocation methods noted as more stable over time as market prices fluctuate \[14\]. More recent studies have extended beyond environmental impact comparisons focused solely on material production to incorporate the typically beneficial influence of SCMs on concrete compressive strength and durability, while simultaneously analyzing the role of allocation methods on the desirability of using SCMs to mitigate environmental impacts \[15, 16\].

In addition to GHG emissions that could be attributed to SCMs from primary processes, transportation also contributes to their net emissions. It has been noted that in some regions, the potential to increase use of FA and BFS is constrained due to the limited supply of suitable quality byproducts \[17\]. With continued demand to increase the use of SCMs in cement (e.g. \[8\]), higher degrees of transportation may become necessary. Depending on mode of transportation and populations exposed, various concerns arise from transportation-related GHG and particulate matter (PM) emissions \[18\].

Air pollutant emissions are not limited to transportation. Such emissions also come from concrete constituent manufacture and batching \[19\]. In the production of cement and concrete, PM emissions can originate from energy resources, such as the combustion of fossil fuels \[20\], and from the materials themselves, such as dust particulates formed during quarrying, kilning, or batching \[21, 22\]. Depending on factors such as particle size, human exposure, and intake fraction, PM emissions can have negative consequences to human health \[23, 24\] raising questions of environmental justice. Higher exposure of low income and minorities to air pollution is well documented (see for example \[25–27\]). Unexplored is whether SCM, PC, and concrete production, which emit PM, are located in lower income or minority communities. While concrete is typically batched near its use \[28\], industrial processes that result in production of SCMs are usually limited to specific areas, thus transport is required to where it will be used in concrete. As a result, a concrete mixture could be batched in one location, with apparent reduction in GHG emissions from use of SCMs, but local health burdens from PM emissions could occur in another location (at the site of the primary processes for SCM production and/or in material transit). Environmental justice issues emerge when these locations are coterminous with economically depressed or minority populations.

The objective of this work is to examine three factors that can affect the environmental sustainability of SCMs: (i) allocation of impacts from the primary good; (ii) the impacts of transportation to the concrete batching site; and (iii) potential environmental justice issues. This study focuses on the three most popular industrial byproduct SCMs: FA, BFS, and SF. In this work, the effects of the no allocation method and mass allocation from primary products on GHG and PM emissions are explored. The global production of these SCMs relative to PC production is assessed as are the ‘tipping points’ of transportation, beyond which additional transportation would lead to greater impacts than using only conventional PC. Finally, location of SCM, PC, and concrete production are examined relative to population demographics to evaluate potential environmental justice associated with these materials.

2. Methodology

2.1. Supplementary cementitious materials production

The production of the SCMs FA, BFS, and SF, was assessed at the country-level using national production statistics of the primary products and factors to indicate the quantity of byproduct. To determine the amount of FA produced in each country, the amount of coal consumed per country for the transformation sector, energy sector, and industrial sector was obtained using the United States Energy Information Administration (EIA) international coal and coke consumption data for 2015 \[29\]. The amount of FA produced from coal consumption was based on available statistics for ash content and FA fraction in ash from coal combustion.
using United States (US) data. Namely, coal consumption in the US for 2018 was 636.5 million short tons for electric power, 49.86 million short tons for industrial purposes, and 0.97 million short tons for commercial purposes [30]. The ash content for those sectors in the same year was 8.34% from coal consumption for electric power, 7.89% for commercial, and 7.66% for industrial sectors [31]. Using these statistics, a weighted average of 8.29% total ash content for combusted coal was derived (see table 1). To estimate the amount of FA, the fraction of FA relative to total ash was modeled as 65.54% (based on data from industry data [32]); similar FA fractions have been reported by others (e.g. [33, 34]). Cumulatively, this ash content and FA fraction resulted in approximately 0.0543 kg of FA produced for every kg of coal consumed. This factor was multiplied by national coal consumption statistics to determine the amount of FA produced in each country. The authors note that varying coal composition could alter ash content [35]; however, such variations are outside the scope of this analysis.

To determine global BFS production, national statistics for pig iron production in 2015 were used (data from [36]). While BFS production per kg of pig iron formed using a blast furnace can vary, the range reported in the US falls between 0.25–0.3 kg of BFS per kg of iron. Therefore, in this work, the average of these values, namely 0.275 kg of BFS per kg of pig iron was modeled (based on data from [37]). This factor was multiplied by national pig iron production statistics to determine the amount of BFS produced in each country.

A similar method was applied to determine global SF production. National statistics for silicon and ferrosilicon production were determined for 2015 (using data from [38]). SF byproduct from silicon and ferrosilicon production was estimated using ratios of 0.55 kg of SF produced per 1 kg of silicon and 0.35 kg of SF produced per 1 kg of ferrosilicon (based on values reported by [39]). These factors were applied to national statistics to determine country-specific SF production. The authors note that, unlike FA and BFS, SF is typically used at low cement replacement levels to achieve high strength concrete mixtures [40].

### 2.2. Environmental impacts of Portland cement and supplementary cementitious materials without allocation from primary processes

To understand the influence of allocation and transportation on the environmental impacts of industrial byproduct SCMs, models were first derived to capture the impacts associated with attaining these reactive compounds. Additionally, environmental impact models for the production of conventional PC and impacts of different modes of transportation were determined to facilitate comparisons made. A cradle-to-gate scope of analysis was used in this work, namely, the environmental impacts from raw material acquisition through the production of PC or SCMs for use in concrete were analyzed. Impacts beyond the production and transportation of these constituents were considered outside the scope of this work. Two main impact categories were studied: GHG emissions and PM emissions. Specifically, three GHGs were considered: CO₂, CH₄, and N₂O. Emissions of these gases were examined concurrently in terms of CO₂-equivalents (CO₂-eq) using 100 years global warming potentials of 25 for CH₄ and 298 for N₂O, based on values by the Intergovernmental Panel for Climate Change [41]. Two sizes of PM were considered in this work: 10 microns or smaller (PM₁₀) and 2.5 microns or smaller (PM₂.₅). These PM emissions were considered due to their being known to be environmental contributors to the burden of disease for humans [24, 42]; while intake of the finer PM₂.₅ has greater potential health implications than PM₁₀, both are known to contribute to health burdens [24].

Prior to allocation of impacts from the primary processes, the environmental impacts associated with the acquisition of reactive SCMs for use in concrete were assessed based on energy demand and energy resources used. FA and SF, were modeled as having no energy demands for the acquisition of the ashes, i.e., 0 MJ kg⁻¹ FA and 0 MJ kg⁻¹ SF based on values reported by the US Environmental Protection Agency (USEPA) [43] for FA, and a report from the Portland Cement Association (PCA) noting the same emissions for SF capture as for FA [44]. For the processes associated with producing reactive BFS, energy demand was modeled as 2.5104 MJ kg⁻¹ BFS according to an environmental product declaration from the Slag Cement Association [45]. This energy demand was modeled as electricity demand. To determine GHG emissions from this electricity requirement, the global average electricity mix was modeled (based on data from the International Energy Agency (IEA)}

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### Table 1. 2018 US coal consumption, ash production, and average ash content (based on data from [30, 31]).

| Sector       | Coal consumed (million short tons) | Ash produced (million short tons) | Ash content (%) |
|--------------|-----------------------------------|-----------------------------------|-----------------|
| Electric     | 636.5                             | 53.1                              | 8.34            |
| Commercial   | 49.9                              | 0.0765                            | 7.89            |
| Industrial   | 0.97                              | 3.82                              | 7.66            |
| Total        | 687                               | 57.0                              | 8.29            |
Table 2. Emissions factors for electricity and thermal energy resources.

| Resource | CO₂ | CH₄ | N₂O | PM₁₀ | PM₂.⁵ |
|----------|-----|-----|-----|------|-------|
| Coal     | 2.60 × 10⁻¹ | 2.64 × 10⁻⁶ | 3.85 × 10⁻⁴ | 2.98 × 10⁻⁵ | 1.25 × 10⁻³ |
| Oil      | 2.12 × 10⁻¹ | 7.32 × 10⁻⁶ | 1.39 × 10⁻⁴ | 2.02 × 10⁻⁵ | 1.83 × 10⁻³ |
| Natural gas | 1.62 × 10⁻¹ | 2.86 × 10⁻⁶ | 6.44 × 10⁻⁷ | 7.54 × 10⁻⁶ | 6.72 × 10⁻⁶ |
| Biomass  | 0        | 1.28 × 10⁻⁴ | 1.70 × 10⁻³ | 5.94 × 10⁻⁴ | 1.30 × 10⁻³ |
| Other¹   | 0        | 0        | 0    | 0    | 0     |

Emissions factors for thermal energy (kg emission/MJ energy)

| Resource | CO₂ | CH₄ | N₂O | PM₁₀ | PM₂.⁵ |
|----------|-----|-----|-----|------|-------|
| Coal     | 9.83 × 10⁻² | 8.14 × 10⁻⁶ | 1.19 × 10⁻³ | 4.07 × 10⁻⁵ | 1.23 × 10⁻³ |
| Oil      | 7.20 × 10⁻² | 2.56 × 10⁻⁶ | 4.69 × 10⁻⁷ | 7.81 × 10⁻⁷ | 4.12 × 10⁻⁷ |
| Biomass  | 7.96 × 10⁻² | 6.04 × 10⁻⁶ | 8.04 × 10⁻⁷ | 5.91 × 10⁻⁸ | 4.65 × 10⁻⁸ |
| Fossil waste | 1.00 × 10⁻¹ | 1.39 × 10⁻⁵ | 3.13 × 10⁻⁴ | 1.12 × 10⁻⁵ | 1.13 × 10⁻⁵ |
| Petroleum coke² | 9.73 × 10⁻² | 2.99 × 10⁻⁶ | 5.98 × 10⁻⁷ | 2.68 × 10⁻⁸ | 2.68 × 10⁻⁸ |
| Natural gas | 5.61 × 10⁻² | 9.96 × 10⁻⁷ | 9.98 × 10⁻⁸ | 2.40 × 10⁻⁹ | 2.40 × 10⁻⁹ |
| Solid waste² | 1.12 × 10⁻¹ | 2.16 × 10⁻⁵ | 3.28 × 10⁻⁶ | 2.86 × 10⁻⁸ | 2.86 × 10⁻⁸ |
| Liquid waste² | 7.35 × 10⁻² | 3.01 × 10⁻³ | 4.00 × 10⁻⁴ | 1.12 × 10⁻⁵ | 1.13 × 10⁻⁵ |

¹Air emissions assumed to be negligible for electricity resources including hydro, wind, solar, and nuclear.
²Due to limitations in available data, PM₁₀ and PM₂.⁵ emissions from these fuel resources were modeled as equivalent in this study.

for the year 2015 [46]). GHG and PM emissions factors for these electricity sources were based on data from the Argonne National Laboratory [47]. Emissions distributions for each energy resource, combustion technology, boiler bottom and firing type, and emissions controls were input into a Monte Carlo simulation (n = 10,000). Emissions distributions were weighted by use ratios in the US to determine emissions factors by fuel resource, with median values from these distributions applied herein (see emissions factors in table 2). Additional PM emissions from handling of the SCMs were also considered. Emissions from unloading SCMs, with emissions controls, were modeled as 6.5 mg of PM per kg of SCM (based on data from [21]). Additionally, PM emissions attributed to each kg of material at the batching stage were considered as 2.8 mg of PM per kg of SCM (based on [21] with the effects of emissions controls approximated based on controls for similar processes). Due to limitations in available data, these handling-related PM emissions were considered to be the same for PM₁₀ and PM₂.⁵ emissions. While these handling emissions occur at the concrete batching stage, they are attributed to the use of SCMs and thus, are included in these models (flow diagram of processes considered in figure 1).

To determine the conditions under which SCMs could potentially exceed the environmental impacts of clinker-based PC, GHG and PM emissions for PC were modeled using global average production data where possible. Kiln energy demand to produce cement clinker was based on world average energy demand for 2050 as reported by the Getting the Numbers Right (GNR) program [21]. This reference provided world average fractions for 2015 of fossil fuel, waste fuel, and biomass fuel used in cement kilns; to capture more specific fuel types, fractions of fuel types within those categories were based on 2018 ratios from [21]. Thermal energy GHG and PM emissions were taken as the median values based on distributions of emissions determined in a study of GHG and air emissions from the global concrete industry [19] (see table 2). The magnitude of electricity required for raw meal preparation, grinding, and other cement operations were again taken for the global average statistics in 2015 from the GNR program [48]. As was done for the SCMs, the electricity mix was modeled as the world average mix for 2015 from the IEA [46] using the same emissions factors for electricity resources (see table 2). In addition to the GHG emissions from energy-derived resources, decarbonation of limestone was determined through stoichiometry and scaled for a 65% lime clinker and a PC system containing 5% gypsum and 95% clinker. Additional PM emissions associated with material handling were also considered. Emissions during raw material acquisition and kilning were based on [19]. Emissions associated with handling PC during batching were modeled as 0.17 mg PM per kg PC, and cement unloading emissions were modeled as 2.8 mg of PM per kg of PC (based on [49]) with the effects of emissions controls approximated based on the effects of controls for similar processes. Again, due to limitations in available data, handling emissions were considered to be the same for assessment of PM₁₀ and PM₂.⁵. Emissions for each of these constituents are shown in table 3.

In addition to the emissions from producing each of these concrete constituents, we examined emissions from the primary processes that result in industrial byproduct SCMs to apply impact allocations (see section 2.3). In this work, four main primary processes were considered: (i) electricity generation from...
Table 3. GHG, PM$_{10}$, and PM$_{2.5}$ emissions from products considered (with no allocation considered for SCMs).

| Product       | Units                  | GHG     | PM$_{10}$ | PM$_{2.5}$ |
|---------------|------------------------|---------|-----------|------------|
| PC           | kg emission/kg material | $8.26 \times 10^{-1}$ | $1.63 \times 10^{-4}$ | $1.16 \times 10^{-4}$ |
| FA           | kg emission/kg material | $0.00$ | $9.30 \times 10^{-6}$ | $9.30 \times 10^{-6}$ |
| BFS          | kg emission/kg material | $3.73 \times 10^{-1}$ | $8.91 \times 10^{-6}$ | $2.79 \times 10^{-5}$ |
| SF           | kg emission/kg material | $0.00$ | $9.30 \times 10^{-6}$ | $9.30 \times 10^{-6}$ |
| Coal electricity | kg emission/kg material | $1.02 \times 10^{-2}$ | $1.16 \times 10^{-6}$ | $4.84 \times 10^{-7}$ |
| Pig iron     | kg emission/kg material | $1.55$ | $4.72 \times 10^{-3}$ | $9.52 \times 10^{-4}$ |
| Silicon      | kg emission/kg material | $1.12 \times 10^{1}$ | $1.28 \times 10^{-2}$ | $1.00 \times 10^{-2}$ |
| Ferrosilicon | kg emission/kg material | $1.00 \times 10^{1}$ | $4.45 \times 10^{-2}$ | $3.94 \times 10^{-2}$ |

*Emissions only reflect those at the handling stage during batching.

*Calculated based on emissions per MJ of electricity and the lower heating value for coal discussed in section 2.2.

c—assessed as the primary process for the production of FA (the focus on allocation of FA from coal electricity was selected as it is the largest user of coal [30]); (ii) pig iron production—assessed as the primary process for the production of BFS; (iii) silicon production—assessed as one potential primary process for the production of SF; and (iv) ferrosilicon production—assessed as another primary process for the production of SF. Coal-based electricity was modeled based on the same coal emissions factors used for electricity resources (table 2). Pig iron was based on a model from the ecoinvent database, reflecting global average production of iron using a blast furnace [50]. Emissions from the production of silicon and ferrosilicon were also based on models from the ecoinvent database, again using global statistics [51]. Emissions for these primary products are shown in table 3.
2.3. Emissions allocation by mass

In order to allocate emissions from primary products to the SCMs, a mass allocation method was implemented following the methods described in [14, 15]. In this methodology, primary process emissions, \( E_{\text{primary}} \), are the emissions from the primary process (e.g., production of pig iron); secondary process emissions, \( E_{\text{secondary}} \), are emissions from processing the secondary materials, namely, the emissions attributable to the SCMs without consideration of impacts from the primary product. The \( E_{\text{primary}} \) and \( E_{\text{secondary}} \) were determined as discussed in section 2.2. Along with these values, a mass fraction, \( C \), was calculated using the following formula:

\[
C = \frac{m_{\text{secondary}}}{m_{\text{secondary}} + m_{\text{primary}}}
\]  

(1)

where \( m_{\text{secondary}} \) is the mass of the SCM produced per unit mass of the primary product, \( m_{\text{primary}} \). The total emissions attributed to the secondary product including allocated emissions, \( E_{\text{allocated}} \), was found by applying this mass fraction in the following equation:

\[
E_{\text{allocated}} = C \cdot E_{\text{primary}} + E_{\text{secondary}}
\]  

(2)

Mass fractions were derived for each SCM from their primary product and in the case of SF, two different mass fractions were derived for the production each from silicon and from ferrosilicon. To derive the mass fraction for the allocation of impacts from coal electricity to FA, the previously determined production ratio of 0.0543 kg of FA production per kg of coal combusted was used, leading to a mass fraction of 0.052 using equation (1). Again, it is noted that variations in coal resources could lead to different mass fractions for FA, which in turn could alter impacts allocated using this method; however, analysis of such variability was outside the scope of this research. For BFS, the ratio of 0.275 kg of BFS produced per kg of pig iron was applied, as noted earlier, and using equation (1), a mass fraction of 0.22 was derived. To develop the mass fractions for each of the two permutations of SF mass allocations, the ratios noted previously of 0.55 kg of SF per kg of silicon production and 0.35 kg of SF production per kg of ferrosilicon production were used. By applying equation (1), these resulted in mass fractions of 0.35 for SF from silicon production and 0.26 for SF from ferrosilicon.

With these mass fractions, \( E_{\text{allocated}} \) emissions were derived for each of the SCMs. The \( E_{\text{primary}} \) for the allocation of coal combustion emissions to FA emissions was calculated from emissions per MJ electricity. Emissions per kg of coal were determined by multiplying emissions per MJ by the hard coal standard factor of 25.8 MJ electricity/kg coal (based on heating values from [52]). For BFS, SF from silicon, and SF from ferrosilicon, \( E_{\text{primary}} \) emissions were taken directly from values determined in section 2.2.

2.4. Environmental impacts from transportation and tipping points

Impacts of transportation were examined based on a ‘tipping point’ beyond which the transportation of SCMs could result in greater environmental impacts than the production of PC. To quantitatively assess this potential threshold, vehicle operation/propulsion emissions by mode of transportation were taken from [18]. This reference was selected because it models GHG emissions using the same warming potentials as this work and models both PM10 and PM2.5 emissions from various modes of transport. Three modes of transportation were considered: truck, rail, and ship. Each mode of transportation was modeled as using diesel fuel, and the trucks were modeled as heavy–heavy duty as defined by [18]. Using these emissions values, the tipping points of each SCM—defined as the distance at which the emissions of the transported SCM is equal to the emissions of non-transported PC—was determined for GHG, PM10, and PM2.5 emissions. The tipping point was calculated using equation (3):

\[
\text{tipping point} = \frac{E_{\text{PC}} - E_{\text{SCM,0 km}}}{E_{\text{ton-km}}}
\]  

(3)

where \( E_{\text{PC}} \) represents the emissions of the PC with no transportation, \( E_{\text{SCM,0 km}} \) represents the emissions of the SCM before transportation, and \( E_{\text{ton-km}} \) represents the emissions to carry 1 metric ton of material a distance of 1 km. Scenarios 1–6 in table 4 were developed in order to compare tipping points (see table 6) to travel lengths for different modes of transportation. To estimate the travel lengths of the scenarios listed in table 4, distances for travel by sea were found using [53], distances for trucks were found using [54], and distances for train were found using [55]. These travel lengths are defined in this work as the distance from a starting location to an end location as defined by references [53–55]. Tipping point values are presented in table 6.

2.5. Factors affecting environmental justice

In this work, consideration of environmental justice is based on location of SCM production and location of concrete production in the US. First, the location of primary processes that result in industrial byproduct SCMs were determined. Coal electricity plants in 2015 by state and county were determined from the EIA (data from
Table 4. List of scenarios presented in section 4 including the start and end location, mode of transportation, travel lengths, and emission discussed (distances calculated using [53–55]).

| Scenario | Starting location   | End location                     | Mode of transport | Travel length (km) | Emission discussed |
|----------|---------------------|----------------------------------|-------------------|--------------------|--------------------|
| 1        | Darwin, Australia   | Shanghai, China                  | Ship              | 5200.78            | CO₂-eq             |
| 2        | Brisbane, Australia | Port of Los Angeles, US          | Ship              | 11 643.52          | CO₂-eq             |
| 3        | Nur-Sultan-Nurly Zhol, Nur-Sultan, Kazakhstan | Shanghai, China | Train | 9004 | CO₂-eq |
| 4        | Nur-Sultan-Nurly Zhol, Nur-Sultan, Kazakhstan | Shanghai, China | Truck | 9004 | CO₂-eq |
| 5        | Morupule Power Station, Botswana | Dar es Salaam, Tanzania | Truck | 3096 | PM₁₀ |
| 6        | Pljevlja Power Station, Pljevlja, Montenegro | Samarqand, Uzbekistan | Truck | 5455 | PM₁₀ |

EIA-860 database, PlantY2015 file [56]). The location of iron production for 2015 was found using data from the USEPA [57]. Data from the US geological survey were used to determine locations for the ferrosilicon production [58] and silicon production [59], both for the year 2015. Then, ready-mixed concrete (RMC) facility locations were found using the National Ready Mixed Concrete Association database, with data representing production in 2020 [60]. In order to find the locations of the PC plants, state-by-state reports published by the PCA were used, representative of plants in the year 2015 [61]. Next, co-location of SCM, PC, and concrete production in the US was determined.

These production locations were then considered concurrently with data retrieved through the USEPA environmental and demographic indicator mapping tool (EJSCREEN). Specifically, the indicator used here was a demographic index (DI), referred to as VULEOPCT by the USEPA. This index is based on the average data representing the percent low-income and percent minority populations, as quantified and reported by the EJSCREEN tool [62]. For this work, indicators from the year 2018 were used. For each county in the US, a range of DI values are reported, so both the maximum and minimum values were considered herein to address variation.

Plots were made to provide a visualization of industrial production and demographic data using inputs from the US Census Bureau. A county shapefile was obtained from [63] representing counties as of 2017. State Federal Information Processing System (FIPS) codes and geographic identifiers (GEOIDs) were used to match data and plot. FIPS codes and GEOIDs both are numeric codes developed to uniquely identify geographic areas. County-level GEOIDs as of 2015 were obtained from [64], and state FIPS codes were obtained from [65].

3. Results

3.1. Environmental impacts of supplementary cementitious materials with allocation

The allocation of impacts from primary products had large effects on the emissions modeled for the SCMs (table 5). One of the largest shifts from the inclusion of impact allocation was for the GHG emissions for SF from both ferrosilicon and silicon productions sources. Without allocation, SF GHG emissions were modeled as 0 kg CO₂-eq per kg material, but with allocation they increase to above the GHG emissions to produce PC (emissions for PC production presented in table 3). This shift is a function of the high CO₂-eq emissions from ferrosilicon and silicon production at 10.0 and 11.2 kg CO₂-eq per kg material, respectively. These emission values are 49%–622% greater than the CO₂-eq emissions from coal combustion for electricity and pig iron production—the primary production processes for FA and BFS—respectively. The production of FA also has 0 CO₂-eq emissions without allocation [43]; however, the allocated CO₂-eq emissions increase to be ~1/2 of the allocated CO₂-eq emissions of BFS. The primary production CO₂-eq emissions of FA, 6.75 kg CO₂-eq per kg coal, is 4.4 times greater than the primary production CO₂-eq emissions of BFS, 1.55 kg CO₂-eq per kg iron. While the primary process resulting in FA production has higher emissions per kg than the process which results in BFS production, the mass fraction used to calculate allocated impacts for FA, 0.052, is only 24% of the mass fraction of BFS (0.22).

While the magnitude of PM₂.₅ and PM₁₀ emissions for the SCMs were small without allocation, when allocation was applied net emissions for the SCMs notably rise. Similar to CO₂-eq emissions, SF exhibited the largest change, increasing four orders of magnitude for ferrosilicon production sourced SF and three orders of magnitude for silicon production sourced SF. With allocation, BFS PM₂.₅ and PM₁₀ emissions increased by 750% and 1170%, respectively, and FA PM₂.₅ and PM₁₀ emissions increased by 180% and 430%, respectively.
emissions of 2.6–3.97 kg CO₂-eq, depending on the primary process. Botswana had notably high production, discussed issues regarding future supplies is the role of shutting down coal power plants, which will diminish production, these values are not representative of anticipated future supplies. Among the most commonly used SCMs studied relative to national cement production; several countries, including Mauritius, Swaziland, and (b), respectively). Coal combustion necessary to produce FA is reported in the most nations. The second most prevalent primary process was pig iron production (necessary for BFS), with ferrosilicon and silicon production (necessary for SF) limited to just a few countries. National BFS and FA production as a fraction of cement production vary widely compared to SF. This low production was coupled with high environmental impacts when mass allocation is applied: GHG emissions of 2.6–3.97 kg CO₂-eq, depending on the primary process. Botswana had notably high production of BFS/PC ratio of 0.406, reflecting the high production of BFS. 2015 production at 25 446 thousand tons [66]. Even with this large amount of cement produced, Japan had a notably high ratio in FA to cement due to the very low production of cement; Kazakhstan appeared to be a large producer of SF. This low production was coupled with high environmental impacts when mass allocation is applied: GHG emissions of 2.6–3.97 kg CO₂-eq, depending on the primary process. Botswana had notably high production of SF. This low production was coupled with high environmental impacts when mass allocation is applied: GHG emissions of 2.6–3.97 kg CO₂-eq, depending on the primary process. Botswana had notably high production of cement compared to cement. Japan provided a rare example of a nation with high SCM production and high cement production: Japan produced 54 830 kt of cement in 2015, which is 115% greater than the average for 2015 production at 25 446 thousand tons [66]. Even with this large amount of cement produced, Japan had a BFS/PC ratio of 0.406, reflecting the high production of BFS.

### 3.2. Global industrial-byproduct supplementary cementitious material production

As noted previously, SCMs are used in large quantities around the world; in 2015 mineral admixtures such as FA, GBS, SF, natural pozzolans, and limestone accounted for approximately 20.3% of the cementitious content for production reported to the GNR program [48]. The SCM production considered in this work was plotted relative to cement production (including PC and blended cement in 2015 as reported by the United States Geological Survey [66]) by weight of material produced based on country or locality data (figure 2). These ratios facilitate assessment of the ability to produce blended cements with limited transportation. Noting that it is typically assumed that only approximately 25% of the total FA produced can be used as a reactive pozzolan [67], two plots are presented for FA relative to cement production. Namely, one plot reflects total FA production and the other reflects the lower production if only 25% FA is considered viable (figures 2(a) and (b), respectively). Coal combustion necessary to produce FA is reported in the most nations. The second most prevalent primary process was pig iron production (necessary for BFS), with ferrosilicon and silicon production (necessary for SF) limited to just a few countries. National BFS and FA production as a fraction of cement production vary widely compared to SF. A few notable examples of national production: Norway had a relatively high ratio of SF production to cement production at 11%; Botswana exhibited a notably high ratio in FA to cement due to the very low production of cement; Kazakhstan appeared to be a large producer of all SCMs studied relative to national cement production; several countries, including Mauritius, Swaziland, Malta, Montenegro, Singapore, and Iceland produced SCMs even though they did not report cement production. For example, Iceland produced 6 thousand tons of FA and 43 thousand metric tons of BFS in 2015, but 0 metric tons of cement.

Focusing on the largest SCM producers around the world provides additional context for our potential utilization of these materials (figure 3). The top 10 cement producers in the world for 2015 did not produce enough SCMs to support high levels of replacement of PC in the industry. Although, among the countries with the top 5 SCM/PC ratios, shown in figures 3(b)–(d), there was potential for a considerable amount of replacement using BFS and FA. Globally, reactive FA could replace up to 2.6% of PC and BFS could replace up to 7.8% of PC. It must be noted, however, that this work is a single point in time. Due to variations in production, these values are not representative of anticipated future supplies. Among the most commonly discussed issues regarding future supplies is the role of shutting down coal power plants, which will diminish FA production [68]. For example, in the US, 25 gigawatts worth of coal capacity is expected to shut down by 2025 [69]. The production of SF was notably low; only 0.1% of global PC production could be replaced by SF. This low production was coupled with high environmental impacts when mass allocation is applied: GHG emissions of 2.6–3.97 kg CO₂-eq, depending on the primary process. Botswana had notably high production of cement compared to cement. Japan provided a rare example of a nation with high SCM production and high cement production: Japan produced 54 830 kt of cement in 2015, which is 115% greater than the average for 2015 production at 25 446 thousand tons [66]. Even with this large amount of cement produced, Japan had a BFS/PC ratio of 0.406, reflecting the high production of BFS.

### 3.3. The role of transportation and tipping points

The further SCMs are transported, the greater the emissions attributed to these constituents become. When emissions are allocated from primary products to industrial byproduct SCMs, they cannot be transported as far while maintaining environmental impacts lower than PC produced locally. To show the effects of transportation on cumulative impacts of the industrial byproduct SCMs studied in this work, tipping points (i.e., thresholds beyond which further transportation would lead to greater impacts
than PC) were determined for each type of SCM, with and without allocation, and for each mode of transportation considered. The rate at which transportation affects cumulative impacts are plotted in figure 4 for the three emissions studied. This figure shows the emissions attributed to the SCM as a ratio of PC emissions plotted against the distance of transportation for the SCM, where higher distances of transportation increase the impacts of the SCM until the impacts of the SCM exceed those of PC (at a value of 1—the tipping point). For example, if FA with allocation were to be transported 20000 km by ship (nearly one half the circumference of Earth), the ratio of its GHG emissions to non-transported PC would be approximately 8.5% of the emissions of PC. Notably, with allocation of impacts from primary processes, PM$_{2.5}$ and PM$_{10}$ emissions for BFS and all emissions for SF exceeded the manufacturing emissions of PC. As such, tipping points were not considered for these cases.

Due to varying emissions factors, the SCMs can be transported different distances depending on mode of transportation before exceeding the tipping point (table 6). Because of their high emissions, transportation by truck had the lowest distances possible before tipping points were reached. Transportation by ship allowed for the greatest distances in terms of GHG emissions but less distance for PM, and transportation by rail allowed for the greatest distances in terms of PM emissions though considerably lower for GHG. Noting that much transportation may occur in areas with limited populations, decisions may potentially be driven for which emission to mitigate based on anticipated inhalation factors. As expected, allocation altered the distance before a tipping point was reached for each of the SCMs studied. For transportation of FA with allocation, the tipping points were reduced by 16%–42% compared to when allocation was not considered. The tipping
Figure 3. (a) Comparison of cement production to total SCM production for the top 10 cement producers in the world and the rest of the world (b) comparison of cement production to 25% of FA production—estimated as the reactive fraction of production—for the countries with the top 5 FA/PC ratios (only 25% of FA production reported to reflect anticipated reactive fraction) (c) comparison of cement production to SF production for the countries with the top 5 SF/PC ratios (d) comparison of cement production to BFS production for the countries with the top 5 BFS/PC ratios.

point for transportation of BFS with allocation was 75% lower than BFS without allocation for CO2-eq emissions. The SF was able to travel the same as FA without allocation, and 21%–110% farther than BFS without allocation.

3.4. Factors affecting environmental justice

While production of GHG emissions is a global concern, production of PM emissions from concrete and its constituents is a local concern, particularly a concern for social justice. As a preliminary approach to this concern, the location of cement producers, RMC producers, ferrosilicon and silicon producers (noted together as (ferro)silicon producers), coal combustion facilities (noted as ‘coal’ producers), and iron producers were plotted for the continental US. Data were examined based on which counties contained one type or multiple types of producers (see figure 5). RMC is produced in the most counties, with 954 counties containing at least one RMC facility. In addition to the large number of RMC producers, figure 5 shows a high presence of coal electricity plants, which were present in 315 counties. Silicon and/or ferrosilicon have the lowest prevalence of production, appearing in only 8 of the counties. While there are several counties containing more than one type of industrial producer studied here, there was no single county that contained a combination of an RMC producer, an iron producer, and a (ferro)silicon producer. Additionally, there was only one county with RMC production, coal-derived electricity, and (ferro)silicon production, which is Jackson County, AL. The number of counties with cement and iron production were 81 and 93, respectively. Moreover, the Northeast and South regions (as defined by the 2010 US Census [70]) contain 43% of all reported coal electricity plants in 2015, while the West region only contains 9.7%.

To examine possible associations between locations of constituent and concrete production relative the locations of either low-income or minority populations, the DI VULEOPCT was plotted for the counties in the continental US (see figure 6). Because these data have higher granularity than the county-level, as plotted here, the lowest percentile reported for a county is shown in figure 6(a) and the highest percentile in figure 6(b); a higher percentile reflects a larger percentage of population belonging to low income and minority groups. There is notable variation of the DI within counties, as depicted by the color variation between the two plots.
Although it has been shown that for energy systems, strategies to cut GHG emissions can have significant co-benefits in reductions of local air pollutants [71], this may not be the case for mitigating GHG emissions through use of SCMs. Rather, it is possible that the use of byproduct SCMs correspond to industrial production of primary processes in areas with disadvantaged populations and commensurate human health implications, with mitigation benefits attributed to a different location where the SCMs are used. Examining the lower percentile of DI for counties in the US, there is little relationship between regions with low-income and/or minority populations with the industrial production considered herein. However, examining the upper percentile, patterns emerge in overlap between areas with low-income and/or minority populations and industrial production, particularly for counties in the southern half of the US.

Table 7 shows the percentage of counties with an upper percentile in each DI group that produce SCMs, RMC, or both and a comparison these percentages to the percentage of each DI group in all counties, referred to herein as a disproportionate impact. A proportion of 1.0 indicates the percentage of a DI subgroup in counties producing SCMs or RMC is equal to the percentage of that DI subgroup in the population (all counties). A proportion of less than 1 indicates the subgroup is underrepresented; a proportion of more than 1 indicates the subgroup is more prevalent than would be expected based on its portion of the total. There is clearly a disproportionate impact. While SCMs can come from natural resources or be imported from other countries, these findings suggest that there is likely a higher propensity for counties with low-income and/or minority populations to have industrial primary processes that negatively impact them while benefiting GHG emissions mitigation strategies in other areas. As to whether polluting industries initially locate in depressed communities or their presence lowers property values and attracts lower status residents, research indicates that both are true [25]. While this examination did not take into account size of counties, it is a preliminary indication of socio-economic disparity between where mitigation strategies alleviate burdens and where they may increase them.
Table 6. Tipping point thresholds beyond which additional transportation will lead to higher impacts for SCMs than PC, considering each type of SCM, with and without allocation, for each mode of transportation.

| Allocation Method | Mode | CO₂-eq | PM₂.₅ | PM₁₀ |
|-------------------|------|--------|--------|-------|
| No allocation, FA | Truck | 12 300 | 6650   | 7660  |
|                   | Train | 59 000 | 121 000| 158 000|
|                   | Ship  | 486 000| 32 300 | 35 600|
| Allocation, FA    | Truck | 70 900 | 5610   | 5660  |
|                   | Train | 33 900 | 102 000| 117 000|
|                   | Ship  | 279 000| 27 200 | 26 300|
| No allocation, BFS| Truck | 6760   | 5490   | 3670  |
|                   | Train | 32 300 | 99 900 | 75 700|
|                   | Ship  | 266 000| 26 600 | 17 100|
| Allocation, BFS   | Truck | 1670   | Values | Values |
|                   | Train | 8010   | exceed | exceed |
|                   | Ship  | 65 900 | PC     | PC     |
| No allocation, SF | Truck | 12 300 | 6650   | 7660  |
|                   | Train | 59 000 | 121 000| 158 000|
|                   | Ship  | 486 000| 32 300 | 35 600|
| Allocation, SF    | Truck | Values | Values | Values |
|                   | Train | exceed | exceed | exceed |
|                   | Ship  | PC     | PC     | PC     |

Figure 5. Map of the contiguous US states colored to indicate the type of production present in each county (iron in figure reflects iron and steel production).

4. Discussion

Methods to mitigate GHG emissions while minimizing harm on local populations can be informed by the factors studied in this work, namely: (a) the location and quantity of industrial byproduct SCM production; (b) the role of allocation methods and transportation on GHG and PM emissions attributed to byproduct SCMs; and (c) implications for production and emissions on environmental justice issues. Considering the relatively low emissions, even when considering allocation and transportation, FA was the typically lowest impact SCM. As such, scenarios are focused on regions with high FA production and potential for the transport of FA without exceeding impacts of producing new PC. See table 4 for a list of scenarios 1–6 to be discussed and table 6 for the tipping point values to compare with the travel distances. Scenario 1 was selected because Australia has a large proportion of FA to PC production, at 70% for full production, and 17% for the reactive FA (25% of total). The travel distance from Darwin, Australia to Shanghai, China is 54 times lower than the CO₂-eq emissions tipping point for FA with mass allocation, transported by ship. In scenario 2, still examining the large FA producing Australia, the travel distance from Brisbane, Australia to the Port of Los Angeles, US is 24 times lower than the CO₂-eq emissions tipping point for FA with mass allocation, transported by ship. Kazakhstan was shown to have excess production of FA; for scenario 3, the tipping point for CO₂-eq emissions of mass allocated FA transported by train is 474% greater than the distance from Nur-Sultan-Nurly Zhol, Nur-Sultan,
Kazakhstan to Shanghai, China. For the same cities, but considering a different mode of transportation (scenario 4), the tipping point for CO\textsubscript{2}-eq emissions associated of mass allocated FA transported by truck is still 20% greater than the 5904 km distance. In this work, Botswana was also noted as producing excess FA. In scenario 5, if FA originates from the Morupule Power Station in Botswana, the tipping point for PM\textsubscript{2.5} emissions with mass allocation and transportation by truck is 81% greater than the distance to Dar es Salaam, Tanzania. Scenario 6 considers Montenegro where 93 thousand metric tons of FA was produced in 2015; however, no cement production was reported [66]. Based on the tipping point values for PM\textsubscript{10} emissions with allocation for transport by truck, the FA could travel all the way to Samarkand, Uzbekistan before emission levels rise to near PC levels. These findings suggest there is great potential for the transport of some SCMs, while lowering cumulative emissions. It must be emphasized, the effects of air pollutant emissions on human health externalities would have to address magnitude and proximity of air emissions to populations, as well as intake fraction [23, 72], which should be addressed in future work.

Examining the implications of SCM production and transportation at a more granular scale facilitates analysis of the effects on the health of communities. In order to get FA to the coast of California, where the majority of construction in the state takes place, transport from the closest US coal electricity plant would
Table 7. Production of SCMs, RMC, both, and disproportionate impact of their production by upper percentile of the DI by county.

| Upper percentile DI | % of counties | Disproportionate impact | % of counties | Disproportionate impact | % of counties | Disproportionate impact |
|---------------------|--------------|-------------------------|--------------|-------------------------|--------------|-------------------------|
| > 0.8               | 33%          | 1.53                    | 38%          | 1.74                    | 50%          | 2.28                    |
| 0.6–0.8             | 26%          | 1.21                    | 27%          | 1.25                    | 28%          | 1.29                    |
| 0.4–0.6             | 26%          | 0.99                    | 21%          | 0.82                    | 19%          | 0.73                    |
| < 0.4               | 15%          | 0.48                    | 14%          | 0.45                    | 4%           | 0.12                    |

*Production of primary processes that could potentially lead to production of SCMs considered here.

A higher DI indicates a higher level of low income and minority residents.

Disproportionate impact is determined by the ratio of the percent of counties within a certain DI range that produce the materials to the total percent of counties within that same DI range; if there is no disproportionate impact, this ratio is equal to 1.

lead to transportation through multiple counties with high upper percentile low-income and/or minority populations. By introducing the additional emissions, the health of these communities could become more at risk. Noting that all ferrosilicon and silicon production in the US is performed on the East Coast, when impacts with allocation are considered, there is no ability to use SF anywhere on the West Coast and reduce impacts on a mass-replacement basis; in fact, with allocation, the emissions attributed to SF even with no transportation exceed PC impacts. Therefore, unless the SF not only replaces PC, but also leads to changes in performance that lowers the overall cementitious content or lowers the volume of concrete needed, this SCM would not facilitate emissions reductions. Iron production is present in several regions throughout the country, so if large enough quantities of BFS are produced, this SCM has higher potential to reduce CO2-eq emissions through partial replacement of PC. However, the high PM emissions from iron production result in BFS with allocation having cumulative PM emissions that exceed those of PC. Those emissions would occur in the community with iron production. Those PM emissions do not occur in the community claiming potential GHG emissions benefits of using the BFS in concrete. Coal electricity plants are also well distributed throughout the US with greater prevalence in the Northeast. Even with the relatively low impacts of FA, tipping points should still be considered to ensure the impacts of PC are not exceeded, and in future work, additional impacts should be considered.

5. Conclusion

This study derived the CO2-eq, PM2.5, and PM10 emissions from the production of three industrial byproduct SCMs, namely FA, SF, and BFS, with and without mass allocation of impacts from primary industrial processes. Data were compiled to determine the global production of these SCMs at the country/locality scale and compared to cement production. The tipping point thresholds for transportation by truck, train, and ship were determined to inform the distance beyond which the emissions of SCMs exceed those of producing PC. Finally, the locations of cement, RMC, and SCM production were compared to demographic indicators in the US to perform an initial assessment of some environmental justice factors associated with certain decisions in mitigating environmental impacts from concrete production.

Some key findings from this work include:

- Emissions allocation can have significant effects on total impacts attributable to SCMs. In the case of FA and SF, both SCMs were modeled with negligible GHG emissions prior to allocation. With allocation, the GHG emissions to produce these SCMs prior to transportation were 0.351 kg CO2-eq per kg FA and 2.60–3.97 kg CO2-eq per kg SF (depending on primary process) respectively. For GHG and PM emissions, allocation results in SF being higher impact than PC; for PM emissions, allocation results in BFS being higher impact than PC.

- There is regional disparity between production of industrial SCMs and cement; however, globally, the total production of industrial SCMs amounts to 10.5% of cement production (considering only reactive FA (i.e., 25% of total FA)).

- In order to best utilize the byproduct SCMs produced, transportation distances and modes of transportation should be considered to minimize impacts. The type of SCM used should benefit material performance, as well as reflect the constituents that are locally available.

- The location of industrial production sites and regions through which SCMs are transported could lead to disproportionate harm to low income and/or minority communities.

In future work, the quality of the SCMs in different regions, their contributions to material performance, and the functionality of lower quality SCMs should be considered in efforts to mitigate emissions. Further study should examine transportation trends for different SCMs as well as potential imbalances between...
export and import emissions for many countries [73]. Linking findings with costs assessments would elucidate whether there are economic benefits or repercussions from replacing PC with local SCMs. Finally, using higher granularity in demographic data could better inform disproportionate effects on certain populations and environmental justice concerns associated with damages from climate and health burdens.

Conflict of interest

The authors declare no competing financial interest.

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Data availability statement

The data that support the findings of this study are available upon reasonable request from the authors.

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