THESIS

THE IMPACT OF PINE BEETLE INFESTATION ON MONOTERPENE EMISSIONS AND

SECONDARY ORGANIC AEROSOL FORMATION IN WESTERN NORTH AMERICA

Submitted by

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ABSTRACT

THE IMPACT OF PINE BEETLE INFESTATION ON MONOTERPENE EMISSIONS AND

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Over the last decade, an extensive beetle outbreak has impacted western North America

resulting in the mortality of over 100,000 km^2 of forest throughout British Columbia and the

western United States. Climate change has aided the expansion and continuation of this beetle

infestation for more than a decade as beetles survive milder winters and expand northward and to

higher elevation areas. Studies have been conducted to investigate the impact of this disturbance

on forest carbon stocks, beetle-fire interactions, and meteorological variables, as well as to affirm

the importance of including beetle infestation in models. In recent years there has been increased

interest in the impact of beetle mortality and attack on atmospheric composition. Numerous

studies have demonstrated that insect attack can prompt elevated emissions of volatile organic

compounds (VOCs) in a variety of plant and tree species, including mountain pine beetle

attacking lodgepole pine, the main beetle-host combination in the current outbreak. These

enhanced VOC emissions are likely a defense mechanism of the tree, consisting of increasing

emissions of compounds that are toxic to the beetles and attract predators of the beetles as well

as increasing sap flow to help remove beetles from the trunk. This impact has not yet been

modeled; however, beetle attack may have a significant impact on atmospheric composition and

air quality in western North America.

In this study, we use 14 years of beetle mortality data for 13 beetle species and beetle-

induced monoterpene concentration data in the NCAR Community Earth System Model (CESM)

to investigate the impact of beetle mortality and attack on monoterpene emissions and secondary

organic aerosol (SOA) formation in western North America. Needleleaf vegetation is decreased each year based on the annual mortality data while emissions of certain compounds in needleleaf trees under attack are scaled-up based on recent beetle-induced VOC data for lodgepole pine (pine scenario) and Engelmann spruce (spruce scenario). As the mountain pine beetle has had the most extensive impact on mortality, we compare changes in emissions of VOCs and subsequent SOA formation caused by the mountain pine beetle to changes caused by the other 12 beetles combined.

Beetle infestation impacts monoterpene emissions through both decreased emissions as trees are killed off (mortality effect) and increased emissions in trees under attack (attack effect). Regionally, beetle infestation may have a significant impact on monoterpene emissions and SOA concentrations with up to a 4-fold increase in monoterpene emissions and up to a 40% increase in SOA concentrations in some years. Responses to beetle attack can vary greatly over space and time as the areas affected as well as the magnitude of the impact depend on the extent of previous mortality and the number of trees under attack in a year. The model captures highly localized impacts on smaller-scales, while on larger-scales, the cumulative mortality effect often mutes the ongoing attack effect. The mountain pine beetle alone has an impact similar to that of the other 12 beetles combined, and the spruce scenario has an impact 3-4 times greater than the pine scenario due to differences in the magnitude of the observed enhancement in monoterpene emissions. In North America, the pine scenario would likely dominate since lodgepole pine is the main species impacted; however, smaller regions of spruce may see higher localized impacts on monoterpene emissions and SOA concentrations. Placed in the context of OM and PM2.5 IMPROVE network measurements, the changes in SOA concentrations due to beetle attack are in most cases small compared to the large annual and interannual variability in the measurements

of total organic aerosol, indicating that most beetle-induced SOA changes are not likely

detectable in current observation networks. However, in areas with especially large emissions

enhancements (e.g. areas of spruce under attack) and lower variability in measurements of OM,

beetle-induced changes in SOA may be observable. Due to the large potential impacts that

beetle infestation may have on monoterpene emissions, SOA formation, and degradation of air

quality, it is important that beetle infestation be included in future models.

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1 Introduction

In the last decade, western North America has experienced the worst beetle outbreak in recorded history (Taylor et al., 2006). The main beetle impacting the region is the mountain pine beetle (MPB) at epidemic levels with nearly 74% of needleleaf tree mortality due to insects and diseases caused by this beetle (Man 2012). The MPB is native to western North America. It kills large numbers of pine trees annually, attacking mainly lodgepole pine. Ponderosa and whitebark pine are also susceptible to attack (Logan and Powell, 2001). These are bark boring beetles, infesting the trunk and eventually killing the tree as the beetles consume the phloem. The MPB also introduces a fungus into the tree, further weakening its defenses. In the United States, MPB infestation peaked in 2009 with 8.8 million acres infested, with 6.8 million acres infested as of 2010 (Man, 2012). In British Columbia, the area of infestation peaked in 2007 with almost 25 million acres infested (Westfall and Ebata, 2011). Although the area infested in both British Columbia and the western United States has decreased recently, the overall severity in both regions is increasing and beetles continue to expand into new areas (Westfall and Ebata, 2011). Figure 1.1 shows the visible impact of the mountain pine beetle on forest mortality. Other beetle species have also contributed to forest mortality over the last decade, albeit on a smaller scale.

Expansion of the MPB into new areas has been limited in the past only by climate (Carroll et al., 2003). Climate change is currently allowing the MPB to expand its range northward and to higher elevation areas that were previously too cold to support the MPB (Westfall and Ebata, 2011). Several studies have attempted to model the future impacts and timing of the expansion of the MPB based on climate change effects and changing pine stand

Figure 1.1: Lodgepole pine trees killed by mountain pine beetle attack (photo by Jeffrey A. Hicke, University of Idaho).

characteristics (e.g. Taylor and Carroll, 2004; Hicke et al., 2006; Bentz et al., 2010). The studies agree that progressively higher elevations will support MPB outbreaks over the next century, possibly maintaining the intensity of the MPB outbreak for decades. Climate change may also lead to drought and increased surface ozone concentrations, both of which can stress trees and make them more susceptible to beetle attack (Percy, 2003). Though most of these studies focus on the MPB, other beetles may be impacted in similar ways by climate change.

The need to include land-cover disturbances such as insect infestation in models has been discussed in several studies (e.g. Running, 2008) due to the many potential impacts and feedbacks that these disturbances can cause. The terrestrial biosphere can serve as a carbon sink, absorbing a portion of the carbon dioxide emitted by fossil-fuel burning (Schimel et al., 2001). Insect damage may impact forests by shifting them from a carbon sink to a carbon source. A few recent studies have also indicated that northern forests may not be as strong a sink as previously thought (Stephens et al., 2007; Yang et al., 2007; Ito et al., 2008) so increased carbon release by disturbances such as beetles may more easily shift these forests from a sink to a source. Models

may also overestimate the carbon uptake potential of forests, and so impacts from insect damage should be included (Kurz et al., 2008b). Models have already shown examples of forests becoming sources due to insect outbreaks, with continued effects for decades (e.g. Kurz et al., 2008a). Bark beetle infestation alters carbon stocks differently than stand-replacement wildfires or clear-cut harvesting because smaller diameter trees and non-host trees will survive a beetle infestation, leading to a different pattern of mortality (Pfeifer et al., 2010), further indicating the need to model this disturbance.

Beetle infestation can also impact fire susceptibility and activity. In the years immediately following an outbreak, there is an increased risk of active crown fire due to increased dead fuels in the canopy (Page and Jenkins, 2007; Simard et al., 2011). About 3-5 years post-outbreak, needles from dead trees begin to fall, which increases fine surface fuels and fuel bed depth (Page and Jenkins, 2007; Simard et al., 2011). Litter accumulation for dead pine stands can be up to two-fold that of live pine stands (Bigler and Veblen, 2010). This increases the intensity and rate of spread of surface fires (Page and Jenkins, 2007; Jenkins et al., 2008; Jorgenson and Jenkins, 2010; and Simard et al., 2011) although the thinned canopy decreases the risk of crown fire (Simard et al., 2011). Several decades following outbreak, dead woody fuels increase as trees fall and re-growth occurs, increasing live fuels in the understory (Page and Jenkins, 2007). This creates ladder fuels which increase the risk of passive crown fire (Romme et al., 2006). Although beetles alter the complex fuel characteristics, climate and weather conditions are still the most important factors in determining whether a fire will occur (Klutsch et al., 2010; Kulakowski and Jarvis, 2011). Another important beetle-fire impact is that Douglas-fir beetles have been found to be more likely to attack moderately to severely fire-injured trees, so fire may increase the severity of outbreaks of these beetles (e.g. Hood and Bentz, 2007).

Insect attack can also alter surface-atmosphere exchanges of heat, water, and momentum through land surface modification. Wiedinmyer et al. (2012) modeled changes in meteorological and flux variables in Colorado and the western US due to large-scale forest mortality caused by insect infestation. They find an average increase in surface temperature of 2.1 K, as well as an increase in sensible heat flux, a decrease in latent heat flux, and an increase in planetary boundary layer (PBL) height. Because PBL depth and temperature affect the concentrations of gases and particles in the atmosphere, they also conclude that there may be an impact on atmospheric composition and that mortality due to beetle infestation should be included in atmospheric chemistry models. Hais and Kucera (2008) estimate a 3.5 K increase in temperature in a spruce forest after beetle attack. Snowpack in a beetle-killed forest can also be prolonged (Boon, 2007).

Pine beetle attack can prompt elevated monoterpene emissions in trees (e.g. Amin et al., 2012), with potential implications for local air quality. The emissions enhancement is likely a defense mechanism of the tree which consists of increasing resin flow to remove beetles, increasing emissions of compounds which are toxic to the beetles, and attracting predators of the beetles (Pare and Tumlinson, 1999). Monoterpenes are a class of biogenic volatile organic compound (BVOC). BVOC emissions are up to an order of magnitude higher than anthropogenic VOC emissions in many regions including western North America (Hallquist et al., 2009). Went (1960) first recognized the role of terpenes in the formation of aerosol through observations of blue haze forming over pine needles in the presence of ozone. Monoterpenes are oxidized in the atmosphere by OH, ozone, or other oxidants to form lower volatility products which can partition into the particle phase, forming secondary organic aerosol (SOA). These oxidation processes can also lead to ozone formation with important implications for global

atmospheric composition. Monoterpene lifetimes against reaction with oxidants are on the order of hours, so SOA can often form directly downwind of precursor emissions, especially in regions with high precursor emissions (e.g. the Amazon). SOA formed from monoterpenes and other compounds can impact radiative forcing through the direct effect of scattering incoming solar radiation and the indirect effect on cloud albedo and lifetime (Lohmann and Feichter, 2005). Visibility may also be degraded by SOA formation in forests. National parks and other protected wilderness areas are currently impacted by beetle infestation and these are areas where the EPA has mandated an improvement in visibility under the Regional Haze Rule (EPA, 1999).

SOA may also have potential health impacts. Most studies focus on the health impacts of total fine particulate matter (PM2.5) to which SOA contributes (e.g. Samet et al., 2000 and Laden et al., 2006). These studies find associations between PM2.5 levels and cardiovascular disease, respiratory illnesses, and premature mortality. One epidemiological study finds a strong association between summer PM2.5 concentrations and daily mortality (Nawrot et al., 2007). Because production of SOA can contribute significantly to PM2.5 during summer, SOA may be contributing to higher mortality, making increases in SOA a health concern. However, few studies have looked at the specific health impacts of SOA. One recent study finds that ambient SOA concentrations may induce negative effects in lung cells (Baltensperger et al., 2008) and another finds that SOA from α-pinene, limonene, and isoprene causes airway irritation and respiratory reduction in mice (Wolkoff et al., 2000) and conclude that similar effects could occur in humans.

A few studies have examined and quantified monoterpene concentrations and emissions from specific tree species including lodgepole pine, ponderosa pine, and Engelmann spruce and how beetle attack changes these emissions. These studies include both beetle infestation studies

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and fungal inoculation studies. Fungal inoculation alone has been found to create a large increase in monoterpene concentration and emission (e.g. Raffa and Smalley, 1995). Since all beetles in nature carry fungus, the fungal inoculation that occurs with beetle infestation may play a large role in the tree's response. This indicates that studies examining tree responses to fungal inoculation without beetle infestation can be a good approximation for beetle attack in nature. Gara et al. (1993) measure an increase in total monoterpene emissions from 6.8 µg m^{-2} day^{-1} to 32.8 µg m^{-2} day^{-1} after fungal inoculation of lodgepole pine. Litvak and Monson (1998) find that simulated and real herbivory significantly increase monoterpene emission rates in ponderosa pine, lodgepole pine, and white fir trees. This impact may be limited, or more pronounced for specific monoterpenes. Jost et al. (2008) measure large relative increases in sabinene, α-phellandrene, and β-phellandrene in lodgepole pine emissions of 93%, 15%, and 80%, respectively, with β-phellandrene becoming the most abundant monoterpene. Amin et al. (2012) find statistically significant increases of β-pinene, 3-carene, β-phellandrene, methyl chavicol, and p-cymene from lodgepole pine ranging from 5-fold to almost 33-fold, with β-phellandrene again experiencing the largest relative increase. In addition to increases in emissions of these compounds, the amount of monoterpenes stored inside the tree increases significantly. Miller et al. (1986) estimate an almost 3-fold increase in monoterpene concentration in lodgepole pine phloem infected with the MPB. These increases in monoterpene concentration and emission are not limited to bark-boring beetles and needleleaf trees. Other studies have seen similar monoterpene emission enhancements caused by a variety of insect species in a variety of plant types. For example, Prieme et al. (1999) find that needle-eating weevils, which consume the foliage of the tree rather than the phloem in the trunk, can also cause up to a 58x increase in emissions of monoterpenes as compared to an uninfested tree. Other insects and hosts which

display an increase in emissions include aspen infested with weevils and moths (Blande et al., 2007), poplar infested with leaf beetles (Brilli et al., 2009) and holm oak infested with gypsy moth larvae (Staudt and Lhoutellier, 2007).   Insect herbivory can therefore induce both substantial increases in total monoterpene emissions from vegetation and changes in the emission profile, with implications for atmospheric composition.

2 Motivation and Objectives

Although the effect of beetle kill on carbon cycling in forests has been studied and modeled previously, the impact of beetle kill on air quality has not yet been investigated. Arneth and Niinemets (2010) suggest that insects and their impacts on BVOC emissions may be significant and should be included in dynamic vegetation models. Recently, a few studies have quantitatively studied the effect of beetle attack on emissions of specific VOCs (e.g. Amin et al., 2011; Duhl et al., 2012) and so this effect can now be estimated.

In this work, we use beetle mortality data from 1997-2010 and beetle-induced monoterpene data from the recent literature in the NCAR Community Earth System Model (CESM) to study the impact of beetle infestation on monoterpene emissions and SOA formation in western North America. We compare two scenarios based on beetle-induced monoterpene data from lodgepole pine and Engelmann spruce and focus on the spatial and temporal evolution of emissions of the six monoterpenes 3-carene, β-pinene, β-phellandrene, p-cymene, α-pinene, and sabinene and SOA formed from these compounds due to cumulative beetle attack and subsequent forest mortality in the model.

3 Model Description

The NCAR Community Earth System Model (CESM) consists of coupled global models for the atmosphere, ocean, land, land ice, and sea ice (Gent et al., 2011). In this work we use a CESM configuration where the land and atmosphere are coupled with fixed ocean (SST) and sea ice conditions for present day.

The land model component used in this study is version 4 of the Community Land Model (CLM4, Lawrence et al., 2011). The CLM4 describes the physical, chemical, and biological processes of terrestrial ecosystems, including the hydrology and carbon cycling of the terrestrial biosphere. Land cover types include glacier, lake, wetland, urban, and vegetation. Vegetation is specified by 16 plant functional types (PFTs) (Table 3.1). Leaf Area Index (LAI) is also specified for each month for each PFT. The PFT distributions are based on Moderate Resolution Imaging Spectroradiometer (MODIS) land surface data sets (Lawrence and Chase, 2007) and a new cropping dataset (Ramankutty et al., 2008). Figure 3.1 shows coverage of needleleaf forests (PFT 1 and PFT 2), the vegetation types susceptible to beetle attack over western North America.
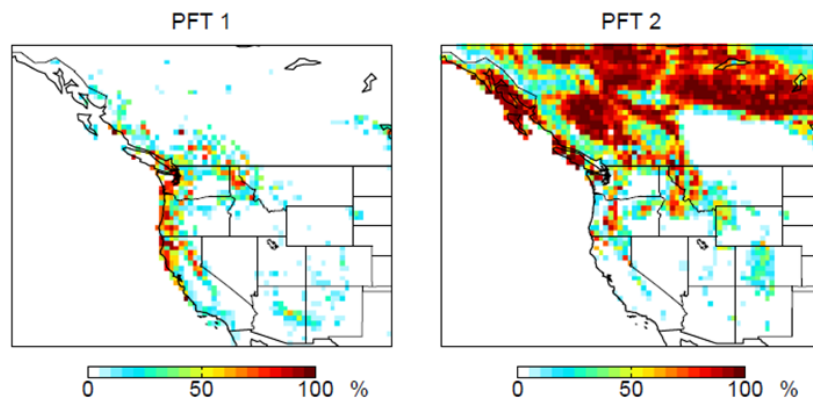
Figure 3.1: Percent of total surface area occupied by PFT 1 (Needleleaf Evergreen Temperate Tree) and PFT 2 (Needleleaf Evergreen Boreal Tree) in western North America

Table 3.1: CLM PFTs

PFT	Description
0	bare
1	needleleaf evergreen temperate tree
2	needleleaf evergreen boreal tree
3	needleleaf deciduous boreal tree
4	broadleaf evergreen tropical tree
5	broadleaf evergreen temperate tree
6	broadleaf deciduous tropical tree
7	broadleaf deciduous temperate tree
8	broadleaf deciduous boreal tree
9	broadleaf evergreen temperate shrub
10	broadleaf deciduous temperate shrub
11	broadleaf deciduous boreal shrub
12	arctic c3 grass
13	cool c3 grass
14	warm c4 grass
15	Crop 1
16	Crop 2

The Model of Emissions of Gases and Aerosols from Nature (MEGAN 2.1) is used in CLM4 to estimate up to 150 different BVOC compounds, including isoprene, monoterpenes, sesquiterpenes, and other oxygenated VOCs (Guenther et al., submitted). MEGAN estimates biogenic emissions from terrestrial vegetation for use in regional to global scale atmospheric chemistry models. A VOC flux in units of $\mu mol\ m^{-2}\ h^{-1}$ is calculated from a baseline emission that is modulated by an emission activity factor, which accounts for emission responses to meteorological and phenological conditions, including light, temperature, leaf age, and LAI. There is a different emission factor for each PFT for each compound. In CLM4 the BVOC emissions are calculated interactively at every time step.

The Community Atmosphere Model (CAM4) is a 3D global atmospheric model. We use specified meteorological fields generated from the GEOS-5 product (Rienecker et al., 2008). CAM can be run with an interactive atmospheric chemistry scheme based on the MOZART-4 (Model of Ozone and Related Chemical Tracers) chemical transport model, a configuration known as CAM-Chem. Lamarque et al. (2012) describe the features of the CAM-Chem model,

as well as validation against observations. The chemical mechanism contains extensive tropospheric chemistry, including O_3, NO_x, SO_x, CO, VOC oxidation processes, and a bulk aerosol scheme including sulfate, ammonium nitrate, carbonaceous aerosols, SOA, sea salt, and dust. Major BVOC species or classes are calculated within CLM4 and fed into the chemical mechanism of CAM-Chem (e.g. isoprene, monoterpenes, acetone, etc.). For our pine beetle simulations, the emissions for select speciated monoterpenes are also input from the CLM4. SOA in our simulations are produced by applying a fixed yield to the first generation products of the precursors. This is described in further detail in Section 4.3. The simulations are run at 1.9x2.5 degree horizontal resolution from 1997 to 2010 (with constant 2008 GEOS-5 meteorological fields) representing 14 years of cumulative beetle kill.

4 Data and Methods

4.1 Land use and beetle mortality

Beetle mortality data were obtained from Meddens et al. (2012). These data were created for the western United States (US) from the US Forest Service aerial detection survey program (1997-2010) and for British Columbia (BC) from the BC Ministry of Forests aerial overview program (2001-2010). The data are provided with 1x1 km grid resolution, and values in each gridcell represent the percent of the gridcell killed by beetles. The bark beetle species and hosts included in the data are listed in Table 4.1. Because the MPB has had such a significant impact in western North America compared to other beetles, the impacts from the MPB are compared to impacts from the other 12 beetles combined (OB) in this study. Figures 4.1 and 4.2 show the temporal and spatial extent of tree mortality in BC and the western US caused by the MPB and the OB. Figure 4.2 shows the extensive damage induced by pine beetles over the last decade, with total vegetated area decreasing by up to 30% in some regions. In Figure 4.1, the mortality area is calculated from the percent mortality data and added for the entire region (western US or BC). In Figures 4.2 and 4.3 the data have been converted to 0.5x0.5 degrees for use with the CESM land surface files as described below.

The original data for BC are percent mortality based on different severity classes. In Meddens et al. (2012) beetle species that contributed more than 0.1% of mortality area from 2001-2010 are used. The aerial overview survey (AOS) polygons are first overlaid with a 1-km grid to define the gridcell boundaries. The percent mortality of forest in a gridcell is calculated by multiplying the fraction of the gridcell covered by the polygon(s) by the percent mortality within the polygon(s). The percent mortality of forest in the gridcell is then multiplied by the

MODIS vegetation continuous fields (VCF) product to convert from percent mortality of forest

to percent mortality of the gridcell.

Table 4.1: Beetle species and host types available in the mortality data (Note: BC data has only mountain pine beetle, Western balsam bark beetle, spruce beetle, and Douglas-fir beetle with no specific host types. 'No data' indicates hosts of unknown type that were killed by that beetle.)

Beetle species	Host type
Mountain pine beetle	Bristlecone pine
	Limber pine
	Lodgepole pine
	Ponderosa pine
	Western white pine
	Whitebark pine
	No data
Pinyon ips	Common pinyon
	Singleleaf pinyon
Ips engraver beetles	Ponderosa pine
	No data
Bark beetles	California red fir
	Ponderosa pine
	Softwoods
	White fir
	No data
Western balsam bark beetle	Subalpine fir
	No data
Fir engraver	California red fir
	Fir species
	White fir
	No data
Spruce beetle	Engelmann spruce
	Spruce species
	No data
Douglas-fir beetle	Douglas-fir
	No data
Western pine beetle	Ponderosa pine
Pine engraver	Ponderosa pine
Jeffrey pine beetle	Jeffrey pine
True fir bark beetle	White fir
Roundheaded pine beetle	Ponderosa pine
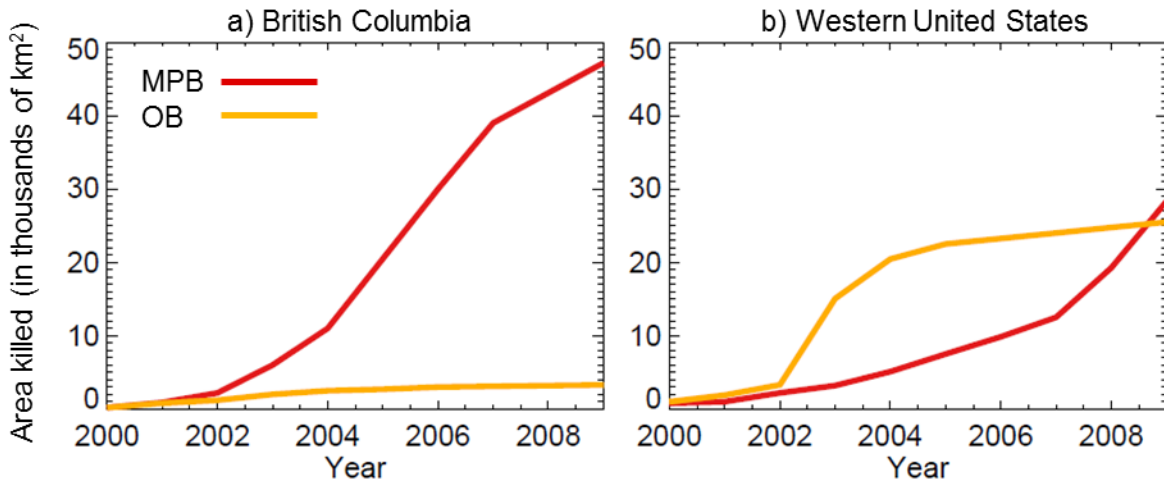
Figure 4.1: Cumulative beetle mortality over time in a) BC and b) western US caused by the mountain pine beetle and other beetles
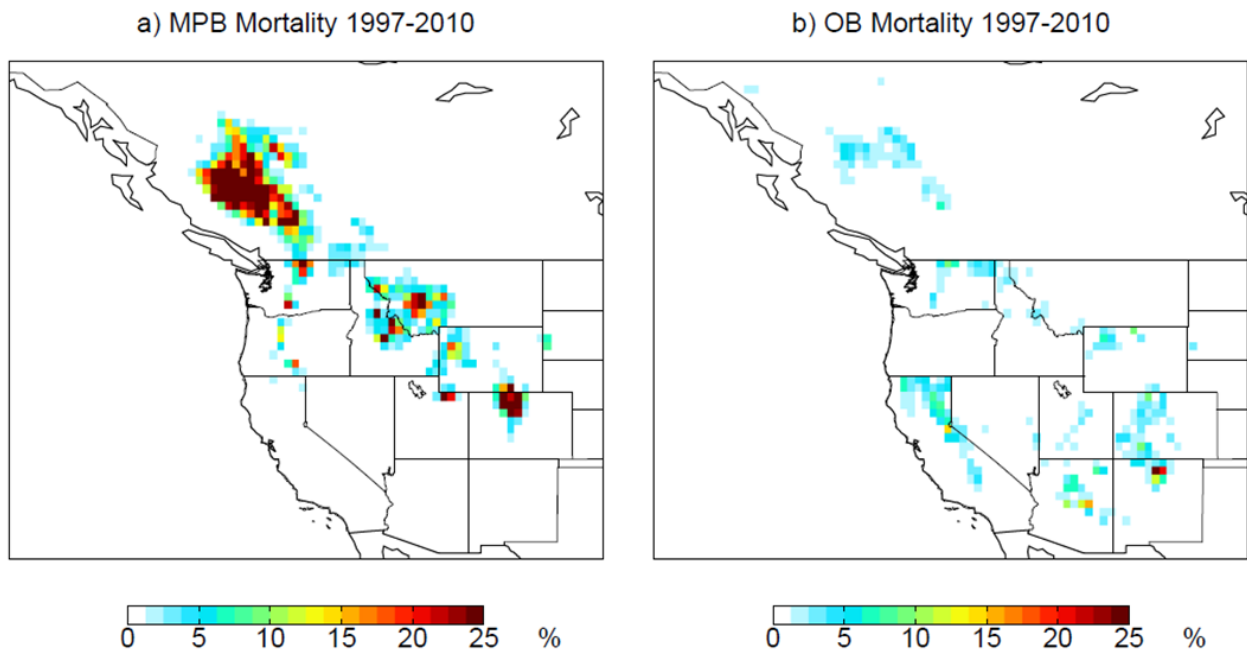
Figure 4.2: Cumulative percent of gridcell total area killed in 2010 in the western US and BC due to MPB (a) and OB (b). Color bar saturated at 25%.

The original data for the US are the number of trees killed. Beetle species that killed greater than 100,000 trees from 1997-2010 were used. Again, in Meddens et al. (2012) the AOS polygons are overlaid with a 1-km grid. The number of killed trees per gridcell is then

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calculated. This number is then multiplied by the crown area of the host tree to get the percent

mortality of the gridcell.

Meddens et al. (2012) indicate that the US aerial detection survey data underestimate the

number of trees killed based on field observations in Idaho, Colorado, and New Mexico. Using

remotely sensed imagery of beetle outbreak locations in Idaho, north -central Colorado, and

northern New Mexico, they developed and applied adjustment factors, ranging from 3.7 to 20.9,

to scale up the US beetle mortality data, improving the agreement with field observations and the

continuity with BC data across the border. The adjusted data are used for the US in this study.

Uncertainties in the mortality data include variability in estimates in space and time due

to differing abilities and techniques of different mappers, the mean tree crown area values used

for calculating mortality in a gridcell, limited data collection areas (wilderness or national park

areas were not regularly surveyed), and the adjustment factors for the US. The adjustment

factors are based on specific regional comparisons, but are applied uniformly throughout the US.

The "jump" in the other beetle mortality seen in 2003 (Fig.4.1a) is due to incomplete surveying

of a pinyon ips beetle outbreak in the Southwest US in 2002. Surveys were not routinely

conducted for pinyon pine forests before 2003 and so pinyon ips mortality added in 2003

includes mortality from the preceding years. This is not corrected for in the dataset. Figure 4.3

shows the region affected by the pinyon ips beetle in 2003.

For use in the CESM, The 1x1 km percent mortality data are converted to degree-space

by re-projection in ArcMap. Using IDL, the data were converted to a 0.5x0.5 degree grid. The

separate US and BC grids were then combined and placed into a single global 0.5x0.5 degree

grid for each year and cumulative mortality grids were created by adding each previous year to

the current year from 1997 to 2010. The land surface datasets were re-gridded to 1.9x2.5 degrees for our simulations.
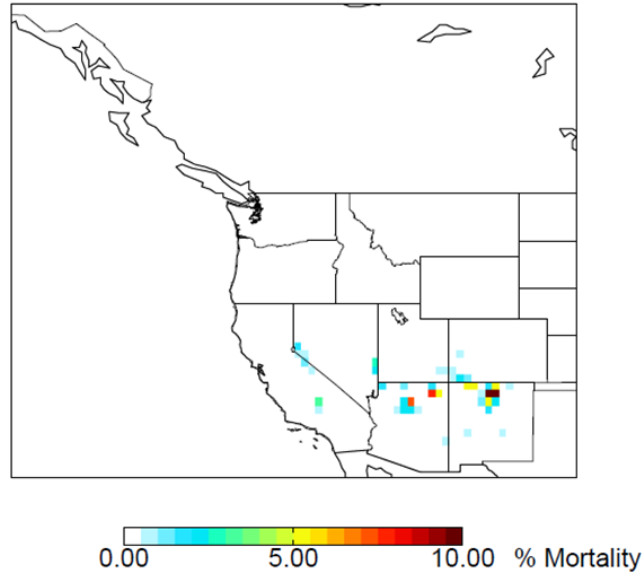
Figure 4.3: 2003 Needleleaf mortality caused by the Pinyon Ips beetle

Cumulative PFT reductions were calculated for each year from 1997-2010. Because the PFTs do not break into specific tree species and maps of sub-PFT species composition are not available, the mortality data were applied to PFTs 1 and 2 (the two needleleaf PFTs covering western North America, Figure 3.1) to simulate beetle attack. However, the same percentage of gridcell is affected whether the cumulative mortality is applied to a general PFT or specific species mortality is applied to each species within that PFT (if sub-PFT species information were available). This is because the mortality is defined as the percent of gridcell affected. For example, if a gridcell contains 50% needleleaf PFT (25% pine and 25% spruce) and 50% non-needleleaf PFT, and the mortality data for pine and spruce is defined as 10% for each species, then 10% of the entire gridcell is killed for each species (and applied only to that species.) for a total gridcell mortality of 20%. Applying the cumulative mortality to the general PFT in this example also results in a total gridcell mortality of 20%. Any reduction in PFT 1 or PFT 2 was

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also reflected as an addition to PFT 0 (bare ground) to ensure that total area coverage was conserved. As a result of the PFT reductions, total LAI decreases with pine beetle kill. The original CLM surface dataset was then used as the baseline. In some gridcells, because the PFT coverage and mortality data are from two different sources, mortality area can exceed PFT coverage. In these cases mortality is capped at this maximum PFT coverage.

4.2 Monoterpene Emissions

Amin et al. (2012) compared sorbent trap concentrations of monoterpenes emitted from healthy Engelmann spruce and lodgepole pine trees and spruce and pine under attack by the mountain pine beetle. In order to simulate the increase in VOCs due to beetle attack in the model, relative increases in the monoterpene compounds 3-carene, β-phellandrene, β-pinene, and p-cymene are calculated from Amin et al. (2012) lodgepole pine data for the first scenario and increases in 3-carene, β-phellandrene, β-pinene, p-cymene, α-pinene, and sabinene from Amin et al. (unpublished) Engelmann spruce data for the second scenario (Table 4.2). Because emitted monoterpenes had little time to react before being collected by sorbent trap filters, the ratio of changes in concentrations of monoterpenes from Amin et al., (2012) are a reasonable proxy for the ratio of changes in emissions in the model. Several compounds showed significant increases due to beetle attack for which increase factors could not be calculated due to the non-detection of the compounds in healthy trees (methyl chavicol for the lodgepole pine study and γ-terpinene, terpinolene, and 1,4-cineole for the spruce study), so in the model these compound emissions are not affected by beetle attack. Overall, the largest relative increase is for β-phellandrene in the pine study and 3-carene in the spruce study. For each scenario, the factors are applied to PFT 1 and PFT 2 and for all beetles, assuming that all trees would respond the same way to different

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beetle attacks. While numerous studies have demonstrated that the VOC increase effect occurs in many plant species caused by many insect species (e.g. Blande et al., 2007; Brilli et al., 2009; Staudt and Lhoutellier, 2007), the differences between the pine and spruce scale-up factors indicate that different tree species may have a very different response to beetle attack. We include these two scenarios here in an effort to characterize this range of response.

Table 4.2. Scale-up factors for beetle-induced changes in monoterpenes calculated from Amin et al. (2012) lodgepole pine and Engelmann spruce data.

Monoterpene	Scale-up Factor	
	Pine	Spruce
β-pinene	7.7	16
3-carene	7.3	65
β-phellandrene	33	5.3
P-cymene	5.4	42
α-pinene	-	3.6
Sabinene	-	18

4.3 SOA Formation

The yields of SOA formed from emissions of several of the monoterpene compounds impacted by beetles have been studied (Tables 4.3 and 4.4). Here we consider SOA formed from both ozonolysis and photooxidation reactions. β-pinene has SOA photooxidation yields from the literature ranging from 3 to 44% (Hatakeyama et al., 1991; Griffin et al., 1999; Hoffman et al., 1997; Jaoui and Kamens, 2003; Lee et al., 2006). 3-carene has SOA yields ranging from 2 to 38% (Griffin et al., 1999; Hoffman et al., 1997; Lee et al., 2006). SOA yields from photooxidation of α-pinene range from 1 to 43% (Hatakeyama et al., 1991; Hoffmann et al., 1999, Noziere et al., 1999, Jaoui and Kamens, 2001; Lee et al., 2006b). SOA yields from ozonolysis of α-pinene range from 8 to 41% (Yu et al., 1999; Griffin et al., 1999; Hoffmann et al., 1997; Winterhalter et al., 2003; Docherty et al., 2005, Lee et al., 2006a). Only Griffin et al. (1999) report SOA yields from photooxidation of sabinene, ranging from 3 to 15%.

Table 4.3: Photooxidation SOA yields from the literature (primary oxidant OH). Values used in this study indicated in bold.

Terpene	SOA Mass Yield (%)	Reference
β-phellandrene	35.68, 40.5, 40.4	Hoffmann et al., 1997
Limonene	8.7-34	Griffin et al., 1999
	58±1	**Lee et al., 2006b**
β-pinene	44±1	Hatakeyama et al., 1991
	30.2	Hoffmann et al., 1997
	3-27	Griffin et al, 1999
	20.9	Jaoui and Kamens, 2003
	31±1	**Lee et al., 2006b**
3-carene	22.9, 27	Hoffmann et al., 1997
	2-18	Griffin et al., 1999
	38±1	**Lee et al., 2006b**
1-methyl-3-n-propylbenzene*	**5.6**	**Odum et al., 1997**
α-pinene	43	Hatakeyama et al., 1991
	1.22-12.5	Hoffmann et al., 1997
	1.22-12.5	Griffin et al., 1999
	4-40	Noziere et al., 1999
	32-45	Jaoui and Kamens, 2001
	32±1	**Lee et al., 2006b**
Sabinene	**10.2**	**Griffin et al., 1999**
*Used for p-cymene		

Table 4.4: Dark ozonolysis SOA yields from the literature (primary oxidant O_3). Values used in this study indicated in bold.

Terpene	SOA Mass Yield (%)	Reference
β-pinene	32.1	Hoffmann et al., 1997
	0-5	Griffin et al., 1999
	4.5-8.3	Yu et al., 1999
	26	Jaoui and Kamens, 2003
	8.3-32.5	Docherty et al., 2005
	17±1	**Lee et al., 2006a**
3-carene	75.9	Hoffmann et al., 1997
	8-13	Griffin et al., 1999
	13	Yu et al., 1999
	47.5, 55.4	Docherty et al., 2005
	54±2	**Lee et al., 2006a**
α-pinene	13.7-18.0	Hoffmann et al., 1997
	8.3-18.6	Griffin et al., 1999
	15.9-18.6	Yu et al., 1999
	16	Winterhalter et al., 2003
	29.5-51.9	Docherty et al., 2005
	41±2	**Lee et al., 2006a**
Sabinene	**3.0**	**Griffin et al., 1999**
	3.5	Yu et al., 1999
	10.4, 38.1	Docherty et al., 2005

Three studies report sabinene ozonolysis SOA yields ranging from 2 to 38% (Yu et al., 1999; Griffin et al., 1999; Docherty et al., 2005).   Only one study has looked at SOA yields from β-phellandrene (Surratt et al., 2008) and this study finds that SOA yields from β-phellandrene may be similar to limonene SOA yields. Therefore, limonene SOA yields are used here to approximate SOA yields for β-phellandrene. No information could be found in the literature on SOA formation from p-cymene. Here we use SOA yields from the similar compound 1-methyl-3-n-propylbenzene from Odum et al. (1997) to approximate SOA yields for p-cymene. These two compounds have a similar chemical structure and likely react with oxidants on a similar timescale, so SOA yields from 1-methyl-3-n-propylbenzene are a reasonable approximation for p-cymene.

For SOA formation from these monoterpenes, we specify reactions for SOA formation in the chemical mechanism using bulk SOA yields from Lee et al. (2006a and b) (unless otherwise specified) and reaction rates at 298°K for the VOCs reacting with OH and O_3 from Atkinson (1997) (Table 4.5). The chamber studies of Lee et al. (2006a; 2006b) were conducted at low HC:NO_x ratio, considered representative of a ponderosa forest. Temperature dependence of the reaction rates is only available for α-pinene and β-pinene, thus the reaction rates are fixed at 298°K. Accounting for the temperature dependence of α-pinene and β-pinene oxidation rate translates to a ~14% and ~11% difference in lifetime, respectively, against OH oxidation from 298°K to 270°K. Given the short lifetimes of these species (Table 4.5) these differences are negligible. We do not include SOA formed from the reaction of monoterpenes with the NO_3 radical due to the difficulty in obtaining yields and reaction rates for the specific monoterpenes considered here. Because monoterpene oxidation by the NO_3 radical may be an important source of SOA at night (Winer et al., 1984; Fry et al., 2009; Fry, 2011), the SOA concentrations here are

a lower limit. Formed SOA is treated as non-volatile, a necessary simplification given the lack

of volatility parameter measurements for all of the specific SOA precursors under consideration.

Because the chemical mechanism already includes reaction of total monoterpenes with OH and

O_3, we recycle the oxidants in our reactions so that they are not consumed twice (Table 4.5). In

this way, we can pull out SOA tracers that are formed from the individual monoterpenes. We

also include a total SOA tracer formed from all monoterpenes ($C_{10}H_{16}$ class) with a yield of 10%.

Table 4.5: SOA Yield reactions added to chemical mechanism. BPHE = β-phellandrene, BPIN = β-pinene, CAR3 = 3-carene, PCYM = p-cymene, APIN = α-pinene, SABI = sabinene.

Chemical Reaction	Reaction Rate cm^3 molecule^{-1} s^{-1}	Monoterpene Lifetime (hours), assuming: [OH] = 1x10^6 molecules cm^{-3} [O$_3$] = 40 ppb
BPHE + OH → 0.55*SOAPH + OH	1.7E-10	1.7
BPIN + OH → 0.29*SOABP + OH	7.9E-11	3.5
CAR3 + OH → 0.36*SOACA + OH	8.8E-11	3.2
PCYM + OH → 0.05*SOACY + OH	1.4E-11	20.
APIN + OH → 0.32*SOAAP + OH	5.4E-11	5.1
SABI + OH → 0.095*SOASA + OH	1.2E-10	2.3
BPIN + O3 → 0.16*SOABP + O3	1.5E-17	17
CAR3 + O3 → 0.51*SOACA + O3	3.7E-17	7.0
APIN + O3 → 0.41*SOAAP + O3	8.7E-17	3.0
SABI + O3 → 0.028*SOASA + O3	8.6E-17	3.0
C10H16 + OH → 0.1*SOATOT + OH	5.3E-11	5.2
C10H16 + O3 → 0.1*SOATOT + O3	8.6E-17	3.0

To evaluate the change in VOC emissions and SOA burden in the model, we separate BC

and the western US (Figure 4.4). It is important to note that the gridcell edges do not line up

with the US/Canada border located at 49° N. A small portion of the BC box is placed in the US

box as a result; however only 1.5% of the total BC vegetation mortality is attributed to the US in
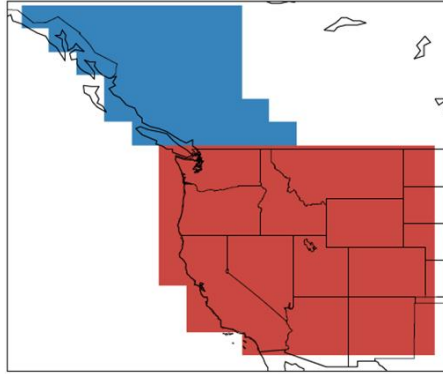
this process.

Figure 4.4: Western US Box (red) and BC Box (blue).

5 Results

5.1 Impacts on Monoterpene Emissions in Western North America

Beetle attack has two main effects on VOC emissions. The first is the mortality effect. This is the decrease in VOC emissions that occurs after trees are killed off year after year. The second is the attack effect. This is the increase in VOC emissions that occurs while a tree is under attack. To simulate the attack effect in the model, the scale-up factors for the monoterpenes in Table 4.2 are applied to the fraction of needleleaf evergreen trees (PFT 1 and PFT 2) that are under attack in each year. The increase in mortality from the current year to the next year corresponds to trees under attack in the current year. Therefore, the fraction of needleleaf evergreen trees under attack in the current year is calculated as PFT coverage in the current year minus PFT coverage in the following year, divided by the current year PFT coverage (Figure 5.1 shows an example of needleleaf trees under attack in 2008). We note that some trees attacked by beetles can survive the attack, and that these trees would not be accounted for here as the mortality dataset only includes trees under attack that subsequently died.
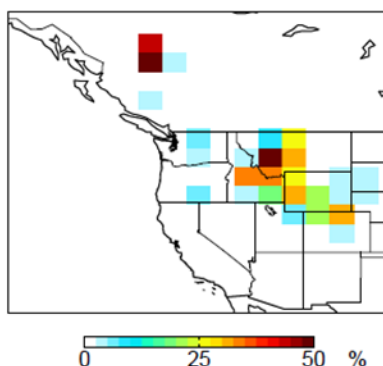
Figure 5.1: Percent of Needleleaf PFT under attack by the MPB in 2008.

We first show the simulated impact of pine beetle kill on monoterpene emissions and SOA formation based on the observed enhancements in lodgepole pine emissions (Amin et al.,

2012; Table 4.2). Lodgepole pine is the main species under attack by pine beetles (Logan and

Powell, 2001). Figure 5.2 shows the simulated summer-mean baseline emissions of the four

monoterpenes impacted in the pine scenario. These emissions peak in the summertime, therefore

we focus on that season through the study. Emissions of β-pinene and 3-carene are the largest

while emissions of β-phellandrene and p-cymene are much smaller in comparison. Emissions of

these monoterpenes are large in the western US, particularly in Washington, Oregon, and

Northern California, due to the extensive coverage of needleleaf trees in this region (Figure 3.1).

Figure 5.2: Simulated summer-mean baseline emissions of the four monoterpenes.

The impact of beetle attack on β-phellandrene emissions is shown as a specific example. This monoterpene had the largest scale-up fa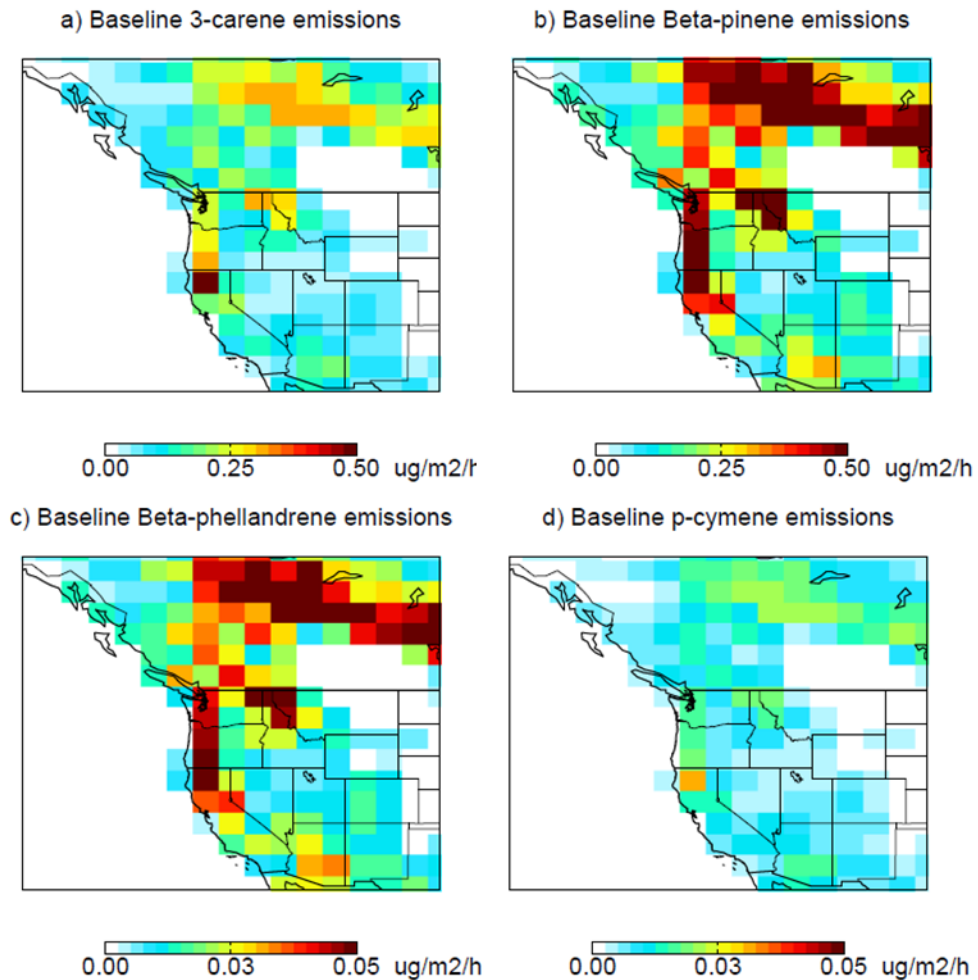ctor for the pine scenario. Figure 5.3 shows maps of simulated β-phellandrene emissions for 2004 under baseline conditions and accounting for 8 years of MPB kill. Although MPB infestation peaked in BC in 2007, the MPB had the largest impact in 2004. This is due to increasing mortality after 2004 which was able to overcome the stronger attack effect in 2007. Emissions of β-phellandrene decrease locally by up to 10% due to the mortality effect. When the attack effect is included, emissions increase throughout the region up to three-fold.

Figure 5.3: Change in simulated summer-mean β-phellandrene emissions due to the impact of mountain pine beetle attack in 2004. a) Change in emissions due to mortality effect alone. b) Change in emissions due to mortality effect plus the attack effect.

Figure 5.4 shows the mortality and attack effect in 2008 after 12 years of MPB kill. Although MPB infestation peaked in 2009 in the western US (Man 2012), the MPB had the largest impact in the western US in 2008. As discussed above for BC, this is likely because in 2009 the cumulative mortality has increased enough to mute the attack effect even though more trees were under attack in 2009. In 2008, emissions of β-phellandrene decrease locally by up to

38% due to the mortality effect, but when the attack effect is included emissions increase throughout most of the region, up to four-fold.
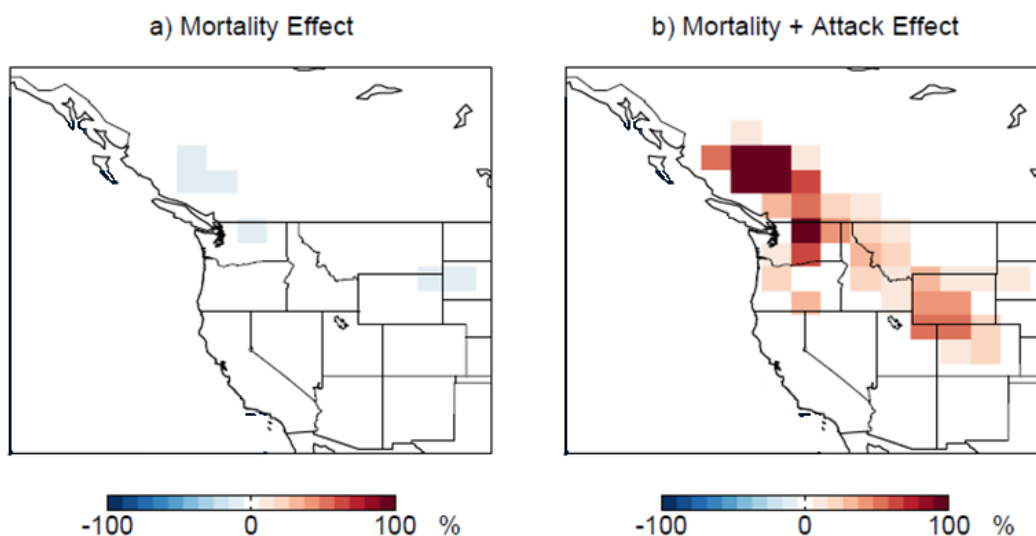

Figure 5.4: Change in simulated summer-mean β-phellandrene emissions due to the impact of mountain pine beetle attack in 2008. a) Change in emissions due to mortality effect alone. b) Change in emissions due to mortality effect plus the attack effect.

Figure 5.5 shows how simulated β-phellandrene emissions are affected by the other beetle attack in 2008. Baseline β-phellandrene emissions are compared to emissions in 2008 after 12 years of OB kill. Cumulative mortality associated with the OB reduces emissions by up to 30% in 2008; however, the number of trees under attack year-to-year has plateaued in 2008 (Figure 4.1) and the OB attack effect on emissions is therefore modest. When both mortality and attack effect are accounted for there are both local increases and decreases in β-phellandrene emissions, but these changes generally do not exceed 20% in 2008.

Figure 5.6 shows maps of simulated β-phellandrene emissions for 2002 under baseline conditions and accounting for 6 years of OB kill. In this year, the OB had the largest impact in both the western US and BC. Emissions of β-phellandrene decrease locally by up to 4% due to

the mortality effect. When the attack effect is included, emissions increase throughout the region

up to four-fold in the US; however, the OB impact in BC is small.

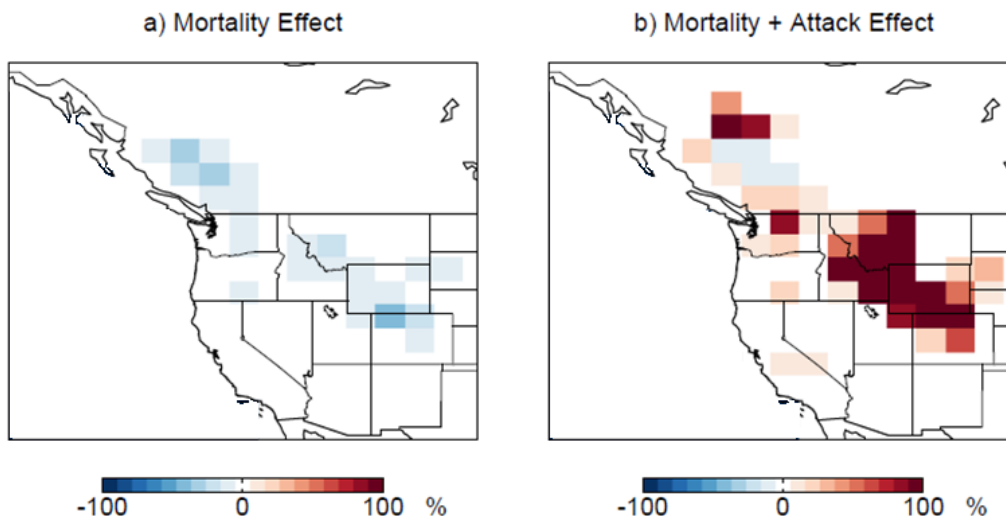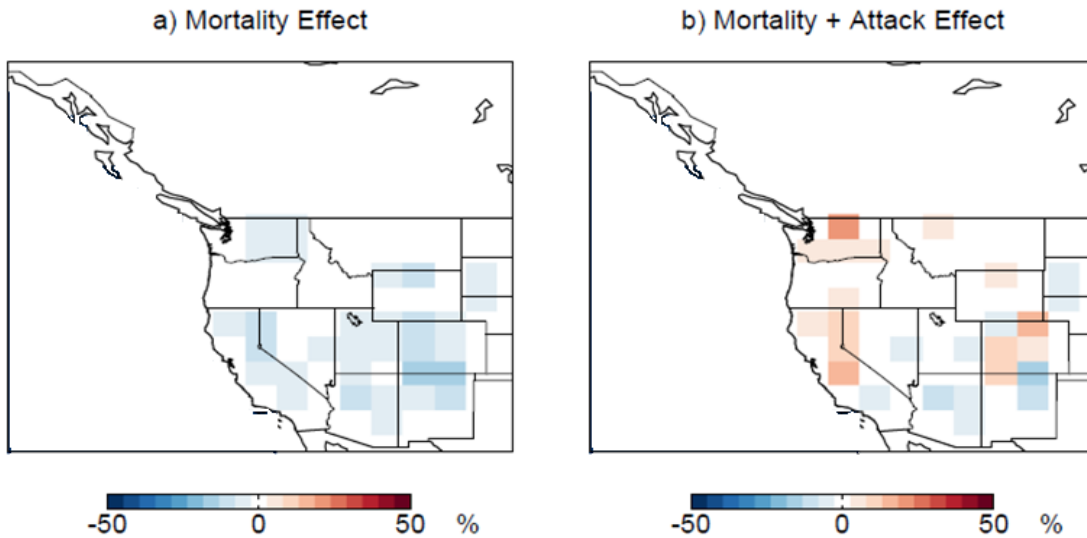**Figure 5.5:** Change in simulated summer-mean β-phellandrene emissions due to the impact of other beetle attack in 2008. a) Change in emissions due to mortality effect alone. b) Change in emissions due to mortality effect plus the attack effect.

Figure 5.6: Change in simulated summer-mean β-phellandrene emissions due to the impact of other beetle attack in 2002. a) Change in emissions due to mortality effect alone. b) Change in emissions due to mortality effect plus the attack effect.

Figure 5.7 shows the impact of mountain pine beetle kill in 2004 and 2008 on the sum of monoterpenes affected. While β-phellandrene emissions exhibit the largest relative increase due to beetle attack, the impact on the other monoterpenes is more modest; therefore, the summed monoterpenes have a lower relative increase. However, the absolute increase is larger due to the inclusion of more abundantly emitted monoterpenes (e.g. β-pinene). With all four monoterpenes included, the largest increase above baseline is 70% in 2004 and 104% in 2008.
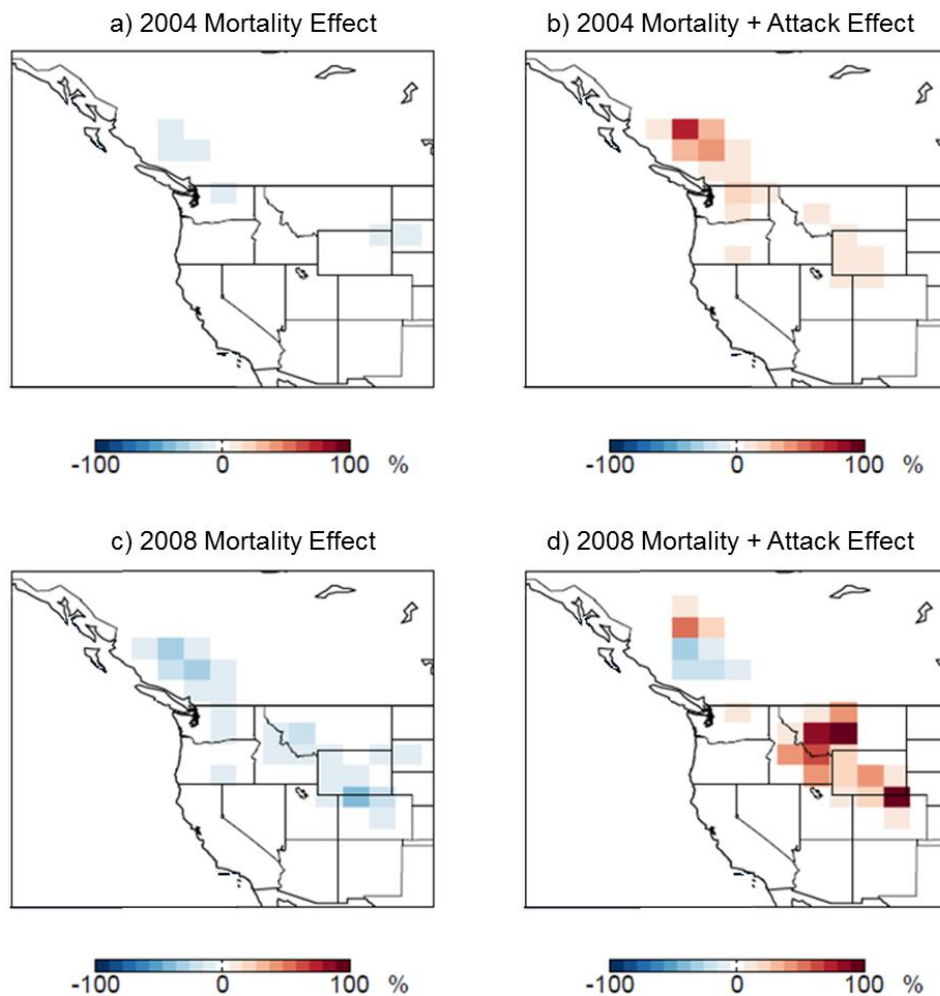
Figure 5.7: Change in simulated summer-mean emissions of the four monoterpenes impacted by mountain pine beetle in 2004 and 2008. a) Change in emissions due to mortality effect alone in 2004. b) Change in emissions due to mortality effect plus the attack effect in 2004. c) Change in emissions due to mortality effect alone in 2008. d) Change in emissions due to mortality effect plus the attack effect in 2008.

Figure 5.8 shows the impact of both mountain pine beetle attack and other beetle attack combined on monoterpene emissions in 2002 and 2004. Although the impact from MPB alone peaks in later years, the impact of MPB and OB together for the entire region peaks in 2002 due to the large impact of OB in the US in 2002 and the fact that the cumulative mortality is lower in 2002 than in later years (Figure 5.8a vs. 5.8c), causing the attack effect to have greater impact in 2002.

Figure 5.8: Change in simulated summer-mean emissions of the four monoterpenes impacted by both mountain pine beetle and other beetles in 2002 and 2004. a) Change in emissions due to mortality effect alone in 2002. b) Change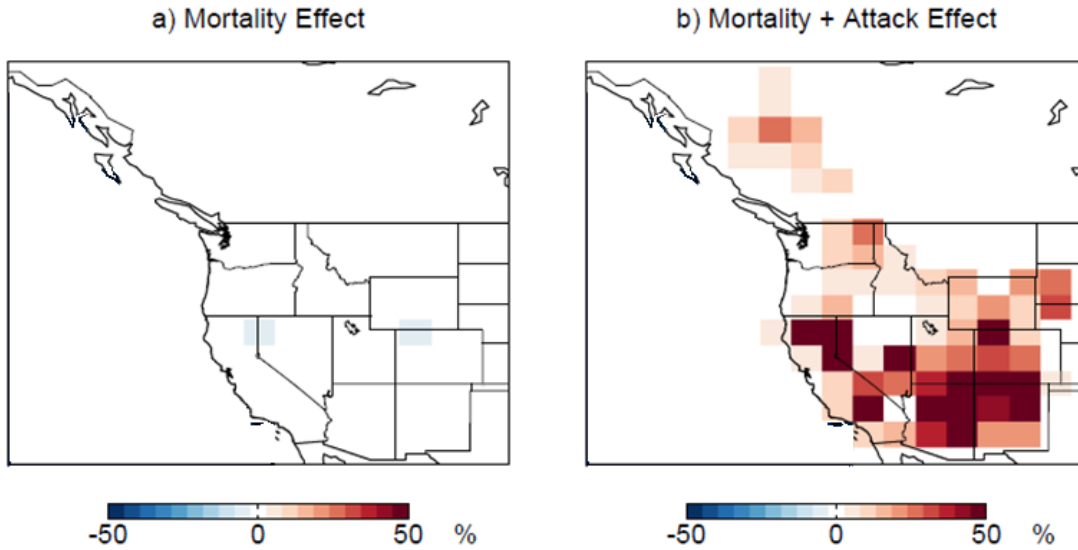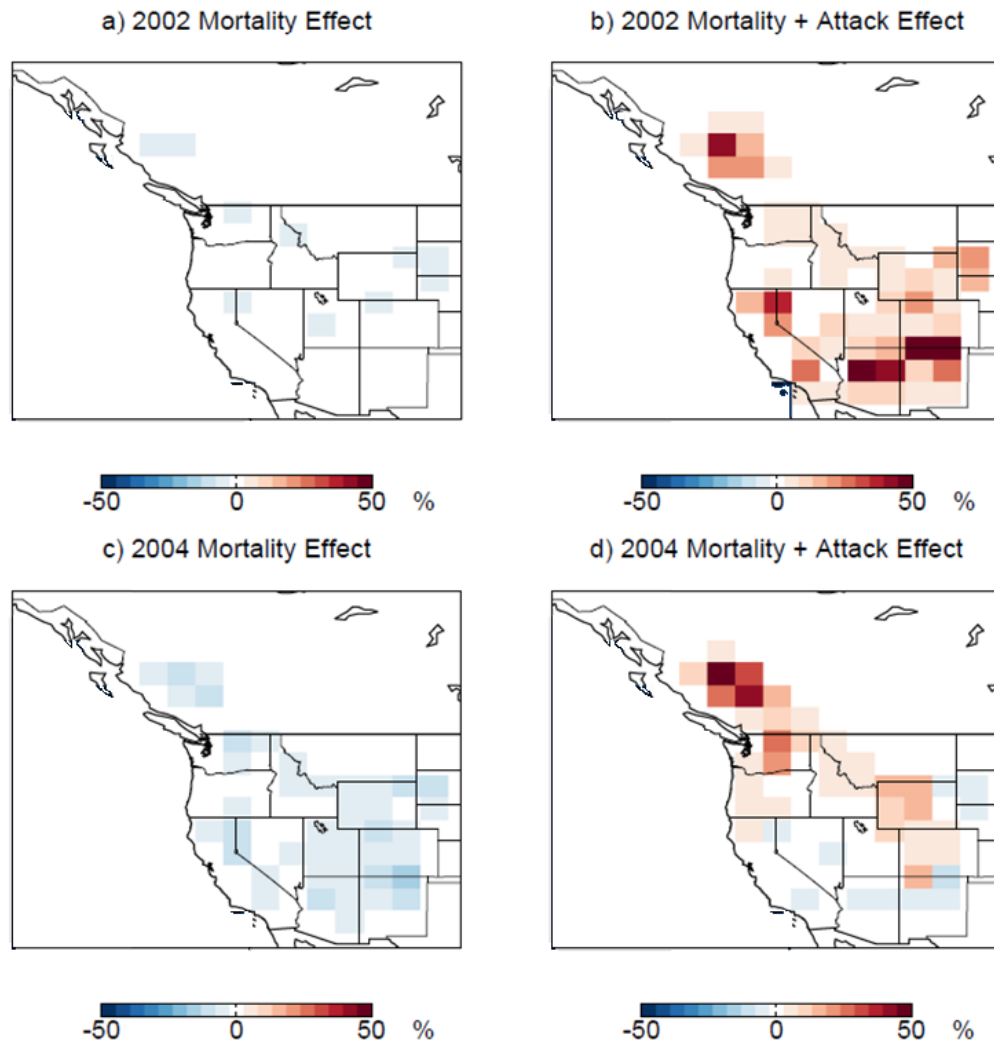 in emissions due to mortality effect plus the attack effect in 2002. c) Change in emissions due to mortality effect alone in 2004. d) Change in emissions due to mortality effect plus the attack effect in 2004.

Figure 5.9 shows how total emissions of the four monoterpenes affected by beetle attack evolved over time in the model in BC and the western US in the MPB run. The emissions for all time series plots are shown as the total emissions from the entire box (either western US or BC). For each plot, the black horizontal line represents the baseline emissions of the compounds. The bars represent the mortality effect or the decrease in emissions over time as trees are killed off. The red line represents the mortality effect plus the attack effect. Because the fraction of trees under attack in a year is calculated based on the change in mortality from the current year to the following year, an attack effect cannot be calculated for the final year of mortality in the dataset (2010). The height of the red line above baseline depends on how many trees are under attack in that year. The more trees under attack, the stronger the attack effect and the higher emissions will be above baseline.
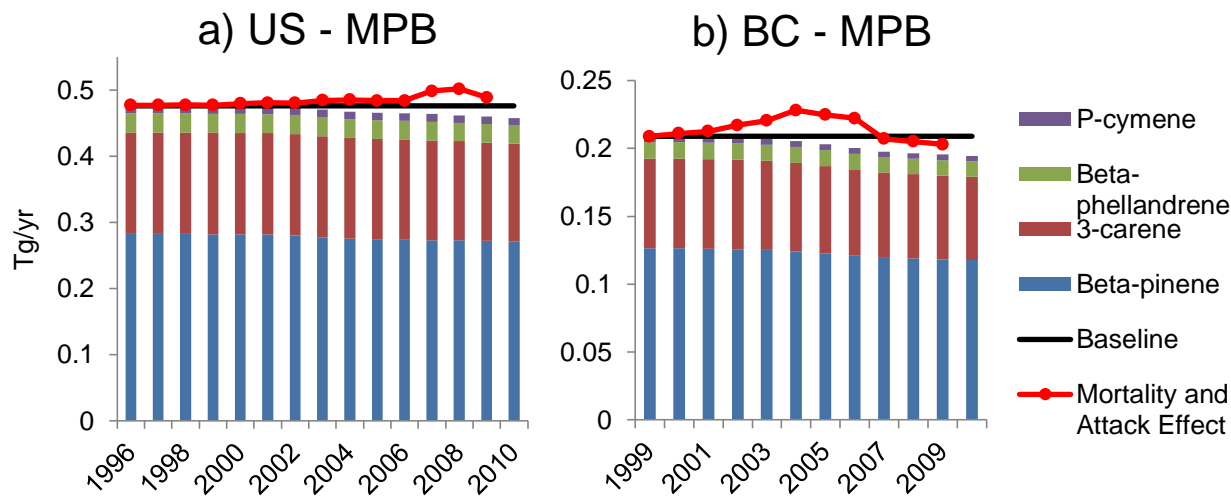
Figure 5.9: Time series showing evolution of the simulated total regional monoterpene emissions affected by mountain pine beetle kill in a) US and b) BC. The black line represents baseline emissions. The bars represent the mortality effect. The red line represents the mortality effect plus the attack effect.

For BC, the maximum increase in total emissions of these four compounds is 7% above baseline in 2004. For the US, the maximum increase in emissions of these four compounds is

3% above baseline in 2008. Since the emissions of the other three compounds are not as enhanced under attack as β-phellandrene (Table 4.2), the attack effect is not seen as strongly in emissions of all four of these compounds combined. While total monoterpene emissions are not significantly perturbed by pine beetle kill, Figure 5.7 shows that local effects can be substantially larger.

For OB impacts in the US, there is a peak in monoterpene emissions in 2002 associated with the pinyon ips outbreak from 2002-2004 discussed above in section 4.1. Aside from this peak, there is little to no increase in emissions above baseline, and after 2003, emissions begin to fall back below baseline. Overall OB impacts in BC are small, and while the MPB outbreak has been growing in the US in the last few years (Figure 5.8) the impact of the OB has declined.

5.2 Impacts on SOA Formation in Western North America

Changes in monoterpene emissions throughout western North America will impact SOA loading in the region. Figure 5.11 shows baseline summertime-average simulated concentrations of SOA from each of the four monoterpenes. These SOA distributions largely mimic the spatial patterns of the monoterpene emissions shows in Figure 5.2 due to the rapid oxidation and formation of the aerosol products in the model. Both 3-carene and β-pinene continue to be the dominant SOA precursors, but β-phellandrene emissions make a more important relative contribution to SOA than VOC emissions as a result of the high estimated SOA yields for this compound (Table 4.5). Here we can assess the spatial and temporal changes in SOA concentration caused by the changes in the monoterpenes discussed above. We show the impact on SOA from β-pinene here since β-pinene produces the most SOA out of the four and so has the largest absolute increase.
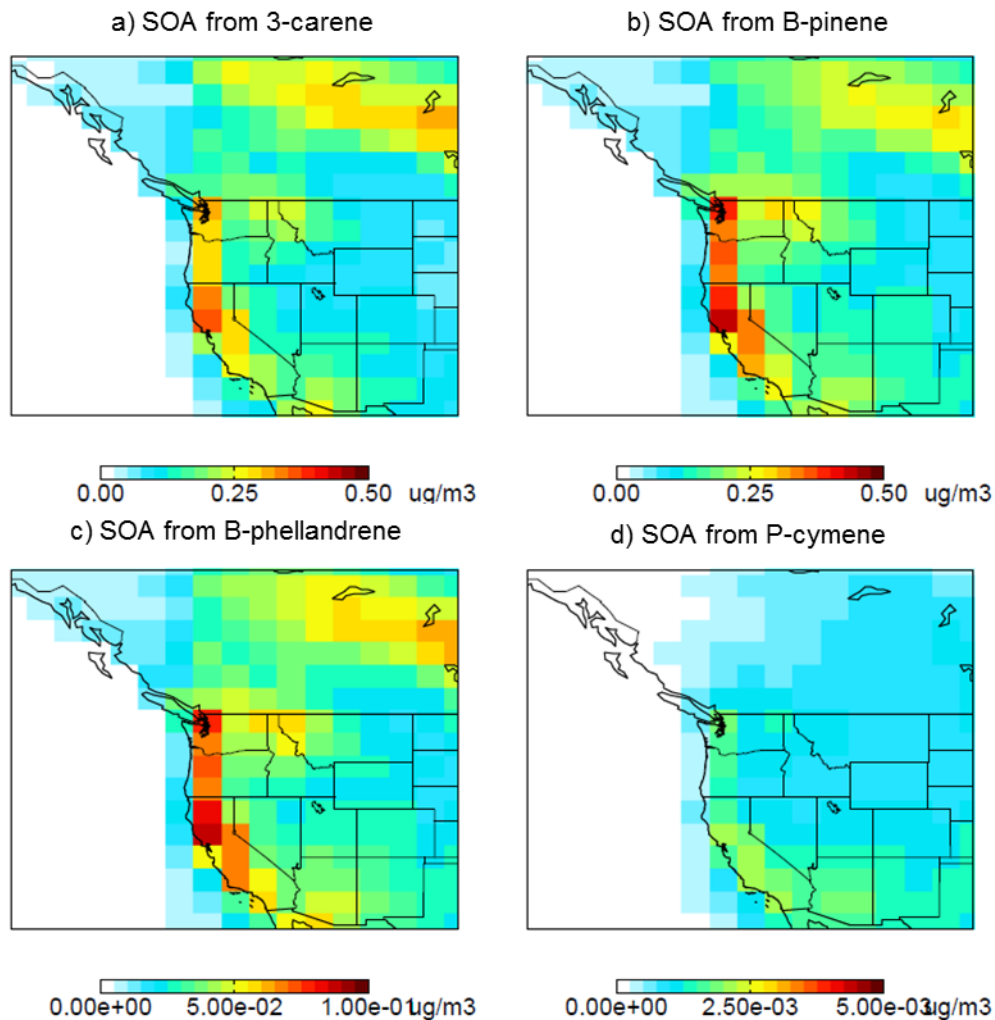
Figure 5.10: Baseline summer-mean concentrations of SOA from a) 3-carene, b) β-pinene, c) β-phellandrene, and d) p-cymene.

Figure 5.11 shows the mortality and attack effect on the concentration of SOA from β-pinene in 2004 and 2008 caused by the MPB for the pine scenario. The impact of beetle infestation is more widespread spatially than the impact on monoterpene emissions because SOA can be formed and transported downwind of precursor sources. In both 2004 and 2008 the mortality effect causes widespread decreases in concentrations of 5-10% with a maximum decrease below baseline of 16% in 2008. When the attack effect is included, SOA concentrations in areas of the western US are as high as 28% above baseline.
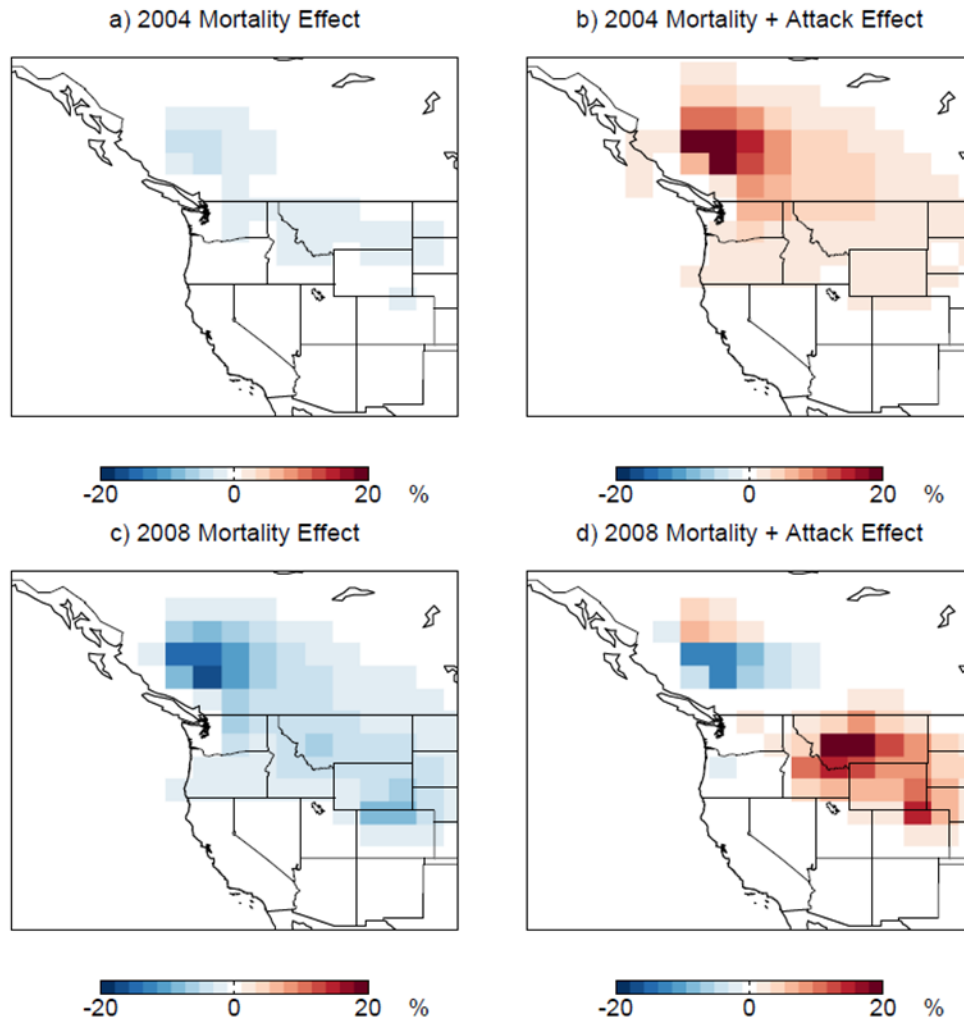
Figure 5.11: Change in simulated summer-mean SOA concentration from β-pinene due to the impact of mountain pine beetle attack in 2004 and 2008. a) Change in emissions due to mortality effect alone in 2004. b) Change in emissions due to mortality effect plus the attack effect in 2004. c) Change in emissions due to mortality effect alone in 2008. d) Change in emissions due to mortality effect plus the attack effect in 2008.

Figure 5.12 shows the mortality and attack effect on SOA from β-pinene in 2002 caused by the OB for the pine scenario. The resulting impact of the OB mortality effect on SOA is a maximum decrease below baseline of 5%. When the attack effect is included, SOA concentrations in areas of the western US are as high as 28% above baseline with the largest impacts in the southwest US.
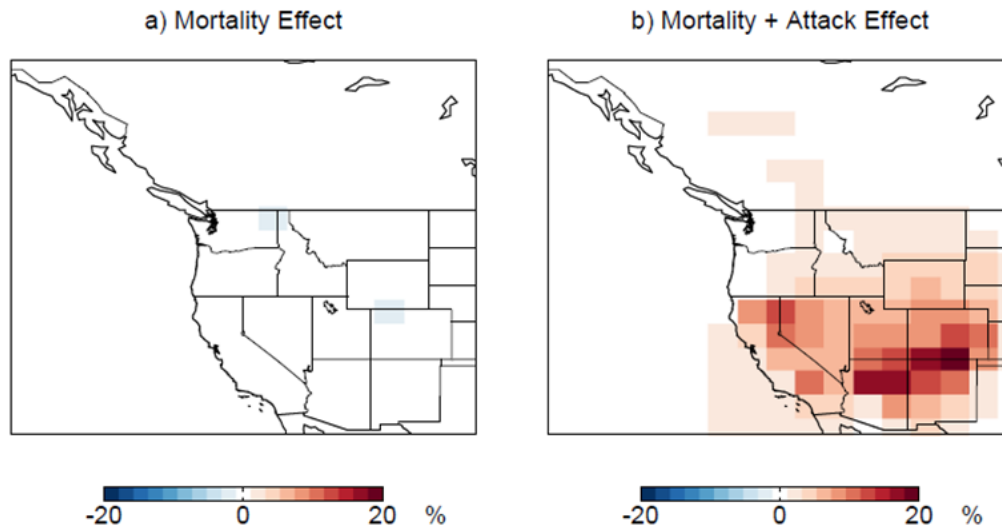
Figure 5.12: Change in simulated summer-mean SOA concentration from β-pinene due to the impact of other beetle attack in 2002. a) Change in emissions due to mortality effect alone. b) Change in emissions due to mortality effect plus the attack effect.

The timelines of the SOA changes are not included here as they follow the timelines of the monoterpene emissions (Figure 5.9). The largest overall impact on SOA concentration from the attack effect from all beetles (MPB plus OB) for the whole region (western US plus BC) is in the year 2002 (Figure 5.13) just as for monoterpenes (Figure 5.8). During this year, the mortality effect causes widespread decreases in SOA from 1-5%. Including the attack effect, specific areas see increases above baseline of up to 30% with a widespread increase above baseline of ~10%. The year with the smallest overall attack effect impact is in the year 2009, likely a combination of decreasing infestation of MPB in BC and a decreasing infestation of the OB in the western US, allowing the mortality effect to overcome the attack effect in this year. Figure 5.13 also includes the impact of MPB and OB attack on SOA in 2004. The largest increase above baseline in this year is 43%.
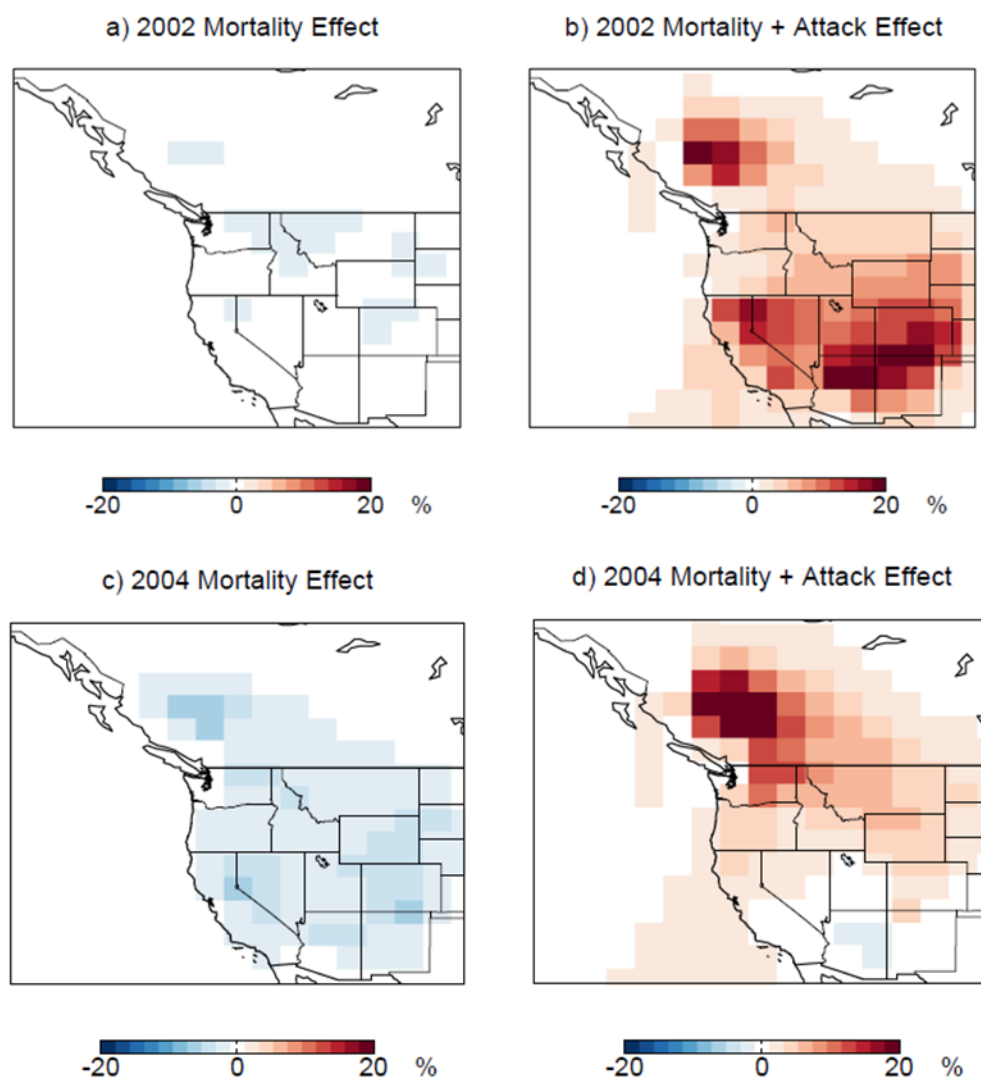
Figure 5.13: Change in simulated summer-mean SOA concentration from all four monoterpenes due to the impact of both mountain pine beetle and other beetle attack in 2002 and 2004. a) Change in emissions due to mortality effect alone in 2002. b) Change in emissions due to mortality effect plus the attack effect in 2002. c) Change in emissions due to mortality effect alone in 2004. d) Change in emissions due to mortality effect plus the attack effect in 2004.

5.3 Spruce Scenario

Next we show the simulated impact of pine beetle kill on monoterpene emissions and SOA formation based on the observed enhancements in Engelmann spruce emissions under attack by MPB (Amin et al., unpublished data; Table 4.2). The spruce tree response differs considerably from the pine tree response as seen in the differences between scale-up factors for

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the monoterpenes in each scenario. The spruce scenario also includes two additional

monoterpenes impacted (α-pinene and sabinene).
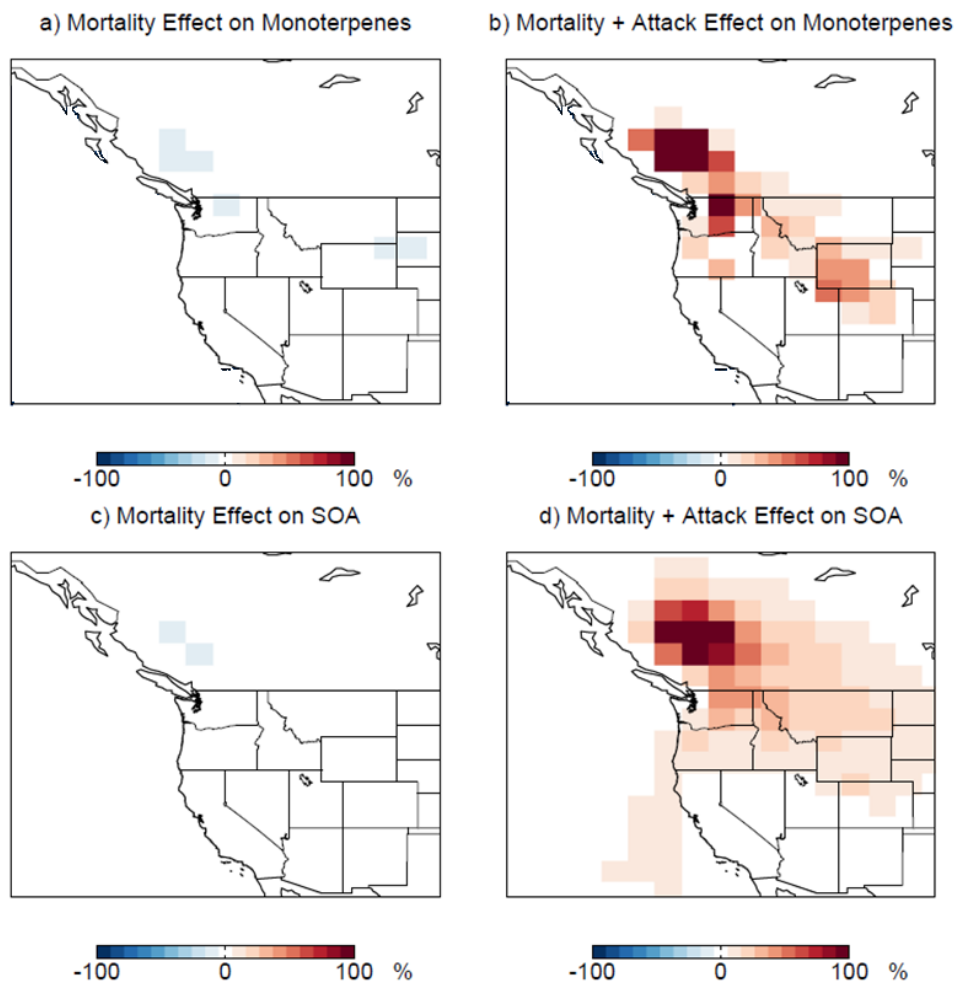
Figure 5.14: Change in simulated summer-mean monoterpene emissions and summer-mean SOA concentrations from all four monoterpenes due to the impact of mountain pine beetle attack in 2004 for the spruce scenario. a) Change in monoterpene emissions due to mortality effect alone. b) Change in monoterpene emissions due to mortality effect plus the attack effect. c) Change in SOA concentrations due to the mortality effect. d) Change in SOA concentrations due to the mortality effect plus the attack effect.

Figures 5.14 and 5.15 show the mortality and attack effect in 2004 and 2008 caused by

the MPB for total monoterpenes and SOA from spruce. The mortality effect is the same for both

the pine and spruce scenarios, while the attack effect, which takes into account the different

scale-up factors, is much different for the spruce scenario. While similar areas are impacted in

both scenarios, the increase in both monoterpene emissions and SOA formation due to the attack effect is a factor of 3-4 higher than the increase seen in the pine scenario. For comparison purposes we have only 2004 and 2008 MPB for the spruce scenario. Out of these two years, the maximum increases in monoterpene emissions (3-fold) and SOA (over 2-fold) occurs in 2008.
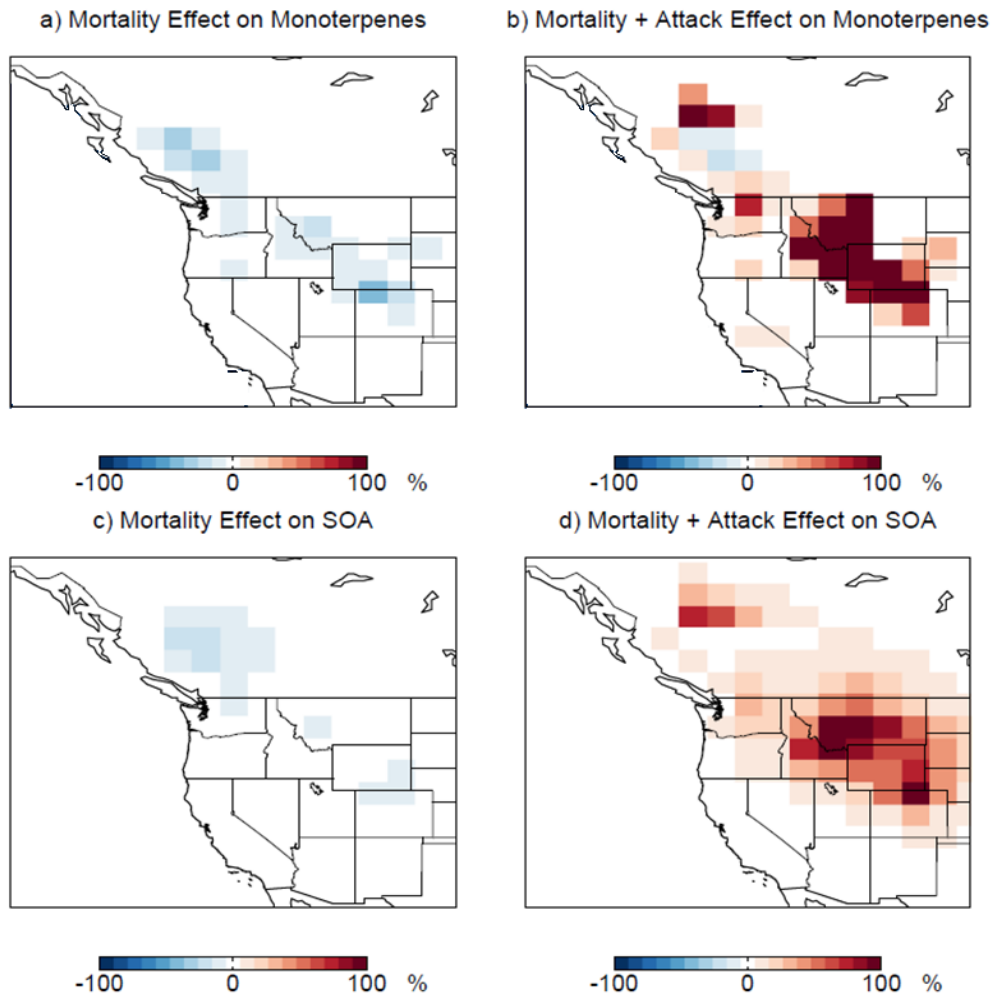
Figure 5.15: Change in simulated summer-mean monoterpene emissions and summer-mean SOA concentrations from all four monoterpenes due to the impact mountain pine beetle attack in 2008 for the spruce scenario. a) Change in monoterpene emissions due to mortality effect alone. b) Change in monoterpene emissions due to mortality effect plus the attack effect. c) Change in SOA concentrations due to the mortality effect. d) Change in SOA concentrations due to the mortality effect plus the attack effect.

We can estimate the contribution of each of these scenarios to the overall beetle impact by looking at the contribution of lodgepole pine and Engelmann spruce forest to the overall

forested area in western North America. Lodgepole pine and Engelmann spruce are the most

dominant tree species in British Columbia. Before the beetle infestation began, lodgepole pine

covered approximately 25% of the forested area in British Columbia and Engelmann spruce

covered approximately 15% (Westfall and Ebata, 2011). Currently both lodgepole pine and

Engelmann spruce cover ~15% of the forested area in British Columbia (Westfall and Ebata,

2011). This indicates that lodgepole pine has been impacted much more than Engelmann spruce

and that the pine scenario impact would likely dominate in British Columbia, while there may be

areas of spruce which see a higher localized impact based on the Amin et al. (2012) data.

In the western US, lodgepole pine covers ~7-10% of the forested area with spruce

covering slightly less. Mountain pine beetle attacking lodgepole pine remains the most

damaging forest pest in western North America, however Engelmann spruce are also under

attack on a smaller scale across the western US (Man, 2012). Again, the pine scenario impact

would likely dominate in the US with some areas of spruce seeing higher localized impacts.

6 Discussion

The impacts of pine beetle attack on monoterpene emissions are in many cases highly localized, meaning that particular regions may have experienced noticeably degraded air quality and visibility, but for the western US and BC on a whole, the impact is muted. The impact can be highly variable both over time and spatially as beetle outbreaks spread to different areas. While we have shown emission increases in years experiencing peak beetle attack, emissions can also fall in years of limited infestation when the mortality impact of previous pine beetle kill overcomes any on-going attack effect. The increased surface temperatures measured in beetle-killed forests by Hais and Kucera (2008) and Wiedinmyer et al. (2012) may also increase VOC emissions beyond what we investigated here, as VOC emissions increase exponentially with increasing temperature (Guenther et al., 1993).

We compare simulated SOA to measurements of OM and PM2.5 from the Interagency Monitoring of Protected Visual Environments (IMPROVE) network in order to provide some context for the simulated aerosol concentration changes due to beetle kill. The IMPROVE network consists of about 200 sites across the US where filters are collected every three days and analyzed for concentrations of speciated PM2.5. SOA cannot be separated from primary organic aersosol (POA) in the measurements, thus we compare our total OM simulation here. The IMPROVE network measures organic carbon (OC) which is multiplied by a factor of 1.4 (based on Grosjean and Friedlander, 1975) to convert to OM. Calculated OM concentrations are currently available for most sites from 2005-2010. Simulated SOA makes up only a fraction of total simulated OM, typically 15-50% in winter and 40-90% in summer in the model simulation. We see that the model simulation underestimates total observed OM over the Western United

States, consistent with previous model studies around the world (Figure 6.1) [e.g. Heald et al., 2005; de Gouw et al., 2005; Volkamer et al., 2006; Lamarque et al., 2011]. The factor of 1.4 used in the conversion of OC to OM may also be too low for many areas of the western US (El-Zanan et al., 2005), further widening the gap between observed and simulated OM. In addition, fire emissions (a large source of organic mass in the western US) in these simulations are monthly mean averages for 2008 from GFED2 (van der Werf et al., 2006). Because fire emissions are the same each year in the model, the model is missing the large annual variability that can occur in fire emissions.
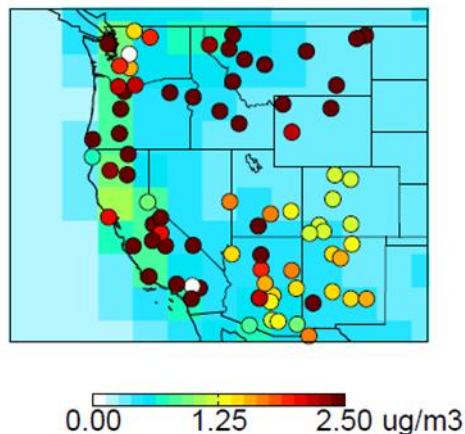
Figure 6.1: Comparison of simulated summertime-average OM (map) and observed summertime-average IMPROVE OM (circles).

To characterize the impact of the effects of pine beetles discussed here, we compare changes in monthly mean SOA and OM concentrations from the model with observed monthly mean concentrations of OM and PM2.5. For the region, OM is typically 15-70% of total PM2.5 measured in summertime in the Western US. Figure 6.3 shows monthly mean measurements of IMPROVE OM from two sites in Montana and Colorado that have been impacted by beetle infestation along with simulated SOA concentrations and simulated OM concentrations sampled to site location. Because the chemistry in the model is a simplified treatment, natural variability is not always captured well in the model. Simulated SOA and OM peak in July in nearly all

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areas in the western US while observed OM in some regions can peak in other months or have

double peaks. These figures also confirm that total OM is underestimated by a factor of 2-3 in

the summertime at these sites.
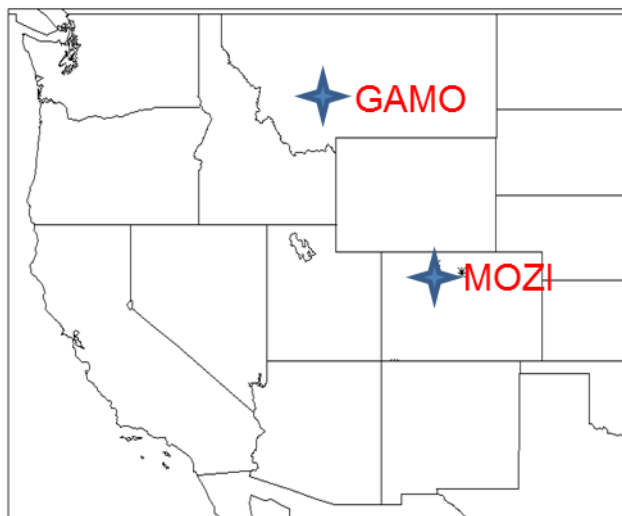
Figure 6.2: Location of IMPROVE sites in Figure 24.

Figure 6.3 also includes 2008 simulated SOA accounting for MPB mortality and attack

from the pine scenario and spruce scenario for both sites. These sites are examples of areas

which saw some of the highest increases in total SOA concentrations above baseline in the

western US. Compared to changes in SOA in the pine scenario, the annual and interannual

variability in observed OM is much larger, indicating that impacts from beetle infestation may

not be detectable in OM or PM2.5 measurements. However, the spruce scenario sees much

larger relative and absolute increases in SOA concentrations due to beetle attack, in this case

more than a doubling of SOA concentration in summertime. The magnitude of these localized

changes may be large enough to observe in surface measurements of OM and PM2.5. However,

due to uncertainties in simulated SOA concentrations and poor overall model agreement for OM

concentrations, it is difficult to make this assessment. Other sources of OM besides SOA include

fossil fuel and biofuel combustion. Wildfires are the main contributor to the large annual and

interannual variability in OM and PM2.5 in the western US (Spracklen et al., 2007) and this may

mask changes in SOA and OM caused by beetle attack in the pine scenario. However, in areas

dominated by spruce under attack and less affected by wildfires or other OM sources (leading to

smaller variability in OM concentrations), beetle-induced SOA changes may be observable.
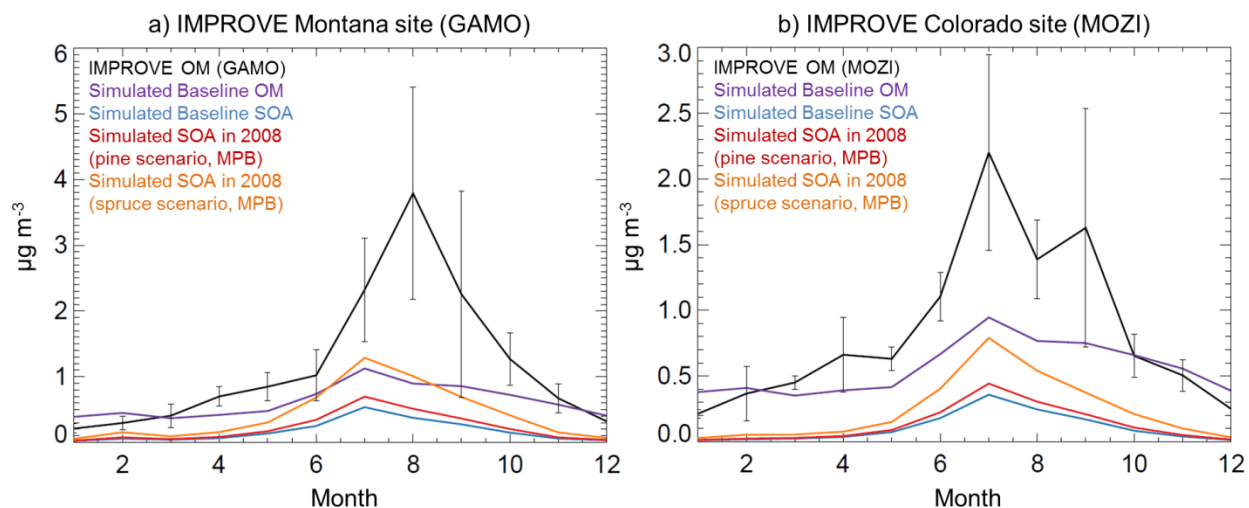
Figure 6.3: Comparison between observed baseline IMPROVE OM concentrations to simulated baseline OM and SOA at a site in a) Montana (GAMO) and b) Colorado (MOZI). Error bars are standard deviation of monthly means.

Following a beetle infestation, there may also be changes in the VOC emissions profile of

forests due to re-vegetation of other species. While re-vegetation dynamics are not fully

understood, Collins et al. (2011) discuss re-vegetation in lodgepole pine stands in Colorado. In

areas of remote forest, deciduous trees such as aspen can grow, which emit mainly isoprene,

another class of biogenic VOCs. However, in other areas, coniferous trees such as subalpine fir

or Engelmann spruce may initially replace lodgepole pine (Collins et al., 2011). While areas

outside of Colorado may experience different re-vegetation effects, any species differing from

the original tree species will cause a change in the amounts and types of VOCs emitted, as well

as subsequent SOA loadings in the region. Tree succession is expected to occur on longer

timescales than the decade of beetle infestation investigated here.

There are several ways that the estimates presented here could be refined or improved. The mortality dataset used here contains specific tree species attacked by beetles, but the CLM4 surface datasets limited the implementation here to broad PFT categories (e.g. Needleleaf Trees). If landcover datasets of specific species become available, future studies can be improved by applying mortality directly to these species, leading to higher spatial accuracy of mortality represented in the model. This can then lead to a more accurate assessment of the regional impact of beetle infestation on atmospheric composition. In addition, some ecosystem models now include actual beetle infestation schemes instead of specifying mortality from data (as discussed in Arneth and Niinemets, 2010). This can lead to higher temporal variability of mortality as the beetles and mortality change during the year, rather than keeping mortality static throughout the year as we do in this study.

The robustness of differences in emissions from healthy trees and beetle-infested trees is difficult to ascertain. Here we use monoterpene-increase data from only two studies (Amin et al. 2012 lodgepole pine data and unpublished Engelmann spruce data). The large differences between the pine and spruce scenarios illustrate the large species-variability in response to beetle attack and the uncertainties that still surround the impact of beetles on atmospheric composition. Very little work has been done to quantify beetle-induced monoterpene changes and more observational studies are certainly needed to constrain how beetles affect VOC emissions both spatially and temporally. Current quantitative studies are only available for small areas in Colorado. Other regions in the western US or BC may see different effects. Duhl et al. (2012) seek to quantify the impacts of MPB on monoterpene emissions from lodgepole pine as well as changes over time; however, their results for a limited number of trees are inconclusive and appear to be dominated by tree-to-tree variability (Duhl et al., 2012). Despite this, this study

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may indicate the possibility that regions of low mortality and high mortality exhibit varying responses as well as the possibility that trees that survive a beetle attack may have an emissions scale-up effect lasting several years, while other trees that succumb quickly to the beetle may only have a scale-up effect lasting a few months. Additional research is needed to confirm these effects. There is also a clear need to study the quantitative change in monoterpene emissions due to other beetles from other tree species (MPB attacking lodgepole pine is by far the most studied combination). Although we currently use effects of MPB on lodgepole pine or spruce as an approximation for effects of other beetles in all needleleaf tree species, there is likely a widely variable response to beetle attack between species as demonstrated by the differing pine and spruce results from Amin et al. (2012). Observational studies done over longer periods of time will help discern the temporal changes in emissions from a tree under attack. The increase in emissions seen in Amin et al. (2012) may remain steady over time or it may vary widely but more studies are needed to determine this.

In recent years the importance of loading-dependent SOA yields has been discussed (e.g. Presto and Donahue, 2006). Typical total OM concentrations in the atmosphere are less than or equal to 5 $\mu g \, m^{-3}$ (Presto and Donahue, 2006). Most SOA chamber experiments are conducted with a much higher organic mass concentration than this. While a high loading can facilitate experimental analysis, it can also lead to high SOA yields that are unrealistic for the real atmosphere. Laboratory experiments at lower mass loadings provide more accurate characterization of SOA formation in atmospherically-relevant conditions (Chan et al., 2009). The Lee et al. (2006) studies, from which we take most of our bulk SOA yields, are examples of high-loading studies, and so SOA yields from this study may be overestimated. For example, OM loadings for the β-pinene ozonolysis and photooxidation experiments are $174 \pm 3 \, \mu g \, m^{-3}$

(Lee et al., 2006a) and 293 ± 4 µg m^{-3} (Lee et al., 2006b) respectively. Recently, more experiments have been done at lower, more atmospherically-relevant mass loadings focusing on ozonolysis of α-pinene (Table 6.1). These studies find that the aerosol mass produced is significantly higher for organic mass concentrations greater than 40 µg m^{-3} than for organic mass concentrations less than 10 µg m^{-3}. Mass yields of SOA from ozonolysis of α-pinene reported in the literature for OM loadings between 0-10 µg m^{-3} range from 2-18% as compared to the Lee et al. (2006a) finding of $41 \pm 2\%$ at 417 ± 5 µg m^{-3} (Table 6.1). Although these more recent studies focus only on ozonolysis of α-pinene, it is possible that the photooxidation and ozonolysis of other monoterpenes have a similar trend of lower aerosol mass yields at lower organic mass concentrations. Thus, given that mass yields at lower organic mass loading were not available for most of the monoterpene species affected by beetle kill, the studies we employed likely overestimate SOA yields by a factor of 2-10. Relative changes to SOA concentrations shown in Section 5 would not be impacted, however the absolute effect of beetle kill on PM would be muted by this effect.

Table 6.1: SOA yields formed from the ozonolysis of α-pinene within a more atmospherically-relevant range of mass loadings (0-10 µg m^{-3})

OM loading (µg m^{-3})	Mass Yield (%)	Reference
0.215-6.43	0.3-8.5	Presto and Donahue 2006
11, 16	2, 4	Chan and Seinfeld 2007
0.5-8.34	3-7	Pathak et al. 2007
13	3	Song et al. 2007
0.15-10	9-18	Shilling et al. 2008

7 Summary and Future Work

This is the first modeling study to assess the impact of pine beetle kill on monoterpene emissions and air quality. We show that at least regionally, beetle infestation may have a significant impact on atmospheric composition in western North America.

Two main effects emerge from this study – the mortality effect and the attack effect. Monoterpenes which have a large scale-up factor in response to beetle attack show the largest relative increases in emissions above baseline (β-phellandrene in the pine study and 3-carene in the spruce study). Spatially, responses to beetle attack can vary greatly with smaller-scale areas showing relatively large changes in monoterpene emissions and SOA concentrations (up to four-fold), while for western North America as a whole, the mortality effect can overcome or greatly mute the attack effect. The response to beetle attack also varies from year-to-year, depending on the number of trees under attack and the magnitude of the cumulative mortality effect.

In both the pine and spruce scenario, the MPB has the largest impact in BC in 2004 and the western US in 2008, with a much larger impact occurring in the spruce scenario. The OB have the largest impact in both BC and the western US in 2002, although the OB impact in BC is small in all years. MPB and OB combined have the largest impact in 2002 over the whole region. The MPB is by far the largest contributor to potential monoterpene and SOA changes in western North America, as it at times has an impact equivalent to the 12 beetles combined in the OB category. Although many of the large relative increases in monoterpene emissions and SOA concentrations we see across western North America are not likely observable in measurements of OM or PM2.5 due to the large annual and interannual variability in these measurements (e.g.

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in the pine scenario), in areas of spruce under attack and lower OM measurement variability, beetle-induced SOA changes may be observable.

This initial model captures the general picture of how beetles may affect monoterpenes and SOA in western North America. Representing beetle mortality in future modeling studies can be improved by using landcover datasets of specific tree species and using beetle infestation schemes rather than fixed annual mortality. Ideally, the aerosol impacts examined here should also be coupled to impacts on meteorology, ozone and fire susceptibility to attain a complete picture of beetle infestation impacts on air quality. More beetle-induced monoterpene emission studies are also needed to better quantify and constrain the impacts of beetle attack on specific monoterpenes.

The MPB outbreak in western North America is severe and still spreading. Furthermore, other regions in the world also experience large-scale herbivory attacks such as the southeast US (Duehl et al., 2011) and Europe (Seidl et al., 2011). Pine beetle attack can affect air quality by disturbing the forest carbon balance. In addition regional air quality may be degraded by increasing monoterpene emissions and resulting SOA formation in the atmosphere. Due to large potential impacts that land disturbances such as beetle attacks may have on regional monoterpene emissions and SOA formation, it is important that they be included in models.

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