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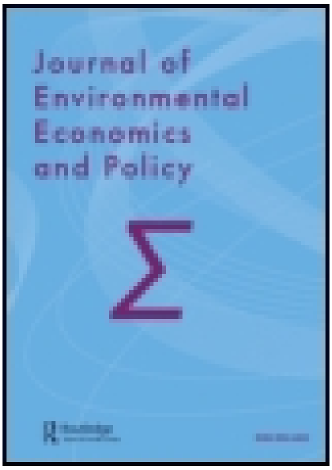

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Exposure to wildfire smoke can increase morbidity in urban areas. Economists are increasingly calling for such health impacts to be included in wildfire damage assessments. However, collecting original health outcome data is costly and time-consuming. Benefits transfer is a more accessible alternative that is often employed. Yet several methodological issues remain unexplored regarding transfers of economic values and air quality concentration-response functions. Ignoring these issues may lead to misinformed wildfire policy based on inexact estimates of smoke-induced health costs. This research provides a case study illustration of a new air quality benefit transfer tool, the US EPA benefits mapping and analysis program-community edition (BenMAP-CE), which is used to estimate smoke damages of a Southwestern US 'mega-fire' event and investigate methodological issues surrounding the analyst's choice between transferring results from 'wildfire-specific' and 'urban air' (unrelated to wildfire) studies. Results indicate that the economic costs of wildfire smoke are substantial. Additionally, transfer of wildfire-specific study results produces substantially higher morbidity estimates and costs compared to use of results from urban air studies. These findings demonstrate (1) that BenMAP-CE can be applied to wildfire events and (2) the importance of transferred study appropriateness when conducting a smoke damage assessment using benefits transfer.

Keywords: wildfire; benefit transfer; health effects; BenMAP-CE; willingness to pay

1. Introduction

Due to prolonged drought, climate change, and fuels build-up on forested lands, the frequency and severity of wildfire events is expected to increase across much of the western United States (US) (Liu, Stanturf, and Goodrick 2010; US Global Change Research Program 2014). In assessing the damage from such events, there is increasing realisation of the need to incorporate wildfire impacts outside of the flame zone such as downstream water quality, regional ecosystem health, and adverse health effects associated with exposure to smoke into wildfire management policy. Economists are interested in measuring the economic benefits and costs associated with wildfire impacts to inform efficient use of limited control resources (Milne et al. 2014). Original estimation of the economic costs of wildfire smoke-related health effects is time consuming and costly, requiring extensive micro-level data collection in affected areas (e.g. Richardson, Champ, and Loomis 2012). However, there is an urgent need to better understand the environmental-health costs of wildfire events, including effects on both nearby and regional urban populations.

Benefits transfer – using existing data to inform decisions in a different setting or context (Rosenberger and Loomis 2003) – is a more accessible way to estimate smoke-exposure

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health costs and has been used in several wildfire studies (e.g. Martin, Brajer, and Zeller 2008; Rittmaster et al. 2006; Hon 1999). Previous research often transfers results from non-'wildfire-specific' studies, which may be inappropriate for estimating costs of severe, short-duration smoke events (Kochi et al. 2010). The objective of this research is to provide a case study illustration of a wildfire benefits transfer using the US Environmental Protection Agency's (EPA) benefits mapping and analysis program-community edition (BenMAP-CE), and investigate the resulting impact on estimated health costs of transferring functions and values from wildfire-specific studies instead of ones from the 'urban air' literature. BenMAP-CE is an open-source Windows-based application that estimates the economic benefits associated with changes in air quality over a geographic area (Davidson et al. 2007). This analysis configures and modifies BenMAP-CE (version 1.0.8)[1] for application to an urban wildfire smoke event in the Albuquerque, New Mexico (NM) metropolitan area caused by the 2011 Wallow Fire (Wallow 'mega-fire', hereafter) that burned 535,000 acres in Arizona and New Mexico (US Forest Service 2011).

This case study addresses several open issues in the literature, including (1) the appropriateness of valuing smoke-induced health impacts through wildfire-specific willingness to pay (WTP) measures and (2) selection of transferred air quality concentration-response (CR) functions. On (1), we contrast mega-fire event health costs using an originally constructed wildfire-specific WTP measure with WTP and cost-of-illness (COI) values estimated using BenMAP-CE's built-in valuation functionality. On (2), comparisons are made between estimated health incidence calculated using wildfire-specific CR functions and BenMAP-CE's built-in urban air quality CR functions, transferred by the EPA from non-wildfire event studies. Again, the objectives are not only to provide a 'proof-of-concept' application of BenMAP-CE to a mega-fire caused smoke event, but to also explore differences in estimated health impacts and costs across types of transferred CR and WTP functions (wildfire-specific or not). Transferred study selection in a wildfire smoke benefit transfer is an unsettled methodological issue, which is likely to be confronted by researchers estimating the costs of a wildfire event.

The contributions of this research are threefold. First, following only one previous study (unpublished), we provide the second case study application and configuration of BenMAP-CE for a wildfire smoke event. Second, we provide preliminary empirical evidence of smoke-related health incidence and valuation sensitivity to choice of transferred air quality CR functions and WTP values (wildfire-specific or not). Finally, using original survey data to complement the case study, we add to the small set of available wildfire smoke WTP valuation estimates.

Results indicate that the economic costs of a wildfire smoke event are substantial. Smoke event morbidity and health costs vary considerably according to whether or not a wildfire CR function is used in place of an urban air CR function. Additionally, use of an originally constructed WTP measure from a wildfire smoke experience questionnaire produces substantially larger health costs than those found by using an urban air quality WTP value. Differences are consistent with the nascent literature on divergences between conventional and wildfire-specific air quality studies (Kochi et al. 2010; Vedal and Dutton 2006).

2. Background

A combination of natural and human caused factors (e.g. prolonged drought, history of fire suppression policy and concomitant fuels build-up in forests, rapid expansion of the wildland urban interface) have increased the risk of wildfires in the US west and elsewhere (Ryan and Opperman 2013). Damages from high severity wildfires have followed an increasing trend, and can be economically significant (US Global Change Research Program 2014). Furthermore, with climate change, recent assessments conclude that wildfire risk and associated damages are expected to increase (Liu, Stanturf, and Goodrick 2010).

Damages from a wildfire event can occur both within the burn area and outside of it (e.g. due to smoke plumes, and subsequent floods and debris flows, etc.). Resource managers must account for effects of fire on ecosystem health, water quality, and soil composition both in and immediately surrounding the flame zone (Dale 2006). Wildfire impacts on environmental and behavioural regimes can extend well beyond the flame zone, to include downstream surface water quality (Smith et al. 2011), ash fallout (Earl and Blinn 2003), and air quality within the smoke plume (Henderson and Johnston 2012; Fowler 2003). Wildfire smoke plumes may contain particulates that are especially harmful to health (Naeher et al. 2007). These plumes have immediate localised effects on communities in and around the flame zone, but can travel great distances (e.g. hundreds of miles) according to their atmosphere injection height, which strongly influences dispersion.

The site for our case study, Albuquerque, is located in Bernalillo County in north-central New Mexico (NM). With a city population of 545,852 and a metropolitan population of 662,564, Albuquerque is the largest urban centre in NM (US Census Bureau 2010). While not situated in forested lands itself, Albuquerque has forest and wildland urban interface (WUI) areas to the east (i.e. the eastern slopes of Sandia Mountains and parts of the Cibola National Forest) and hundreds of miles to the southwest (Gila and Apache National Forests). Predominant winds out of the southwest (Ryan and Opperman 2013) can bring smoke plumes from wildfires into Albuquerque. Wildfire smoke over Albuquerque is not uncommon, with smoke from the 2011 Wallow mega-fire in south-eastern Arizona being the most recent prolonged event. The Wallow mega-fire significantly impacted air quality levels across central and northern NM, some 300 miles away from the burn site, including Albuquerque.[2] Health effects associated with smoke from the fire have been investigated and include respiratory and cardiovascular illnesses (Resnick et al. 2015), though to-date no smoke damage assessment has been performed.

2.1. *Wildfire smoke health impacts*

Recognition that wildfire smoke affects human health is not new (Henderson and Johnston 2012; Fowler 2003; Duclos, Sanderson, and Lipsett 1990). Despite decades of ‘awareness’, it is still unclear what many of the short- and long-term health impacts of wildfire smoke are in an exposed population. Many studies focus on estimating changes in emergency room visits (Rittmaster et al. 2006; Hon 1999; Duclos, Sanderson, and Lipsett 1990), hospital admissions (Crabbe 2012; Delfino et al. 2009; Mott et al. 2005; Shahwahid and Othman 1999), or mortality (Johnston et al. 2011; Hänninen et al. 2009; Vedal and Dutton 2006). Generally, wildfire smoke events are associated with increased cardio-respiratory related physician and hospital visits. However, the link between increased daily mortality and smoke exposure is unclear (Vedal and Dutton 2006).

Health impacts of exposure to air pollutants may vary depending on their source. Research has investigated whether wildfire smoke or wood smoke exposure produces the same health impacts as exposure to urban air pollution (Kochi et al. 2010; Hänninen et al. 2009; Naeher et al. 2007; Seagrave et al. 2006). Results have been mixed. Seagrave et al. (2006) find little evidence to suggest increased toxicity of wood smoke compared to urban pollutants, though only smoke from fireplaces and prescribed forest burns was examined. Naeher et al. (2007) argued that insufficient evidence exists to make any general conclusions regarding relative pollutant source toxicity. Hänninen et al. (2009) observed wildfire-specific mortality increases that were consistent with expected estimates from urban air pollutant models. In contrast, Kochi et al. (2010) concluded that wildfire smoke is 'consistently' more detrimental to cardio-respiratory health than urban air pollutants, though less of a threat to mortality. Kochi et al. (2010) only examined studies that directly investigated smoke known to be from forest or brush fires, which may explain differences in their results compared to earlier studies.

Several causes have been proposed to explain why wildfire smoke may be more toxic than urban air pollution. These include chemical composition differences between smoke plumes and urban air, differences in perceived health risks (people may perceive wildfire smoke as more health damaging), and behavioural response patterns (i.e. averting behaviours) (Hänninen et al. 2009; Kunzli et al. 2006; Vedal and Dutton 2006). If wildfire health impacts substantively differ from urban air health impacts, then CR functions estimated for one class, say urban air, are inappropriate for use in a benefit transfer estimation of health impacts in the other class, wildfire, for example. In the extreme, use of an urban air CR function may lead to incorrectly ascribing mortality or morbidity incidence to a wildfire smoke event. For this reason, Kochi et al. (2010) recommend using wildfire-specific study results whenever possible in a benefit transfer. It remains an open empirical question as to how much bias may be introduced by using urban air study results in a wildfire smoke damage assessment. Comparing smoke exposure health impacts between wildfire-specific and urban air pollutant transferred studies for a specific benefit transfer case study, like the one carried out here, may shed additional light on this open debate.

2.2. *Evaluating health impacts and costs of wildfire smoke events*

Wildfire damage assessments often use benefit transfers to capture the economic costs of smoke exposure (e.g. Martin et al. 2008; Rittmaster et al. 2006; Cardoso de Mendonça et al. 2004; Butry et al. 2001), though not in all cases (e.g. Richardson, Champ, and Loomis 2012; Hon 1999; Shahwahid and Othman 1999). One advantage of benefit transfer is that it can be used in circumstances where original data are unavailable. Existing data and information originally collected for use at one site are transferred to a different context or 'policy site' (Rosenberger and Loomis 2003). Ideally, the context and site to which the original research is being adapted are similar in many respects, so that the transferred values are as close to the 'true' value as possible.

In a wildfire smoke benefit transfer, two types of transfers are made from prior studies: (1) transfers of CR air quality functions to estimate changes in health incidence and (2) transfers of COI and WTP values to estimate the economic costs of health incidence changes. In (1), CR functions from either the urban air quality or wildfire smoke literature are used to relate changes in pollutant concentrations to changes in health incidence. The economic costs of these changes are valued in (2) using estimates from previous studies (e.g. cost of an emergency room visit, WTP to avoid a smoke related health impact, etc.). For example, in a simple benefit transfer, an existing CR function for PM2.5 (particulate matter up to 2.5 μm in size) from the urban air literature might be used to estimate changes in mortality due to a wildfire smoke event. WTP results from the value of a statistical life literature might then be used to determine the economic costs of the event in terms of increased mortality.

Mixed-transfer or 'hybrid' wildfire smoke benefit transfers have also been used. In a hybrid transfer, the researcher(s) transfers either CR functions or economic cost values, but not both (e.g. Kochi et al. 2012; Cardoso de Mendonça et al. 2004). The non-transferred results are then estimated using original data specific to the wildfire event. In this case study, we perform both a benefit transfer and a hybrid benefit transfer, where in the latter we estimate WTP to avoid a wildfire smoke health impact using original data from a smoke health impact questionnaire.

2.3. Using WTP and COI to estimate costs

Economic costs of a wildfire smoke event can be estimated either using COI or WTP approaches. In a COI approach, the direct and indirect resource costs of a smoke-related illness are identified and measured. These costs include expenditures on medical care and medications, opportunity costs of time spent acquiring medical treatment, and the value of lost wages due to time spent sick (Richardson, Loomis, and Champ 2013). COI is an imperfect measure of the total burden of wildfire smoke on society because it does not fully capture the disutility of illness (Richardson, Loomis, and Champ 2013; Richardson, Champ, and Loomis 2012; Kochi et al. 2010; Freeman 2003). WTP to avoid a wildfire smoke health impact is the preferred utility-theoretic alternative and has been used in a handful of studies (e.g. Martin et al. 2008; Rittmaster et al. 2006; Hon 1999). However, *all* these previous studies (except for Richardson, Champ, and Loomis 2012) use a WTP measure derived from the urban air literature. In the only published WTP value to avoid health damages estimated for wildfire smoke specifically, Richardson, Champ, and Loomis (2012) found a WTP of $93.15 (2014$) per exposed person per day for a large wildfire in southern California. The authors suggest that their result is consistent with the health literature, but do not make a direct comparison with urban air study results. Is WTP to avoid a smoke health impact meaningfully different than WTP to avoid an illness caused by urban air pollutants? If yes, then a wildfire benefit transfer utilising urban air economic valuation measures may over- or under-value smoke-related health costs. Kochi et al. (2010) posited that such value differences might exist, but to-date no comparison has been made. Using survey data for Albuquerque, NM, we contribute the second estimate of WTP to avoid illnesses from wildfire smoke specifically, and compare our value to a commonly used urban air WTP value from Dickie and Messman (2004).

2.4. Benefit transfer using BenMAP-CE

The US EPA's BenMAP-CE is a benefit transfer tool that utilises transferred CR functions and economic values to estimate benefits (for an improvement) or costs (for a decrement) associated with changes in air quality for non-overlapping health endpoints (Davidson et al. 2007). First, the user inputs modelled or monitored air quality data such as particulate matter (PM) into an air quality grid over a defined geographic space. After specifying the analysis timeframe, health endpoints of interest, and population-grid size, the user defines an air quality policy change (called a control). For example, this could be a hypothetical reduction in PM2.5 of 1 $\mu g/m^3$ annually over a decade in Albuquerque. Using the supplied data and selection of health endpoints, BenMAP-CE calculates point estimates of changes in incidence for each endpoint associated with the air quality change within each grid-cell. Finally, COI and WTP values are transferred from air quality and health endpoint literatures or transferred from original estimates by the researcher to value estimated incidence changes at the grid-cell level. At the discretion of the user, incidence and economic valuations can be spatially aggregated.

BenMAP-CE has been applied to analyse health impacts of changes in air quality standards in the US (Fann et al. 2012; US EPA 2010, 2013), and internationally

(Voorhees et al. 2014). In an unpublished thesis, Douglass (2008) provides the only prior application to wildfire smoke. Given the open-source nature and peer-reviewed development of BenMAP-CE, it's a robust benefit transfer tool for estimating and valuing wildfire smoke health impacts. The potential of using BenMAP-CE for wildfire events is tremendous because it provides a way to quickly estimate health impacts of a wildfire event using hourly and daily pollutant measures. Until recently, this was not easy to do in a systematic and controlled way.

This study will configure BenMAP-CE for a wildfire smoke event and apply it in a case study of a Southwestern US mega-fire caused smoke event in Albuquerque, NM. To evaluate differences between transfers of wildfire-specific and urban air study results, two benefit transfers will be performed in BenMAP-CE. In the first, built-in urban air study CR functions, COI, and WTP values will be employed to estimate smoke exposure incidence and associated health costs. In the second, wildfire-specific CR functions, COI, and an originally estimated wildfire WTP will be utilised. Comparisons will be made between estimated incidence and health costs across transfers. This is the first case study illustration of results sensitivity to choice of transferred study results (wildfire or urban air) and only the second case study application of BenMAP-CE to a wildfire smoke event. The value-added of our analysis over the previous BenMAP-CE application (Douglas 2008) is that we investigate sensitivity to transferred study type and use originally estimated economic valuation results.

3. Methods

3.1. Modifications to BenMAP-CE

BenMAP-CE (v.1.0.8) has more than forty built-in 'health impact' functions covering twenty-two health endpoints from acute bronchitis to hospital admissions to work loss days. In BenMAP-CE, a health impact function refers to the relationship between a change in pollutant concentration (Δp) and a change in health incidence (Δy). It's numerically derived from an air quality CR function, which relates pollutant concentration (p) to health incidence (y). Each health impact function comes from a unique CR function.[3] Importantly, none of the built-in health impact functions were transferred from wildfire-specific studies.

The open-source nature of the program means that practically any health impact function can be input by the user as long as it has a specific CR function that relates incremental changes in pollutant levels to changes in incidence, such as those from regression estimation of relative risk or odds ratios. Focusing our search of the wildfire smoke literature to studies where a CR function was estimated produces a handful of results. Higher morbidity rates during wildfire smoke events are consistently observed, with increases in cardio-respiratory emergency room visits and hospital admissions being common (Resnick et al. 2015; Crabbe 2012; Henderson et al. 2011; Delfino et al. 2009; Hanigan, Johnston, and Morgan 2008; Moore et al. 2006). Other CR functions estimated in the literature include pharmaceutical dispensations for salbutamol (Elliott, Henderson, and Wan 2013) and non-hospital physical visits (Henderson et al. 2011). A previously mentioned, evidence linking wildfire smoke to mortality is mixed (see Kochi et al. 2010).

For the wildfire-specific benefit transfer explored as part of our case study, CR functions estimated by Delfino et al. (2009) and Resnick et al. (2015) are used. These two studies are selected because they are recent and are specific to western US wildfires, where our study area is located. Only statistically significant results are utilised from each study and only functions covering individuals aged 0-99 or 'all ages' are included. We coded into BenMAP-CE four health impact functions for the following endpoints:

(1) *emergency room visits, asthma*; (2) *hospital admission, all respiratory*; (3) *hospital admission, asthma*; and (4) *hospital admission, pneumonia*. Relative risks of each health endpoint were converted to coefficients for use in a log-linear health impact function, following the BenMAP User's Manual (Abt Associates 2012).

In our benefit transfer utilising built-in urban air quality transfers, we select CR functions from Mar, Koenig, and Primomo (2010), Zanobetti et al. (2009), Slaughter et al. (2005), Ito (2003), and Sheppard (2003) for the same set of health endpoints as in the wildfire-specific analysis. Additionally, CR function results from Ostro and Rothschild (1989) are used to estimate incidence of minor restricted activity days (MRAD).[4] MRADs capture most symptoms and illnesses associated with smoke exposure, including those requiring physician or hospital treatment. Thus, it is a broad-based measure of health effects. Incidence of MRAD closely matches our originally constructed WTP measure to avoid any wildfire smoke-related health impact. This allows us to compare health costs estimated using the built-in WTP measure with a wildfire-specific one.

The next step was to define the relevant air quality grid. PM2.5 was chosen for the geographic grid-area of Bernalillo County, NM, which completely contains Albuquerque and a majority of the metropolitan population. A Community Multi-scale Air Quality Model (CMAQ) 12 $\times$ 12 km grid was used in BenMAP-CE. This is the finest grid size available in the program and is commonly used in the literature (US EPA 2010). A 50 km radius around each air quality monitoring station was used to define the extent of the CMAQ grid. Incidence and valuation results were calculated for each grid-cell and aggregated to the county level (Bernalillo).

BenMAP-CE calculates incidence based on differences in air quality between a baseline and a treatment. Our treatment is the air quality level in Albuquerque, NM during the Wallow mega-fire. The fire started in southeastern Arizona on 29 May 2011, but due to atmospheric conditions, quickly developed a smoke plume that affected the Albuquerque area (some 200 miles away) by the next day. Sporadically for several weeks in June, smoke from the mega-fire significantly impacted PM levels in Albuquerque and across much of northern and western New Mexico, with measured daily PM2.5 levels spiking to 70.5 $\mu g/m^3$ at one monitoring site in Albuquerque on June 4; an increase of 675% above the three-year site average for that day.

A Wallow mega-fire smoke event day was identified as monitored daily-average PM2.5 levels in excess of the 99th percentile of daily average readings per monitor site over the previous five years (2008–2012). Only smoke event days occurring during the wildfire event period (29 May−8 July) were considered. The 99th percentile threshold for an event day is consistent with recent literature in this area (Johnston et al. 2011). If PM2.5 levels did not exceed the 99th percentile for the site, then that day was not considered a smoke event day for that monitoring station and baseline PM2.5 readings were used. Monitoring stations were excluded from this part of the analysis if they did not report PM levels for the entire month of June or had at least three missing observations during the Wallow mega-fire event.

3.2. *WTP estimation using the defensive behaviour method*

We employ the defensive behaviour method to estimate wildfire-specific WTP. This method captures individual WTP for changes in health status caused by a pollutant, incorporating costly averting and mitigating behaviours. Examples of averting behaviour include staying indoors, wearing a mask, or avoiding work during a wildfire smoke event, while examples of mitigating behaviour would be buying medicine or being admitted to the hospital. The model applied here is from Freeman (2003) and Dickie (2003).

The basis of the model is the individual's health production function, $h(.)$, which relates exogenous environmental exposure to a pollutant (p), averting actions (a), and mitigating actions (m) to changes in health status (s). An individual produces a health output according to a health production function,

$$s = h(p, a, m, \boldsymbol{x}), \tag{1}$$

where $\boldsymbol{x}$ is a vector of socioeconomic and demographic variables affecting health. Exposure to the pollutant decreases health ($h_p < 0$) and averting and mitigating actions are assumed to be non-harmful ($h_a, h_m \geq 0$). Effects of socioeconomic variables are not *a priori* clear. For this application, p captures exposure to wildfire smoke and s reflects whether or not an individual experienced a wildfire smoke-related health effect.

Using Equation (1) within a utility maximisation framework, Freeman (2003) derives the marginal WTP for an exogenous reduction in illness from an averting action as

$$WTP = -\frac{p_a}{\frac{\partial s}{\partial a}}. \tag{2}$$

That is, the WTP for a reduction in illness is the negative ratio of the price of the averting action to the marginal effect of that action on health status. Using Equation (2) relies on an empirical estimate of $\partial s/\partial a$, obtained from Equation (1) as a post-estimation marginal effect.

For estimation of Equation (1), a linear-in-the-parameters probit model is applied,

$$\Pr(s = 1 \mid p, a, m, \boldsymbol{x}) = \Phi(\beta_0 + \beta_1 p + \beta_2 a + \beta_3 m + \boldsymbol{x}'\boldsymbol{\gamma}), \tag{3}$$

where $\Phi(.)$ is the standard normal cumulative distribution function. The binary dependent variable (s) is whether the respondent experienced a wildfire smoke health effect (Yes/No). The coefficient of interest for an averting action is β_2 - the marginal effect of averting actions (a) on health status (s). A probit model is employed because the data indicates whether or not an individual experienced a smoke-related health effect, but not the duration or symptom days of health effects. This data limitation precludes using a symptom count model as done by Richardson, Champ, and Loomis (2012) for wildfire smoke. Fitting a count data model without a clear picture of the temporal symptom profile would necessitate potentially strong assumptions on average symptom days and their intensity as experienced by individuals, potentially biasing estimated WTP. A probit specification of the type employed here does not require assumptions on symptom profiles and is more consistent with available data, though is arguably not as informative as count models in determining the relationship between averting actions and health status because it only captures a discrete change instead of intensity of symptoms.

Endogeneity is often a concern in estimation of Equation (3) as an individual might take an averting action to prevent an illness (averting behaviour causing health status) or an ill individual might take an averting action to limit future symptoms (health status causing averting behaviour). The direction of causation is unclear. To prevent biased coefficient estimates (Wooldridge 2011), endogeneity will be purged using a two-stage maximum likelihood instrumental variables approach (Freedman and Sekhon 2010).

To econometrically estimate (3), we follow Richardson, Champ, and Loomis (2012) and investigate the relationship between health status and the averting behaviour *used air filter/cleaner*. This is because *used air filter/cleaner* is endogenous to the model[5] and is the only averting action whose coefficient is negative, both as theoretically predicted.

Furthermore, prices of air filters, cleaners, and purifiers (p_a) are readily available from many sources.[6] Equation (3) is estimated as a two-stage bivariate probit model with *used air filter/cleaner* as the primary independent variable of interest. Other covariates include smelling smoke at home, a control for chronic respiratory disease, controls for symptoms (headaches, coughs, shortness of breath, asthma, allergies, and other symptoms), education, years in NM, Latino, and number of children under age five in the home. We use *income* as an instrument in a second-stage equation prediction of *used air filter/cleaner*. This is consistent with Richardson, Champ, and Loomis (2012). *Income* is found to be a relevant, but not particularly strong instrument. The post-estimation marginal effect of *used air filter/cleaner* will be used in conjunction with the average price of an air filter/cleaner, \$29.71, to determine WTP for a reduction in wildfire smoke health effects.

4. Data

4.1. Air quality data

Daily monitored PM2.5 air quality data from stations in the Albuquerque metropolitan comes from US EPA AirData[7] for the years 2008−2012, inclusive. The raw data contain daily 24-hour average PM2.5 levels in units of $\mu g/m^3$ per air monitoring site. There are six sites in Albuquerque and one site in Valencia County located in the town of Los Lunas (immediately south and adjacent to Albuquerque). Figure 1 shows the locations of the seven monitoring sites.

For creation of the baseline air quality grid, an average for each site was calculated using five years (2008−2012) of site-specific daily mean PM2.5 concentrations. This yields 365 (366 for a leap year) observations per site. PM2.5 deviations from the five-year daily average during a wildfire event are the source of variation BenMAP-CE uses to identify health impacts.

4.2. Wildfire experience survey

Researchers at the University of New Mexico developed and administered a survey questionnaire on wildfire risk and surface water supply to a sample of households in Albuquerque, in September-December, 2014. The focus of the survey was on household support for implementation of a payment for ecosystem services model to reduce wildfire risk in the larger forested watershed approximately 100 miles or more from Albuquerque. The survey was part of a larger New Mexico EPSCOR/NSF-funded grant to study wildfire and water in New Mexico. Focus groups, individual interviews, and pre-tests were used to aid development of the survey instrument. Up to five contacts were mailed to 2596 households in Albuquerque selected from a sampling frame of 190,000 Bernalillo County homeowners, consisting of (1) an initial cover letter; (2) a survey packet; (3) a reminder postcard; (4) a second survey packet; and (v) a final letter with an additional survey packet. Contacts were mailed until a returned survey packet was obtained, the respondent notified us they did not want to participate, or a contact came back as undeliverable. Respondents could choose to complete the questionnaire online or via mail. Out of the 2596 questionnaires mailed, 133 were undeliverable. 911 were returned (751 by mail and 160 online) for a response rate of 37%.

A section of the questionnaire asked about past wildfire smoke experience, health effects of past exposure, and averting or mitigating actions taken to avoid exposure. Respondents were asked if they had ever smelled smoke from a wildfire at their home

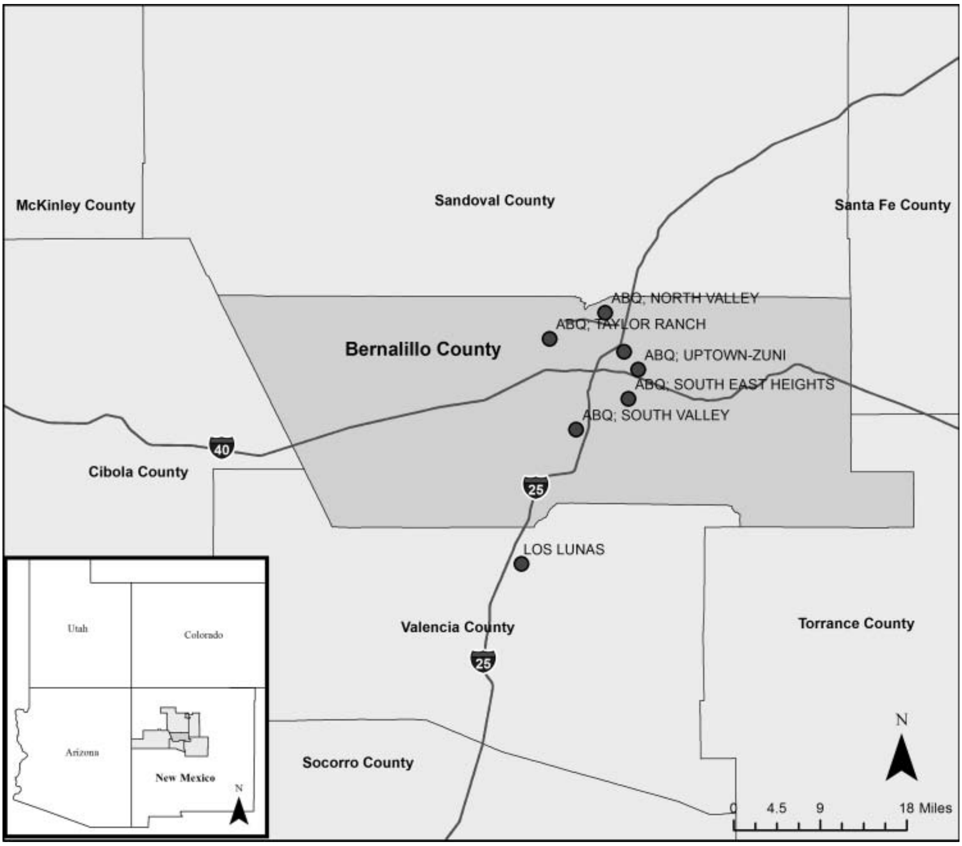

Figure 1. US EPA air quality monitoring sites for Bernalillo and Valencia counties, NM (Albuquerque metro area).
Source: constructed by the authors in ESRI ArcMap 10.1.

and if exposure to smoke had ever affected their health. If they had smelled smoke, they were asked to indicate symptoms experienced (e.g. headache, cough, etc.), averting actions taken (e.g. stayed indoors, used an air purifier, etc.), and if they took a mitigating action (e.g. went to the hospital/physician).

The survey results on health effects and demographics are used to econometrically estimate WTP to avoid a wildfire smoke-related illness. Variable definitions on health and demographic questions and their summary statistics are presented in Table 1. Results indicate that 71% of respondents took at least one averting action during previous wildfire smoke events, compared to 1% of respondents who took a mitigating action.

Survey results also indicate that a significant percentage of respondents have smelled wildfire smoke at their home (88%), with 26% reporting that wildfire smoke has impacted their health at some point. The most commonly reported symptom associated with wildfire smoke is a cough (25%) followed closely by allergies (21%). Fifty-five percent of respondents reported staying indoors more than usual during a smoke event, with 42% reporting avoiding normal outdoor recreation and exercise. Overall, survey results demonstrate considerable health and behavioural impacts of wildfire smoke in the Albuquerque metropolitan area.

Table 1. Wildfire survey variable definitions and summary statistics ($n = 911$).

Variable	Coding	Mean	Std. dev.	Min.	Max.
Wildfire smoke exposure					
Smelled smoke at home	1 = yes, 0 = no	0.88	0.33	0	1
Own a home north of Albuquerque	1 = yes, 0 = no	0.10	0.29	0	1
Smoke has affected health	1 = yes, 0 = no	0.26	0.44	0	1
Changed routine because smelled smoke	1 = yes, 0 = no	0.47	0.50	0	1
Averting actions					
Evacuated	1 = yes, 0 = no	0.05	0.22	0	1
Covered face with mask	1 = yes, 0 = no	0.07	0.25	0	1
Used air filter/cleaner	1 = yes, 0 = no	0.16	0.37	0	1
Avoided work	1 = yes, 0 = no	0.01	0.11	0	1
Removed ashes from property	1 = yes, 0 = no	0.04	0.20	0	1
Stayed indoors	1 = yes, 0 = no	0.55	0.50	0	1
Avoided outdoor recreation/exercise	1 = yes, 0 = no	0.42	0.49	0	1
Took no averting action	1 = yes, 0 = no	0.29	0.45	0	1
Mitigating actions					
Visited physician or admitted to hospital related to smoke exposure	1 = yes, 0 = no	0.01	0.12	0	1
Symptoms					
Headaches	1 = yes, 0 = no	0.14	0.35	0	1
Coughs	1 = yes, 0 = no	0.25	0.43	0	1
Dizziness	1 = yes, 0 = no	0.02	0.13	0	1
Blurred vision	1 = yes, 0 = no	0.03	0.17	0	1
Shortness of breath	1 = yes, 0 = no	0.11	0.31	0	1
Asthma	1 = yes, 0 = no	0.11	0.31	0	1
Allergies	1 = yes, 0 = no	0.21	0.41	0	1
Experienced none of the above symptoms	1 = yes, 0 = no	0.49	0.50	0	1
Health history					
Chronic respiratory disease	1 = yes, 0 = no	0.33	0.47	0	1
Heart disease	1 = yes, 0 = no	0.14	0.35	0	1
Demographics					
Female	1 = yes, 0 = no	0.37	0.48	0	1
White	1 = yes, 0 = no	0.80	0.40	0	1
Hispanic/Latino	1 = yes, 0 = no	0.28	0.45	0	1
College graduate	1 = yes, 0 = no	0.62	0.49	0	1
Income	1 = <14,999; 2 = 15,000–24,999; 3 = 25,000–34,999; 4 = 35,000-49,999; 5 = 50,000–74,999; 6 = 75,000–99,999; 7 = 100,000-149,999; 8 = 150,000–199,999	5.30	1.74	1	8
Years lived in New Mexico	Continuous	35.26	19.32	1	85
Number of children under 5 in house	Continuous	0.12	0.39	0	3

(*continued*)

Table 1. (*continued*).

Variable	Coding	Mean	Std. dev.	Min.	Max.
Beliefs					
Effectiveness of averting actions	1 = 'not at all effective'; 2 = 'slightly effective'; 3 = 'somewhat effective'; 4 = 'moderately effective'; 5 = 'highly effective'	3.19	0.97	1	5

5. Results

5.1. Wildfire smoke health incidence

Increases in incidence associated with smoke exposure from the Wallow mega-fire event are substantial (Table 2). Health endpoints are listed separately, by column, and incidence results are categorised by source of transferred CR function. Reported incidences are the number of additional visits per Albuquerque resident estimated by BenMAP-CE over the five-week smoke event.[8] For MRADs, the reported incidence is in units of person-days. Across all endpoints, MRADs are the largest health effect associated with wildfire smoke. Over 14,700 person-days of minor health impacts are created, which corresponds to an average of 0.03 days or about 44 minutes per Albuquerque resident over the event. *Emergency room asthma* visits are the second largest health impact in the wildfire-specific benefit transfer, with 16 additional cases over the five-week event. Less than one additional emergency room visits is estimated by BenMAP-CE when an urban air CR function is used. *Hospital admissions for any respiratory illness* increase between 3 and 5 cases, depending on type of CR function. Increases in admissions due to pneumonia (2.7 cases) are the largest component of overall respiratory admissions based on urban air CR functions, though when wildfire functions are used, increases in asthma admissions (2.2) drive overall respiratory admissions.[9]

Incidence estimates based on wildfire-specific CR functions are between 43% (*hospital admission all respiratory*) and 2617% (*emergency room asthma*) larger than incidences estimated using urban air study results. This is consistent with the literature on greater morbidity impacts of wildfire smoke compared to urban air pollutants (Kochi et al. 2010). However, *hospital admission pneumonia* incidence is 30% lower across the two sources, suggesting that perhaps wildfire smoke has less of an impact on pneumonia morbidity than typical urban air pollutants in this case study. MRAD incidence is identical across CR function sources because no wildfire-specific health impact function exists for this broad endpoint.

5.2. Health costs (COI and WTP) of smoke exposure

Built-in BenMAP-CE COI valuation functions are used to value changes in incidence associated with endpoints *emergency room asthma* and *hospital admission respiratory*. Inclusion of endpoints *hospital admission asthma* and *hospital admission pneumonia* would overestimate event COI because endpoint *hospital admission all respiratory* already includes all respiratory-related illnesses. Costs are estimated separately for urban air and wildfire-specific incidence results and are inflation-adjusted to 2014 US dollars (USD). Total COI health costs of Wallow mega-fire smoke exposure in the Albuquerque

Table 2. Smoke exposure health incidences by CR function source for Wallow Mega-Fire.

	Incidence endpoint (number of cases)				
Source of CR function	Emergency room: asthma	Hospital admission: all respiratory	Hospital admission: asthma	Hospital admission: pneumonia	Minor restricted activity days (MRAD)
Urban air quality literature	0.6	3.5	1.2	2.7	14,786.4
Wildfire smoke literature	16.3	5.0	2.2	1.9	14,786.4
Percentage change	(+) *2616.7%*	(+) *42.9%*	(+) *83.3%*	(−) *29.6%*	(+) *0.0%*

Notes: Emergency room asthma incidence for urban air quality results is average of incidences from Mar et al. (2010) and Slaughter et al. (2005) transferred functions. Percentage change in incidence from urban air to wildfire smoke is reported in the final row.
Source: BenMAP-CE calculations (v.1.0.8). Urban air quality literature results come from BenMAP-CE calculations using CR functions from the urban air literature that were selected from the existing BenMAP-CE database. Wildfire smoke literature results come from BenMAP-CE calculations using CR functions from the wildfire smoke literature that were manually and individually added by the authors to the BenMAP-CE database. Incidence of minor restricted activity days is identical across CR function sources because only one literature estimate exists for this incidence endpoint.

metropolitan area are estimated by BenMAP-CE at $74,000 in the urban air function analysis and $111,000 in the wildfire function analysis (Table 3). These are the medical costs associated with diagnosis and treatment plus lost wages due to illness, estimated by BenMAP-CE and aggregated by the software for Bernalillo County. *Emergency room asthma* costs are 2535% higher and *hospital admission all respiratory* costs are 44% higher in the wildfire benefit transfer, reflective of greater estimated morbidity when wildfire-specific CR functions are used.

To estimate WTP to avoid a wildfire smoke health effect, the survey data is used, and a bivariate probit version of Equation (3) is estimated. For brevity, individual results from estimation of that model are reported in the appendix. The marginal effect associated with *used air filter/cleaner* is estimated at −0.227 with a 95% confidence interval of (−0.400, −0.054). Dividing the negative of the average price of an air cleaner, $29.71, by −0.277 and adjusting for inflation, produces $130.79 (range: $74.19−$551.33), which is the marginal WTP for a reduction in wildfire smoke health effects. This is larger than a

Table 3. COI health costs (2014$) by CR function source for Wallow Mega-Fire.

	Incidence endpoint costs		
Source of CR function	Emergency room: asthma	Hospital admission: all respiratory	Total
Urban air quality literature	$177	$73,760	$73,937
Wildfire smoke literature	$4,664	$106,405	$111,069
Percentage change	(+) *2535.0%*	(+) *44.3%*	(+) *50.2%*

Notes: Cost-of-illness (COI) event costs reported. Emergency room asthma costs for urban air quality results is average of costs based on estimated incidence from Mar et al. (2010) and Slaughter et al. (2005) transferred functions. Percentage change in COI from urban air to wildfire smoke is reported in the final row.
Source: BenMAP-CE calculations (v.1.0.8). COI calculated by BenMAP-CE separately by source of CR functions. Urban air quality literature results come from BenMAP-CE calculations of the COI associated with health incidence estimated using CR functions from the urban air literature. Wildfire smoke literature results come from BenMAP-CE calculations of the COI associated with health incidence estimated using CR functions from the wildfire smoke literature.

Table 4. WTP health costs (2014$) by CR function source for Wallow Mega-Fire.

Source of WTP metric	Minor restricted activity days (MRAD)	WTP-COI ratio
Urban air quality literature	$337,623	4.6
Wildfire smoke (originally estimated)	$429,156	3.9
Percentage change	(+) *27.1%*	*NA*

Notes: Willingness to pay (WTP) event costs reported. NA = not applicable. Percentage change in WTP from urban air to wildfire smoke is reported in the final row. WTP-COI ratio is the ratio of WTP and COI (Table 3) health costs per economic value transfer.
Source: BenMAP-CE calculations (v.1.0.8). Urban air quality literature results come from BenMAP-CE calculations of the WTP health costs associated with minor restricted activity days when an urban air WTP value ($98) is used from the existing BenMAP-CE valuation database. Wildfire smoke results come from BenMAP-CE calculations of the WTP health costs associated with minor restricted activity days using an originally estimated wildfire-specific WTP value ($130.79) that was manually coded into the software's valuation database.

comparable urban air WTP value of $98 (range: $68−$146) from Dickie and Messman (2004) and also larger than the only other wildfire-specific WTP of $93.15 (no bounds provided) estimated by Richardson, Champ, and Loomis (2012).

To arrive at an estimate of WTP health costs using wildfire-specific study results, MRAD results from Table 2 were put into per capita terms and multiplied by the previously estimated marginal WTP of $130.79. Table 4 reports the results from these wildfire-specific calculations, performed in BenMAP-CE, and the results from a separate urban air specific estimation in BenMAP-CE using the Dickie and Messman (2004) urban air WTP value. Estimated health costs utilising WTP to avoid an air pollutant-related symptom range from $338,000 (urban air value) to $429,000 (wildfire smoke value); an increase of 27%. Additionally, WTP health costs are between 3.9 and 4.6 times larger than comparable COI health costs. This is suggestive evidence of differences in estimated wildfire smoke health costs depending on the source of transferred economic value (urban air vs. wildfire-specific) and economic value utilised (WTP vs. COI). Although, had we instead transferred the Richardson, Champ, and Loomis (2012) WTP value estimated for Southern California, the resulting WTP health costs would be similar to the urban air costs. While our estimated WTP value is arguably more appropriate – it is specific to wildfire and the study site (Albuquerque) – in other settings (e.g. California) perhaps urban air and wildfire WTP values are not meaningfully different. That said, there is clearly room for additional original valuation studies for health effects associated with wildfire events and comparisons of WTP values in other contexts, accounting for differences in study design, estimation methods, and sampled populations.

6. Conclusions

This case study illustrates how a robust benefit transfer tool, BenMAP-CE, can be applied to wildfire smoke damage assessments in an urban area. Application results demonstrate sensitivity to choice of transferred CR function and economic values. Use of transferred results from urban air quality studies undervalues the health impacts of wildfire smoke compared to use of wildfire-specific study results. We find higher incidences of emergency room asthma visits, all respiratory, and asthma hospital admissions when CR functions from the wildfire smoke literature are utilised. Increases range from 43% to 2617%. Health costs in the wildfire benefit transfer are 27% (WTP) to 50% (COI) larger than costs in the urban air benefit transfer.

These findings provide the first empirical support (albeit from a case study) of Kochi et al.'s (2010) recommendation to use wildfire-specific study results when possible in wildfire damage assessments. Short-duration, extreme shocks to air quality created by wildfire smoke plumes appear to have more significant impacts on human health than the same pollutant concentration change to levels of urban air quality. This may be because a concentration, *X*, of wildfire smoke is more toxic than the same concentration, *X*, of urban air (Vedal and Dutton 2006). For these reasons, it's our recommendation that analysts performing wildfire smoke benefit transfers carefully consider the source of transferred functions so as not to over- or under-value morbidity changes and associated health costs of wildfire smoke.

The urgency to better understand how wildfire affects health, across many dimensions, especially in the western US, is underscored by climate change, possible connections to carbon sequestration capacity of standing forests (see Koirala and Mysami 2015), increased depletion of water in the west, continued growth of western cities, and recent increases in the number and severity of wildfires associated with prolonged drought in many western US states. BenMAP-CE is a useful tool for studying wildfire health impacts that researchers should be aware of and we hope this empirical demonstration spurs additional applications of the program to wildfire damage events. While our study shows where some of the important choices are, it also illustrates where additional data collection may be needed, including wildfire-specific CR functions for health impacts (mortality, especially) and wildfire-specific WTP measures.

There are several limitations of this analysis. First, the only air quality impact considered was PM2.5 and not other pollutants created by wildfires such as ozone, NO2, etc. Including these in the analysis would raise both incidence and health costs of a wildfire smoke event. Second, only monitored air quality data were used. A more sophisticated approach would incorporate modelled air quality data to interpolate PM in areas where sites are not located. It is unclear how this would alter results, but given the significant number of monitoring sites in Bernalillo County and close proximity to one another, the advantages of modelling are diminished. However, modelled data might be appropriate in areas with fewer proximal sites. Third, statistical differences in study design, sampled populations, or estimation methods across the urban air and wildfire-specific CR functions used in this analysis were not investigated. Previous research (e.g. Kochi et al. 2010; Vedal and Dutton 2006) has examined this issue and offered various explanations on how and why wildfire smoke may be more damaging to health. The focus of the present research is on the meaningful application of a benefit transfer tool to wildfire smoke, inclusive of previous literature demonstrating differences between urban air and wildfire smoke. A formal statistical analysis exploring how or why wildfire smoke may be more damaging to health is outside this scope. Thus, an important caveat is that while a comparison of BenMAP-CE calculated point estimates is informative and illustrative of how differences in CR functions and WTP values manifest into overall damage assessment heterogeneity, it is not sufficient evidence to draw general conclusions regarding the relationship between smoke and urban air.

Fourth, a formal statistical analysis to assess the consistency of BenMAP-CE with previously estimated costs of wildfire smoke events was not performed. This case study is only a proof-of-concept that BenMAP-CE can be adapted to study wildfire smoke effects in urban areas. While an important first step, future work should use BenMAP-CE to replicate wildfire smoke economic assessments carried out using other means (e.g. questionnaires, hospital admissions, emergency room visits, etc.). Results from BenMAP-CE, after being modified to assess wildfire smoke events, could be statistically compared.

Such an approach would provide more conclusive evidence on the viability of the program for wildfire smoke analyses. We hope that this case study spurs such further investigations.

Disclosure statement

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Notes

1. BenMAP-CE is accessible at: http://www2.epa.gov/benmap.
2. As an example of the increasingly common mega-fires seen in recent years in the western USA, the Wallow Fire burned more than 538,000 acres (841 square miles), and is the largest fire on record in Arizona (Ryan and Opperman 2013). The fire covered parts of four counties in eastern Arizona and one in southwestern New Mexico. The fire started near the Bear-Wallow Wilderness Area in the Apache National Forest, and the ignition source was from an unattended campfire. More than 6000 people were evacuated and physical property damages have been estimated to be over $109 million (Ryan and Opperman 2013). While not the source of any known fatalities, the smoke plume extended across New Mexico and into Texas and Oklahoma.
3. Appendix C of the BenMAP User's Manual (Abt Associates 2012) describes the health impact function derivation process, though we note that it's easily performed by-hand.
4. A MRAD is defined as any day on which an individual was forced to alter his or her normal activities due to minor illnesses, including both respiratory and nonrespiratory conditions (Ostro and Rothschild 1989). One shortcoming is that the single MRAD CR function that exists (Ostro and Rothschild 1989) is estimated for urban air quality.
5. Endogeneity was tested for using a Wu-Hausman F-test ($p < 0.01$) and a Durbin-Wu-Hausman χ^2-test ($p < 0.01$).
6. The inflation-adjusted price reported in Richardson, Champ, and Loomis (2012) of $29.71 (2014$) is used. This result is an average of self-reported prices (including $0). Prices from other sources (e.g., Home Depot, Amazon.com) were investigated and results are available upon request.
7. http://www.epa.gov/airdata/ad_data_daily.html.
8. Results presented in the Table 2 row labelled 'urban air quality literature' are estimated by BenMAP-CE using urban air quality CR functions selected by us in the programs' graphical user interface, which had already been coded into the software by US EPA programmers for the five health endpoints listed. Similarly, results in the row labelled 'wildfire smoke literature' are estimated by BenMAP-CE using wildfire-specific CR functions that we manually and individually coded into the program for four of the five incidence endpoints. All individual results in Table 2 are produced by BenMAP-CE calculations.
9. The health endpoint 'hospital admission: all respiratory' is comprised of both asthma and pneumonia admissions, in addition to any other respiratory illnesses (not estimated in this analysis). The two endpoints separately illustrate specific drivers of overall respiratory admissions using endpoint-specific CR functions.

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