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How Will Climatic Change Affect Air Quality in Parks and Wilderness?

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Introduction

As global temperatures and populations increase and demands on natural resources intensify through the 21st century, management options will become more constrained and more trade-offs will have to be evaluated. For example, in the USA land managers use prescribed fire for restoring and maintaining ecosystems (Allen et al. 2002). In landscapes in which fire severity was low prior to active suppression but fuel loadings are now higher than they were historically, prescribed fire can also reduce the risk of catastrophic wildfire that would threaten key resources or human communities. Land managers in protected areas (national parks and wilderness) have adopted a policy of "wildland fire use," whereby they allow naturally ignited fires to burn unless they threaten one or more values—typically fire risk to structures or ambient air quality—held to be of higher priority.

Fire effects on air quality can be both local and regional. On actual burns and in watersheds immediately downwind of prescribed fires, smoke exposure causes respiratory problems even in healthy people, but is especially problematic for those with asthma or other chronic respiratory problems. Particularly hazardous are the particulate emissions smaller than 2.5 μ in diameter ($PM_{2.5}$), which can be breathed more deeply and cross protective membranes in the lungs (Kreyling et al. 2004).

These same particulates and other elements of the smoke plume can contribute significantly to visibility impairment hundreds of kilometers downwind from emissions sources (Malm 1999). In the western United States, regional haze from fires and other sources reduces visibility in most of the protected areas at some time during a typical year.

To maintain air quality in Class I areas into the future we need to understand not only present-day emissions from fires, but also how they may change over time in response to climatic changes, land use, and management strategies. Fire regimes will likely evolve in response to temperature increases and vegetation changes associated with them (McKenzie et al. 2004). Specifically, annual area burned by wildfire is expected to increase across the western United States and Canada (Flannigan et al. 1998; McKenzie et al. 2004). Fires in many ecosystems are already becoming more severe than they were historically because of increasingly severe fire weather, unnatural fuel buildup from fire suppression, or both (Agee 1997; Allen et al. 2002). Increases in area burned and fire severity increase biomass consumption and smoke emissions, and consequently atmospheric dispersion of particulates and aerosols that produce regional haze.

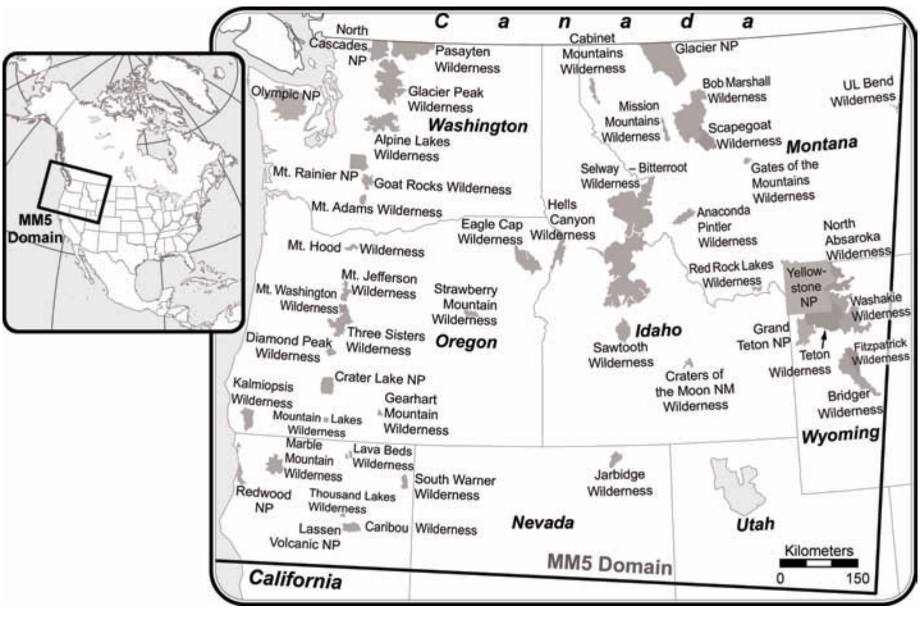
In this paper we describe the integration of four simulation models, an array of GIS raster layers, and a set of algorithms for fire danger calculations (National Fire Danger Rating System, or NFDRS; Cohen and Deeming 1985) into a modeling framework for simulating regional-scale smoke dispersion. We focus on a representative fire season (2003) in the Pacific Northwest, USA, and track the simulated dispersion and concentration of $PM_{2.5}$ over the course of the season. We compare summary statistics for simulations to real data for the same time period, and briefly discuss implications for management of parks and wilderness into the future.

Study area

Our study area is the Pacific Northwest 12-km domain used in real-time forecasts from the MM5 mesoscale meteorological model (Grell et al. 1994; Mass et al. 2003) as shown in Figure 1. In this region, steep gradients in elevation, precipitation, and temperature exist across multiple scales. The diversity of climatic conditions, topography, and elevations supports a variety of ecosystem types, including coastal temperate rainforest, subalpine parkland and alpine meadows, drier mixed-coniferous forests, and semi-arid shrublands and grasslands.

Fire regimes within the Pacific Northwest include large, stand-replacing fires (Agee and Smith 1984); mixed-severity, medium-frequency fires (Morrison and Swanson 1990); and

Figure 1. Class I wilderness areas in the MM5 12-km modeling domain for the Pacific Northwest. Inset shows the full extent of the domain, which includes parts of southwestern Canada and the northeastern Pacific Ocean.



low-severity, high-frequency fires (Agee 1993). Lightning is the main source of wildfire ignitions in our study area (Rorig and Ferguson 1999), and climatic variability, both within and among fire seasons, is the dominant control on fire occurrence and fire extent within the region (Hessl et al. 2004; Gedalof et al. 2005).

Methods

The framework of the integrated modeling system is shown in Figure 2. Multiple dependencies exist among elements. For example, climate affects fire severity directly through fire weather, but also indirectly through its effects on vegetation and associated abundance and distribution of fuels. Within the conceptual framework, we delineated three modules: (1) a fire scenario builder (FSB) that simulates fire starts and fire sizes as a function of fire meteorology and historical fire frequency, (2) a consumption and emissions module that calculates particulate and aerosol emissions from biomass consumed in the fires, and (3) a smoke dispersion module that simulates the smoke plume and atmospheric dispersion of emissions from each fire. For this study we simulated only lightning-caused wildfires.

Fire scenario builder. The FSB uses climatic information (historical observations or future climate simulations) to determine a scenario of fire starts, sizes, and locations that can be then used by the consumption module. The FSB is designed to accept three input layers: meteorology, vegetation/fire regime, and management, but for this exercise we omitted any management options. We used the "natural background" of annual area burned associated with potential natural vegetation in the region, and used simulated daily meteorological output to downscale annual area burned to individual fires and increase or decrease it proportionally based on fire weather.

From the MM5 model, we extracted all necessary meteorology variables needed for the modeling system, including surface temperature, relative humidity, and rainfall. As a proxy for atmospheric instability, and therefore the probability of lightning, we calculated the maximum CAPE (convection-available potential energy) statistic for each day at each 12-km grid cell. Lightning was simulated when max(CAPE) > 1000, creating 4–5 episodes of sufficient lightning potential during the fire season, similar to what is observed.

The potential for lightning to trigger a fire was estimated using the NFDRS (Cohen and Deeming 1985), which provides a set of algorithms for estimating fire danger. We used the equations in Cohen and Deeming (1985) to calculate daily equilibrium moisture content (EMC) from MM5 output in the size classes of surface fuels (0.6–8.0 cm diameter) most important for fire spread. A fire was "ignited" on the ignition day for a cell

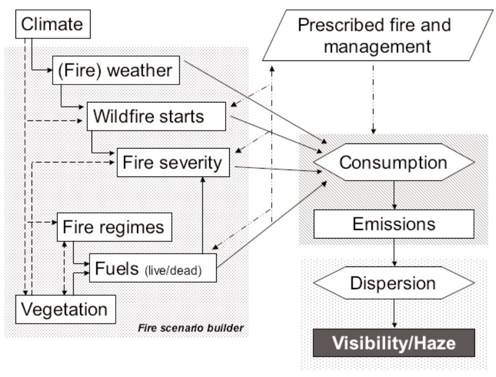


Figure 2. Integrated modeling framework for simulating regional haze from wildland fire. Interactions with solid arrows are activated in the current paper. Dotted arrows indicate interactions that are turned off or for which default values are assumed. See text for explanation.

if the weighted average fuel moisture percentage in the 0.6–8.0 cm size class was below 25%, considered a default threshold for fire danger modeling (Cohen and Deeming 1985).

Fire sizes were simulated in the following way. The fuel moisture damping coefficient from NFDRS (Cohen and Deeming 1985) was calculated with an extinction level of 25%. Using the "expected" area burned for each cell as a mean, the damping coefficient defined the quantile of a negative exponential distribution with that mean. The cell's "fire size" was adjusted to the associated quantile. Fires produced in this way ranged from the miniscule up to 2,800 ha, with the majority being under 40 ha. Fires under 40 ha were then eliminated. Because real fires under this size are not tracked, they are excluded from emissions inventories and thus should be absent from our simulated inventories.

Potential fire duration was a linear function of the adjusted fire size. Total area burned was then assigned to "ignition days" and days following, if any, proportionally to the weighted-average fuel moisture values for each day. A fire "went out" if fuel moisture reached 25%, but area burned was not truncated; rather, it was renormalized to occur in the consecutive days after ignition whose fuel moisture was below 25%.

Consumption and emissions module. The consumption and emissions modules are currently nested in the BlueSky Smoke Modeling framework (http://www.fs.fed.us/bluesky/; O'Neill et al. 2003). Fuel loadings in the Pacific Northwest domain were obtained from a 1-km GIS layer developed by the U.S. Forest Service (Hardy et al. 1998). Within BlueSky, area burned for each day and fuel loadings for each cell were passed to the Emissions Production Model (EPM; Sandberg and Peterson 1984), which calculates hourly consumption, heat release, and smoke emissions ($PM_{2.5}$ and PM_{10}, CO_2, CO, VOC, NMHC) from fires based on an exponential mixture model of flaming and smoldering stages of combustion.

Dispersion module. The emission estimates from EPM, along with meteorology from MM5, are processed for the CALPUFF Gaussian dispersion model (Scire et al. 2000). CALPUFF is a puff dispersion model that simulates point, volume, or area sources, assuming that plume dispersion occurs in a Gaussian pattern. CALPUFF also estimates plume rise and accounts for density differences between the plume and the ambient air. A pre-processing program, EPM2BAEM, converts the emissions from EPM into an area emission source suitable for input into CALPUFF. It calculates flame height (Cetegen et al. 1982) using the heat-release estimates from EPM and vertical velocity of the smoke plume, assuming conservation of buoyancy flux proportional to heat-release rate.

Data output. We ran the simulations through a 61-day period in the summer of 2003, producing $PM_{2.5}$ concentrations across the MM5 Pacific Northwest domain. In this paper we focus on $PM_{2.5}$ concentrations in selected Class I Wilderness Areas within the domain (Figure 1). We recorded the maximum of 24-hour running means of $PM_{2.5}$ over all 12-km cells included in the Class I area. We then calculated an extinction coefficient to represent the worst-case reduction in visibility from pristine conditions associated with the 24-hour concentrations of $PM_{2.5}$ from fire only (Engling et al. 2004).

Engling et al. (2004) found, in a study of aerosols in Yosemite National Park, that $PM_{2.5}$ from fire was 80% organic carbon (OC). Assuming that this finding is applicable to fire across the western U.S.; assuming that the ratio of OC to elemental carbon (EC) from fire is 9:1; and neglecting sulfate, nitrate, and fine soil, the extinction coefficient from fire only is:

$$\beta ext = 4[OC] + 10[EC] = (4)(0.8)[PM_{2.5}] + (10)(0.8)[PM_{2.5}]/9$$

where [ ] indicates concentration ($\mu g\ m^{-3}$), OC is organic carbon, EC is elemental carbon, and βext is in units of Mm^{-1} (USEPA 1999).

We used the WinHaze Visual Air Quality Modeler (Air Resource Specialists 2004) to visualize the visibility reduction from modeled $PM_{2.5}$ concentrations. This allowed us to compare simulated reductions in visibility to a library of photographs of Class I areas (IMPROVE 2004), thereby qualitatively estimating the percentage of regional haze attributable to smoke dispersion from fire by comparing WinHaze output for days with the highest extinction coefficients to library photos of days with the worst visibility. One can also quantitatively compare results to the highest extinction coefficients reported for a particular Class I area on the IMPROVE network (http://vista.cira.colostate.edu/improve/).

Results

Area burned tracked the number of fires started for most days, reflecting the contribution of fuel moisture calculations, and particularly the extinction threshold of 25% in woody fuels, to both variables. One exception occurred on 16 July (day 197), when 48 fires started but only about 4,000 ha burned, reflecting extensive simulated atmospheric instability but without, on average, dry woody fuels. In late August, total fire activity was greatly reduced, reflecting widespread precipitation across much of the domain.

Using these simulated fires, consumption and dispersion were calculated to yield smoke concentrations throughout the domain. From 24-hour mean concentrations of $PM_{2.5}$, light extinction coefficients (βext) were computed at each Class I area in the domain. Figure 3 shows βext for selected Class I areas in northern Idaho and western Montana. The maximum βext is 144 Mm^{-1} in the Selway-Bitterroot Wilderness Area in late August (Figure 3).

The cause of this August spike is due to much of the fire activity in late August being concentrated upwind of the Selway-Bitterroot. Based on comparison with data from the IMPROVE visibility monitoring program, the maximum simulated value (144 Mm^{-1}

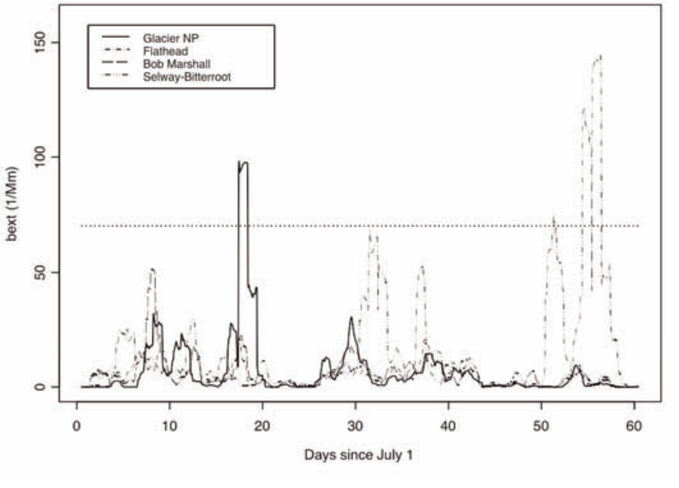


Figure 3. Twenty-four-hour running means of maximum extinction coefficients predicted for four Class I areas in Idaho and Montana (see Figure 1 for locations). Predictions for the Selway-Bitterroot are for the northern half only (see text). The horizontal line at 70 Mm^{-1} represents "significant degradation" according to national air quality standards.

in the Selway-Bitterroot) exceeds the 20 worst days average of any western national parks, but is below the maximum observed values for many western national parks.

Discussion

We simulated the contribution of wildfire to fine particulates ($PM_{2.5}$) that cause visibility reduction (regional haze) in Class I areas of the Pacific Northwest, USA, under historical (natural background) fire regimes, but using current fuel conditions.

The modeling system produced light extinction coefficient values at Class I areas within the range observed historically at western U.S. national parks. The simulated days of maximum reduction in visibility, in late August in the Selway-Bitterroot Wilderness Area and in early July in Glacier National Park, are analogous to observed visibility impairment from wildland fire. We expect that wildfires upwind of Class I areas will consistently reduce visibility—if not to record levels of degradation, then at least to levels associated with worst-case days by regulators and with unacceptable loss of scenic vistas to the public.

How will wildfire affect visibility in the future? Both empirical models (McKenzie et al. 2004) and process-based models (Lenihan et al. 1998) suggest that wildfire area will increase in the western USA with a warming climate. We can therefore expect the contribution of fire to regional haze and reduced visibility to increase. Our modeling system provides a framework for translating estimates of area burned into pollutant concentrations in Class I areas, provided that appropriate meteorological time series are available.

Given the anticipated complexity of future management and policy decisions, integrated multidisciplinary models are needed to guide management alternatives in the face of dynamic ecosystems and a warming climate. For example, adding prescribed fire scenarios or other means of fuel reduction will allow us to estimate the potential value that fuel treatments on multiple-use lands might have for maintaining pristine air quality in protected areas. Clearly there will be trade-offs between air quality and ecosystem restoration, and precise quantitative estimates of the effects of treatments will help land managers across the West to make informed choices.

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