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Disproportionate impacts of building materials production facilities on neighboring communities

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Abstract:
The construction and building materials (CBM) production industries, such as cement, steel, and plastics that are responsible for a substantial share of global CO_2 emissions, face increasing pressure to decarbonize. Recent legislative initiatives like the United States (US) federal Buy Clean Initiative and the World Green Building Council's decarbonization plan for Europe highlights the urgency to reduce emissions during CBM production stages. However, there remains a gap in addressing the localized environmental and social impacts of these industries as well as a necessary understanding of how decarbonization efforts may change local impacts. This study introduces a framework for quantifying the disproportionate impacts (I_d) of 12 CBM production facility categories on communities of color and low-income demographics across the US. Using geographical and environmental data from the 2017 National Emissions Inventory (NEI), we assess these impacts at four spatial scales: census tract, county, state, and national. Results show that across all scales, many CBM production facilities impose disproportionate impacts. The geographical disproportionate impact ($I_{G,d}$) shows the greatest burdens at the broadest spatial scales, whereas the environmental disproportionate impact ($I_{E,d}$) indicates highest burdens at more localized levels. Based on this spatial understanding, we provide methods that can be implemented to support community engagement and mitigate damages to populations neighboring industrial materials manufacturing. These findings offer valuable insights into the relationship between facility locations, emissions, and demographic groups, providing a basis for more targeted environmental justice policies aimed at mitigating these disproportionate impacts.

Keywords: Environmental justice; air pollution; Spatial modeling; Localized burdens; Decarbonization.

1. Introduction

The buildings and construction sector accounts for an estimated 37% of global energy and process-related CO_2 emissions[1]. A notable amount of these emissions is released during the materials production stage (sometimes referred to as the embodied carbon). As a result, recent decarbonization policies aim to significantly impact the high greenhouse gas (GHG) emitting industries within this sector. For example, in the United States (US), executive order 14057 launched a federal Buy Clean Initiative to prioritize use of lower-carbon construction materials[2], and the World Green Building Council (WorldGBC) launched a plan for the European Union to decarbonize the buildings and construction industries with an emphasis on embodied carbon[3]. Policies for climate change mitigation tied to fuel and energy use (e.g., for fuel standards in California[4]) have been reported to have co-benefits, showing improvements in impacts to air quality that can reduce burdens to local communities near the associated combustion sites.[5] The interlinkages between decarbonization efforts and localized burdens must integrate Environmental Justice (EJ). EJ, according to the US Environmental Protection Agency (USEPA), is the "fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income, with respect to the development, implementation, and enforcement of environmental laws, regulations, and policies." Although environmental injustice and activism have been taking place for many years throughout the world, the formal EJ movement in the US began around the 1980s.[6] As the building material sectors progress in their decarbonization efforts, it is crucial to examine EJ impacts concurrently to avoid unintended consequences or worsen disproportionate impacts on specific, localized communities. Yet while EJ impacts from industries like transportation[7–10] and energy production[4,11–14] are widely reported, the EJ for the building materials sector is not.

Materials-production GHG emissions are driven by a combination of energy-derived sources (e.g., from combustion of fuels) and process-derived sources (e.g., through chemical conversion and material handling). For example, the production of Portland cement (referred to herein as cement), a hydraulic binder used to make construction materials like concrete and mortar, results in an estimated 7% of global anthropogenic CO_2 emissions annually.[15] The emissions from cement production are a result of using fossil fuels in the cement kilns and process-derived CO_2 emissions from calcination (in which limestone is decarbonated to create a reactive calcium compound for the formation of calcium silicates in cement),[16] as well as the enormous amount of consumption of cement, in excess of 4 billion metric tons (Gt) annually.[17] Other popular construction materials also have varied emissions sources. Steel production, of which over 50% goes into the built environment,[18] is responsible for another estimated 7% of anthropogenic CO_2 emissions[19]. It has CO_2 emissions from energy resources, as well as process emissions from the use of coal as a reducing agent and from limestone decarbonation (as lime is used as a flux to remove impurities steel alloys).[20] For plastics production, which contributes over 3% of global GHG emissions[21] and where nearly 20% of production is for construction use[22], there are energy-derived emissions as well as emissions from processes and other factors, such methane leakage.[23] As a result, there is a global burden from materials-derived GHG emissions, and decarbonization efforts for many building materials must tackle both energy-derived and process-derived emissions.

In addition to global GHG emissions, construction and building materials (CBM) production are also responsible for local environmental burdens, such as air pollution related particulate matter (PM) with diameter less than 2.5 μm ($PM_{2.5}$) and PM with diameter less than 10 μm (PM_{10}) – herein referred to jointly as PM emissions, heavy metals exposure, and localized resource scarcities. Exposure to PM emissions can cause a wide range of diseases, which impacts quality of life, and currently contributes to millions of premature deaths annually.[24,25] Heavy metals exposure can similarly cause human health issues; however, for heavy metal exposures there are commonly concerns associated with a number of neurological, cardiac and other diseases[26]. Unlike CO_2 emissions, PM and metal emissions are more likely to be indicators of localized burdens of a production facility, and there can be a range of impacts on neighboring communities depending on factors such as degree of exposure and underlying health issues.[27] And just as with CO_2 emissions, these emissions are not only driven by energy resources, there are process-derived impacts as well. Quarrying activities, which are necessary for most conventional mineral-based materials, can produce PM emissions[28], as well as other impacts, including altering land use and creating overburden waste[29]. Metal mining and smelting activities can lead to gaseous emissions, solid waste, and wastewater containing heavy metals.[30] And the chemicals used in the production of materials like plastic can release processing compounds with significant burden of disease to exposed populations.[31] There are many other forms of ecosystem damages that can accrue from industrial production facilities, but among the more unique issues for materials production are localized resource scarcities, in which expected demand for a resource is greater than its local availability. Such impacts have been noted for common resources like sand and water.[32,33] Further, because quarrying activities are needed for raw material acquisition, there are a series of quarrying impacts that can occur, including altering land use and creating overburden waste.[29]

The impacts from manufacturing on a local level has been shown to systematically cause disproportionate impact on historically marginalized communities.[34] In the US, studies have shown an inequality and disproportionality in exposure to PM emissions for particular racial groups compared to others.[34,35] For instance, Greer et al. 2024[36] utilized systems analysis to address air quality planning in the Port of Oakland, highlighting the environmental justice concerns associated with industrial operations near vulnerable populations. Furthermore, Greer et al. 2022[37] found that pavement resurfacing and transportation supply chains are significant contributors to $PM_{2.5}$ exposure in urban areas, providing evidence of the localized air pollution burdens tied to specific industrial activities. The effects from historical practices, such as redlining, have resulted in impacts to present-day health risks and outcomes.[38] However, current frameworks to improve environmental sustainability do not always promote EJ.[39] Most EJ research has focused on social implications;[8] the emphasis of such studies when examining materials production and demand have typically tied to planning decisions in specific locations (e.g., Ezeugoh *et al.* 2020[40] and De Sousa Silva *et al.* 2018[41]). Switches in technology to decarbonize CBM can alter such parameters as well as potentially create new emissions to air, water, soil, and waste depending on the resources used. Some innovations in decarbonizing other sectors, such as electricity generation from renewables instead of fossil resources,[42] have contributed to co-benefits in reducing health impacts.[43] Yet initial studies suggest co-benefits may not be as consistent from decarbonization methods for CBM due to the combination of process and energy-derived impacts as well as factors such the need for large quantities of resource consumption, such as the disproportionate burdens from using industrial byproducts in concrete to lower GHG emissions.[44]

In this work, we develop a method framework to measure both the geographic disproportionate impact ($I_{G,d}$) and environmental disproportionate impact ($I_{E,d}$) of CBM production facilities within the US. Our analysis focuses on two key demographic indicators: people of color and people considered low-income. We apply this method at four spatial scales to determine the disproportionate impact relative to demographics within the (a) census tract (b) county (c) state and (d) nation. We investigate 12 CBM categories based on the North American Industry Classification System (NAICS), mapping their production facilities and calculating both geographical and emissions-based (environmental) disproportionate impacts for each category, referred to herein as $I_{G,d}$ and $I_{E,d}$, respectively. The study further explores how $I_{G,d}$ and $I_{E,d}$ values vary across spatial scales and in relation to demographic indicators. We identify trends that highlight localized and broader impacts. We synthesize key additional analysis methods that can be paired with this form of spatiotemporal analysis to understand effects to localized communities. Finally, we detail limitations within this study and provide suggestions for future work to apply this framework.

2. Methods

2.1 Data sources

For this study, we leverage 2017 National Emissions Inventory (NEI) point data summaries[45] to identify CBM facilities which release emissions monitored by the USEPA, namely criteria air pollutants[46]. The CBM categories are created by organizing 2017 NAICS codes[47] based on construction material type (see Supplemental Methods for detailed explanation of categories), and these 2017 NAICS codes are used to match the NEI datasets. The 12 CBM categories investigated in this study include: (a) wood, (b) asphalt, (c) plastics and rubber, (d) clay products, (e) glass, (f) cement, (g) concrete, (h) lime, (i) gypsum, (j) iron and steel, (k) alumina and aluminum, and (l) non-ferrous metals. Regional demographic information (e.g., people of color and people of low-income) at the census block group level is collected from the US Census Bureau's American Community Survey (ACS) 5-year summary (2017-2021)[48]. The Demographic Index (DI) is defined as the average between the percentage of people of color and percentage of people considered low-income in a region, based on methods from the USEPA's Environmental Justice Screening and Mapping tool (EJScreen) 2.2[49] (**Eq 1**). In this work, all mention of demographic groups specifically refers to people of color and people considered low-income, which are jointly categorized as DI.

$$DI = \frac{\%\, people\ of\ low\ income + \%\, people\ of\ color}{2} \qquad \text{(Eq. 1)}$$

2.2 Disproportionate impact equation

Disproportionate impact (I_d) is determined by applying proportionality indices, which account for both geographic and environmental disproportionality. These indices provide insights into how CBM facilities affect specific demographic groups, such as people of color and low-income people, either by the location of the facilities or the emissions produced by these facilities. It is important to note that our approach to calculating I_d is agnostic of specific demographic assumptions. This means any population, regardless of demographic composition, may experience disproportionate impacts based on the concentration of facilities and emissions exposure in their area; we note that any location without residents reported in the US Census is not included in this study.

2.2.1 Proportionality indices and calculation framework

Both geographical and environmental I_d are assessed by calculating the ratio of subgroup's representation in areas affected by CBM facilities (by location or emissions) to their overall representation in the total population. In this work, a region of analysis is defined by a Census Block Group (CBG), as it is the smallest geographic area with demographic information from the US Census Bureau's ACS 5-year summary[50]. As such, this study determines an I_d value for each CBG in the US that has a CBM facility. The proportionality index, I_d, is determined using the following relationship:

$$I_d = \frac{F_i}{F_{total}} \quad \text{(Eq. 2)}$$

Where F_i is the percentage of CBGs with a specified CBM category affecting a particular DI range, and F_{total} is the total in the outcome group for that DI range. A ratio greater than 1 indicates disproportionate impact (i.e., a subgroup is more prone to being affected by a facility), while a ratio less than 1 suggests that the subgroup is less affected. This analysis is conducted at each of the four considered spatial scales – census tract, county, state, and national.

2.2.2 Geographical disproportionality (I_d by location)

Geographical disproportionate impact ($I_{G,d}$) focuses on the spatial distribution of CBM facilities in relation to the demographics of people living in those areas. Namely, $I_{G,d}$ is an indicator of the disproportionate burden associated with CBM facilities being located in areas with people of color and people considered low-income (i.e., agnostic of the quantity of environmental burden). This approach prioritizes analyzing spatial distribution of facilities rather than on their type or quantity of emissions. We apply this spatial analysis method because it captures the overall impact of individual production facilities, as well as the cumulative effects of multiple facilities located in close proximity. The $I_{G,d}$ is determined using the following relationship:

$$I_{G,d} = \frac{F_{G,i}}{F_{G,total}} \quad \text{(Eq. 3)}$$

where $F_{G,i}$ is defined as subgroup i of those in a particular outcome for a specified CBM category and DI range (i.e., subgroup i is a dependent on CBM category and DI range). Here, we address variation in demographic groups, broken down by 10% increments, where DI is between [0, 0.1], (0.1, 0.2], (0.2, 0.3], (0.3, 0.4], (0.4, 0.5], (0.5, 0.6], (0.6, 0.7], (0.7, 0.8], (0.8, 0.9], (0.9, 1]. Then, we define percent of regions with a selected CBM industry (e.g., of all regions with a given industry, those with 60%-70% low-income persons). This ratio can be calculated as:

$$F_{G,i} = \frac{F_{G,Industry,i}}{total\ number\ of\ CBGs\ with\ that\ CBM\ industry\ within\ the\ region} \quad \text{(Eq. 4)}$$

where $F_{G,Industry,i}$ = the number of CBGs with a specified CBM industry in a particular DI range in a region (e.g., census tract, county, state, nation). $F_{G,total}$ = total CBGs in a particular DI range broken down by 10% increments (e.g., of all regions, those with 60%-70% low-income persons). This ratio can be calculated as:

$$F_{G,total} = \frac{F_{G,DI,i}}{total\ number\ of\ CBGs\ within\ the\ region} \quad \text{(Eq. 5)}$$

where $F_{G,DI,i}$ = the number of instances with a particular DI range (corresponding to the range for $F_{G,Industry,i}$, but inclusive of all CBM industries). So, when calculating the $I_{G,d}$ the numerator represents how (much or little) a particular CBM industry is impacting regions within a particular DI range, while the denominator describes the occurrence of all CBM facilities in this particular DI range relative to all DI ranges within the region.

2.2.3 Environmental disproportionality (I_d by emissions)

The environmental disproportionate impact ($I_{E,d}$) reflects the emissions from the CBM facilities on demographic groups. Namely, $I_{E,d}$ is the disproportionate burden associated with pollutants released into the environment from CBM facilities within regions with demographic groups (i.e., an indicator of environmental burden). Similar to the geographical index, this ratio examines the extent to which emissions from the facilities considered disproportionately affect certain groups. The $I_{E,d}$ will be determined using the following relationship:

$$I_{E,d} = \frac{F_{E,i}}{F_{E,total}} \quad \text{(Eq. 6)}$$

Where $F_{E,i}$ is defined as subgroup *i* of those in a particular outcome for a specified CBM category and DI range. This parameter is used herein to represent the percent of CBGs with a selected emissions from a selected CBM category affecting a particular DI range. Again, we address variation in demographic groups, broken down by 10% increments, where DI is between [0, 0.1], (0.1, 0.2], (0.2, 0.3], (0.3, 0.4], (0.4, 0.5], (0.5, 0.6], (0.6, 0.7], (0.7, 0.8], (0.8, 0.9], (0.9, 1]. This ratio can be calculated as:

$$F_{E,i} = \frac{F_{E,Industry,DI,i}}{total\ emissions\ in\ regions\ with\ that\ CBM\ industry} \quad \text{(Eq. 7)}$$

Where $F_{E,Inudstry,DI,i}$ = sum of E_i (the selected emissions from a specified industry) in a particular DI range in a region (e.g., census tract, county, state, nation), and $F_{E,total}$ = total emissions affecting a particular DI range broken down by 10% increments (e.g., of all regions, those with 60%-70% low-income persons). This ratio can be calculated as:

$$F_{E,total} = \frac{F_{E,DI,i}}{total\ emissions\ within\ the\ region} \quad \text{(Eq. 8)}$$

Where $F_{E,DI,i}$ = the amount of a type of emissions within the region, affecting a particular DI range corresponding to the range for $F_{E,Inudstry,DI,i}$, but inclusive of all CBM industries. It should be noted that any emissions type can be used when calculating $F_{E,i}$ and $F_{E,total}$; however, the unit of emissions used in this study is short tons (herein, references of tons refer to short tons; metric tons will be explicitly labeled) of $PM_{2.5}$ emissions per year.

2.2.4 Example calculation for environmental disproportionality ($I_{E,d}$)

Similar to $I_{G,d}$, when calculating $I_{E,d}$, the numerator represents the emissions impact of a particular CBM industry within a specified DI Range, while the denominator describes the emissions from all CBM facilities within that particular DI range relative to emissions in all DI ranges within the region.

To provide an example of implementing these equations, consider calculating the $I_{E,d}$ for plastics & rubber facilities at the county level (region of analysis), for CBGs with DI range 4 (i.e., the average between the percentage of people of color and percentage of people considered low-income is 30-40%). Suppose this county has 100 CBM facilities emitting a total of 10,000 tons of $PM_{2.5}$ (i.e., the denominator in the $F_{E,total}$ calculation = 10,000 tons), and this county has 10 plastics & rubber facilities emitting a total of 1,000 tons of $PM_{2.5}$. (i.e., the denominator in the $F_{E,i}$ calculation = 1,000 tons). Of these, three facilities are in DI range 4, and they are emitting 200 tons of $PM_{2.5}$ (i.e., the numerator in the $F_{E,i}$ calculation, $F_{E,Industry,DI,i}$, = 200 tons). Across all CBGs in DI range 4 (inclusive of all industries) there are 15 CBM facilities emitting a total of 5,000 tons of $PM_{2.5}$ (i.e., the numerator in the $F_{E,total}$ calculation, $F_{E,DI,i}$, = 5,000 tons).

In this $I_{E,d}$ calculation, the numerator captures the 200 tons of emissions from these three plastics & rubber facilities in DI range 4, relative to the emissions of all plastics & rubber facilities in the county. The denominator captures the total 5,000 tons emitted by all 15 CBM facilities in DI range of 4, relative to the emissions from all 100 CBM facilities within the county. The $F_{E,i}$ calculation equates to 0.2 and the $F_{E,total}$ calculation equates to 0.5. The result ($I_{E,d}$ = 0.4) indicates that plastics & rubber facilities are not disproportionately located (or more prone to exist) in areas with DI range 4 compared to other DI ranges in the county.

3. Results

3.1 Construction building material facility locations and Demographic Index (DI) maps

To visualize the regions of interest, **Figure 1a** shows the number CBM facilities per US county; an individual map of facilities for each of the 12 CBMs is provided in the Supporting Information (Figure S1). In 2017, the region with the highest number of CBM facilities was Cook County in Illinois (162 facilities), followed by Los Angeles County in California (104 facilities), and San Joaquin County in California at 62 facilities. **Figure 1b** displays the total tons of $PM_{2.5}$ emissions per US county released by CBM facilities. Here, the region with the highest total $PM_{2.5}$ emissions was Hamblen County in

Tennessee (4688 tons) entirely due to one plastics & rubber facility in the region. This is followed by Lake County in Indiana (2415 tons) due to multiple iron & steel facilities in the county. Jefferson County in Alabama has the third highest emissions with 1598 tons, again due to multiple iron & steel facilities in the county. **Figure 1c** presents the average DI among all CBGs (including those which do not have CBM facilities present) for each US county, using demographic data from the ACS 5-year summary for years 2017-2021. Any CBM facility located in a CBG with no residents is not included in this study. Among all CBGs where at least one CBM facility exists, the US average DI value is 33% (slightly lower than the US average for all CBGs, which is 35%). Among all CBGs included in this study, for state averages, Arizona has the highest county DI, with the mean and median at 54% and 55%, respectively. Conversely, New Hampshire records the lowest average county DI, with both the mean and median at 8%. Nottoway County in Virginia experiences the maximum value for county level DI (i.e., average DI of all CBGs per county) in the US at 100%. This peak is due to the only CBM facility in the entire county being in a CBG where 100% are people of color and people considered low-income. Union County in Ohio experiences the lowest-value county level DI at 2%, where two CBM facilities exist.

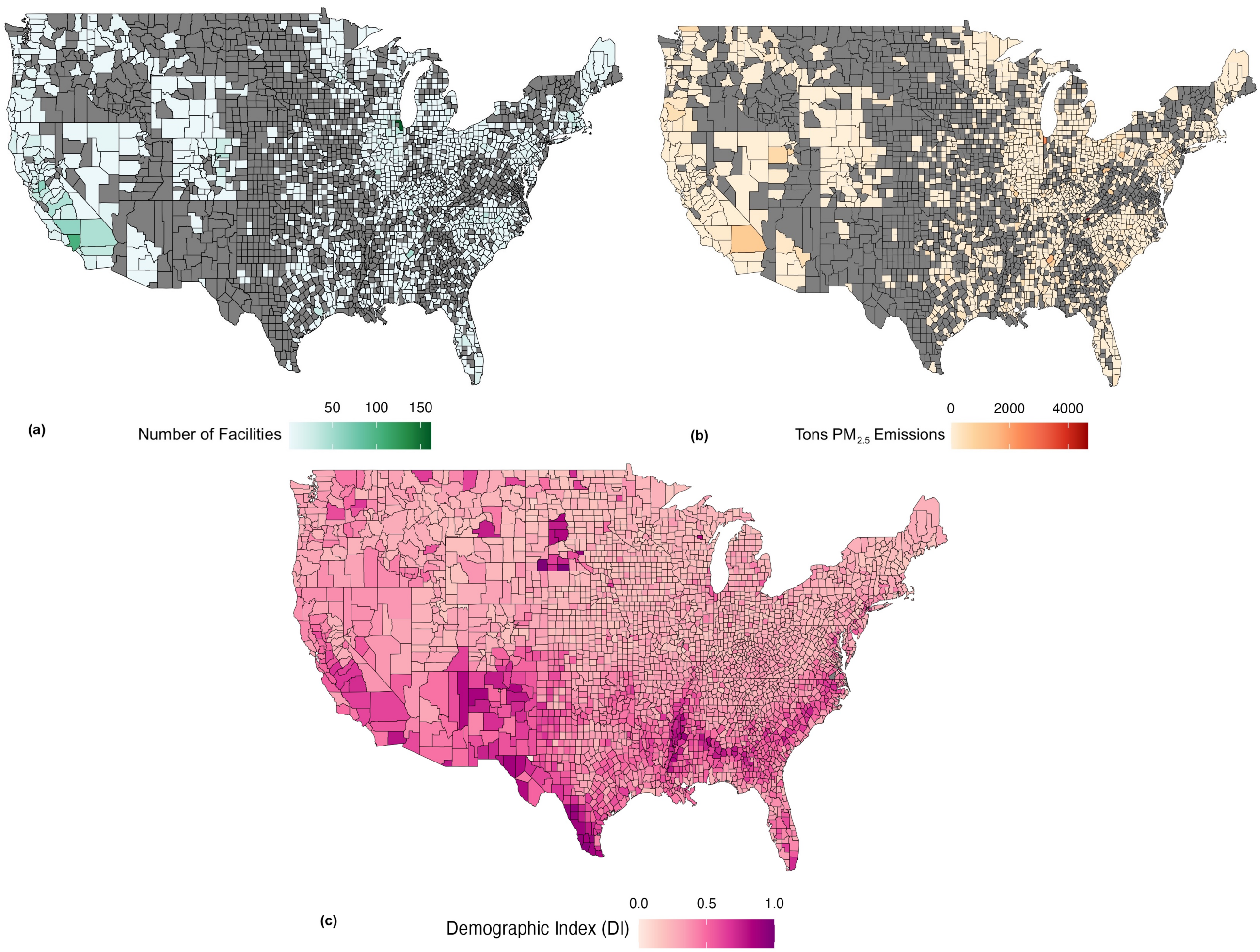

Figure 1. (a) Count of Construction and Building Materials (CBM) facilities per county in the contiguous United States and **(b)** total particulate matter with diameter less than 2.5 µm ($PM_{2.5}$) emissions per county for CBM facilities in the United States **(c)** average percentage of people of color and people considered low-income per county (i.e., Demographic Index (DI)) in the United States, including all Census Block Groups (CBGs).

3.2 Production emissions for each construction building material category

Figure 2 displays the GHG emissions and $PM_{2.5}$ emissions released by CBM production facilities in the US, reported by the NEI in 2017. The cement industry exhibits the highest GHG emissions contributing over 70 million tons in 2017, followed closely by iron & steel (~67 million tons). These two industries are significant contributors of global anthropogenic GHG emissions, due to both the emissions from their production and the magnitude of production. The plastics & rubber and lime industries also demonstrate substantial GHG emissions (both ~20 million tons). Emissions from glass, wood, and alumina & aluminum industries are relatively lower in GHG emissions comparing to the top contributors (~8 million tons, 6 million tons, and 4 million tons, respectively).

However, the industries with the highest GHG emissions do not always correlate with the highest $PM_{2.5}$ emissions. For instance, the cement industry releases the largest GHG emissions but has notably lower $PM_{2.5}$ emissions compared to magnitudes reported for iron & steel and wood. For $PM_{2.5}$ emissions, iron & steel exhibits the highest amount (~25 thousand tons), followed by wood (~20 thousand tons). Plastics & rubber and cement industries are next and release ~10 thousand tons, respectively. These findings emphasize the importance of monitoring differing environmental pollutants to perform comprehensive analyses. Further, the data underscores the need for targeted interventions aimed at reducing specific emissions from key industries.

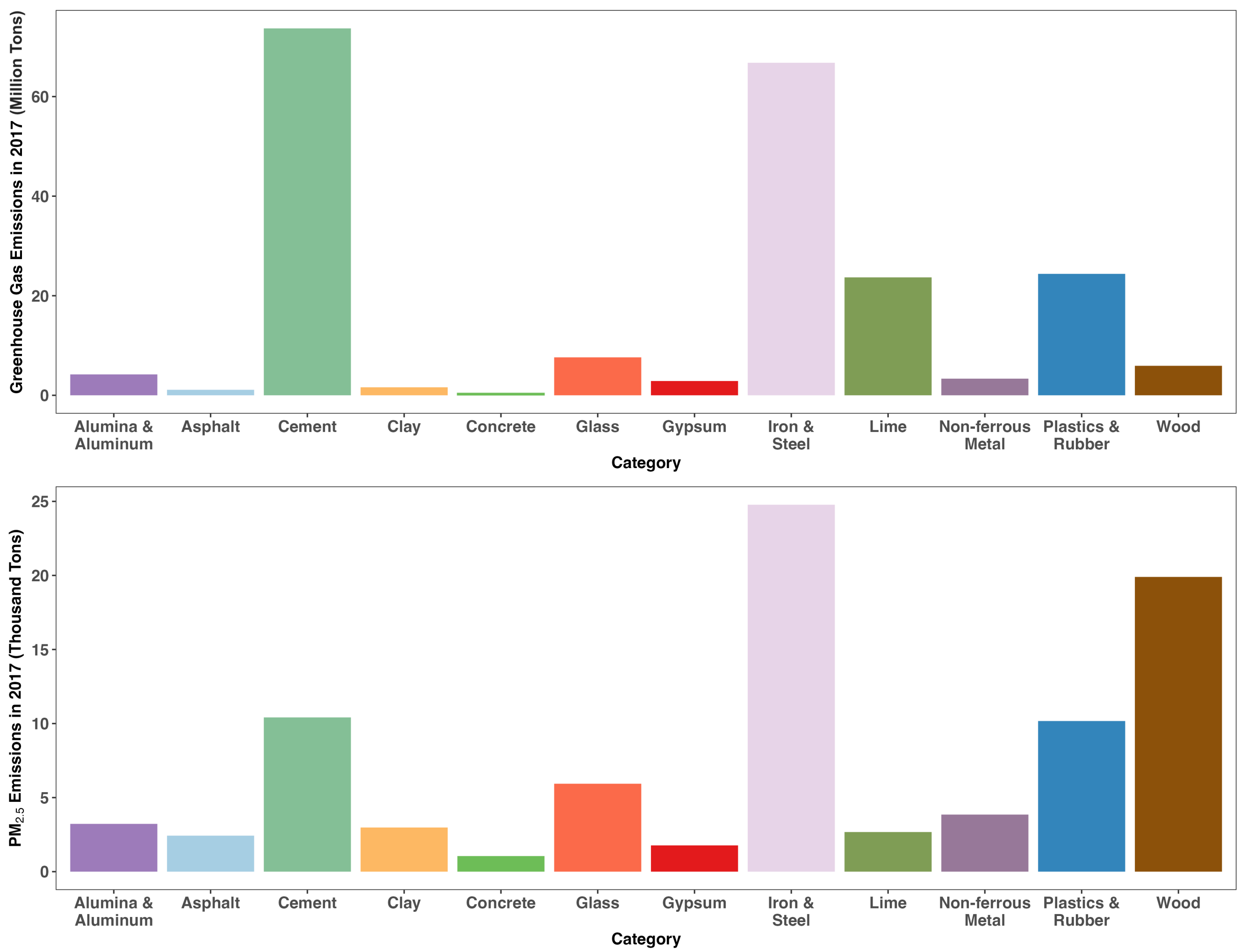

Figure 2. Comparison of greenhouse gas (GHG) emissions and particulate matter with diameter less than 2.5 µm ($PM_{2.5}$) emissions by Construction Building Material (CBM) category in the United States in 2017.

3.3 Geographical Disproportionate Impact (I_d) for each construction building material category

To compare results across CBM categories, **Table 1** displays the range of mean $I_{G,d}$ values for each CBM category relative to those demographic groups within (a) the census tract (b) the county (c) the state and (d) the nation. Across all four spatial scales, CBM facilities tend to be located in regions that impose disproportionate impacts on people of color and/or low-income, i.e., with $I_{G,d} > 1$ indicating that a subgroup is more prone to being affected by a facility.

At the census tract level, the overall mean $I_{G,d}$ value across all CBM categories is 1.08, indicating that facilities are slightly more likely to be located in areas that impose disproportionate burdens relative to census tract level demographic groups. While each CBM has a different mean $I_{G,d}$, the values all exceed 1.0, indicating a disproportionality (with the spread of CBM category mean geographic disproportionate impact showing lime facilities with the smallest value of 1.04, up to alumina & aluminum facilities having a mean $I_{G,d}$ value of 1.14). The low variability in the $I_{G,d}$ values at this scale, with a hierarchical average of 1.08, suggests that most CBM categories impose a similar level of disproportionate burdens across respective census tracts.

At the county level, the overall mean $I_{G,d}$ value is 2.22, again with the mean exceeding 1.0 for all CBMs assessed. Cement, non-ferrous metal, and alumina & aluminum facilities exhibit the highest ratios in disproportionate impact calculations, namely, 3.07, 2.71, 2.69, respectively. These higher ratios reflect both the localized burden and the concentration of these facilities within particular counties, which increases at this larger spatial scale. In contrast, concrete facilities exhibit the lowest ratio 1.53, which is a function of their geographical spread resulting in less disproportionate impact relative to the population. Yet, notably, despite this broader geographic distribution, the mean remains above 1.0, indicating a disproportionate impacted on the populations studied. The hierarchical average of 2.22 at this scale indicates higher variability and localized clustering of impacts.

For the state level, the overall mean $I_{G,d}$ value across CBM categories is 2.44. Lime and gypsum facilities stand out with the highest $I_{G,d}$ values of 5.26 and 4.48, respectively. However, plastics & rubber and concrete facilities, with $I_{G,d}$ values around 1.22 and 1.17, respectively, are more evenly distributed relative to state-level demographic averages.

Finally, at the national level, the $I_{G,d}$ ratios are even larger, with an overall mean $I_{G,d}$ of 9.64. The highest impacts are observed for lime (11.40), alumina & aluminum (10.21), and cement (10.16) facilities, which are concentrated in regions where demographic groups are significantly overrepresented compared to the national population. The smallest $I_{G,d}$ values, while still substantially above 1, are seen for plastics & rubber (8.78) and concrete (8.79). This increase in $I_{G,d}$ values at the national level is largely due to the cumulative count of CBGs across the nation, which average out more localized effects observed at smaller scales.

These findings highlight the importance of spatial scale in the analysis of disproportionate impacts. At finer scales, such as the census tract and county levels, more localized impacts are captured. At broader scales, such as state and national levels, a more general distribution of disproportionate burdens is observed, with increasing $I_{G,d}$ values due to the aggregation of demographic data across wider geographic regions.

Table 1. Geographical Disproportionate Impact ($I_{G,d}$) across four spatial scales (census tract, county, state, and nation) for various Construction Building Material (CBM) Categories.

Geographical Disproportionate Impact ($I_{G,d}$)								
	Census Tract		**County**		**State**		**Nation**	
	Category	**Mean**	**Category**	**Mean**	**Category**	**Mean**	**Category**	**Mean**
1	Alumina & Aluminum	1.139	Cement	3.073	Lime	5.255	Lime	11.40
2	Non-ferrous Metal	1.127	Non-ferrous Metal	2.712	Gypsum	4.485	Alumina & Aluminum	10.21
3	Wood	1.088	Alumina & Aluminum	2.686	Alumina & Aluminum	3.020	Cement	10.16
4	Iron & Steel	1.078	Clay	2.545	Cement	2.906	Glass	9.787
5	Asphalt	1.077	Glass	2.531	Glass	2.665	Gypsum	9.763
6	Glass	1.072	Gypsum	2.184	Clay	2.343	Non-ferrous Metal	9.592
7	Concrete	1.071	Iron & Steel	2.079	Non-ferrous Metal	2.049	Clay	9.571
8	Plastics & Rubber	1.065	Lime	2.079	Iron & Steel	1.672	Iron & Steel	9.297

9	Cement	1.065	Asphalt	1.907	Asphalt	1.288	Asphalt	9.181
10	Clay	1.061	Plastics & Rubber	1.676	Wood	1.251	Wood	9.161
11	Gypsum	1.040	Wood	1.670	Plastics & Rubber	1.227	Concrete	8.791
12	Lime	1.036	Concrete	1.527	Concrete	1.171	Plastics & Rubber	8.780
	Hierarchical Average	**1.077**	**Hierarchical Average**	**2.222**	**Hierarchical Average**	**2.444**	**Hierarchical Average**	**9.641**

Relative to national demographics, the overall mean I_d value is significantly greater than both the county and state analysis at 48.1.

3.4 Environmental Disproportionate Impact (I_d) for each construction building material category

The $I_{E,d}$ assess the effect of actual emissions from CBM facilities on demographic groups, and the results at all four spatial scales are displayed in **Table 2**. Here, with the exception of plastics & rubber at the national level, all CBM facilities tend to impose an exposure-based disproportionate impact to local populations.

At the census tract level, the overall average $I_{E,d}$ for all categories is 2.93, indicating that certain facilities are disproportionately contributing to localized environmental burdens of $PM_{2.5}$ emissions. The highest $I_{E,d}$ is observed for plastics & rubber (6.37), concrete (6.22), and clay (6.05), suggesting that these facilities have significant emissions burdens at the smallest geographic scale on average. Conversely, even the lowest mean $I_{E,d}$ values, i.e., those for cement (1.03), lime (1.03), and iron & steel (1.04), exceed 1.0, indicating that on average, each group of CBM facilities results in disproportionate impacts at the census tract level.

At the county and state levels, the average $I_{E,d}$ values grow. At the county level, the overall average $I_{E,d}$ is 21.46, with notably high ratios for plastics & rubber (77.63), wood (42.03), and non-ferrous metal (36.50). Lower $I_{E,d}$ is values, e.g., gypsum and glass with 2.22 and 4.79, respectively, indicate relatively even emissions distributions among the populations within this scale on average. For the state level, the overall average $I_{E,d}$ is 7.12. Clay has the highest state-level $I_{E,d}$ (38.10), followed by plastics & rubber

(10.67) and concrete (5.78). However, the state-level average wood facilities (1.48) and iron & steel (1.94) still exceed 1.0.

Finally, at the national level, the average $I_{E,d}$ noticeably drops to 1.13, indicating a much more balanced distribution of emissions when viewed across the entire country. Here, alumina & aluminum (1.28), non-ferrous metal (1.21), and lime (1.19) exhibit the highest $I_{E,d}$ values at the national scale, though these values are substantially lower compared to the finer geographical scales. However, plastics & rubber, which has the highest $I_{E,d}$ at the county level, has the lowest national-level $I_{E,d}$ (0.94). At a $I_{E,d}$ less than 1.0, the average national-level plastics & rubber emissions are the only average category examined that do not contribute to a disproportionate impact. This trend suggests that although the emissions from plastic & rubber facilities disproportionately affect certain communities at local scales, their overall impact is less pronounced when averaged nationally.

The $I_{E,d}$ results reveal trends based on the spatial scale of analysis. At finer scales, such as census tract and county, materials like plastics & rubber, concrete, and clay have a notable effect on local communities. However, as the scales become greater in size to the state and national levels, the average $I_{E,d}$ values tend to decrease, reflecting a more equitable emissions distribution across broader regions. This pattern further highlights the importance of considering multiple spatial scales when assessing environmental justice impacts. For example, plastics & rubber shows high emissions impacts at finer scales that may not be evident at broader scales, where emissions become more evenly distributed across the population. These findings suggest that interventions (e.g., policy, industry) aimed at mitigating environmental justice impacts should be tailored to particular geographic scales. As smaller scales, targeted strategies may be necessary to address severe disproportionate impacts experienced by certain communities near specific CBM facilities. At larger scales, interventions focusing on overall emissions reductions may be more appropriate.

Table 2. Environmental Disproportionate Impact ($I_{E,d}$) across four spatial scales (census tract, county, state, and nation) for various Construction Building Material (CBM) Categories.

Environmental Disproportionate Impact ($I_{E,d}$)								
	Census Tract		County		State		Nation	
	Category	**Mean**	**Category**	**Mean**	**Category**	**Mean**	**Category**	**Mean**
1	Plastics & Rubber	6.372	Plastics & Rubber	77.63	Clay	38.10	Alumina & Aluminum	1.279
2	Concrete	6.217	Wood	42.03	Plastics & Rubber	10.67	Non-ferrous Metal	1.210
3	Clay	6.052	Non-ferrous Metal	36.50	Concrete	5.776	Lime	1.190
4	Non-ferrous Metal	3.708	Asphalt	30.50	Gypsum	5.644	Asphalt	1.186
5	Asphalt	3.284	Concrete	20.83	Lime	5.396	Clay	1.181
6	Wood	2.318	Alumina & Aluminum	10.24	Alumina & Aluminum	4.307	Wood	1.145
7	Alumina & Aluminum	1.998	Lime	10.13	Non-ferrous Metal	3.929	Concrete	1.101
8	Glass	1.059	Clay	8.523	Asphalt	3.542	Iron & Steel	1.097
9	Gypsum	1.057	Iron & Steel	8.108	Glass	2.616	Gypsum	1.094
10	Iron & Steel	1.038	Cement	5.986	Cement	2.030	Cement	1.069
11	Lime	1.033	Glass	4.795	Iron & Steel	1.943	Glass	1.032
12	Cement	1.026	Gypsum	2.223	Wood	1.484	Plastics & Rubber	0.938
	Hierarchical Average	**2.930**	**Hierarchical Average**	**21.46**	**Hierarchical Average**	**7.120**	**Hierarchical Average**	**1.127**

3.5 Comparison of geographical disproportionate impact ($I_{G,d}$) and environmental disproportionate impact ($I_{E,d}$)

This analysis calculates two forms of disproportionate impacts, geographical and environmental, across four spatial scales (census tract, county, state, nation). A comparison of these two metrics shows key similarities and differences that reflect both the geographical distribution of facilities and the actual emissions burdens from CBM industries. At finer scales, such as the census tract and county levels, the differences between $I_{G,d}$ and $I_{E,d}$ are large, with $I_{E,d}$ higher for certain materials like plastics & rubber and concrete. As the scale grows larger to the state and national levels, both metrics converge, suggesting that the environmental burdens become more averaged across larger populations, reducing the appearance of disproportionate impacts. Overall, this comparison underscores the importance of incorporating emissions data in environmental justice analyses, as geographical proximity alone may underestimate the true impacts on local communities, particularly at finer spatial scales.

4. Discussion

4.1 A review of additional factors impacting neighboring communities

Although this work provides a starting point in integrating environmental sustainability and social impact through spatial analysis, it is crucial to consider other factors to develop comprehensive and community-centered models. Understanding the impacts of industrial facilities on neighboring communities is a multidisciplinary endeavor that often involves analyzing environmental, health, social, and economic factors. With all these methods, data can be paired with socioeconomic characters of communities from census data and GIS data, as was done in our analysis above, to identify disproportionate equity impacts. Here, we review a collection of methodological approaches that can allow for considerations of these additional factors and expand and refine this work.

4.1.1 Environmental monitoring of pollutants

Environmental monitoring can be used to quantify environmental burdens from facility practices. Localized monitoring methods can be used to identify areas of concern and monitoring over time and location can allow evaluation of probable causes of adverse effects on the environment.[51] For example, analysis of data from before and after a mitigation is instituted can indicate whether the mitigation is reducing the characteristics measured. Analysis of data from multiple monitors at set distances from a cement facility can indicate the extent of impact and suggest whether pollution is coming from the facility or from other sources. These assessments can be of biophysical characteristics (such as changes in air, water, and soil quality) and biophysical impacts (such as waste)[52]. Monitoring efforts to assess potential areas of concern and change, can include[53]:

- Air Quality Monitoring - setting up stations to continuously monitor air quality, which can be focused on pollutants of key concern or pollutants known to be emitted by the facility.

- Water Quality and Quantity Monitoring - sampling and analysis of water bodies near the facility to measure potential contamination (e.g., heavy metals, chemicals) caused by processes on site, as well as measurement of ground water level.
- Soil Analysis - examining soil samples for contaminants to identify any leachate or spillage from the facility that can affect local agriculture and ecosystems.
- Noise Monitoring – using sound level meters to indicate potential damage to, stress on, or interference with sleep or other behaviors of humans and fauna.
- Waste Monitoring - measuring quantities and types of waste (e.g., mine tailings, hazardous waste) generated by the facility, as well as transport and disposal pathways.

Each of these monitoring techniques requires professional/technical support to design a study, calibrate equipment, establish data gathering points, and analyze data.

4.1.2. Assessment of health impacts

Assessing community health impacts has been linked to assessing environmental impacts since the National Environmental Policy (1969) established a focus on the effects of large projects and recently, has expanded to include disproportional impacts based on socio-economic factors to focus on health-equity[54]. A primary means of assessing community health impacts is through analysis of quantitative epidemiological data that indicate incidence and prevalence of disease often across time and across geographic areas. These can be used to compare the health of populations living near the facility with that of populations farther away, looking for correlations between proximity to the facility and health issues (e.g., Wong and Raabe 2000[55]). Qualitative health surveys can be conducted to collect descriptive data from populations of interest, such as local communities. Such surveys can inquire about personal health as well as health concerns,[56] including any potentially linked to a specific industrial facility. Reliance on such surveys must be tempered by consideration of the size and representativeness of the sample, response rates, and accuracy of memory of health history. Even with these caveats, health surveys can offer suggestions of areas to further examine. A substantial amount of structured and unstructured data is collected by hospitals and clinics, though access might be limited if data bases have not been designed to provide anonymity for patients. As with epidemiological data, analysis of hospital medical data by location can identify trends in medical conditions and can be correlated to quantitative measures of pollution from materials production[57].

4.1.3. Economic parameters

There are several economic parameters beyond demographics that could be considered in the examination of an industry, its effects on neighboring communities, and GHG emissions mitigation strategies. Assessments can include consideration of job creation versus job loss, changes in property values, and the facility's overall economic contribution to the local economy. Tracking such parameters can be used to indicate potential economic effects on neighboring communities. Beyond the employment directly in the manufacturing facilities, there are upstream and downstream industries that manufacturing influences, which could also lead to employment (e.g., mining, transportation, product assembly)[58] as well as employee expenditures within the community. Such assessment can also be paired with consideration for the local housing market, including how the presence of or distance from the facility influences property values, rental rates, and housing demand. For example, multiple studies in Europe have shown that residential property values go down with increased proximity to industrial manufacturing sites[59,60].

4.1.4. Community engagement and ethnographic studies

Obtaining input from members of adjacent or nearby communities and understanding their goals and priorities is critical to mitigating negative outcomes from production facilities. Such engagement can foster buy-in for changes that are going to be made and ensures that the community is heard if there are any concerns. This is uniquely different from corporate social responsibility, which has received criticism. Namely, in some cases, employees have had a limited role in corporate social responsibility, which has limited its inclusivity[61]. It has been argued that a corporate code of social responsibility without community engagement can conceal a strategy of simply business as usual[62]. This issue has been emphasized for mining-related industries[62]. It has been highlighted that organizations should move towards recognizing the interconnectedness between local communities, particularly indigenous communities, and future sustainability goals [63]. Community involvement that relies on tours and contributions to local non-profits is not the same as engagement. Methods for community engagement could employ surveys and interviews, which can help gather insights directly from the residents about their perceptions, concerns, and experiences related to the industrial facility. As with health surveys, sample size and representativeness, as well as response rate, can limit usability. Focus groups can also be conducted, bringing together diverse community members to discuss specific aspects of the facility's impact, offering qualitative data and nuanced understandings. Organizations can run listening sessions to collect information about members of the neighboring communities' experiences.

While distinct from community engagement, ethnographic studies can further bolster understanding of neighboring communities, their goals, and their concerns. Namely, through fieldwork, living in a community, a better understanding can be gained about the circumstances of the people being studied[64]. Often called participant observation, researchers immerse themselves in the community, observing daily life and community-facility interactions, gaining a deeper understanding of the lived experiences of residents. However, while participant observation can provide rich, qualitative data on individual and collective experiences, perceptions, and attitudes towards the industrial facility, its time-intensity and lack of reproducibility of results limits it utility.

4.1.5. Secondary data analysis (including legal and policy analysis)

There are several forms of secondary data analysis that can provide perspective on community response to neighboring cement plants. Secondary data are data that were collected for another purpose. A review of the existing academic studies, industry reports, and case studies from similar contexts can be used to predict and understand potential impacts. Ideally, such work would be organized thematically or methodologically, synthesizing findings to illustrate the current state of knowledge and the evolution of the field, as well as highlighting existing limitations. Analysis of secondary data can also play a critical role in determining expected impacts on neighboring communities (as well as burdens external to the community) in the absence of primary data (such as direct emissions monitoring or community health data). Quantitative secondary data such as data gathered by government agencies, NGOs, and other organizations can often provide material for analysis of potential impacts without the cost of original research. For example, an air pollutant emissions calculations based on industry-dependent energy demands[65,66], state reported energy resources used[67], and nationally reported emissions factors by energy resource and combustion type[68] can permit assessments comparable to various forms of monitoring.

Reviews of legal compliance of the facility with environmental, health, and safety regulations, can provide useful insights for the community. Most pointedly, these could include court cases, recorded violations, and grievances against industrial facilities (e.g., William and Onciano 2022[69]) and alternative technology companies offering GHG emissions reduction methods. For violations on environmental aspects, inadequate monitoring, reporting, or action to mitigate impacts to water, air, and soil, there are direct implications of potential effects on the neighboring communities. However, cases involving labor-relations can also be used to understand if there are potential other issues that may affect the community. Further, complaints regarding management methods can be strong indicators of the potential efficacy of regulatory measures. For example, in the past there were many complaints filed by the Federal Trade Commission with regard to vertical integration and mergers with cement companies and ready-mixed concrete producers[70]. And in Europe, it has been argued that there are both legal and illegal cartels that have influenced cement industry monitoring efforts, information exchange, and pricing schedules[71].

Part of such work can also include both assessments of the effectiveness of current policies and assessments of the effectiveness of policies in other areas in protecting the community, as well as guiding responsible industrial practices. Examining policy effectiveness could include checking if it has measurable goals, utilizing before-and-after data to assess performance metrics, soliciting expert and public feedback, examining stakeholder benefits against costs, noting any unintended effects, and ensuring transparency and accountability. In considering other policies, several policies have been implemented in the US to quantify or address the embodied carbon of materials (which would encapsulate emissions such as those from cement production) (e.g., toolkit by the Carbon Leadership Forum[72]). Internationally, policies addressing embodied carbon have also been explored and/or implemented (e.g., Rowland et al 2023[73] and report by the French Ministry of Ecological Transition and Territorial Cohesion[74]).

4.2 Limitations

While this study provides insights into the disproportionate impacts of building material production facilities on communities of color and low-income populations, several limitations should be acknowledged. First, our analysis relies on primary $PM_{2.5}$ emissions data from the NEI, which captures direct emissions released from production facilities, including potential heavy metals in particulate form. However, this analysis does not include the chemical transformation of precursor emissions (i.e., secondary $PM_{2.5}$ emissions). Furthermore, the current analysis does not incorporate dispersion modeling, meaning it does not account for the long-range transport of pollutants, which could significantly affect the broader regional impact of these emissions. This limitation is particularly relevant at larger spatial scales, where emissions can disperse far beyond their point of origin, potentially affecting populations not captured in our localized analysis.

Additionally, our analysis focuses on two primary demographic indicators: income and person of color status. These were selected both for their relevance to EJ concerns and for methodological consistency with the US Environmental Protection Agency's (EPA) EJScreen tool, which defines the DI based on these two factors. Other important demographic factors, such as age, education level, health status, or housing quality, are not considered, which may limit the comprehensiveness of our understanding of vulnerability and environmental justice concerns. These additional factors could contribute to further insights into the disproportionate impacts experienced by various subgroups within the population, and future studies should seek to expand the demographic scope to capture these variables.

Further, our analysis allows for any demographic group to be identified as disproportionately impacted based on their proximity to CBM facilities or emissions exposure. This framework is designed to avoid a deficit-based approach, ensuring that disproportionate burdens are revealed without prior assumptions or biases about which groups are more likely to be affected. As such, a region with low concentrations of people of color and people who are low-income (e.g., DI range 1) can be discussed as disproportionately impacted by the location and emissions of a CBM facility. However, the authors acknowledge that regions with high concentrations of historically marginalized groups have experienced long-standing disproportionate environmental burdens and health impacts due to systemic inequalities.

Future studies should apply these equations to draw trends to more directly assess how these historical inequities may be exacerbated by building materials industries.

Moreover, although the use of census tract, county, state, and national spatial scales provides broad insights into disproportionate impacts, it may obscure critical localized effects, particularly at smaller geographic levels such as neighborhoods. Although we included a fine-scale analysis at the census tract level, broader geographic regions may mask extreme disparities at a localized scale, where environmental and social burdens may be significantly more concentrated. Future work should consider integrating finer geographic resolution data to better address these hyper-localized impacts.

Lastly, the U.S. Census data used to define demographic characteristics can often underreport communities that are particularly vulnerable to environmental injustice, such as indigenous populations, migrant workers, and unhoused individuals. These populations, despite being among the most exposed, are not fully captured in the demographic data, which limits the study's ability to account for their disproportionate burden.

5. Conclusion

Recent policies aimed at industrial decarbonization are anticipated to significantly impact construction and building materials industries, as they will be required to adopt practices that reduce their environmental burdens. This new focus provides an opportunity to monitor and address social burdens, such as historical EJ concerns, at the same time. However, there is a current data gap in applying EJ concepts to building materials production. We provide a methodological framework which measures the disproportionate impacts of building material production facilities on concentrations of communities of color and of low-income, at four spatial scales (census tract, county, state, nation). We find that, across each of these spatial scales, a majority of CBM facilities are causing disproportionate impacts.

The range of $I_{E,d}$ found in this work and their average values at different spatial scales reveal that the concentration of CBM facilities in certain regions exacerbates disproportionate environmental burdens, with the severity of these impacts varying significantly depending on the spatial scale. This demonstrates the importance of analyzing both local and broader spatial scales to fully capture the extent of environmental injustices.

These findings suggest that targeted, localized interventions—such as policies to reduce emissions in areas with higher disproportionate impacts—are critical to mitigating the environmental burdens on overburdened communities. At broader spatial scales, interventions focused on reducing overall emissions across the industry may be more effective. Future work should continue to explore these geographic and emissions-based trends to better inform policies that address environmental justice in the building materials sector.

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Data Availability Statement

All data that support the findings of this study are included within the article (and any supplementary files).

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