

## AN ABSTRACT OF THE DISSERTATION OF

Garrett W. Meigs for the degree of Doctor of Philosophy in Forest Science presented on September 12, 2014.

Title: Mapping Disturbance Interactions from Earth and Space: Insect Effects on Tree Mortality, Fuels, and Wildfires across Forests of the Pacific Northwest.

Abstract approved:

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Given the vital role of forest ecosystems in landscape pattern and process, it is important to quantify the effects, feedbacks, and uncertainties associated with forest disturbance dynamics. In western North America, insects and wildfires are both native disturbances that have influenced forests for millennia, and both are projected to increase with anthropogenic climate change. Although there is acute concern that insect-caused tree mortality increases the likelihood or severity of subsequent wildfire, previous research has been mixed, with results often based on individual fire or insect events. Much of the ambivalence in the literature can be attributed to differences in the particular insect of interest, forest type, and fire event, but it is also related to the spatiotemporal scale of analysis and a general lack of geospatial datasets spanning enough time and space to capture multiple forest disturbances consistently and accurately.

This dissertation presents a regional-scale framework to map, quantify, and understand insect-wildfire interactions across numerous insect and fire events across the Pacific Northwest region (PNW). Through three related studies, I worked with many collaborators to develop regionally extensive but fine-grained maps to assess the spatiotemporal patterns of wildfires and the two most pervasive, damaging forest insects in the PNW – mountain pine beetle (MPB; *Dendroctonus ponderosae* Hopkins [Coleoptera: Scolytidae]; a bark beetle) and western spruce budworm (WSB; *Choristoneura freemani* Razowski [Lepidoptera: Tortricidae]; a defoliator). The proximate objectives of developing new maps and summarizing where and when insects have occurred before wildfires enable us to address the ultimate question: How does forest insect activity influence the likelihood of subsequent wildfire?

In a pilot study focused on the forest stand scale (Chapter Two), we leveraged a Landsat time series change detection algorithm (LandTrendr), annual forest health aerial detection surveys (ADS), and field measurements to investigate MPB and WSB effects on spectral trajectories, tree mortality, and fuel profiles at 38 plots in the Cascade Range of Oregon. Insect effects were evident in the Landsat time series as combinations of both short- and long-duration changes. WSB trajectories appeared to show a consistent temporal evolution of long-duration spectral decline followed by recovery, whereas MPB trajectories exhibited both short- and long-duration spectral declines and variable recovery rates. When comparing remote sensing data with field measurements of insect impacts, we found that spectral changes were related to cover-based estimates (e.g., tree basal area mortality and down coarse woody detritus). In contrast, ADS changes were related to count-based estimates (e.g., dead tree density). Fine woody detritus and forest floor depth were not well correlated with Landsat- or aerial survey-based change metrics. This study demonstrated the utility of insect mapping methods that capture a wide range of spectral trajectories, setting the stage for regional-scale mapping and analysis.

In a regional assessment of MPB and WSB effects on tree mortality (Chapter Three), we developed Landsat-based insect maps and presented comparisons across space, time, and insect agents that have not been possible to date, complementing existing ADS maps by: (1) quantifying change in terms of field-measured tree mortality; (2) providing consistent estimates of change for multiple agents, particularly long-duration changes; (3) capturing variation of insect impacts at a finer spatial scale within ADS polygons, substantially reducing estimated insect extent. Despite high variation across the study region, spatiotemporal patterns were evident in both the ADS- and Landsat-based maps of insect activity. MPB outbreaks occurred in two phases – first during the 1970s and 1980s in eastern and central Oregon and then more synchronously during the 2000s throughout the dry interior conifer forests of the PNW. Reflecting differences in habitat susceptibility and epidemiology, WSB outbreaks exhibited early activity in northern Washington and an apparent spread from the eastern to central PNW during the 1980s, returning to northern Washington during the 1990s and 2000s. Across the region, WSB exceeded MPB in extent and tree mortality impacts in all ecoregions except for one, suggesting that ongoing studies should account for both bark beetles and defoliators, particularly given recent and projected increases in wildfire extent.

By combining these insect maps with an independent wildfire database (Chapter Four), we investigated wildfire likelihood following recent MPB and WSB outbreaks at ecoregional and regional scales. We computed wildfire likelihood with two-way binary matrices between fire and insects, testing for paired differences between percent burned with and without prior insect activity. All three disturbance agents occurred primarily in the drier, interior conifer forests east of the Cascade Range, with recent wildfires extending through the southern West Cascades and Klamath Mountains. In general, insect extent exceeded wildfire extent, and each disturbance typically affected less than 2% annually of a given ecoregion. In recent decades across the PNW, wildfire likelihood is not consistently higher in forests with prior insect outbreaks, but there is evidence of linked interactions that vary across insect agent (MPB and WSB), space (ecoregions), and time (interval since insect onset). For example, fire likelihood is higher following MPB activity in the North Cascades and West Cascades, particularly within the past 10 years, whereas fire likelihood is lower at various time lags following MPB in the Northern Rockies, East Cascades, and Blue Mountains. In contrast, fire likelihood is lower following WSB outbreaks at multiple time lags across all ecoregions. In addition, there are no consistent relationships between insect-fire likelihood and interannual fire extent, suggesting that other factors (such as climate) control the disproportionately large fire years accounting for the majority of regional fire extent. Although insects and wildfires do not appear to overlap enough to facilitate consistently positive linked disturbance interactions, specific fire events and years – such as 2003 and 2006 in the North Cascades – demonstrate high insect-fire co-occurrence and potential compound disturbance effects at the landscape scale.

The results from this dissertation highlight the key ecological roles that native disturbances play in PNW forests. WSB, MPB, and wildfire have been relatively rare at the regional scale, but all three have had and will continue to have profound effects on particular forest stands and landscapes. Because scale is such an important aspect of both the disturbance phenomena themselves as well as our ability to detect the ecological changes they render, our results also underscore the importance of geospatial datasets that span multiple scales in space and time. Given concerns about forest health in a rapidly changing climate, long-term monitoring will enable forest managers to quantify and anticipate the independent and interactive effects of insects, wildfires, and other disturbances.

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Mapping Disturbance Interactions from Earth and Space:  
Insect Effects on Tree Mortality, Fuels, and Wildfires across Forests of the Pacific Northwest

by  
Garrett W. Meigs

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APPROVED:

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I understand that my dissertation will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my dissertation to any reader upon request.

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Garrett W. Meigs, Author

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## CONTRIBUTION OF AUTHORS

Robert E. Kennedy assisted with study design, data analysis, and writing of Chapters 2 and 3. Warren B. Cohen assisted with writing of Chapter 2. Andrew N. Gray and Matthew J. Gregory assisted with data analysis and writing of Chapter 3. John L. Campbell and Harold S. J. Zald assisted with study design, data analysis and writing of Chapter 4. John D. Bailey, Tiffany S. Garcia, Robert E. Kennedy, and David C. Shaw assisted with writing of Chapter 4. A manuscript version of Chapter 2 was published in *Remote Sensing of Environment* in 2011. Manuscripts associated with Chapters 3 and 4 were in the process of peer review at the time this thesis was submitted.

# TABLE OF CONTENTS

	<u>Page</u>
<b>CHAPTER 1    INTRODUCTION.....</b>	<b>1</b>
<b>CHAPTER 2    A LANDSAT TIME SERIES APPROACH TO CHARACTERIZE BARK BEETLE AND DEFOLIATOR IMPACTS ON TREE MORTALITY AND SURFACE FUELS IN CONIFER FORESTS.....</b>	<b>3</b>
ABSTRACT.....	4
INTRODUCTION .....	6
METHODS .....	9
RESULTS AND DISCUSSION.....	14
SUMMARY.....	22
REFERENCES .....	24
FIGURES AND TABLES .....	28
<b>CHAPTER 3    SPATIOTEMPORAL DYNAMICS OF RECENT MOUNTAIN PINE BEETLE AND WESTERN SPRUCE BUDWORM OUTBREAKS ACROSS THE PACIFIC NORTHWEST REGION.....</b>	<b>36</b>
ABSTRACT.....	36
INTRODUCTION .....	38
METHODS .....	41
RESULTS .....	48
DISCUSSION.....	52
REFERENCES .....	58
FIGURES.....	62
<b>CHAPTER 4    DO INSECT OUTBREAKS INCREASE WILDFIRE LIKELIHOOD IN CONIFER FORESTS? .....</b>	<b>74</b>
ABSTRACT.....	74
INTRODUCTION .....	76
METHODS .....	79
RESULTS .....	84
DISCUSSION.....	87
REFERENCES .....	95
FIGURES AND TABLES .....	100
<b>CHAPTER 5    CONCLUSION .....</b>	<b>112</b>
SYNTHESIS.....	112
BROADER IMPLICATIONS .....	114
<b>BIBLIOGRAPHY .....</b>	<b>116</b>

## LIST OF FIGURES

<u>Figure</u>	<u>Page</u>
<b>Fig. 2.1.</b> Study area locations within the Pacific Northwest Region .....	28
<b>Fig. 2.2.</b> Aerial detection survey and Landsat-based maps of forest disturbance .....	29
<b>Fig. 2.3.</b> Example LandTrendr spectral trajectories and plot photographs.....	30
<b>Fig. 2.4.</b> Relationships of field-measured tree mortality with LandTrendr disturbance and aerial detection survey insect maps .....	31
<b>Fig. 2.5.</b> Relationships of field-measured surface fuels with aerial detection survey insect maps and LandTrendr disturbance maps.....	32
<b>Fig. 2.A1.</b> Field plot sampling design.....	34
<b>Fig. 3.1.</b> Study area map and recent insect dynamics across the PNW .....	62
<b>Fig. 3.2.</b> Datasets and processing steps for Objective 1 .....	63
<b>Fig. 3.3.</b> Plot-scale relationship between LandTrendr-based spectral change and field-based dead tree basal area change .....	64
<b>Fig. 3.4.</b> Example landscape maps comparing cumulative impacts of mountain pine beetle and western spruce budworm according to aerial detection surveys and LandTrendr thresholds .....	65
<b>Fig. 3.5.</b> Difference of onset year and count among aerial detection survey and LandTrendr maps of MPB and WSB .....	66
<b>Fig. 3.6.</b> Temporal patterns of extent of recent MPB and WSB activity according to aerial detection survey and LandTrendr change detection across forested ecoregions of the PNW .....	67
<b>Fig. 3.7.</b> Spatial patterns of cumulative insect effects on tree mortality (dead tree basal area; 1984-2012) from LT high threshold maps across the PNW.....	68
<b>Fig. 3.8.</b> Temporal patterns of insect effects on tree mortality (dead tree basal area) from LandTrendr change detection across forested ecoregions of the PNW .....	69
<b>Fig. 3.A1.</b> Map of stratified random sample plots in seven Landsat path/row scenes with MPB or WSB activity according to aerial detection surveys across the PNW .....	70
<b>Fig. 3.B1.</b> Example of thresholds for simple attribution of presumed ‘insect’ and ‘noninsect’ LandTrendr disturbance segments intersected by aerial survey polygons.....	71

## LIST OF FIGURES (Continued)

<u>Figure</u>	<u>Page</u>
<b>Fig. 3.C1.</b> Cumulative extent of recent MPB and WSB activity according to aerial detection survey and LandTrendr change detection across forested ecoregions of the PNW .....	72
<b>Fig. 3.D1.</b> Temporal patterns of cumulative insect effects on tree mortality (dead tree basal area) from LandTrendr change detection across forested ecoregions of the PNW .....	73

## LIST OF TABLES

<u>Table</u>	<u>Page</u>
<b>Table 2.1.</b> Spectral trajectories associated with insect activity, stable conditions, and growth... 33	
<b>Table 2.A1.</b> Statistical relationships among remote sensing and fuels variables..... 35	
<b>Table 4.1.</b> Insect-fire likelihood across forested ecoregions affected by recent insect activity. 109	
<b>Table 4.2.</b> Statistical summary of paired differences in insect-fire likelihood across forested ecoregions affected by mountain pine beetle activity at different time lags..... 110	
<b>Table 4.3.</b> Statistical summary of paired differences in insect-fire likelihood across forested ecoregions affected by western spruce budworm activity at different time lags ..... 111	

## PREFACE

*Everything changes and nothing stands still.*

-Heraclitus

## CHAPTER 1 || INTRODUCTION

Life is defined by change, and disturbance ecology is the study of how ecosystems change across time and space. Given the vital role of forests in ecosystem composition, structure, and function, it is important to quantify the effects, feedbacks, and uncertainties associated with forest disturbances (Running 2008). In western North America, wildfire and insect outbreaks are the two most prevalent natural disturbance processes. Both fire and insects have increased in recent decades, resulting in episodes of relatively sudden tree death, and both are expected to accelerate with climate change (e.g., Westerling et al. 2006, Raffa et al. 2008, Bentz et al. 2010, Littell et al. 2010), potentially destabilizing terrestrial carbon storage (Kurz et al. 2008). Because fires sometimes follow insect outbreaks, there is acute concern that insect-caused tree mortality results in fuel conditions conducive to extreme, fuel-driven fire behavior, which in turn could increase fire likelihood or severity (Negrón et al. 2008, Hicke et al. 2012). These potential disturbance interactions represent a crucial frontier for ecosystem research and management.

Studies to date have produced mixed results; insect effects on fuels and fire appear to vary widely across different systems and scales (e.g., Fleming et al. 2002, Hummel and Agee 2003, Negrón et al. 2008, Simard et al. 2009, Hicke et al. 2012). Most previous research has assessed insect-fire interactions with simulation models at the stand scale (e.g., Simard et al. 2011) or empirical observations of a small number of insect or wildfire events at stand to landscape scales (e.g., Kulakowski and Veblen 2007, but see Fleming et al. 2002, Lynch and Moorcroft 2008). Here, I suggest that neither approach is sufficient to characterize possible interactions. Because existing fire behavior models were constructed with limited reference to insect-altered fuels, it is unlikely that they contain appropriate mechanisms to model such effects. Further, because fire is an inherently stochastic, site-specific process, it is difficult to generalize the causes and effects of any particular fire event. A regional-scale approach is necessary for robust inference to general system behavior. Moreover, although much of the ambivalence in the literature can be attributed to differences in the particular insect of interest, forest type, and fire event, it is also related to the spatiotemporal scale of analysis and a general

lack of geospatial datasets spanning enough time and space to capture multiple forest disturbances consistently and accurately.

This dissertation presents a regional-scale framework to map, quantify, and understand insect-wildfire interactions across numerous insect and fire events across the Pacific Northwest region (PNW). This project develops and presents regionally extensive but fine-grained maps (30-m) to assess the spatiotemporal patterns of wildfires and the two most pervasive, potentially damaging forest insects in the PNW – mountain pine beetle (MPB; *Dendroctonus ponderosae* Hopkins [Coleoptera: Scolytidae]; a bark beetle) and western spruce budworm (WSB; *Choristoneura freemani* Razowski [Lepidoptera: Tortricidae]; a defoliator). The proximate objectives of developing new maps and summarizing where and when insects have occurred before wildfires enable us to address the ultimate question: How does forest insect activity influence the likelihood of subsequent wildfire?

In the chapters that follow, I present three related studies that build from stand-scale observations and remote sensing responses associated with MPB and WSB damage (Chapter 2) to ecoregional- and regional-scale maps and comparisons of WSB and WSB activity over the past few decades (Chapter 3) and finally to an ecoregional- and regional-scale assessment of fire likelihood in forests with and without prior MPB and WSB outbreaks at various time lags. I generally focus on the forested ecoregions (Omernik 1987) where MPB and WSB have been most active since 1970, predominantly in the interior PNW mixed-conifer forests where mixed-severity wildfire is also native (Hessburg et al. 2000). I use a variety of geospatial and statistical methods to develop the core maps, leveraging them to describe the recent natural disturbance history of these forests with a consistent breadth and depth that has not been available in prior studies.

Despite these advances, many uncertainties remain, as detailed at the end of each chapter. I conclude the dissertation by synthesizing these chapters in the context of emergent themes and broader forest health implications (Chapter 5).

**CHAPTER 2 || A LANDSAT TIME SERIES APPROACH TO CHARACTERIZE BARK  
BEETLE AND DEFOLIATOR IMPACTS ON TREE MORTALITY AND SURFACE FUELS IN  
CONIFER FORESTS**

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## ABSTRACT

Insects are important forest disturbance agents, and mapping their effects on tree mortality and surface fuels represents a critical research challenge. Although various remote sensing approaches have been developed to monitor insect impacts, most studies have focused on single insect agents or single locations and have not related observed changes to ground-based measurements. This study presents a remote sensing framework to (1) characterize spectral trajectories associated with insect activity of varying duration and severity and (2) relate those trajectories to ground-based measurements of tree mortality and surface fuels in the Cascade Range, Oregon, USA. We leverage a Landsat time series change detection algorithm (LandTrendr), annual forest health aerial detection surveys (ADS), and field measurements to investigate two study landscapes broadly applicable to conifer forests and dominant insect agents of western North America. We distributed 38 plots across multiple forest types (ranging from mesic mixed-conifer to xeric lodgepole pine) and insect agents (defoliator [western spruce budworm] and bark beetle [mountain pine beetle]). Insect effects were evident in the Landsat time series as combinations of both short- and long-duration changes in the Normalized Burn Ratio spectral index. Western spruce budworm trajectories appeared to show a consistent temporal evolution of long-duration spectral decline (loss of vegetation) followed by recovery, whereas mountain pine beetle plots exhibited both short- and long-duration spectral declines and variable recovery rates. Although temporally variable, insect-affected stands generally conformed to four spectral trajectories: short-duration decline, short- then long-duration decline, long-duration decline, long-duration decline then recovery. When comparing remote sensing data with field measurements of insect impacts, we found that spectral changes were related to cover-based estimates (tree basal area mortality [ $R^2_{\text{adj}} = 0.40$ ,  $F_{1,34} = 24.76$ ,  $P < 0.0001$ ] and down coarse woody detritus [ $R^2_{\text{adj}} = 0.29$ ,  $F_{1,32} = 14.72$ ,  $P = 0.0006$ ]). In contrast, ADS changes were related to count-based estimates (e.g., ADS mortality from mountain pine beetle positively correlated with ground-based counts [ $R^2_{\text{adj}} = 0.37$ ,  $F_{1,22} = 14.71$ ,  $P = 0.0009$ ]). Fine woody detritus and forest floor depth were not well correlated with Landsat- or aerial survey-based change metrics. By characterizing several distinct temporal manifestations of insect activity in conifer forests, this study demonstrates the utility of insect mapping methods that capture a wide

range of spectral trajectories. This study also confirms the key role that satellite imagery can play in understanding the interactions among insects, fuels, and wildfire.

#### KEYWORDS

bark beetle; change detection; defoliator; fire; fuel, insect disturbance; Landsat time series; mountain pine beetle; Pacific Northwest; spectral trajectory; western spruce budworm

## INTRODUCTION

In western North America, native defoliators and bark beetles cause pervasive, regional-scale tree mortality, a profound ecosystem impact that varies from year to year but may be increasing (Swetnam and Lynch 1993, Raffa et al. 2008). Tree-killing insects influence forest structure and function directly, and they also may influence other forest disturbances, particularly wildfire. Abiotic and biotic factors, including drought, tree condition (e.g., density, age, vigor), and landscape contiguity, render many forests susceptible to both insect and wildfire disturbances. This co-occurrence of insects and wildfire, coupled with recent and predicted increases in both disturbances due to climate change (Logan et al. 2003, Westerling et al. 2006, Kurz et al. 2008, Raffa et al. 2008, Balshi et al. 2009, Bentz et al. 2010, Littell et al. 2010), raise concerns that insect-caused tree mortality may increase the likelihood of extreme fire behavior and may amplify disturbance interactions (Geiszler et al. 1980, Negron et al. 2008, Simard et al. 2011). To date, however, studies indicate that insect effects on tree mortality, fuels, and fire can diverge widely, depending on forest type, insect type, time since outbreak, vegetation response, fire weather, and fire suppression (e.g., Fleming et al. 2002, Hummel and Agee 2003, Lynch and Moorcroft 2008, Page and Jenkins 2007b, Simard et al. 2011). A key impediment to better understanding insect-fire interactions is the lack of empirical maps that track both insects and fire consistently over space and time. Although mapping fire with remote sensing approaches is well-established (e.g., Eidenshink et al. 2007), the science of mapping insect effects is relatively nascent.

Previous remote sensing studies have employed a variety of datasets and approaches to map insect disturbance. At broad spatial extents, MODIS-based algorithms are emerging for the detection of insect effects (e.g., Verbesselt et al. 2009), but for many resource-based goals, finer-grained data are required (Cohen and Goward 2004). Numerous studies have targeted bark beetle outbreaks (mountain pine beetle [*Dendroctonus ponderosae* Hopkins]) in British Columbia (e.g., Coops et al. 2006, Goodwin et al. 2008, Goodwin et al. 2010) and the U.S. Rocky Mountains (e.g., Lynch et al. 2006, Dennison et al. 2010), using diverse remote sensing datasets, including aerial sketch mapping and photography, Hyperion, Quickbird, GeoEye-1, ASTER, and Landsat (Wulder et al. 2006, Dennison et al. 2010). Many approaches conceptualize insect outbreak stages as relatively abrupt, short-term anomalies from stable forest

conditions. Accordingly, a time series of spectral values (hereafter, the ‘spectral trajectory’) is expected to demonstrate a period of relative spectral stability followed by an abrupt change in spectral value. For example, Goodwin et al. (2008) used a decision tree approach with a temporal sequence of Landsat imagery to identify abrupt changes in the Normalized Difference Moisture Index associated with the onset of mountain pine beetle outbreaks (red attack stage) in north-central British Columbia, achieving > 70% accuracy. Yet, insects do not consistently cause abrupt changes in spectral trajectories. Vogelmann et al. (2009) observed relatively long-term, gradual declines in vegetation-based spectral indices on water-stressed slopes in New Mexico, attributing the spectral trajectories to defoliation by western spruce budworm (*Choristoneura occidentalis* Freeman). Indeed, Goodwin et al. (2010) further examined mountain pine beetle effects in British Columbia, developing mixed linear models to characterize the spectral variability occurring over several years of progressive beetle attack. Thus, recent studies suggest that the temporal dynamics of insect disturbance—and the associated spectral trajectories—can vary considerably. To map defoliator and bark beetle impacts consistently across large geographic areas, remote sensing algorithms must characterize the full variation of spectral trajectories.

In addition, to be useful in understanding potential insect and fire interactions, insect mapping must move toward quantifying tree mortality and how that mortality relates to fuel dynamics. Unlike timber harvest, which removes vegetation from a site, insect-related mortality transforms the condition and arrangement of plant material (i.e., fuels) in place. In conifer forests of western North America, bark beetle outbreaks can induce relatively rapid tree mortality and associated foliage color change from green to red, lagged shedding of foliage, bark, and branches, and eventual tree fall (e.g., Page and Jenkins 2007a, Klutsch et al. 2009, Simard et al. 2011). When western spruce budworm defoliation persists for several consecutive years, it also can result in widespread tree mortality (often in conjunction with bark beetles, Hummel and Agee 2003, Goheen and Willhite 2006, Vogelmann et al. 2009). This insect-caused tree mortality alters canopy fuel moisture and eventually transfers fuels from canopy to surface strata, namely forest floor (litter and duff), fine woody detritus, and down coarse woody detritus (Brown et al. 1982). As canopy, ladder, and surface fuel distributions change, potential fire behavior and fire effects shift accordingly (e.g., Page and Jenkins 2007b). Empirical testing of

this interaction, however, requires pre-fire maps distinguishing insect-caused fuel accumulations. It is unclear from existing remote sensing studies how insect-induced spectral trajectories are related to key biophysical drivers on the ground.

To improve generalized mapping of insect-related effects in forests, this study integrates a Landsat time series change detection algorithm (LandTrendr; Kennedy et al. 2010), annual forest health aerial detection surveys, and ground-based measurements to investigate cumulative insect impacts on tree mortality and surface fuels in conifer forests. The overall goal is to advance a remote sensing framework to capture the effects of multiple insect agents consistently across variable spatial and temporal scales. In this paper, we demonstrate the potential of our approach with a pilot study in conifer forests of the Cascade Range, Oregon, USA. Our specific objectives are to:

1. Characterize spectral trajectories in Landsat time series associated with defoliator and bark beetle disturbances of varying duration and severity;
2. Relate spectral trajectories to ground-based measurements of insect-caused tree mortality and surface fuels to assess biophysical drivers of spectral change.

The Cascade Range is an ideal region to explore the variability of insect impacts during the Landsat era. Ongoing research leverages dense Landsat time series to assess forest disturbance and recovery processes in the Pacific Northwest Region, including insect activity (e.g., Cohen et al. 2010, Kennedy et al. 2010). A complementary regional dataset is the Cooperative Aerial Detection Survey (ADS), where human observers identify a wide variety of disturbances across forested lands of Oregon and Washington annually (K. Sprengel, USDA Forest Service, personal communication). Coinciding with the digitized ADS record (since 1980) and the launch of the TM sensor on Landsat 5 (1984), mountain pine beetle and western spruce budworm outbreaks have affected dry forest landscapes across much of the region, particularly in the eastern Cascade Range of Oregon. Previous studies have characterized endemic and outbreak levels of insect populations in relatively productive mixed-conifer forests and less productive lodgepole pine (*Pinus contorta* Douglas ex Louden; e.g., Geiszler et al. 1980, Franklin et al. 1995, Hummel and Agee 2003).

## METHODS

### *Analytical approach and study areas*

Our analysis employed a Landsat time series change detection algorithm (LandTrendr; Kennedy et al. 2010; see below), annual forest health aerial detection surveys (ADS), and field observations to investigate recent insect activity in the eastern Cascade Range, Oregon. Using maps from LandTrendr and ADS as guides, we distributed 38 survey plots across a range of forest conditions (forest type, insect agent type [bark beetle vs. defoliator] and severity, and time since onset of insect outbreak). We then measured tree mortality and surface fuels at these plots to evaluate and interpret the remotely derived disturbance maps.

We focused on two landscapes that have experienced contrasting insect activity according to the ADS (Figs. 2.1, 2.2). The generally higher productivity, mixed-conifer forests in the Mt. Hood Zone experienced widespread defoliation by western spruce budworm (WSB), beginning in the mid-1980s and peaking in the early 1990s, followed by sporadic and locally intense mountain pine beetle (MPB) activity in the early 2000s. In contrast, lower productivity lodgepole pine forests in the Cascade Lakes Zone experienced minimal WSB activity due to the lack of suitable host trees but some of the highest cumulative mortality from MPB in the region beginning in the early 1980s.

Both landscapes are defined by steep environmental gradients. In the Mt. Hood Zone, dominant tree species are Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), grand fir (*Abies grandis* [Douglas ex D. Don] Lindl.), western larch (*Larix occidentalis* Nutt.), ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & C. Lawson), and occasional lodgepole pine. Study plot elevation ranges from 1000 to 1600 m. Averaged across study plots, the 32-year (1978-2009) mean annual precipitation is 2250 mm (SD = 475 mm) and mean annual temperature is 5.8 °C (SD = 1.0 °C) (Daly et al. 2002; PRISM Group, Oregon St. Univ., <http://prism.oregonstate.edu>). In the Cascade Lakes Zone, dominant tree species at lower elevations are lodgepole pine and ponderosa pine, which intergrade at higher elevations with mixed-conifer species, particularly *Abies spp.*, mountain hemlock (*Tsuga mertensiana* [Bong.] Carrière) and whitebark pine (*Pinus albicaulis* Engelm.). Study plot elevation ranges from 1200 to 2100 m, and 32-year mean annual precipitation and temperature are 895 mm (SD = 525 mm) and 5.7 °C (SD = 0.8 °C), respectively (<http://prism.oregonstate.edu>). Both study landscapes

experience warm, dry summers (Mediterranean climate type), and soils are volcanic in origin (andisols).

### ***LandTrendr disturbance mapping***

We developed maps of forest disturbance and growth using outputs from LandTrendr algorithms and analysis, which are described in detail by Kennedy et al. (2010). Briefly, we acquired georectified annual Landsat TM/ETM+ images from the USGS Landsat archive for two Landsat scenes that covered our study areas (path 45/ rows 29 and 30; Fig. 2.1) from 1984 to 2007, with multiple images used in years when clouds obscured part of the study area. We corrected a single image to approximate surface reflectance using the COST approach (Chavez 1996) normalized all other images in the same path/row to that reference image using the MADCAL relative radiometric normalization (Canty et al. 2004), and conducted cloud screening manually. For this temporal stack of normalized Landsat imagery, we then calculated the spectral index referred to as the Normalized Burn Ratio (NBR; Key and Benson 2006), which contrasts Landsat bands four and seven (near infrared [NIR] and shortwave infrared [SWIR], respectively):

$$\text{NBR} = (\text{Band 4} - \text{Band 7}) / (\text{Band 4} + \text{Band 7}) \quad (1)$$

In its contrast between the NIR and SWIR bands, NBR is similar to spectral indices used in other studies to track insect effects (e.g., Wulder et al. 2006, Goodwin et al. 2008, Vogelmann et al. 2009, Goodwin et al. 2010). In our study region, NBR is particularly applicable because sensitivity analyses have shown its utility for capturing disturbance processes relative to other indices (Kennedy et al. 2010, Cohen et al. 2010) and because of its familiarity to the fire science community (Eidenshink et al. 2007).

Following preparation of the Landsat image stacks, we applied LandTrendr temporal segmentation algorithms to the time series of NBR values at each pixel. Segmentation distills an often-noisy yearly time series into a simplified series of segments to capture the salient features of the trajectory while avoiding most false changes (Kennedy et al. 2010). We derived disturbance maps at the Landsat pixel scale using key characteristics of these segments. Disturbance segments were defined as those segments experiencing a decline in NBR over time (Cohen et al. 2010, Kennedy et al. 2010). To minimize commission errors due to background

variation and statistical noise, we only accepted segments whose change in NBR exceeded a threshold value of 50 units ( $\text{NBR} * 1000$ ), adjusted to higher values for shorter duration segments. For each disturbance segment, we mapped the absolute change in NBR from start to finish as an estimate of disturbance magnitude. We also mapped the onset year and duration of the segment (in years). We classified the disturbance segments according to whether they were associated with a long-duration decline (defined here as lasting six or more years) or short-duration decline (less than six years). For those segments associated with long-duration trends, we also identified those with evidence of post-disturbance vegetation growth (positive change in NBR) lasting three or more years. These maps were used first to aid in locating field plots (see section 2.4) and later to characterize disturbance at each field plot (see section 2.6).

Strictly for the purposes of display (Fig. 2.2), we derived filtered maps to emphasize disturbances likely caused by insects and to de-emphasize those likely caused by other agents (e.g., logging, fire). The filtering process included a combination of classification algorithms based on training data acquired by trained interpreters, followed by manual removal of overlapping fire perimeters (based on maps from <http://mtbs.gov>) and logging (based on shape and NBR magnitude). The filtered maps were not intended as a final mapping product but rather to show the potential of Landsat time series data to map insect impacts.

### ***Aerial detection survey mapping***

Since the mid-twentieth century, state and federal agencies have conducted annual aerial surveys of forested lands in Oregon and Washington (USDA Forest Service Region 6). Human observers identify a wide variety of biotic and abiotic forest disturbances, including insect and disease impacts on specific host tree species (Ciesla 2006, K. Sprengel, USDA Forest Service, personal communication). Observations are recorded as thematic polygons, digitized versions of which are available since 1980 (<http://www.fs.fed.us/r6/nr/fid/as>). We used this multiple decade record to identify insect agents associated with LandTrendr disturbance segments.

We also used the ADS record to extract cumulative insect effects for comparison with LandTrendr outputs and field data, enabling us to assess the robustness of the ADS dataset. We converted all ADS data from 1980 to 2009 into raster format (30 m grain size) and used GIS-type queries in the IDL programming language (<http://www.itvis.com>) across years in each cell

to identify the onset and duration of mountain pine beetle and western spruce budworm. We tallied the total magnitude of each agent's effects, using the quantitative trees per acre count for MPB (converted to trees per hectare) and the qualitative defoliation severity estimate for WSB. The WSB severity data were recorded on a numeric scale (1-4) from 1994-1998 and a thematic scale (low, medium, high, and very high) for all other years. We converted the thematic scale to a standard numeric scale (low to 1, medium to 2, high and very high to 3) and summed the severity estimates across all years. Additionally, to derive a common quantitative metric across all plots, we calculated the cumulative duration of both agents observed at each plot.

### ***Field plot selection***

With limited resources for field surveys and with a study focus on description rather than prediction, we used a purposive sampling approach designed to observe quickly a range of conditions across the study zones. Using LandTrendr and ADS maps as guides, we established 38 field plots in September-November 2009 (14 plots in the Mt. Hood Zone, 24 plots in the Cascade Lakes Zone; Figs. 2.1, 2.2). Because one of our objectives was to describe spectral trajectories associated with insect disturbance and recovery processes, we excluded stands with evidence of substantial anthropogenic activity (e.g., salvage harvest, thinning) or fire occurrence since 1984 (verified via GIS data [<http://mtbs.gov>]). To reduce backcountry travel time, we limited sample points to distances within 1000 m of roads and trails after determining that pixels closer to roads spanned the same range of remotely sensed insect impacts as all pixels across the study landscapes.

### ***Field measurements of tree mortality, surface fuels, and landcover***

We designed plot-level measurements as a rapid assessment of tree mortality and surface fuel distributions within a single Landsat pixel (30 \* 30 m). At each plot, we quantified live and dead trees in four circular subplots located 14 m from plot center in the sub-cardinal directions (NE, SE, SW, NW; Fig. 2.A1). At each subplot, we used variable-radius subplots (prism sweep; basal area factor 10) to estimate live and dead tree basal area and fixed-radius subplots (9 m radius) to estimate live and dead trees per hectare. For direct comparison with aerial detection survey estimates of cumulative tree mortality, we tallied trees in three height strata (dominant,

codominant, understory), presenting here the estimates from dominant and codominant strata only. Within the fixed-radius subplots, we also tallied all dead, down trees that were likely rooted within the 9 m radius before being killed by recent insect outbreaks. Although we did not attempt to date the death of individual trees, we classified each down tree based on our confidence that the tree died during the time period identified by ADS data (three confidence levels—90%, 75%, 50%—based on their decay class and evidence of insect disturbance [i.e., bark beetle galleries]). In this paper, we present a conservative estimate of down, dead trees, including only the highest confidence level ( $n = 1409/1574$ ; 90% of sampled down trees ).

We measured woody surface fuels along line intercept transects (Brown 1974, Brown et al. 1982, Harmon and Sexton 1996) originating at plot center and extending 21 m in the sub-cardinal directions (Fig. 2.A1). For down coarse woody detritus (CWD; 1000 hour fuels; all woody pieces  $\geq 7.62$  cm diameter), we recorded species, diameter (cm), decay class (1-5), and char class (0: no char, 1: bark char; 2: wood char) along the full transect length (84 m). For fine woody detritus (FWD), we recorded time lapse-based size class (1 hour:  $< 0.65$  cm; 10 hour: 0.65–2.54 cm; 100 hour: 2.55–7.62 cm), decay class, and char class along size class-specific segments (1, 10, 100 hour fuels along 12, 24, 84 m, respectively). We converted line intercept counts to volume per unit area using standard equations after Harmon and Sexton (1996) and estimated total CWD and FWD mass with decay class-, species-, and ecoregion-specific wood density values (Hudiburg et al. 2009, Meigs et al. 2009). We sampled ground layer fuels by measuring litter and duff depth at two points along each transect (8 total).

To capture potential insect effects on live overstory and understory fuels at each fixed-area subplot, we completed a standard landcover classification with ocular estimates of percent cover for each of the following classes: live and dead needleleaf tree, live and dead broadleaf tree, shrub, herb, light litter/duff, dark litter/duff, rock/soil. To aid in imagery interpretation, we collected six photographs from each subplot center (24 total per plot).

### ***Data analysis***

We scaled all field measurements to per-unit-area values for comparison among insect agents, forest types, and study areas. At each plot location, we extracted via GIS the onset, duration, and magnitude of change from LandTrendr and ADS maps at the co-occurring pixel

(applying no spatial filter). We compared the remotely sensed disturbance magnitude values with our field measurements of tree mortality and surface fuels. We calculated tree basal area percent mortality from the variable-radius plot data, recognizing that this metric would provide an integrated measure of vegetation change for standing trees only. Because the ADS sketch maps identify trees per unit area affected by various mortality agents, we used the fixed-radius overstory dead tree counts to calculate an analogous field-based metric. We related field measurements to remotely sensed indices with simple linear regression (lm procedure; R Development Core Team 2011). Where appropriate, we subsetted the dataset by study zone (e.g., WSB only prevalent in Mt. Hood Zone) and excluded outlier plots that were not from the sample population of interest in specific statistical comparisons (e.g., old forest plots with high levels of down coarse woody detritus not associated with recent changes). We assessed these relationships with  $R^2$ , adjusted  $R^2$  ( $R^2_{adj}$ ),  $P$  values, and regression coefficients.

## RESULTS AND DISCUSSION

### *Spectral trajectories of bark beetles and defoliators*

Because previous studies have identified diverse, seemingly disparate temporal signals associated with insect disturbance (e.g., Goodwin et al. 2008, Vogelmann et al. 2009, Goodwin et al. 2010), we sought to advance a remote sensing framework to capture a variety of spectral trajectories (i.e., temporal trajectories of spectral response to insect activity) objectively and consistently. Across both study zones, the LandTrendr algorithm detected several spectral trajectories at insect disturbance areas identified by aerial detection survey sketch maps. In general, insect effects were evident in the Landsat time series as combinations of both short- and long-duration spectral change. In fact, the majority of plots exhibited long-duration declines ( $\geq 6$  years) in the NBR index, corresponding to relatively slow disturbance processes at an annual time step (Table 2.1). Although study plots were distributed across a broad range of conditions (tree structure and composition, site productivity, insect agent and severity), each plot affected by insect disturbance conformed to one of four generalized spectral trajectories: short-duration decline, short- then long-duration decline, long-duration decline, long-duration decline then recovery (Table 2.1). Two additional spectral trajectories not associated with insect mortality—

stable and growth—occurred frequently across the study landscapes. Together, these six trajectories represent a conceptual basis for interpreting the most important sequences of change across these dynamic landscapes.

Stands affected by mountain pine beetle (MPB) did not exhibit a single diagnostic disturbance trajectory. Some MPB plots exhibited short-term declines in NBR (e.g., Fig. 2.3A), but these mortality “events” were often followed by continued declines, suggesting an initial spike in mortality followed by subsequent mortality and dead tree fall. Other MPB plots showed gradually declining NBR (e.g., Fig. 2.3B), which was sometimes followed by spectral recovery. In general, these plots exhibited relatively lower proportional tree mortality, more robust lodgepole pine regeneration, or higher abundance of non-host species (e.g., *Abies spp.*, *Tsuga mertensiana*) than plots with more abrupt spectral declines. Our findings of both short- and long-duration vegetation decline associated with MPB are consistent with previous studies (e.g., Wulder et al. 2006, Goodwin et al. 2008, Goodwin et al. 2010) and highlight the diverse effects of bark beetles on forest cover and structure.

In contrast to the high variability in spectral response among mountain pine beetle plots, stands affected by western spruce budworm (WSB) defoliation consistently showed long-duration spectral declines, typically followed by relatively strong spectral recovery (e.g., Fig. 2.3C). WSB plots occurred only in the Mt. Hood Zone, a landscape characterized by relatively productive mixed-conifer forest compared to the Cascade Lakes Zone. We suggest that the long-duration decline and recovery signal is a diagnostic spectral trajectory for WSB in these mixed-conifer forests, where partial overstory tree survival and understory vegetation growth result in relatively rapid re-greening of defoliated plots. Our observations of gradual spectral changes associated with WSB support the findings of Vogelmann et al. (2009), who documented gradually increasing SWIR/NIR reflectance in New Mexico associated with WSB damage in spruce-fir forests, although they did not have temporal coverage of post-outbreak years that might indicate a vegetation recovery signal.

The observed differences between MPB trajectories (variable combinations of both short- and long-duration disturbance) and WSB trajectories (consistently long-duration disturbance) reflect the distinct biological effects of these two types of insect. Our results show that bark beetles (MPB) can indeed induce rapid forest changes but that their pixel-scale impacts typically

evolve over several years. By definition, WSB defoliation takes several consecutive years to yield substantial vegetation loss, often in conjunction with tree-killing bark beetles in late stages of defoliation (Hummel and Agee 2003, Goheen and Willhite 2006). Despite these cumulative impacts, our observation of rapid spectral recovery at WSB sites suggests relatively transient defoliator impacts in these mixed-conifer forests. These results demonstrate that no single model of temporal change can capture the full range of insect-induced changes across heterogeneous landscapes. Instead, remote sensing approaches require: (1) the capacity to detect the full variety of short- and long-duration changes, including both spectral losses and gains; (2) attribution datasets such as ADS and field observations.

### ***Spectral trajectory and aerial survey relationships with insect-caused tree mortality and surface fuels***

The satellite and aerial survey maps both captured elements of ground-based measurements of insect effects. Across both study landscapes, LandTrendr disturbance magnitude (NBR units) was positively correlated with tree basal area mortality, particularly when two field plots experiencing change outside the Landsat time series interval were excluded ( $R^2_{\text{adj}} = 0.40$ ,  $F_{1,34} = 24.76$ ,  $P < 0.0001$ ; Fig. 2.4A). Although this relationship indicates that LandTrendr accounts for only a portion of the variation in observed tree mortality, this result is particularly encouraging because field plots were distributed across highly heterogeneous stands and landscapes without replication. In addition, the basal area mortality metric did not capture insect-killed trees that had already fallen at the time of field measurements.

Combining the ADS MPB and WSB into a single metric of cumulative insect presence enabled the direct comparison of the two study landscapes, showing a positive correlation of insect duration with field measurements of dead overstory trees per hectare (TPH;  $R^2_{\text{adj}} = 0.26$ ,  $F_{2,35} = 7.38$ ,  $P = 0.002$ ; Fig. 2.4B). Similarly, the dead TPH count was positively correlated with the ADS metric of cumulative dead TPH attributed to MPB, particularly in the Cascade Lakes Zone ( $R^2_{\text{adj}} = 0.37$ ,  $F_{1,22} = 14.71$ ,  $P = 0.0009$ ; Fig. 2.4D), which experienced more intense and pervasive MPB impacts than the Mt. Hood Zone. Although the TPH metrics were positively correlated, the field-based estimates were about one order of magnitude higher than the ADS estimates (Fig. 2.4D). The cumulative ADS damage from WSB was not well correlated with the

number of dead overstory trees per hectare (data not shown), but there did appear to be a threshold effect with tree basal area mortality, where substantial tree mortality (> 20%) occurred with ADS WSB damage above five units (i.e., for Mt. Hood plots,  $R^2_{\text{adj}} = 0.25$ ,  $F_{1,10} = 4.72$ ,  $P = 0.055$ ; Fig. 2.4C). This result suggests that an appropriate use of the ADS budworm data could be as a presence/absence indicator of heavy defoliation when total impacts exceed the threshold (five damage units). This result also highlights the complementarity of the two remote sensing indices, where ADS sketch maps indicate specific insect agents, and LandTrendr provides a more objective, consistent, quantitative measure of the magnitude of change. Future research could investigate the robustness of this threshold effect.

In contrast to the consistent positive correlations of remotely sensed change maps with overstory tree mortality, the relationships of LandTrendr and ADS change magnitudes with surface fuels were highly variable (Fig. 2.5, Table 2.A1). For example, although down coarse woody detritus (CWD) mass was not associated with cumulative ADS presence of MPB and WSB ( $R^2_{\text{adj}} = -0.03$ ,  $F_{1,34} = 0.13$ ,  $P = 0.721$ ; Fig. 2.5A), CWD mass did show a positive correlation with the magnitude of Landsat spectral change (NBR units), particularly when four undisturbed plots were excluded ( $R^2_{\text{adj}} = 0.29$ ,  $F_{1,32} = 14.72$ ,  $P = 0.0006$ ; Fig. 2.5B). This association, in addition to the positive correlation of LandTrendr disturbance magnitude with tree basal area mortality, is consistent with our understanding of the Landsat spectral data being sensitive primarily to vegetative cover (Carlson and Ripley 1997 Teillet et al. 1997).

To our knowledge, no previous published study has reported a statistically significant relationship between Landsat data and CWD accumulation. CWD accumulation (i.e., snag fall) following disturbance is a highly stochastic process that depends on many factors (Russell et al. 2006). For example, Russell et al. (2006) estimated half-lives for standing, fire-killed ponderosa pine and Douglas-fir of 9-10 and 15-16 years, respectively, and Simard et al. (2011) observed a tripling of MPB-killed CWD across a 35 year chronosequence. Klutsch et al. (2009) did not observe significant differences in CWD between stands affected and unaffected by MPB 0-7 years following outbreak, but they did project significant increases of CWD over decadal time scales more applicable to the change maps evaluated in the current study. Similarly, Hummel and Agee (2003) observed a near doubling of CWD after eight years in stands affected by WSB, a time interval consistent with our observations.

Despite evidence of an apparent relationship between LandTrendr disturbance magnitude and CWD, neither LandTrendr nor ADS measures were correlated with fine woody detritus (FWD) or forest floor depth (Table 2.A1), indicating that the cumulative overstory impacts of defoliators and bark beetles did not translate to changes in fine surface fuels. The lack of significant differences in FWD is consistent with field measurements up to seven years following MPB outbreak in Rocky Mt. lodgepole pine forests (Klutsch et al. 2009, Simard et al. 2011), although both studies did document an initial increase in forest floor (litter) depth.

FWD is an important contributor to surface fire behavior (e.g., fire line intensity; Page and Jenkins 2007b, Simard et al. 2011), whereas CWD is not a key element in surface or crown fire models (Hummel and Agee 2003). Indeed, although several studies have demonstrated increasing CWD surface fuels following insect outbreaks (e.g., Hummel and Agee 2003, Klutsch et al. 2009, Simard et al. 2011), these studies have not shown significant increases in simulated crown fire potential. At our study plots, the significant increase in CWD but lack of change in FWD highlights the variability and relatively high levels of fine surface fuels in conifer forests—whether insect affected or not. It also suggests that insect impacts on fuels and fire will most likely be manifested as increased soil heating and fire residence time associated with CWD, rather than increased fire intensity or spread associated with FWD. Future studies should continue to investigate the fuels mechanisms associated with insect outbreak cycles, including the role of live vegetation fuels, which were not assessed in this study. A larger field sample size and replication/stratification likely would result in stronger statistical relationships.

### *Limitations and uncertainties*

This study presents a promising integration of remote sensing and field observations (see section 3.4 below). As in any remote sensing investigation, our pilot study was limited by uncertainties associated with each component dataset. For example, the 38 field plots were distributed purposively across diverse landscape gradients, precluding formal replication, and they were measured at only one point in time. As such, the scope of inference is limited to those specific locations and times.

Because insect disturbance processes have been occurring and continue to evolve, maps derived from the LandTrendr and ADS datasets (Figs. 2.1, 2.2) represent snapshots of very

dynamic landscapes, retrospectively integrating up to three decades of change into a single image. With our Landsat-based approach, mapping insect effects that evolve over several years is not possible until after the event has started, highlighting the need for yearly direct observations (via ADS) or detection alarms with other satellite tools (e.g., MODIS). In addition, the signal of spectral change likely depends on overall site productivity. Higher-productivity sites have more vegetation to lose, potentially improving detectability, but they typically have greater understory vegetation that may dampen the spectral impact of change. These sites also have higher potential for robust growth/recovery signals. Future studies should investigate detectability thresholds along productivity gradients.

ADS sketch maps are rich in information but have important caveats. A single polygon in any given year can be influenced by many factors, including sun angle, phenological variation, observer-to-observer variation, observer fatigue, and the scale of observation (K. Sprengel, USDA Forest Service, personal communication). In addition, polygons are recorded as homogenous patches of insect damage but actually represent heterogeneous mortality at individual trees within polygons (see scale mismatch discussion below). Further, the interannual variability of ADS observations results in sporadic coverage of what may be continuous, long-duration changes (e.g., the contrasting temporal dynamics between ADS and LandTrendr trajectories; Fig. 2.3). This patchy spatiotemporal coverage is one contributor to the order of magnitude underestimate of ADS cumulative mortality due to MPB relative to ground-based estimates. Another key factor is the mismatch between plane- and ground-based estimates of trees per hectare (particularly in lodgepole pine stands, where ground-measured density can reach  $10^4$  trees per hectare). Still, as a relative measure of insect effects on tree mortality, the ADS data captured a substantial proportion of the variation in ground-based observations (statistically significant, positive relationships; Fig. 2.4B-D), and the human observations provided invaluable identification of specific insect agents. To our knowledge, this is the first published study to present direct comparisons between plane-based ADS sketch maps and ground-based observations of tree mortality caused by MPB (but see Nelson et al. [2006] for evaluation of helicopter-based surveys).

Scale mismatch also affects our ability to precisely link field-based measurement to satellite and ADS data. Because of the cost associated with extensive field data collection, we

chose to represent approximately one Landsat pixel with our field measurements (Fig. 2.A1). Linking these measurements to a single Landsat pixel is challenging because of inaccuracy in both the field-based positional data and misregistration in any satellite imagery (particularly a time series). Similarly, the ADS data are collected to represent broad spatial extents, and the condition of any single point within the ADS polygons may or may not be representative of the entire polygon. Because these mismatches introduce noise into quantitative relationships among field and remote estimates, the exploratory results reported here are particularly encouraging.

### *Implications and recommendations*

Despite the limitations described above, this pilot study presents several previously undocumented findings. By advancing a disturbance mapping methodology that consistently identifies spectral trajectories of varying type, duration, and magnitude (LandTrendr; Kennedy et al. 2010), we have established a framework to integrate results from previous studies that used different spectral indices to assess differing insect agents and spatiotemporal scales. This framework enables the detection of seemingly disparate insect impacts due to defoliators and bark beetles (in addition to other disturbances, including fire and logging), while confirming that bark beetles induce both abrupt and gradual tree mortality and growth trajectories. We have also developed new maps to compare stand- and landscape-scale estimates of insect impacts according to multiple remote sensing datasets, showing that the ADS cumulative mortality maps and filtered LandTrendr insect maps reveal similar geographic patterns of change (Fig. 2.2), particularly across the Mt. Hood landscape. Finally, we have linked these remote sensing datasets with field observations to quantify relationships between spectral trajectories and tree mortality and, to a lesser degree, changes in surface fuels (CWD).

Future studies could build on these initial findings by leveraging larger field datasets covering multiple time intervals, such as the regional forest inventory networks of the Pacific Northwest (e.g., Current Vegetation Survey, Forest Inventory and Analysis). These large databases sample the full gradient of vegetation, insect, and fire conditions, and they also provide valuable locations for satellite and airphoto interpretation through the TimeSync validation procedure (Cohen et al. 2010). In addition to their systematic spatiotemporal coverage of forested regions, these inventory plots also sample a larger spatial footprint (~1 ha) than the

Landsat pixel size (0.09 ha) rapid assessment plots established in the current study. The larger footprint would enable the assessment of the variation and uncertainty among multiple Landsat pixels within patches of insect-killed trees. For further sampling, analysis, prediction, and interpretation, additional geographic datasets, such as the National Land Cover Data set (Vogelmann et al. 2001), biomass, productivity, and elevation maps, could help constrain spectral trajectory analysis to forests that are particularly vulnerable to specific insect agents and temporal sequences of disturbance, as well as determine detectability thresholds, as described above. Through ongoing studies integrating multiple, regional-scale remote sensing and field datasets, we aim to develop insect disturbance maps encompassing a large population of fire events, thus enabling the observation and analysis of a wide range of potential insect-fuel-fire interaction trajectories.

## SUMMARY

This study investigated spectral trajectories associated with insect disturbance and related those spectral trajectories to ground-based measurements of tree mortality and surface fuels. A Landsat time series segmentation algorithm (LandTrendr; Kennedy et al. 2010) captured both short- and long-duration changes in spectral reflectance, indicating complex temporal dynamics in insect-affected forests. We identified four general spectral trajectories that could be diagnostic of insect disturbance at the 38 conifer forest stands investigated, but future studies should quantify the variation and uncertainty associated with these trajectories at landscape and regional scales. Both LandTrendr and aerial survey estimates of overstory change were related to field measurements of tree mortality. LandTrendr disturbance magnitude was positively, weakly associated with the accumulation of coarse surface fuels, suggesting the potential for future studies to advance the development of Landsat-based fuel maps. In contrast, neither remote sensing dataset assessed in this study was associated with fine surface fuels, whose high variability suggests limited effects of insects on surface fire intensity. These results highlight the utility of insect mapping methods that capture a wide range of spectral signals and indicate that methods focusing on relatively short-term anomalies may miss substantial spatial and temporal manifestations of insect disturbance. Given the likely increase of fire and insect activity in western North American forests (Logan et al. 2003, Westerling et al. 2006, Littell et al. 2010), the accurate characterization of insect effects on forests, fuels, and subsequent wildfire will become increasingly important.

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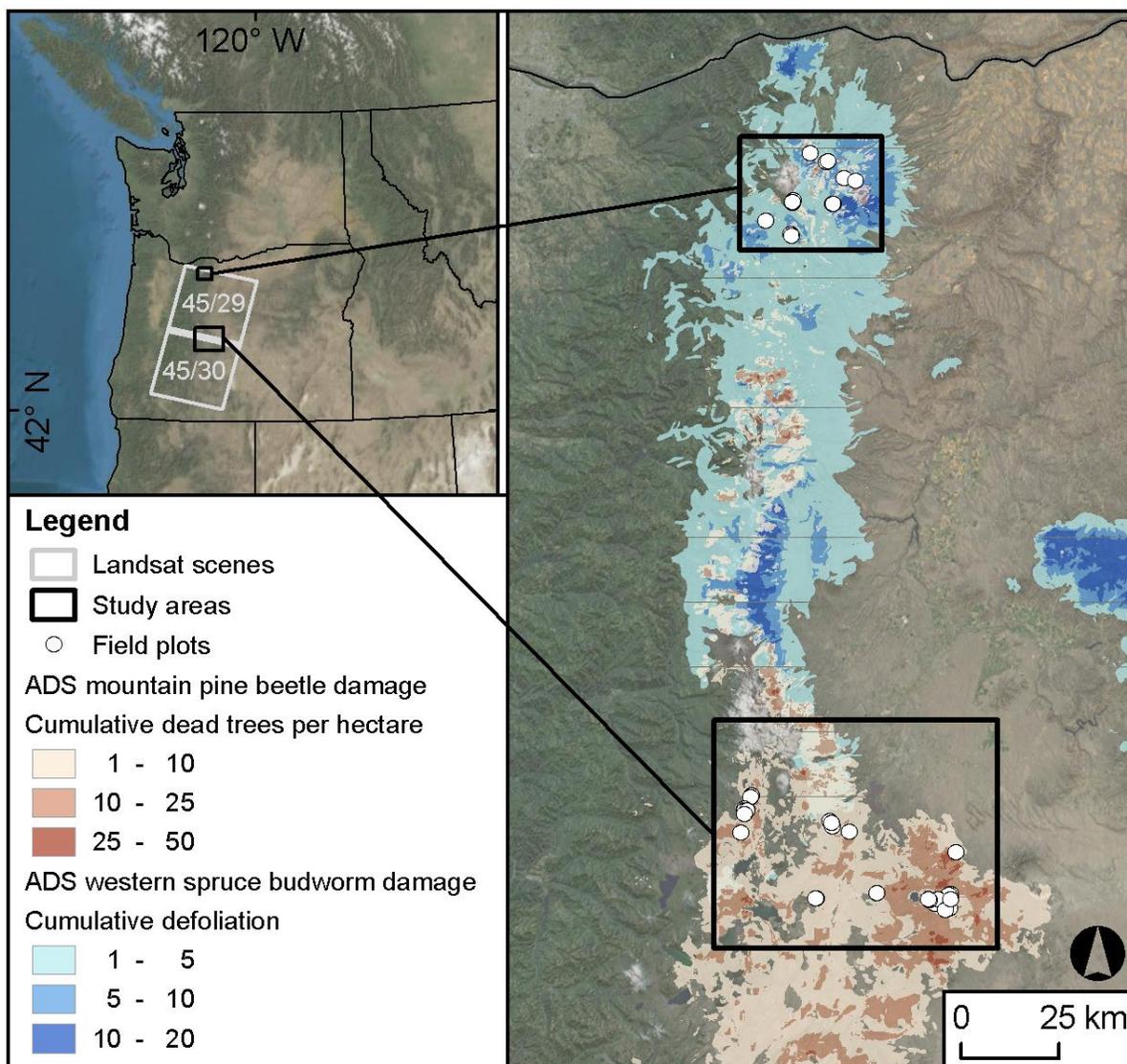
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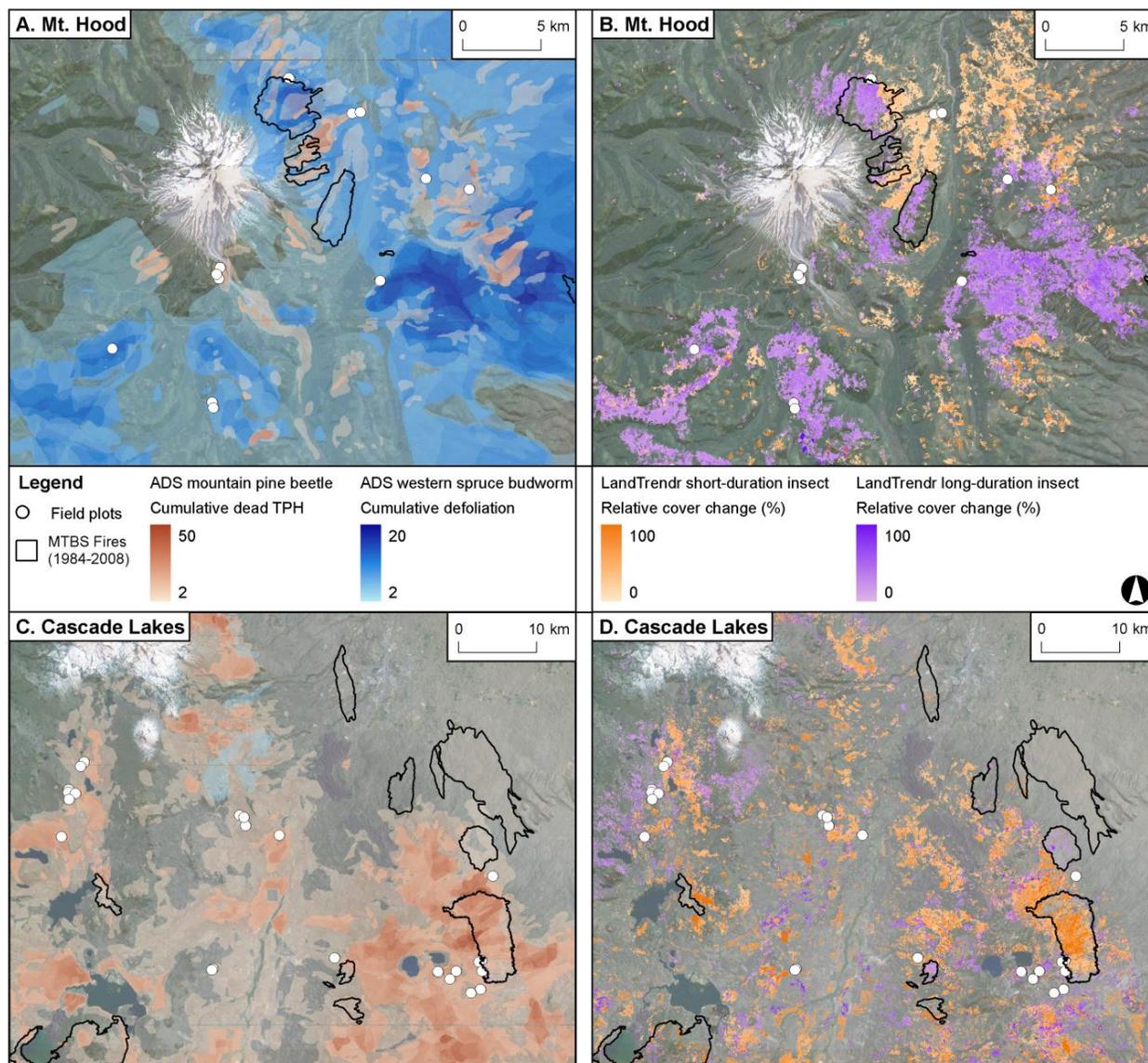
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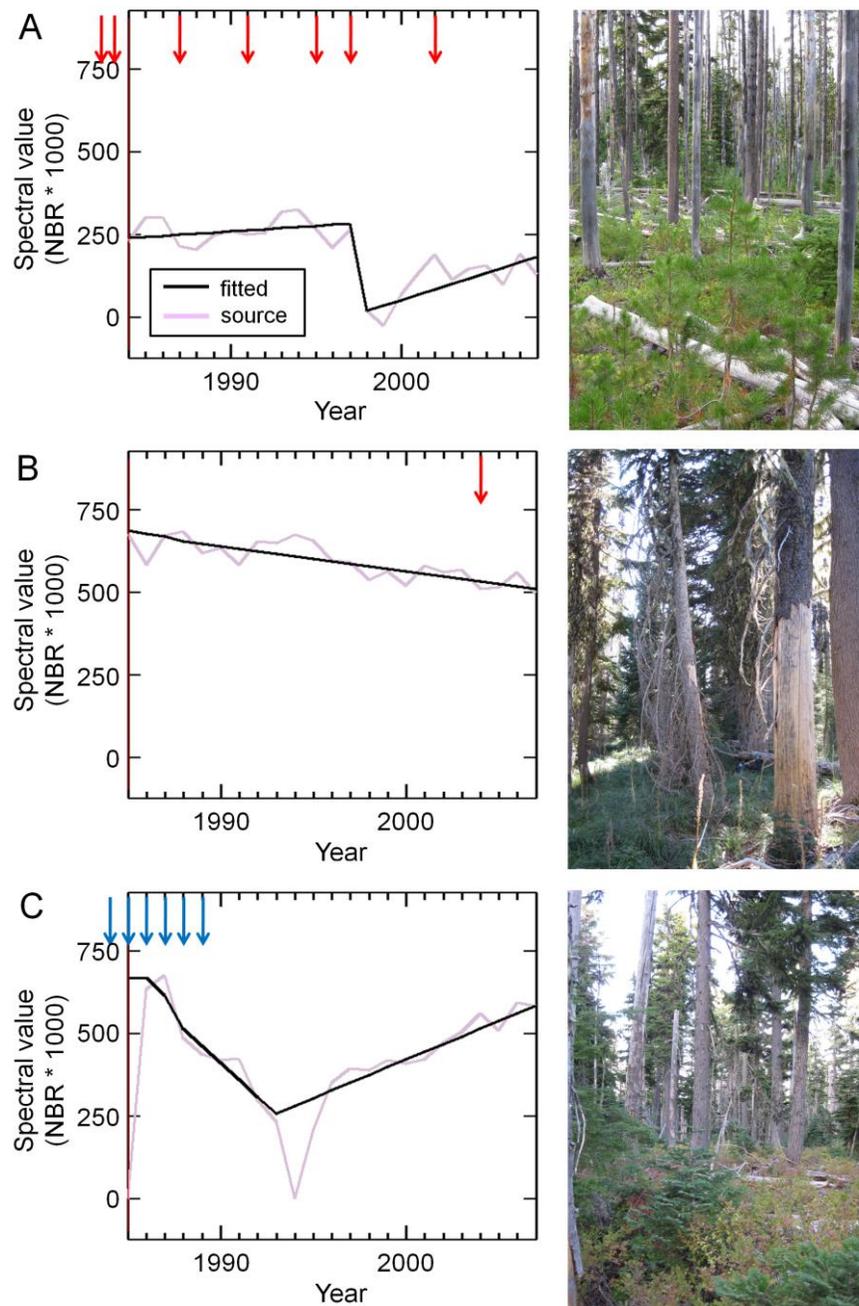
## FIGURES AND TABLES



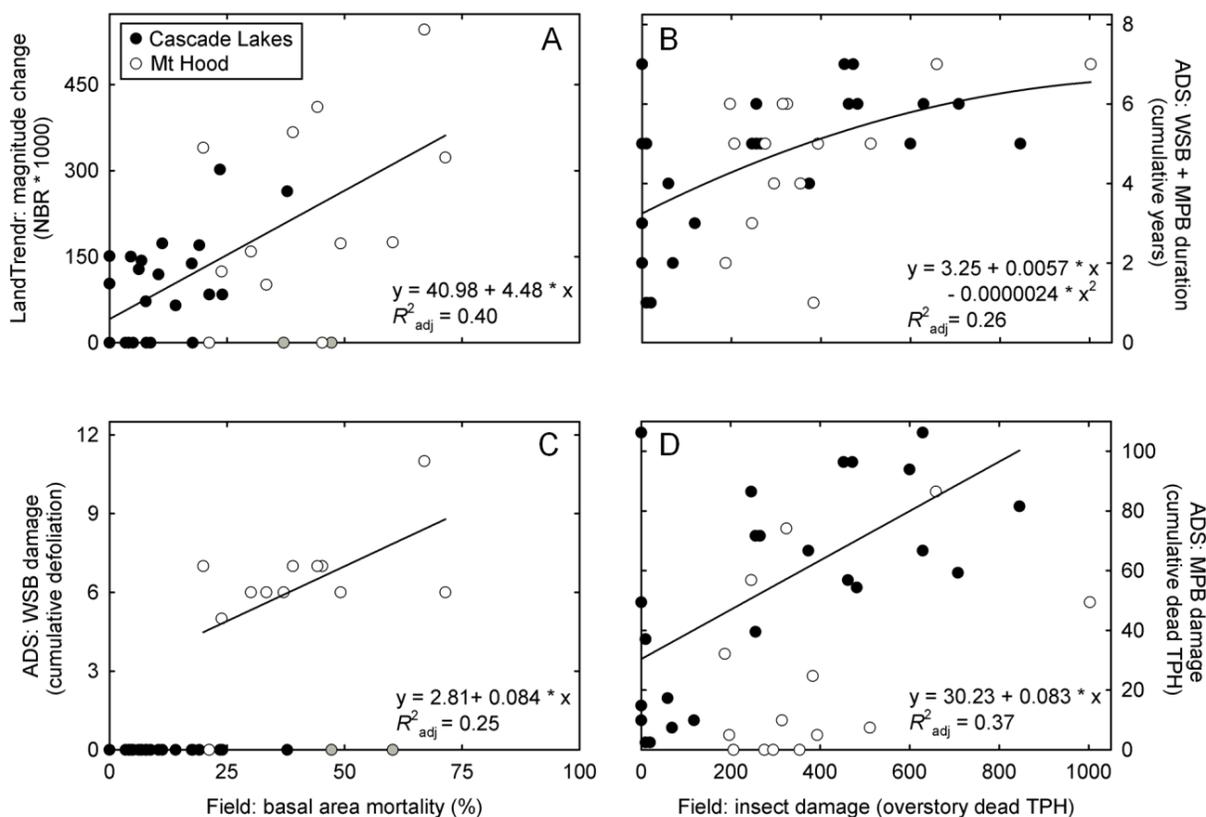
**Fig. 2.1.** Study area locations within the Pacific Northwest Region (inset) and Oregon (Landsat scenes 45/29 and 45/30). Insect damage represents cumulative impacts mapped by aerial detection surveys (ADS) from 1980 to 2009. ADS data report MPB damage quantitatively (trees per acre, converted to trees per hectare here) but WSB damage qualitatively on a numeric scale (standardized here to 1-3; see Methods). MPB data overlap WSB data where the two co-occur, and both are displayed with 30% transparency. Black boxes denote map extents in Fig. 2.2. Base map: ESRI Imagery World 2D from <http://server.arcgisonline.com>. Projection: Albers NAD83. Note the broad spatial extent of insect disturbance across forests of the Eastern Cascade Range.



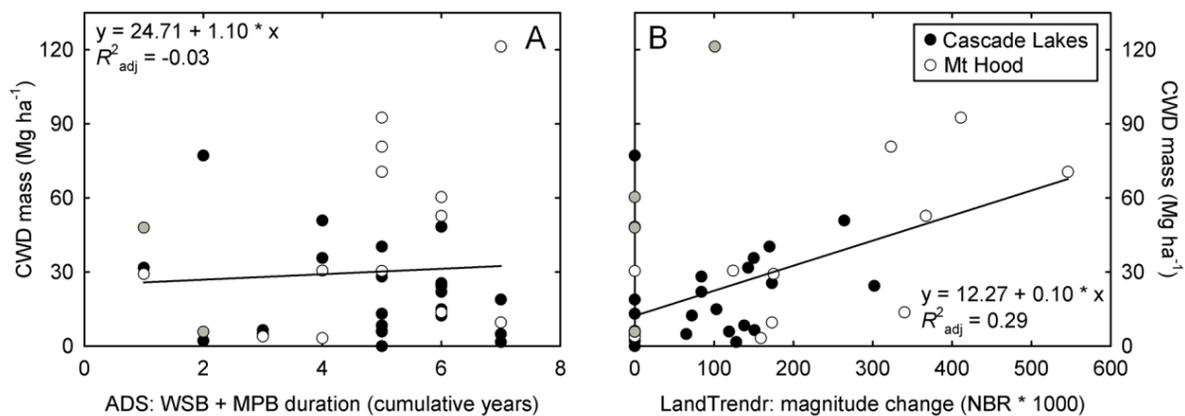
**Fig. 2.2.** Aerial detection survey and Landsat-based maps of forest disturbance across study landscapes. Note different spatial extents in two study areas. Panels (A) and (C) show aerial observations of cumulative damage from mountain pine beetle (MPB; red/brown) and western spruce budworm (WSB; blue) from 1980-2009 (displayed with 30% transparency; MPB overlaps WSB in A, WSB overlaps MPB in B). Panels (B) and (D) show LandTrendr (Kennedy et al. 2010) maps of the magnitude of short-duration (<6 years) and long-duration ( $\geq 6$  years) insect disturbance, excluding non-forested areas. Fire perimeters from: <http://mtbs.gov>. Base map: ESRI Imagery World 2D from <http://server.arcgisonline.com>. Projection: Albers NAD83. Note: (1) contrasting recent insect disturbance histories between study landscapes; (2) location of field plots; (3) broad spatial extent of ADS polygons versus the spatially-constrained Landsat pixels; (4) general agreement of heavy insect damage areas among both data sources; (5) Overlap of some but not all fires with insect disturbance areas.



**Fig. 2.3.** Example LandTrendr spectral trajectories (Kennedy et al. 2010) and plot photographs. NBR: Normalized Burn Ratio derived from Landsat bands 4 and 7 (Key and Benson 2006), multiplied by 1000. A: short-duration disturbance followed by spectral recovery. B: long-duration disturbance. C: long-duration disturbance followed by spectral recovery. Zero values in C are from clouds excluded from LandTrendr fits. Vertical arrows indicate aerial detection of insect activity (red: mountain pine beetle; blue: western spruce budworm).

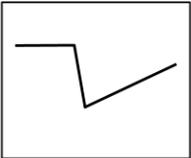
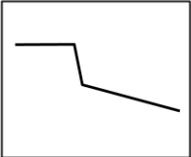
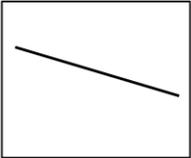
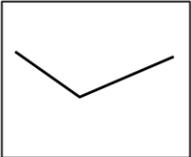
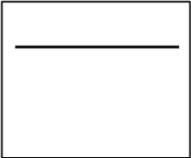
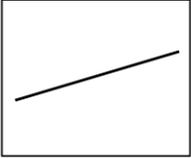


**Fig. 2.4.** Relationships of field-measured tree mortality with LandTrendr disturbance (A) and aerial detection survey (ADS) insect maps (B-D). The linear fit in (A) excludes two plots (shown as gray) where disturbance occurred outside of LandTrendr temporal coverage. TPH: dead trees per hectare. The quadratic fit in (B) indicates a saturation point (7 years of ADS detection) exceeded by none of the field plots. The linear fit in (C) excludes the Cascade Lakes plots (where minimal WSB occurred) and two additional plots in the Mt. Hood Zone (one where disturbance occurred outside the ADS temporal coverage and one where tree mortality was caused by MPB). The linear fit in (D) excludes the Mt. Hood plots (where MPB impacts were less frequent and intense [Fig. 2.2]).

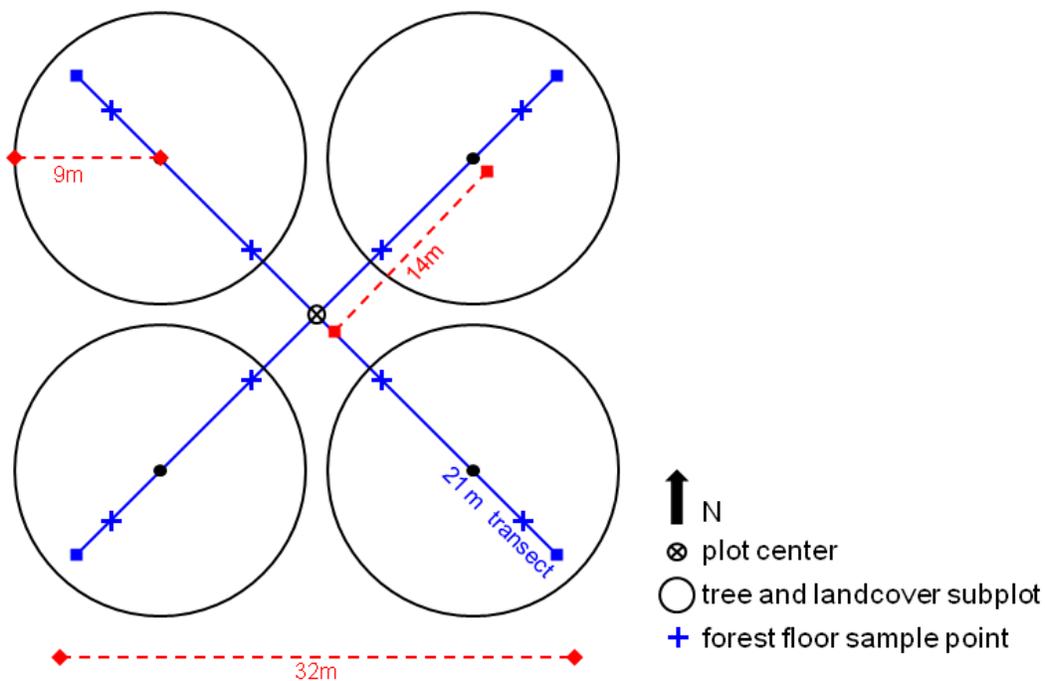


**Fig. 2.5.** Relationships of field-measured surface fuels (down coarse woody detritus: CWD) with aerial detection survey (ADS) insect maps (A) and LandTrendr disturbance (B) maps. Gray points indicate plots outside of ADS and LandTrendr temporal coverage.

**Table 2.1.** Spectral trajectories associated with insect activity, stable conditions, and growth.

Spectral trajectory	Interpretation	Insect mortality agent (from ADS, field obs.)	Number of plots ( <i>n</i> = 38)			Environmental conditions
			MH	CL	Total	
	stable, rapid mortality, recovery	mountain pine beetle	0	4	4	relatively productive sites
	stable, rapid mortality, slow mortality	mountain pine beetle	0	2	2	multiple disturbance processes such as mortality followed by tree fall
	long, slow mortality	mountain pine beetle western spruce budworm	4	4	8	long-term presence of insects or multiple insect agents affecting different hosts within a stand
	slow mortality, recovery	mountain pine beetle western spruce budworm	7	6	13	relatively productive, mesic sites exhibiting rapid recovery
	stable (no change)	none	3	3	6	stable over time interval
	growth or recovery	potential prior insect	0	5	5	potentially linked with natural or anthropogenic disturbance

## CHAPTER APPENDICES

**Fig. 2.A1.** Field plot sampling design.

*Notes:* The subplots (black circles) and fuels transects (blue lines) are designed to sample the area within one Landsat pixel (30 x 30 m). Results presented in this paper are averaged among subplots and scaled to per-unit-area values. Figure not to scale.

**Table 2.A1.** Statistical relationships among remote sensing and fuels variables.

Dataset <sup>a</sup>	<i>n</i>	Response variable (y)	Explanatory variable (x)	$\beta_0$	(se)	$\beta_1$	(se)	adjusted $R^2$	$F$ -stat <sub>1,n-2</sub>	$P$ -value
All plots	38	CWD	LandTrendr disturbance magnitude (NBR * 1000)	19.28	(6.07)	0.083	(0.033)	0.127	6.402	0.016
All - 4 error plots <sup>b</sup>	34	CWD	LandTrendr disturbance magnitude (NBR * 1000)	12.27	(5.16)	0.102	(0.026)	0.294	14.720	0.0006
All plots	38	CWD	ADS cumulative presence of WSB and MPB (years)	25.74	(13.59)	0.888	(2.718)	-0.025	0.107	0.746
All - 2 error plots <sup>c</sup>	36	CWD	ADS cumulative presence of WSB and MPB (years)	24.71	(15.67)	1.103	(3.058)	-0.025	0.130	0.721
MH plots	14	CWD	ADS damage by WSB (cumulative defoliation)	14.51	(18.42)	5.424	(3.017)	0.147	3.233	0.097
All plots	38	CWD	ADS damage by MPB (trees per hectare)	34.40	(7.49)	-0.103	(0.134)	-0.011	0.598	0.444
All plots	38	FWD	LandTrendr disturbance magnitude (NBR * 1000)	5.29	(0.80)	0.004	(0.004)	-0.004	0.847	0.364
All plots	38	FWD	ADS cumulative presence of WSB and MPB (years)	1.68	(1.51)	0.879	(0.301)	0.169	8.516	0.006
MH plots	14	FWD	ADS damage by WSB (cumulative defoliation)	5.21	(2.15)	0.394	(0.353)	0.019	1.248	0.286
All plots	38	FWD	ADS damage by MPB (trees per hectare)	4.95	(0.91)	0.020	(0.016)	0.012	1.452	0.236
All plots	38	Forest floor depth	LandTrendr disturbance magnitude (NBR * 1000)	4.47	(0.48)	0.002	(0.003)	-0.011	0.611	0.440
All plots	38	Forest floor depth	ADS cumulative presence of WSB and MPB (years)	4.29	(1.00)	0.094	(0.200)	-0.021	0.223	0.640
MH plots	14	Forest floor depth	ADS damage by WSB (cumulative defoliation)	5.00	(1.21)	0.150	(0.199)	-0.034	0.569	0.465
All plots	38	Forest floor depth	ADS damage by MPB (trees per hectare)	4.98	(0.55)	-0.006	(0.010)	-0.018	0.334	0.567

*Notes:*

Simple linear regression model for all comparisons:  $y = \beta_0 + \beta_1 * x$ .

Abbreviations and references: CWD: down coarse woody detritus (Mg ha<sup>-1</sup>); FWD; fine woody detritus (Mg ha<sup>-1</sup>); Forest floor depth is the sum of litter and duff depth (cm); LandTrendr (Kennedy et al. 2010); NBR: Normalized Burn Ratio (Key and Benson 2006); ADS: aerial detection survey; WSB: western spruce budworm; MPB: mountain pine beetle).

<sup>a</sup> Error plots removed where field observations indicated that response variable (e.g., CWD) was not associated with the time interval covered by the LandTrendr or ADS dataset. WSB occurred only at plots in Mt. Hood Zone.

<sup>b</sup> Model fit shown in Fig. 2.5b.

<sup>c</sup> Model fit shown in Fig. 2.5a.

### **CHAPTER 3 || SPATIOTEMPORAL DYNAMICS OF RECENT MOUNTAIN PINE BEETLE AND WESTERN SPRUCE BUDWORM OUTBREAKS ACROSS THE PACIFIC NORTHWEST REGION**

#### **ABSTRACT**

Across the western US, the two most pervasive, potentially damaging native forest insects are mountain pine beetle (MPB; *Dendroctonus ponderosae*; a bark beetle) and western spruce budworm (WSB; *Choristoneura freemani*; a defoliator). MPB outbreaks have received more research and forest management attention than WSB outbreaks, but studies to date have not compared their cumulative mortality impacts in an integrated, regional framework. The goal of this study is to map and compare the effects of both insects on tree mortality across the Pacific Northwest Region from 1970 to 2012. Our specific objectives are to: (1) map tree mortality associated with MPB and WSB outbreaks using multi-date forest inventory data, forest health aerial detection surveys (ADS; 1970-2012), and Landsat time series (LandTrendr; 1984-2012); (2) compare the timing, extent, and cumulative impacts of recent MPB and WSB outbreaks across the forested ecoregions of Oregon and Washington, USA. For our first objective, we related the change in plot-based dead tree basal area to the contemporaneous change in a Landsat spectral index (NBR) using reduced major axis regression. We then mapped MPB and WSB impacts (dead basal area at 30-m pixel resolution) across the region by applying the tree mortality model to LandTrendr disturbance trajectories collocated with ADS polygons. Our Landsat-based insect maps facilitate comparisons across space, time, and insect agents that have not been possible to date, complementing existing ADS maps in three important ways. They (1) quantify change in terms of field-measured tree mortality; (2) provide consistent estimates of change for multiple agents, particularly long-duration changes; (3) capture variation of insect impacts at a finer spatial scale within ADS polygons, substantially reducing estimated insect extent. For our second objective, we evaluated MPB and WSB impacts and synchrony at ecoregional and regional scales. Despite high variation across the study region, spatiotemporal patterns were evident in both the ADS- and Landsat-based maps of insect activity. MPB outbreaks occurred in two phases—first during the 1970s and 1980s in eastern and central

Oregon and then more synchronously during the 2000s throughout the dry interior conifer forests of the PNW. Reflecting differences in habitat susceptibility and epidemiology, WSB outbreaks exhibited early activity in northern Washington and an apparent spread from the eastern to central PNW during the 1980s, returning to northern Washington during the 1990s and 2000s. Across the region, WSB exceeded MPB in extent and tree mortality impacts in all ecoregions except for one, suggesting that ongoing studies should account for both bark beetles and defoliators. Given projected increases of insect and fire activity in western forests, the accurate assessment and monitoring of these disturbances will be crucial for sustainable ecosystem management.

#### KEYWORDS

bark beetle; change detection; defoliator; forest health; insect disturbance; Landsat time series; mountain pine beetle; Pacific Northwest; tree mortality; western spruce budworm

## INTRODUCTION

Insects are important forest disturbance agents, influencing ecosystem structure, function, and dynamics from local to global scales. Across western North America, native forest insects affect more area annually than wildfire (e.g., Williams and Birdsey 2003, Raffa et al. 2008, Littell et al. 2009), and, like wildfire, insects have received increasing research and public attention. In addition to their direct effects on forest mortality and succession, insects also influence fuel structure, setting the stage for potential interactions with fire, anthropogenic climate change, and forest management (e.g., McKenzie et al. 2004, Kurz et al. 2008, Littell et al. 2010, Simard et al. 2011). Different insects vary substantially in their distribution and effects, however, and it is important to contrast the tree mortality impacts of different insect types.

In the western US, the two most pervasive and potentially damaging native insects are mountain pine beetle (MPB; *Dendroctonus ponderosae* Hopkins [Coleoptera: Scolytidae]; a bark beetle) and western spruce budworm (WSB; *Choristoneura freeman* Razowski [Lepidoptera: Tortricidae]; a defoliator; Williams and Birdsey 2003). Although numerous studies have assessed the causes and consequences of these insects from fine to coarse scales (e.g., Swetnam and Lynch 1993, Simard et al. 2011, Preisler et al. 2012, Meddens et al. 2012) few to date have compared their cumulative impacts at a regional scale (but see Williams and Birdsey [2003]). This lack of comparison is due in part to a perception that defoliators are not as significant a forest health threat as bark beetles; it is also due to a lack of spatiotemporally explicit datasets that quantify the impacts of various agents in consistent units. This paper integrates spatially and temporally extensive satellite, aerial, and field data to map and compare bark beetle and defoliator effects on tree mortality across the Pacific Northwest Region from 1970 to 2012.

Here, we suggest that a broad, regional scale is essential to elucidate potentially unprecedented changes and emergent patterns. Particularly in the context of global environmental change, it is essential to evaluate recent insect outbreaks across spatiotemporal scales that are broad enough to capture multiple outbreaks of the same insect species (e.g., Swetnam and Betancourt 1998, Meddens et al. 2012, Flower et al. 2014). In addition, a regional framework enables the comparison of multiple insect species across different forest types and the observation of emergent impacts that may not be apparent at finer spatiotemporal scales. Moreover, a regional framework may allow researchers and managers to assess whether

particular outbreaks are driven by landscape-scale endogenous drivers (e.g., insect population dynamics, host abundance and distribution) or regional-scale exogenous drivers (e.g., climate, synchronous drought stress of host trees). Datasets spanning broad spatial and temporal scales are limited, however, and existing maps have key limitations and uncertainties (Meigs et al. 2011 [This dissertation, Chapter 2], Meddens and Hicke 2014). New, regionally extensive but fine-grained maps are necessary to advance our understanding of these insect outbreak phenomena.

In western North America, the most readily available maps to assess landscape and regional insect dynamics are forest health aerial detection surveys (ADS). These surveys provide an unmatched record of insect activity and other forest disturbances and have been collected annually for decades, documenting multiple outbreaks across space and time (Williams and Birdsey 2003). Although valuable for coarse-scale assessments, ADS data have critical uncertainties related to the spatial and temporal variability of observation conditions, methodology, and personnel (Meigs et al. 2011, Preisler et al. 2012, Meddens and Hicke 2014). More importantly, the aerial survey focus on detection and use of inclusive polygons creates the appearance of widespread, homogeneous insect activity across entire landscapes and regions, whereas actual insect impacts on tree mortality are highly variable at finer scales (Meigs et al. 2011). Diverse stakeholders, including forest managers and policy makers, are concerned about the apparent forest health crises that ADS maps imply. In contrast to this potential overestimation of insect *extent* at the regional scale, studies have shown that the ADS units of damage, at least in the case of bark beetles, are a strong underestimate of actual *impacts* at the forest stand scale (Meigs et al. 2011, Meddens and Hicke 2014). Finally, different agents are characterized in different units, hindering their comparison. Thus, complementary geospatial and field datasets are necessary to facilitate regional analysis.

In many ways, the Landsat satellite archive represents an ideal complementary dataset. As reviewed by Wulder et al. (2006) and Meigs et al. (2011), recent studies have used a variety of remote sensing platforms to map insect disturbance, and Landsat-based approaches show considerable promise. Due to its moderate spatial resolution (30-m pixels), broad temporal scope (1984-present), and consistent, seamless coverage, the Landsat TM sensor captures a variety of disturbance dynamics at forest stand, landscape, and regional scales (Cohen and Goward 2004, Kennedy et al. 2014). Previous studies have demonstrated wide variability in

Landsat spectral indices and trajectories associated with insect outbreaks (e.g., Vogelmann et al. 2009, Goodwin et al. 2010, Meigs et al. 2011). Although more studies have focused on MPB outbreaks (due to their generally more damaging outbreaks; e.g., Wulder et al. 2006, Goodwin et al. 2010, Simard et al. 2011), Landsat-based analyses capture WSB dynamics as well (Vogelmann et al. 2009, Meigs et al. 2011). Like the ADS data, however, Landsat-based estimates have important limitations. Specifically, satellite data lack any inherent attribution of specific disturbance agents, and the raw reflectance data have limited biological meaning without being linked to ground-based estimates of change (e.g., tree mortality). Here, we advance a multi-faceted approach, leveraging Landsat time series and aerial survey data, as well as ground-based observations from regional forest inventory plots, to map and compare MPB and WSB disturbance.

Mountain pine beetle and western spruce budworm are the two most prevalent species of bark beetle and defoliator, respectively, across western North America (Williams and Birdsey 2003). Both insects exhibit cyclic outbreaks associated with endogenous and exogenous factors, including insect population dynamics, host tree vigor and distribution, and both short- and long-term climatic variability (Sanders et al. 1985, Swetnam and Betancourt 1998, Raffa et al. 2008, Bentz et al. 2010). Under sustained outbreak conditions, both insects can cause pervasive tree mortality and associated changes in forest structure and composition, although the mechanisms and rates differ (Swetnam and Lynch 1993, Raffa et al. 2008, Meigs et al. 2011). MPB adults mass attack the stems of pine trees (*Pinus spp.*, particularly mature lodgepole pine [*P. contorta* Douglas ex Louden]) to feed on cambium and lay eggs (Goheen and Willhite 2006), inducing variable but relatively rapid tree mortality across large areas (e.g., regional-scale outbreaks in British Columbia and US Rocky Mountains [Meddens et al. 2012]). In contrast, WSB larvae typically consume only current year's foliage of host trees (especially understory true firs [*Abies spp.*], spruces [*Picea spp.*], and Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco]) (Goheen and Willhite 2006, Vogelmann et al. 2009, Meigs et al. 2011). Despite these subtler impacts, multiple years of WSB defoliation can result in tree mortality, often in conjunction with secondary bark beetles (not including MPB, which is a primary bark beetle; Hummel and Agee 2003, Goheen and Willhite 2006, Vogelmann et al. 2009). In the US Pacific Northwest Region (PNW; defined here as the states of Oregon and Washington), multiple widespread outbreaks of

both insects have occurred in recent decades (Williams and Birdsey 2003), creating a strong need for a multi-decadal, retrospective analysis of their impacts.

In this study, we build on a recent pilot study (Meigs et al. 2011), applying new approaches across a much broader area to develop a regional, Landsat-based atlas of insect activity. Our overall goal is to map and compare the effects of two native insects on tree mortality across the PNW Region from 1970 to 2012. Our specific objectives are to: (1) map tree mortality associated with MPB and WSB outbreaks using multi-date forest inventory data, aerial detection surveys, and Landsat time series; (2) compare the timing, extent, and cumulative impacts of recent MPB and WSB outbreaks across the forested ecoregions of Oregon and Washington.

## METHODS

### *Study area*

The Pacific Northwest is a geographically diverse region with ecologically, economically, and socially important forest landscapes. Mixed-conifer forests are widespread, and their composition and structure vary across gradients of climate, topography, soil parent material, and disturbance and management history (Franklin and Dyrness 1973, Hessburg et al. 2000). Although precipitation and temperature regimes vary substantially, a common climatic feature is that little precipitation occurs in summer months (Franklin and Dyrness 1973). The resulting warm, dry conditions, coupled with abundant biomass and slow decomposition, are conducive to periodic insect and wildfire disturbances (e.g., Littell et al. 2010). This study focuses on the forested ecoregions (Omernik 1987) where MPB and WSB have been most active historically, which are primarily the drier interior mixed-conifer forests east of the crest of the Cascade Range. Our specific study area is defined as forests affected by MPB or WSB since 1970 according to aerial detection surveys (approximately 8 M ha; Fig. 3.1).

In general, these forests inhabit remote, mountainous terrain and are managed primarily by the US Forest Service. These forest landscapes have undergone dramatic historical and recent anthropogenic changes, including widespread fire suppression and exclusion, logging, and grazing, which have increased the potential for landscape-to-regional-scale outbreaks of native

bark beetles and defoliators (Hessburg et al. 2000). Given the widespread extent of similar geographic conditions and anthropogenic pressures, these forests and recent MPB and WSB dynamics are also broadly representative of analogous vegetation types and recent insect outbreaks throughout western North America.

### *Datasets*

Here, we leverage three spatially and temporally extensive datasets in a retrospective, empirical framework.

#### *Aerial Detection Survey (ADS)*

The ADS is a cooperative effort between federal and state agencies, conducted nationally and administered at the regional level. In US Forest Service Region 6 (Oregon and Washington; data available online: <http://www.fs.usda.gov/main/r6/forest-grasslandhealth/>), these surveys have been conducted by observers in fixed-wing aircraft since 1947 with the objective of recording a wide range of forest insects and diseases based on characteristic aerial signatures (Ciesla 2006). Compared to the western US region (Meddens et al. 2012), the PNW ADS has been more consistently and completely surveyed on an annual basis.

In this study, we utilized ADS data from 1970 to 2012, converted the polygons to raster format (30-m grain), and selected all MPB and WSB pixels for further analysis. For each affected pixel, we identified the onset (first year of detection), cumulative magnitude (trees per hectare for MPB, defoliation units on a 1-3 scale for WSB), and count of years detected (as in Meigs et al. 2011) for two time periods (1970-2012, 1984-2012). Due to the limitations and uncertainties mentioned above, ADS data are not intended for assessment of specific stands or points on the ground but rather landscape-to-regional patterns and trends (Ciesla 2006, Meigs et al. 2011). To this end, we focused on insect patterns at the ecoregional and regional scales, and we combined the ADS data with Landsat time series and forest inventory data to map the impacts of MPB and WSB (1) at a finer spatial scale than the homogeneous ADS polygons and (2) in terms of field-based estimates of tree mortality.

### *Landsat imagery and LandTrendr disturbance mapping*

We used Landsat TM/ETM+ imagery from 1984 to 2012 covering Oregon and Washington (including at least part of 35 path/row scenes; Fig. 3.A1) and applied LandTrendr algorithms and analyses, which are described in detail by Kennedy et al. (2010). Briefly, we downloaded georectified images from the USGS Landsat archive and applied a series of pre-processing, processing, and analysis steps to assess trajectories of change (vegetation decline and growth) associated with disturbance dynamics (Kennedy et al. 2010, Meigs et al. 2011).

Pre-processing involved atmospheric correction using the LEDAPS algorithms (Masek et al. 2006) and cloud masking using the FMASK algorithm (Zhu and Woodcock 2012), with occasional cloud mask additions added after quality control screening. All images were reprojected from the standard USGS UTM projection to the standard US Continental Albers Equal Area projection to allow mosaicking across Landsat scenes. The result was a temporal stack of images with minimal atmospheric and cloud interference, from which the clearest pixel per year near the median date of each stack (generally August 1st) was used as input to the pixel-based temporal segmentation algorithms.

We then calculated the Normalized Burn Ratio spectral index (NBR; Key and Benson 2006). By contrasting Landsat bands four (near infrared) and seven (short wave infrared), NBR is similar to spectral indices used in other remote sensing assessments of insect effects (e.g., Wulder et al. 2006, Vogelmann et al. 2009, Goodwin et al. 2010, Meigs et al. 2011, Meddens and Hicke 2014). NBR is also widely familiar to the fire science community (Eidenshink et al. 2007), and sensitivity analyses have shown its utility for capturing various disturbance dynamics in our study region (Kennedy et al. 2010, Cohen et al. 2010).

Following these pre-processing steps, we processed the imagery by applying LandTrendr temporal segmentation algorithms to the NBR time series at the pixel scale (30-m grain). Segmentation distills an often-noisy annual time series into a simplified set of vertices and segments to capture the salient features of a trajectory while omitting most false changes (Kennedy et al. 2010). First, we identified disturbance segments as those where NBR declined between vertices. Then, for each disturbance segment, we mapped the absolute change in NBR as an estimate of disturbance magnitude. We also mapped the onset year, duration (in years), and slope of each segment (where slope is magnitude divided by duration). This latter variable –

NBR slope – has the potential to distinguish abrupt, severe disturbances like stand-replacing wildfire and logging from more gradual, subtle changes due to insect activity. It also integrates short-duration, lower magnitude changes associated with some MPB activity with longer-duration, higher magnitude changes associated with WSB activity in more productive forest types (Meigs et al. 2011).

*Field-based estimates of tree mortality from forest inventory data*

We related LandTrendr-based changes in NBR to concurrent field-based estimates of tree mortality from the federal Current Vegetation Survey (CVS) database. The CVS began as a systematic forest inventory on US Forest Service lands in the Pacific Northwest Region (Max et al. 1996) and subsequently was integrated into the national Forest Inventory and Analysis program (Bechtold and Patterson 2005). In the PNW, most CVS plots were measured twice, as early as 1993 and as late as 2007 (intervals vary from 1 to 14 years). The CVS plots are located on a systematic, regular grid (one plot every 2.7 km in non-wilderness, 5.5 km in wilderness; Ohmann et al. 2007) and represent an unbiased sample of Forest Service land. Each CVS plot covers one hectare and includes nested subplots for different vegetation attributes, including live and dead tree species, diameter, and height, land use, stand age, and, via repeat measures, tree growth, mortality, and regeneration. For each CVS measurement occasion, we computed per-unit-area total live and dead tree basal area and density, as well as their absolute and relative changes. Our previous research showed that LandTrendr change detection was most strongly associated with basal area mortality (Meigs et al. 2011), so we used the absolute change in dead tree basal area as our primary estimate of plot-scale tree mortality and response variable in regression modeling (methods below; Section 2.3.2).

Because we were interested in capturing insect effects (MPB, WSB) on forest stands during CVS intervals, we subsetted the inventory data in several important ways. Specifically, we focused on CVS plot intervals with these characteristics: aerial detection of MPB or WSB within three years of either measurement (an inclusive interval accounting for potential lagged mortality and/or missed detection), at least 10 measured trees, total tree basal area of at least two units ( $\text{m}^2 \text{ha}^{-1}$ ), and a remeasurement interval of at least three years. We also excluded CVS plots with decreasing dead tree basal area (i.e., plots where more tree basal area died before the first observation than during the interval; Azuma 2010), increasing NBR (i.e., spectral evidence of

vegetation growth rather than loss or no change), and evidence of human disturbance within the interval (i.e.,  $\geq 5\%$  of live tree stems removed by human activity). We thus narrowed the sample population substantially from the overall CVS database ( $n \approx 12,000$ ) to our final subset ( $n = 282$ ).

We recognized that this subset includes some plots with both MPB and WSB effects as well as other non-anthropogenic disturbances (e.g., other insects, fire, wind) between CVS observations. Rather than further narrowing the database to specific MPB and WSB subsets and excluding locations with potential secondary disturbances, we retained this more general mortality subset to cover a bigger range of mortality effects across the region. In addition, by combining both insects in the same mortality subset, we were able to retain a broader scope of inference, calibrate insect damage to a more general biophysical relationship, and compare MPB and WSB maps in consistent units of change.

### *New maps of insect impacts on tree mortality*

Following the data acquisition, pre-processing, and processing steps described above, we leveraged the Landsat spectral trajectories in two primary data pathways to create new maps of MPB and WSB impacts on tree mortality (Fig. 3.2). All geospatial processing and statistical analyses were performed with the ArcGIS, IDL, GDAL, Python, or R programs.

#### *Combining Landsat and aerial survey data*

First, to capture the Landsat-scale impacts associated with ADS polygons, we selected all pixels with LandTrendr disturbance temporal segments intersected in at least one year by ADS damage greater than or equal to a baseline cumulative threshold (two TPA for MPB; three defoliation units for WSB). Although these pixel-based segments overlapped spatially and temporally with one or more ADS polygons, the change also could have been driven by non-insect disturbance. To resolve this potentially confounding effect, we used the NBR slope variable to distinguish insect, stand-replacing fire, and logging disturbances at stratified-random airphoto and Landsat time series interpretation sites ( $n = 480$ ). These sites were stratified by insect agent (MPB, WSB) and Landsat scene (40 sites for each agent in each of six Landsat scenes across the study area; Fig. 3.A1). Following the identification of non-insect ( $n = 72$ ) and presumed insect ( $n = 408$ ) samples, we randomly selected two-thirds of the sample for training

and used the other one-third for validation, combining the MPB and WSB data because their slope distributions were not significantly different ( $p > 0.05$ ; ANOVA; data log-transformed to meet normality and equal variance assumptions).

To identify key thresholds where slope values balanced the omission and commission errors of both non-insect and presumed insect samples, we plotted producer's and user's accuracy curves (Fig. 3.B1), repeating this process 10 times and averaging the threshold values to determine a low (NBR slope = 29) and high threshold (NBR slope = 239). We then created two sets of maps by excluding pixels with slope values above the low and high thresholds to both examine the sensitivity of the final change maps to the thresholds and to assess map uncertainty (methods below; Section 2.5). Finally, we extracted the annual incremental change from the Landsat-based insect segments (NBR magnitude), assembled regional mosaics for each year (1984-2012), and derived cumulative disturbance summaries analogous to the ADS cumulative summaries (onset, cumulative magnitude, and count).

#### *Statistical modeling of Landsat- and field-based estimates of change*

To relate these maps of Landsat pixel-scale change associated with MPB and WSB to a meaningful, field-based estimate of change, we performed Reduced Major Axis (RMA) regression modeling between Landsat and CVS inventory plot data (Fig. 3.3). RMA regression has several advantages over ordinary least squares or other simple linear approaches, particularly in a remote sensing framework (Cohen et al. 2003). Foremost among these is the symmetry of the variance structure around the RMA fit (i.e., sum of squares are minimized with regard to both the X and Y axes rather than the Y axis only as in OLS regression; Cohen et al. 2003). This symmetry recognizes that both the X and Y variables are measured with error and also enables the interchanging of axes such that either variable can be evaluated on the X or Y axis. Because our specific objective was to create maps of the change in dead tree basal area based on the change in NBR, this simple, symmetrical, univariate approach was optimal.

For each CVS plot, we first extracted the mean annual NBR values within a 13-pixel footprint anchored on the plot's X-Y coordinates (Ohmann et al. 2007). This footprint is analogous to the one hectare CVS sample plot and provides a small buffer for spatial offsets between raster data and CVS plot centers as well as GPS coordinate errors (Zald et al. 2014). We then calculated the change in NBR during the CVS interval and used this as our predictor

variable in the RMA regression. Our *a priori* response variable for tree mortality mapping was the change in dead tree basal area ( $\text{m}^2 \text{ha}^{-1}$ ) during the CVS interval. This univariate model (change in dead basal area as a function of the change in NBR) performed as well as or better than other potential models with other tree change metrics (live basal area, live and dead tree density) and spectral indices (relative differenced NBR [Miller and Thode 2007], pre-disturbance and differenced tasseled-cap indices [Crist and Cicone 1984]; data not shown). To complete this mapping process, we applied the coefficients from this RMA regression to all disturbance segments in the NBR insect maps to derive maps of estimated change in dead tree basal area attributable to MPB and WSB activity in all years (Fig. 3.2).

### ***Comparison of recent MPB and WSB activity***

For our second objective, we summarized the timing, extent, and cumulative impacts of MPB and WSB across forested ecoregions of the PNW. We compared three different sets of maps – original ADS, Landsat-based low slope threshold, Landsat-based high slope threshold – for each insect at ecoregional and regional scales. We also combined the map displays with time series of the extent and per-unit-area impacts on estimated tree mortality associated with MPB and WSB. These time series enabled the comparison of the onset, duration, and overall synchrony of outbreaks between the two insect agents and among ecoregions. We hypothesized that widespread synchrony would be evidence of broad, exogenous drivers such as drought, whereas asynchrony would imply independent drivers such as local to ecoregional history of land use and other disturbances.

### ***Uncertainty analysis***

Recognizing that there are important sources of uncertainty throughout this data processing and analysis framework, we evaluated the sensitivity of mapped outputs and relationships at multiple steps in the mapping workflow (Fig. 3.2). For example, to assess the variability within the ADS polygons, we compared the extent and cumulative damage from the ADS data to the new Landsat-based insect maps (which, by definition, capture insect damage only within the ADS polygons but at a finer resolution). We also compared the onset year and count (number of years) of the ADS- and Landsat-based maps by differencing the two types of

maps where they overlapped in space (i.e., excluding ADS pixels without Landsat change pixels) and time (i.e., 1985-2012, excluding potential confounding effects of the initial Landsat change year [1984-1985]). In addition, we assessed the sensitivity of the insect/non-insect NBR slope threshold by comparing the two sets of Landsat-based maps (low and high slope threshold) in terms of their extent and cumulative impacts (dead tree basal area). Finally, although we focused on recent MPB and WSB dynamics as captured by the ADS, Landsat, and CVS data, many other disturbance processes influence these forests. The impacts of other insects, wildfires, pathogens, logging, drought, and additional factors contribute to the variation in these datasets but are beyond the scope of this study.

## RESULTS

### *New Landsat-based maps of tree mortality associated with insect activity*

At forest inventory plots with aerial detection of either MPB or WSB, increases in field-based dead tree basal area were associated with decreases in the NBR spectral vegetation index (Fig. 3.3; RMA regression;  $n = 282$ ;  $R^2 = 0.67$ ). By applying the Landsat-based tree mortality model to disturbed pixels intersecting ADS polygons, we created novel insect damage maps (Figs. 3.2, 3.4), thus enabling subsequent comparisons across insect agents, landscapes, and ecoregions.

At the forest landscape scale, the cumulative Landsat-based maps of insect impacts revealed much more fine-grained variability than the cumulative ADS polygons (Fig. 3.4). Cumulative mortality due to both insects was highly variable among Landsat pixels (30-m), whereas the ADS polygons were more extensive in space, particularly lower-magnitude WSB defoliation. As expected, high NBR slope threshold maps (Figs. 3.4C, D) showed a higher range of mortality and larger cumulative extent than the low NBR slope threshold maps (Figs. 3.4A, B). The Landsat high threshold map also resulted in the commission of some contemporaneous disturbances (e.g., stand-replacing fire in the MPB high map; Fig. 3.4C). In contrast, by definition, the low threshold map omitted higher magnitude impacts attributable to insects (i.e., in locations without fire or logging).

Across the PNW Region, comparisons among the ADS and Landsat-based maps illustrate some key similarities in onset year and differences in number of years mapped (Fig. 3.5). Regarding onset, there was a high level of correspondence between the ADS and Landsat maps. In the difference images between the ADS and Landsat onset maps, a large spike at zero showed that the first year of insect-caused change was identical in both maps (Figs. 3.5A, B). In contrast, the difference images between the ADS and Landsat count maps showed that the Landsat maps captured change in more years than the ADS maps at a given location, reflecting the lasting vegetation change signal captured by LandTrendr temporal segmentation (Figs. 3.5C, D). For both insects, annual counts were highest close to zero (i.e., no difference in number of years), although both insects exhibited a small increase in count near 25 years, indicating the presence of some relatively long-duration Landsat disturbance segments. Finally, the Landsat high and low slope threshold maps were similar across most of the range of count difference but diverged close to zero, as expected, because the low threshold maps exclude relatively short-duration, severe disturbance segments.

### ***Regional comparisons of MPB and WSB timing, extent, and cumulative impacts***

Across the region, although there was high variability in insect activity and impacts, spatiotemporal patterns were evident in both the aerial survey- and Landsat-based maps. Because MPB and WSB were negligible in the Coast Range and Klamath Mountains (Fig. 3.1), we exclude those ecoregions from the following results and discussion.

#### *Recent insect timing and extent*

MPB outbreaks occurred in two distinct phases (Fig. 3.6). An early phase of activity initiated in the Blue Mountains and East Cascades of Oregon during the 1970s and 1980s, respectively. These were the most extensive MPB outbreaks, approaching 10% of the East Cascades ecoregion at the mid-1980s peak (Fig. 3.6F) and exceeding 2 M cumulative ha in both ecoregions according to the ADS data (Fig. 3.C1). A second wave of MPB activity occurred throughout the PNW Region after the year 2000, affecting dry forests in all of the ecoregions except for the Blue Mountains, where MPB had been widespread 30 years earlier according to the ADS data. These latter outbreaks appeared to be synchronous across ecoregions, potentially

driven by exogenous factors such as regional climate (e.g., drought-stressed host trees). At the regional scale, MPB outbreaks did not exceed 1% of the PNW in any given year (Fig. 3.6A).

In all ecoregions except for the East Cascades, WSB extent exceeded MPB extent according to the ADS data (Figs. 3.5, 3.6). Reflecting differences in epidemiology and host susceptibility, WSB outbreaks first appeared in the North Cascades of Washington in the 1970s. WSB also exhibited an apparent spread west from the Blue Mountains and Northern Rockies ecoregions into the East and West Cascades during the 1980s, returning to the North Cascades of Washington during the 1990s and 2000s (Fig. 3.6). Although there was some WSB activity in other dry forest ecoregions, the WSB outbreak that initiated in the Blue Mountains ecoregion in the 1980s and spread into the East and West Cascades was the most extensive insect outbreak in the regional record (Fig. 3.6). According to the ADS, that outbreak exceeded 25% of the Blue Mountains ecoregion (ca. 12 M cumulative ha; Fig. 3.6B, Fig. 3.C1) and 5% of the PNW Region at its peak (Fig. 3.6A). Indeed, WSB affected substantially more area than MPB across all ecoregions except for the East Cascades, where they were similar in extent (Fig. 3.6).

Where they overlapped temporally (after 1984), the Landsat- and ADS-based time series of outbreak extent exhibited similar temporal patterns, but the Landsat-based maps were much smoother from year to year, illustrating stability associated with lasting vegetation change (Fig. 3.6, Fig. 3.C1). In contrast, the ADS maps covered more years (back to 1970) but showed much larger interannual variability, demonstrating higher uncertainty due in part to sporadic observations as well as the ADS focus on the current year's insect activity rather than multiple years of durable change.

Although the Landsat high slope threshold maps were more extensive than the low threshold maps by definition, the differences were small relative to the difference between the ADS and Landsat maps (Fig. 3.6, Fig. 3.C1). We thus displayed the Landsat high slope threshold (Fig. 3.7) to compare regional patterns of insect activity and cumulative impacts with the ADS-based record (Fig. 3.1). As with the landscape-scale example (Fig. 3.4), the regional Landsat-based maps exhibited finer-grained impacts and substantially reduced overall extent (Fig. 3.7).

*MPB and WSB impacts on tree mortality*

By showing the impacts of both MPB and WSB in the same disturbance units (basal area mortality), the Landsat-based maps facilitated comparisons among agents and ecoregions that were not possible with the ADS data alone (Figs. 3.7, 3.8). The temporal patterns of Landsat-based tree mortality were similar to Landsat-based patterns of insect extent but demonstrated higher interannual variability (Figs. 3.6, 3.8). For example, in the North Cascades, the range of MPB-caused mortality (defined by the Landsat high and low thresholds) overlapped the range of WSB-caused mortality (Fig. 3.8C, Fig. 3.D1), whereas the Landsat-based extent estimates did not overlap (Fig. 3.6C, Fig. 3.C1). Similarly, in the East Cascades, WSB-caused mortality exceeded MPB-caused mortality for several years around the year 2000 (Fig. 3.8F), a pattern that was not as pronounced in the extent data (Fig. 3.6F) or in cumulative mortality trajectories (Fig. 3.D1).

Across the region, WSB-associated mortality exceeded MPB-associated mortality from 1985 to 2002, and the two insects had very similar amounts of mortality for the last 10 years of the Landsat record (Fig. 3.8A). As expected due to the previously described extent patterns, about 75% of regional WSB impacts occurred in the Blue Mountains (Fig. 3.8B). The earlier peak in MPB-associated mortality occurred largely in the East Cascades, whereas the later peak occurred in all ecoregions except for the Blue Mountains (Fig. 3.8). Although patterns of cumulative tree mortality were similar between the two insects, the WSB low threshold exceeded the MPB high threshold estimate of cumulative tree mortality in three out of the five ecoregions; cumulative mortality associated with MPB outbreaks was higher only in the East Cascades (Fig. 3.D1).

## DISCUSSION

### *Key aspects of new Landsat-based maps*

Our Landsat-based insect maps complement existing ADS maps in three important ways, thereby enhancing the ecological resolution for analysis, interpretation, and management of insect impacts. Specifically, the Landsat-based maps (1) quantify change in terms of field-measured tree mortality (dead basal area); (2) provide consistent estimates of change for multiple agents, particularly long-duration changes; (3) capture variation of insect impacts at a finer scale (30 m) within ADS polygons, substantially reducing estimated insect extent. Because they are derived from three complementary datasets that span the PNW Region (forest inventory plots, aerial detection surveys, Landsat time series), these new maps facilitate comparisons across space, time, and insect agents that have not been possible to date.

At the forest landscape scale, the fine-grained, Landsat-based insect maps highlight the coarse-scale uncertainties in the ADS polygons, particularly the low damage polygons (Fig. 3.4). For WSB, the much larger ADS extent is evident in all ecoregions (Figs. 3.4, 3.6). This result illustrates the biophysical contrast between transient defoliation detected by ADS observers versus the lasting vegetation impacts identified by the Landsat time series. The more durable changes captured by the Landsat trajectories have important implications for vegetation succession, fuel profiles, and carbon, nutrient, and hydrological cycling.

At the regional scale, the Landsat-based maps substantially reduce the extent and cumulative impacts estimated from the ADS maps (Figs. 3.1, 3.6, 3.7, 3.C1), demonstrating the potential to overestimate insect damage from ADS polygons alone. We suggest that whereas the ADS polygons represent a useful coarse filter to track short-term insect activity, the Landsat trajectories provide a fine filter to identify actual, long-term insect impacts. In this way, Landsat-based maps help to resolve how the ADS maps appear to overestimate regional-scale insect extent while simultaneously underestimating stand-scale effects (Meigs et al. 2011, Meddens and Hicke 2014). Researchers and forest managers could use both types of maps to identify locations across the region where MPB and WSB are currently active (ADS polygons) and where they have had especially severe, cumulative impacts (Landsat trajectories).

Previous mapping studies have focused primarily on MPB and other bark beetle outbreaks, leveraging either ADS (e.g., Williams and Birdsey 2003, Meddens et al. 2012,

Preisler et al. 2012) or Landsat data (e.g., Wulder et al. 2006, Goodwin et al. 2010, Meddens and Hicke 2014). To our knowledge, this study is the first to capture both bark beetle and defoliator impacts in an integrated mapping framework, combining complementary remotely sensed and ground-based data to render fine-grained maps of insect impacts on tree mortality at a regional scale. It is also the first study to quantify and compare MPB and WSB impacts across the PNW Region.

### ***Regional spatiotemporal patterns of MPB and WSB impacts***

MPB outbreaks exhibited both landscape-scale asynchrony (in the early outbreaks) and regional-scale synchrony (in the late outbreaks). The former affected much more area in the PNW Region, creating a large pulse of tree mortality and setting the stage for potential interactions with wildfires (Hicke et al. 2012). The latter may be particularly important in the context of global change and broader insect-climate dynamics across western North America (Raffa et al. 2008, Meddens et al. 2012).

Recent MPB outbreaks have been the subject of intense scientific and public concern, particularly the regional-scale outbreaks in British Columbia and US Rocky Mountains (e.g., Meddens et al. 2012). Although less prominent, recent MPB outbreaks in the PNW have affected a substantial proportion of the ecoregions where host trees (lodgepole pine) are present (Fig. 3.6), resulting in widespread tree mortality (Figs 3.7, 3.8). Preisler et al. (2012) quantified MPB outbreaks in OR and WA, attributing recent activity (ADS dead tree estimates at the km<sup>2</sup> scale) to beetle pressure and climate drivers. Our MPB time series data are consistent with the ADS summaries of Preisler et al. (2012) but focus on ecoregional rather than statewide summaries, as well as the lasting, cumulative impacts on tree mortality due to bark beetles and defoliators. Our fine-grained mortality maps have the potential to aid future analyses of the drivers of MPB and WSB outbreaks – including insect population dynamics, climate, and land use history – from forest stand to regional scales.

Despite the dominance of MPB studies in recent literature on insect outbreaks and mapping, our maps show that the WSB is a very important disturbance agent in the PNW Region. WSB extent exceeds MPB extent in all ecoregions except for the East Cascades, where cumulative extents were similar (Figs. 3.6, 3.C1). Indeed, the 1980s WSB outbreak in the Blue

Mountains dominates the regional record, accounting for much of the insect extent and mortality impacts across all forested ecoregions (Figs. 3.6, 3.8). Although the Landsat-based annual and cumulative MPB impacts initially exceed WSB impacts in the North Cascades (Figs. 3.8C, 3.D1), this is the result of the Landsat record beginning after a widespread earlier WSB outbreak (Figs. 3.6C, 3.C1). Subsequent WSB impacts in the North Cascades exceed MPB impacts, resulting in very similar cumulative mortality by the end of the Landsat record (Figs. 3.8C, 3.D1), highlighting the importance of repeat defoliation events in WSB-prone forests (Flower et al. 2014). These complex WSB and MPB dynamics in the North Cascades – and indeed in all ecoregions – illustrate the importance of accounting for both types of insects, as well as incorporating as many years as possible in future analyses.

### *Uncertainties and future work*

This study advances insect mapping methods by combining regionally extensive field, aerial, and satellite remote sensing datasets in a novel framework. Though spatially and temporally extensive, the datasets we used have particular strengths and weaknesses (e.g., Meigs et al. 2011, Meddens and Hicke 2014), underscoring the need for incorporating multiple, complementary sources of data. Specifically, our approach benefits from the use of CVS field plots for mortality estimates, ADS polygons for attribution/detection of specific insect agents, and LandTrendr time series for fine-grained, consistent change estimates. Nevertheless, many uncertainties remain.

Although this assessment is unprecedented in terms of capturing the annual and cumulative impacts of multiple insects from forest stand to regional scales, it extends only to 1984 for the Landsat-based maps and to 1970 for the ADS-based maps. Recent research demonstrates the potential to extend Landsat time series back to 1972 with MSS data (Pflugmacher et al. 2012), which would enable further comparisons of Landsat and ADS data, as well as the capture of additional insect impacts. Longer-term studies based on tree ring data are vital to assess multiple outbreaks and to compare outbreak trends over time (e.g., Swetnam and Betancourt 1993, Flower et al. 2014). Similarly, although the PNW is a broad, heterogeneous region with many similarities to other regions (e.g., US Rocky Mountains), the datasets we used

may not be as readily available or spatiotemporally consistent elsewhere (Meddens et al. 2012). Future studies thus should investigate the integration of similar field, aerial, and satellite datasets.

In addition, we focused on the two most pervasive insects in the study region (MPB and WSB), but forests of the PNW are influenced by numerous other disturbance agents, including other insects (e.g., fir engraver beetle [*Scolytus ventralis* LeConte {Coleoptera: Curculionidae}], Douglas-fir tussock moth [*Orgyia pseudotsugata* {McDunnough} {Lepidoptera: Lymantriidae}]; Goheen and Willhite 2006), wildfire, diseases, logging, and climate change, among others. Future studies should consider these other agents in a comprehensive attribution analysis. Because our high NBR slope threshold resulted in commission of some non-insect disturbances, and our low NBR slope threshold omitted some insect disturbances (Fig. 3.4), attribution based solely on spectral change collocated with ADS damage may be insufficient. Future studies could incorporate ancillary data on other disturbance types (e.g., fire and logging inventories) and spatial patterns (e.g., patch size and shape), as well as known insect drivers such as drought stress and host tree distribution and structure. It is particularly challenging to differentiate the effects of insects alone versus the combined effects of insect outbreaks and management actions, as both insects and humans effectively thin forest stands. Although we excluded CVS inventory plots with evidence of anthropogenic disturbance from our basal area mortality model, our approach still includes pixels within ADS polygons that may have experienced thinning and/or salvage harvest, management responses that have been typical in the study area (e.g., Waring et al. 1992, Azuma 2010). Finally, our approach limited the Landsat-based maps to pixel trajectories intersected in at least one year by ADS polygons; future work could include disturbance pixels outside of those locations, as well as post-disturbance spectral recovery trajectories (Kennedy et al. 2010, Meigs et al. 2011).

### ***Management implications***

This regional-scale assessment provides novel insights regarding the spatial and temporal patterns of tree mortality associated with mountain pine beetle and western spruce budworm outbreaks. Both native insects have been very active in the dry interior conifer forests of the PNW, and it will be increasingly crucial to monitor their impacts on forest health in the coming years. Our new maps based on multiple, integrated, regional-scale datasets provide an important

context for current forest composition and structure, informing potential forest management actions in these insect-prone forests.

By capturing the effects of both MPB and WSB and constraining mapped impacts to locations where substantial tree mortality has occurred, our analysis also identifies particular hotspots of insect activity where both bark beetles and defoliators have been active. These mixed-conifer forest stands and landscapes, particularly in the Blue Mountains of Oregon and the East and North Cascades of Washington, may warrant special management attention. Recent insect outbreaks in these forests may also amplify the need for proactive ecosystem restoration (Hessburg et al. 2000). The role of these recent outbreaks in subsequent fire probability or severity is a particularly important management concern (Hicke et al. 2012), as insects and wildfires have been and will continue to be fundamental disturbance processes in North American forests (e.g., Littell et al. 2009, Bentz et al. 2010, Littell et al. 2010).

## ACKNOWLEDGEMENTS

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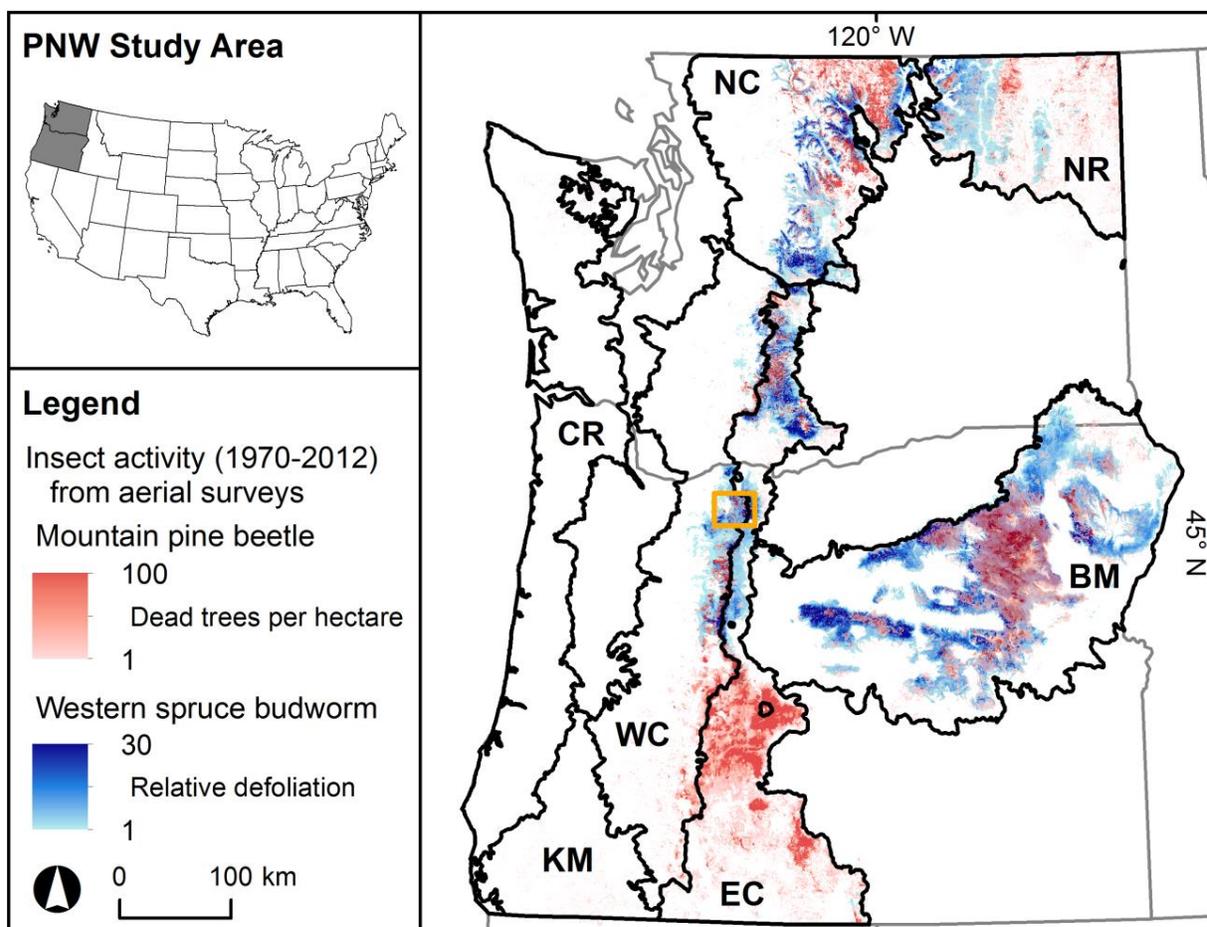
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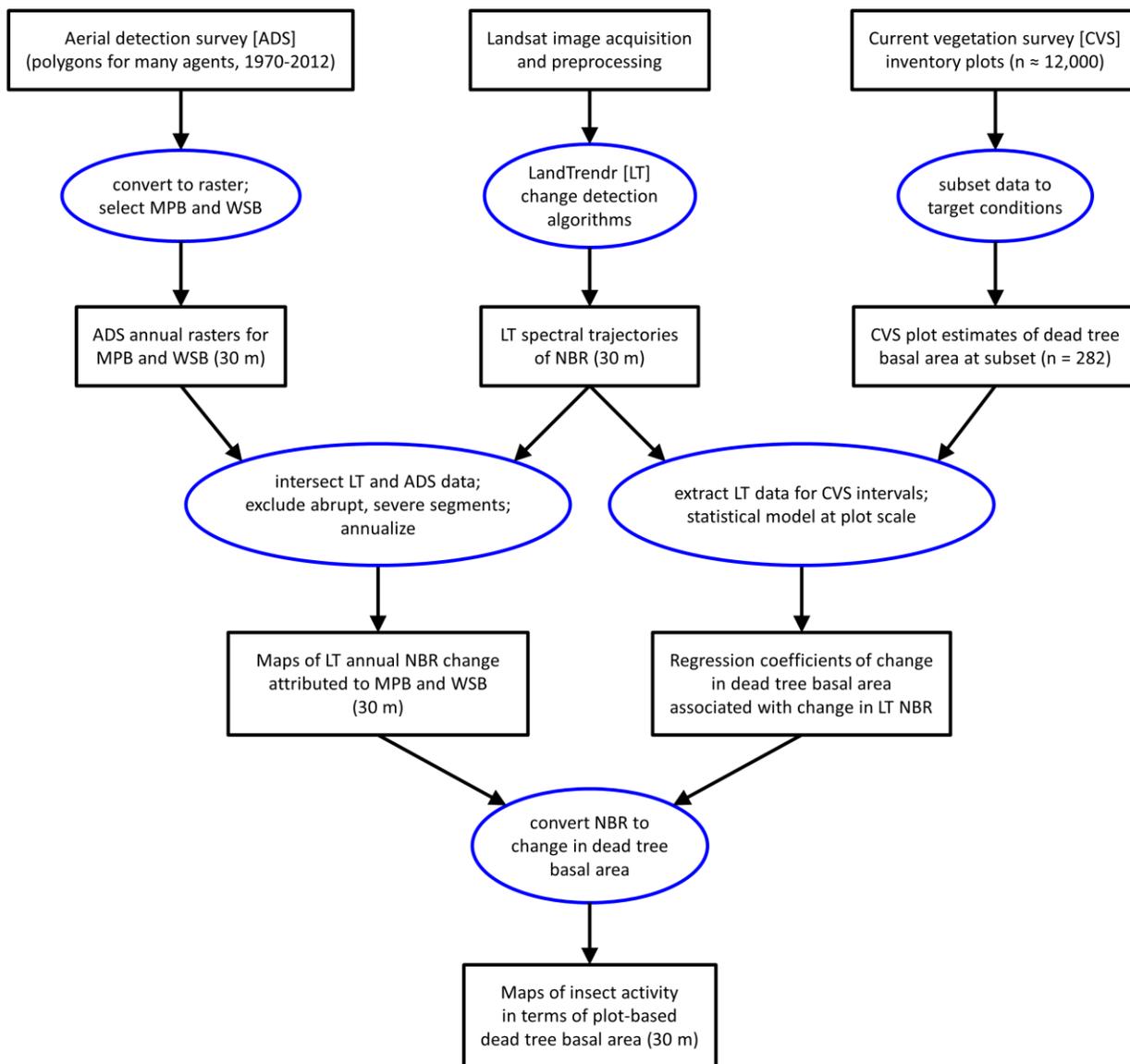
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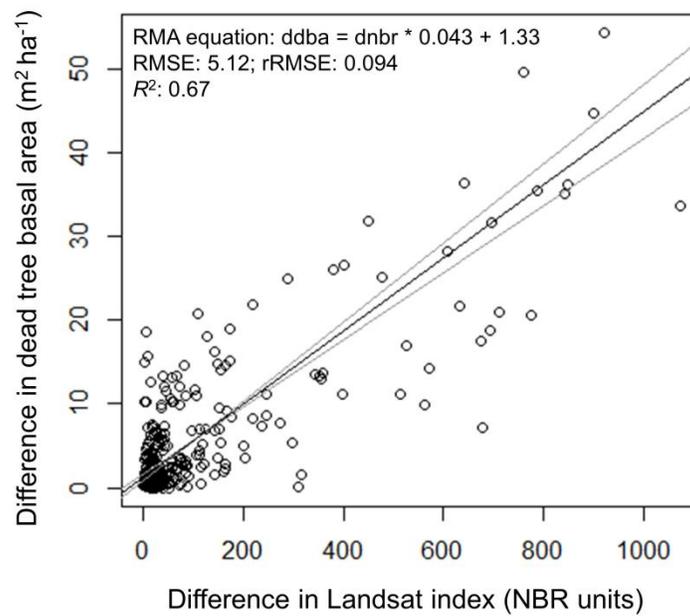
## FIGURES



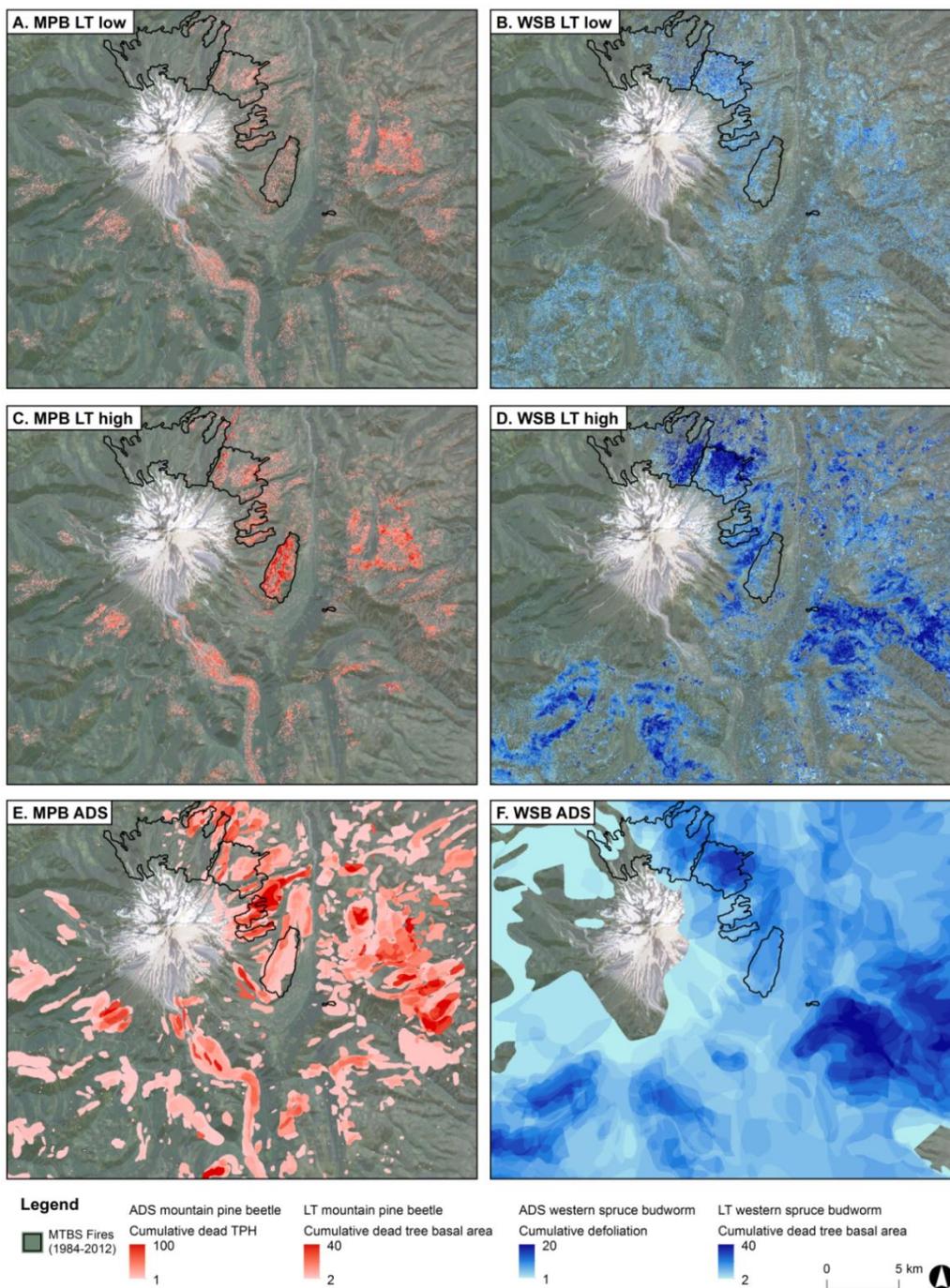
**Fig. 3.1.** Study area map. This study focuses on insect activity within the forested ecoregions of the Pacific Northwest (PNW; Oregon and Washington; inset). Total area of forested ecoregions is ~25 M ha, and area affected by either insect is ~8 M ha according to ADS data (1970-2012). Note that MPB and WSB have been active primarily in the interior mixed-conifer forests and are generally absent in the CR, KM, and much of the WC ecoregions. Note also that MPB overlaps WSB activity in this display (MPB shown with 30% transparency), revealing purple locations where both insects have been active. Orange rectangle denotes extent of Fig. 3.4. Ecoregion (Omernik 1987) abbreviations: NC: North Cascades; NR: Northern Rockies; CR: Coast Range; BM: Blue Mountains; WC: West Cascades; EC: East Cascades; KM: Klamath Mountains. Note that the other ecoregions are not labeled and that we include only the portions of ecoregions within the two states.



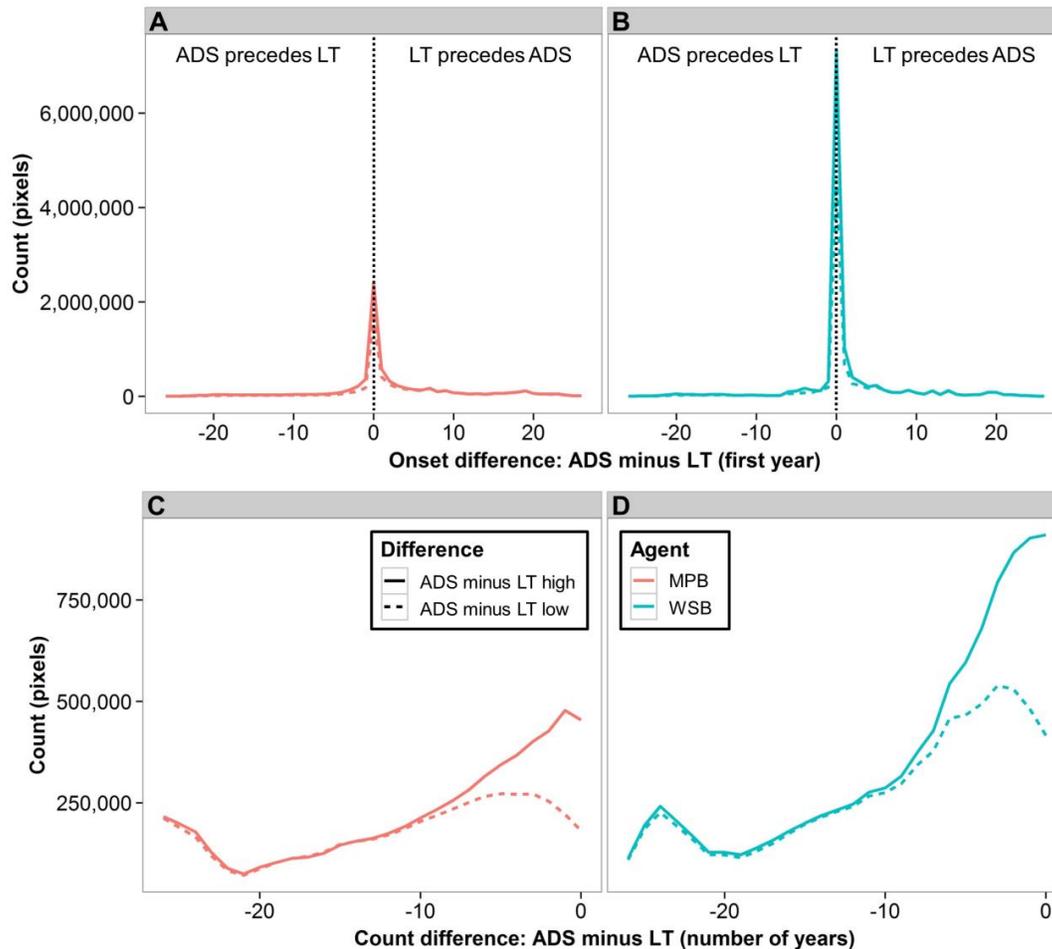
**Fig. 3.2.** Datasets and processing steps for Objective 1. See Methods for details. Abbreviations: ADS: aerial detection survey; CVS: current vegetation survey; MPB: mountain pine beetle; WSB: western spruce budworm; LT: LandTrendr change detection (Kennedy et al. 2010); NBR: normalized burn ratio (Key and Benson 2006). Note that at the end of the process there are actually four maps: high and low NBR slope thresholds for both MPB and WSB.



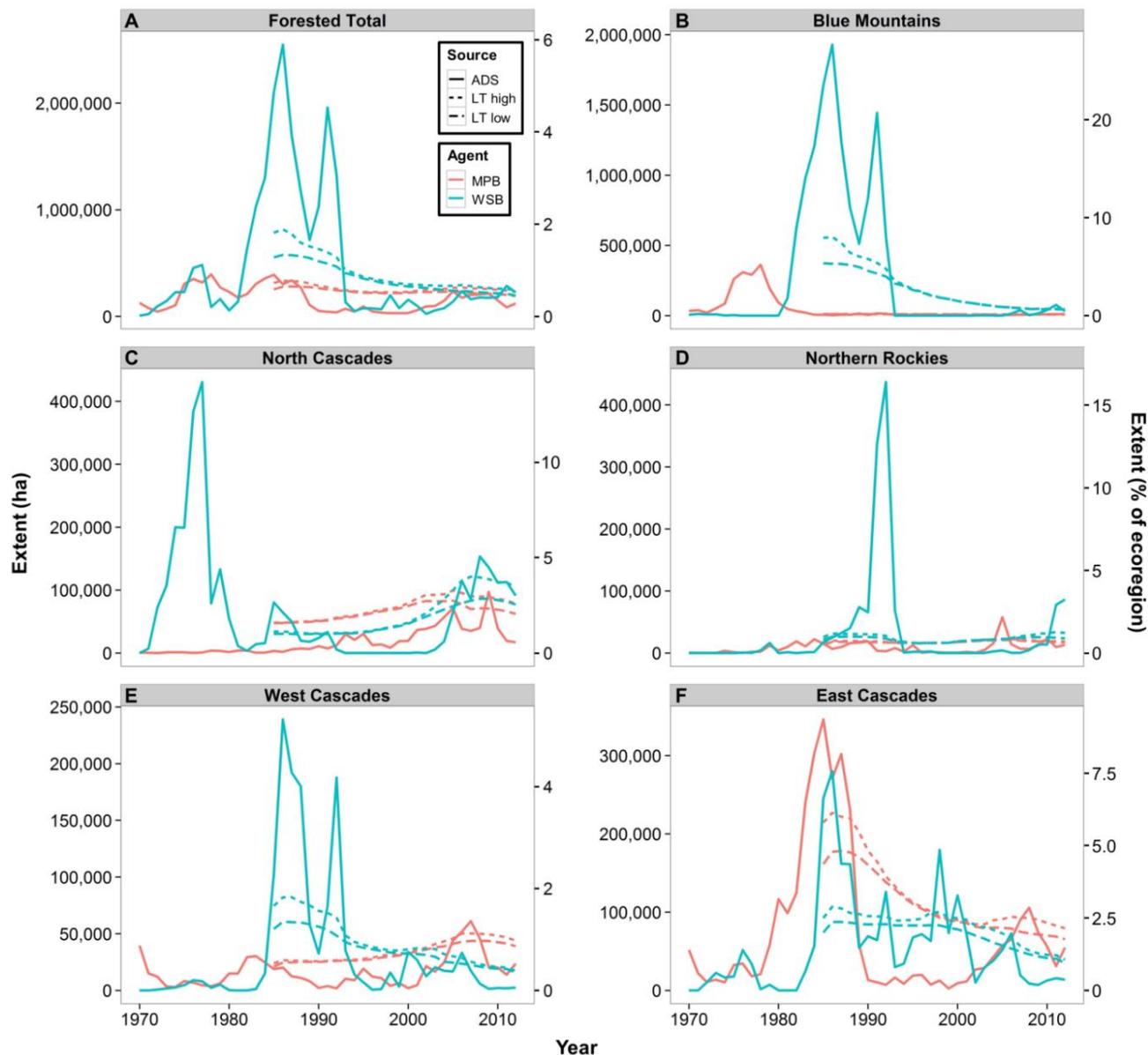
**Fig. 3.3.** Plot-scale relationship between LandTrendr-based spectral change and field-based dead tree basal area change (RMA regression line with 95% CI). See Methods for criteria used to subset CVS inventory plots from the overall database ( $n \approx 12,000$ ) to this refined sample focusing on disturbance associated with MPB or WSB activity according to aerial surveys ( $n = 282$ ).



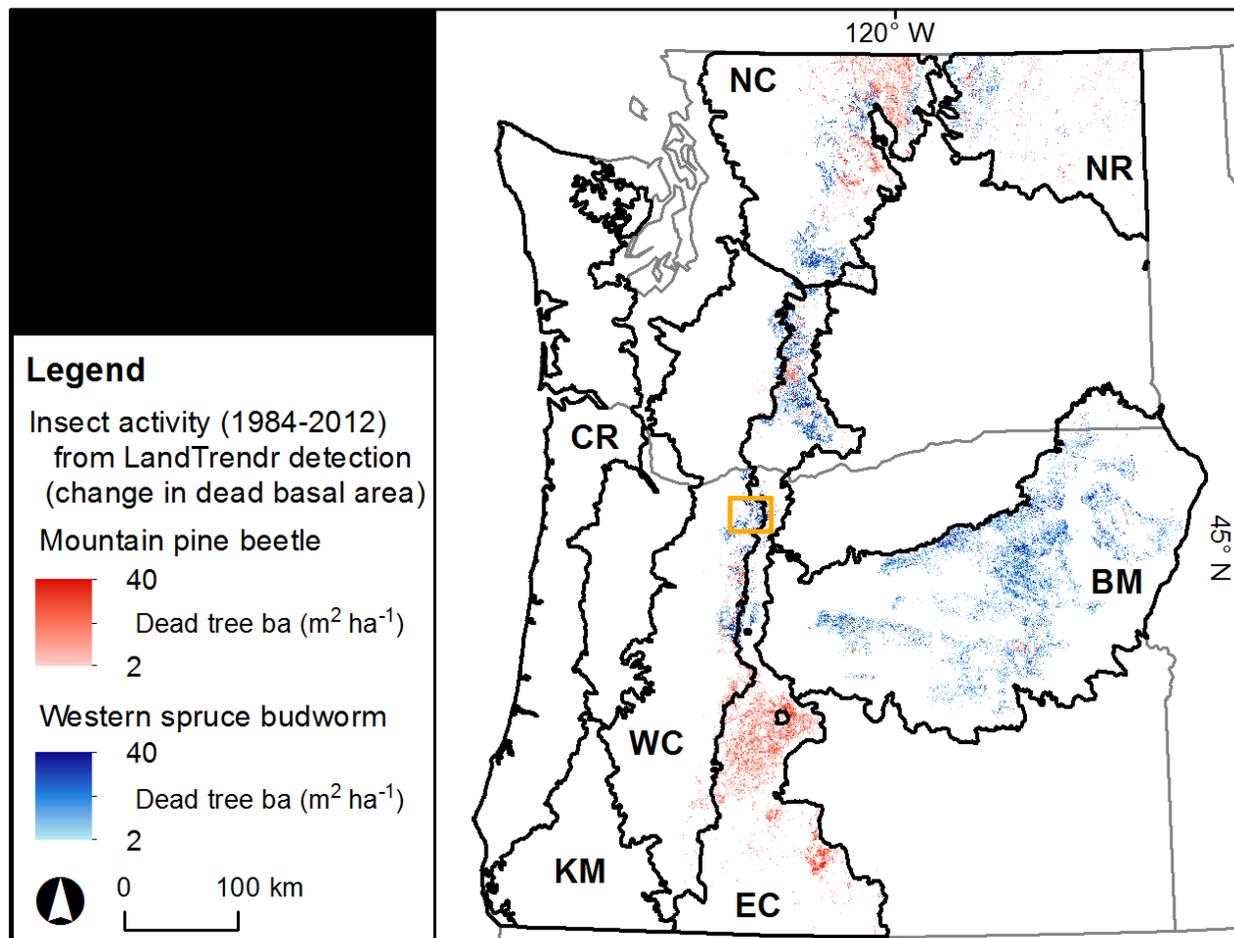
**Fig. 3.4.** Example landscape maps comparing cumulative impacts (1984-2012) of mountain pine beetle (MPB) and western spruce budworm (WSB) according to aerial detection surveys (ADS) (E, F) and two thresholds of LandTrendr (LT). This Mt. Hood area has representative examples of both insects in a centrally located landscape (Fig. 3.1). Fire perimeters from Monitoring Trends in Burn Severity (MTBS; <http://mtbs.gov>) show potential for fires to be included in insect maps at the high LT slope threshold. Note that LT insect maps are constrained within ADS polygons but exhibit finer-grained detail and change estimates in consistent tree mortality units. Base maps: World Terrain Base and ESRI Imagery World 2D from <http://server.arcgisonline.com>.



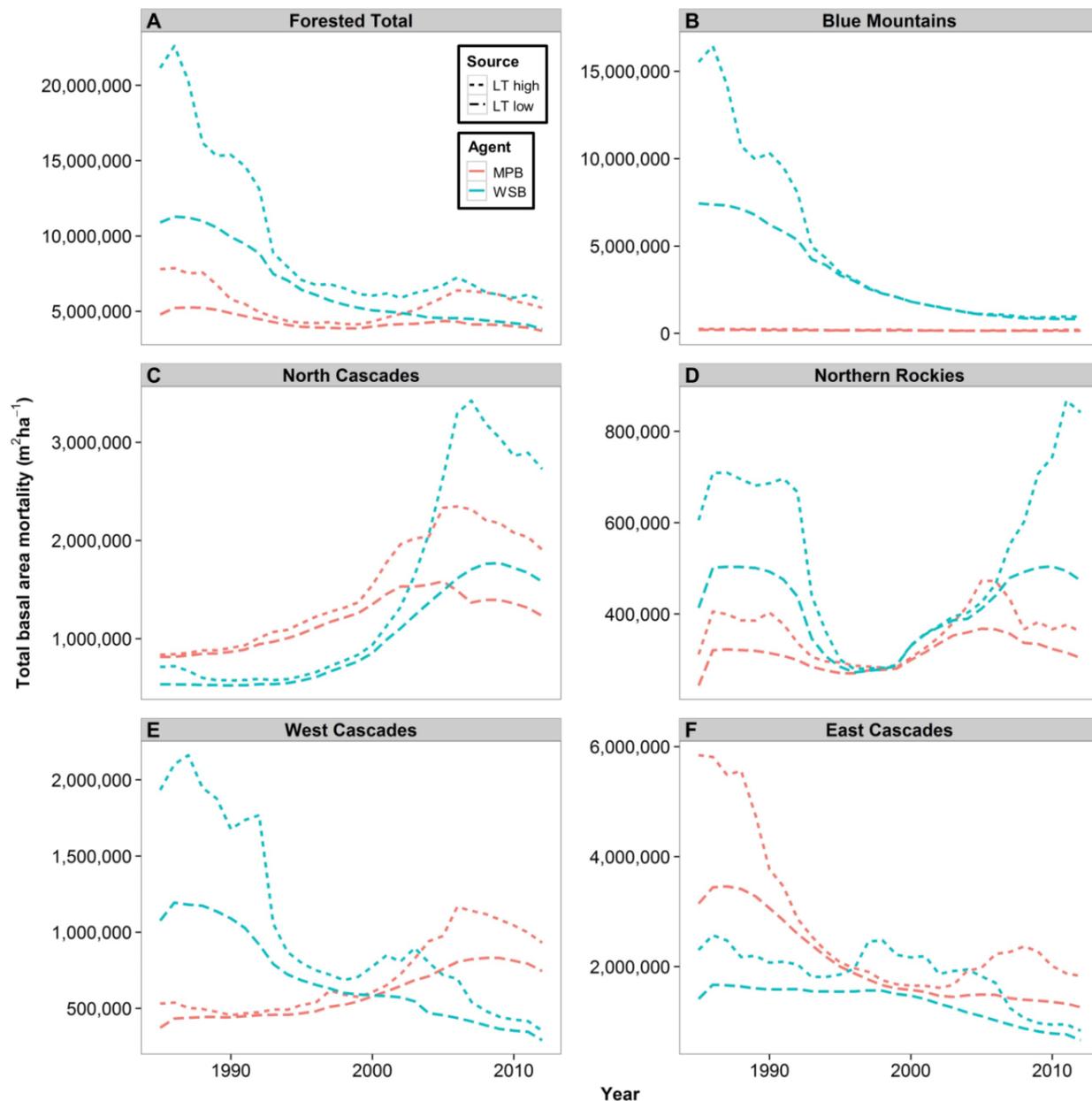
**Fig. 3.5.** Difference of onset year and count (number of years) among aerial detection survey (ADS) and LandTrendr (LT) maps of MPB (A, C) and WSB (B, D). Note that these comparisons exclude locations with ADS detection only as well as the first year of the Landsat time series (1984-1985), thereby focusing on locations and years of mutual coverage (1986-2012 in both data sources). LT high refers to the high NBR slope threshold; LT low refers to the low NBR slope threshold (See Methods).



**Fig. 3.6.** Temporal patterns of extent (ha and percent of ecoregion) of recent MPB and WSB activity according to aerial detection survey (ADS; 1970-2012) and two thresholds of LandTrendr change detection (LT; 1984-2012) across forested ecoregions of the PNW. LT high refers to the high NBR slope threshold; LT low refers to the low NBR slope threshold (See Methods). Note that the “Forested Total” area is the sum of insect activity across all ecoregions and that the percentage is out of the full PNW Region, but the Coast Range and Klamath Mountains ecoregions are not shown here due to the absence of MPB and WSB activity. Note also that the y axes vary by ecoregion. See Fig. 3.C1 for cumulative extent. See Figs. 3.1 and 3.7 for regional spatial patterns.



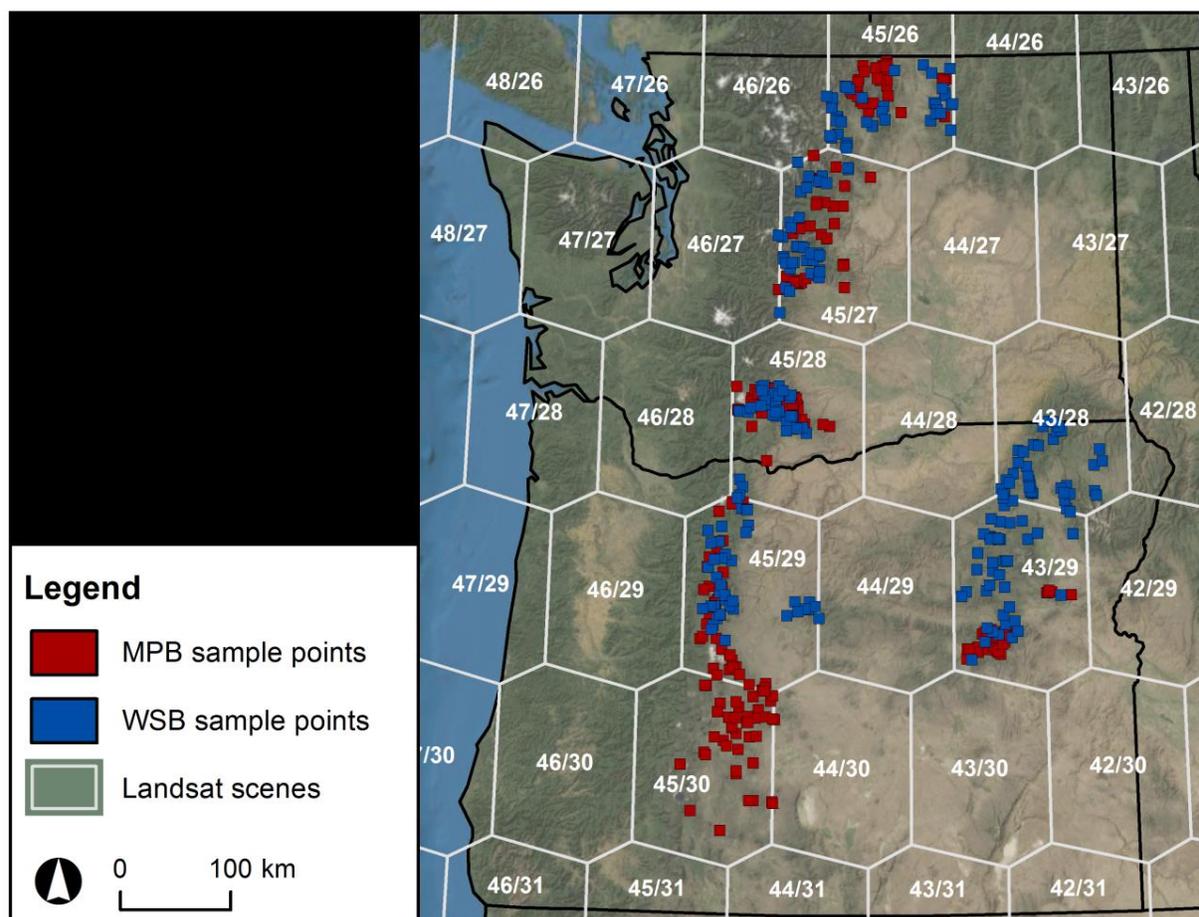
**Fig. 3.7.** Spatial patterns of cumulative insect effects on tree mortality (dead tree basal area; 1984-2012) from LT high threshold maps (See Fig. 3.4) across the PNW. Note that MPB overlaps WSB activity in this display, and both are much less extensive than the analogous aerial survey-based maps (Figs. 3.1, 3.4, 3.6). Note also the absence of MPB and WSB in the CR, KM, and most of the WC ecoregions. Orange rectangle denotes extent of Fig. 3.4. Temporal patterns shown in Fig. 3.8. Ecoregion (Omernik 1987) abbreviations: NC: North Cascades; NR: Northern Rockies; CR: Coast Range; BM: Blue Mountains; WC: West Cascades; EC: East Cascades; KM: Klamath Mountains. Note that we include only the portions of ecoregions within the two states.



**Fig. 3.8.** Temporal patterns of insect effects on tree mortality (dead tree basal area) from LandTrendr change detection (LT; 1984-2012) across forested ecoregions of the PNW. LT high refers to the high NBR slope threshold; LT low refers to the low NBR slope threshold (See Methods). Note that the “Forested Total” area is the sum of insect activity across all ecoregions and that the percentage is out of the full PNW Region, but the Coast Range and Klamath Mountains ecoregions are not shown here due to the absence of MPB and WSB activity. Note also that the y axes vary by ecoregion. Spatial patterns shown in Fig. 3.7. Cumulative effects shown in Fig. 3.C1.

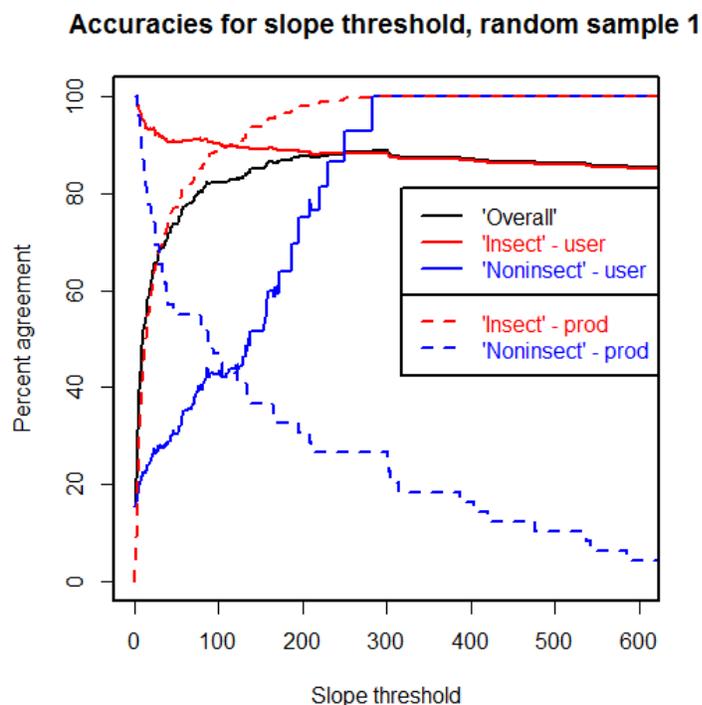
## CHAPTER APPENDICES

**Fig. 3.A1.** Map of stratified random sample plots in seven Landsat path/row scenes with MPB or WSB activity according to ADS across the PNW.



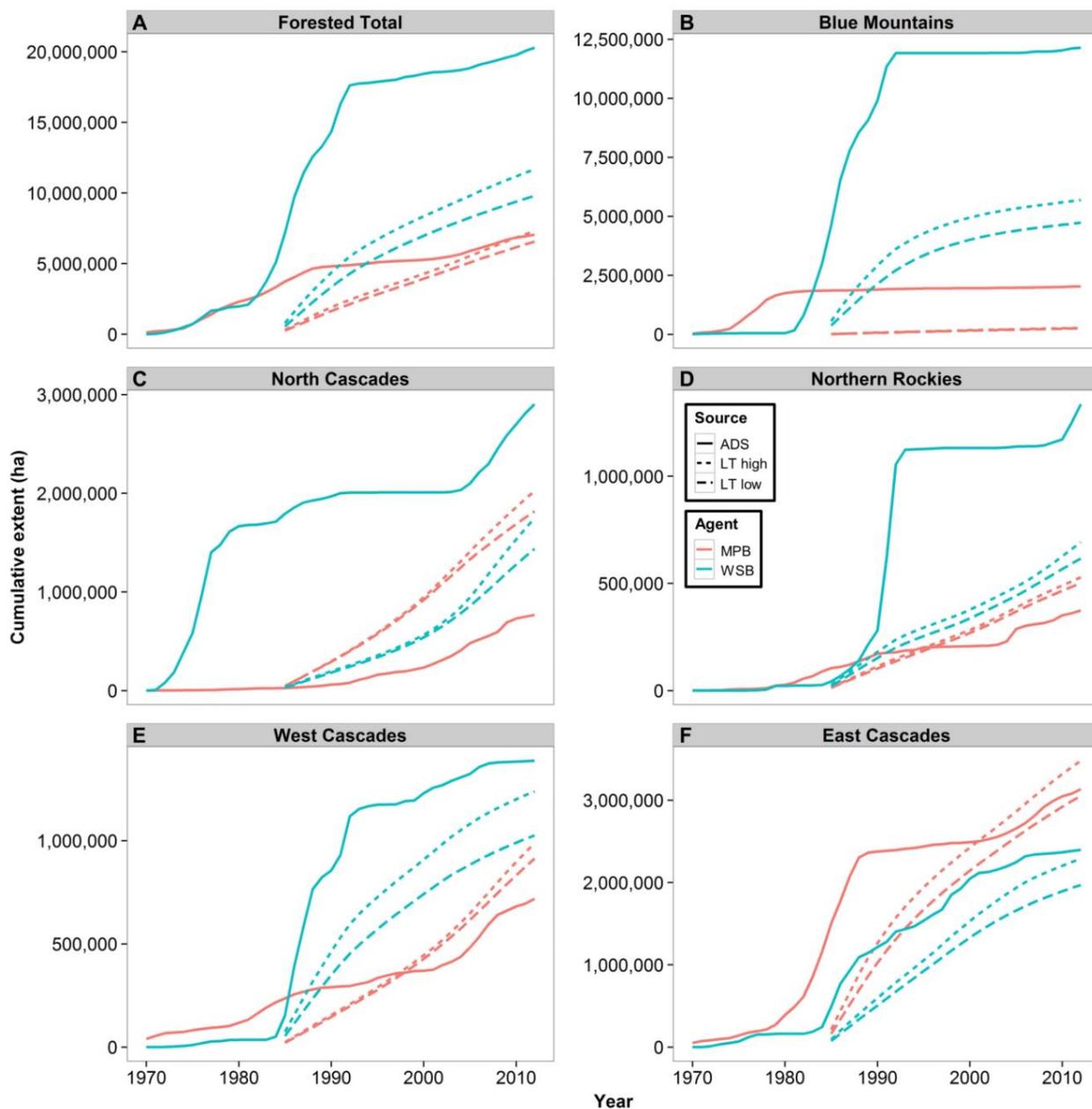
*Notes:* 240 plots for each agent were combined and assessed for evidence of insect and noninsect disturbance to develop low and high thresholds for inclusion in the final LandTrendr-based maps (Fig. 3.B1). See Methods for further details. 35 Theissen scenes (polygons account for scene overlap) are intersected by OR or WA. Base maps: World Terrain Base and ESRI Imagery World 2D from <http://server.arcgisonline.com>.

**Fig. 3.B1.** Example of thresholds for simple attribution of presumed ‘insect’ and ‘noninsect’ LandTrendr disturbance segments intersected by aerial survey polygons.



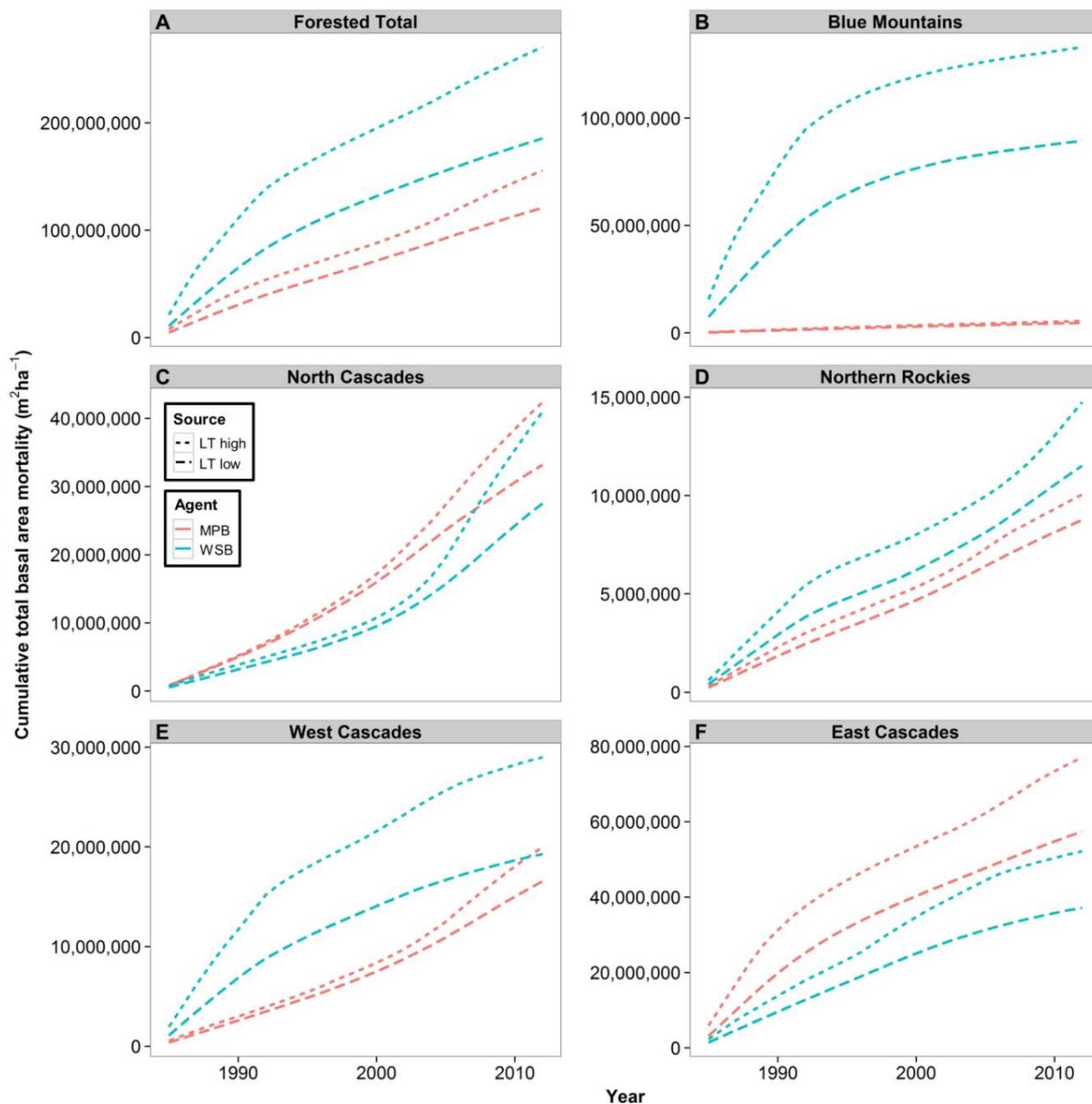
*Notes:* ‘Overall’ classification accuracy is sensitive to imbalanced sample size. The low slope threshold (29) is defined as the intersection of the two producer’s accuracy curves and includes only lower slope values in the insect class. The high slope threshold (239) is defined as the intersection of the two user’s accuracy curves and includes many more segments. Final thresholds were averaged from 10 iterations where 2/3 of the sample set ( $n = 320$ ) was used for training and 1/3 ( $n = 160$ ) for validation.

**Fig. 3.C1.** Cumulative extent (ha) of recent MPB and WSB activity according to aerial detection survey (ADS; 1970-2012) and two thresholds of LandTrendr change detection (LT; 1984-2012) across forested ecoregions of the PNW.



*Notes:* LT high refers to the high NBR slope threshold, and LT low refers to the low NBR slope threshold (See Methods). Note that the “Forested Total” area is the sum of insect activity across all ecoregions in the PNW, but the Coast Range and Klamath Mountains ecoregions are not shown here due to the absence of MPB and WSB activity. Note also that the y axes vary by ecoregion.

**Fig. 3.D1.** Temporal patterns of cumulative insect effects on tree mortality (dead tree basal area) from LandTrendr change detection (LT; 1984-2012) across forested ecoregions of the PNW.



*Notes:* LT high refers to the high NBR slope threshold; LT low refers to the low NBR slope threshold (See Methods). Note that the “Forested Total” area is the sum of insect activity across all ecoregions in the PNW, but the Coast Range and Klamath Mountains ecoregions are not shown here due to the absence of MPB and WSB activity. Note also that the y axes vary by ecoregion. Spatial patterns shown in Fig. 3.7. Annual effects shown in Fig. 3.8.

## CHAPTER 4 || DO INSECT OUTBREAKS INCREASE WILDFIRE LIKELIHOOD IN CONIFER FORESTS?

### ABSTRACT

Insects and wildfires are important forest disturbances, and both are projected to increase with anthropogenic climate change. Although there is acute concern that insect-caused tree mortality increases the likelihood or severity of subsequent wildfire (i.e., linked disturbance interactions), previous research has been equivocal, with results often based on individual fire or insect events. This study investigates wildfire likelihood following the two most prevalent insects in the Pacific Northwest (PNW) – mountain pine beetle (MPB; *Dendroctonus ponderosae*; a bark beetle) and western spruce budworm (WSB; *Choristoneura freemani*; a defoliator). Working at landscape and regional scales across numerous insect and fire events, we leverage seamless census data to (1) summarize the extent and overlap of MPB, WSB, and wildfire across forested ecoregions of the PNW and (2) quantify the likelihood of wildfires occurring in areas with and without recent insect activity across different insect agents, ecoregions, time lags, and fire sizes. We first map annual fire extent (1984-2012) and cumulative insect extent (1970-2012) for each fire year. We then compute wildfire likelihood with two-way binary matrices between fire and insects, testing for paired differences in fire extent with and without prior insect activity (Wilcoxon signed-rank test). All three disturbance agents have occurred primarily in the drier, interior conifer forests east of the Cascade Range, with recent wildfires extending through the southern West Cascades and Klamath Mountains. MPB and WSB outbreaks occur over multiple years at the ecoregion scale, whereas wildfires occur more stochastically. In general, WSB extent exceeds MPB extent, which in turn exceeds wildfire extent, and each disturbance typically affects less than 2% annually of a given ecoregion. In recent decades across the PNW, wildfire likelihood is not consistently higher or lower in forests with prior insect outbreaks, but there is evidence of linked interactions that vary across insect agent (MPB and WSB), space (ecoregions), and time (interval since insect onset). For example, fire likelihood is higher following MPB activity in the North Cascades and West Cascades, particularly within the past 10 years, whereas fire likelihood is lower at various time

lags following MPB in the Northern Rockies, East Cascades, and Blue Mountains. In contrast, fire likelihood is lower following WSB outbreaks in the Northern Rockies and Blue Mountains and at multiple time lags across all ecoregions. In addition, there are no consistent relationships between insect-fire likelihood and interannual fire extent, suggesting that other factors (such as climate) control the disproportionately large fire years accounting for the majority of regional fire extent. Although insects and wildfires do not appear to overlap enough to facilitate consistently positive linked disturbance interactions, specific fire events and years – such as 2003 and 2006 in the North Cascades – demonstrate high insect-fire co-occurrence and potential compound disturbance effects at the landscape scale. Given concerns about forest health in a rapidly changing climate, long-term monitoring will enable forest managers to quantify and anticipate the independent and interactive effects of insects, wildfires, and other disturbances.

#### KEYWORDS

bark beetle; defoliator; disturbance interactions; fire; forest health; insect outbreak; mountain pine beetle; Pacific Northwest; tree mortality; western spruce budworm

## INTRODUCTION

Insect outbreaks and wildfires are important agents of change in forest ecosystems, shaping forest pattern and process throughout the world. In western North America, insects and wildfires are both native disturbances that have influenced forests for millennia, both have been widespread in recent years, and both are projected to increase with anthropogenic climate change (e.g., Swetnam and Lynch 1993, McKenzie et al. 2004, Raffa et al. 2008, Heyerdahl et al. 2008, Littell et al. 2010). Recognizing that disturbances do not function in isolation, ecologists have focused increasingly on the complex interactions among multiple disturbances (e.g., Paine et al. 1998, Kulakowski and Veblen 2007, Simard et al. 2011). Indeed, recent bark beetle outbreaks and acute concerns about forest health have prompted numerous studies on the interactions between insects and wildfires (Negrón et al. 2008, Hicke et al. 2012, Jenkins et al. 2012). Results to date have been mixed, however, with evidence of positive (e.g., Perrakis et al. 2014, Prichard and Kennedy 2014), negative (e.g., Lynch and Moorcroft 2008, Simard et al. 2011), and neutral (e.g., Crickmore 2011, Kulakowski and Jarvis 2011) effects of insects on subsequent fire likelihood, behavior, or severity.

Much of the ambivalence in the literature can be attributed to differences in the particular insect of interest, forest type, fire weather, type and spatiotemporal scale of analysis, and response and predictor variables evaluated. In addition, most studies have been geographically limited, focusing on individual fires or insect outbreaks on a case by case basis (Bisrat 2010). A more systematic, regional approach is necessary to elucidate general system behavior across numerous insect and fire events. In this study, we integrate robust, spatially and temporally seamless insect and fire census data to assess insect-wildfire interactions across multiple spatiotemporal scales in a region where improving forest health and resilience is a high priority for managers and policy makers (Hessburg et al. 2000).

Ecological theory provides the context to conceptualize and analyze disturbance interactions. A recent assessment of bark beetle effects on fuels and simulated fire behavior in the Rocky Mountains presents a framework for evaluating “compound” versus “linked” disturbance interactions (Simard et al. 2011). Whereas the concept of compound disturbances (*sensu* Paine et al. 1998) focuses on the synergistic, emergent ecological effects and unpredictable responses of disturbance events occurring in rapid succession, the concept of

linked disturbances emphasizes the effects that one disturbance may have on the likelihood, severity, or extent of a subsequent disturbance (Simard et al. 2011, Harvey et al. 2013). Here, we investigate linked disturbance interactions, focusing on wildfire likelihood following outbreaks of the two most prevalent insects in the Pacific Northwest Region – mountain pine beetle (MPB; *Dendroctonus ponderosae* Hopkins [Coleoptera: Scolytidae]; a bark beetle) and western spruce budworm (WSB; *Choristoneura freemani* Razowski [Lepidoptera: Tortricidae]; a defoliator).

Most recent analyses of insect-fire interactions focus on fire effects in individual wildfire events (e.g., Lynch et al. 2006, Kulakowski and Veblen 2007, Prichard and Kennedy 2014) or simulate fire behavior and/or effects in insect-altered forests assuming they burn (e.g., Page and Jenkins 2007, Simard et al. 2011). To address concerns that forest managers and policy makers have about linked disturbances, it is first necessary to determine how often insects and wildfires actually overlap at landscape and regional scales. Although they are generally widespread disturbances, insects and wildfires still affect only a small portion of forested landscapes in a given year, even in regional outbreak or fire years (Littell et al. 2009, Meigs et al. 2014 [This dissertation, Chapter 3]). Thus, their relative rarity reduces the likelihood of overlap as well as the detectability of linked disturbance effects. Moreover, because forest susceptibility to both insect outbreaks and wildfires is associated with the same abiotic and biotic drivers (e.g., drought, forest structure, and landscape contiguity; Meigs et al. 2011 [This dissertation, Chapter 2], Simard et al. 2011, Prichard and Kennedy 2014), they likely coincide with or without a strong mechanistic link. This study quantifies the extent and interannual variation of insect-fire overlap since 1984 across all forested ecoregions of Oregon and Washington (hereafter “PNW”).

MPB and WSB, respectively, are the two most prevalent species of bark beetle and defoliator in the PNW, and both have irrupted in extensive outbreaks in recent decades (Williams and Birdsey 2003, Meigs et al. 2014). Outbreaks of both insects are associated with endogenous (e.g., insect population dynamics, host tree vigor and distribution) and exogenous factors (e.g., short- and long-term climate variability; Sanders et al. 1985, Waring and Pitman 1985, Waring et al. 1992, Raffa et al. 2008, Bentz et al. 2010, Meigs et al. 2014). During sustained outbreaks, both insects can cause pervasive tree mortality and associated changes in forest composition and fuel structure, although the mechanisms and rapidity differ (Swetnam and Lynch 1993, Raffa et

al. 2008, Meigs et al. 2011). Time since insect outbreak is a particularly important factor for fuel succession and associated fire potential, as dead trees transfer through characteristic “red”, “gray”, and “silver” stages with highly variable effects on surface, ladder, and crown fuels and associated fire potentials (e.g., Hicke et al. 2012, Donato et al. 2013).

Although generally similar in that both insects can cause pervasive tree mortality, there are also important differences between MPB and WSB biology, host tree distribution, and recent dynamics. MPB adults mass attack the cambium of host trees (*Pinus spp.*, especially mature lodgepole pine [*P. contorta* Douglas ex Louden]; Goheen and Willhite 2006, Raffa et al. 2008), causing variable but relatively rapid tree mortality across large areas (e.g., recent regional-scale outbreaks in British Columbia, US Rocky Mountains, and PNW [Meddens et al. 2012, Preisler et al. 2012, Meigs et al. 2014]). Although MPB effects on fuels and fire potential have been studied intensively, empirical evidence of linked disturbance interactions has been weak or lacking (e.g., Negrón et al. 2008, Kulakowski and Jarvis 2011, Simard et al. 2011, Hicke et al. 2012, but see Perrakis et al. 2014, Prichard and Kennedy 2014).

Illustrating a very different life history, WSB larvae typically consume the current year’s foliage of host trees (primarily understory true fir [*Abies spp.*], spruce [*Picea spp.*], and Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco]; Sanders et al. 1985, Goheen and Willhite 2006), and they have received less research attention. Although WSB impacts typically are more transient than MPB impacts, WSB defoliation also can result in widespread tree mortality – often in conjunction with bark beetles (Goheen and Willhite 2006, Meigs et al. 2014) – and altered fuel profiles (Hummel and Agee 2003). Several studies have analyzed WSB interactions with wildfire, finding negative feedbacks in British Columbia (Lynch and Moorcroft 2008), limited effects on simulated fire behavior in Washington (Hummel and Agee 2003), and no detectable effects on burn severity in Oregon (Crickmore 2011). Here, we assess the potential linked disturbance effects of both insects at different time lags in an integrated framework.

Since the mid-1980s, wildfire extent in western North America has increased in association with longer, hotter fire seasons (Westerling et al. 2006, Littell et al. 2009), and policy makers are particularly concerned about the vulnerability of human communities in dry forests at the wildland-urban interface (Kennedy and Johnson 2014). If insect outbreaks further increase the likelihood of wildfire, particularly in close proximity to people and valued resources, then

these disturbance interactions have profound implications for forest management. In the PNW, a pulse of large wildfires following recent MPB and WSB outbreaks and regional-scale, seamless datasets provide a unique opportunity to characterize linked disturbance interactions in a retrospective, empirical framework. Working at landscape and regional scales across numerous insect and fire events, our specific objectives are to (1) summarize the extent and overlap of MPB, WSB, and wildfire across forested ecoregions of the PNW region and (2) quantify the likelihood of wildfires occurring in areas with and without recent insect activity across different insect agents, ecoregions, time lags, and fire sizes from 1984 to 2012.

For our first objective, we hypothesize that (1a) all three disturbances occur primarily in dry interior conifer forests east of the crest of the Cascade Range due to favorable climate, vegetation composition and structure, and ignitions and that (1b) insect outbreaks have occurred in spatiotemporally autocorrelated episodes at the ecoregion scale due to their synchronous epidemiology, whereas wildfires have occurred stochastically due to variability in ignition and fire management response. For our second objective, we hypothesize that (2a) wildfires are more likely in areas with recent insect activity due to relatively strong insect effects on fuels versus other fire drivers (e.g., ignitions, topography, weather, climate, management) and that (2b) wildfire likelihood varies with insect agent and time since outbreak due to differences in the cumulative severity and timing of insect effects on fuel profiles and altered fire potential.

## METHODS

### *Study area and design*

The Pacific Northwest The Pacific Northwest is a geographically diverse region with ecologically, economically, and socially important forest landscapes. Mixed-conifer forests are widespread, and their composition and structure vary across gradients of climate, topography, soil parent material, and disturbance and management history (Franklin and Dyrness 1973, Hessburg et al. 2000, Meigs et al. 2014). Although precipitation and temperature regimes are variable across the region, a common climatic feature is that little precipitation occurs in summer months (Franklin and Dyrness 1973). These warm, dry summer conditions, coupled with

abundant biomass and slow decomposition, are conducive to periodic insect and wildfire disturbances (e.g., Littell et al. 2010).

In general, PNW forests occur in remote, mountainous terrain and are managed primarily by the US Forest Service. PNW forests are also defined in part by high variability in historic fire frequency, effects, and patch dynamics typical of mixed-severity fire regimes (Hessburg et al. 2007, Tepley et al. 2013). These forest landscapes have undergone dramatic historical and recent anthropogenic changes, including widespread fire suppression and exclusion, logging, and grazing, which have increased the potential for landscape-to-regional-scale outbreaks of native bark beetles and defoliators as well as wildfires (Hessburg et al. 2000, Meigs et al. 2014).

This observational study is retrospective, leveraging spatially and temporally extensive datasets encompassing all forested lands of the PNW. We define our scope of inference as the generally forested ecoregions (Omernik 1987), which contain ~20,000,000 ha of forest, and the years covered by the census data (1970-2012 for insect data; 1984-2012 for fire data; Fig. 1). For the fire likelihood analysis (below), our scope is limited to those years in which fire occurred in a given ecoregion. Given the widespread extent of similar geographic conditions, vegetation types, and anthropogenic pressures, the recent insect and wildfire dynamics in PNW forests are broadly representative of contemporary disturbance regimes in conifer forests of western North America.

### ***Insect data from aerial detection surveys (ADS)***

We accounted for MPB and WSB activity with geospatial data from the ADS, a cooperative effort between federal and state agencies conducted nationally and administered at the regional scale. These surveys have been flown since 1947 in US Forest Service Region 6 (Oregon and Washington; data available online: <http://www.fs.usda.gov/main/r6/forest-grasslandhealth/>), with observers in fixed-wing aircraft recording a wide range of forest insects and diseases based on aerial signatures (Ciesla 2006). Although ADS data provide an unmatched record of insects and other forest disturbances, there are important limitations related to the spatial and temporal variability of observation conditions, methodology, and personnel (Meigs et al. 2011, Preisler et al. 2012, Meddens et al. 2012). Due to these uncertainties, ADS data are not intended for assessment of specific points on the ground but rather landscape-to-

regional patterns and trends (Ciesla 2006, Meigs et al. 2011). To this end, we focus on insect patterns at the ecoregional and regional scales and integrate multiple years of cumulative ADS damage to estimate prefire insect activity.

We utilized ADS data on MPB and WSB from 1970 to 2012 (Fig. 1), converting the polygons to raster format (30-m grain) for further analysis. For each affected pixel, we identified the onset (first year of detection), cumulative magnitude (trees per hectare for MPB, defoliation units on a 1-3 scale for WSB), and count of years detected after Meigs et al. (2011). We also derived cumulative insect extent, onset, magnitude, and count data for each fire year, focusing primarily on patterns in extent and onset.

### ***Fire data from Monitoring Trends in Burn Severity (MTBS)***

We assessed annual fire extent data from MTBS, a national effort to track fire extent and severity consistently (Eidenshink et al. 2007; <http://mtbs.gov/>). In the western US, the MTBS program objective is to map all fires greater than 404 ha since 1984 using before-after change detection with Landsat TM/ETM+ imagery (30-m grain; Eidenshink et al. 2007). MTBS analysts use the NBR spectral index (Key and Benson 2006) to delineate fire perimeters and compute severity.

We clipped fire perimeters within forested ecoregions (Omernik 1987) and state boundaries ( $n = 685$  fire polygons; Fig. 1). As with the ADS data, we converted the polygons to raster format for spatial analysis at 30-m resolution. For this assessment, we did not examine variability in burn severity, and we included all area within fire perimeters. We recognize that there is substantial variability, including unburned areas, within MTBS perimeters (Kolden et al. 2012), but our goal was to assess all potential overlap with the similarly inclusive and coarse-scale insect polygon data.

### ***Data summary and analysis***

For our first objective, we mapped the three disturbance agents (MPB, WSB, wildfire) and derived time series across the PNW. We included the full MTBS record (1984-2012) and additional years of ADS data (1970-2012) to capture a more extensive range of prefire insect activity for the earlier fire years. For each fire year, we also computed the fire extent with and

without prior MPB or WSB activity of any magnitude, including all potential prefire insect years (minimum 14 years for 1984 fires; maximum 42 years for 2012 fires) and all fire extent (no forest mask as in the analysis below).

To address our second objective, we integrated the ADS and MTBS data in a binary, two by two likelihood matrix of fire and prior insect activity (Fig. 2). For each fire year (1984-2012), we computed fire extent with prior insect extent (A), unburned prior insect extent (B), fire extent without prior insect extent (C), and the total remaining forest extent in each ecoregion with neither fire nor prior insect extent (D). We conducted these geospatial computations on rasterized insect and fire data at a 30-m grain. To mitigate the confounding effect of fire in nonforested areas, we limited this analysis to forested areas using a forest mask based on USGS GAP and NLCD maps (30-m grain; Ohmann et al. 2012; data available online: (<http://gapanalysis.nbi.gov/>, <http://landcover.usgs.gov/>)).

Then, for all pixels in a given year and ecoregion, we calculated empirical fire likelihood indices as the percent burned of the available area that did and did not have prior insects (Fig. 2):

1. Fire likelihood in areas with prior insect activity:  $FL_I: A / (A + B) * 100$ ;
2. Fire likelihood in areas without prior insect activity:  $FL_{NI}: C / (C + D) * 100$ .

We compared these fire likelihoods for all years in which fire occurred to address the essential question: Given a fire event, do areas with recent insect activity have the same fire likelihood as areas without recent insect activity? We note that because we excluded years with no fire extent, actual annual fire likelihood rates are slightly lower than our estimates. In addition, our fire likelihood metrics are similar to a recent conditional probability analysis of bark beetle-fire interactions (Bisrat 2010), but we cover many more years, use much finer-grained data, and focus on a specific bark beetle and defoliator.

To assess potential changes in fire likelihood associated with fuel succession at different times following insect outbreak (Hicke et al. 2012), we compared the distribution of fire likelihood in areas with and without MPB or WSB activity at multiple time lags. For our overall summary and most inclusive calculation, we retained any prefire insect activity for a given fire year (range: 14-42 years). We also assessed fire likelihood in areas with and without onset of MPB or WSB outbreak in five year intervals (1-5, 6-10, 11-15, 16-20). For this time lag analysis, we treated the first year of ADS detection as the onset of insect outbreak, recognizing

that actual insect activity likely initiated one year before it was observed by aerial surveyors (Kulakowski and Veblen 2007, Meddens et al. 2012). We also considered alternative metrics for timing of insect outbreak, such as peak year or last year of outbreak, but the onset year was a more consistent metric across such a wide range of fire and insect events. We address some of the uncertainty in the timing of insect onset and peak damage by focusing on five year intervals rather than individual years. To retain as much of the fire population as possible in the full analysis of time since outbreak, we included intervals up to 20 years only (i.e., our ADS data begin in 1970, so all fires from 1990 onward have the potential for all insect time lags up to 20 years).

For each ecoregion, insect agent, and time lag, we tested for differences between the percent burned of the available area in each condition (with and without prior insect activity) at the scale of individual fire years. After determining that these differences were not temporally autocorrelated but did exhibit outliers and positive skewness, we used a nonparametric test (Wilcoxon signed-rank, abbreviated SR) to evaluate the differences between fire likelihood with and without prior insect activity. Because we tested these differences separately for each ecoregion, insect agent, and time interval (60 simultaneous comparisons), we used the false discovery rate (FDR; Benjamin and Hochberg 1995, Pike 2011) to control for potential Type I errors. Concurrently, we interpreted FDR-adjusted  $P < 0.05$  as strong evidence of differences and FDR-adjusted  $P < 0.1$  as moderate evidence, controlling for potential Type II errors after Donato et al. (2013).

We also assessed the effect of fire size on insect-fire likelihood. To this end, we related the difference in fire likelihood between areas with and without prior insect activity to the total fire extent in a given year. The years in our sample population vary considerably in interannual fire extent (Fig. 3), and this final analysis enabled us to test whether insect-affected areas were more or less likely to burn in years with few/small fires versus many/large fires (i.e., regionally synchronous, climate-driven fire years; Heyerdahl et al. 2008). We conducted all geospatial analyses with ARCGIS, IDL, or GDAL, and we derived statistics and graphics with R (packages plyr, ggplot2 package; Wickham 2009, Wickham 2011).

## RESULTS

### *Extent and overlap of MPB, WSB, and wildfire across the PNW*

The three disturbances we assessed we assessed – MPB, WSB, and wildfire – have not been distributed evenly across space and time, according to the ADS and MTBS datasets (Figs. 1, 3). MPB and WSB have been active primarily in the drier, interior conifer forests east of the crest of the Cascade Range, whereas wildfire has occurred more widely in interior PNW forests as well as portions of the West Cascades and Klamath Mountains. In all ecoregions except for the Klamath Mountains, cumulative WSB extent has exceeded cumulative MPB extent, which in turn has exceeded wildfire extent (Fig. 3). In most years, the extent of any of the three disturbances has been less than 2% of the available ecoregion area (Fig. 3).

Each ecoregion has illustrated a distinct disturbance history since 1970. The eastern half of the North Cascades experienced all three disturbances in abundance (Fig. 1), particularly since 2000 (Fig. 3A), with moderate overlap of fire on prior insect activity (Figs. 4, 5). The Northern Rockies had widespread WSB activity (including a particularly large spike in the 1990s up to 15% of the ecoregion [400,000 ha]; Fig. 3B), moderate MPB, and minimal fire. The West Cascades experienced a substantial WSB outbreak in the 1980s and 1990s and all three disturbances later in the time series (Fig. 3C). The East Cascades had widespread outbreaks of both insects and relatively low fire extent (Fig. 3D), with relatively low overlap of fire with prior insect activity (Figs. 4, 5). The Blue Mountains experienced the most widespread disturbance – dominated by WSB outbreaks in the 1980s encompassing more than 25% of the ecoregion (~2,000,000 ha) – and moderate overlap of fires with prior insect activity (Figs. 4, 5). The Klamath Mountains had major fire episodes – punctuated by the ~200,000 ha Biscuit Fire in 2002 – but minimal insect activity. We thus exclude the Klamath Mountains from the insect-fire results and discussion below. Similarly, because the Coast Range was virtually devoid of fire and both insects (logging has been the dominant recent disturbance; Spies et al. 2007), we exclude that ecoregion from further results and discussion.

At the ecoregion scale, insect outbreaks have occurred continuously over the course of several years, whereas fire occurrence has been relatively more stochastic (Fig. 3). In addition, the majority of fire extent has not had prior insect activity in most years. Despite later fire years

having many more years of potential prior insect activity (up to 42 years), the proportion of fire extent with prior activity of either insect has not increased.

Three specific large fire events stand out in the ecoregional insect-fire overlap assessment, with greater than 50% of fire extent in those years overlapping prior insect activity. In the North Cascades, the 2006 Tripod Fire (~70,000 ha) occurred in an area with a very recent/ongoing, heavy MPB outbreak (Fig. 4A; Prichard and Kennedy 2014). In the West Cascades, the 2003 B&B Fire (~37,000 ha) occurred in an area with widespread, heavy WSB activity 10-15 years earlier (Fig. 5C; Crickmore 2011). Finally, in the East Cascades, the 2012 Table Mt. Fire (~27,000 ha) occurred in an area with widespread but low intensity prior WSB activity (Fig. 5D). Although important as ecoregional fire events, these three fire complexes appear to be exceptions to the general patterns exhibited across space and time. In addition, to determine whether these or any other fire events had a higher likelihood of burning in locations with recent insect activity, it is necessary to account for the cumulative insect extent that was available to burn in each year, as we did with our fire likelihood indices.

### ***Fire likelihood with and without recent insect activity***

In recent decades across the PNW, wildfire likelihood was not consistently associated with prefire insect outbreaks initiating as far back as 1970 (Fig. 6). Fire likelihood was generally low (average median percent burned of available forest: 0.12%; Fig. 6, Table 1), and the relationship of fire likelihood and prefire insect outbreaks varied by agent and ecoregion. Specifically, when accounting for all possible years of prefire insect activity, fire likelihood was higher following MPB activity in the North Cascades, West Cascades, and across the forested ecoregions as a whole (FDR-adjusted  $P < 0.05$ ; SR test; Fig. 6, Table 2). In contrast, fire likelihood was lower following recent WSB activity in the Northern Rockies and Blue Mountains (FDR-adjusted  $P < 0.1$ ; SR test; Fig. 6, Table 3). Across the other combinations of insect agent and ecoregion, there were no apparent differences in fire likelihood between areas with and without prior insect activity, although fire likelihood following insect activity tended to be more variable (Fig. 6, Table 1).

MPB and WSB associations with fire likelihood were more evident in different time intervals since insect onset, but these relationships were inconsistent among insects and

ecoregions (Figs. 7, 8, Tables 2, 3). For example, fire likelihood was significantly higher with the onset of MPB activity in the prior 5 years in the North Cascades and across all forested ecoregions ( $P < 0.05$ ; SR test; Fig. 7, Table 2), whereas fire likelihood was lower at particular time lags following MPB in the Northern Rockies, East Cascades, and Blue Mountains ( $P < 0.1$ ; SR test; Fig. 7, Table 2). In contrast, fire likelihood consistently was lower following WSB outbreaks at various time lags in all ecoregions ( $P < 0.1$ ; SR test; Fig. 8, Table 3). Across both insects and all time lags, even when statistically significant, the average differences were small in magnitude (pseudo-median difference  $\sim 0.1\%$ ; Tables 2, 3).

The two insects illustrated several other distinct relationships among time lags and ecoregions. MPB difference distributions were tended to be positively skewed, driven by a larger number of extreme years with more fire occurring in recent MPB outbreaks (Fig. 7), whereas WSB difference distributions tended to be negatively skewed (Fig. 8). For MPB, the North Cascades had the largest number of extreme positive differences (outlier data not shown). For WSB, despite the generally lower fire likelihoods in areas with recent outbreaks, there were several extreme positive values, particularly in the West Cascades, East Cascades, and Blue Mountains (outlier data not shown).

There were no clear associations between fire likelihood and fire size (annual fire extent) for either insect at the ecoregional or regional scales (Fig. 9). Most years had relatively low total fire extent and small differences between forests with and without prior insect activity, although variability in the insect-fire effect increased with increasing fire extent. One particularly extreme fire year in the North Cascades (driven by the 2006 Tripod Fire) illustrated very high fire extent and increased likelihood of fire in forests with recent MPB activity (Fig. 9A). Although there were more examples of positive than negative differences in the North Cascades and West Cascades (and vice versa in the Blue Mountains), most difference values were close to zero.

## DISCUSSION

Whereas previous research has focused on individual insect outbreaks or wildfires at finer scales, this study presents a novel, landscape- and regional-scale synthesis across numerous insect and fire events. In recent decades across the PNW, insect outbreaks do not appear to be a primary driver of fire extent, nor is there a consistent increase or decrease in fire likelihood in forests with previous insect outbreaks. Wildfires, bark beetles, and defoliators all influence PNW forests, but they may not overlap enough to facilitate linked disturbance interactions as defined by Simard et al. (2011). Here, we evaluate our four hypotheses in the context of existing literature. We then discuss some of the uncertainties of our study, opportunities for future research, and implications for forest management.

### *Evidence of linked disturbance interactions between insects and wildfires*

As we hypothesized, all three disturbance agents occurred primarily in the drier, interior conifer forests east of the Cascade Crest, although wildfire extended in an arc through the southern West Cascades and Klamath Mountains (Hypothesis 1a partially supported; Figs. 1, 3). Notably, the largest fire event in recent Oregon history – the Biscuit Fire, which burned ~10% of the Klamath Mountains ecoregion – occurred in an ecoregion with minimal influence from MPB and WSB.

In general, the distributions of WSB and MPB match their host tree distributions, so it is not surprising that WSB has been the most extensive of the three disturbances (Figs. 1, 3). WSB affects a variety of host tree species, including true firs, spruce, and Douglas-fir, whereas MPB epidemics are limited to areas with substantial lodgepole pine (Goheen and Willhite 2006). That said, WSB has the potential to influence Douglas-fir forests throughout the western PNW but so far has been limited to drier, interior forests.

As expected for biotic versus abiotic disturbances, insect outbreaks have been relatively continuous and autocorrelated at the ecoregion scale compared to the more sporadic, stochastic occurrence of wildfire, although there is substantial interannual variability in the extent of all three disturbances (Hypothesis 1b partially supported; Fig. 3). In most years, fire extent illustrates distinct temporal patterns among ecoregions, but 2002 stands out as a regional fire episode, consistent with climate-driven synchrony (Heyerdahl et al. 2008).

In general, wildfires are not consistently more likely in areas with previous insect activity, but there are important examples of higher fire likelihood following MPB outbreaks and lower fire likelihood following WSB outbreaks (Hypothesis 2a partially supported; Fig. 6). All three disturbances are relatively constrained across the forested ecoregions of the PNW (rarely exceeding 2% of ecoregion extent in a given year; Fig. 3), indicating that there are large areas with neither insect nor wildfire throughout the study time period. Thus, although insect extent generally exceeds fire extent, and recent insect outbreaks peaked in years preceding most large fire years in the PNW (Fig. 3), insect associations with fire likelihood appear to vary by insect type and ecoregion (Figs. 4, 5, 6). In addition, the relatively low fire likelihood in forests with or without prior insect activity indicates that wildfire is a relatively rare phenomenon, consistent with a general fire deficit relative to historic conditions (Marlon et al. 2012). Finally, the higher variability and negatively skewed distributions in fire likelihood in areas with recent insect activity (Fig. 6, Table 1) are the result of multiple years without fire in insect-altered areas (i.e., because insects affect a small area in any given year relative to the available forest area; Fig. 3).

Despite the high variability in fire likelihood among insect agents and time since insect onset, many five-year intervals demonstrate important differences between areas with and without recent insect outbreaks (Figs. 7, 8, Tables 2, 3; Hypothesis 2b partially supported). For MPB, the five year intervals reveal substantial variation, including inconsistent time lags for positive linked effects as well as several examples of negative linked effects (Fig. 7), suggesting that time since outbreak remains a key consideration (Hicke et al. 2012). In contrast, WSB outbreaks exhibit more consistent effects, with examples of lower fire likelihood at multiple time lags in all ecoregions (Fig. 8). Thus, although both bark beetles and defoliators may alter fuel profiles and associated fire potentials at different time lags (e.g., Hummel and Agee 2003, Page and Jenkins 2007, Simard et al. 2011, Hicke et al. 2012, Donato et al. 2013), these windows of opportunity for increased or decreased fire potential are too narrow – and/or fire is too rare – for a consistent signal to emerge across PNW conifer forests.

There are specific fire years, however, with unusually strong positive insect-fire signals, (Figs. 7, 8, 9). These extreme positive years may be important examples of insect-fire co-occurrence at the landscape scale, resulting in compound disturbance effects even without broader evidence of linked disturbance effects (as in Harvey et al. [2014]). In addition, the lack

of association between insect-fire likelihood and fire extent (Fig. 9) suggests that the relationship between fire likelihood and prior insect activity is not more pronounced in mild, moderate, or extreme fire years. This result also suggests that other factors (such as climate; Heyerdahl et al. 2008) control the disproportionately large fire years accounting for the majority of regional fire extent (e.g., Strauss et al. 1989).

Our findings that insect-wildfires interactions appear to vary across insect agents, space, and time may help to reconcile divergent results of previous studies. Looking across all possible years of prefire insect activity, we found examples of no difference in fire likelihood for each insect and in multiple ecoregions (Fig. 6), consistent with previous studies showing a lack of linked disturbance interactions between insects and wildfires (e.g., Kulakowski and Veblen 2007, Crickmore 2011, Kulakowski and Jarvis 2011, Harvey et al. 2013). Our findings of positive insect-fire interactions at multiple scales (Figs. 6, 7, 8) and particular years with large insect-fire overlap (Figs. 4, 5), driven by individual fires like the 2006 Tripod Fire, are also consistent with studies showing positive linked interactions between MPB and the spatial patterns of fire at the event scale (Lynch et al. 2006, Prichard and Kennedy 2014). Our result that MPB within the previous five years is associated with both increased and decreased fire likelihood (Fig. 7) partially supports findings from a regional assessment of bark beetle-fire interactions that employed conditional probabilities similar to our fire likelihoods but only up to five years prefire (Bisrat 2010). In addition, our results showing consistently lower fire likelihood in areas affected by WSB (Figs. 6, 8) are consistent with a recent study of negative feedbacks between WSB and fire in British Columbia (Lynch and Moorcroft 2008). We also found examples of higher fire likelihood following WSB activity in extreme years in multiple ecoregions (Fig. 8), suggesting that the feedbacks between defoliators and wildfires merit further research.

Our empirical results are not directly comparable to studies that use simulation models to estimate fire behavior and/or effects in particular post-insect forest stands or landscapes (e.g., Hummel and Agee 2003, Page and Jenkins 2007, Simard et al. 2011). These studies assume that a given insect-altered forest burns, whereas our findings suggest that fire affects a relatively small proportion of insect- and non-insect-affected forest in any given year. Additional retrospective studies have used dendroecological methods, remote sensing, and landscape pattern

analysis of individual fire events to assess the empirical evidence of insect-fire interactions over longer time periods (e.g., Bebi et al. 2003, Bigler et al. 2005, Kulakowski and Jarvis 2011). Although these studies have focused on non-MPB bark beetles or ecosystems outside of the PNW, our results are generally consistent with their findings of mixed or relatively weak positive feedbacks among insects and wildfires.

### ***Uncertainties, limitations, and future research***

This study addresses insect-fire interactions at relatively broad landscape and regional scales, empirically quantifying wildfire likelihood across ecoregions where numerous fine-scale processes and feedbacks are operating. Foremost among these are the potential effects of management actions at the forest stand scale. Salvage logging and/or thinning in insect outbreak areas and fire suppression are management activities that have been prevalent on public and private forests in recent decades (e.g., Sanders et al. 1985, Waring and Pitman 1985, Waring et al. 1992, Negron et al. 2008, Azuma 2010, Marlon et al. 2012), reducing the chance of fire encountering insect-altered forests and fuels. It is possible that insect and wildfire events would overlap more frequently in less intensively managed forests (i.e., wilderness areas) or in the future if climate change increases either or both types of disturbance (Heyerdahl et al. 2008, Bentz et al. 2010, Littell et al. 2010). Although fire extent has increased in recent decades, many forests in the PNW remain in a fire deficit relative to historic fire frequency and extent (e.g., Littell et al. 2009, Marlon et al. 2012). The relatively low fire likelihoods observed here (Figs. 3, 6, Table 1) would have to increase dramatically to affect the amount of insect-fire overlap. Moreover, interannual climate is the most important top-down control of regional fire dynamics (Heyerdahl et al. 2008), and it appears that interannual fire extent is not associated with the differences between forests with and without prior insect activity (Fig. 9).

In addition to these uncertainties associated with the ecological phenomena of interest, there are also limitations in our datasets. The ADS data likely overestimate insect extent while simultaneously missing some insect impacts due to detectability challenges and spatial inaccuracies in the polygon data (Meddens et al. 2012, Meigs et al. 2014). In addition, the ADS polygons encompass high heterogeneity in forest conditions and insect effects, and future studies could evaluate whether more constrained insect maps would result in different fire likelihood

estimates, either focusing on higher damage ADS polygons or leveraging a more consistent change detection dataset like Landsat imagery (Meigs et al. 2014). Similarly, the MTBS fire perimeters contain a wide range of fire effects – including unburned or unburnable patches (Kolden et al. 2012) – that could mask potential insect-fire signals. Future research could conduct similar analyses with high- or moderate-severity fire only, but this approach would require a severity classification scheme that is consistent across the full spectrum of forest types and prefire conditions in the region (e.g., the 685 fire polygons in our study). Future research could also use ADS and MTBS data to investigate insect-fire interactions in other regions (e.g., US Rocky Mountains), although the ADS data may not be as spatiotemporally consistent (Meddens et al. 2012).

Additionally, although the Omernik (1987) ecoregions are very useful to assess broad landscape differences across the PNW, they also encompass substantial variation in forest composition, structure, and disturbance history. All three of these factors are important for MPB, WSB, and wildfire behavior and effects (e.g., Sanders et al. 1985, Hessburg et al. 2000, Raffa et al. 2008). It is possible that similar analyses with finer-grained vegetation maps (such as potential vegetation type; [Henderson et al. 2011] or gradient nearest neighbor-based attributes [Ohmann et al. 2012]), would yield different or more context-specific results.

Finally, although our census data cover a relatively large number of fires and insect outbreaks, future studies could account for longer time periods, additional disturbance agents, and other response variables. For example, bark beetles have been shown to double coarse fuels 25-30 years post-outbreak (Donato et al. 2013), but we limited our time lag analysis to 20 years to retain balance in our fire sample. In addition, PNW forests are influenced by numerous, interacting biotic and abiotic disturbance factors, including other insects (e.g., fir engraver beetle [*Scolytus ventralis* LeConte {Coleoptera: Curculionidae}], Douglas-fir tussock moth [*Orgyia pseudotsugata* {McDunnough} {Lepidoptera: Lymantriidae}]; Goheen and Willhite 2006), diseases, drought, storms, logging, and grazing. These factors contribute to the variation in our fire likelihood results, but the net effect remains a lack of insect-fire overlap in the forests we assessed.

Even where insects and wildfires do overlap in space and time, it is important to distinguish the overlap that would be expected by chance or given other covarying drivers.

Indeed, the three disturbances we assessed are catalyzed by the same abiotic and biotic factors (e.g., drought, tree composition, structure, and vigor, and landscape contiguity; Meigs et al. 2011, Simard et al. 2011, Prichard and Kennedy 2014), and future research could focus on these drivers explicitly. Future research could also investigate linked disturbance effects of insects on fire severity rather than likelihood or extent, focusing only on fires with substantial prior insect activity, a small subset of fire events representing a high research and management priority.

### ***Management implications***

Our central findings that wildfire likelihood is not consistently higher or lower in areas affected by recent insect outbreaks and that insect-fire interactions appear to be context- and scale-dependent have several important implications for forest management. First, although relatively rare in any given year or ecoregion (Fig. 3), each of the three disturbances – MPB, WSB, and wildfire – will continue to influence PNW forests, and it may be efficient to focus on the forest health impacts of each agent separately more than their interactions. Because these disturbances are not evenly distributed across the region, it also makes sense to focus management activities on specific landscapes with higher disturbance impacts and/or more valuable ecosystem services. For example, portions of the North Cascades and West Cascades ecoregions have experienced recent pulses of all three disturbances (Figs. 1, 3) and substantial insect-fire overlap events (Figs. 6, 7, 8), representing an opportunity to study and adaptively manage for resilience, particularly where these disturbances occur outside of wilderness areas where management options are limited.

Second, when and where fires do burn in recent or ongoing insect outbreaks, fire behavior and/or effects may change (e.g., higher rate of spread or severity; Prichard and Kennedy 2014, Perrakis et al. 2014), fire management will likely be more challenging (Hicke et al. 2012, Page et al. 2013), and compound disturbance effects are possible (e.g., Harvey et al. 2013). Because fire likelihood and the extent of insect-fire overlap are generally low (Figs. 4, 5, 6, Table 1), however, these compound disturbance effects are relatively rare. Moreover, our finding that WSB outbreaks are associated with decreased fire likelihood suggests that these defoliators may function as forest thinning agents. The mixed relationships between MPB and fire likelihood versus the more consistently negative WSB relationships suggest that forest

managers should continue to use insect-specific management strategies in the forests these two insects inhabit. In addition, both MPB and WSB are native to PNW forests and likely contribute to the pyrodiversity (Martin and Sapsis 1994) inherent to mixed-severity fire regimes. Because these insects may not represent a regional forest health crisis, there are higher priorities for ecological restoration programs, such as fuel and fire dynamics at the wildland-urban interface (e.g., Kennedy and Johnson 2014).

Third, forests will continue to burn whether or not there was prior insect activity, and known fire drivers like fuel structure and vegetation stress will likely play a more important role in a warmer, potentially drier future (McKenzie et al. 2004, Westerling et al. 2006, Heyerdahl et al. 2008, Littell et al. 2010). Long-term monitoring will remain essential for research and adaptive management, and it is imperative that programs like the ADS and MTBS continue to track these disturbances with consistent protocols and data accessibility. Given societal concerns about forest health in a rapidly changing climate, ongoing research and monitoring will enable forest managers to quantify and anticipate the independent and interactive effects of insects, wildfires, and other disturbances, both native and novel.

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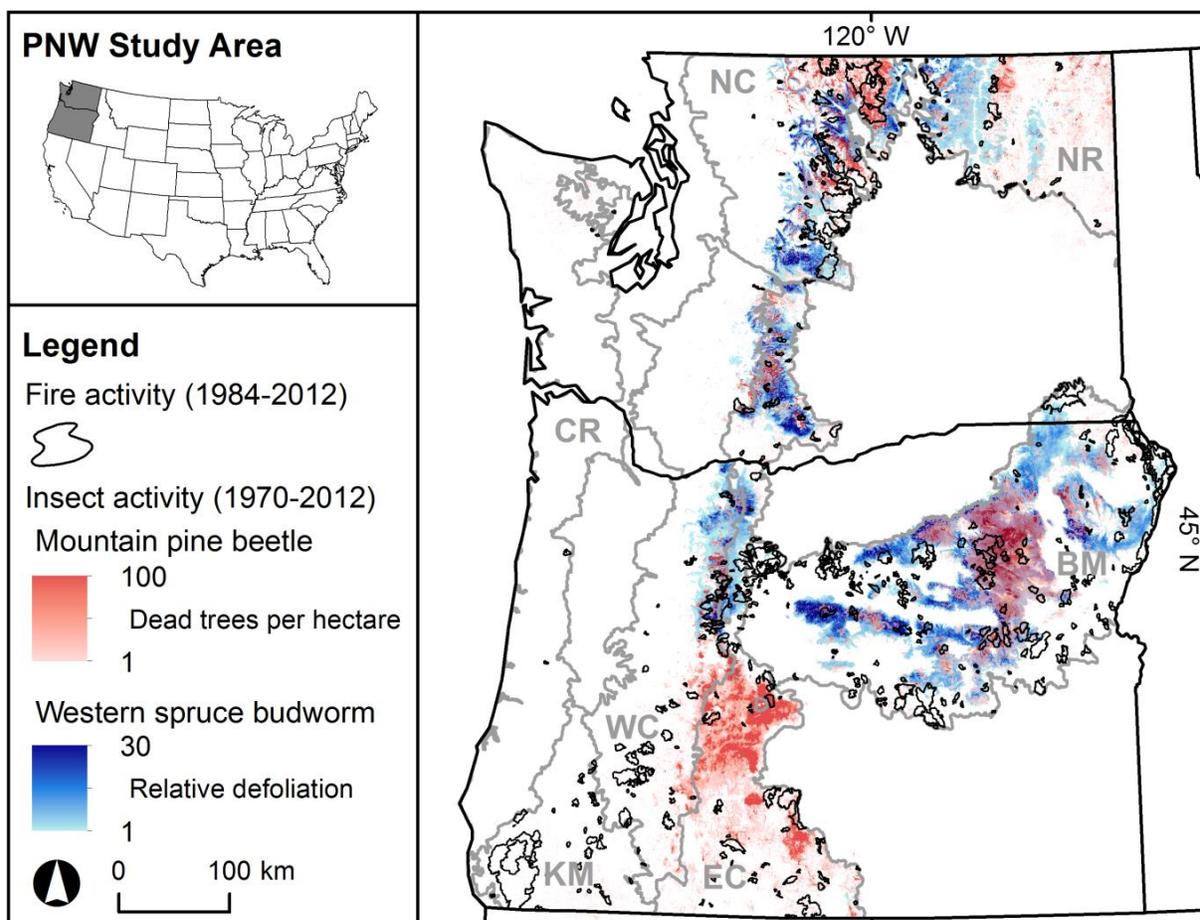
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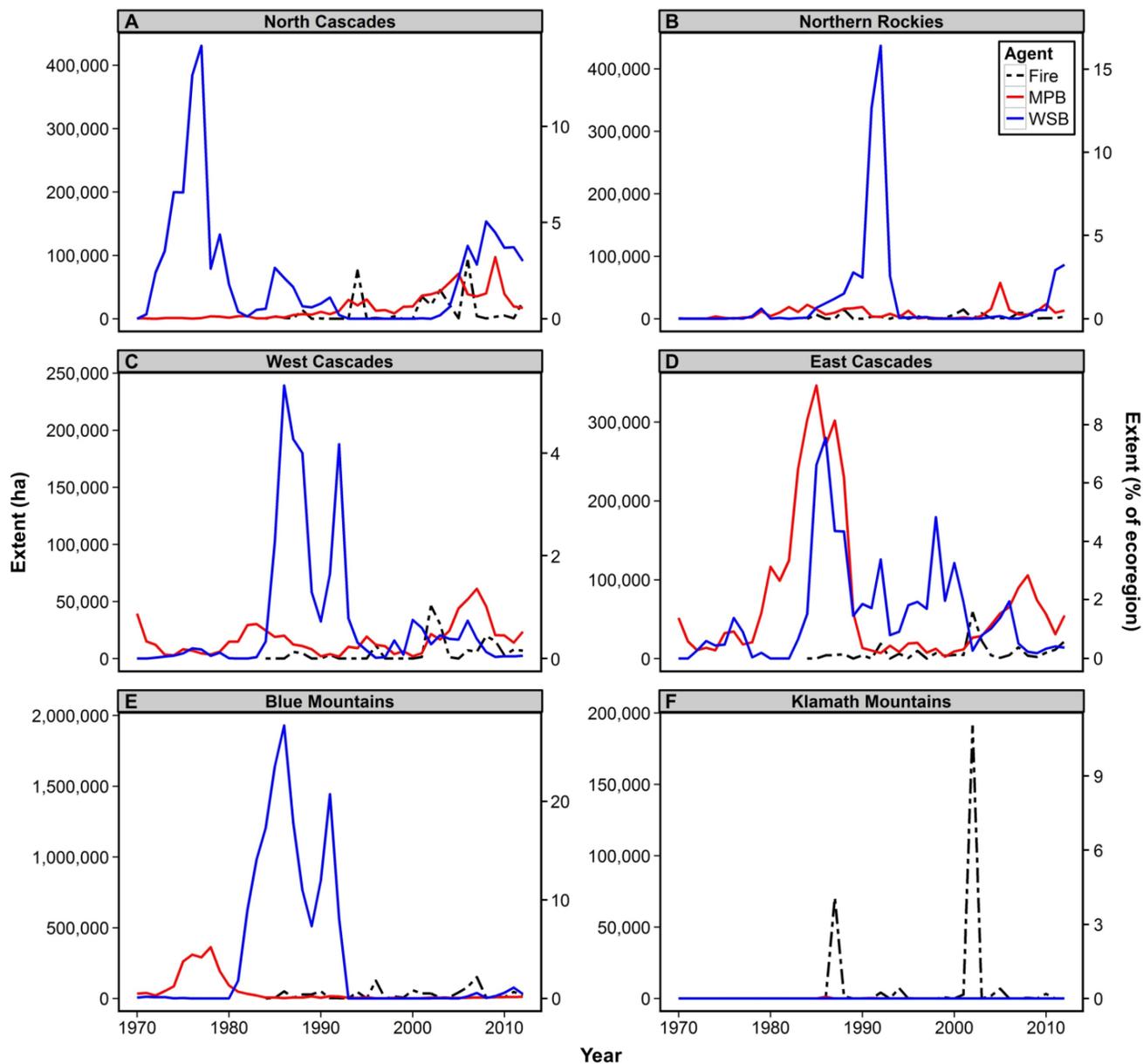
## FIGURES AND TABLES



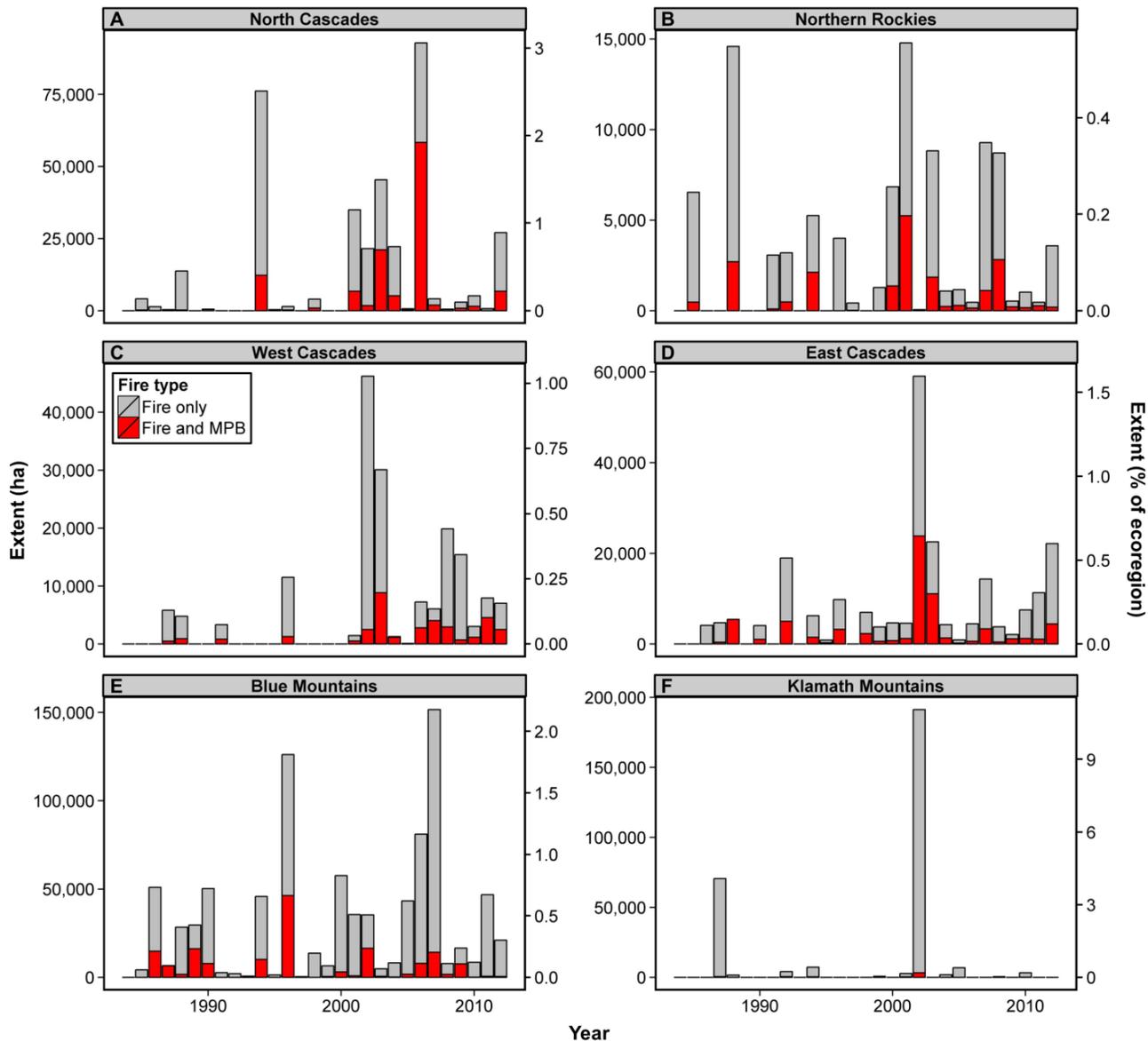
**Fig. 4.1.** Study area map and distribution of insects and wildfires across forested ecoregions of the Pacific Northwest (PNW; Oregon and Washington; inset). Fire perimeters (MTBS; 1984-2012; <http://mtbs.gov>) are clipped within forested ecoregion and state boundaries. All insect activity from aerial detection surveys (ADS; 1970-2012; see Methods) is included. Note that mountain pine beetle (MPB) and western spruce budworm (WSB) have been active primarily in the dry, interior conifer forests east of the crest of the Cascade Range, whereas wildfire has been distributed widely in the interior forests as well as portions of the WC and KM ecoregions. All three disturbances have been absent in the CR. Note also that MPB overlaps WSB activity in this display (MPB shown with 30% transparency), revealing purple locations with both insects. Total area of forest cover in forested ecoregions is ~20 M ha, and area affected by either insect is ~8 M ha according to ADS data (1970-2012). Ecoregion (Omernik 1987) abbreviations: NC: North Cascades; NR: Northern Rockies; CR: Coast Range; BM: Blue Mountains; WC: West Cascades; EC: East Cascades; KM: Klamath Mountains. Note that the other ecoregions are not labeled and that we include only the portions of ecoregions within the two states.

		FIRE	
		Yes	No
INSECT	Yes	A	B
	No	C	D

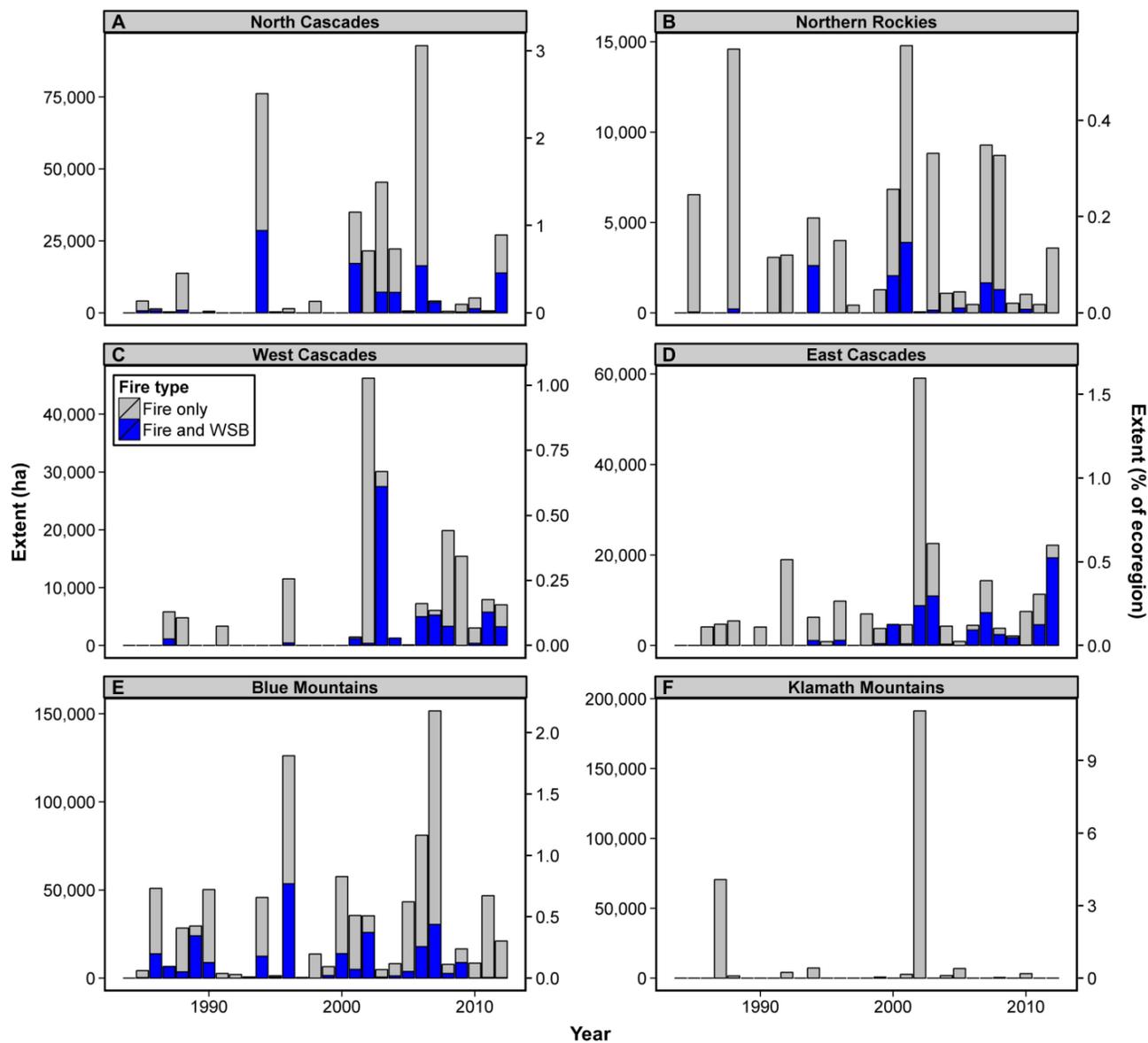
**Fig. 4.2.** Two-way insect-fire likelihood matrix, where fire likelihood is expressed as the percent burned of the area available in two conditions in a given year. Fire likelihood in areas with prior insect activity:  $FLI: A / (A + B) * 100$ . Fire likelihood in areas without prior insect activity:  $FLNI: C / (C + D) * 100$ .



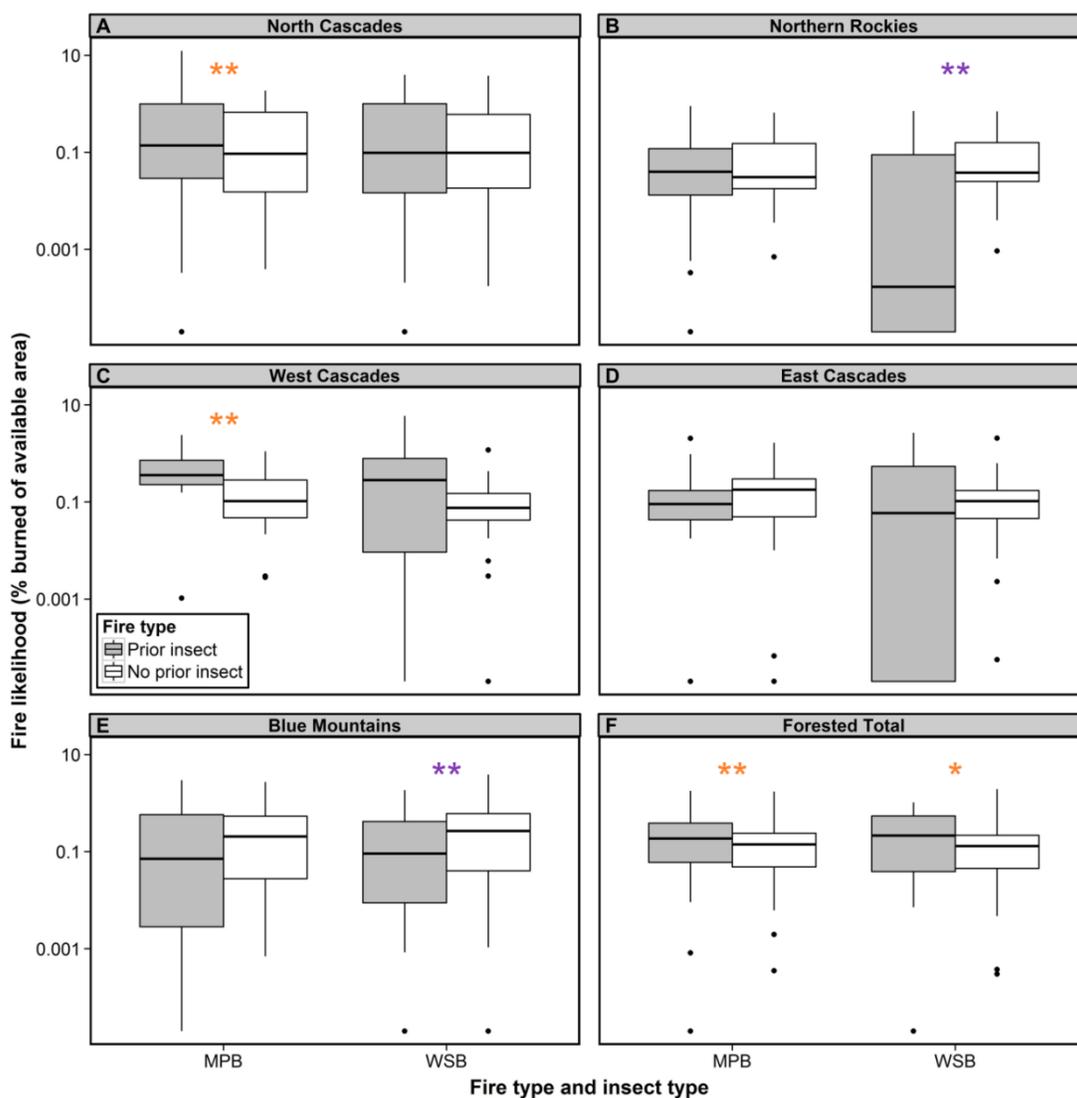
**Fig. 4.3.** Wildfire, mountain pine beetle (MPB), and western spruce budworm (WSB) activity across forested ecoregions of the PNW. Note that insect extent, particularly WSB, is much greater than interannual and cumulative fire extent (except for Klamath Mountains). Note also the relative continuous insect occurrence and more sporadic fire occurrence. Coast Range ecoregion excluded due to negligible insect and fire activity. Fire extent from perimeters (MTBS; 1984-2012; <http://mtbs.gov>) within forested ecoregion and state boundaries. Insect activity from aerial detection surveys (ADS; 1970-2012; see Methods).



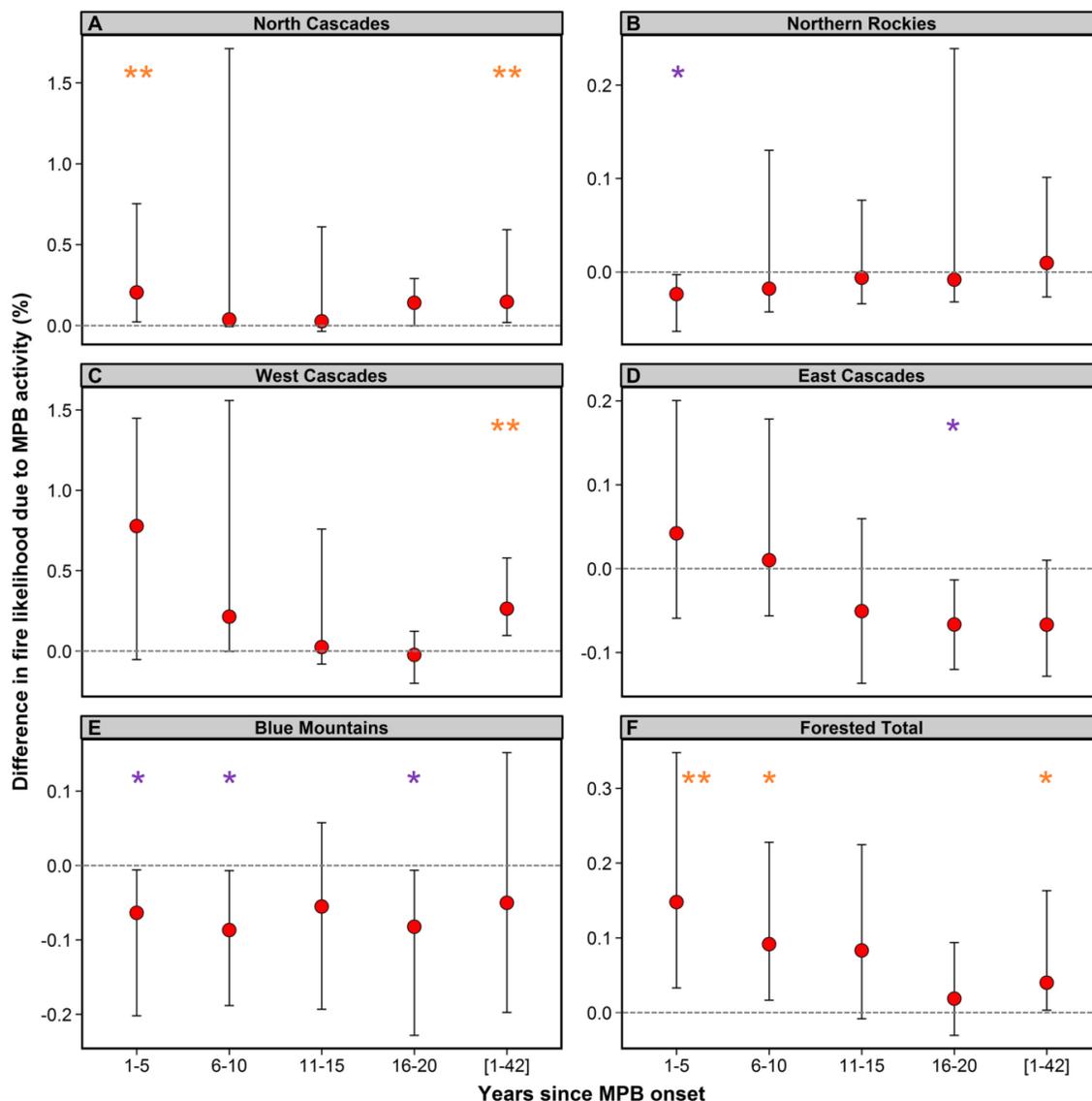
**Fig. 4.4.** Fire extent with and without recent mountain pine beetle activity (MPB) across forested ecoregions of the PNW. The majority of fire extent does not have recent insect activity in most years, although later fire years have many more years of potential prior insect. Note that the total fire extent is equal to the top of a bar in a given year (i.e., bars are not stacked). Coast Range ecoregion excluded due to negligible insect and fire activity. Fire extent from perimeters (MTBS; 1984-2012; <http://mtbs.gov>) within forested ecoregion and state boundaries. Insect activity from aerial detection surveys (ADS; 1970-2012; see Methods).



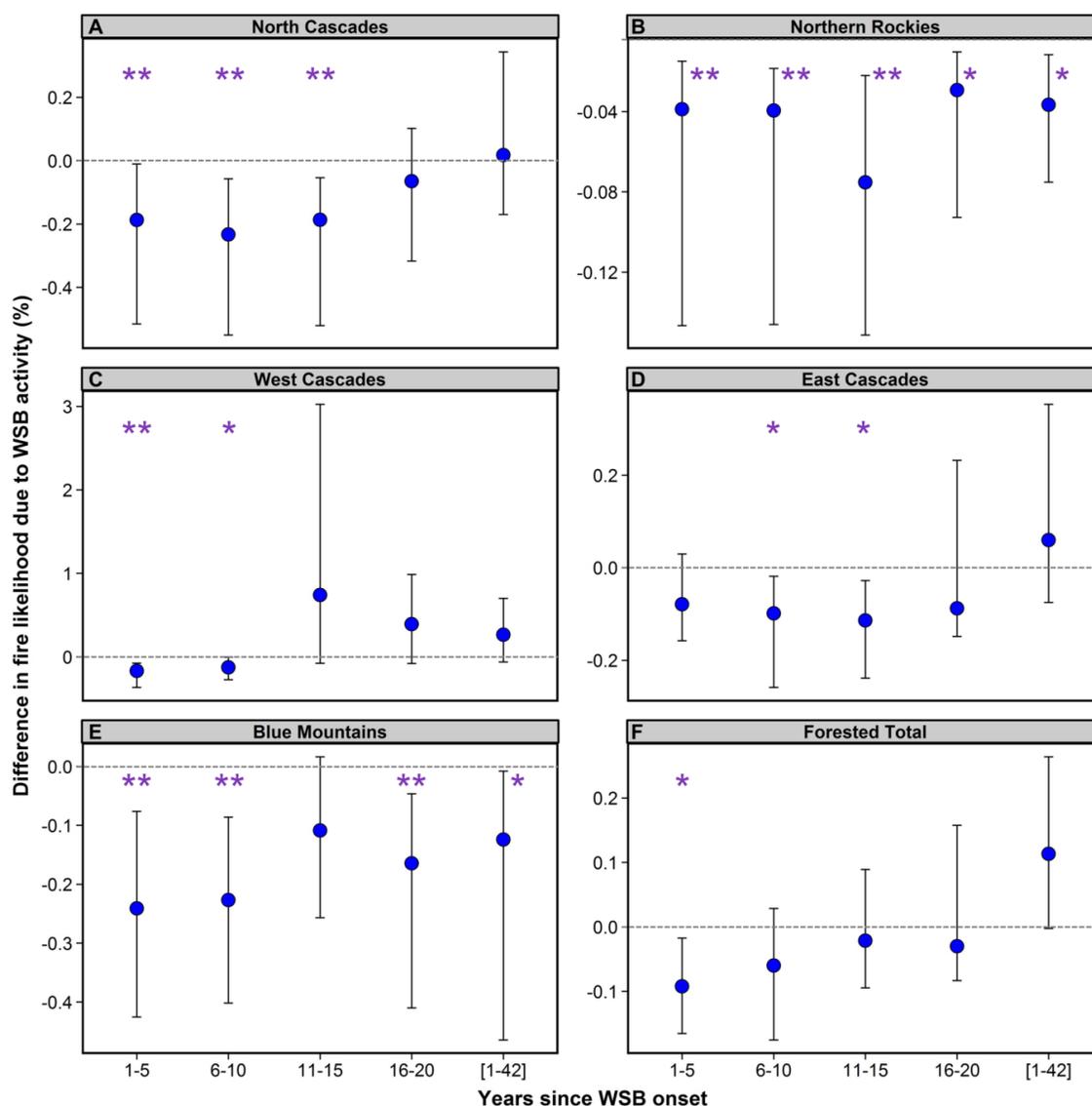
**Fig. 4.5.** Fire extent with and without recent western spruce budworm activity (WSB) across forested ecoregions of the PNW. The majority of fire extent does not have recent insect activity in most years, although later fire years have many more years of potential prior insect. Note that the total fire extent is equal to the top of a bar in a given year (i.e., bars are not stacked). Coast Range ecoregion excluded due to negligible insect and fire activity. Fire extent from perimeters (MTBS; 1984-2012; <http://mtbs.gov>) within forested ecoregion and state boundaries. Insect activity from aerial detection surveys (ADS; 1970-2012; see Methods).



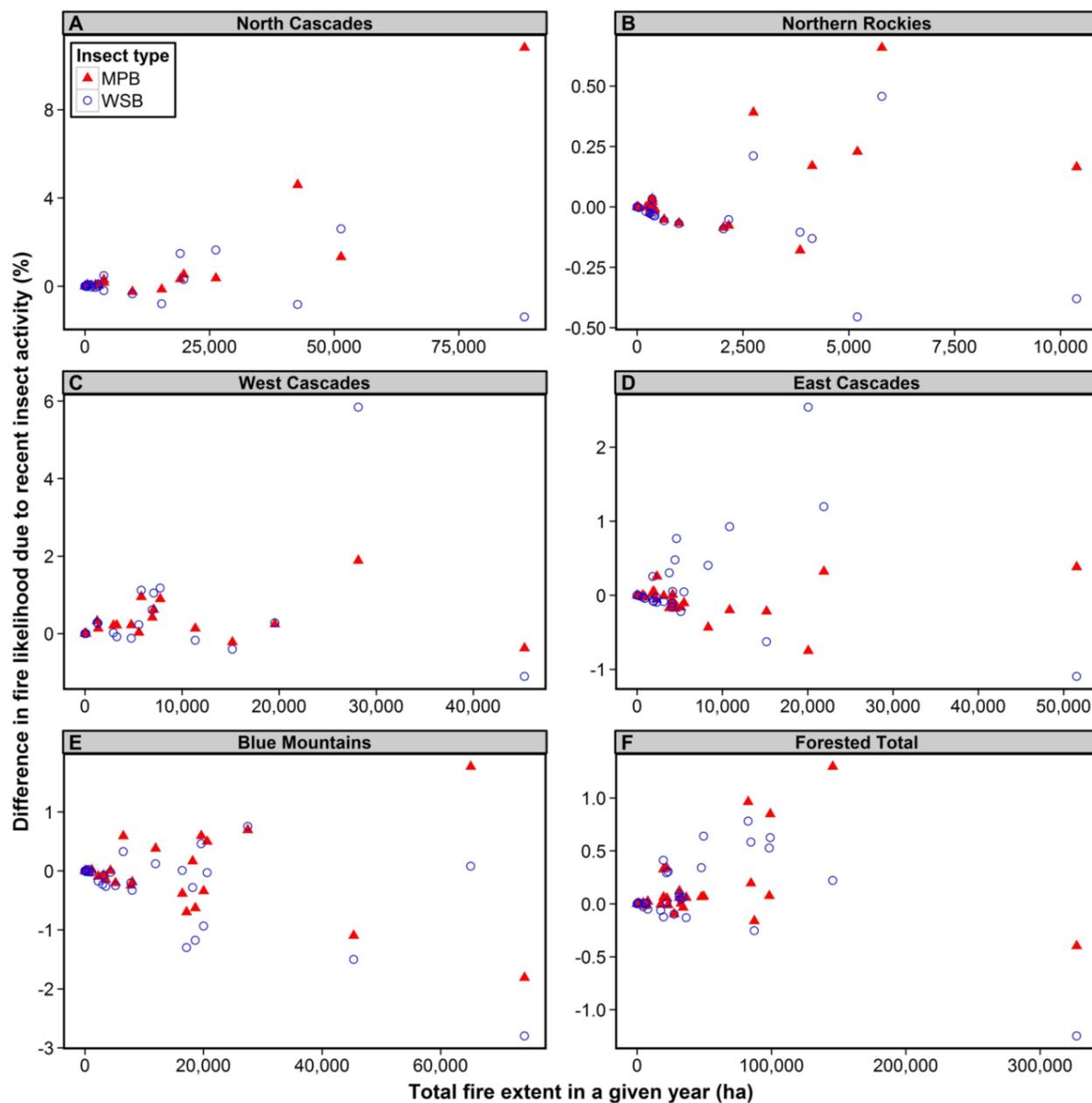
**Fig. 4.6.** Interannual fire likelihood with and without recent insect activity across forested ecoregions of the PNW. For this summary, recent insect activity is defined as any mountain pine beetle (MPB) or western spruce budworm (WSB) activity in any year prior to a given fire year (minimum 14, maximum 42 years). Data are individual fire years, log-transformed for display (constant value half the size of the smallest value in the dataset added to retain zeroes). See Table 1 for values and number of fire years. Significant paired differences (Wilcoxon signed-rank test) indicated by asterisks (\*\* = FDR-adjusted  $P < 0.05$ , \* = FDR-adjusted  $< 0.1$ ). Orange asterisks denote positive differences; purple asterisks denote negative differences. See Tables 2 and 3 for additional details. Note that this figure excludes years with zero fire extent in both classes (at ecoregion scale), so the average annual fire likelihood values are lower than those displayed here. Note also that the Coast Range and Klamath Mountains ecoregions are excluded due to negligible insect activity, although they are included in the Forested Total category. Fire extent from perimeters (MTBS; 1984-2012; <http://mtbs.gov>) within forested ecoregion and state boundaries. Insect activity from aerial detection surveys (ADS; 1970-2012).



**Fig. 4.7.** Interannual differences in fire likelihood associated with mountain pine beetle (MPB) activity at recent time lags across forested ecoregions of the PNW. Intervals of insect onset are arranged with the most recent five year interval on the left and the interval of all available years on the right. Points are average differences (Wilcoxon signed-rank test pseudomedian). Error bars are nonparametric 95% confidence intervals. Significant paired differences indicated by asterisks (\*\* = FDR-adjusted  $P < 0.05$ , \* = FDR-adjusted  $< 0.1$ ). Orange asterisks denote positive differences; purple asterisks denote negative differences. See Table 2 for additional details. Note that this figure excludes years without fire in both classes (at ecoregion scale) and that y axes vary with ecoregion. Note also that the Coast Range and Klamath Mountains ecoregions are excluded due to negligible insect activity, although they are included in the Forested Total category. Fire extent from perimeters (MTBS; 1984-2012; <http://mtbs.gov>) within forested ecoregion and state boundaries. Insect activity from aerial detection surveys (ADS; 1970-2012).



**Fig. 4.8.** Interannual differences in fire likelihood associated with western spruce budworm (WSB) activity at recent time lags across forested ecoregions of the PNW. Intervals of insect onset are arranged with the most recent five year interval on the left and the interval of all available years on the right. Points are average differences (Wilcoxon signed-rank test pseudomedian). Error bars are nonparametric 95% confidence intervals. Significant paired differences indicated by asterisks (\*\* = FDR-adjusted  $P < 0.05$ , \* = FDR-adjusted  $< 0.1$ ). Orange asterisks denote positive differences; purple asterisks denote negative differences. See Table 3 for additional details. Note that this figure excludes years without fire in both classes (at ecoregion scale) and that y axes vary with ecoregion. Note also that the Coast Range and Klamath Mountains ecoregions are excluded due to negligible insect activity, although they are included in the Forested Total category. Fire extent from perimeters (MTBS; 1984-2012; <http://mtbs.gov>) within forested ecoregion and state boundaries. Insect activity from aerial detection surveys (ADS; 1970-2012).



**Fig. 4.9.** Fire likelihood vs. interannual fire extent across forested ecoregions of the PNW affected by recent mountain pine beetle (MPB) and western spruce budworm (WSB) activity (all years). Note that this plot excludes years with no area burned in both classes (at ecoregion scale). Note also that both the X and Y axes vary among ecoregions and that the Coast Range and Klamath Mountains ecoregions are excluded due to negligible insect activity, although they are included in the Forested Total category. Fire extent from perimeters (MTBS; 1984-2012; <http://mtbs.gov>) within forested ecoregion and state boundaries. Insect activity from aerial detection surveys (ADS; 1970-2012; see Methods).

**Table 4.1.** Insect-fire likelihood across forested ecoregions affected by recent insect activity.

Ecoregion <sup>1</sup>	Number of fire years <sup>2</sup>	Insect <sup>3</sup>	Fire type <sup>4</sup>	FL <sup>5</sup> (min) <sup>6</sup>	FL (25%) <sup>7</sup>	FL (median) <sup>8</sup>	FL (75%) <sup>7</sup>	FL (max) <sup>6</sup>	FL (mean)
North Cascades	21	MPB	Prior insect	0.000	0.029	<b>0.138</b>	0.996	1.337	1.265
			No prior insect	0.000	0.015	<b>0.093</b>	0.669	1.501	0.397
		WSB	Prior insect	0.000	0.015	0.097	0.999	2.381	0.647
			No prior insect	0.000	0.018	0.097	0.606	1.339	0.499
Northern Rockies	21	MPB	Prior insect	0.000	0.013	0.040	0.119	0.119	0.173
			No prior insect	0.001	0.018	0.031	0.152	0.298	0.116
		WSB	Prior insect	0.000	0.000	<b>0.000</b>	0.089	0.177	0.092
			No prior insect	0.001	0.025	<b>0.038</b>	0.159	0.308	0.133
West Cascades	16	MPB	Prior insect	0.001	0.233	<b>0.359</b>	0.722	0.995	0.567
			No prior insect	0.003	0.047	<b>0.105</b>	0.288	0.511	0.213
		WSB	Prior insect	0.000	0.054	0.280	0.806	1.233	0.748
			No prior insect	0.000	0.046	0.076	0.162	0.283	0.187
East Cascades	24	MPB	Prior insect	0.000	0.043	0.090	0.177	0.311	0.226
			No prior insect	0.000	0.049	0.178	0.309	0.642	0.293
		WSB	Prior insect	0.000	0.000	0.059	0.544	1.093	0.397
			No prior insect	0.000	0.046	0.104	0.173	0.231	0.217
Blue Mountains	28	MPB	Prior insect	0.000	0.003	0.073	0.577	1.214	0.371
			No prior insect	0.001	0.028	0.205	0.539	1.197	0.421
		WSB	Prior insect	0.000	0.009	<b>0.091</b>	0.418	0.726	0.298
			No prior insect	0.000	0.041	<b>0.267</b>	0.613	1.309	0.579
Forested Total	28	MPB	Prior insect	0.000	0.060	<b>0.187</b>	0.387	0.590	0.36
			No prior insect	0.000	0.049	<b>0.141</b>	0.241	0.489	0.223
		WSB	Prior insect	0.000	0.039	0.213	0.551	1.031	0.331
			No prior insect	0.000	0.046	0.130	0.218	0.377	0.223

*Notes:* This table corresponds directly to the boxplots in Fig. 6.

<sup>1</sup> We include only forested ecoregions (Omernik 1987) with substantial MPB or WSB activity (1970-2012, from ADS). Forested total includes the ecoregions shown here plus the Coast Range and Klamath Mountains, which are otherwise excluded due to minimal insect activity.

<sup>2</sup> We exclude years with no fire in a given ecoregion.

<sup>3</sup> MPB: mountain pine beetle; WSB: western spruce budworm.

<sup>4</sup> Fire type: with or without insect activity in all years preceding a given fire year (Fig. 2).

<sup>5</sup> FL: Fire likelihood based on percent burned of available area and calculated with two-way likelihood matrix (Fig. 2).

<sup>6</sup> Minimum and maximum values correspond to end of boxplot vertical lines (Fig. 6) and do not include outliers.

<sup>7</sup> 25% and 75% quartiles correspond to the outside edges of the boxplots (Fig. 6).

<sup>8</sup> Significant paired differences (Wilcoxon signed-rank test) indicated by orange (positive) and purple (negative) text at two FDR-adjusted  $\alpha$  levels (bold indicates FDR-adjusted  $P < 0.1$ ; bold and italic indicates FDR-adjusted  $P < 0.05$ ). See Tables 2 and 3 for additional details.

**Table 4.2.** Statistical summary of paired differences in insect-fire likelihood across forested ecoregions affected by mountain pine beetle (MPB) activity at different time lags.

Ecoregion <sup>1</sup>	Number of fire years <sup>2</sup>	Time lag <sup>3</sup>	Difference (median) <sup>4</sup>	Difference (95% CI) <sup>5</sup>	P-value <sup>6</sup>
North Cascades	21	1 - 5	0.204	(0.022 - 0.753)	<b><i>0.046</i></b>
	21	6 - 10	0.037	(-0.007 - 1.712)	0.246
	21	11 - 15	0.026	(-0.036 - 0.609)	0.649
	17	16 - 20	0.140	(-0.002 - 0.290)	0.110
	21	All (1 - 42)	0.145	(0.019 - 0.592)	<b><i>0.046</i></b>
Northern Rockies	21	1 - 5	-0.024	(-0.064 - -0.003)	<b><i>0.085</i></b>
	21	6 - 10	-0.018	(-0.043 - 0.130)	0.277
	21	11 - 15	-0.006	(-0.034 - 0.077)	0.663
	19	16 - 20	-0.008	(-0.032 - 0.239)	0.588
	21	All (1 - 42)	0.009	(-0.027 - 0.101)	0.706
West Cascades	16	1 - 5	0.777	(-0.053 - 1.449)	0.216
	16	6 - 10	0.213	(-0.003 - 1.559)	0.105
	16	11 - 15	0.024	(-0.081 - 0.759)	0.756
	14	16 - 20	-0.024	(-0.201 - 0.123)	0.705
	16	All (1 - 42)	0.263	(0.097 - 0.579)	<b><i>0.046</i></b>
East Cascades	24	1 - 5	0.042	(-0.059 - 0.200)	0.478
	24	6 - 10	0.010	(-0.056 - 0.178)	0.812
	24	11 - 15	-0.051	(-0.137 - 0.060)	0.517
	20	16 - 20	-0.067	(-0.120 - -0.013)	<b><i>0.068</i></b>
	24	All (1 - 42)	-0.067	(-0.128 - 0.010)	0.163
Blue Mountains	28	1 - 5	-0.064	(-0.202 - -0.006)	<b><i>0.073</i></b>
	28	6 - 10	-0.087	(-0.188 - -0.007)	<b><i>0.065</i></b>
	28	11 - 15	-0.055	(-0.193 - 0.058)	0.170
	23	16 - 20	-0.082	(-0.228 - -0.007)	<b><i>0.069</i></b>
	28	All (1 - 42)	-0.050	(-0.197 - 0.152)	0.345
Forested Total	28	1 - 5	0.148	(0.033 - 0.348)	<b><i>0.047</i></b>
	28	6 - 10	0.091	(0.017 - 0.228)	<b><i>0.068</i></b>
	28	11 - 15	0.083	(-0.008 - 0.225)	0.173
	23	16 - 20	0.019	(-0.030 - 0.094)	0.589
	28	All (1 - 42)	0.040	(0.003 - 0.163)	<b><i>0.072</i></b>

*Notes:* This table corresponds directly to Fig. 7. We assess paired differences with the Wilcoxon signed-rank test because it is robust to skewed distributions (see Methods).

<sup>1</sup> We include only forested ecoregions (Omernik 1987) with substantial MPB or WSB activity (1970-2012, from ADS). Forested total includes the ecoregions shown here plus the Coast Range and Klamath Mountains, which are otherwise excluded due to minimal insect activity.

<sup>2</sup> Number of years is reduced for the longest time lag we assessed since the earliest fire year in that subset is 1990.

<sup>3</sup> Time lags represent the years since first aerial survey observation of insect activity.

<sup>4</sup> Pseudomedian of the paired difference in fire likelihood with and without prior insect activity.

<sup>5</sup> Nonparametric confidence interval calculated after Bauer (1972).

<sup>6</sup> Significant paired differences indicated by orange (positive) and purple (negative) text at two  $\alpha$  levels (bold indicates FDR-adjusted  $P < 0.1$ ; bold and italic indicates FDR-adjusted  $P < 0.05$ ).

**Table 4.3.** Statistical summary of paired differences in insect-fire likelihood across forested ecoregions affected by western spruce budworm (WSB) activity at different time lags.

Ecoregion <sup>1</sup>	Number of fire years <sup>2</sup>	Time lag <sup>3</sup>	Difference (median) <sup>4</sup>	Difference (95% CI) <sup>5</sup>	P-value <sup>6</sup>
North Cascades	21	1 - 5	-0.187	(-0.516 - -0.011)	<b><i>0.049</i></b>
	21	6 - 10	-0.233	(-0.550 - -0.057)	<b><i>0.003</i></b>
	21	11 - 15	-0.187	(-0.520 - -0.054)	<b><i>0.003</i></b>
	17	16 - 20	-0.065	(-0.317 - 0.102)	0.265
	21	All (1 - 42)	0.018	(-0.170 - 0.344)	0.705
Northern Rockies	21	1 - 5	-0.039	(-0.147 - -0.015)	<b><i>0.029</i></b>
	21	6 - 10	-0.040	(-0.146 - -0.018)	<b><i>0.021</i></b>
	21	11 - 15	-0.075	(-0.151 - -0.022)	<b><i>0.001</i></b>
	19	16 - 20	-0.029	(-0.093 - -0.010)	<b><i>0.064</i></b>
	21	All (1 - 42)	-0.037	(-0.075 - -0.012)	<b><i>0.064</i></b>
West Cascades	16	1 - 5	-0.167	(-0.366 - -0.076)	<b><i>0.014</i></b>
	16	6 - 10	-0.126	(-0.273 - -0.003)	<b><i>0.095</i></b>
	16	11 - 15	0.740	(-0.077 - 3.025)	0.505
	14	16 - 20	0.392	(-0.079 - 0.988)	0.183
	16	All (1 - 42)	0.266	(-0.062 - 0.701)	0.170
East Cascades	24	1 - 5	-0.079	(-0.158 - 0.030)	0.163
	24	6 - 10	-0.098	(-0.259 - -0.019)	<b><i>0.069</i></b>
	24	11 - 15	-0.114	(-0.238 - -0.028)	<b><i>0.050</i></b>
	20	16 - 20	-0.088	(-0.148 - 0.232)	0.138
	24	All (1 - 42)	0.060	(-0.075 - 0.353)	0.702
Blue Mountains	28	1 - 5	-0.241	(-0.425 - -0.076)	<b><i>0.001</i></b>
	28	6 - 10	-0.227	(-0.402 - -0.086)	<b><i>0.001</i></b>
	28	11 - 15	-0.109	(-0.257 - 0.016)	0.138
	23	16 - 20	-0.164	(-0.410 - -0.046)	<b><i>0.010</i></b>
	28	All (1 - 42)	-0.124	(-0.464 - -0.008)	<b><i>0.064</i></b>
Forested Total	28	1 - 5	-0.092	(-0.165 - -0.017)	<b><i>0.064</i></b>
	28	6 - 10	-0.060	(-0.175 - 0.029)	0.255
	28	11 - 15	-0.021	(-0.094 - 0.089)	0.682
	23	16 - 20	-0.030	(-0.083 - 0.158)	0.345
	28	All (1 - 42)	0.113	(-0.002 - 0.264)	0.110

*Notes:* This table corresponds directly to Fig. 8. We assess paired differences with the Wilcoxon signed-rank test because it is robust to skewed distributions (see Methods).

<sup>1</sup> We include only forested ecoregions (Omernik 1987) with substantial MPB or WSB activity (1970-2012, from ADS). Forested total includes the ecoregions shown here plus the Coast Range and Klamath Mountains, which are otherwise excluded due to minimal insect activity.

<sup>2</sup> Number of years is reduced for the longest time lag we assessed since the earliest fire year in that subset is 1990.

<sup>3</sup> Time lags represent the years since first aerial survey observation of insect activity.

<sup>4</sup> Pseudomedian of the paired difference in fire likelihood with and without prior insect activity.

<sup>5</sup> Nonparametric confidence interval calculated after Bauer (1972).

<sup>6</sup> Significant paired differences indicated by orange (positive) and purple (negative) text at two  $\alpha$  levels (bold indicates FDR-adjusted  $P < 0.1$ ; bold and italic indicates FDR-adjusted  $P < 0.05$ ).

## CHAPTER 5 || CONCLUSION

### SYNTHESIS

#### *Research summary*

This dissertation develops novel geospatial datasets and integrates these with existing regional databases to provide unique perspectives on highly relevant forest disturbance phenomena. My research is rooted in field-based observations of tree mortality associated with two regionally important insects in the Pacific Northwest (PNW) – mountain pine beetle (MPB; *Dendroctonus ponderosae* Hopkins [Coleoptera: Scolytidae]; a bark beetle) and western spruce budworm (WSB; *Choristoneura freemani* Razowski [Lepidoptera: Tortricidae]; a defoliator). By integrating field observations with LandTrendr spectral trajectories (Kennedy et al. 2010), my colleagues and I were able to capture the variable timing and severity of insect impacts on tree mortality and fuels (Chapter 2). Then, building from a small number of forest plots in the Oregon Cascades ( $n = 38$ ) to a regional subset of federal forest inventory plots ( $n = 282$ ), and from two Landsat scenes to the entire region, we created new maps with unprecedented resolution to summarize insect disturbance dynamics (Chapter 3). These maps leverage the spatial grain (30-m) and temporal depth (1984-2012) of the Landsat TM/ETM+ sensor but are constrained within areas affected by MPB or WSB via an independent remote sensing dataset, aerial detection surveys (ADS), and they are displayed in field-based units of dead tree basal area mortality. In processing the ADS data for the insect mapping analysis, we realized that the ADS data themselves could be leveraged to assess insect-fire interactions, and that their temporal depth to 1970 was a particular strength for investigating different time lags of prefire insect activity (Chapter 4). Although the insect-fire likelihood analysis yielded tangible results that insects do indeed influence wildfire likelihood within particular scales and conditions, many uncertainties remain, and it is possible that refined datasets or methods will reveal additional or different interactions. In this sense, my dissertation serves primarily to establish a framework within which to evaluate the new data and address additional questions.

### *Emergent themes*

Several key themes emerge from this research. First is the primacy of scale. Because scale is such an important aspect of both the disturbance phenomena themselves, as well as our ability to detect the ecological changes they render, our results underscore the importance of geospatial datasets that span multiple scales in space and time. The analyses in this dissertation cover a range of spatial scales (individual trees in 1-ha inventory plots, 30-m Landsat pixels, insect and fire polygons of varying extent, broader landscapes encapsulated by ecoregions, and the PNW region as a whole), as well as temporal scales (years to decades). That said, the phenomena of interest cover many finer scales, and they are influenced by very broad top down controls as well. MPB adults attack an individual tree by the thousands to overcome its defenses at the cellular level, while swarms of WSB caterpillars consume individual tree needles, with both insects operating at sub-daily time scales. Both insects can kill individual trees, and large outbreaks can result in many dead trees across landscapes and even regions over the course of years and decades (Raffa et al. 2008, Swetnam and Lynch 1993). Similarly, wildfire operates at a fine scale as it consumes individual tree needles and branches, interacting with microtopography and fine-scale wind patterns at the time scale of seconds or even finer. An underlying assumption of this dissertation research is that these fine-scale insect and fire processes manifest themselves consistently at the forest stand scale, and that a 30-m Landsat pixel is an adequate size to capture tree mortality patterns at an annual time step. Previous research supports this assumption (e.g., Cohen and Goward 2004, Kennedy et al. 2014), but future studies nevertheless should pursue fine-scale mechanisms and interactions, accounting for uncertainties across multiple scales wherever possible.

Another emergent theme from this research, particularly the insect-fire likelihood analysis (Chapter 4), is the disproportionate effects of rare events. Wildfire distributions are very positively skewed (i.e., there are many small fire events and few large events), and the very largest fire events account for a majority of interannual fire extent (e.g., 1% of fires account for 80-96% of burned area; Strauss et al. 1989). Thus, the large fire events that really transform forest landscapes (such as the 200,000 ha Biscuit Fire) may happen only once in a 29 year fire dataset like the one used here if we researchers are lucky, or they may not happen at all. Because there is not a compelling relationship between annual fire extent and the difference in fire

likelihood due to prior insect activity (Fig. 4.9), this study suggests that other factors (such as climate and fire suppression) control the disproportionately large fire years. That said, there were several prominent examples of extreme insect-fire overlap events. Even if these were not the largest fire years, they still represent key isolated examples of fire extent being higher in areas with prior insect activity, and they are inherently difficult to account for with standard statistical tools.

A final emergent theme from this research is the importance of intermediate disturbances. Most PNW forests fall into the broad category of mixed-severity fire regimes (e.g., Hessburg et al. 2007, Tepley et al. 2013), which is an intermediate disturbance (by definition between low- and high-severity fire regimes). Although these chapters did not explicitly account for burn severity, the majority of the area burned in a given year is low- to moderate-severity fire (Meigs 2009). In addition, the insect outbreaks examined here are intermediate in both severity (i.e., tree mortality) and frequency. They kill a portion of overstory trees but rarely all trees, even in forests dominated by host trees like relatively pure lodgepole pine in South Central Oregon. They also affect PNW forests on an intermediate, decadal time scale. Although these intermediate disturbances by definition do not have as severe per-unit-area impacts as stand-replacing fire or logging, their extent can be quite vast, as is the case with WSB. Thus, the cumulative impacts of the intermediate disturbance may surpass those of more abrupt disturbances (e.g., Stueve et al. 2011).

## BROADER IMPLICATIONS

### *Forest health context*

The results from this dissertation highlight the widespread and potentially fundamental ecological roles that native disturbances play in PNW forests. WSB, MPB, and wildfire have been relatively rare at the regional scale in any particular year, but all three have had and will continue to have profound cumulative effects, as well as episodic, intense impacts on particular forest stands and landscapes. Given concerns about forest health in a rapidly changing climate, long-term monitoring will enable forest researchers and managers to quantify and anticipate the independent and interactive effects of insects, wildfires, and other disturbances. In so doing,

they may be able to help define baseline conditions and variability in effects that represent these native disturbances functioning as important agents of healthy forests versus potentially unprecedented effects that could destabilize ecosystems and the services they provide. All bets are off for the future, but this dissertation does indicate that recent effects of insects and wildfires are not the forest health crisis that some people fear.

### *Final thoughts*

A summer day hardly passes in the Pacific Northwest without mention of wildfires, and insect outbreaks are often mentioned in the same sentence. The conventional wisdom that insect outbreaks necessarily lead to wildfires – thereby increasing their likelihood – is firmly embedded in the public consciousness. Although this dissertation does present some evidence of positive feedbacks between insects and wildfires, it also presents evidence of negative feedbacks, as well as numerous examples of no apparent feedbacks. Like many questions in ecology, the answers depend on multiple factors, the answers are complicated, and the answers could lead to increased confusion and potential exploitation by those with particular agendas. Continued field-based and remotely sensed monitoring of forest dynamics likely will help to determine how these native disturbances do or do not interact across time and space, and how these interactions might shift with climate change and forest management actions. Ongoing research will also help track the impacts of each disturbance separately, which may be just as useful to managers and policy makers if the rates of overlap and interaction remain low. Given the nonstationarity in disturbance dynamics in the accelerating Anthropocene, long-term research and methods that span multiple scales will continue to be essential components of adaptive management. Indeed, this dissertation, though relatively extensive and data rich, is still just a snapshot in forest time.

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