Smoke health costs and the calculus for wildfires fuel management: a modelling study

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Summary

Background Smoke from uncontrolled wildfires and deliberately set prescribed burns has the potential to produce substantial population exposure to fine particulate matter ($PM_{2·5}$). We aimed to estimate historical health costs attributable to smoke-related $PM_{2·5}$ from all landscape fires combined, and the relative contributions from wildfires and prescribed burns, in New South Wales, Australia.

Methods We quantified $PM_{2·5}$ from all landscape fire smoke (LFS) and estimated the attributable health burden and daily health costs between July 1, 2000, and June 30, 2020, for all of New South Wales and by smaller geographical regions. We combined these results with a spatial database of landscape fires to estimate the relative total and per hectare health costs attributable to $PM_{2·5}$ from wildfire smoke (WFS) and prescribed burning smoke (PBS).

Findings We estimated health costs of AU$ 2013 million (95% CI 718–3354; calculated with the 2018 value of the AU$). $1653 million (82·1%) of costs were attributable to WFS and $361 million (17·9%) to PBS. The per hectare health cost was of $105 for all LFS days ($104 for WFS and $477 for PBS). In sensitivity analyses, the per hectare costs associated with PBS was consistently higher than for WFS under a range of different scenarios.

Interpretation WFS and PBS produce substantial health costs. Total health costs are higher for WFS, but per hectare costs are higher for PBS. This should be considered when assessing the trade-offs between prescribed burns and wildfires.

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Introduction

Since 2017 different regions in the world, including Australia, Canada, the USA, Brazil, and parts of Europe, have been affected by unprecedented wildfires (also called bushfires, forest fires, or wildland fires).[1,2] Wildfires can have large effects on society, including injuries and loss of life, destruction of homes and other infrastructure, disrupted productivity and water supplies, and pollution of the air by smoke.[3] Wildfire smoke is a mixture of airborne particles and gases, many of which are harmful to health. At a population level, the most important and best studied pollutant is fine particulate matter ($PM_{2·5}$), which has a similar spectrum of health effects to the well characterised $PM_{2·5}$ from sources such as industrial and traffic emissions.[4] $PM_{2·5}$ from wildfire smoke is associated with an increase in a myriad of adverse health outcomes, including exacerbations of asthma and chronic obstructive pulmonary disease and increases in cardiovascular and respiratory hospital admissions and all-cause mortality.[4,5] Other pollutants in smoke include carbon monoxide, nitrogen oxides, ozone, and volatile organic compounds; the specific composition of smoke constantly changing due to ongoing chemical reactions and the accumulation and deposition of particles.[4] Smouldering and incomplete combustion produce a wider range of compounds and more severe pollution than more intense flaming combustion. $PM_{2·5}$ is accepted as the most appropriate pollutant for the assessment of the health effects of fire smoke, but it is acknowledged that analysis of $PM_{2·5}$ alone will not capture the full range of health harms caused by the entire smoke mixture.[6]

Prescribed burning, also referred to as planned burning or hazard reduction burning, is the planned application of fire to a landscape. The generic term, landscape fire, encompasses all fire types including wildfires and prescribed burns. Prescribed burning is commonly used to modify fuel loads to reduce the risks associated with future wildfires by making them more controllable.[7,8] Prescribed burning is also done for cultural, ecological, and silvicultural purposes. Non-burning approaches for fuel management include the manual thinning of woody fuels, and the mowing or grazing of grass fuels; however, non-burning approaches are typically applied at local scales, whereas prescribed burning is applied at the broad landscape scale.

Like wildfires, smoke produced by prescribed burning has a negative effect on air quality and health.[9] Prescribed fires, although generally smaller in scale than wildfires, are frequently done at interfaces between urban and rural environments to protect the built environment;

Research in context

Evidence before this study

Current evidence shows that air pollution, and specifically fine particulate matter ($PM_{2·5}$), is associated with a wide range of adverse health effects, including but not restricted to, increased mortality and admissions to hospital for cardiovascular and respiratory conditions. Multiple studies have estimated the health burden attributable to short-term and long-term exposure to $PM_{2·5}$, and its associated health costs, by implementing health impact assessments at local, national, and global scales. Since 2012, several studies have applied these methods to systematically assess the impacts associated with elevated smoke-related $PM_{2·5}$ attributable to landscape fire activity, including wildfires and prescribed burns. Some studies have shown that prescribed burns, which tend to be smaller in duration and extension, might produce large increases in $PM_{2·5}$ and public health impacts, when large, populated areas are affected. However, only one study has assessed the relative health impacts and costs of wildfire smoke (WFS) and prescribed burning smoke (PBS), and this study was restricted to days when the average daily $PM_{2·5}$ was above 25 μg/m³.

Added value of this study

This study assessed the historical smoke-related health costs attributable to WFS and PBS, over a 20 year period, in New South Wales, Australia. We combined a spatial fires database with publicly available air quality, health, demographic, and economic data to attribute smoke-related health costs to wildfires and prescribed burns. We included the use of machine learning methods, particularly random forest, for two purposes: 1) to classify days as being affected by WFS or PBS, when more than one fire type was active on the same day and 2) to estimate daily wood heater smoke, to exclude it from the analysis. We then estimated the total and per hectare costs for WFS and PBS. Our results establish new estimates for the smoke health costs of these two major types of fire. Total costs are higher for WFS than for PBS, but on a per hectare basis they are higher for PBS. Our sensitivity analyses show that these results remain consistent across different scenarios.

Implications of all the available evidence

Our study found similar results compared with other studies with respect to the level of landscape fire smoke $PM_{2·5}$ exposure: exposure was relatively larger for WFS compared with PBS. Our results also suggest that smoke-related health impacts are coupled with area burned in both wildfires and prescribed burns, with estimated health costs per area burned higher for prescribed burns than for wildfires. There is considerable variation in health impacts at a regional level, and important inter-regional effects of $PM_{2·5}$ exposure and public health outcomes. Current evidence for New South Wales shows that to reduce the risk of burning of 1 hectare due to a wildfire, 3–6 hectares need to be burned through prescribed fires. Given that our results always yield higher per hectare health costs for PBS than for WFS (between 1·2-times and 4·6-times higher), it would be expected that if prescribed fires impacts follow the historical patterns described in our analysis, then the smoke from these practices might produce health costs at least 3·6 times higher than the avoided smoke impacts from wildfires. These results should be investigated elsewhere in Australia and other fire prone regions in the world, and incorporated into the population risk calculus for wildfire management strategies.

therefore, they usually occur closer to populated areas.[10] Although $PM_{2·5}$ emissions tend to be lower following prescribed burning, population exposure to air pollution can be substantial because of the closer proximity of the fires, the more frequent exposure, and weather conditions that generally favour accumulation rather than dispersal of smoke.[11–13] To better understand the existing trade-offs and effectiveness of fire management tools, such as prescribed burning, the relative public health burden attributable to wildfire and prescribed fire smoke exposure needs to be considered.[11,14] An improved understanding of the public health burden is especially important in the context of a changing climate in which extreme wildfires might increase both in magnitude and frequency[15] and the availability of suitable days for prescribed burning might be restricted or changed in seasonality.[16,17]

Health impact assessments have been extensively used to estimate the health impacts associated with air pollution;[18] and in recent years these methods have been increasingly applied to assess the health impacts of $PM_{2·5}$ associated with smoke from wildfires and prescribed burns.[2,19–23] Nevertheless, little is known about the relative contribution of wildfires and prescribed burns to smoke-related health impacts.A study published in 2020 estimated the health impacts attributable to smoke from wildfires and prescribed burns for Western Australia, Australia, but the analysis was restricted to days with daily $PM_{2·5}$ concentrations exceeding the national daily standard (25 μg/m³), which excluded a large number of days with moderate pollution levels.[19]

Between October, 2019, and February, 2020, the eastern seaboard of Australia was affected by an unprecedented fire season, with more than 7 million hectares of Eucalyptus forests burned causing harm to biodiversity and substantial carbon emissions.[24,25] The fires caused harm to human populations through the destruction of more than 3000 homes, 33 direct deaths, widespread social trauma, and chronic smoke pollution.[24] The state of New South Wales, Australia, was the most affected, and estimates show that the $PM_{2·5}$ that covered the state during this period probably translated into an additional 200 deaths, 1600 hospital admissions, and 700 asthma emergency department visits, with health costs of more than AU$1 billion.[2,26] This catastrophic event triggered a national inquiry: the Royal Commission into National

Natural Disaster Arrangements.[27] The inquiry noted the importance of fuel management in mitigating fire risk through the application of Aboriginal traditional knowledge and use of deliberately set fires (prescribed burns) to reduce fuel loads. Yet, the inquiry highlighted that smoke generated through these activities might pose significant harm to the population, particularly for vulnerable groups (eg, children, adults aged 65 years and older, and people with pre-existing health conditions).[27] Resolving the trade-offs of the benefits of bushfire risk reduction through intentional burning against health costs remains a major science and policy challenge, which accordingly demands economic analyses of costs and benefits across different sectors of the economy.

New South Wales is an ideal setting for such an analysis because of the flammable native Eucalyptus forests close to many large population centres and a longstanding programme of prescribed burning to manage the risks associated with severe wildfires. In the Sydney region about 1% of forests get treated with prescribed burns each year, often in close proximity to large populations;[10] additionally an average of 4% of forests are burnt by wildfire per year.[28] Previous studies in New South Wales have found that to effectively reduce the area burned by wildfires an area about three-times as large needs to be burned through prescribed fires.[28] Furthermore, New South Wales is also an ideal setting because it has a reliable fire history, well developed air quality monitoring network, and regularly updated health statistical information that provide the necessary data for such an analysis. In this study, we aimed to estimate the health costs of $PM_{2·5}$ attributable to wildfire smoke (WFS) and prescribed burning smoke (PBS) in New South Wales, and to estimate a per hectare health cost indicator to help incorporate the potential fire smoke-related health impacts of fire management practices.

Methods

In this modelling study, we applied a quantitative health impact assessment framework to estimate the health costs of $PM_{2·5}$ attributable to WFS and PBS in New South Wales, since July, 2000. Additionally, we estimated a per hectare health cost indicator, to help incorporate the potential fire smoke-related health impacts associated with fire management practices.

We estimated the health burden attributable to landscape fire smoke (LFS) in New South Wales between July 1, 2000, and June 30, 2020. We combined a spatial landscape fires database containing detailed characterisation of all registered fires within the state, with estimated daily health costs attributable to LFS-related $PM_{2·5}$ (figure 1). The analysis approach used the following three steps: (1) we analysed the National Parks and Wildlife Service (NPWS) fires database, cleaned the data, estimated the daily area burned by each fire type (wildfire or prescribed burn), and applied a machine learning algorithm (random forest) to classify each day across the state as being affected by WFS or PBS; (2) we did a health impact assessment to quantify the LFS-attributable health impacts and estimate the daily smoke-related health costs; and (3) we combined results from the first two steps to estimate the total and per hectare health costs attributable to WFS and PBS. For the health impact assessment, we estimated the burden for all-cause mortality, hospitalisations for circulatory and respiratory diseases, and emergency department presentations for asthma-like illnesses, using previously tested methods.[2,19–21] Our main analysis considered the entire state of New South Wales as a single spatial unit, but we also did a regional analysis, which considered grouping fires and health impacts according to their location using the eight geographical administration branches defined by the NPWS (appendix p 3).

See Online for appendix

Procedures

We used the NPWS Fire History database,[29] and considered fires within New South Wales that were active during the study period (July 1, 2000, to June 30, 2020). Information in this database included fire season (12 consecutive months between July and June), fire type (prescribed burn or wildfire), start date, end date, and area burned (appendix p 35). Every fire is given a unique identifier in the NPWS database. We identified duplicate fires, based on the NPWS identifiers, and wrong or missing dates, and used the following procedure to estimate the daily area burned for each fire. First, we identified fires that had missing start and end dates and those with complete start and end dates. Second, for fires with complete start and end dates, we estimated the daily area burned for each fire. Daily burned areas were calculated as the total area burned divided by the number of days the fire was active (from start day to end day). Third, for each fire season and fire type, we aggregated the daily area burned for fires with complete start and end dates (identified in step two) and estimated the daily area burned distribution. Finally, for fires with missing start and end dates, we assumed that their total area burned had the same temporal distribution as those in step three (ie, we assumed that fires with missing dates had the same temporal distribution as those with complete dates).

Once we had an estimation of the daily area burned for each fire, we used these results to identify if each day was dominated by PBS or WFS. For each day, we calculated the proportion of area burned by a wildfire (perc ha wf), and then classified each day according to the likely dominant LFS category.

We used a random forest model machine learning algorithm to predict the dominant LFS category for each day. A random forest is a supervised learning algorithm that can be trained with observations having known categories (eg, wildfire or prescribed burn) to predict these categories on a new set of observations.[19,30] First, we identified days that were dominated by wildfires (perc ha wf $\geq 0·9$) or prescribed burns (perc ha wf $\leq 0·1$).

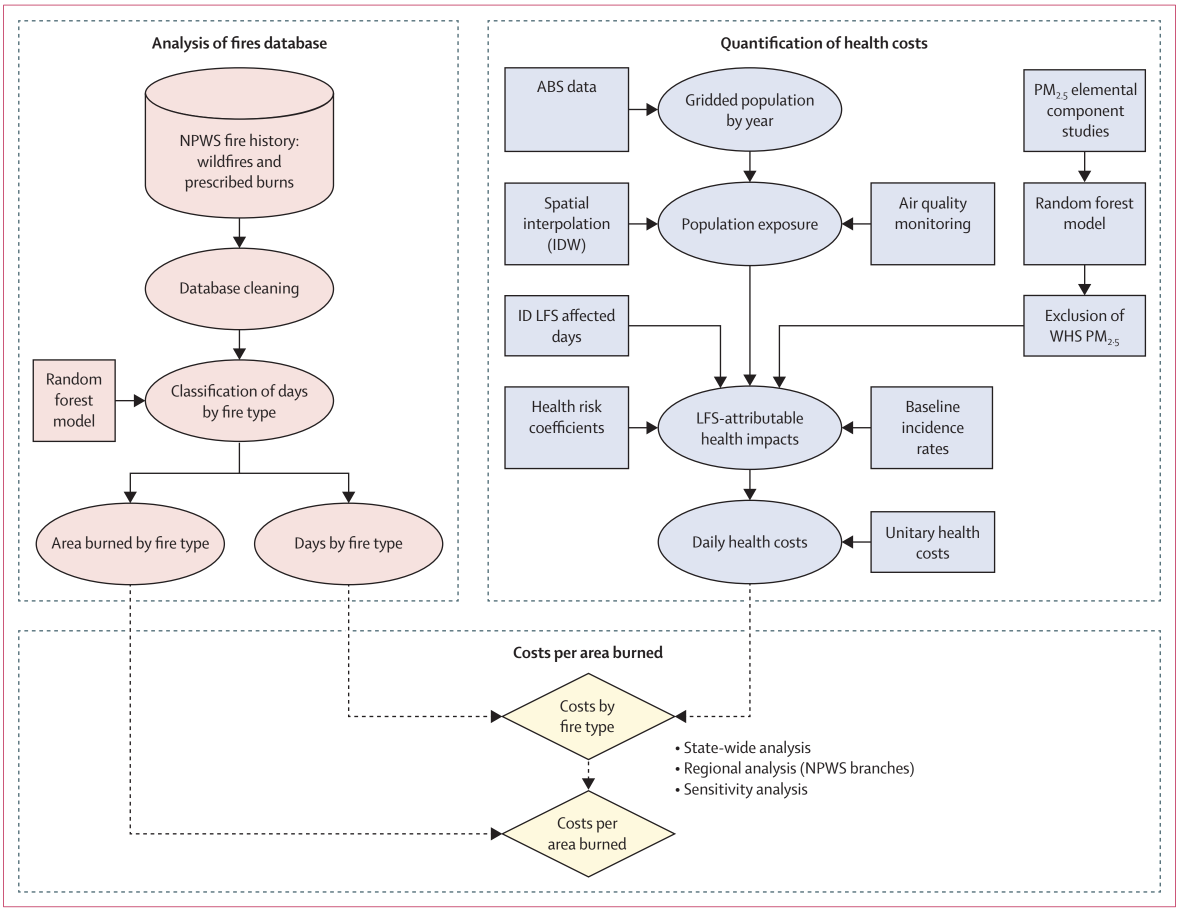

Figure 1: **Summary diagram for analysis**
ABS: Australian Bureau of Statistics. ID=identifier. IDW=inverse distance weighting. LFS=landscape fire smoke. NPWS=National Parks and Wildlife Service. WHS=wood heater smoke.

We trained a random forest model with this data, and used this model to predict the LFS category on other days (0·1< perc ha wf <0·9). This model used the following explanatory variables: year, month, week, weekday, fire season, area burned, mean temperature, minimum temperature, maximum temperature, rainfall, solar exposure, vapour pressure, and relative humidity. With this model, we achieved an accuracy score of over 96% (appendix pp 4–6).

In the health impact assessment we estimated health costs for each day (d) using the following equation:

$$\mathrm{Costs}_d = \sum_0 [\mathrm{IR}_0 \times \mathrm{Pop}_d \times (e^{\beta 0 \times \Delta \mathrm{Cd}} - 1)] \times \mathrm{UnitCost}_0$$
$$= \sum_0 [\mathrm{Cases}_0 \times \mathrm{UnitCost}_0$$

We used baseline incidence rates (IR_0) for all-cause mortality, circulatory disease, and respiratory disease hospitalisations, and asthma emergency department presentations (outcome o), by state and year; exposed population (Pop_d) by Statistical Area Level 2 (SA2; medium sized geographical area with average population of 10 000 people) and year; and exposure response functions or health risk coefficients (β_0) for short-term exposure to $PM_{2\cdot5}$. For unit costs ($UnitCosts_0$) we used the value of a statistical life (VSL) for mortality, and Australia-specific average costs for morbidity, adjusted to the value of the Australian dollar in 2018 (appendix pp 7–8).

LFS-related $PM_{2\cdot5}$ exposure (ΔC_d) was calculated as the difference between the estimated daily $PM_{2\cdot5}$ for days identified as being affected by LFS and the long-term mean daily $PM_{2\cdot5}$ by SA2 and month, excluding the contribution of wood heater smoke. To estimate exposure for each SA2, we first used station level $PM_{2\cdot5}$ data, and applied an inverse distance weighting method to interpolate values at mesh block (smallest geographical area defined by the Australian Bureau of Statistics) centroids, and then calculated a population weighted $PM_{2\cdot5}$ for each SA2 (each SA2 is composed of multiple mesh blocks). We considered a day to be affected by LFS whenever the daily $PM_{2\cdot5}$ on any given SA2 was above the long-term (2000–20) 95th percentile for that SA2 month, a method that has already been validated in Australian settings.[2,31,32] Additionally, we excluded 105 days having high particulate matter levels due to the presence of dust storms between 2002 and 2018.[33] The wood heater smoke mass contribution has been previously estimated for a series of sites across New South Wales for different periods between 1998 and 2015 (appendix pp 9–15). We used this data, to train a random forest model, and estimate the contribution of wood heater smoke to daily $PM_{2·5}$ across the study period. We fitted the random forest model with the following explanatory variables: daily $PM_{2·5}$, day of the week, longitude, latitude, solar exposure, mean temperature, year, daily change in temperature (maximum temperature minus minimum temperature), minimum temperature, day of the year, relative humidity, maximum temperature, month, heating degree days, day, and rainfall.

To assess the association between health cost and burned area, we estimated an average cost per hectare by fire type, by fitting a linear regression with the following shape, and estimated the goodness of fit (measured by R^2):

$$\text{Health cost}_{ft,t} = \beta_0 + \beta_{1,ft,t} \times \text{Area burned}_{ft,t}$$

Health $\text{cost}_{ft,t}$ was the total estimated health cost for fire type (ft) aggregated for time (t); β_0 represented the intercept (we assumed β_0=0; ie, health costs attributable to LFS were zero when there was no area burned). Area $\text{burned}_{ft,t}$ represented the total area burned by fire type (ft) aggregated for time (t). $\beta_{1,ft,t}$ represented the regression coefficient or slope (in this context meaning health cost per area burned, $ per hectare). We estimated this relationship for all LFS days, WFS days, and PBS days. We also considered alternative fire day type classifications and temporal grouping variables as detailed in the sensitivity analyses.

To estimate health costs per hectare for the regional analysis, we aggregated the 8 NPWS branches into 5 groups: (1) Greater Sydney and Blue Mountains, (2) Hunter Central Coast, (3) Northern branches (North Coast and Northern Inland branches), (4) Southern branches (South Coast and Southern Ranges branches), and (5) West branch. Additionally, we looked at correlation matrices and did visual analyses to assess for inter-regional effects, that is, smoke produced in one region affecting the population health of other regions. We estimated the Pearson correlation coefficient using R (4.0.2), between area burned and health costs between regions.

Sensitivity analyses

We considered the baseline incidence rates and the health risk coefficients, with 95% CI calculated for both variables. Because of the potential for misclassification

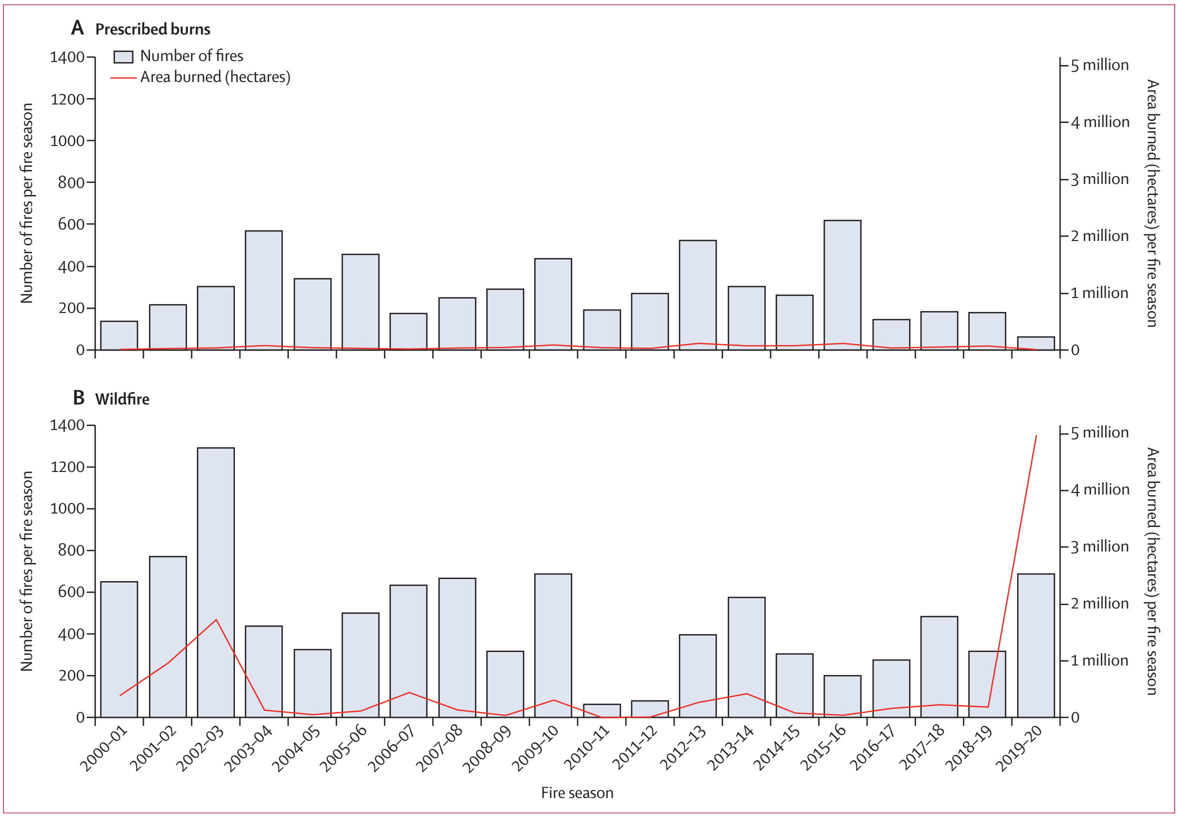

Figure 2: **Number of fires and area burned by fire season**

	LFS days	AU$ million (95% CI)	Proportion of costs	Mean daily cost (AU$/LFS day)
Total	2037 (100·0%)	2013·4 (718·2–3353·9)	100·0%	988 410
By fire type (random forest)				
PB	826 (40·5%)	360·8 (129·8–596·0)	17·9%	436 848
WF	1211 (59·5%)	1652·6 (588·4–2757·9)	82·1%	1 364 620
By fire type (random forest; only fires with correct dates)				
PB	758 (37·2%)	337·2 (121·3–557·0)	16·9%	444 830
WF	1175 (57·7%)	1659·0 (590·7–2768·5)	83·1%	1 411 873
By fire type (categorical)				
WF 0–10%, PB 90–100%	615 (30·2%)	255·0 (91·7–421·2)	12·7%	414 630
WF 10–50%, PB 50–90%	208 (10·2%)	104·1 (37·5–172·0)	5·2%	500 629
WF 50–90%, PB 10–50%	212 (10·4%)	80·8 (29·1–133·4)	4·0%	381 246
WF 90–100%, PB 0–10%	1002 (49·2%)	1573·4 (559·9–2627·3)	78·1%	1 570 298
By season				
Summer	383 (18·8%)	1016·4 (360·7–1701·6)	50·5%	2 653 742
Autumn	585 (28·7%)	291·9 (105·0–482·3)	14·5%	499 002
Winter	566 (27·8%)	146·7 (52·9–241·9)	7·3%	259 186
Spring	503 (24·7%)	558·4 (199·6–928·0)	27·7%	1 110 125
By fire season				
2000–01	43 (2·1%)	49·1 (17·7–81·2)	2·4%	1 142 848
2001–02	79 (3·9%)	229·6 (81·7–384·0)	11·4%	2 906 702
2002–03	153 (7·5%)	287·1 (103·0–476·2)	14·3%	1 876 430
2003–04	116 (5·7%)	105·1 (37·9–173·5)	5·2%	906 338
2004–05	80 (3·9%)	42·5 (15·3–70·2)	2·1%	531 352
2005–06	56 (2·7%)	16·7 (6·0–27·6)	0·8%	298 729
2006–07	86 (4·2%)	67·8 (24·4–112·4)	3·4%	788 913
2007–08	49 (2·4%)	6·1 (2·2–10·1)	0·3%	124 691
2008–09	66 (3·2%)	26·3 (9·5–43·6)	1·3%	399 175
2009–10	87 (4·3%)	36·4 (13·1–60·2)	1·8%	418 193
2010–11	52 (2·6%)	7·6 (2·7–12·6)	0·4%	146 349
2011–12	78 (3·8%)	10·9 (3·9–18·0)	0·5%	139 490
2012–13	88 (4·3%)	16·9 (6·1–27·9)	0·8%	191 913
2013–14	94 (4·6%)	75·4 (27·0–124·8)	3·7%	801 964
2014–15	74 (3·6%)	21·7 (7·8–35·7)	1·1%	292 579
2015–16	87 (4·3%)	67·7 (24·3–112·1)	3·4%	778 197
2016–17	75 (3·7%)	15·1 (5·4–25·0)	0·8%	201 503
2017–18	101 (5·0%)	47·5 (17·1–78·4)	2·4%	469 831
2018–19	246 (12·1%)	60·5 (21·7–100·0)	3·0%	246 024
2019–20	327 (16·1%)	823·3 (291·4–1380·6)	40·9%	2 517 625
By branch				
Blue Mountains branch	748 (36·7%)	68·3 (24·3–114·1)	3·4%	91 274
Interbranch impacts	1613 (79·2%)	676·8 (242·0–1125·1)	33·6%	419 567
Greater Sydney branch	323 (15·9%)	818·2 (291·8–1362·4)	40·6%	2 533 021
Hunter Central Coast branch	645 (31·7%)	235·3 (84·1–391·2)	11·7%	364 737
North Coast branch	451 (22·1%)	9·7 (3·4–16·5)	0·5%	21 550
Northern Inland branch	728 (35·7%)	41·6 (14·7–70·0)	2·1%	57 189
South Coast branch	463 (22·7%)	108·6 (38·7–181·3)	5·4%	234 612
Southern Ranges branch	343 (16·8%)	45·1 (15·7–76·9)	2·2%	131 343

Data are n (%), unless otherwise stated. LFS= landscape fire smoke. PB=prescribed burns. WF=wildfires.

***Table 1*: Summary statistics of estimated LFS-related health costs**

bias of LFS day types, we tested two alternative classifications of days: (1) season, because certain seasons are strongly dominated by either WFS (summer), PBS (autumn and spring), or wood heater smoke (winter); and (2) daily area burned by wildfires and prescribed burns (simple categorical classification). For the simple categorical classification, we assigned a category to each day according to the estimated proportion of area burned by a wildfire and by a prescribed burn: (1) wildfire 0–10%, prescribed burn 90–100%; (2) wildfire 10–50%, prescribed burn 50–90%; (3) wildfire 50–90%, prescribed burn 10–50%; and (4) wildfire 90–100%, prescribed burn 0–10%. For example, a simple category of wildfire 0–10% and prescribed burn 90–100% would be interpreted as having less than 10% of total daily area burned by wildfires and more than 90% of the daily area burned by prescribed burns.

Because of the possible errors associated with wrong or missing dates in the fires database, and the presence of time lags between smoke production, exposure, and onset of symptoms, we varied the time grouping variable to account for multiple days (eg, regression and correlation results were aggregated by day, week, month, year, and fire season), and we included a sensitivity analysis that assessed the effect of excluding fires with wrong or missing dates.

Role of the funding source

The funder of the study had no role in study design, data collection, data analysis, data interpretation, or writing of the report.

Results

A total of 15 536 fires (9638 [62·0%] wildfires) burning 11 940 699 hectares (10 767 390 [90·2%] by wildfires) were active between July 1, 2000, and June 30, 2020 (appendix p 18). 5001 (32·2%) fires, which burned 2 881 092 (24·1%) hectares, had wrong or missing dates in the NPWS database. Summer and spring periods accounted for most (11 621 [74·8%]) fires, which accounted for 10 258 381 (85·9%) hectares of the area burned. The 2002–03 fire season had the highest number of all fires (1592 [10·2%]), but the 2019–20 fire season, which was dominated by unprecedented wildfires (figure 2) had the largest area burned (5 million hectares [41·9%]) and the highest mean area burned of 6692 hectares per fire (or 15 123 hectares per fire day). Both number of fires and area burnt were highest in the NPWS Northern Inland branch, followed by the North Coast branch in the number of fires and by the Blue Mountains branch in terms of area burned.

Population weighted $PM_{2.5}$ exposure across all fire seasons was higher for LFS days compared with all days, with an estimated daily average of 6·8 µg/m³ (SD 9·2) attributable to LFS (appendix p 19). On average, LFS-related $PM_{2.5}$ was higher for WFS during summer and spring and was highest in the 2001–02, 2002–03, and

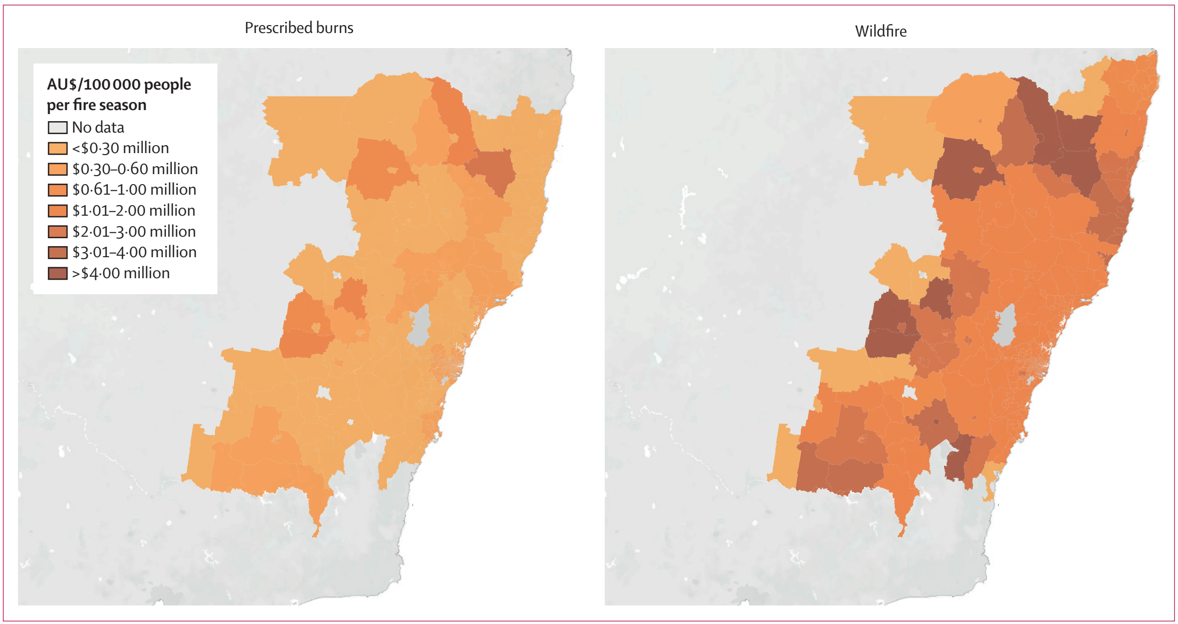

Figure 3: **Map of average estimated health costs by statistical area level 2 and fire type**
Health costs were measured as AU$ per 100 000 people per fire season.

2019–20 fire seasons. There was also spatial variation, with some NPWS branches such as the North Coast, the Southern Ranges and the Greater Sydney, having higher LFS-related $PM_{2·5}$ than the rest. The long-term population weighted annual mean $PM_{2·5}$ (all sources) for NSW was around 6·6 μg/m³. On average, wildfires contributed 0·47 μg/m³ to annual $PM_{2·5}$ exposure per year. $PM_{2·5}$ exposure was 0·94 μg/m³ in the 2001–02, 1·18 μg/m³ in the 2002–03, and 3·97 μg/m³ in the 2019–20 fire seasons.

yThe 2019–20 fire season was by far the most burdensome ($823·3 million [95% CI 291·4–1380·6]; 40·9% of total costs) followed by the 2002–03 fire season ($287·1 million [103·0–476·2]; 14·3% of total costs). As expected, the two most affected regions had the largest populations in New South Wales, the Greater Sydney branch ($818·2 million [291·8–1362·4]) and the Hunter Central Coast branch ($235·3 million [84·1–391·2]). $676·8 million (33·6% [242·0–1125·1]) were inter-branch effects (ie, fires in some regions produced smoke that affected other, probably adjacent, regions). Health costs (normalised per 100 000 people) and fire season varied across New South Wales and by fire type; costs were historically higher for wildfires, with an estimated $1·6 million per 100 000 people and fire season, compared with prescribed burns, $0·33 million per 100 000 people and fire season (figure 3). The highest wildfire costs were seen in the Southern Ranges ($2·68 million per 100 000 people and fire season) and the Northern Inland branches ($2·47 million per 100 000 people and fire season). The highest costs for prescribed burning were reported in Northern Inland ($360 000 per 100 000 people and fire seasons) and the Greater Sydney branch ($341 000 per 100 000 people and fire season; appendix p 21).

For New South Wales as a whole and when aggregating health costs and area burned by fire season, we estimated an average cost of $105 per hectare for all LFS days ($104 per hectare for WFS and $477 per hectare for PBS), but these results were highly influenced by the 2002–03 and 2019–20 fire seasons (figure 4). When these two fire seasons were excluded the difference between WFS and PBS is reduced substantially, but our estimates still found that the average smoke-related health costs of prescribed burning was 1·4-times higher than that of wildfires ($478 per hectare for prescribed burns *vs* $344 per hectare for wildfires). Although these results changed when excluding fires with incorrect dates the differences were minor, with average health costs of prescribed burns being 4·0-times higher than that of wildfires considering all fire seasons, and 1·2-times higher when excluding the 2002–03 and 2019–20 fire seasons (table 2). When increasing or changing the grouping variable (year, month, week) the goodness of fit (measured by the R^2 squared) was reduced, but the difference in costs was sustained. The average cost of days classified as wildfire 0–10%, prescribed burns 90–100% versus wildfire 90–100%, prescribed burns 0–10% (when excluding the 2002–03 and 2019–20 fire seasons) was cost 1·2-times more ($440 per hectare for wildfire 0–10%, prescribed burns 90–100% *vs* $356 per hectare for wildfire 90–100%, prescribed burns 0–10%). All fires in the autumn cost 1·1-times more than all fires in the summer ($503 per hectare in the Autumn *vs* $455 per hectare in the summer; appendix pp 27–28).

We found that fire activity, $PM_{2·5}$ exposure, and smoke-related health effects have different distributions across NPWS branches. The highest average costs (when

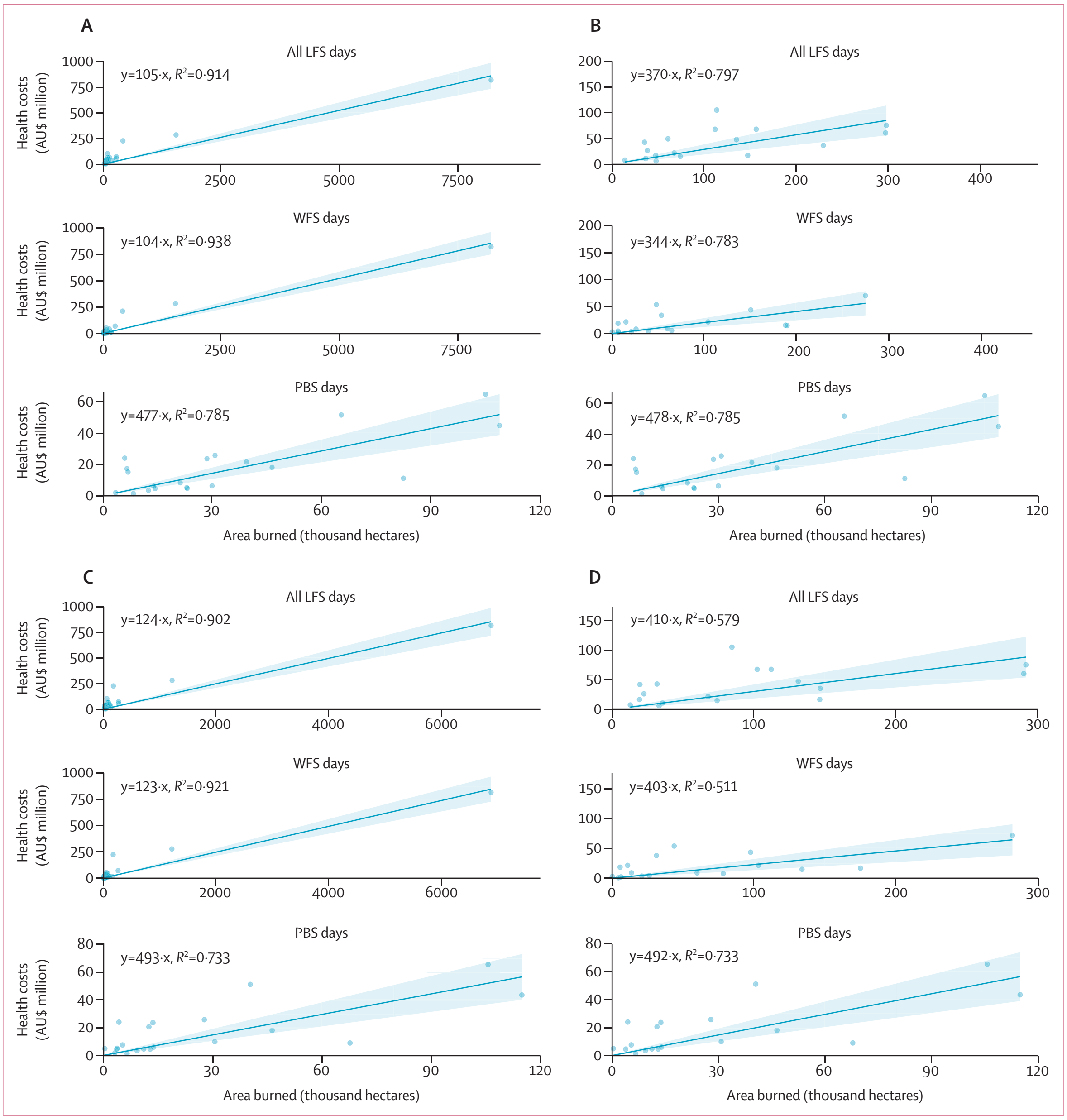

Figure 4: **Linear relationship between estimated health costs and total area burn by type of LFS day**
(A) All LFS days (i), WFS days (ii), and PBS days (iii) across all fire seasons (2000–01 to 2019–20). (B) All LFS days (i), WFS days (ii), and PBS days (iii) across all fire seasons, excluding the 2002–03 and 2019–20 fire seasons. (C) All LFS days (i), WFS days (ii), and PBS days (iii) across all fire seasons with the correct dates, and D) All LFS days (i), WFS days (ii), and PBS days (iii) across all fire seasons with the correct dates, excluding 2002–03 and 2019–20 fire seasons. The linear equation is presented in AU$ and hectares. Shaded areas are 95% CI.

excluding the 2002–03 and 2019–20 fire seasons) occurred in the Greater Sydney and Blue Mountains branches ($494 per hectare) and the Hunter Central Coast branch ($1022 per hectare; appendix p 22). For all NPWS branches, except for the West branch, average PBS costs were higher than WFS (appendix p 31), with the largest difference reported in the Southern branches, where PBS was 7·1-times more expensive than WFS ($321 per hectare for PBS *vs* $45 per hectare for WFS; appendix pp 24–31).

For NPWS branches, a cost per area burned was estimated whenever LFS-related health costs were estimated on the same day a fire was occurring in the same branch. Therefore, it should be noted that these results are probably an underestimation because the inter-branch effects are not included. There were periods of time when landscape fires occurred in some branches of New South Wales and smoke-related health impacts (health costs) were observed in other branches (appendix pp 32–34). For example, a high correlation was observed between the area burned in the Greater Sydney and Blue Mountains branches with costs in the Hunter Central Coast branch (correlation 0·83), Northern branch (correlation 0·91), Southern branches (correlation 0·95), and West branch (correlation 0·93).

Discussion

To our knowledge, this is the first study to thoroughly assess the relative contribution of $PM_{2\cdot5}$ associated with smoke from wildfires and prescribed burns to health costs. Previously, studies have focused on assessing the impact of LFS on air quality, the study of specific events, or the estimation of the total LFS health burden. We found that over the past 20 years in New South Wales, fire history has been dominated by wildfires, accounting for more than 62% of fire events and more than 90% of area burned. Wildfires occurred mainly during spring and summer, with high fire severity and number variability through time, and the 2002–03 and 2019–20 fire seasons being the most severe on record in New South Wales. We estimated these fires caused substantial increases in $PM_{2\cdot5}$ exposure, equivalent to more than the 1 μg/m³ annual average for the 2002–03 and 2019–20 fire seasons. We calculated that wildfires account for most of the health costs (82·1%), but that per hectare health costs were larger for prescribed fires.

Previous studies have assessed the impact of fires on society, including the smoke-related health burden and costs.[2,20,21] In the USA, 30% of the population has an estimated annual LFS-related $PM_{2\cdot5}$ exposure of between 0·56 μg/m³ and 1·5 μg/m³, with 10% of the population exposed to more than 1·5 μg/m³ LFS-related $PM_{2\cdot5}$.[20,34] In Canada, a large proportion of the population is exposed to wildfire season, which lasts from May to September; average LFS-$PM_{2\cdot5}$ exposure is reported to be between 0·2 μg/m³ and 1 μg/m³, with exposures of more than 5–10 μg/m³ reported for some years.[21] In Australia, we estimated an average annual exposure to $PM_{2\cdot5}$ due to wildfires to be 0·47 μg/m³, which increased to 3·97 μg/m³ for the 2019–20 fire season.

Other studies have assessed the relative impact of wildfires and prescribed fires on air quality. A systematic review of studies from the USA found that $PM_{2\cdot5}$ exposure from wildfires was significantly lower than prescribed burns, but these findings were attributed to differences in the methods between studies, because particulate matter was measured further away for wildfires and closer for prescribed burns.[12] These results were not in keeping with our findings or those of other studies. Jaffe and colleagues[35] found that for 2017, maximum daily $PM_{2\cdot5}$ values were higher for wildfires (125–550 μg/m³) compared with prescribed burns (29–49 μg/m³) for the top five states for annual area burned, with both fire types having a similar magnitude of area burned (260–640 thousand hectares per year).[35] Between 2004 and 2018, there seemed to be a positive association between the percentage of monitor-days exceedances (>35 μg/m³) and area burned for California and Washington, with exceedance frequency also related to fire activity across states.[35] These results are in line with our findings, with higher $PM_{2\cdot5}$ exposure levels for wildfires compared with prescribed burns. Our results also suggest that health costs driven by $PM_{2\cdot5}$ exposure are coupled to area burned in both wildfires and prescribed burns, with a considerable contribution of $PM_{2\cdot5}$ exposure due to inter-branch fire activity (ie, fires occurring in some NPWS branches and $PM_{2\cdot5}$ exposure observed in other branches).

In Australia, Price and colleagues[11] assessed the effect of two prescribed burns, with burned areas of 52 hectares and 700 hectares. They found that the small burn produced larger spikes in $PM_{2\cdot5}$ closer to the burn (about 500 m downwind), whereas the larger event might have produced important air pollution increases in locations up to 14 kms away. The smoke levels from these prescribed fires was found to be severe, reaching levels of above 1000 μg/m³, and daily averages of above 200 μg/m³, far exceeding national air quality standards. The authors suggest that prescribed fires can probably produce high local effects and lower regional level effects, both increasing with fire size. This situation outlined by Price and colleagues[11] has been observed in Australia, with some prescribed burning programmes producing similar large air pollution episodes to those observed for serious wildfire events.[14]

Generally, LFS-related health costs estimated by different studies are not comparable because of existing differences in the magnitude of exposed populations, the duration of fires (or studies), and the values used to quantify and monetise health impacts and costs. One way to compare health-related costs across different contexts is by normalising these values by a constant amount of population, generally 100 000 people (ie, similar to a per capita estimate). In our study, we estimated average LFS-related health costs of $1·61 million per 100 000 people fire season for wildfires ($0·33 million per 100 000 people fire season for prescribed burns), which is lower than estimates for

	Variable	All fire seasons	All fire seasons excluding 2002–03 and 2019–20
By fire type (random forest)			
PB	PBS costs (AU$ per hectare)	$477	$478
WF	WFS costs (AU$ per hectare)	$104	$344
	PBS to WFS ratio	4·6	1·4
By fire type (random forest; fires with correct dates)			
PB	PBS costs (AU$ per hectare)	$493	$492
WF	WFS costs (AU$ per hectare)	$123	$403
	PBS to WFS ratio	4·0	1·2

PB=prescribed burns. PBS=prescribed burning smoke. WF=wildfire. WFS=wildfire smoke.

Table 2: **Summary of PBS and WFS health costs per hectare and PBS to WFS cost ratio**

Canada (AU$3 million per 100 000 people) and the USA ($6·1 million per 100 000 people).[20,21] The main difference between our study and the estimates for the USA and Canada probably lies in the magnitude of the VSL used to monetise mortality. These studies used a VSL of CA$6·5 million (at the 2007 value of the Canadian dollar) and US$10·1 million (at the 2010 value of the US dollar), substantially higher than the AU$4·5 million used for this study.

The main strengths of this study relate to the application of a quantitative health impact assessment in a setting ideal for this analysis: New South Wales. New South Wales has forests close to many population centres; a longstanding programme of prescribed burning; a reliable fire history database, with more than 20 years of data; a well developed air quality monitoring network; and up to date demographic and health statistical information.

Like all health and economic modelling studies, this analysis has some inherent assumptions, uncertainties, and limitations that are well recognised and have been discussed elsewhere.[36] Our aim was to use consistent and accepted methods to compare the relative health costs of the two types of landscape fires. This required using some additional approaches, which have additional specific limitations: the use of administrative fire data (and its inherent data errors) to identify individual fire events, the association of fire type days with identified LFS-affected days, and the estimation of a relationship between LFS-attributable health costs and fire activity measured by area burned. The potential errors present in the NPWS fire history database could lead to the wrong classification of fire type days and the overestimation or underestimation of area burned for each of these fire types. We sought to overcome this limitation by doing some additional analyses and presenting our results excluding fires with incorrect original dates. As can be seen in our analysis, the exclusion of fires with wrong dates did not have a substantial effect on the results. Additionally, we did our analysis at different aggregate temporal levels (days to years) and classified fire type days under alternative categorical variables (simple categorical classification based on area burned by fire type) and season (spring, summer, autumn, winter) that might be representative of wildfires and prescribed burns, and observed consistency in our results. The association of fire type days with identified LFS affected days was one of the most complex methodological decisions we had to make for this analysis. If we chose to analyse individual fires, we would have been incapable of attributing smoke exposure to many of these fires, particularly because substantial population exposure due to multiple fire events happening across the landscape and not just individual fires could occur. Therefore, instead of trying to identify how specific fire events affected $PM_{2·5}$ in surrounding locations, we assigned the most likely smoke source (WFS or PBS) to each day. Additionally, we excluded days potentially affected by dust storms and the estimated contribution of wood heater use on increased $PM_{2·5}$ during colder days. If this was not done, the smoke-related health costs would have been overestimated. Nevertheless, our approach was conservative with respect to the identification of LFS days because, by considering only the days with the highest 5% of $PM_{2·5}$ exposure (excluding days affected by dust storms), we excluded a considerable number of days that could have been affected by fire smoke but had a moderate level of pollution. Finally, we did several sensitivity analyses to test the robustness of our results to the approach we used to characterise the relationship between LFS attributable health costs and fire activity, using different fire type classification methods and different temporal levels of aggregation. Across all combinations, we consistently observed that the per hectare health costs for prescribed burns were higher than those for wildfires.

The historical health burden calculated in this study is probably an underestimation of the real impact. We used short-term health risk coefficients (lower than those used for chronic exposure) usually used for acute exposure, although our results show that the LFS-related $PM_{2·5}$ exposure translated to relevant annual $PM_{2·5}$ values, with more than 30% of days within the study period being affected by LFS. Additionally, our analysis only includes the effect of $PM_{2·5}$ on health, but other pollutants—such as ozone—might also produce a relevant smoke-related health burden.[37]

The future will probably bring an increase in frequency and magnitude of wildfires in different regions across the globe with increasing pressure to better manage the inherent risks. Interventions used to reduce wildfire risk affect future fuel availability, wildfire occurrence, and wildfire magnitude. Although prescribed fires might be an effective fire management tool, the burn window for a safe and effective application of prescribed burns might be reduced or changed in seasonality. Many factors—such as fire intensity, fuel availability, and type and fire duration—have an effect on smoke volume, dispersion, and composition and, as a result, on health impacts.[14] Nevertheless, our results show a high correlation between estimated health costs and area burned, and the most interesting and surprising result of this study is that per hectare health costs are higher for prescribed burns than for wildfires. This can be due to various reasons, including the larger magnitude of wildfire surface, proximity to people, or because prescribed burns tend to occur in weather more conducive to produce smoke accumulation.[11,12] Prescribed burns are usually done at interfaces between rural and urban environments; the aim of these burns is to protect the built environment. Therefore, prescribed burns probably produce a substantial population exposure to air pollution. Crawford and colleagues[13] found that smoke particulate concentrations on winter days in Sydney, NSW, Australia, were 11-times higher when the atmosphere was stable than when it was not.[13] Stable days are preferred for prescribed burning because the presence of light winds reduces the risk of an uncontrolled fire. In New South Wales, to effectively reduce the area burned by wildfires an area about three-times the size needs to be burned through prescribed fires.[28] It is expected that in this setting, the implementation of a prescribed burning programme that successfully reduces the area burned by wildfires will produce an overall increase in smoke-related harms.

The health effects of LFS should be adequately incorporated into the assessment of the trade-offs between wildfires and prescribed burns. Alternatives to prescribed burning in areas close to human populations—such as herbivory, the manual removal of fuels, vegetation modification, and the creation of so called green firebreaks[38]—will probably have major economic benefits, provide similar levels of risk mitigation, and lower smoke-related health harm outweighing the additional costs of these alternatives compared with prescribed burning. The likely increase in frequency and intensity of fire seasons, coupled with changes in the seasonality and availability of days with ideal conditions for prescribed fires, will increase the complexity and risk trade-offs associated with prescribed burning programmes in the future. This requires an increase in the extent and diversity of fuel management programmes, and systems for better mitigation of the adverse population health impacts of smoke from prescribed burns.

Contributors

NB-A and FHJ conceived the paper. NB-A contributed to the methods, conducted the analyses, and drafted the manuscript. FHJ contributed to the health assessment methods and analyses. DMJSB contributed to the fire-related methods and analyses. AJP contributed to the health economic methods and analysis. OP and SS contributed to the fire-related data collection and analysis. HC contributed to the overall methodological approach. GS contributed to the statistical and machine learning methods used. NB-A and GS had access to all the data. All authors helped revise the final version of this article and were responsible for the decision to submit the manuscript.

Declaration of interests

We declare no competing interests.

Data sharing

All data, codes, and materials used in the analysis are available from the authors upon reasonable request.

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References

1 Xu R, Yu P, Abramson MJ, et al. Wildfires, global climate change, and human health. *N Engl J Med* 2020; **383:** 2173–81.

2 Johnston FH, Borchers-Arriagada N, Morgan GG, et al. Unprecedented health costs of smoke-related $PM_{2.5}$ from the 2019–20 Australian megafires. *Nat Sustain* 2020; **4:** 42–47.

3 Johnston FH. Bushfires and human health in a changing environment. *Aust Fam Physician* 2009; **38:** 720–24.

4 Cascio WE. Wildland fire smoke and human health. *Sci Total Environ* 2018; **624:** 586–95.

5 Reid CE, Brauer M, Johnston FH, Jerrett M, Balmes JR, Elliott CT. Critical review of health impacts of wildfire smoke exposure. *Environ Health Perspect* 2016; **124:** 1334–43.

6 Rice MB, Henderson SB, Lambert AA, et al. Respiratory impacts of wildland fire smoke: future challenges and policy opportunities. An official American Thoracic Society workshop report. *Ann Am Thorac Soc* 2021; **18:** 921–30.

7 Fernandes PM, Botelho HS. A review of prescribed burning effectiveness in fire hazard reduction. *Int J Wildl Fire* 2003; **12:** 117–28.

8 Hunter ME, Robles MD. Tamm review: The effects of prescribed fire on wildfire regimes and impacts: a framework for comparison. *For Ecol Manage* 2020; **475:** 118435.

9 Haikerwal A, Reisen F, Sim MR, et al. Impact of smoke from prescribed burning: is it a public health concern? *J Air Waste Manag Assoc* 2015; **65:** 592–98.

10 Price OF, Bradstock RA. The spatial domain of wildfire risk and response in the wildland urban interface in Sydney, Australia. *Nat Hazards Earth Syst Sci* 2013; **13:** 3385–93.

11 Price OF, Horsey B, Jiang N. Local and regional smoke impacts from prescribed fires. *Nat Hazards Earth Syst Sci* 2016; **16:** 2247–57.

12 Navarro KM, Schweizer D, Balmes JR, et al. A review of community smoke exposure from wildfire compared to prescribed fire in the United States. *Atmosphere* 2018; **9:** 185.

13 Crawford J, Chambers S, Cohen D, Williams A, Griffiths A, Stelcer E. Assessing the impact of atmospheric stability on locally and remotely sourced aerosols at Richmond, Australia, using Radon-222. *Atmos Environ* 2016; **127:** 107–17.

14 Williamson GJ, Bowman DMJS, Price OF, Henderson SB, Johnston FH. A transdisciplinary approach to understanding the health effects of wildfire and prescribed fire smoke regimes. *Environ Res Lett* 2016; **11:** 125009.

15 Clarke H, Penman T, Boer M, et al. The proximal drivers of large fires: a pyrogeographic study. *Front Earth Sci* 2020; **8:** 90.

16 Kupfer JA, Terando AJ, Gao P, Teske C, Hiers JK. Climate change projected to reduce prescribed burning opportunities in the south-eastern United States. *Int J Wildl Fire* 2020; published online June 1. https://doi.org/10.1071/WF19198.

17 Clarke H, Tran B, Boer MM, Price O, Kenny B, Bradstock R. Climate change effects on the frequency, seasonality and interannual variability of suitable prescribed burning weather conditions in south-eastern Australia. *Agric For Meteorol* 2019; **271:** 148–57.

18 Héroux ME, Anderson HR, Atkinson R, et al. Quantifying the health impacts of ambient air pollutants: recommendations of a WHO/Europe project. *Int J Public Health* 2015; **60:** 619–27.

19 Borchers Arriagada N, Palmer AJ, Bowman DM, Johnston FH. Exceedances of national air quality standards for particulate matter in Western Australia: sources and health-related impacts. *Med J Aust* 2020; **213:** 280–81.

20 Fann N, Alman B, Broome RA, et al. The health impacts and economic value of wildland fire episodes in the U.S.: 2008–2012. *Sci Total Environ* 2017; **610–11:** 802–09.

21 Matz CJ, Egyed M, Xi G, et al. Health impact analysis of $PM_{2.5}$ from wildfire smoke in Canada (2013–2015, 2017–2018). *Sci Total Environ* 2020; **725:** 138506.

22 Johnston FH, Henderson SB, Chen Y, et al. Estimated global mortality attributable to smoke from landscape fires. *Environ Health Perspect* 2012; **120:** 695–701.

23 Broome RA, Johnston FH, Horsley J, Morgan GG. A rapid assessment of the impact of hazard reduction burning around Sydney, May 2016. *Med J Aust* 2016; **205:** 407–08.

24 Filkov AI, Ngo T, Matthews S, Telfer S, Penman TD. Impact of Australia's catastrophic 2019/20 bushfire season on communities and environment. Retrospective analysis and current trends. *J Saf Sci Resil* 2020; **1:** 44–56.

25 Godfree RC, Knerr N, Encinas-Viso F, et al. Implications of the 2019–2020 megafires for the biogeography and conservation of Australian vegetation. *Nat Commun* 2021; **12:** 1023.

26 Borchers-Arriagada N, Palmer AJ, Bowman DM, Morgan GG, Jalaludin BB, Johnston FH. Unprecedented smoke-related health burden associated with the 2019–20 bushfires in eastern Australia. *Med J Aust* 2020; **213:** 282–83.

27 Binskin M, Bennett A, Macintosh A. Royal Commission into National Natural Disaster Arrangements Report. 2020 https://naturaldisaster.royalcommission.gov.au/publications/royal-commission-national-natural-disaster-arrangements-report (accessed Nov 15, 2020).
28 Price OF, Bradstock RA. Quantifying the influence of fuel age and weather on the annual extent of unplanned fires in the Sydney region of Australia. *Int J Wildl Fire* 2011; **20:** 142.
29 New South Wales Department of Planning Industry and Environment. NPWS fire history - wildfires and prescribed burns. 2020. https://data.nsw.gov.au/data/dataset/fire-history-wildfires-and-prescribed-burns-1e8b6 (accessed July 6, 2020).
30 Breiman L. Random forests. *Mach Learn* 2001; **45:** 5–32.
31 Johnston FH, Hanigan IC, Henderson SB, et al. Creating an integrated historical record of extreme particulate air pollution events in Australian cities from 1994 to 2007. *J Air Waste Manag Assoc* 2011; **61:** 390–98.
32 Hanigan IC, Morgan GG, Williamson GJ, et al. The biomass smoke validated events database. 2015. https://researchdata.edu.au/biomass-smoke-validated-events-database/934712 (accessed Aug 22, 2021).
33 New South Wales Department of Planning Industry and Environment. Air quality annual report. 2020. https://www.environment.nsw.gov.au/topics/air/air-quality-annual-reports (accessed Oct 15, 2020).
34 Rappold AG, Reyes J, Pouliot G, Cascio WE, Diaz-Sanchez D. Community vulnerability to health impacts of wildland fire smoke exposure. *Environ Sci Technol* 2017; **51:** 6674–82.
35 Jaffe DA, O’Neill SM, Larkin NK, et al. Wildfire and prescribed burning impacts on air quality in the United States. *J Air Waste Manage Assoc* 2020; published online April 2. DOI:10.1080/10962247.2020.1749731.
36 Johnston FH. Burning to reduce fuels: the benefits and risks of a public health protection strategy. *Med J Aust* 2020; **213:** 246–48.
37 Ballesteros-González K, Sullivan AP, Morales-Betancourt R. Estimating the air quality and health impacts of biomass burning in northern South America using a chemical transport model. *Sci Total Environ* 2020; **739:** 139755.
38 Bowman DMJS, Balch J, Artaxo P, et al. The human dimension of fire regimes on Earth. *J Biogeogr* 2011; **38:** 2223–36.