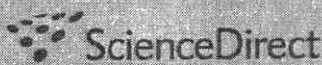
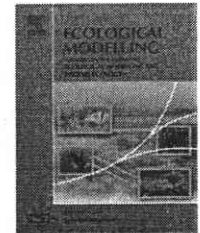


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## Integrating models to predict regional haze from wildland fire

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### ABSTRACT

Visibility impairment from regional haze is a significant problem throughout the continental United States. A substantial portion of regional haze is produced by smoke from prescribed and wildland fires. Here we describe the integration of four simulation models, an array of GIS raster layers, and a set of algorithms for fire-danger calculations into a modeling framework for simulating regional-scale smoke dispersion. We focus on a representative fire season (2003) in the northwestern USA, on a 12 km domain, and track the simulated dispersion and concentration of PM<sub>2.5</sub> over the course of the season. Simulated visibility reductions over national parks and wilderness areas are within the ranges of measured values at selected monitoring sites, although the magnitudes of peak events are underestimated because these include inputs other than fire. By linking the spatial and temporal patterns of haze-producing emissions to climatic variability, particularly synoptic weather patterns, and the stochastic nature of fire occurrence across the region, we can provide a robust method for estimating the quantity and distribution of fire-caused regional haze under climate-warming scenarios.

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### 1. 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 throughout the western states and in the Southeast (Brockway et al., 1997, 2002; Fulé et al., 1997; Stephenson, 1999; 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” (WFLU—Miller and Landres, 2004), 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.

Human-set prescribed fires face similar constraints on public, tribal, or private lands. The risk of fires escaping their designated perimeters is the most severe constraint, causing the timing of fires to be changed from the hours, days, or seasons in which they would most likely have burned historically. Prescribed fires that escape, because of mismanagement or unforeseen rapid changes in conditions, or both, have had high visibility with both policy makers and the public (e.g., Cerro Grande in 2000—<http://www.nps.gov/cerrogrande/Report.pdf>).

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Therefore, both prescribed fires and wildland fire use are typically prohibited when fire behavior may turn extreme due to low fuel moisture or unstable meteorology, whereas these were precisely the conditions under which the greatest area formerly burned. Scheduled prescribed fires are also frequently postponed or cancelled because unacceptable reductions in air quality are expected (Hardy et al., 2001). Fire effects on air quality can be both local and regional. On actual burns and in airsheds 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  $2.5\ \mu\text{m}$  in diameter ( $\text{PM}_{2.5}$ ), which can be breathed more deeply and cross protective membranes in the lungs (Dockery et al., 1993; US Environmental Protection Agency, 1996; 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. In accordance with the Clean Air Act and in order to maintain visibility standards in pristine, or "Class 1" areas, the US Environmental Protection Agency (EPA) adopted the Regional Haze Rule in 1999 (<http://www.epa.gov/air/visibility/facts.pdf>), requiring that visibility impairment in Class 1 areas be returned to "natural" levels by 2064. Thus the cumulative effect of anthropogenic influences – including pollution sources, altered fire regimes and fire severity, and other human activities – must be zero by 2064. Total carbon is the largest contributor to  $\text{PM}_{2.5}$  concentrations in the western US and elevated contributions at the upper extremes of aerosol fine mass likely correspond to biomass burning (Ames and Malm, 2001). Because high fine mass concentrations often translate to substantially degraded visibility (Ames and Malm, 2001), implementation of the regional haze rule necessarily includes the future management of wildland fire.

To maintain air quality in Class 1 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 (Veblen et al., 2003; Cook et al., 2004; McKenzie et al., 2004b; Pierce 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., 2004a). 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; Flannigan et al., 1998; Covington, 2000; 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.

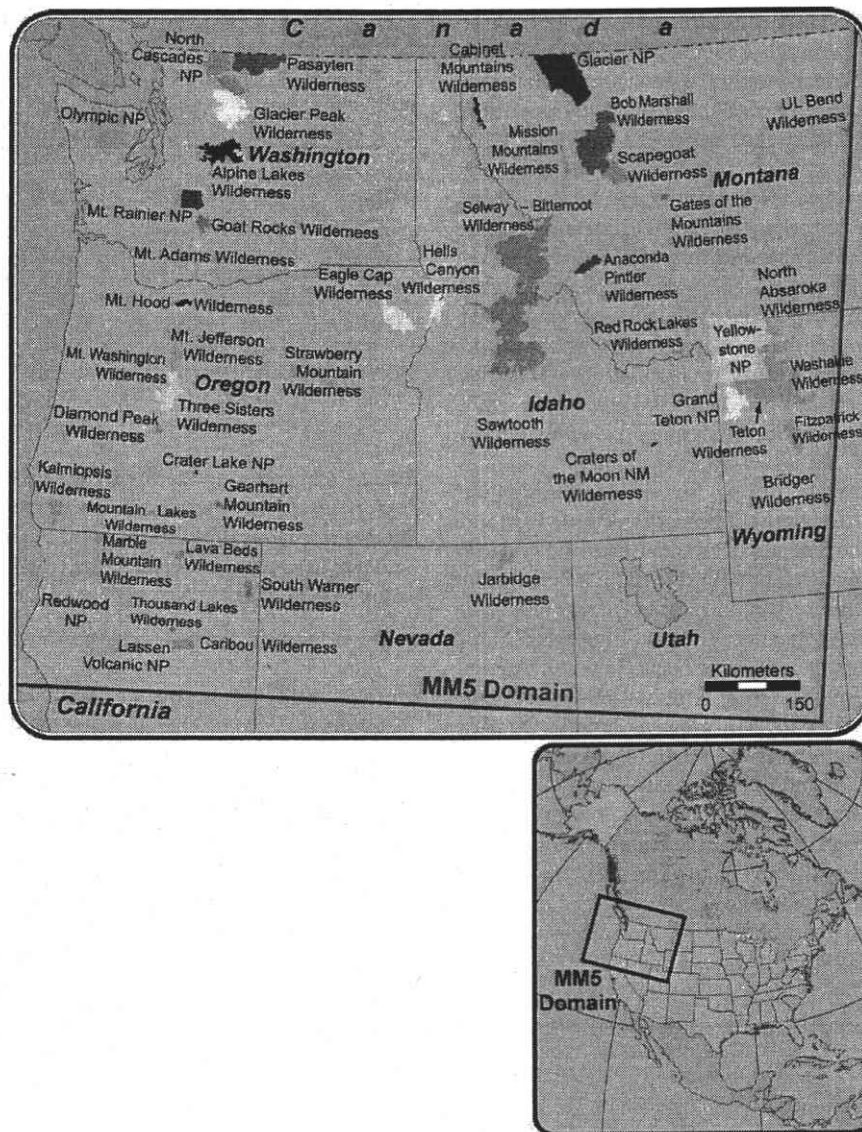
Understanding the multiple causes of regional haze, and the chain of physical and ecological processes from a fire start to the eventual dispersion of smoke emissions, requires an interdisciplinary approach and expertise drawn from climatol-

ogy/meteorology, fire and landscape ecology, ecosystem geography, and fire physics and chemistry. Furthermore, predicting future patterns of regional haze requires a modeling framework that is not driven by empirical observations, because clearly these do not exist for the future. To date, there has been considerable effort, both within the USA and around the world, to estimate patterns of regional haze, but these efforts have been mainly data-driven. For example, the Indonesian fires of 1998 caused unprecedented haze concentrations and were the subject of considerable interdisciplinary research (Hisham-Hashim et al., 1998; Leech et al., 1998; Radojevic and Hassan, 1999). Other sources besides fire are more important in arid regions that have little combustible vegetation (Draxler et al., 2001), or regions heavily affected by anthropogenic pollutants such as sulphur (Engardt, 2001; Engardt and Leong, 2001). The majority of these studies have had the immediate objective of quantifying existing pollution and haze. Other efforts focusing on future projections still generally extrapolate from current conditions, and are essentially data-driven (Norman et al., 2001; Vallack et al., 2001; RMC, 2004).

In this paper, we take a different approach. We simulate the occurrence of the initial source of haze, wildland fires, without calibrating to existing fire records. Rather, we base both the likelihood of fire occurrence and simulated fire size on environmental conditions conducive to wildfire. We have built an integrated system, using new and existing computer models and continental-scale geographic databases. This system not only estimates regional haze under current conditions, but also will allow us to extrapolate to future climate and land use. There are three modules. A climate-fire-vegetation module estimates the effects of climate on wildland fire regimes and vegetation succession, combining a stochastic fire-scenario builder with either historical vegetation and fire regimes (current fire) or a dynamic vegetation and disturbance model (future fire). The fire-scenario builder uses real-time mesoscale meteorology to project wildfire events. A consumption and emissions module calculates particulate and aerosol emissions from biomass consumed in the fires. A smoke dispersion module then simulates the smoke plume and atmospheric dispersion of emissions from each fire. We compare summary statistics for simulations to real data for the same time period, and discuss elements of the modeling framework that need improvement. Finally, we describe enhancements, already under way, to the system that will allow us to project dispersion and concentration of pollutants under future climate, as represented in Global Circulation Model (GCM) scenarios, and apply different scenarios of land use and management, particularly the use of prescribed fire or wildland fuel reduction by mechanical means.

## 2. 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 Fig. 1. In this region, steep gradients in elevation, precipitation, and temperature exist across multiple scales. Climate is a result of the interaction of three air masses: (1) moist marine



**Fig. 1** – Class I wilderness areas in the MM5 12 km modeling domain for the northwestern US. Inset shows the full extent of the domain, which includes parts of southwestern Canada and the northeastern Pacific Ocean. Total area = 16,650 km<sup>2</sup>. Polygons correspond to those on the line graphs in Figs. 5 and 6 are in matched colors.

air from the west, (2) continental air from the east and south, and (3) dry arctic air from the north (Ferguson, 1997). Summer drought, caused by a seasonal northward shift in the jet stream in conjunction with high-pressure over coastal Oregon and Washington, is common, even in areas with high annual precipitation. Summer monsoon conditions sometimes arise in the eastern part of the region from warm air masses moving up from the Southwest (Higgins et al., 1997). 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 (Daubenmire, 1969; Lassoie et al., 1985).

Fire regimes within the study area include large, stand-replacing fires (Agee and Smith, 1984; Henderson et al.,

1989); mixed-severity, medium-frequency fires (Morrison and Swanson, 1990; Taylor and Halpern, 1991); and low-severity, high-frequency fires (Bork, 1985; Kertis, 1986). Severe fires, particularly in moist, high-elevation forests, are usually associated with synoptic weather patterns (Agee, 1993; Ferguson, 1997; Schmoldt et al., 1999; Gedalof et al., 2005). In drier ecosystems on or east of the crest of the Cascade Range in the Interior Columbia River Basin, large fire years are associated with drought, and to a lesser degree with dry phases of the Pacific Decadal Oscillation [PDO] and El Niño Southern Oscillation [ENSO] (Hessl et al., 2004). Lightning is the main source of wildfire ignitions in our study area (Rorig and Ferguson, 1999), whereas in other regions of the USA and in other countries humans are the predominant cause (Guyette and Dey, 2000; Moritz et al., 2003).

Native Americans first settled the inland Pacific Northwest approximately 13,000 years ago (Robbins, 1999), and may have also been an ignition source, burning low elevation ponderosa pine forest and grasslands, prior to the major population and cultural changes of the early 1900's (Robbins and Wolf, 1994; Boyd, 1999; Robbins, 1999; Ross, 1999). In 1878, the Northern Pacific Railroad forbade native burning (Ross, 1999), and in 1908 the US Forest Service began a program of fire suppression (Pyne, 2001), which probably became fully effective around the mid-20th century. Other disturbances – cattle and sheep grazing and selective logging of ponderosa pine forests – increased after the 1880's (Galbraith and Anderson, 1991; Robbins and Wolf, 1994). Despite the variety of human influences, however, climatic variability, both within and among fire seasons, is still the dominant control on fire occurrence and fire extent within the region (Hessl et al., 2004; Gedalof et al., 2005).

Weather in the region during our study period (July 1–August 30, 2003) alternated between periods of high and low stability. During the first part of July, low-pressure systems moved south, triggering light showers across the Pacific Northwest. Although precipitation was widespread, it was relatively light, with heavier rain showers to the east in western Montana. In mid-July an upper level ridge began to build, causing decreased precipitation and rising temperatures. This ridge maintained high temperatures and relative drought through the end of July. In early August the ridge moved east and an area of low-pressure formed off the coast. Moving inland in the second week of August, this low-pressure system caused widespread precipitation. Partially because of this rain, temperatures were significantly cooler than in July. Toward the middle to end of August the Pacific Northwest dried out again as the weakening low-pressure system moved east and was replaced again by a high-pressure ridge.

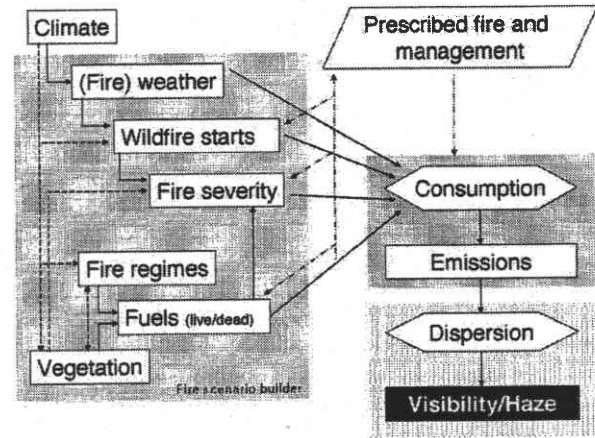
### 3. Methods

The framework of the integrated modeling system is shown in Fig. 2. Multiple dependencies exist among elements. For example, climate affects fire severity directly through its effects on fire weather, but also indirectly through its effects on vegetation and associated abundance and distribution of fuels. Prescribed fire and other management activities primarily affect fuel loadings, thereby affecting consumption and emissions indirectly, but can also modify emissions directly via specific emissions reduction techniques (Hardy et al., 2001).

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 simulate only lightning-caused wildfires.

#### 3.1. Fire Scenario Builder

The Fire Scenario Builder, developed for this work, uses climatic information (historical observations, future climate



**Fig. 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.**

simulations) to determine a scenario of fire starts, sizes, and locations that can be then used by the consumption module. Fire is modeled as a stochastic process, of which each time series of fire events on the landscape is one realization; multiple runs of the FSB therefore will yield different realizations of specific fires on the landscape. Our over-arching criterion for the fire simulations was that each realization should be consistent with all known climatic influences on fire occurrence, rather than a close approximation of real fire events in time and space. Validation of the model is therefore based on comparisons of summary statistics rather than explicit correspondence to empirical data.

The FSB is designed to accept three input layers (Fig. 3), but for this exercise, because it was the first test of the integrated modeling framework for regional haze, 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.

We used a GIS polygon layer for Küchler (1964) potential natural vegetation and resampled it to the 12 km MM5 domain for the northwestern USA. Non-burnable areas (agricultural, urban, barren, and water) were masked out at 1 km resolution, so that each 12 km cell had a percentage (0–100) burnable area. An “expected” area burned per year was calculated based on the mean fire-return interval for each vegetation type as represented in a USA-wide fire-regime database (Olson and McKenzie, 2004).

For fire-weather input, we obtained hourly output for the period of July 1–August 30, 2003 from the University of Washington forecast system (Mass et al., 2003). The MM5 model (Grell et al., 1994), developed by Pennsylvania State University and the National Center for Atmospheric Research (NCAR), Boulder, CO, is a widely used mesoscale atmospheric circulation model. From the MM5 hourly data, we extracted meteorology variables needed for the modeling system: surface

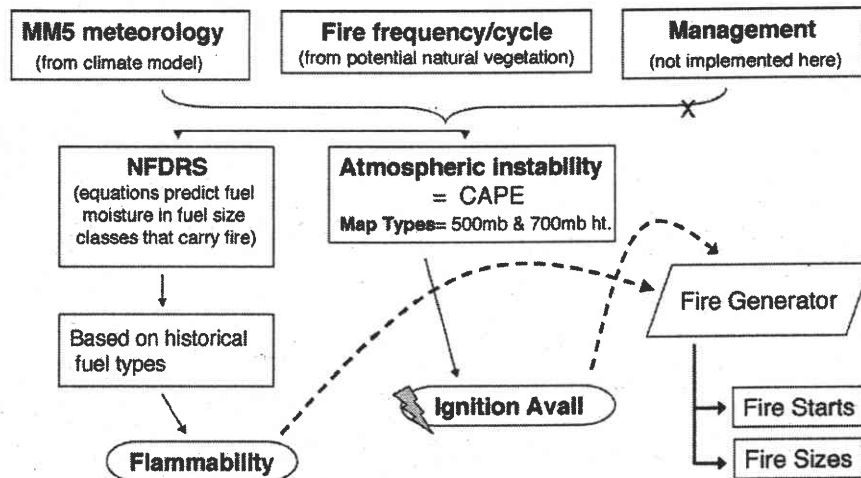


Fig. 3 – Overview of the Fire Scenario Builder. With the management module turned off, fire starts depend on lightning ignitions as a function of atmospheric stability, and fire sizes are controlled by availability of flammable fuels whose moisture content is below a specified threshold. See text for specific values used in the present scenario.

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—Petersen et al., 1996) statistic for each day at each 12 km grid cell. Maximum CAPE has positive correlation with lightning occurrence (Petersen et al., 1996).

Lightning was simulated when  $\max(\text{CAPE}) > 1000$ , creating 4–5 episodes of sufficient lightning potential during the fire season, similar to what is observed. Use of CAPE for lightning potential was an initial approximation, with the understanding that better proxies (e.g., using combinations of cloud temperature, available moisture, and vertical velocity) may exist (e.g., Solomon and Baker, 1998).

The potential for lightning to trigger a fire was estimated with the National Fire Danger Rating System (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 (Anderson, 1982; Cohen and Deeming, 1985). A normalized cumulative distribution of CAPE values was built for each cell, with all values  $< 1000$  set to zero. A uniform random number on  $[0, 1]$  identified an “ignition day” for each cell, based on where it fell in the normalized cumulative distribution function (CDF) of CAPE values. A fire was “ignited” on the ignition day for a cell 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. Fuel moisture in the 0.6–8.0 cm size class was used to define the quantile of a negative exponential distribution with the “expected” area burned for each cell as a mean. The quantile was defined by a reverse linear interpolation (the drier the higher) between zero moisture and the maximum moisture for the season (ca. 71%). For quantiles between 0 and 95%, the cell’s “fire size” was adjusted to the associated quantile. For quantiles above 95% (lowest 5% possible fuel moisture), we considered “fire

weather” to be extreme, and modeled fire sizes with an excess function (Reiss, 2001) using a truncated Pareto distribution, following Alvarado et al. (1998). Alvarado et al. (1998) found that the truncated Pareto provided the best fit to large-fire data. The cumulative distribution function (cdf) is:

$$F_x(x) = P(X \geq x) = \left[1 - \left(\frac{k}{k'}\right)\right]^{-1} \left[\left(\frac{k}{x}\right)^a - \left(\frac{k}{k'}\right)^a\right],$$

for  $k < x < k'$ , 0 for  $x \leq k$ , 1 for  $x \geq k'$  (1)

where  $k$  is a specified lower bound,  $k'$  the specified upper bound, and  $a$  is a parameter to be estimated. Alvarado et al. (1998) found parameter  $a$  to be very close to 0.5 (0.48 and 0.52 for two different datasets).

By choosing  $a=0.5$ , Eq. (1) can be easily solved for  $x$ . Rearranging and noting that the quantile of interest  $Q_x = 1 - P(X \geq x)$ , we have

$$x = \frac{k}{((1 - Q_x)[1 - (k/k')^{1/2}] + (k/k')^{1/2})^2} \quad (2)$$

where  $x$  is in the same units as  $k$  and  $k'$ .  $k$  was set to the area associated with the 95% quantile of the negative exponential distribution and  $k'$  set to the maximum burnable area within each cell ( $\leq 144 \text{ km}^2$ , depending on the proportion of the cell available—see above).

Fires produced in this way were between miniscule and 6500 ha, with the majority being under 40 ha. Fires under 40 ha were then eliminated. This size is the de minimis threshold for fire-tracking procedures specified by the Regional Haze Rule. 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. We used the average duration for wildfires reported to the USA National Interagency Fire Center (NIFC) in 2003 under 6500 ha as the maximum (ca. 8 days), and interpolated the



potential duration of our simulated fires accordingly, i.e., in eight bins spanning the range 40–6500 ha. 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%.

### 3.2. 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 across the domain were obtained from a 1 km GIS layer developed by the first author using the Fuel Characteristic Classification System (FCCS—Sandberg et al., 2001, GIS layer posted at <http://faculty.washington.edu/dmck/feradata/FCCS-lower48.zip>). 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<sub>2.5</sub> & PM<sub>10</sub>, CO<sub>2</sub>, CO, volatilized organic carbon (VOC), non-methane hydrocarbons (NMHC)) from fires based on an exponential mixture model of flaming and smoldering stages of combustion.

### 3.3. 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.

### 3.4. Data output

We ran the simulations through a 61-day period in the summer of 2003, producing PM<sub>2.5</sub> concentrations across the MM5 Pacific Northwest domain. In this paper we focus on PM<sub>2.5</sub> concentrations in selected Class I Wilderness Areas within the domain (Fig. 1). We recorded the maximum of 24 h running means of PM<sub>2.5</sub> over all 12 km cells included in Class I area. We then calculated an extinction coefficient to represent the worst-case reduction in visibility from pristine conditions associated with the 24 h concentrations of PM<sub>2.5</sub> from fire only.

McMeeking et al. (2005) and Engling et al. (2004) found, in a study of aerosols in Yosemite National Park, USA, that PM<sub>2.5</sub> from fire was 80% organic carbon (OC). Assuming that this finding is applicable to fire across the Western US, and assuming that the ratio of OC to elemental carbon (EC) from fire is 9:1 (Engling et al., 2004; McMeeking et al., 2005), and neglect-

ing sulfate, nitrate and fine soil, the extinction coefficient from fire only is:

$$\beta_{\text{ext}} = 4[\text{OC}] + 10[\text{EC}] = (4)(0.8)[\text{PM}_{2.5}] + \frac{(10)(0.8)[\text{PM}_{2.5}]}{9} \quad (3)$$

where [] indicates concentration (μg m<sup>-3</sup>), and β<sub>ext</sub> is in units of M m<sup>-1</sup> (US Environmental Protection Agency, 1999).

Pitchford and Bachman (personal communication) suggest the following thresholds to translate the extinction coefficient into a qualitative index of visibility:

- β<sub>ext</sub> < 20—Natural Conditions
- β<sub>ext</sub> > 20—Noticeable Impairment
- β<sub>ext</sub> > 41—Moderate Degradation
- β<sub>ext</sub> > 70—Severe Degradation

These are heuristic values only, based on an extensive examination of photographs, and were not used for any computations.

We used the WinHaze Visual Air Quality Modeler (Air Resource Specialists, 2004) to visualize the visibility reduction from modeled PM<sub>2.5</sub> 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.

## 4. Results

Fig. 4 shows the area burned per day and number of fires per day simulated by the FSB for the PNW domain between July 1 and August 30, 2003. 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. In late August, total fire

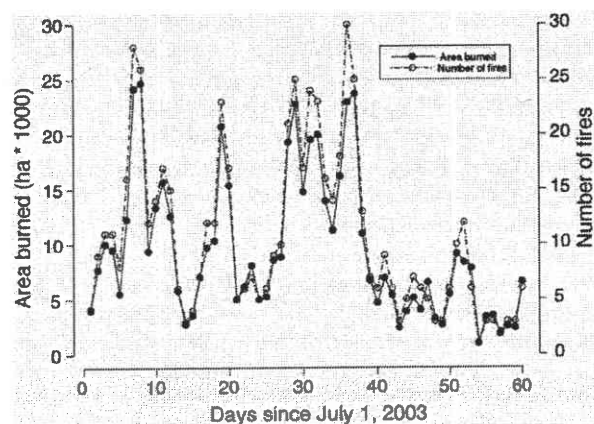


Fig. 4 – Total hectares burned per day and number of fires per day for the domain, during the period July 1 (Day 182 of the year) to August 29 (Day 242).

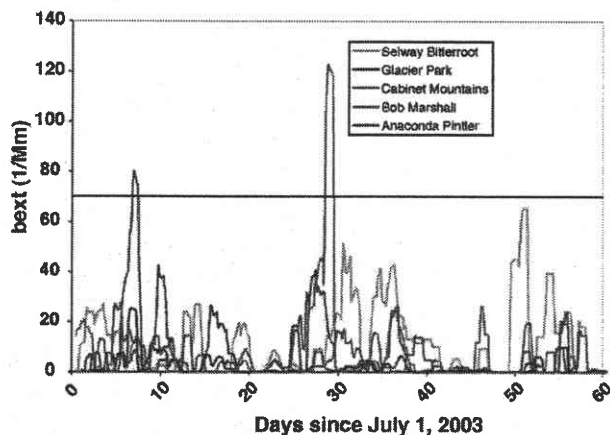


Fig. 5 – Twenty-four hours running means of maximum extinction coefficients predicted for five Class I areas in Idaho and Montana (see Fig. 1 for locations—polygon colors match line graphs). Predictions for the Selway–Bitterroot are for the northern half only (see text).

activity was greatly reduced, reflecting widespread precipitation across much of the domain (see Section 2).

Using these simulated fires, consumption and dispersion were calculated to yield smoke concentrations throughout the domain. From 24 h mean concentrations of particulate matter less than  $2.5 \mu\text{m}$  ( $\text{PM}_{2.5}$ ), light extinction coefficients ( $\beta_{\text{ext}}$ ) were computed from Eq. (3) at each Class 1 area in the domain (Figs. 5 and 6). Fig. 5 shows  $\beta_{\text{ext}}$  for selected Class 1 areas in northern Idaho and western Montana; Fig. 6 shows  $\beta_{\text{ext}}$  for selected areas in Washington and Oregon. Low  $\beta_{\text{ext}}$  values (clear conditions) exist for every location at some point, but lower  $\beta_{\text{ext}}$  values are typical for areas further to the west, particularly along the crest of the Cascade Mountains in Washington and Oregon (Fig. 6). Higher  $\beta_{\text{ext}}$  values (more visibility degradation) are typically found to the east, produced by the

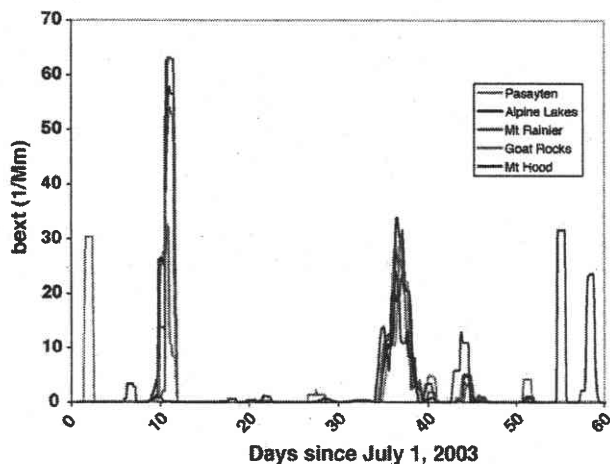


Fig. 6 – Twenty-four hours running means of maximum extinction coefficients predicted for five Class I areas along the crest of the Cascade Range, Washington and Oregon (see Fig. 1 for locations—polygon colors match line graphs).

Table 1 – Light extinction coefficient values in western US national parks

National park	20 worst days average ( $\beta_{\text{ext}}$ in $\text{M m}^{-1}$ )	Maximum value ( $\beta_{\text{ext}}$ in $\text{M m}^{-1}$ )
Canyonlands	35.5	98.0
Crater Lake	41.8 (9.3)	391.0 (13.6)
Glacier	63.9 (64.3)	224.0 (80.4)
Grand Canyon	33.7	195.0
Great Basin	31.9	98.0
Mesa Verde	33.6	98.0
Mount Rainier	75.9 (49.6)	391.0 (63.3)
Point Reyes	76.2	196.0
Rocky Mountain	37.4	196.0
Yellowstone	38.1	196.0
Yosemite	55.5	245.0

Mean values for the 20 worst days during 1988–1998, and maximum values during the same period. Values in parentheses are modeled mean values of  $\beta_{\text{ext}}$  for the 2% worst data points and modeled maximum  $\beta_{\text{ext}}$  values (based on data in Figs. 5 and 6). From IMPROVE site data (<http://vista.cira.colostate.edu/improve/>).

typical west-to-east transport of smoke by prevailing winds. The maximum  $\beta_{\text{ext}}$  is  $123 \text{ M m}^{-1}$  in the Bob Marshall Wilderness in late July (Fig. 5).

Peaks in the extinction coefficient roughly correspond to total area burned per day across the domain. We use data from the IMPROVE visibility monitoring program (<http://vista.cira.colostate.edu/improve/>) to compare the simulated light extinction values with observations. Table 1 presents the average  $\beta_{\text{ext}}$  from the 20 worst days during 1988–1998 and absolute maximum  $\beta_{\text{ext}}$  over the same period calculated from station observations taken at national parks under the IMPROVE program. Three of the national parks listed in Table 1 fall within the model domain: Crater Lake, Glacier, and Mount Rainier. Data in parentheses are modeled mean values of  $\beta_{\text{ext}}$  for the 2% worst data points and modeled maximum  $\beta_{\text{ext}}$  values. If the IMPROVE station is functional for the full 10 year period, then 20 days comprise approximately 2% of the total number of measurements. The maximum simulated value ( $123 \text{ M m}^{-1}$  in the Bob Marshall Wilderness) exceeds the 20 worst days average of any western national parks, but is below the maximum observed values for many eastern national parks.

The second highest  $\beta_{\text{ext}}$  values in the simulation occurred in Glacier National Park (NP), which also has an IMPROVE monitoring station. From March 1988 to December 2003, IMPROVE data show 55 days where  $\beta_{\text{ext}}$  was  $>70 \text{ M m}^{-1}$  (indicating severe visibility degradation), with a peak extinction coefficient of  $224 \text{ M m}^{-1}$  occurring in September 2001. Using daily IMPROVE data from the period July 2–20, 2003 we find that the average observed  $\beta_{\text{ext}}$  was  $16 \text{ M m}^{-1}$  compared to our simulated average  $\beta_{\text{ext}}$  of  $8.5 \text{ M m}^{-1}$ . Both of these values are based on light extinction due to organic and elemental carbon only. The maximum simulated  $\beta_{\text{ext}}$  of  $80.4 \text{ M m}^{-1}$  exceeded the 20 worst days average, but is well within the maximum value and the average of the top 2% data ( $64.3 \text{ M m}^{-1}$ ) is very similar to the IMPROVE 20 worst day average of  $63.9 \text{ M m}^{-1}$ . During this period, fire is likely to have been the principal source of  $\text{PM}_{2.5}$ —observed organic carbon/elemental carbon ratios between 4.5 and 9.3 indicate a substantial fire input

(McMeeking et al., 2005). These results indicate that we successfully predicted visibility reduction contributing to the 20 worst days, but did not simulate as extreme an event as the 10-year peak, which likely included sources of haze other than fire. Observed organic carbon to elemental carbon ratios ranging from 4.9 to 14.3 at Crater Lake also indicate a substantial fire input, but simulated results indicate we underestimated the average of the 20 worst days. Mount Rainier is located downwind of the city of Seattle, a large metropolitan area, therefore because we excluded sources of  $PM_{2.5}$  other than fire, simulated values would be expected to be lower than observed maximum values.

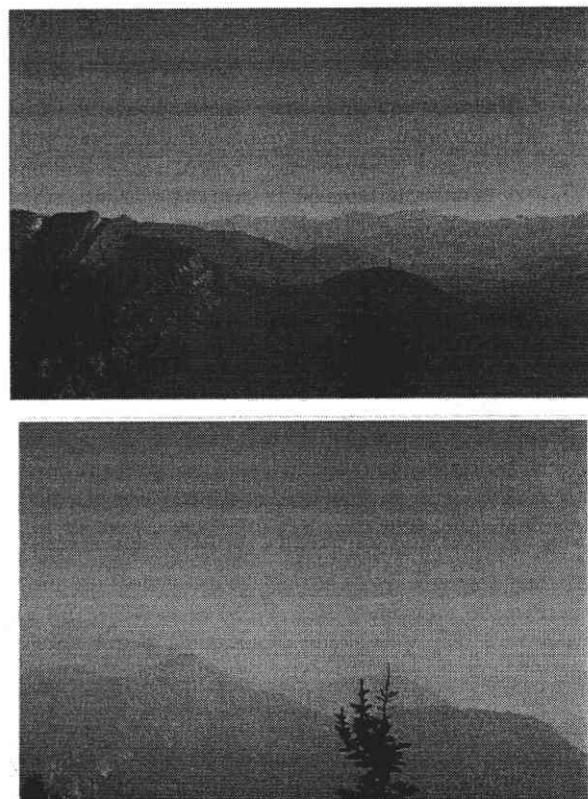
## 5. Discussion

We integrated four simulation models (MM5, FSB, EPM, and CALPUFF) with GIS raster layers, threshold calculations for fire ignition (based on CAPE), and fire-extent calculations (based on NFDRS). With this integrated system, we simulated the contribution of wildfire to fine particulates ( $PM_{2.5}$ ) that cause visibility reduction (regional haze) in Class 1 areas of the Pacific Northwest, USA, under historical (natural background) fire regimes, but current fuel conditions.

The FSB, in its current configuration, overestimated the number of fire starts while slightly overestimating the total area burned. The FSB simulated a total of 340 fires larger than 40 ha in the MM5 domain for the period July 1–September 1, 2003, but only approximately half this number were reported by the National Interagency Fire Center (NIFC) for the same region and period (NIFC 209 report summaries). It is known, however, that many of the smaller fires, even over 40 ha, go unreported, so this discrepancy is less an issue than it might appear.

The total area burned simulated by the FSB was about 108% of that listed on the NIFC reports (based on a proportional representation of state totals by the area of each state included in the domain: WA, OR, ID = 100%, MT = 65.7%, UT = 45.0%, NV = 44.4%, WY = 31.9%, CA = 20.8%). The largest fire simulated by the FSB was just over 6500 ha, whereas the largest actual fire during that period was 36,000 ha. Although our simulated area burned per day is consistent with the daily area burned in real fires, our longest fire burned 8 days, whereas the 2003 Little Salmon Wildfire burned for 33 days. Even with an extreme value distribution implemented for large fires, the FSB “smoothed” the peaks, while closely approximating seasonal total area burned.

To make the FSB consistent with our mechanistic understanding of fire extinctions, fires “went out” above a moisture threshold. This is a simplification of reality, because many of the largest recorded fires have gone through dormant phases only to erupt again under favorable weather. These dormant phases are unpredictable, however, so the smoothing provides a surrogate robust method for reproducing the seasonal totals, which are of interest to decision-makers. The modeling system produced light extinction coefficient values at Class 1 areas within those observed historically at western US national parks. The simulated days of maximum reduction in visibility, in late July in the Bob Marshall Wilderness and in early July in Glacier National Park, are analogous to



**Fig. 7 – Views of a scenic vista in the Bob Marshall Wilderness in northwestern Montana under pristine conditions ( $\beta_{ext} = 10 M m^{-1}$ ) and severely degraded conditions ( $\beta_{ext} = 123 M m^{-1}$ ). See text for details.**

observed visibility impairment from wildland fire. Fig. 7 shows a visual comparison of pristine versus maximum degradation for the maximum simulated degradation at the Bob Marshall Wilderness in northwestern Montana. We expect that wildfires upwind of Class 1 areas will consistently reduce visibility, if not to record levels of degradation, at least to levels associated with worst-case days by regulators and with unacceptable loss of scenic vistas by the public.

To ensure the most accurate estimates possible of emissions, the baseline GIS fuels layer needs to be dynamic, i.e., be rebuilt at regular intervals as land cover changes across the domain. We also need to be able to incorporate changes in vegetation cover (and therefore fuels) over time, for example, using MODIS quantitative vegetation layers (<http://edcdaac.usgs.gov/modis/mod44b.asp>) for empirical estimates or ecosystem models (Keane et al., 1996; Bachelet et al., 2000) for dynamic estimates. Once this is implemented, management options in the FSB can then be applied to modify the fuels layer, thereby changing potential and actual biomass consumed and smoke produced from the emissions simulator.

No published evaluation of the EPM model currently exists, but it was originally designed for application in “activity” fuels—those specifically created by logging operations (Sandberg and Peterson, 1984). Work is underway to integrate a refined set of algorithms for the effects of fuel moisture on combustion, canopy fuel consumption, and plume rise (this



last an input to smoke dispersion models) into our modeling framework (Sandberg et al., 2004). EPM is known to underestimate emissions in crown fires; a more accurate computation will enable our system to simulate visibility reduction commensurate with peak observations.

Over-calibration of any simulation model at an early stage of development is unwise (Schmoldt et al., 1999; McKenzie et al., 2004b); these modifications should precede any “tuning” to more closely correspond to real data. Ensemble runs, though computationally expensive and beyond the scope of this initial effort, should also be made before any substantial calibration. Because error propagation can be difficult to control when models are combined or extrapolated to new conditions (Rastetter et al., 1992; McKenzie et al., 1996), sensitivity analysis, both of individual modules and the whole system, will need to be ongoing.

How will wildfire affect visibility in the future? Both empirical models (McKenzie et al., 2004a) and process-based models (Lenihan et al., 1998) suggest that wildfire area will increase in the western USA with a warming climate. Clearly we can 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 1 areas, provided that appropriate meteorological time series are available. A collaborative effort is under way to simulate mesoscale meteorology across the domain for the mid-21st century, using MMS, and link output to burned-area estimates and the FSB.

These future “mean-field” estimates of potential area burned can come from process-based models (Keane et al., 1996; Lenihan et al., 1998; Bachelet et al., 2000), statistically based models (Flannigan et al., 1998; McKenzie et al., 2004a; Gedalof et al., 2005), or both. For example, Lenihan et al. (1998) produced monthly estimates of area burned and biomass consumed within each landscape unit by applying fire danger and fire behavior calculations and downscaled future climate to simulated future vegetation. In contrast, McKenzie et al. (2004a) used statistical relationships between known climate and seasonal area burned to extrapolate to predicted future climate. Both methods could provide baseline area estimates to use as input to the FSB.

## 6. Conclusions

Given the expected 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. We present here the first output from a simulation system that incorporates climatology and meteorology, ecosystem geography, fire physics and fire ecology, and atmospheric chemistry, to predict temporal and spatial patterns of visibility impairment from wildland fire. Results for the northwestern USA both show promise and identify shortcomings while suggesting ways to improve the system. We expect to improve the scientific understanding and data quality associated with each module, combining deterministic and stochastic elements that produce results consistent with known physical principles. By doing so, we will maintain the robustness of the system to extrapolation

beyond current conditions and optimize its capacity to accept future enhancements.

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