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The unaccounted costs of slash pile burning: Carbon emissions and $PM_{2.5}$ exposure in central Oregon, USA

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ABSTRACT

Slash pile burning is a commonly used practice for disposing of unmerchantable forest residues resulting from forest thinning and timber harvest to lower wildfire intensity and risk, but can also generate significant greenhouse gas (GHG) and particulate matter ($PM_{2.5}$) emissions. These effects of pile burning generate costs that are rarely taken into account in forest management budgets, expenses that could significantly affect decision-making if considered. This study quantifies those external costs of pile burning in the Deschutes National Forest in central Oregon, relating modeled emissions from 85 pile burns in 2022 to population exposure. We estimate that the piles burned produced approximately 61,413 metric tons of CO_2, 104 metric tons of CH_4, and 250 metric tons of $PM_{2.5}$, resulting in costs ranging from \$2.4 million to \$116 million in combined local health and global climate costs, depending on the assumptions used. While pile burning can reduce wildfire risk, our results highlight the unaccounted effects of pile burning, suggesting a potential benefit from considering alternative disposal strategies, such as the production of bioenergy or biochar. These findings emphasize the importance of integrating the external costs of pile burning into forest management decisions and demonstrate an adaptable framework for evaluating the external costs of biomass disposal in other regions.

1. Introduction

Slash pile burning is a common practice used to dispose of woody residues that result from forest thinning and forest harvests in the western US. This approach reduces hazardous fuel loads which can then lower the intensity and risk of wildfires. The benefits of fuel reduction treatments, which commonly include pile burning, are well documented (Mott et al., 2021). The external costs of pile burning remain largely undocumented, and these costs rarely influence forest management decisions. Here, we define external costs as the unpriced social damages of pile burning, specifically the local health and global climate impacts from smoke emissions. These costs occur outside the direct operational expenses of conducting pile burns and are typically omitted from forest management analyses. This study addresses this gap by estimating the external costs of pile burning in central Oregon, combining emissions modeling, population exposure analysis, and economic valuation. By providing cost estimates with ranging assumptions, our framework highlights the trade-offs of current practices and offers a transferable approach to evaluate biomass disposal strategies in other forest regions.

The primary purpose for which slash piles are formed is to dispose of unmerchantable material left after forest treatments to reduce the risk of wildfires. With wildfires becoming more frequent and intensified in the past 50 years (Jolly et al., 2015; Mansoor et al., 2022), slash piles and slash pile burns have become an important component of addressing this problem. To reduce fuel loads for potential fires, thinning forests and timber harvest are commonly used, resulting in additional slash piles (Banerjee, 2020; Kang et al., 2014; Agee and Johnson, 1988). While forest thinning and forest harvests reduce hazardous fuel loads, which can improve forest health and decrease the intensity and risk of wildfires (Vorster et al., 2024; Wang et al., 2023; Collins et al., 2023; Brodie et al., 2024), they leave unmerchantable material as byproducts. The positive impacts of pile burning and fuel reductions are well recognized, however these practices generate greenhouse gases (GHGs) and smoke that can negatively affect local and regional air quality and human health (Anenberg et al., 2012; Ganguly et al., 2018; Sifford et al., 2016; Pierobon et al., 2022; Barker et al., 2025). Pile burns can also create burn scars that persist for decades and inhibit vegetation regeneration (Rhoades et al., 2021; Halpern et al., 2014) and promote invasive species (Sexton et al., 2020).

Throughout US forests, thousands of brush pile burns contribute to both local and global externalities. An externality exists when the activity of one party (individual, household, firm, management agency)

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affects the well-being of another party without being priced. Because the effect is unpriced, there is no incentive for the emitting party to consider the effect, either beneficial or detrimental, on the affected party, and an inefficient amount of resources may be devoted to the activity. The benefits of pile burns are typically recognized in forest management decisions and accounted for in budget targets (Charnley et al., 2015), while the associated external costs remain largely undocumented and rarely influence the final decision on treatment selection (Afkhami et al., 2024). Since the primary benefit of pile burning comes from the removal of excess fuels, alternative removal strategies can generate lower external costs, providing greater net benefits for forest treatment methods such as forest thinning.

Biomass combustion is the largest source of primary fine carbonaceous particles and the second largest source of trace gases[1] in the global atmosphere (Akagi et al., 2011). Removing excess sources of greenhouse gas emissions and trace gases, such as wildfires and slash pile burns, is a significant concern in addressing climate change. Because slash pile burning can release 92%–94% of its carbon content in a short period of time (Temple, 2022), thousands of pile burns annually can significantly impact global emissions totals.

To evaluate forest management strategies and the best potential uses for slash, the complete cost of pile burning must be considered. Wildfires are a substantial source of biomass burning and have been the focus of evaluating impacts of smoke pollution on human health (Cascio, 2018; Reid et al., 2016). However, fewer studies have attempted to identify the effect of burning slash piles on human health (Pierobon et al., 2022; Nance, 2023), with no known literature that quantifies these damages as an external cost. Understanding these external costs can help decision makers allocate the appropriate resources to the problem of slash disposal or explore alternative disposal methods.

Some of this slash can be used to make wood products, but due to high costs, up to 60% of harvested material remains on site (Parikka, 2004) and is frequently piled and burned. Many landowners consider prescribed pile burning wasteful and prefer to find useful applications of slash (Ynfante et al., 2024), but economic and operational barriers constrain alternatives. The collection, processing and especially the transport of low value slash is costly; for example, biomass collection can exceed $45 per ton, costing far more than on-site burning (Han et al., 2002; Bisson et al., 2016). While unmerchantable brush and small-diameter trees could supply renewable energy or bioproducts, high transport costs limit feasibility (Han et al., 2002, 2018). As a result, prescribed pile burning remains a common solution for slash disposal, despite creating a significant source of local air pollution (Sifford et al., 2016). These emissions can contribute to acute and chronic health effects, particularly for vulnerable groups such as children, the elderly, and individuals with asthma or other respiratory conditions (Sifford et al., 2016; Schwartz, 1993; Dockery and Pope, 1994; Pierobon et al., 2022). Alternative disposal methods such as bioenergy or biochar production may reduce these impacts and warrant further evaluation (Puettmann et al., 2020; Adhikari et al., 2024).

This study uses the Deschutes National Forest in Oregon (see Fig. 1) as a case example to assess the broader cost to society of brush pile burning for one year. This choice was made because the Forest's large number of piles and significant population nearby offered an ideal setting to assess the local and global costs associated with pile burning. Using a single year in a single forest, we analyze the emission profile of pile burning while measuring external costs on local and global scales. With these methods, the external cost of burning brush piles in the Deschutes National Forest in 2022 is estimated to be between $2.4 million and $116 million depending on the assumptions used. The external costs of these pile burns are critical considerations for policy makers, forest managers, and timber manufacturers to ensure that the goals of managing healthy forests are aligned with the associated costs of brush removal methods. Knowing the external cost associated with pile burns occurring within an average year allows for more comprehensive analyses of different disposal methods, enabling informed decisions on which practices best align with forest managers' goals.

[1] Trace gases are gases in the Earth's atmosphere other than nitrogen, oxygen, and argon which make up the vast majority of gases in the global atmosphere. This includes CO_2, CH_4, and other GHG.

(a) The U.S. State of Oregon

(b) Location of the Deschutes National Forest within Oregon.

Fig. 1. The study area of the Deschutes National Forest in Oregon (a). This National Forest is in the center of the Oregon (b) in the Pacific Northwest.

2. Data and methods

The central objective of this research is to understand and quantify external costs of burning brush piles to rigorously show there are substantial costs not being considered in forest management decisions. The empirical analysis leverages a database maintained by the United States Forest Service (USFS) that records all brush piles formed by purchasers of National Forest timber. The data contain detailed information on the locations, sizes, and other characteristics related to thousands of brush piles. In conjunction with this dataset, we utilize the BlueSky modeling framework (Larkin et al., 2009) to provide emission profiles and air quality measurements for each pile burn. Bluesky integrates the VSmoke model, which calculates the maximum concentration of $PM_{2.5}$ over 24 h. This combination allows analysis of both the impacts of smoke dispersion of $PM_{2.5}$ and GHG emissions from pile burns. Using estimated impact functions of $PM_{2.5}$ exposure (U.S. Environmental Protection Agency, 2019), we calculate the number of people expected to experience adverse health outcomes and the associated cost of treatment. This incremental exposure is independent from background $PM_{2.5}$, as we only consider the effects of pile burns. This approach allows for a comprehensive estimate of the global environmental impact of GHG emissions and the local impacts of exposure to particulate matter that results from burning brush piles, expressed in dollar terms.

2.1. USFS - Brush Disposal Funded Activities

The Brush Disposal Funded Activities data set is a free and regularly maintained dataset provided by the USFS. The data document the location of brush piles formed by all purchasers of national forest timber and various characteristics of each pile. Due to these specifications, the data do not contain all brush removal activities. Information is missing on piles formed on private land and in the eastern and southern regions of the United States. The lack of data in the East and South is because most of the forest lands in these regions are privately owned (Butler et al., 2016), and therefore are not tracked by the USFS. Although the data does not include all sources of slash piles, it does provide high-quality spatial data on the locations of each pile, the treatment area in acres, and the date of disposal, which are used in this analysis.

Since 2000, more than 42,000 treatment areas have been explicitly labeled as piled and burned. These treatment areas represent sites where forest residues are gathered in piles and then burned, with the majority concentrated in the western and northern US. Between 2000 and 2023, 13,100 treatment sites were treated with pile burns in the Pacific Northwest, where the Deschutes National Forest is located. Although most of the regions have burned a consistent number of brush piles annually, the Pacific Northwest has seen an increase from around 500 in 2005 to 1500 in 2022. In 2022, the Deschutes National Forest had 85 total treatment areas, with an average pile comprised of residual material from a treatment area of 49.6 acres, a minimum treatment area of 4 acres, and a maximum of 246 acres.

When the data describe a pile, it is not to be inferred as one singular mass of brush. Instead, it is a representation of the area where harvesting or treatment has occurred. This means that a ‘pile’ labeled as having 50 acres is a collection of piles formed from the material resulting from treatment activities across those 50 acres. Because of this, there is no explicit information on the exact tonnage or number of individual piles burned within each area in the data. The emissions of each pile burn are calculated assuming a prescribed burn of the estimated fuel totals in the area provided in the data. Pile collections are typically consolidated before burning or processing, but this is not considered for emissions calculations.

2.2. BlueSky and VSmoke modeling

The BlueSky model enables estimation of several types of emissions according to the location of each pile (Larkin et al., 2009). The USFS and the University of Washington developed this model to simulate the emission outputs of fire events across the United States. The model also considers the size and composition of each pile to allow for greater accuracy.

This model has been used in a variety of applications, including the estimation of wildland fire emissions (Jaffe et al., 2020), the balance of public health and land management goals (Oakley, 2022), and other analyses of the impacts of slash pile burns on air quality in Washington State (Pierobon et al., 2022). The precision of the BlueSky model was reviewed and compared with the fire events observed by Pan et al. (2020). Their research concluded that the BlueSky model captured most of the fire signals detected by multiple observations and was useful for predicting and evaluating fire impacts. Due to its extensive use in the fire literature, the BlueSky model is suitable for the research goal of modeling the emission profiles of pile burns.

The BlueSky program outputs emission levels in tons of $PM_{2.5}$, PM_{10}, carbon monoxide (CO), carbon dioxide (CO_2), methane (CH_4), nitrous oxide (NO_x), volatile organic compounds (VOC), ammonia (NH_3), and sulfur dioxide (SO_2). The emission levels depend on the parameters inputted into the model, such as the type of fuel, its moisture content, the amount of fuel consumed, and the timing of the burns. BlueSky also estimates the fuel bed in specified locations based on over 400 different fuel bed types. A fuel bed refers to the characteristics of various tree species in an area and their combustible material that influence the behavior and effects of the fire. Each fuel bed is characterized by a different tonnage per acre of various fuel types, or categories of burnable material like sound wood, ladder fuels, and dead branches. Among the numerous fuel beds in Bluesky, four are located in the Deschutes National Forest, shown in Table 1.

2.3. Model configuration and assumptions

Fuel beds BlueSky assigns each pile a fuel bed based on its location, with preset values in tons per acre. For this analysis, we restricted emissions to fuel types typically present in slash piles. These are the middle and lower understory, class 1 snags, ladder fuels, shrubs and herbs, 1–100 h sound wood, 1000 h rotten wood, and litter (Chang et al., 2023; Schnepf et al., 2009; Agee and Skinner, 2005; Stephens et al., 2012). Excluded fuel types include tree overstory, class 2–3 snags, ground lichen, moss, tree stumps, duff layers, sound wood exceeding 100 h, and rotten wood exceeding 1000 h. This normalization means that all piles assigned a given fuel bed share the same fuel loads per acre, ensuring comparability between sites. Although emissions from individual piles can vary, we standardize emissions to capture the composition of an average slash pile. With this methodology, a ponderosa pine savanna pile in Oregon will possess the same composition as a ponderosa pine savanna fuel bed in California.

Burn assumptions Burn assumptions for each simulated pile burn are standardized to typical pile burn conditions. This includes setting the fuel moisture to very dry and fixing the duration of pile burn to one day, consistent with most pile burn operations (Barker et al., 2025). We assume that piles burn 90% of the estimated fuels (Barker et al., 2025; Hardy, 1996) in the primary results, and report sensitivity analyses for 50% and 70% consumption factors. The assumed tonnage per acre is then representative of the available burnable material present at each pile location, with the consumption factor representing how much of that material is burned in pile burns. These assumptions enable consistent comparisons across scenarios and allow global social costs to be calculated directly from BlueSky’s greenhouse gas and particulate matter emission outputs.

Meteorology and dispersion Dispersion was simulated with VSmoke under typical meteorological conditions of the fall burn season (October–December). This time window is typically used to minimize the risk of wildfire and help restrict a fire’s ability to spread due to cooler temperatures and higher humidity (Sifford et al., 2016; Pierobon et al., 2022). For each simulation, wind speed was set to 7 mph, with prevailing winds from the south, based on hourly data from the Redmond Airport station located 20 miles north of Bend (Midwestern Regional Climate Center, 2025). This aligns with Oregon’s prescribed burn regulations that prohibit operations at wind speeds greater than 15 mph (Groth et al., 2020). Additional VSmoke parameters were set to standard neutral conditions. These conditions include a mixing height of 1500 ft, surface temperature of 59 °F, surface pressure of 1013.25 mb, relative humidity of 25%, and background $PM_{2.5}$ equal to zero to only measure the incremental increase in $PM_{2.5}$ resulting from pile burns. These settings represent average November burn conditions in central Oregon. Further details on seasonal wind regimes, including wind data from both Weatherspark and Redmond Airport, are provided in the appendix.

2.4. Global social costs of brush pile burns

Global costs are essential for estimating the total damages from pile burns, since GHG emissions contribute to long-term climate damages and represent an important component of their overall effects. The costs resulting from GHG emissions are calculated by normalizing all emissions to a CO_2 equivalent based on their global warming potential (GWP). The US EPA defines global warming potential as the amount of energy a gas will absorb over a specified period relative to the

Table 1
The number of burned piles, assumed tonnage of brush per acre, and total acreage of burned slash piles in 2022. Data are summed across each of the four fuelbeds a pile can belong to.

Fuel bed	Number of piles	Tonnage per acre	Total acres
Mature lodgepole pine forest	27	7.24	1416
Pacific silver fir - mountain hemlock forest	6	5.49	276
Pacific ponderosa pine - Douglas-fir forest	29	14.3	1610
Ponderosa pine savanna	23	6.61	917

energy absorbed by the emission of 1 ton of CO_2 (U.S. Environmental Protection Agency, 2024b).

Normalizing emissions with the GWP allows a direct calculation of the global social cost of the emissions resulting from each burn using the social cost of carbon dioxide emissions (SC-CO_2). The social cost of CO_2 emissions[2] (SC-CO_2) is an estimate, in US dollars, of the net present value of future damages caused by 1 metric ton (t) increase in CO_2 emissions in a given year (Interagency Working Group on Social Cost of Greenhouse Gases, U.S. Government, 2016; U.S. Environmental Protection Agency, 2023). These damages include changes in agricultural productivity, human health, property damage from an increased risk of natural disasters, and loss of ecosystem goods and services due to changes in climate.

Of the emissions produced by the BlueSky model, only carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) have considerable potential for global warming (U.S. Environmental Protection Agency, 2024b). Since most of the BlueSky outputs estimate very low quantities of N_2O (< 0.01 tons), we do not consider it in social cost calculations. Methane has a global warming potential that is 27 times greater than that of CO_2 over a century-long timeframe. Although other emissions from the BlueSky outputs may have associated social costs, these costs cannot be standardized with global warming potentials and are beyond the scope of this research. Since BlueSky provides the tonnage of CO_2 and CH_4 from each pile burn, it enables a straightforward calculation of the global social cost of these emissions. This cost is calculated using the tonnage of CO_2 plus the product of 27 times the tonnage of CH_4 times the SC-CO_2.

$$CO_{2e} = CO_2 + (27 \times CH_4)$$

$$\text{Global Cost} = CO_{2e} \times \text{SC-}CO_2$$

The SC-CO_2 is a vital parameter that significantly influences global cost calculations. The SC-CO_2 used by US federal agencies has changed in the last 10–15 years (Rennert and Prest, 2023). In early 2023, the US EPA proposed an increase in the SC-CO_2 from $ 51 per ton to $190 per ton, highlighting the instability in consensus on the appropriate value (U.S. Environmental Protection Agency, 2023; Rennert and Prest, 2023; Rennert et al., 2022). Following the Office of Management and Budget Circular A-4 in 2023 (U.S. Office of Management and Budget, 2023), the SC-CO_2 estimates used in this research are calculated to include effects experienced by non-citizens residing abroad and domestically within the US. This implies that any calculation using SC-CO_2 estimates that only consider domestic damages to the US would be significantly lower. Other countries use a higher SC-CO_2 than the US, with Germany citing a range of $235 to $820 per ton and the United Kingdom citing a range of $20–$100 per ton (Wagner et al., 2021).

Given this evolving policy context and the lack of a single universally accepted SC-CO_2 value, it is important to reflect the uncertainty of these estimates. The SC-CO_2 can vary widely depending on a range of assumptions, including whether damages are considered domestically or globally, population projections, the persistence of CO_2 in the atmosphere, the discount rate applied, and how climate impacts are expected to affect different sectors of the economy. To account for these uncertainties, a sensitivity analysis is conducted using a range of benchmark SC-CO_2 values, including $51, $100, and $190 per ton of CO_2. This range helps illustrate how key assumptions, particularly the choice of discount rate, influence the estimated social cost of pile burning.[3]

We reiterate that the most important part of this analysis is the CO_{2e} emissions themselves. The SC-CO_2 is simply a multiplicative factor to place dollar values on potential damages caused by emissions. Most of the uncertainty enters this analysis through the SC-CO_2, while the emissions themselves are more certain.

2.5. Local health costs of brush pile burns

Deriving the health cost of a change in air quality is less straightforward than determining the social cost of global emissions due to the complex nature of air quality and its interaction with weather and population, from wind speed to the amount of human health impact associated with changes in $PM_{2.5}$ concentrations. To accommodate this variability, average impact measurements are used wherever possible to ensure a consistent benchmark across all piles and their associated air quality impacts. Most piles are burned far enough away from population centers to have a relatively low impact on human health, but when large piles are burned and considerable amounts of smoke are released, significant deterioration of air quality can affect surrounding populations. Because our data are limited in the precise parameters surrounding the timing of each pile burn, a one-day burn assumption is used to gauge the resulting emissions for each burn in the data, per standard practice (Barker et al., 2025; Hardy, 1996). Therefore, the total emissions resulting from each burn could spread over more days and at lower total concentrations, but this factor is not considered for analysis. Since we only consider how smoke emissions from pile burns result in additional health costs to individuals, our measurements for local effects are a local health cost, rather than an all-encompassing local social cost. In contrast, since global costs are calculated using a value of SC-CO_2, which includes all estimated damages from CO_2 emissions, global damages are considered global social costs. Therefore, when presenting final results, the term total external cost is used to signify the sum of global social cost and local health costs.

We estimate local costs using 24-hour peak $PM_{2.5}$ concentrations attributable to each pile burn, assuming a 24-hour exposure window per event. We value only the incremental increase in concentration from each modeled pile burn rather than total $PM_{2.5}$ (which would be including background $PM_{2.5}$) to gauge the discrete impact of the pile burn regardless of prevailing air quality factors. These outputs provide five tiers of potential air quality change based on proximity to the fire's location. These tiers, representative of areas covered by varying smoke concentrations, are moderate (38–88 $\mu g/m^3$), unhealthy for sensitive groups (88–138 $\mu g/m^3$), unhealthy (138–351 $\mu g/m^3$), very

[2] The SC-CO_2 has been commonly referred to as the social cost of carbon. This can lead to confusion because carbon is generally transferred to the atmosphere in CO_2, and the SC-CO_2 was defined as a metric ton of CO_2 emission. To ensure clarity, the Interagency Working Group on the Social Cost of Greenhouse Gases began to refer to social cost of carbon dioxide rather than social cost of carbon (Interagency Working Group on Social Cost of Greenhouse Gases, U.S. Government, 2016).

[3] It is important to emphasize that the SC-CO_2 values used here are not definitive, but rather serve as a reference framework for evaluating the carbon-related damages due to pile burns.

unhealthy (351–526 μg/m^3), and hazardous (526+μg/m^3). These exposure tiers are the BlueSky/VSmoke model tiers, which do not align with EPA/WHO thresholds and are the basis for this exposure analysis. The 24-hour peak $PM_{2.5}$ concentration levels remain consistent regardless of the pile's size, and therefore a larger pile only extends the area for each concentration tier than with a smaller fire.

To measure the amount of damage these air quality changes cause relative to baseline air quality, the midpoint, upper bound, and lower bound of each concentration range are considered to provide a robust measure of $PM_{2.5}$ damages across each affected area. Responses to a change in $PM_{2.5}$ concentration are measured in increments of 10 μg/m^3 at a time (U.S. Environmental Protection Agency, 2019) and, therefore, the assumed exposure for each exposure tier is rounded to the nearest ten to facilitate calculation. This implies that for the moderate, unhealthy for sensitive groups, unhealthy, very unhealthy, and hazardous levels, the assumed midpoint exposure concentrations are 60 μg/m^3, 110 μg/m^3, 250 μg/m^3, 440 μg/m^3, and 530 μg/m^3 respectively.[4] For the hazardous tier, the model reports an open-ended upper bound, and we therefore assume 530 μg/m^3 as the representative concentration for lower, upper, and midpoint calculations. Because the tiers do not aggregate overlapping events and the events are valued separately in time, each person is counted once per event and there is no double counting of the exposed population.

With air quality measurement standards in place, the number of people in each affected area is determined using population data mapped to 1 km grids for the region (CIESIN - Columbia University, 2018). Because the air quality surfaces are irregularly shaped polygons over population grid cells, the percent coverage of the air quality surface and each population grid cell is measured and then summed to reveal the affected populations by each brush pile burn's air quality impacts. It should be noted that this approach may overestimate or underestimate exposure near urban edges and does not capture time activity patterns or vulnerable subgroups (e.g., outdoor workers). Fig. 2 shows an example of this calculation. The red ellipse shape represents the 24-hour peak $PM_{2.5}$ concentrations reflective of moderate air quality from a simulated pile burn over 24 h, while the blue shape at the bottom of the ellipse represents the area where pile burns occurred. We then overlap the air quality surface with the population grid, with population densities shown by the colored rectangles ranging from black to white.

The pile in Fig. 2 is a 69-acre pile burn directly south of Bend, and since the assumed wind conditions have prevailing winds from the south, this air quality surface is able to reach the suburban edge. The cell outlined in green is an example where, since roughly 50% of the purple cell overlaps with the red air quality surface, we assume that the change in air quality impacts 50% of the population within that cell. This proportion of coverage is calculated precisely for all five air quality concentration surfaces for every pile burn in the data. Because health and mortality costs are derived from expected statistical exposure consistent with EPA guidelines (U.S. Environmental Protection Agency, 2024a), we consider exposure to fractional individuals when calculating health costs.

With an affected population associated with each $PM_{2.5}$ concentration tier and an assumed level of exposure within each, health impacts can be estimated. The estimated health impacts of $PM_{2.5}$ are derived from the US EPA 2019 Integrated Science Assessment for Particulate Matter (U.S. Environmental Protection Agency, 2019). This EPA analysis is a review of the literature that gauges the correlation between a 10 μg/m^3 increase in $PM_{2.5}$ concentrations and the associated health impacts of that change. For short-term exposure, the report concludes that an increase in $PM_{2.5}$ concentration is "likely to be causal" for respiratory effects. At the same time, cardiovascular hospitalization and cardiovascular mortality are deemed causally influenced by short-term exposure to increases in $PM_{2.5}$ concentrations (U.S. Environmental Protection Agency, 2019).

A 10 μg/m^3 increase in $PM_{2.5}$ concentration raises asthma hospital admissions by 1.3% to 6.8%, cardiovascular hospital admissions by 0.5% to 3.6%, and cardiovascular mortality by 0.47% to 0.94%. In comparison, deSouza et al. (2021) found a 0.9% increase in cardiovascular hospital admission rates per 10 μg/m^3 increase in $PM_{2.5}$ concentrations. The EPA report indicates that the concentration–response relationships for $PM_{2.5}$ are linear and do not have a threshold, which means that the effects scale proportionally between different concentrations. Similar findings from (Delfino et al., 2009) state that average increases of 70 μg/m^3 during heavy smoke conditions were associated with an increase of 34% in asthma admissions, further supporting a linear concentration response function without threshold relationship. Therefore, a 50 μg/m^3 increase would result in these impacts being five times greater (U.S. Environmental Protection Agency, 2019).

Since each of these impacts has a range, the lower, midpoint, and upper bound values of each range are evaluated to ensure a complete assessment of the effects under different assumptions. The primary findings are based on the midpoints of EPA estimated impacts on asthma, cardiovascular health, and cardiovascular mortality. Although non-accidental mortality is also causally influenced by an increase in $PM_{2.5}$ concentrations, this measure is separately considered for local cost calculations to highlight the cost of illness and cardiovascular mortality outcomes.

The increases outlined in the 2019 EPA report are based on an increase in respiratory, cardiovascular, and mortality cases from the baseline levels of each. Respiratory baseline levels are obtained from a 2020 study that found 5% of workers nationally had at least one asthma-related medical event between 2011 and 2015 (Syamlal, 2020), with an average cost of $8,238 for each hospital visit. Cardiovascular disease is studied much more closely since it is the number one cause of mortality in the US, resulting in an annual report of cardiovascular statistics published by the American Heart Association (AHA) (Virani et al., 2020). The 2020 AHA research makes a point to note that air pollution from $PM_{2.5}$ "is associated with elevated HBP (high blood pressure), poor endothelial function, incident cardiovascular disease (CVD) events, and all-cause mortality" (Virani et al., 2020). The report also finds there were 4,840,000 emergency department discharges for CVD in 2016, indicating a rate of 1.49% of US citizens admitted to hospitals for CVD. The study finds that the overall CVD death rate in 2017 was 219.4 per 100,000 Americans, but these rates were lower in Oregon, at 191.8 per 100,000. For our analysis, we used all the available data specific to the average Oregon resident and national data where Oregon specific data is not available. Therefore assumed baseline health incidence rates used are the 2011–2015 asthma rate of 5%, the 2016 CVD hospitalization rate of 1.49%, and the 2017 CVD mortality rate of 0.1918%. Values determined from prior to 2020 were prioritized to avoid conflation of data with the COVID-19 pandemic.

The rate of change from a baseline level of injury or mortality measures how many additional people are affected given a shift in $PM_{2.5}$ concentrations, allowing the final step of associating a dollar amount to each additional case of illness or mortality. Each medical case, whether cardiovascular or respiratory, is given an average cost for that course of treatment, allowing the average cost to be estimated based on the increase in estimated hospitalizations. As stated previously, asthma-related medical events had an average cost of $8,238 for each inpatient visit (Syamlal, 2020), while cardiovascular emergency department visits had an average cost of $1,942 for each of the

[4] The upper and lower bounds of exposure assumptions are similarly rounded to the nearest 10 μg/m^3, with assumed exposure within each tier of 40 μg/m^3 and 90 μg/m^3 for moderate exposure, 90 μg/m^3 and 140 μg/m^3 for unhealthy for sensitive groups, 140 μg/m^3 and 350 μg/m^3 for unhealthy, and 350 μg/m^3 and 530 μg/m^3 for very unhealthy exposure calculations.

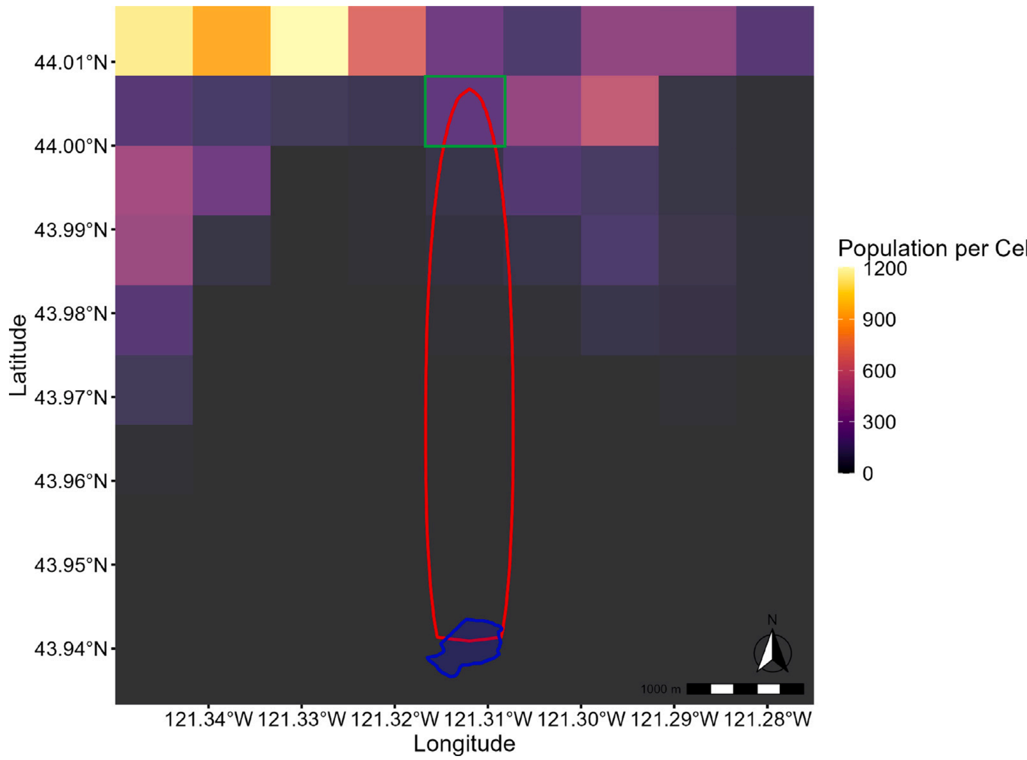


Fig. 2. Illustration of how population exposure to $PM_{2.5}$ from a pile burn is calculated. The red shape represents the extent of exposure to moderate $PM_{2.5}$ concentrations, while the blue shape represents the documented area of the selected pile burn. Each square in the grid represents a 1 km *x* 1 km area that is colored based on the number of people who live in that area.

hospital discharges[5] (Virani et al., 2020). For changes in cardiovascular mortality, the 2024 value of statistical life of \$13.1 million is used to measure mortality costs, with no adjustment for age (Kearsley, 2024).

In summary, we follow a structured approach for each burn event to estimate the average health impacts of changes in $PM_{2.5}$ concentrations from pile burns. First, we determined the number of people exposed to various $PM_{2.5}$ concentration tiers using population data combined with the output of the BlueSky model. Let N_i represent the number of people exposed to the ith $PM_{2.5}$ concentration tier. Next, we identify baseline levels of medical cases within each $PM_{2.5}$ tier. Let H_j represent the baseline rate of a specific health outcome j (e.g., baseline asthma hospitalization or cardiovascular hospitalization). The baseline number of health outcomes, B_{ij}, is given by:

$$B_{ij} = H_j \times N_i \tag{1}$$

Using the 2019 EPA report, we estimate changes in the number of individuals affected by each type of medical incidence due to increased $PM_{2.5}$ levels. Let ΔH_j represent the change in the health outcome rate per 10 μg/m^3 increase in $PM_{2.5}$ concentrations for health outcome j, and let ΔC_{ij} represent the change in medical cases for the ith tier for health outcome j. Here, ΔC_{ij} represents only the incremental increase in cases above baseline incidence, ensuring that B_{ij} is not double-counted.

$$\Delta C_{ij} = B_{ij} \times \left(\Delta H_j \times \left(\frac{\Delta PM_{2.5,i}}{10} \right) \right) \tag{2}$$

where $\Delta PM_{2.5,i}$ is the increase in $PM_{2.5}$ for the assumed concentration for the ith tier (this is the midpoint of each tier in the base case and endpoints of each tier in sensitivity). $\Delta PM_{2.5,i}$ is divided by 10 to account for the concentration response for every 10 μg/m^3 increase to $PM_{2.5}$ exposure. The total cost of illness for a single pile is calculated by multiplying the change in the number of medical cases by the cost associated with each medical incidence and then summing across all 5 $PM_{2.5}$ concentrations, i (moderate, unhealthy for sensitive groups, unhealthy, very unhealthy, and hazardous) and across all health outcomes, j (asthma or cardiovascular hospitalizations). To account for uncertainty, these calculations are made for the midpoint, upper bound, and lower bounds for both the health-concentration response and the assumed exposure tiers. Let $C_{\mathrm{j,med}}$ represent the cost of health outcome j:

$$\text{Cost of Illness} = \sum_{i=1}^{5} \sum_{j=1}^{2} \Delta C_{ij} \times C_{\mathrm{j,med}} \tag{3}$$

Using a similar approach, we determine the cost of increased cardiovascular mortality. Let M_{cv} represent the baseline rate of cardiovascular mortality and $B_{i,\mathrm{cv}}$ represent the baseline rate of cardiovascular mortality at the concentration level i. ΔH_{cv} then represents the change in the cardiovascular mortality outcome rate per 10 μg/m^3 increase in $PM_{2.5}$ concentrations, and let $\Delta D_{i,\mathrm{cv}}$ represent the incremental increase in number of cardiovascular mortalities for the ith tier::

$$B_{i,\mathrm{cv}} = M_{\mathrm{cv}} \times N_i \tag{4}$$

$$\Delta D_{i,\mathrm{cv}} = B_{i,\mathrm{cv}} \times \left(\Delta H_{\mathrm{cv}} \times \left(\frac{\Delta PM_{2.5,i}}{10} \right) \right) \tag{5}$$

The cost of increased cardiovascular mortality is calculated by multiplying changes in cardiovascular mortality rates by the 2024 value of statistical life of \$13.1 million (Kearsley, 2024), denoted as VSL_{2024}:

$$\text{Cardiovascular Mortality Cost} = \sum_{i=1}^{5} \Delta D_{i,\mathrm{cv}} \times VSL_{2024} \tag{6}$$

[5] When considering hospital inpatient stay statistics, there were 7,971,000 inpatient cardio operations at a cost of \$97.5 billion, indicating \$12,231 per visit. This statistic is not used since the EPA report cites an increase in total cardiovascular hospital admissions due to a change in $PM_{2.5}$ concentrations, not specifically inpatient stays.

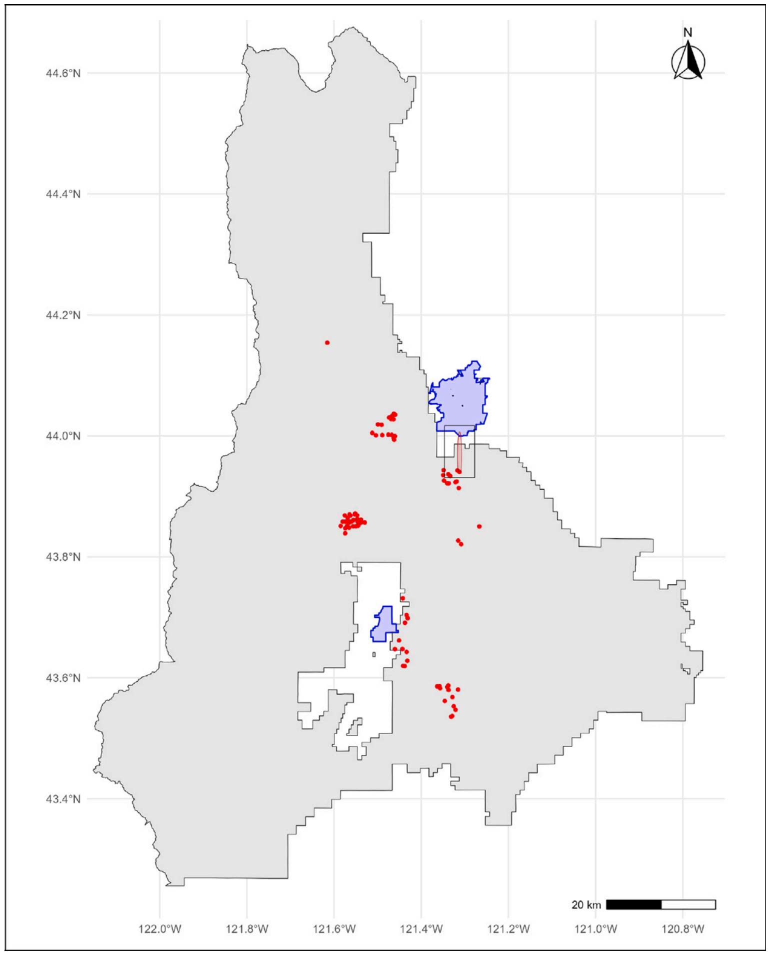


Fig. 3. Locations of brush piles burned in the Deschutes National Forest in 2022. Each point in red represents a recorded pile burn, while the areas in blue represent the city limits of Bend (North) and La Pine (South). The air quality surface shown in Fig. 2 is included to the south of Bend, with the black box representing the map's bounds in Fig. 2.

Using both local cost estimates, we calculate the total local health cost by summing the cost of illness and the cost of increased cardiovascular mortality for each pile in the data. By applying this structured methodology, we aim to comprehensively estimate the local health costs associated with changes in $PM_{2.5}$ concentrations from pile burns.

3. Results

The primary results presented use the baseline assumptions of a 2023 EPA SC-CO_2 of $190 per ton, a 90% consumption factor, the midpoint of the concentration response function of $PM_{2.5}$ exposure on asthma and cardiovascular hospitalization and cardiovascular mortality, and the midpoint of each concentration tier range for assumed $PM_{2.5}$ concentration.[6] The SC-CO_2 proposed by the EPA in 2023 is used because it has extensive evidence for each assumption used in their analysis and aligns with the SC-CO_2 values used in other nations (U.S. Environmental Protection Agency, 2023; Rennert et al., 2022). The midpoints of other assumptions are used to avoid overestimation or underestimation of the effects. A consumption rate of 90% is used, aligning with pile burn literature (Barker et al., 2025; Hardy, 1996). With these assumptions, we estimate that for all (85) piles burned in 2022 in the Deschutes National Forest, the total external cost of all pile burns is $26,023,000, where the average external cost per acre of pile burns is $6,168. Across all pile burns, the total carbon equivalent (CO_{2e}) emissions are 64,222 tons, corresponding to an average global social cost of $2,892 per acre. Similarly, the 85 pile burns produced 250.91 tons of $PM_{2.5}$, resulting in an average local health cost of $3,276 per acre. In this case study, local health costs exceed global GHG costs, though this balance may differ for the other 154 national forests depending on their proximity to population centers.

Fig. 3 shows the locations of all the brush piles burned in the Deschutes National Forest in 2022. The example air quality surface used in Fig. 2 is included in this map as a scaling reference. Because the example air quality surface is representative of the air quality change resulting from a 69-acre pile burn, any larger pile burn will result in a larger air quality surface and potentially affect more people.

In Fig. 4, piles are mapped with color shading based on the total external cost per acre of burning those piles. Local health costs drive much of the variation between external cost calculations and are the source of outliers south of Bend. Since local health costs occur when simulated pile burns occur near major population centers, the piles south of Bend have the highest external costs per acre. Fig. 4 illustrates this relationship, with the piles located directly to the west and south of Bend having the darkest color shading. These piles extend to urban edges and spaces, thereby affecting people and contributing to local health cost estimates.

In contrast, pile burns far from population centers typically have more uniform costs per acre. When a pile burn occurs and the resulting air quality changes do not reach populations, the full cost of that burn is attributed solely to global social costs. Since global social costs are

[6] Summary statistics of several variations of these results based on different assumptions are provided in the Appendix.

Table 2
Mean emissions per acre of slash piles burned across consumption-factor (CF) scenarios (CF = 50%, 70%, 90%). For each CF, we report the sample mean with the 5th–95th percentile confidence intervals in parentheses (the central 90% of pile-level estimates). These results hold all other assumptions constant relative to the baseline assumptions.

Emission type	Consumption factor		
	50%	70%	90%
CO_2 tons per acre	7.82 (4.57 – 12.00)	10.94 (6.39 – 16.80)	14.06 (8.22 – 21.50)
CH_4 tons per acre	0.0132 (0.0078 – 0.0204)	0.0185 (0.0108 – 0.0286)	0.0238 (0.0139 – 0.0367)
$PM_{2.5}$ tons per acre	0.0319 (0.0186 – 0.0492)	0.0447 (0.0259 – 0.0688)	0.0574 (0.0334 – 0.0884)

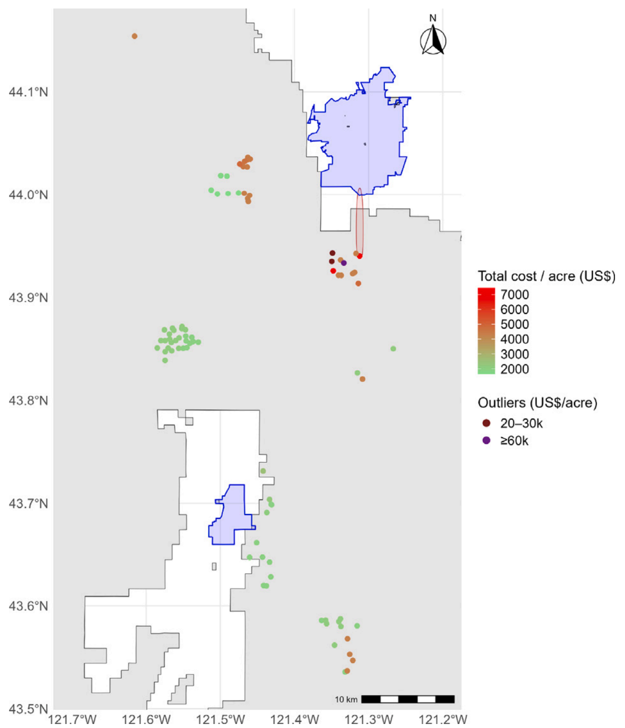


Fig. 4. All burned brush pile locations in the Deschutes National Forest in 2022, where piles are colored based on their estimated external cost per acre. Cities are colored blue, representing Bend (North) and La Pine (South). The air quality shape mapped in Fig. 2 is included as a scaling reference for the size of air quality surfaces.

dependent on the designation of the fuel bed and the fuel loads, which are standardized across fuel beds, piles without a local health cost will have the same cost per acre as other piles with the same fuel bed classification. This is shown in Fig. 4, where the piles clustered in the west have very similar estimates of the external cost per acre due to being assigned the same fuel bed. Conversely, when piles are assigned different fuel beds, there is a clear break in the external cost per acre, as shown with the group of piles in the southeast.

Notably, there are three outliers, with one outlier exceeding $60,000 in total external costs per acre. These high per acre costs are due to the piles being larger, as well as their simulated smoke plumes carrying to the western side of Bend, which is more populated. In addition, the pile with an external cost per acre higher than $60,000 is from the pile directly south of Bend at 180 acres in size, resulting in a large smoke plume that affects more people than any other pile, subsequently resulting in the highest local health cost per acre.

Fig. 5 shows the distribution of per acre local health costs and per acre global social costs for each of the 85 pile burns. Local health costs can vary widely because they depend on whether smoke plumes intersect with nearby population centers, with many piles carrying little

Table 3
Mean global social cost per acre under alternative SCC values. Each cell reports the sample mean with the 5th–95th percentile confidence intervals in parentheses. All other assumptions held constant relative to the baseline.

$SC\text{-}CO_{2e}$	Total global social cost per acre
$51	$749 ($438 – $1149)
$100	$1470 ($859 – $2253)
$190	$2793 ($1633 – $4281)

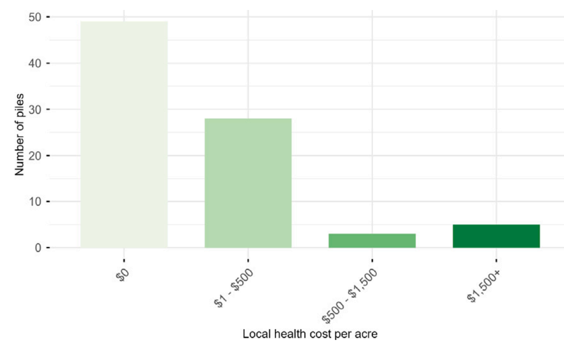
or no local health impact. However, global social costs for piles with the same fuel bed scale linearly with pile size. This is because fuel types are standardized by fuel bed, resulting in the same per acre tonnage of fuels for each emission calculation.

Estimated total external costs are dependent on the assumptions used, which can have varying effects on the final cost calculation. The consumption factor significantly influences both global and local cost estimates. Fig. 6 shows this relationship, where consumption factors are varied while all other assumptions are held constant to the baseline assumptions.

The consumption factor represents the share of the available fuel that is consumed during a pile burn. Thus, for a fuel bed with 14 tons of available burnable material per acre, applying a 50% consumption factor implies that 7 tons of fuel are actually combusted. This affects both local and global cost calculations, since a higher consumption factor will result in larger air quality surfaces and therefore more people being affected. Likewise, a larger consumption factor will linearly increase the total estimated emissions and the subsequent global cost calculations.

While the relationship between the consumption factor and the estimated global social cost is linear, the relationship between the consumption factor and the local health cost is more nuanced. This change is most clearly illustrated in Fig. 6, where between 50% and 70% consumption, the average external cost per acre doubles, while the median goes up less than 50%. This occurs because when the consumption factor is 50%, the air quality surfaces generated reach just outside of Bend. However, when 70% or more consumption is assumed, those air quality surfaces are more extensive and affect a larger population within the city limits of Bend, generating more local costs.

Table 2 shows how the emissions calculations change with different consumption factors. Since the consumption factor applies a percentage reduction in total available fuels burned, that same percentage increase or decrease will be reflected in the resulting emissions. These emissions are used to calculate global social costs, which scale linearly with any selection of $SC\text{-}CO_2$. With our baseline assumptions, on average, a pile emits 14.703 tons of CO_{2e} per acre (CO_2 plus CH_4 converted to CO_{2e} using a factor of 27) at the 90% consumption level. To estimate the global social cost of a burn, this per acre value is multiplied by the pile's acreage and the chosen $SC\text{-}CO_2$.



(a) Local health costs per acre

(b) Global social cost per acre

Fig. 5. Distribution of piles by per acre local health costs and per acre global social costs. Each bar represents the number of piles falling within a given per acre cost range. These results reflect the baseline assumptions.

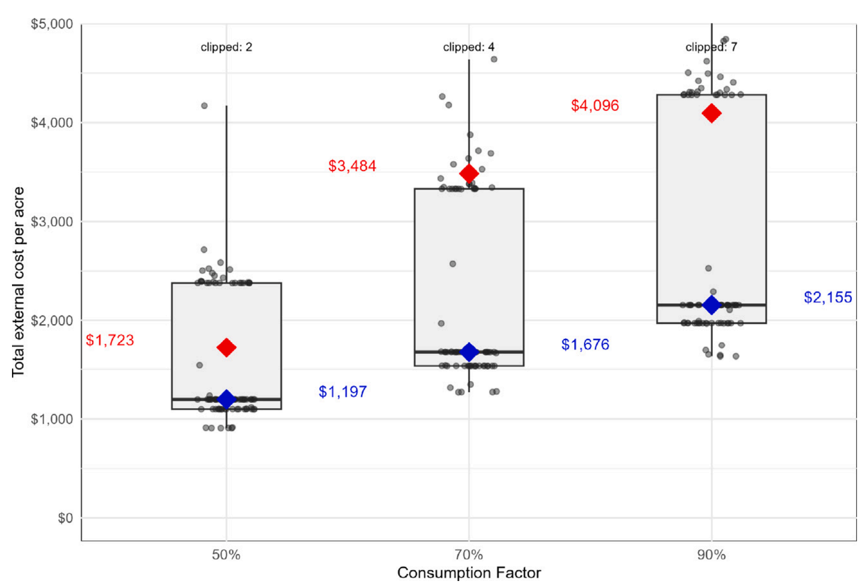


Fig. 6. Distribution of per acre total external costs of pile burns, grouped by the assumed consumption factor used. Here, the 50% column contains data points representing the estimated per acre external cost of all pile burns, assuming 50% of the available material is actually burned. The red diamond represents the mean external cost per acre under that assumption, while the blue diamond represents the median. The bottom of each box is the 1st quartile, while the top of each box is the 3rd quartile.

The SC-CO_2 is a debated topic due to varying assumptions that substantially affect the final calculated value. We provide estimated external costs when applying SC-CO_2 values of $51, $100, and $190. Table 3 illustrates how different values of SC-CO_2 change global social cost calculations.

As an extension of the emissions results, we illustrate how these emissions would change under alternative disposal methods, such as biochar production. Biochar production, a commonly considered alternative to pile burning, can result in 2–40 times lower CO_{2e} emissions, depending on the production method used (Puettmann et al., 2020). If the 64,222 tons of CO_{2e} emissions from piles burned in 2022 were instead processed through biochar production systems, the global CO_{2e} emissions would drop by between 32,111 and 62,616 tons CO_{2e}. At an SC-CO_2 of $190 per ton, this corresponds to an avoided external cost

Table 4
Mean external cost per acre (US$) under alternative assumptions about the health response to $PM_{2.5}$ and the assumed $PM_{2.5}$ exposure within concentration tiers. Rows compare three concentration–response function choices that represent the lower bound (LB), midpoint (MD) and upper bound (UB) of the estimated health effect per $10\mu g/m^3$ unit increase in $PM_{2.5}$. Columns compare three exposure assumptions within each concentration tier (LOW, MID, UP), corresponding to the lower bound, midpoint and upper bound of $PM_{2.5}$ concentration used to calculate local health damages. Each cell reports the sample mean followed by the 5th–95th percentile interval in parentheses. All other model assumptions are held constant to baseline assumptions when computing each cell (see beginning of Results).

		Concentration tier		
		LOW	MID	UP
Concentration Response Function	UB	$3,992 ($1,700, $6,438)	$4,570 ($1,733, $7,509)	$5,430 ($1,781, $9,115)
	MD	$3,672 ($1,682, $5,862)	$4,096 ($1,706, $6,646)	$4,725 ($1,741, $7,823)
	LB	$3,352 ($1,664, $5,285)	$3,621 ($1,679, $5,784)	$4,021 ($1,702, $6,532)

of $6.1–$11.9 million. Similar findings from Susott et al. (2002) find that $PM_{2.5}$ emissions from pile burns can be reduced by a factor of 23 when producing biochar instead, implying even greater cost reductions when considering local health impacts as well.

Similar effects occur when the concentration response function and the assumed concentration are changed within each concentration tier between the upper bound, lower bound, and midpoint of each of their respective ranges. Table 4 provides a two-dimensional sensitivity analysis of total external costs when using different local health impact assumptions. Across columns, costs differ depending on whether the lower, midpoint, or upper bound of the modeled $PM_{2.5}$ exposure range is assumed, while across rows, costs shift depending on whether the concentration–response function (CRF) is specified at its lower bound, midpoint, or upper bound. This layout highlights how the combined variation in exposure assumptions and health response estimates affects the per acre cost estimations.

As an illustration, when all assumptions are held constant relative to the baseline assumptions except for the assumed health concentration response to an increase in $PM_{2.5}$ concentrations,[7] costs can range from $3621 when using the lower bound to $4,570 when using the upper bound of each of those ranges. A similar interpretation can be had for changes to the assumed concentration within each concentration tier of $PM_{2.5}$ exposure, where holding all assumptions constant relative to the baseline assumptions except for the assumed concentration exposure within each concentration tier results in external costs per acre of piles burned changing between $3672 when using the lower bound to $4725 when assuming the upper bound of exposure. Since the range of values is larger when varying the concentration response function, this suggests that the concentration response function has a greater effect on local cost calculations than the assumed concentration within each concentration tier range.

4. Discussion

In this research, we developed a methodology to estimate the external cost of burning brush piles. The methods described here quantify, in dollar terms, a cost to society that is seldom considered in forest management practices. Following this structured approach, we demonstrated substantial costs to society from the continued practice of burning slash piles, which we estimate to be more than $26 million in 2022 for the Deschutes National Forest.

Overall, we estimate that an average ton of burned biomass in a brush pile generates approximately 1.5 tons of CO_2 emissions. This value is comparable to the findings reported by Barker et al. (2025), who estimate that each ton of biomass generates, on average, 1.8 tons of CO_2 emissions in the Western US. The unique characteristics, such as the specific pile size, burn timing, and exact pile composition, are not known with precision for each pile. We therefore estimate the average impact of a pile burn. It is important to note that since we do not take into account all local effects, the total estimated external cost of these pile burns could be lower than the total social cost of the pile burns when all local factors are considered. In future research, this model could be extended to more fuel bed types and burning conditions in different forests, revealing a more comprehensive scope of the total social cost of all piles burned over time.

Beyond these technical extensions, the broader management implications are equally important. Recent work shows that the external effects of slash disposal, particularly pile burning, are often unaccounted for in carbon assessments and rarely influence forest management decisions (Afkhami et al., 2024; Charnley et al., 2015). Since the primary benefit of pile burning lies in fuel reduction, alternative removal strategies such as biofuel or biochar production can achieve the same objective with lower external costs and yield greater net societal benefits. Because these unpriced damages represent an additional cost of thinning, pile burning effectively reduces the net benefits of forest treatments. Incorporating less costly disposal methods could therefore improve the overall benefit–cost ratio of forest management operations. Our findings complement this perspective from the forest operations side. We estimate the external costs of those same emissions and local health costs, thus reinforcing that management choices about slash disposal can meaningfully change the emissions and local health impact of timber production and forest treatments. By attaching a dollar cost to each pile burn, a more comprehensive analysis can be pursued to attain optimal disposal outcomes.

Applying these methods to other settings would require similar pile data, including harvest sizes and pile locations, as well as sufficient population data. Since the BlueSky model is US-focused, this process is not guaranteed to work for international settings. Environmental regulations are only considered to inform burn timing and typical meteorology, with adjustments necessary for analysis in different regions.

Our estimates should be interpreted as illustrative averages under standardized assumptions rather than the exact emissions effects that occurred from each pile burn considered. First, we could not validate BlueSky/VSmoke $PM_{2.5}$ estimates against observed data because many piles have very localized effects that are difficult to find data for, and these effects may overlap, therefore limiting our ability to attribute certain costs to specific piles over 24-hour effects. Nonetheless, prior evaluation by Pan et al. (2020) found that BlueSky emissions estimates perform well against observed air quality data, lending confidence to modeled emissions estimates. Second, exposures are calculated using a 1-km population raster, which can overestimate or underestimate populations at urban edges and sparse rural housing. Third, meteorological and dispersion simulations are simplified, excluding specific plume rise

[7] These will be based on the ranges of concentration response functions for asthma hospital admissions, cardiovascular hospital admissions, and cardiovascular mortality for a 10 μg/m^3 increase in $PM_{2.5}$ concentration as provided in Section 2.5.

schemes, inversions, or lingering smoke. Fourth, the assumptions used may vary further than what is assumed here, where real piles may have varying fuel type compositions, fuel moisture, and pile construction methods, we consider an average pile. Fifth, we quantify the costs of burning, but do not monetize the benefits of reduced wildfire risk, which could be significant. Finally, we do not differentiate vulnerable subgroups such as outdoor workers, children, the elderly, and sensitive groups, who may have higher risks, and we exclude other potential externalities (visibility loss, avoidance costs). Despite these limitations, the framework provides a replicable foundation for incorporating the unpriced health and climate damages of pile burning into future forest management analyses, though results should be interpreted with appropriate caution when generalized beyond the study area.

5. Conclusions

This study develops and applies a framework to estimate the external cost of burning brush piles by combining modeled emissions, population exposure, and economic valuation. Under our baseline assumptions, pile burning in the Deschutes National Forest in 2022 generated substantial global and local health effects, which translate to an external cost of $26 million.

Since wildfire mitigation will continue to produce more brush piles in the future, our results underscore that accounting for the external costs of burning provides strong motivation to evaluate alternative removal strategies such as bioenergy, bioproducts, or biochar. Recognizing that burning these piles can incur significant costs to society provides additional motivation for considering alternative removal strategies. Alternative disposal methods, such as biochar production, can produce up to 40 times fewer CO_{2e} emissions (Puettmann et al., 2020) and 23 times fewer $PM_{2.5}$ emissions (Susott et al., 2002), implying significant avoided global social costs and local health costs when piles are processed rather than burned. Incorporating these external costs with operational costs can shift the feasibility and attractiveness of alternative disposal strategies to help identify piles that are too costly to burn when external damage is considered. Our results indicate that explicitly including the emissions pathway of slash disposal and the resulting external costs could materially affect forest management decision making.

Our estimates are framed as illustrative averages tied to standardized assumptions. Future research can expand on these results by validating modeled emissions when burn timing is more precise and more pile characteristics are available. There is also a pressing need to quantify the avoided wildfire damages to fully assess net benefits of pile burning and other forest treatments. Improved representation of meteorological conditions can also be considered in future research, including changes to wind speed, plume rise, and elevation effects. Exposure estimates can be improved if time-activity patterns are considered and more granular population data are utilized. A key extension is to directly apply these methods and individually assess where alternative disposal is most warranted. This could help land managers and policy makers evaluate practical pathways to reduce both carbon emissions and local health damages from slash disposal while simultaneously creating a marketable product.

CRediT authorship contribution statement

Caleb E. Axlund: Writing – review & editing, Writing – original draft, Visualization, Supervision, Software, Resources, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Jordan F. Suter:** Writing – review & editing, Validation, Supervision, Project administration, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Daniel W. McCollum:** Writing – review & editing, Validation, Supervision, Resources, Project administration, Funding acquisition, Conceptualization.

Author participation

DWM conceived the project. CEA took the lead in data collection and analysis with contributions from JFS and DWM. All authors contributed substantially to interpretation of results and writing the paper.

The authors acknowledge the valuable comments and insights provided by the editor and the three reviewers. The paper is better as a result. Thank you.

Generative AI

During the preparation of this work, the author(s) used the Grammarly writing tool in order to check for spelling and grammar. After using this tool, the author(s) reviewed and edited the content as needed and assumes full responsibility for the content of the publication.

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Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Caleb Axlund reports financial support was provided by USDA Forest Service Rocky Mountain Research Station. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary material related to this article can be found online at https://doi.org/10.1016/j.tfp.2025.101081.

Data availability

Data will be made available on request.

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