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Technical Note

Potential Health Impacts from a Wildfire Smoke Plume over Region Jämtland Härjedalen, Sweden

Andreas Tornevi [1,*], Camilla Andersson [2], Ana Carvalho [2], Joakim Langner [2] and Bertil Forsberg [1,*]

1 Section of Sustainable Health, Department of Public Health and Clinical Medicine, Umeå University, SE 90187 Umeå, Sweden
2 Swedish Meteorological and Hydrological Institute (SMHI), SE 60176 Norrköping, Sweden; camilla.andersson@smhi.se (C.A.); ana.carvalho@smhi.se (A.C.); joakim.langner@smhi.se (J.L.)
* Correspondence: andreas.tornevi@umu.se (A.T.); bertil.forsberg@umu.se (B.F.)

Abstract: In the summer of 2018, Sweden experienced widespread wildfires, particularly in the region of Jämtland Härjedalen during the final weeks of July. We previously conducted an epidemiological study and investigated acute respiratory health effects in eight municipalities relation to the wildfire air pollution. In this study, we aimed to estimate the potential health impacts under less favorable conditions with different locations of the major fires. Our scenarios focused on the most intense plume from the 2018 wildfire episode affecting the largest municipality, which is the region's only city. Combining modeled PM2.5 concentrations, gridded population data, and exposure–response functions, we assessed the relative increase in acute health effects. The cumulative population-weighted 24 h $PM_{2.5}$ exposure during the nine highest-level days reached 207 μg/m^3 days for 63,227 inhabitants. We observed a small number of excess cases, particularly in emergency unit visits for asthma, with 13 additional cases compared to the normal 12. Overall, our scenario-based health impact assessment indicates minor effects on the studied endpoints due to factors such as the relatively small population, limited exposure period, and moderate increase in exposure compared to similar assessments. Nonetheless, considering the expected rise in fire potential due to global warming and the long-range transport of wildfire smoke, raising awareness of the potential health risks in this region is important.

Keywords: PM2.5; asthma; chemistry transport model; forest smoke; health care visits; respiratory health; wildfires

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1. Introduction

Wildfires have a significant impact on air pollution, contributing to the emission of greenhouse gases and air pollutants that are associated with various health problems. The smoke from wildfires contains a range of substances, including carbon dioxide, carbon monoxide, particulate matter, complex hydrocarbons, nitrogen oxides, trace minerals, and numerous other compounds. Recent literature reviews on the community health effects of wildfire smoke have consistently shown a strong association with respiratory morbidity [1,2], particularly in relation to acute effects on asthma [3]. Epidemiological studies have also found associations between exposure to fine particles (particles less than 2.5 μm in diameter ($PM_{2.5}$)) from wildfire smoke and mortality [4,5], as well as a possible association with cardiovascular outcomes [6,7].

Wildfire regimes are influenced by climate, weather conditions, fuel availability, and human activities as ignition sources [8]. Climate change has led to extended fire seasons in the European Boreal forests, posing an increasing threat from forest fires [9]. Lightning-induced wildfires are projected to rise, given the predicted increase in lightning strikes in northern Europe [10]. Lehtonen et al. (2016) used climate model data and found that large (≥10 ha) boreal forest fires in Finland may double or even triple in size by the end of the 21st century under different emission scenarios, albeit with considerable inter-model variability [11]. Using large ensembles of climate model data, Lund et al. (2023) show an increased frequency of the meteorological conditions conducive to high wildfire risk in Fennoscandia and other boreal forest regions and calculate an associated increase in the number of days with a moderate-to-high fire weather index with global warming [12]. On the other hand, changes in climatic conditions and forestry practices are expected to result in an increased presence of broad-leaved trees in Swedish forests. This shift may help reduce wildfire risk and fire emissions since broad-leaf trees locally decrease surface temperatures [13].

Sweden has recently experienced two severe fire years: 2014 and 2018 [14]. The 2014 forest fire occurred in the Västmanland region, approximately 150 km northwest of Stockholm, burning around 14,000 hectares of land, leading to 1000 evacuations and resulting in one fatality. In 2018, the country recorded fires covering more than 24 kha of burnt area. The Gävleborg and Jämtland Härjedalen regions burned the most, each accounting for around 8500 hectares [15].

During the summer of 2018, the episodes of wildfire smoke were most intense in the sparsely populated region of Jämtland Härjedalen. Our previous study aimed to investigate the short-term respiratory health effects resulting from changes in the composition of the atmosphere due to emissions of pollutants from biomass burning [16]. Using the MATCH chemistry transport model, we calculated daily population-weighted concentrations of $PM_{2.5}$ for each municipality in the Jämtland Härjedalen region. We examined the potential respiratory problems associated with $PM_{2.5}$ levels from wildfires by analyzing daily health care contacts related to respiratory complications in each municipality, employing a quasi-Poisson regression model that accounted for weather conditions, weekday patterns, and long-term trends. In Härjedalen, the municipality most exposed to smoke from wildfires, there were nine days with a daily mean $PM_{2.5} > 20\ \mu g/m^3$, which resulted in a significant increase in asthma visits on the same day and the following two days after such an event (RR = 2.64, 95% CI: [1.28–5.47]). A meta-analysis involving all eight municipalities showed a statistically significant increase in asthma visits (RR = 1.68, 95% CI: [1.09–2.57]), as well as an increase in visits for any lower airway disorders (RR = 1.40, 95% CI: [1.01–1.92]).

Härjedalen municipality, with Sveg as its major settlement, had the highest number of wildfire smoke days and the highest population-weighted concentrations of $PM_{2.5}$ during the summer of 2018. However, this municipality has a sparse population, with six times more people residing in Östersund (n = 63,227 as of 1 January 2019), the only city in this region. The objective of this study is to estimate the potential health impacts considering a scenario where the most intense plume from the 2018 wildfire episode affects the largest population in the region—Östersund. For this purpose, we constructed two scenarios.

2. Methods

2.1. Exposure Modeling

Smoke exposure from forest fires in the Jämtland Härjedalen region was modeled by using the chemical transport model MATCH [17,18]. MATCH simulates the evolution of atmospheric concentrations of gas- and particle-phase chemical compounds on an Eulerian grid accounting for emissions, transport, chemistry, and deposition using meteorological input from operational weather prediction systems. MATCH is one of the models in the ensemble used to produce the day-by-day forecast of the atmospheric composition over Europe as part of the services provided by CAMS (Copernicus Atmosphere Monitoring Service). MATCH is also used for the assessment of air quality and deposition in Sweden on behalf of the Swedish Environmental Protection Agency, for reporting to the European Union, and for evaluating the air quality in Sweden's municipalities [19–21]. The chemistry scheme used in MATCH is based on Simpson et al. (2012) [22], with modified isoprene chemistry according to Carter et al. (1996) [23]. The aerosol simulation considers primary and secondary aerosols; the formation of the latter follows the formulations developed by Bergström et al. (2015) and Hodzic et al. (2016) [24,25]. Dry deposition of gases and aerosols to land and water surfaces is described with a resistance model using parameters depending on surface type and season. For vegetated surfaces, the deposition of gases is driven by various meteorological factors, such as vapor pressure deficit, soil moisture, temperature, and photosynthetically active radiation. Precipitation deposition is modeled as being proportional to precipitation intensity for most gaseous and aerosol components, using the parameters proposed by Simpson et al., 2012 [22].

Exposure to PM2.5 was simulated using two nested model domains. In order to capture contributions from long-range transport, a larger domain covering major parts of Europe was used. A smaller, high-resolution domain (with grid cells with lateral lengths of 4 km) centered on the county of Jämtland Härjedalen that extends to other parts of Sweden and Norway was used to capture the detailed distribution and evolution of the wildfire smoke plumes (Figure 1). The large domain had a $0.1° \times 0.1°$ (ca. 6 by 12 km) horizontal resolution.

Figure 1. MATCH domains: in green covering part of Europe, with a $0.1° \times 0.1°$ grid cell size, and in green the smaller domain at 4 km × 4 km resolution covering Jämtland Härjedalen and surrounding areas.

To account for atmospheric emissions from wildfires, we utilized the Global Fire Assimilation System (GFAS) daily product [26]. This product provides data at a resolution of $0.1° \times 0.1°$. We considered the most important compounds provided by GFAS, including gaseous emissions of carbon monoxide (CO), methane (CH4), nitrogen oxides (NOx), non-methane volatile organic compounds (NMVOC), and sulfur dioxide (SO2) as well as emissions of particle-phase elemental carbon (EC), organic carbon (OC), PM10, and $PM_{2.5}$. The emitted compounds of the GFAS database were interpolated to each one of the two MATCH domains' horizontal resolutions. In each horizontal grid cell containing wildfire emissions, these were then distributed from the surface to the top of the plume by applying a parabolic function as in Walter et al. (2016) [27]. After a statistical analysis of the top of the plume during the period of interest, which is also available as a parameter in GFAS, the top of the plume was set at 1500 m. In the temporal dimension, a Gaussian distribution was applied to wildfire emissions that peak during the day (as suggested in Kaiser et al. (2012) [26]).

Anthropogenic emissions of NOx, SOx, NH3, NMVOC, and CO for the European domain were retrieved from the CAMS REG AP_v3.1/2016 database, while emission data from the Nordic WelfAir project emission inventory v5 on a 1×1 km^2 resolution were used for the inner domain.

The meteorological fields used to force the MATCH model are from the operational weather forecasting model (IFS) of the European Centre for Medium-Range Weather Forecasts (ECMWF). These model results, with a temporal resolution of three hours, were then interpolated for the two domains shown in Figure 1. For the European domain, the data were based on Corine land cover with adjustment to EMEP classes (described in Simpson et al., 2012 [22]), while for the smaller domain, which covers the county of Jämtland Härjedalen, the land-use classes were updated using the 2012 Corine land cover dataset. To separate the contribution of total $PM_{2.5}$ from wildfire emissions, two sets of simulations were carried out on the nested domain for June–August 2018, which means that one set includes wildfire emissions from GFAS, while in the other set, these emissions are masked over a region covering almost all of Sweden.

Figure S1 shows the simulated distributions of diurnal maximum PM2.5 over the most affected region in central Sweden for 15–29 July 2018, the time period with the strongest fire impacts on PM2.5. As can be seen, the simulated impact of the fire plumes is widespread, especially on 16–21 July. From 29th July onward, the impact of fires is reduced due to a change in weather conditions and active firefighting. Figure 2, top left panel, shows the simulated distribution of maximum 1 h concentrations over the same region for the period June–August 2018. Locations of meteorological stations and one air quality station are indicated on the map, whereas the locations of the city of Östersund and the small town Sveg are indicated in Figure 2, top right panel. Although the simulated impact of the wildfires is widespread, it is clear that the areas with the highest 1 h concentrations exceeding 200 ug/m^3 are quite narrow. Given the low population density in the region, the highest concentrations may have missed the major population centers in the region during the episode in 2018.

Simulations of air quality over Europe using MATCH have recently been evaluated with observations, e.g., Frohn et al. (2022) [28] and Tsyro et al. (2022) [29], documenting the ability of the model system to simulate realistic distributions of PM2.5 as well as other criteria pollutants. However, observations to validate the details of the simulated distributions in the Jämtland Härjedalen region in connection with the fires in 2018 are scarce. There is only one air quality station, Bredkälen, with surface observations of PM2.5 available in the northern part of the region of interest (see Figure 2). Figure S2 shows a comparison between observed and simulated diurnal average concentrations of PM2.5 at Bredkälen with and without wildfire emission for the period June–August 2018. There is fair agreement between observed and simulated concentrations at Bredkälen over the time period with a correlation of 0.74 and a positive bias of 57% (51% without wildfire emissions). However, the simulated peak due to wildfire emissions on the 20th of July is not seen in the observations. In order to extend the evaluation, we utilized visibility observations from 11 meteorological stations in the region operated by the Swedish Meteorological and Hydrological Institute (Figure S3). It must be emphasized that although fire plumes clearly have an impact on visibility, there is no direct relationship that can be used to relate visibility observations to PM2.5 concentrations. Also, there are other effects such as fog that can affect the visibility observations. Figure S4 shows comparisons between simulated diurnal average PM2.5 and observed inverse visibility for the period June–August 2018. Both PM2.5 and inverse visibility were normalized with the maximum in each time series. Days with precipitation were excluded to minimize the influence of fog and low-level clouds. For the stations located to the south, major peaks in PM2.5 and visibility often coincide suggesting that the fire plumes are causing the observed reduced visibility. For the three stations located to the north (Storlien, Hallåxsåsen, and Korsvattnet), the correlation between PM2.5 and visibility is low, suggesting that the simulated peak in PM2.5 around the 20th of July is exaggerated. Also, several peaks observed in the visibility records are missing in the simulation. This could be due to detection problems in the GFAS fire emission product.

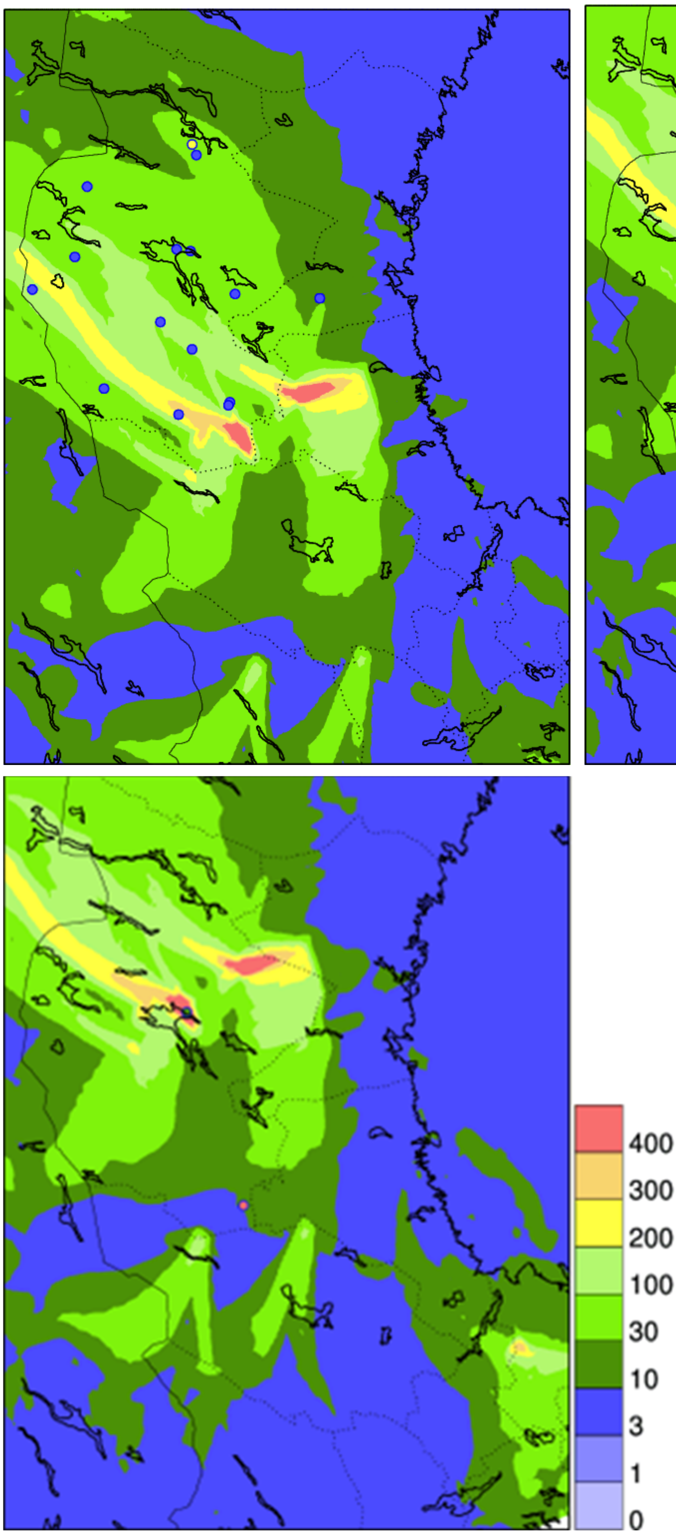

Figure 2. Period maximum 1 h concentration of PM2.5 for the base case (**top left**), scenario I (**top right**), and scenario II (**bottom**). Blue circles (**top left**) indicate visibility observation sites. Yellow circle (**top left**) indicates the PM2.5 observation site Bredkälen. Light blue circles (**top right**) indicate the city of Östersund to the north and the village of Sveg in the center of the map. The location of the geographical and temporal maximum of PM2.5 in the summer of 2018 (**bottom**; pink circle) and the location of the center of the population distribution in Östersund (**bottom**, green circle) that was shifted to the geographical concentration maximum are shown. Unit: μg m^{-3}.

We constructed two exposure scenarios specifically for the city of Östersund. During the 2018 fires, the small town of Sveg was the populated area closest to the largest fire in Sweden (Figure 2, top right panel). Scenario I (Figure 2, top right panel) aimed to illustrate a situation where the larger city of Östersund was exposed to $PM_{2.5}$ concentrations similar to those reaching Sveg. To achieve this, we horizontally shifted the population distribution in Östersund (lat/lon 63.183/14.667) to Sveg (lat/lon 62.035/14.365).

Scenario II (Figure 2, bottom panel) aimed to illustrate a situation where the most adverse plume from the largest fire directly affected Östersund. For this scenario, we horizontally shifted the population distribution in Östersund (lat/lon 63.188/14.658) to the location of the maximum plume (lat/lon 61.762/14.401). The choice of the Östersund location in Scenario II differed from that in Scenario I to ensure that the plume impacted a larger fraction of the more densely populated urban area. Given that the overall forest cover in Sweden in 2020 was 68% and in Jämtland 70% (Statistics Sweden, https://www.scb.se/en/finding-statistics/statistics-by-subject-area/environment/land-use/land-use-in-sweden/pong/tables-and-graphs/land-use-in-sweden-2020/, accessed on 12 September 2023), it is not an unrealistic scenario to assume that Östersund might be directly impacted by nearby forest fires.

2.2. Exposure Simulation Scenario

To estimate the population's exposure to $PM_{2.5}$ concentrations with and without wildfire smoke, we combined the geographically resolved $PM_{2.5}$ concentrations from the simulations with gridded population data for Östersund. The population data were obtained from the National Population Register (100 × 100 m resolution). The daily population-weighted exposure (E^k) was calculated using the following formula:

$$E^k = \sum_{i=1}^{n^k} P_i C_i / P_k \tag{1}$$

For each scenario k, the population P is multiplied by the concentration of the pollutant C in each cell i, summed up to the n grid cells covering the municipality, and normalized by the total population. The population-weighted daily $PM_{2.5}$ exposure from wildfire smoke was then aggregated to calculate the cumulative population-weighted excess exposure over the entire episode, expressed as $\mu g\ m^{-3} \times$ persons × days.

2.3. Health Impact Calculations

To quantify the change in the number of health events due to the increase in $PM_{2.5}$ population-weighted exposure in Östersund caused by wildfire smoke, we employed published exposure–response functions (ERFs) and reported baseline frequencies of health outcomes (Table 1).

Table 1. Included health outcomes with applied relative risks and baselines and their references.

Health Outcome	RR/% Increase	Reference	Baseline
StTM	RR 1.021 (95% CI 1.018, 1.024) per 10 $\mu g/m^3$ increase in daily mean	Chen et al., 2021 [7]	2.9 per day per 100,000 inhabitants of the JH county (2019), National Board of Health and Welfare
StCVDHA	3.68% (95% CI −1.73, 9.09) per 10 $\mu g/m^3$ increase in daily mean	Karanasiou et al., 2021 [6]	5.5 per day per 100,000 inhabitants of the JH county (2019), National Board of Health and Welfare
StRDHA	9.19% (95% CI 5.71, 12.68) per 10 $\mu g/m^3$ increase in daily mean	Karanasiou et al., 2021 [6]	0.6 per day per 100,000 inhabitants of the JH county (2019), National Board of Health and Welfare
StAERV	RR 1.37 (95% CI 1.08, 1.73) per 10 $\mu g/m^3$ increase in daily mean	Calculated from Tornevi et al., 2021 [16]	2.10 per day per 100,000 inhabitants, Tornevi et al., 2021 [16]

The baseline rates reported per year are in the table converted into the number of cases per day in 100,000 persons. The selection of ER functions was based on their relevance to this scenario.

For the calculation of the increase in adverse health outcomes (ΔY), the following equation was used:

$$\Delta Y = Y_{0/100,000} \times (\mathrm{RR} - 1) \times E^T, \tag{2}$$

where Y_0 represents the baseline rate, RR is the relative risk for a one-unit increase in exposure, and E^T is the estimated cumulative population-weighted excess exposure over the entire episode as $\mu g\ m^{-3} \times$ persons $\times$ days.

2.4. Mortality

The short-term effect on total mortality (StTM) was estimated using the ERF for the daily number of deaths from the largest study identified [5]. The study reported a relative risk of 1.021 (95% CI: 1.018–1.024) per 10 $\mu g/m^3$ increase in $PM_{2.5}$ at lag 0, which was used in our calculations. Additionally, for a three-day moving average, the relative risk was 1.019. The effect of wildfire smoke exposure on mortality tended to diminish after three days.

2.5. Cardio-Respiratory Hospital Admissions

A recent review paper on biomass burning (BB) emissions, following the WHO Global Air Quality Guidelines protocol for systematic reviews and meta-analyses on air pollutants and health effects, identified 63 studies on the impact of BB emissions, with a focus on wildfires, on cardiorespiratory morbidity [6].

To estimate the short-term effect on cardiovascular hospital admissions (StCVDHA), we utilized the overall estimate from ten included studies evaluating $PM_{2.5}$ originating from BB. This resulted in a summary estimate of a 3.68% (95% CI −1.73, 9.09) increased risk of total cardiovascular admissions and emergency visits per 10 $\mu g/m^3$ increase in $PM_{2.5}$.

For asthma hospital admissions, we considered nine papers that resulted in a meta-estimate of a 9.19% (95% CI 5.71, 12.68) increase per 10 $\mu g/m^3$ increase in $PM_{2.5}$. This relative risk was applied to admissions for chronic respiratory diseases in the lower airways (StRDHA).

2.6. Respiratory Emergency Visits

The short-term effect on acute visits for asthma to primary health care centers and emergency units (StAERV) is estimated from our epidemiological study of wildfire smoke in 2018 and health care utilization in eight municipalities in the Jämtland Härjedalen region [16]. For this health impact assessment, we recalculated a linear ERF of 1.37 per 10 $\mu g/m^3$ increase in PM2.5 at lag 0 for the most highly exposed municipality Härjedalen.

3. Results

Figure 3 displays the modeled daily mean population-weighted exposure to $PM_{2.5}$ from wildfire smoke in Östersund for scenario II, which was used for the impact calculations. Figure 4 shows the corresponding daily maximum 1 h concentration (not used for impact calculations).

In scenario I, the sum of population-weighted 24 h $PM_{2.5}$ exposure from wildfire smoke was 101 $\mu g/m^3$ days. Among the nine days, they contributed to 97.3 $\mu g/m^3$ days or 96.3% of the total exposure. Considering the population, the total population exposure in Östersund due to the wildfire smoke was 6,386,559 $\mu g/m^3$ person days, with the nine most contributing days accounting for 6,151,987 $\mu g/m^3$ person days. The estimated increase in cases was calculated for these nine influential days. In scenario II, the sum of exposure from the nine days amounted to 207 $\mu g/m^3$ days, resulting in a population exposure of 17,102,904 $\mu g/m^3$ person days.

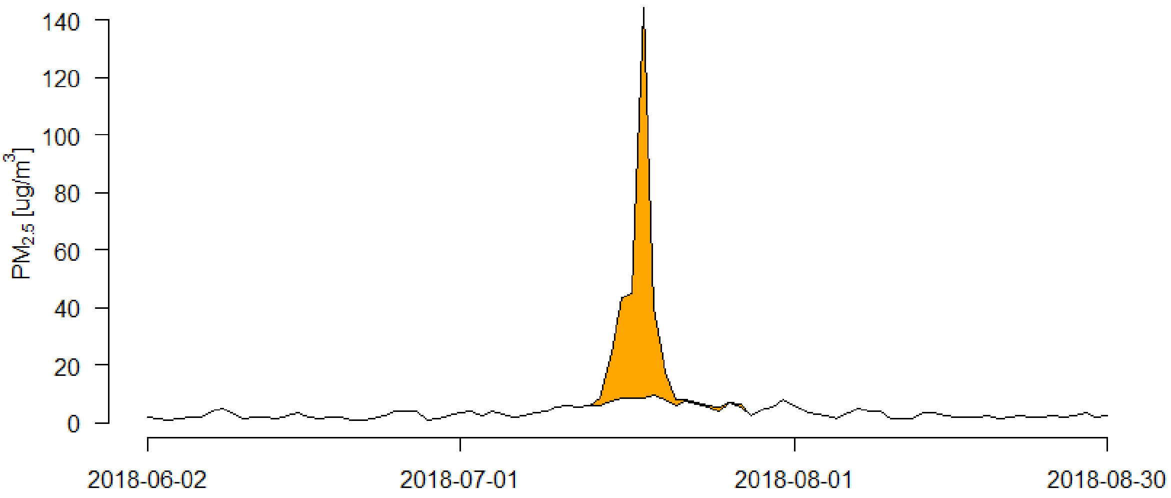

Figure 3. Illustration of the simulated $PM_{2.5}$ exposure data (daily mean) with wildfire-generated concentrations highlighted in orange.

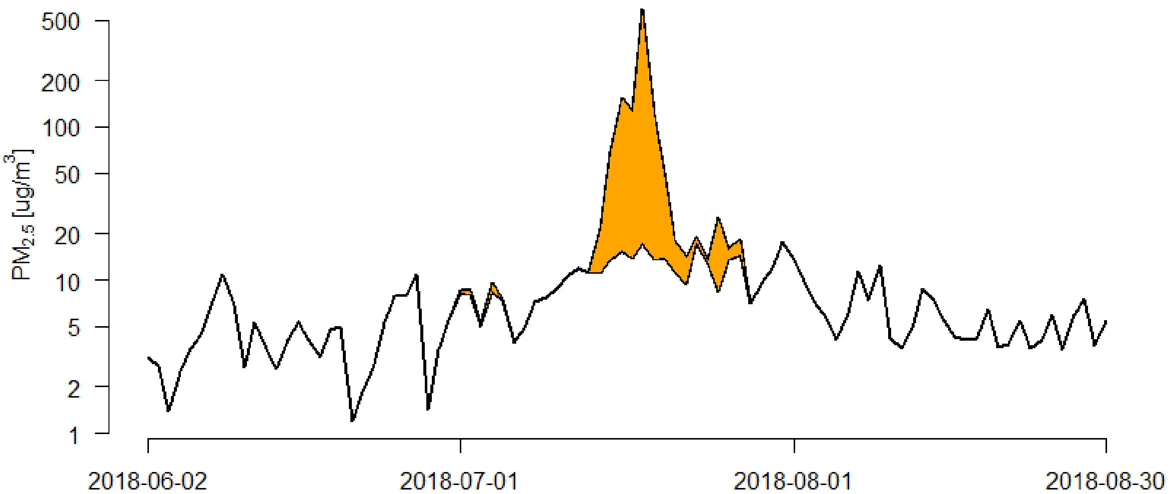

Figure 4. Illustration of the simulated $PM_{2.5}$ exposure data (mean of 1 h maximum per day) with wildfire-generated concentrations highlighted in orange (logarithmic *y*-axis).

Table 2 presents both the expected number of cases during the nine days without wildfire smoke based on regional baselines and the estimated number of excess cases due to the modeled exposure to wildfire $PM_{2.5}$ in the two scenarios. Over the nine days without fire exposure, the expected numbers range from 3 cases (respiratory disease hospital admissions) to 31 cases (cardiovascular hospital admissions), resulting in low numbers of excess cases estimated. The largest effect is observed in emergency unit visits for asthma, with 13 excess cases in scenario II (Table 2).

Table 2. The expected number of cases during 9 days without wildfire smoke and the estimated number of excess cases due to the smoke exposure in the two scenarios.

Health Outcome	Baseline Number	Scenario I Excess Cases (95% CI)	Scenario II Excess Cases (95% CI)
StTM	16.5	0.37 (0.32, 0.43)	1.04 (0.89, 1.19)
StCVDHA	31.3	1.2 (−0.59, 3.08)	3.46 (−1.63, 8.55)
StRDHA	3.4	0.34 (0.21, 0.47)	0.94 (0.59, 1.30)
StAERV	11.9	4.8 (1.03, 9.43)	13.3 (2.87, 26.1)

4. Discussion

In previous health impact assessments, different methods have been used to estimate the actual exposure to smoke at higher concentrations and in larger populations [30]. We aimed to study a "worst-case scenario" in this sparsely populated region. Yet, the effects on the studied endpoints were determined to be minor. Several factors contributed to this result, including the relatively small population size, the limited exposure period of nine days, and the moderate increase in $PM_{2.5}$ exposure compared to similar health impact assessments conducted in other regions [30,31]. For instance, the wildfire smoke episode in Washington State in 2020 alone was projected to have led to more than 90 excess mortality cases [32].

While our study considered some specific health outcomes in the overall population, it would be good to also explore other and milder impacts of wildfire smoke exposure on vulnerable populations, such as individuals with pre-existing respiratory conditions, the elderly, and children. These subpopulations may experience heightened susceptibility and broader health risks from wildfire smoke, and understanding their specific vulnerabilities could guide targeted public health interventions.

Much higher exposure to wildfire smoke at northern latitudes has been recorded elsewhere. Hahn et al. (2021) reported monthly mean $PM_{2.5}$ in July on wildfire days of 98.33 ug/m^{-3} (STD 61.97) for the time period 2008–2019 in Fairbanks, Alaska [33]. The corresponding concentrations on non-wildfire days were 8.73 ug/m^{-3} (STD 17.64). This corresponds to a 10–30 times higher exposure than estimated in our scenario II, so clearly, consequences could be much worse if fires in Sweden were to become more intense and of longer duration. Our study focused on short-term health impacts during the specific wildfire episode. However, there is growing evidence suggesting potential long-term health effects of wildfire smoke exposure, including respiratory and cardiovascular conditions. Discussing these long-term effects and the need for longitudinal studies could provide a comprehensive understanding of the health risks associated with wildfires.

During the same period when the study area was exposed to $PM_{2.5}$ from wildfires, unusually high temperatures were also recorded. Previous research conducted in Stockholm, Sweden, using a piece-wise linear model, described the effect of heat on daily mortality [33]. From the 90th percentile of the 0–2 lag maximum temperature (21.7 °C), the model estimated a risk increase of 1.4% per degree increase. Although not directly applicable to Östersund, which has a different climate, it can be crudely estimated that the heat alone would result in an additional number of deaths. However, further analysis would be needed to determine the specific impact.

It is important to note that our impact calculations are subject to various uncertainties, with major uncertainties relating to the validity of the exposure assessment and the inclusion of exposure–response functions. Additionally, due to data limitations, we had to use regional annual mean baselines for some of the health outcomes studied. With regard to the exposure assessment, the wildfire emissions themselves are most likely the dominant uncertainty including also the vertical distribution of these emissions. Uncertainties in the type of satellite-based emission estimates used here depend on the quality of fire-activity data as well as on the knowledge and availability of vegetation data and cover, fuel conditions and loadings, and emission factors. AMAP (2021) compared estimates from five different satellite-based fire emission products for the Arctic, including GFAS which was used here. They report differences of about a factor of five between the lowest and highest estimates for BC and PM2.5 aggregated annually and over different latitude bands from 45° N to 80° N [34]. The GFAS estimates were in the middle of the range of estimates for BC and PM2.5. Clearly, uncertainties for smaller domains and episodes can be larger. Unfortunately, the exposure modeling could not be directly validated since there were no measurements available in the vicinity of the major fire plumes.

Exposure–response functions used in epidemiological studies of wildfire smoke have shown significant variability among individual studies [6] and even between different years within the same region [35]. Furthermore, the choice of exposure model has been found to impact epidemiological results related to wildfire smoke and morbidity [36] (Gan et al., 2017). It is also worth noting that there may be additional health impacts beyond those included in our study due to wildfire smoke exposure [2].

Although distinct episodes of wildfire smoke are not common in northern European countries, studies have indicated that long-range transport of wildfire particles could have short-term effects on mortality in Nordic countries [7,37]. Given the expected increase in fire potential due to global warming and the ability of wildfire smoke to travel long distances, it is still essential to consider proactive measures to mitigate and adapt to the potential health risks of future wildfire events. This may include improving smoke forecasting and early warning systems, implementing effective communication strategies to raise public awareness, developing guidelines for vulnerable populations, and enhancing urban planning to minimize exposure to wildfire smoke.

In conclusion, while our findings suggest minor health impacts in Östersund during the studied wildfire smoke episode, it is important to acknowledge the uncertainties and limitations of our analysis. Continued monitoring, research, and preparedness for future wildfire events are essential to fully understand and mitigate the potential health risks associated with wildfire smoke exposure.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/atmos14101491/s1, Figure S1: Simulated diurnal maximum concentration of PM2.5 in the region affected by forest fires in central Sweden 2018 for 15–29 July 2018. Unit: ug m^{-3}; Figure S2: Daily mean concentration of PM2.5 at Bredkälen. Modelled total PM2.5 (mod tot), modelled PM2.5 excluding wild fire contribution (mod noWF) and observed (obs) PM2.5. Unit: ug m^{-3}; Figure S3: Station locations and names for meteorological stations with visibility observations; Figure S4: Model simulated diurnal average PM2.5 (blue) and observed diurnal average inverse of visibility (1/m, red) at meteorological stations operated by the Swedish Meteorological and Hydrological Institute in the region hit by fires in 2018. For station locations see Figure S3. Both PM2.5 and inverse of visibility were normalized to one using the maximum value in each time series. Correlations (R) are provided in the top right corner for each station.

Author Contributions: Conceptualization and methodology, B.F., A.T. and C.A.; validation, J.L., A.C. and C.A.; formal analysis, A.T., C.A., B.F., A.C. and J.L.; investigation, A.T., C.A. and B.F.; data curation, C.A., J.L. and A.C.; writing—original draft preparation, A.T., B.F. and C.A.; writing—review and editing, A.T., J.L., B.F., C.A. and A.C.; visualization, A.C., C.A. and J.L.; project administration, J.L. and B.F.; funding acquisition, J.L. and B.F. All authors have read and agreed to the published version of the manuscript.

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