Landscape vegetation change, pattern detection, and interpretation in a subalpine fir forest infested with balsam woolly adelgid

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Abstract

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Invasive insects and pathogens, climate change, and anthropogenic factors are causing broad-scale change in forest ecosystems. Identifying, understanding, and maintaining historic and functioning ecosystems in changing landscapes can be challenging because multiple internal and external factors interact across scales. In this dissertation, I present a hierarchical approach to investigate patterns of health and mortality of subalpine fir (*Abies lasiocarpa* (Hook) Nutt. var. *lasiocarpa*) associated with a non-native herbivorous insect, the balsam woolly adelgid (BWA), *Adelges piceae* Ratzeburg, and landscape disturbance on the Olympic Peninsula, Washington, U.S.A. Assessment across broad and fine spatial and temporal scales allowed the identification of distinct boundaries of change associated with climate, topography, vegetation, and disturbance processes of fire, winter weather events, mass movement, and endemic and introduced insects and pathogens. I used Landsat imagery at broad spatial and temporal scales to differentiate patterns of very-high-, high-, and low-magnitude disturbance in subalpine-fir forest using the normalized burn ratio (NBR). Low-magnitude decline in NBR gradually increased in area over time (1994-2007) and occurred predominantly at high elevations and southern aspects during years of warm PDO index values, suggesting interactive effects of an invasive biotic agent and climate. I conducted field studies that identified BWA as the biotic agent that was consistently present in affected areas and associated with poor health and mortality of subalpine fir. Using dendrochronology, I determined that variability in the growth of subalpine fir was positively associated with the arrival of BWA in a forest along Hurricane Ridge in Olympic National Park. I compared aerial detection surveys, Landsat imagery, and dendrochronology methods and confirmed that all three methods independently detected effects of BWA for the same time period. These methods provided a more complete assessment that would not have been possible with one method alone. This dissertation demonstrates the utility of a hierarchical approach that combines remote sensing and field validation to detect patterns of disturbance and identify potential fine- and broad-scale causal mechanisms of forest change.

Dedication

This dissertation is dedicated to my daughter Reyn on her 16th birthday.

We have learned much together, and I am grateful to have been able to share the scientific process with my hard-working and conscientious critical thinker.

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Chapter 1

INTRODUCTION

Maintaining pristine, historic conditions may no longer be feasible in many protected areas (Baron et al. 2009, Stephenson 2014). The threat of permanent change to vegetation composition and structure is increasingly recognized (Seastedt et al. 2008, Hobbs et al. 2009).Managers are faced with unpredictable introductions of non-native insects and disease such as balsam woolly adelgid (*Adelges piceae*), hemlock woolly adelgid (*Adelges tsugae*), emerald ash borer (*Agrilus planipennis*), and sudden oak death (*Phytophthora ramorum)* (Hayes and Ragenovich 2001, Ellison et al. 2005, Pimentel et al. 2005, Evans and Oszaco 2007, Tomback and Resler 2007, Dukes et al. 2009). Infestation by non-native insects and diseases is occurring simultaneously with climate change which can have significant but unexpected ecological consequences (Mckenzie et al. 2003, Hennon et al. 2006, Stephenson et al. 2006, Adams et al. 2009, Kliejunas et al. 2009, Bentz et al. 2010, Littell et al. 2010, Allen et al. 2010, Sturrock et al. 2011) and cause alteration of disturbance regimes (Flannigan et al. 2000, McKenzie et al. 2004, 2009, Westerling et al. 2006, Littell et al. 2009). Prevention of species introductions may be difficult because factors that originate outside land management boundaries may be beyond the control of land managers. Monitoring change in forested landscapes is also important so that actions can be taken before damage becomes extensive (Kennedy et al. 2007a, Fancy et al. 2009, Tu 2009). After non-native species introductions have already occurred, the next steps are learning, adaptation, and acting to preserve ecosystem function and integrity (e.g., Westley 2002, Baron et al. 2009, Tweed 2010, Millar et al. 2014). Monitoring and research have important roles in each of these phases to inform management regarding the location and dynamics of change.

COMPLEMENTARY MONITORING AND RESEARCH TOOLS

A collaborative approach in research and management is needed to develop innovative solutions that meet multiple objectives (Shindler et al. 2011). This is especially true in the development and application of landscape monitoring and research tools. Monitoring for change detection with field sampling and aerial surveys has been conducted for decades to inform management decisions (USDA 1992, Ciesla 2006, Fancy et al. 2009). Although advances have been made in the application of remote sensing tools (Cohen et al. 2010, Kennedy et al. 2010), these methods may be adopted slowly because of apparent start-up costs and the need for advanced training. Furthermore, land managers may not appreciate the value of a combined approach that uses remote sensing in concert with other field and modeling efforts. My research lays the groundwork for complementary use of change detection methods by applying multiple approaches to detect patterns and quantify relationships between landscape change and agents of forest disturbance. I demonstrate how satellite imagery can be used to supplement ADS data and vice versa, and I provide examples of how further development and applications of this approach are needed in forest research and management (see last section of Chapter 4).

DISSERTATION CHAPTERS AND OBJECTIVES

In the following four chapters I assess forest vegetation change within a protected area, and I focus on subalpine fir of Olympic National Park, Washington, USA, that is infested with the invasive insect balsam woolly adelgid (BWA). I use a hierarchical approach that illustrates the importance of conducting investigations at multiple scales with complementary methods that range from remote sensing to field sampling and dendrochronology. By integrating methods with a multi-scale approach and an awareness of hierarchical relationships I provide a more holistic assessment of spatial and temporal distributions, emergent patterns, and ecological processes that may be interacting to cause vegetation change. Chapter 2 presents the hierarchical patch dynamics (HPD) model as a conceptual framework for identifying and communicating factors that interact across scale and influence system dynamics. This framework is applied to set the context for my research of forest change patterns. Chapter 3 uses remote sensing tools to observe landscape patterns of vegetation change from 1984-2012, and to relate this change to climate patterns, weather events, and insect activity. Chapter 4 uses field-sampled data and remote sensing tools to compare vegetation change, tree mortality, and tree health in areas with and without BWA. Chapter 5 investigates the effects of BWA on tree growth and mortality in an infested subalpine fir forest, compares growth to an uninfested forest, and identifies a growth signal for infestation activity, comparing onset date with change detected in remotely sensed data. Chapter 6 concludes the dissertation with an overview of research results and a discussion of management strategies and opportunities.

REFERENCES

Adams, H. D., M. Guardiola-Claramonte, G. a Barron-Gafford, J. C. Villegas, D. D. Breshears, C. B. Zou, P. a Troch, and T. E. Huxman. 2009. Temperature sensitivity of drought-induced tree mortality portends increased regional die-off under global-change-type drought. Proceedings of the National Academy of Sciences of the United States of America 106:7063–7066.

Allen, C. D., A. K. Macalady, H. Chenchouni, D. Bachelet, N. McDowell, M. Vennetier, T. Kitzberger, A. Rigling, D. D. Breshears, E. H. (Ted) Hogg, P. Gonzalez, R. Fensham, Z. Zhang, J. Castro, N. Demidova, J.-H. Lim, G. Allard, S. W. Running, A. Semerci, and N. Cobb. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management 259:660–684.

Baron, J. S., L. Gunderson, C. D. Allen, E. Fleishman, D. McKenzie, L. a Meyerson, J. Oropeza, and N. Stephenson. 2009. Options for national parks and reserves for adapting to climate change. Environmental Management 44:1033–42.

Bentz, B. J., J. Regniere, M. Hansen, J. L. Hayes, and J. A. Hicke. 2010. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. Bioscience 60:602–613.

Ciesla, W. M. 2006. Aerial signatures of forest insect and disease damage in the western United States. Forest Health Technology Enterprise Team. FHTET-01-06. USDA Forest Service, Fort Collins, Colorado.

Cohen, W. B., Z. Yang, and R. Kennedy. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 2. TimeSync - Tools for calibration and validation. Remote Sensing of Environment 114:2911–2924.

Dukes, J. S., J. Pontius, D. Orwig, J. R. Garnas, V. L. Rodgers, N. Brazee, B. Cooke, K. a. Theoharides, E. E. Stange, R. Harrington, J. Ehrenfeld, J. Gurevitch, M. Lerdau, K. Stinson, R. Wick, and M. Ayres. 2009. Responses of insect pests, pathogens, and invasive plant species to climate change in the forests of northeastern North America: What can we predict? Canadian Journal of Forest Research 39:231–248.

Ellison, A. M., M. S. Bank, B. D. Clinton, E. A. Colburn, K. Elliott, C. R. Ford, D. R. Foster, B. D. Kloeppel, J. D. Knoepp, G. M. Lovett, J. Mohan, D. A. Orwig, N. L. Rodenhouse, W. V Sobczak, K. A. Stinson, J. K. Stone, C. M. Swan, J. Thompson, B. Von Holle, and J. R. Webster. 2005. Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Frontiers in Ecology and the Environment 3:479–486.

Evans, H., and T. Oszaco, editors. 2007. Alien Invasive Species and International Trade. Forest Research Institute, Warsaw.

Fancy, S. G., J. E. Gross, and S. L. Carter. 2009. Monitoring the condition of natural resources in US national parks. Environmental Monitoring Assessment 151:161–174.

Flannigan, M. D., B. J. Stocks, and B. M. Wotton. 2000. Climate change and forest fires. The Science of the Total Environment 262:221–9.

Hayes, J. L., and I. Ragenovich. 2001. Non-native invasive forest insects of Eastern Oregon and Washington. Northwest Science 75:77–84.

Hennon, P., D. Wittwer, A. Johnson, P. Schaberg, G. Hawley, C. Beier, S. Sink, and G. Juday. 2006. Climate warming, reduced snow, and freezing injury could explain the demise of yellow-cedar in Southeast Alaska, USA. World Resource Review 18:427–450.

Hobbs, R. J., E. Higgs, and J. a. Harris. 2009. Novel ecosystems: implications for conservation and restoration. Trends in Ecology and Evolution 24:599–605.

Kennedy, R. E., W. B. Cohen, A. A. Kirschbaum, and E. Haureiter. 2007. Protocol for Landsat-based monitoring of landscape dynamics of North Coast and Cascades Network Parks: 2-G1. U.S. Geological Survey Techniques and Methods, Reston, VA.

Kennedy, R. E., Z. Yang, and W. B. Cohen. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 1. LandTrendr — temporal segmentation algorithms. Remote Sensing of Environment 114:2897–2910.

Kliejunas, J. T., B. W. Geils, J. M. Glaeser, E. M. Goheen, P. Hennon, M. Kim, H. Kope, J. Stone, R. Sturrock, and S. J. Frankel. 2009. Review of Literature on Climate Change and Forest Diseases of Western North America. USDA Forest Service, Pacific Southwest Research Station. General Technical Report PSW-GTR-225., Albany, CA.

Littell, J., E. Oneil, D. McKenzie, and J. Hicke. 2010. Forest ecosystems, disturbance, and climatic change in Washington State, USA. Climatic Change:129–158.

Littell, J. S., D. McKenzie, D. L. Peterson, and A. L. Westerling. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916-2003. Ecological Applications 19:1003–1021.

McKenzie, D., Z. Gedalof, D. L. Peterson, and P. Mote. 2004. Climatic change, wildfire, and conservation. Conservation Biology 18:890–902.

McKenzie, D., D. L. Peterson, and J. Littell. 2009. Global warming and stress complexes in forests of western North America. Pages 319–337 *in* A. Bytnerowicz, M. J. Arbaugh, A. R. Riebau, and C. Anderson, editors. Wildland Fires and Air Pollution. Elsevier, Amsterdam/London.

Mckenzie, D., D. W. Peterson, D. L. Peterson, and P. E. Thornton. 2003. Climatic and biophysical controls on conifer species distributions in mountain forests of Washington State , USA. Journal of Biogeography 30:1093–1108.

Millar, C. I., C. W. Swanston, and D. L. Peterson. 2014. Adapting to climate change. Pages 183–222 *in* D. L. Peterson, J. M. Vose, and T. Patel-Weynand, editors. Climate Change and United States Forests. Springer Science + Business Media, Dordrecht Heidelberg New York London.

Pimentel, D., R. Zuniga, and D. Morrison. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics 52:273–288.

Seastedt, T. R., R. J. Hobbs, and K. N. Suding. 2008. Management of novel ecosystems: are novel approaches required? Frontiers in Ecology and the Environment 6:547–553.

Shindler, B., R. Gordon, S. McCaffrey, and E. Toman. 2011. Collaborating for healthy forests and communities: A guide for building partnerships among diverse interests. Corvallis, OR, USA.

Stephenson, N. L. 2014. Making the transition to the third era of natural resources management. The George Wright Forum 31:227–235.

Stephenson, N., D. Peterson, D. Fagre, C. Allen, D. McKenzie, J. Baron, and K. O’Brian. 2006. Response of western mountain ecosystems to climatic variability and change: the western mountain initiative. Park Science 24:24–29.

Sturrock, R. N., S. J. Frankel, a. V. Brown, P. E. Hennon, J. T. Kliejunas, K. J. Lewis, J. J. Worrall, and a. J. Woods. 2011. Climate change and forest diseases. Plant Pathology 60:133–149.

Tomback, D. F., and L. M. Resler. 2007. Invasive pathogens at alpine treeline: consequences for treeline dynamics. Physical Geography 28:397–418.

Tu, M. 2009. Assessing and managing invasive species within protected areas. *in* J. Ervin, editor. Protected Area Quick Guide Series. The Nature Conservancy, Arlington, VA.

Tweed, W. C. 2010. Uncertain Path. University of California Press, Berkeley and Los Angeles, California.

USDA. 1992. Forest Service resource inventories: an overview. USDA Forest Service. Forest Inventory, Economics, and Recreation Research Staff, Washington, D.C.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science 313:940–943.

Westley, F. 2002. The devil in the dynamics: adaptive management on the front lines. Pages 333–360 *in* L. H. Gunderson and C. S. Holling, editors. Panarchy. Island Press, Washington, D.C.

Chapter 2

A HIERARCHICAL LANDSCAPE FRAMEWORK FOR INVESTIGATING VEGETATION CHANGE IN PROTECTED AREAS

SUMMARY

Invasive species, climate change, and anthropogenic factors interact across spatial and temporal scales and pose significant challenges to land managers of protected areas. Consequently, a conceptual framework is needed that facilitates understanding and communication of complex ecosystem dynamics. I demonstrate the use of a hierarchical framework to conceptualize constraints and processes of disturbance in a forested ecosystem on the Olympic Peninsula in Washington State, USA. I present and apply these hierarchical concepts to examine physical geographical and disturbance factors that influence large-scale patterns in subalpine fir (*Abies lasiocarpa* var. *lasiocarpa*). Climate constrains and influences vegetation patterns at broad spatial and temporal scales and fluctuates at decadal and annual scales (Pacific Decadal Oscillation and El Niño Southern Oscillation). At intermediate spatial and temporal scales, vegetation composition and structure are influenced by topography and variable exposure to sun, wind, adiabatic cooling, and orographic effects. At finer scales, vegetation interacts with cyclical abiotic and biotic disturbance regimes, including fire, winter weather events, endemic and non-native insects and pathogens, and synergistic disturbance complexes. Vegetation patterns are the result of multiple factors at different levels of scale. This hierarchical framework incorporates simultaneous broad- and fine-scale approaches to identify emergent patterns, trends, and causal mechanisms, and can be used by land managers to anticipate potential trajectories of landscape change and to develop short- and long-term ecological goals for protected areas.

INTRODUCTION

Managers of protected areas are faced with increasingly difficult management decisions related to the preservation of native vegetation and functioning ecosystems (West et al. 2009). Permanent changes to vegetation that are now occurring due to direct and indirect effects of invasive species, climate change, and other anthropogenic factors may have cascading effects for many ecosystems and ecosystem services (Rabenold et al. 1998, Van Hemert et al. 2014). Some protected areas have experienced change that cannot be remedied by traditional management strategies and encompass local landscapes that cannot be feasibly returned to historic conditions (Seastedt et al. 2008). Collaborative development of monitoring and research is often presented as a means to identify and understand management-related ecosystem problems and to guide adaptive responses (Kennedy et al. 2007a, Fancy et al. 2009, Woodward et al. 2009, Halofsky et al. 2011, Peterson et al. 2011). For this to be effective, managers and scientists need conceptual tools to visualize and communicate the factors involved.

In this chapter I demonstrate the use of a hierarchical framework for establishing context for research. The dominant structure for this framework is provided by the hierarchical patch dynamics (HPD) model (Allen and Starr 1982, Levin 1992, Wu and Loucks 1995). A focal subject is presented in relation to influencing constraints and processes, cycles, and mechanisms, with an awareness of scale and interaction among factors. The use of HPD does not exclude other conceptual models that highlight details of interaction (e.g., Manion’s disease spiral, disturbance complex diagrams, or the disease triangle). My objectives are to (1) present concepts from the hierarchical patch dynamics model, and (2) apply the concepts to an examination of factors influencing large-scale vegetation change and resulting landscape patterns in subalpine fir (*Abies lasiocarpa* var. *lasiocarpa*) forest of the Olympic Peninsula in Washington State, USA. I describe physical geographical context, key processes and interactions within the subalpine fir ecosystem, and then discuss the management implications for loss of a foundation species (Ellison et al. 2005).

HIERARCHICAL PATCH DYNAMICS MODEL

The hierarchical patch dynamics (HPD) model describes a theoretical hierarchical structuring of spatial, temporal, and organizational elements that are interrelated and linked across and within scales (O'Neill et al. 1989, Levin 1992, Wu and Loucks 1995). Elements are grouped in nested levels based on strength of interaction, similar speed, and geometric or spatial attributes, e.g., processes occurring within a forest, a tree, a leaf, or a cell (Simon 1974, Holling 2001). This model may be used to describe components of any system (e.g., ecological, economic, or social) but will be applied here to describe factors influencing a forested ecosystem.

Ecosystem studies benefit by incorporating three nested levels to evaluate broad- and fine-scale factors that influence the pattern being investigated (Wu 1999; **Fig. 2.1a**). The three levels are (1) constraints composed of broad-scale, slow-acting physical geographical elements (e.g., climate, topography, and vegetation distribution) that are hierarchically dominant and constrain patterns and processes at finer scales; (2) the focal-level that contains the pattern, object, or process of interest (e.g., abundance or distribution of tree mortality); and (3) fine-scale, fast-acting processes (e.g. interacting disturbance agents, demographics, or chemical cycling) that create emergent patterns at the focal level.

The concept of hierarchical patch dynamics supports and explains the co-existence of dynamic heterogeneity, meta-stability, and cyclical processes, as well as the possibility for systems to transition into altered states (Holling 2001, Holling and Gunderson 2002). For example, at one location a forest stand may have experienced sudden change in structure and composition due to disturbance. Successional dynamics will continue to drive structural and compositional change at that location until the next disturbance, in a dynamic and cyclical process. At a larger scale, many patches are in different phases of disruption and regrowth across the landscape, and this creates a heterogeneous and metastable system. Constraining factors such as climate help to maintain the system (and support resilient processes such as seed production). Over geologic time scales, constraining factors change, and metastable systems transition to new states. Anthropogenic climate change and introduced insects and disease can reduce these time scales and shift ecosystems into altered states more quickly than the adaptive capacity of most plants and animals (**Fig. 2.1c**).

Importance of scale in detection and interpretation of patterns

Ecological data that are collected at broad spatial and temporal scales may reveal emergent patterns resulting from interactions between constraints and processes; while data collected with a fine grain size can document details of underlying mechanisms (White and Pickett 1985, Wiens 1989, Wu 1999). Effective research is designed to (1) encompass the spatial extent of the interaction or relationship being studied, (2) span the duration or temporal extent of the process being observed, preferably bracketed by inactive periods and long enough to capture cyclical patterns, (3) include data collected at relevant time intervals or seasons, (4) have enough detail to investigate important mechanisms in the pattern-process relationship, and (5) have appropriate replication to cover spatial and temporal variability in system responses.

Ecological studies may be limited by observational tools, land management boundaries, funding, time, and effort. Limitations and assumptions need to be identified, but can also be supplemented by substitutions or complementary methods. For example, when examining a process that occurs over very long time scales, space may be substituted for time by placing sample plots in a range of developmental or progressive stages (Pickett 1989) assuming that stages can be identified and other differences are considered. Dendrochronological methods can also extend the time period for tree response research to 100 or more years in the past. If tree core data cover a small spatial extent (as in Chapter 5), they may be complemented by satellite data which have a broad spatial extent but a shorter time period; if both methods detect a tree response, the data from field studies and satellites may be used together to increase spatial and temporal extent.

Patches as flexible and unit-less delineations of space

The patch is a central concept in the HPD model and is often used to describe pattern. Patches or patchiness are defined as spatial units or pattern that may vary in size, internal homogeneity, and discreteness, and differ from their surroundings in nature or appearance (White and Pickett 1985, Wiens 1989). A patch may define spatial boundaries of a biotic community, an area experiencing disturbance, or a different phase in the adaptive cycle (e.g., Holling 2001). Patches can range in scale from part of a leaf to a whole tree or an entire landscape. They may be described in relation to the size and distribution of adjacent patches occurring on the landscape, or in terms of process details occurring within the patch such as tree mortality or succession. Temporal characteristics such as duration or frequency of recurrence may be attributed to a patch or patch type that is associated with a particular process. The flexibility of the patch concept has proven useful for describing ecosystem heterogeneity, patterns of change associated with disturbance, and for explaining coexistence of species that are able to utilize different spatial and temporal niches (Levin 1992).

Patterns correspond to processes and constraints

The patchwork of patterns on a landscape at any moment in time is the product of interacting processes that are limited or driven by pre-existing patterns and constraints. Thus, patterns are expected to be correlated with distributions of causal processes in space and time and are bound by the spatial or temporal extent of constraints. Process-driven patterns may be stochastic, indiscriminate or species-specific, and they may show diagnostic signs and symptoms. For example, wildfire in the subalpine fir zone is stochastic, indiscriminate, and severe. It is often influenced by topography and weather at short time scales, or climate on longer time scales. Insect and disease agents, in contrast, often target specific tree species, and patterns are constrained by tree species distribution and geography. Patterns from the effects of climate-change may have a broad extent, and occur simultaneously across a landscape with predictably patchy distribution where temperature and moisture are altered by topography or where species differ in susceptibility.

The strength of the relationship between pattern and process can be determined with statistical methods. Further evidence may be derived from complementary detection methods and supported by observation, modeling (Cairns et al. 2008), and experiments that evaluate cause-and-effect relationships (Levin 1992, Raffa et al. 2008). After a specific process-pattern relationship is established, pattern may be used as an indication of process, e.g., shape, size, location, and duration of patches associated with landslides (Antonova et al. 2011) or to indicate when broad-scale constraints may be affecting processes (McIntire and Fajardo 2009).

Change: metastable systems vs. systems in transition

Change occurs in any system, and the perceived effect of this change depends on scale. Change that occurs slowly, infrequently, or in small patches generally does not interfere with system function. Terms describing such conditions include baseline mortality, metastability, and historic range of variability (HRV). Ecological systems operating within HRV are expected to (1) persist without interference from human activities, (2) experience change from cyclic processes such as disturbance, competition, and succession, and (3) support habitat and species that have adapted to conditions over an extended time period and geographical area (Swanson et al. 1994, Landres et al. 1999). System stability is preserved by properties of resilience, resistance, and persistence that may be found in adaptations such as vegetative re-sprouting, defense mechanisms, rooting depth or bark thickness, seed bank storage, or the ability of animal populations to move from disturbed habitat or into vacated habitat. In this manner, plant and animal communities are perpetuated within a system of historic changes and cycles.

Holling and Gunderson (2002) recognized four phases of the cyclic process which they call the adaptive cycle: (1) release (disturbance or disruption), (2) reorganization (renewal, opportunity), (3) exploitation (growth), and (4) conservation (stability, connectedness, rigidity, wealth). The adaptive cycle fits within the HPD model because cycles operate at each level of scale, simultaneously moving through different phases at different rates, and interacting with other cycles within and across levels. Knowing the phase of a cycle provides some predictive power regarding system response to disturbance or management actions.

System-altering change, in contrast, results from a change in constraints or processes beyond the adaptive capacity of components. It may be detected initially as a disruption in the frequency, intensity, or extent of disturbance (Landres et al., 1999; **Fig. 2.1b**), such as an increase in wildfire (Littell et al., 2010), bark beetle activity (Bentz et al. 2010), or tree mortality associated with an introduced insect such as the balsam woolly adelgid (Mitchell and Buffam 2001). System change may be more evident when pattern disruption is extreme, persistent, frequent, or associated with novel processes. Disruption patterns may begin locally as fine-scale processes and spread to affect larger areas, or may be detected as simultaneous broad-scale disruptions occurring over thousands of kilometers (spatial disjunctions due to topography or vegetation) as has been occurring with climate change (van Mantgem and Stephenson 2007, Vogelmann et al. 2009). Process changes can cascade upward and affect constraints as well. For example, climate change originated from an increase in carbon dioxide (CO_2) production from combustion processes, and then CO_2 interacted with atmospheric processes and altered global dynamics of heat transfer (IPCC 2013). These effects on climate have cascaded downward to affect processes at lower hierarchical levels including weather events, insect populations, and fire (IPCC 2014).

APPLYING A HIERARCHICAL FRAMEWORK IN OLYMPIC NATIONAL PARK

The balsam woolly adelgid (BWA; *Adelges piceae* (Ratzeburg)) has become a wide-spread agent of disturbance in North America (Smith and Nicholas 1998, Livingston et al. 2000, Mitchell and Buffam 2001) and is interacting with regional physiography and other disturbance agents to alter the structure and composition of true fir (*Abies* spp.) forests (Dale et al. 1991, Hollingsworth and Hain 1994). For an examination of declining vegetation health in subalpine fir forest of Olympic National Park, I use the hierarchical framework described above to organize constraints and processes relative to their scales of influence (**Fig. 2.2**).

Constraints: physical geographical context

Geography and geology are largely responsible for climate, topography, and vegetation distribution. Climate is the overarching constraint on the distribution of plant species, disturbance regimes, and other processes at lower hierarchical levels, and the effects of climate are modified by topography. At finer spatial scales, the influence of climate becomes less apparent as micro-environment and local processes have a more direct role. Topography modifies energy and moisture distributions through adiabatic cooling (elevation), exposure to solar radiation (aspect), orographic effects on precipitation, soil development, and soil moisture (slope and topographic position). Climate and topography constrain vegetation through their interaction and create finer-scale species assemblages categorized in plant association groups (Crawford et al. 2009). Vegetation acts as a constraint for disturbance agents by influencing their distribution and behavior.

Climate and topography of the Olympic Peninsula

The Olympic Peninsula is located on the western edge of North America and is adjacent to the Pacific Ocean, the Strait of Juan de Fuca, and the Puget Sound (**Fig. 2.3a, b**). Several mountain ranges reach 1,800 m in elevation at the core of the peninsula with peaks over 2,100 m and glacially carved, broad river valleys radiating into surrounding lowlands (Franklin and Dyrness 1973, Henderson et al. 1989). High-elevation forest on the Olympic Peninsula is geographically isolated from similar forests in the Cascade Range to the east and has been protected from human development within Olympic National Park (ONP) (373,380 ha) and wilderness of Olympic National Forest (35,674 ha).

The climate of the Olympic Peninsula is moderated by the surrounding water bodies and is classified as mild maritime with high winter precipitation and summer drought. Micro-climatic conditions vary due to steep topography and variations in slope and aspect; orographic precipitation occurs where moisture moving off the Pacific Ocean is released on the west slope of the Olympic Mountains. On the east side of the Olympics the climate is relatively dry and cold at high elevations, with low winter soil temperatures, and average minimum January temperature of -2.8 ^{o}C. Summers are warm with an average maximum temperature of 18.9 ^{o}C in August and relatively high soil temperatures, especially along south-facing slopes of sparsely vegetated stands (Henderson et al. 1989) (temperature data extracted to 500 random points for ONP at 960-1900 m elevation; PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, accessed 23 Apr 2015). Annual precipitation ranges from ≥ 500 cm in west-side mountains, to 250 cm in east-side mountains where subalpine fir dominates, and to 41 cm in the northeastern lowlands (**Fig. 2.3c**). Adiabatic cooling causes about 60% of winter precipitation to fall as snow at high elevations. This creates a moisture reservoir that distributes water to lower elevations during the spring and summer. Spring snow-melt has been occurring increasingly early as a result of climate change (Mote 2003, Hamlet et al. 2005). Summer thunderstorms may occur in May through August (Estberg et al. 2008) with lightning activity that generally moves south to north across the mountains. The most heavy thunderstorm activity occurs during the driest months of late summer (Pickford et al. 1980) when fire risk is moderate to high. From October through February, mid-latitude cyclones produce 65-115 km/h winds that move off the Pacific Ocean from the southwest.

Climate cycles such as El Niño Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO) also affect climate conditions on the west coast of the Olympic Peninsula (Mantua et al. 1997, Diaz et al. 2001, Mass 2008). These cycles are associated with annual to decadal trends in monthly sea-surface temperature and pressure differentials. ENSO completes a cycle every 3-7 years with two alternating climate patterns: (1) El Niño periods that are warm and dry, and (2) La Niña periods that are cool and wet. Transition years are neutral but may be associated with increased storm occurrence. Similar to ENSO, the PDO index registers as monthly warm or cool phases of varying intensity but has a broad cyclic trend of 20 to 30 years that fluctuates between cool, negative phases with high snowpack, and warm, positive phases with low snowpack. When ENSO and PDO are both in a negative or positive phase, effects are compounded.

Climate change: an altered constraint

Warming temperatures, changing precipitation patterns, and more frequent extreme weather events are expected with climate change (Mote and Salathé 2010). This may affect vegetation directly via limiting environmental factors, and indirectly via competition with other species and altered disturbance regimes (Dale et al. 2001, McKenzie et al. 2004, Westerling et al. 2006, Littell et al. 2010, Ayres et al. 2014). Climate affects the dynamics of biotic disturbance through interactions with the agent (physiology and distribution), the host (stress, defense, and tolerance), and predators, competitors, or mutualists (Sturrock et al. 2011, Ayres et al. 2014). Fire extent and frequency are expected to increase, as are drought, winter storm severity (precipitation), and potential for frost events. Average winter snowpack (measured as snow water equivalent) has decreased by 60% in the Olympic Mountains over the period from 1950 to 2000 (Mote 2003), and dates of peak snow accumulation and 90% melt have been occurring increasingly early in the year (Hamlet et al. 2005). This initially may benefit tree growth at high elevations where trees are limited by growing season length but would reduce growth where trees are moisture limited (Ettl and Peterson 1995). Accurately predicting an outcome from interactions that include an altered climate and multiple processes is difficult (Gilman et al 2010); however, novel plant and animal communities are likely to result (Hobbs et al. 2009).

Vegetation: conifers of subalpine forest

Temperature and moisture gradients interact with species tolerances, competitive abilities, growth, and successional dynamics to create the distribution of conifer species in subalpine forest (Fonda and Bliss 1969, Franklin and Dyrness 1973, Henderson et al. 1989, Ettl and Peterson 1995, Peterson et al. 1997, Woodward 1998). For this research, high-elevation forests of the Olympic Mountains were separated by National Vegetation Classification (NVCS) macrogroup (Federal Geographic Data Committee, 2008): (1) Rocky Mountain Subalpine & High Montane Subalpine Forest (RMSF) occurs in the dry northeast and (2) Vancouverian Subalpine Forest (VSF) occurs in the wetter southwest **(Fig. 2.3a**). The NVCS was developed to create a consistent National vegetation classification for federal agencies. It is based on existing vegetation data gathered at local, regional, or national levels as plot data and from scientific literature. Macrogroups have a common set of growth forms and diagnostic plant taxa, and sharing similar geographic region, climate, and disturbance. Subalpine fir is the dominant tree species in RMSF where it co-occurs with lodgepole pine (*Pinus contorta* var. *latifolia*) on dry slopes, and with mountain hemlock (*Tsuga mertensiana*) on north-facing slopes.

Forest transitions into parkland and meadows at higher elevations where tree establishment is hampered by snow (Woodward et al. 1995, Rochefort and Peterson 1996). Whitebark pine (*Pinus albicaulis*) also occurs near timberline, and western white pine (*Pinus monticola*) occurs in forests below. At lower elevations subalpine fir is commonly associated with Pacific silver fir (*Abies amabilis*) and transitions to Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) and western hemlock (*Tsuga heterophylla*). Alaska yellow cedar (*Callitropsis nootkatensis*) is intermixed with subalpine fir along high-elevation riparian areas and avalanche chutes. Engelmann spruce (*Picea engelmannii*) very rarely occurs together with subalpine fir in moist areas.

In the Pacific Northwest, subalpine fir grows between 1200 m and 1800 m in elevation, and as low as 600 m along cold stream bottoms. At high elevation, subalpine fir is a dominant tree species and a pioneer because it can establish in mineral soil, litter, duff, or decaying wood, develop a root system under severe conditions, withstand snowpack, and shed snow with less branch breakage than other species (Alexander et al. 1984). Within the RMSF macrogroup, subalpine fir is self-perpetuating and shade tolerant, able to reestablish in the understory, and in VSF, it plays a seral role relative to mountain hemlock and Pacific silver fir and requires disturbance to re-establish (e.g., Franklin and Mitchell 1967).

Subalpine fir is an adaptable species and long-time component of western forests, surviving both the warm, dry, Medieval Optimum from 1000 to 1300, and the cold, wet period of the Little Ice Age, from 1300 to 1850 (Henderson et al., 1989). Olympic Peninsula subalpine fir constitutes only a small portion of the total species range within western North America, which extends from southeast Alaska, south across Canada into Washington, Oregon, Idaho, Montana, Wyoming, Colorado, Utah, and in scattered mountain ranges of Nevada and New Mexico (Uchytil 1991, Earle 2014; **Fig. 2.3b**). Plant hardiness zones (USDA 2012) increase in warmth across the range of subalpine fir from Zone 2a through Zone 8a with the species optimum in Zone 5 (Bannister and Neuner 2001). Subalpine fir on the Olympic Peninsula occurs at the warm end of the hardiness range (Zone 8a); the average minimum January temperature is − 6 °C at the highest elevation where subalpine fir grows (PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, accessed 23 Apr 2015). For reference, population growth of BWA may be reduced at this temperature but not eliminated; BWA occurs in hardiness zones above 4a, with a threshold of – 11 °C, average January minimum (Quiring et al. 2008).

Disturbance processes

Disturbance processes and their interactions influence vegetation patterns from the hierarchical level below, with their distributions constrained by physical geography. Processes that affect tree health on the Olympic Peninsula can be divided into abiotic, biotic, invasive, and complexes of disturbance processes (**Fig. 2.4**).

Abiotic disturbance in subalpine forest

Fire, avalanches, landslides, riparian activity, and occasionally strong mid-latitude cyclones (**Figures 2.4 a, b, and c**) cause high-magnitude and abrupt changes to vegetation within Olympic National Park. Fire behavior and severity depend on fuel conditions, topography, and weather. More area is burned during dry years (e.g., 2010), more frequent fires occur on the drier east side of the Olympic Peninsula, and spread is greater when dry winds blow from the east (foehn winds; Agee, 1993; **Fig. 2.5a**). Risk of ignition increases during late summer thunderstorms and 81% of area burned has been attributed to lightning ignition (Pickford and Ottmar 1980; **Fig. 2.5c)**. In the subalpine fir zone, fire has the potential to be severe, extensive, and stand-replacing, because of abundant fuels that allow fire to spread easily into the crown during dry conditions (Taylor and Fonda 1990, Huff 1995). Also, thin-barked trees (e.g., subalpine fir and lodgepole pine) lack fire resistance and thus succumb to heat. Because high-elevation fuels remain moist most of the year, the fire return interval is relatively long and averages from about 100 to 275 or more years depending on the local climate (Agee 1993, Edmonds et al. 2011a).

Fire has influenced subalpine vegetation composition and structure on the Olympic Peninsula by maintaining meadows, resetting successional pathways, and contributing to heterogeneous tree establishment conditions. The severity and size of fire determines how forests will reestablish (Agee and Smith 1984). If scattered subalpine fir trees do not burn because of discontinuous fuels, broken and rocky terrain, and a moist and cool environment, they provide a subalpine fir seed source (Uchytil 1991). However, if lodgepole pine is present, it usually grows faster than subalpine fir and a predominantly single-age stand may establish. Subalpine fir can regenerate beneath lodgepole pine and eventually replace the pine with a multi-age forest in the absence of fire. Where lodgepole is absent, harsh conditions can limit tree reestablishment to 30-50 years or longer. Subalpine fir may produce cones at age 20, but trees are most productive at 150-200 years (Uchytil 1991).

Fire suppression may not have impacted subalpine fir forest dynamics because fire return intervals are greater than 100 years for this forest type, which is longer than the period of fire suppression. However, suppression is likely to result in an increased proportion of older forest than would have been present without suppression (Agee, 1994b), and may increase the susceptibility of trees to insects and disease. Since the development of the ONP Fire Management Plan in 2005, management guidelines have allowed lightning-caused fires to burn within the Wildland Fire Use Unit (WFU) with limitations (Kailin and Nickey 2005). The WFU unit comprises 57% of park area and a large portion (22% of park area) is in the elevation range of subalpine fir. Fires are to be limited by management to 81 ha yr^{-1} with a maximum of 526 ha in a peak year. It is difficult to compare this fire area restriction to past area burned. Prehistoric fire was variable and linked to climate fluctuations. In 1701, a 404,700 ha fire extended along the north and east sides of the peninsula where Douglas-fir dominates today (Henderson et al. 1989). After this, the largest historic fires burned 12,141 ha (Dungeness Fire in 1890/91), 5,180 ha (Sol Duc Burn in 1907), and 13,355 ha (Forks Fire in 1951). However, between 1938 and 2001 only 2,189 ha burned inside ONP; 67% of total area burned was lightning caused and 60% of the fires were human caused.

Wind from temperate cyclonic storms is a dominant disturbance agent during winter (Mass 2008) and can cause widespread low- to localized high-magnitude damage. Storms approach the Oregon and Washington coastline from the southwest and move across the flat lowlands and into the mountains causing abrupt, patchy structural damage in forests across the Olympic Peninsula. At intervals of approximately 30 years, wind events cause extensive high-magnitude damage to trees (Agee 1994a; **Fig. 2.5b**). Wind storms are more common during neutral ENSO years and least common during El Niño years when the Pacific Jet stream splits at the latitude of Washington and flows toward Alaska and California (Mass 2008). When combined with heavy precipitation, such as occurs during La Niña years, windstorms can be especially damaging to trees as a result of decreased strength of saturated soils and snow or ice-burdened branches. Avalanches and landslides are likely to contribute to disturbance during cool wet La Niña years as well, compared to years with low precipitation and low snowpack.

Wind disturbance has greatest impact on the west side of the Olympic Peninsula along the coast and on south-facing slopes (**Fig. 2.5a**; Agee 1994a). However, cyclonic storms originating from the southwest also blow down trees on north-facing slopes and valleys in the northern part of the Olympic Peninsula (e.g. Heart of the Hills), indicating that wind movement is complex and often channeled by topography (Woodward et al. 2011). Wind disturbance interacts synergistically with insects and disease by breaking and blowing down trees that have been killed or are structurally weakened by root and stem disease (Edmonds et al. 2011b), and by providing substrate for bark beetle reproduction in downed and damaged wood.

Biotic disturbance in subalpine forest

Endemic tree pathogens and herbivores have co-evolved and coexist with vegetation on the Olympic Peninsula causing damage in small scattered patches. Generally only weak and stressed trees are attacked (Bleiker et al. 2005). On occasion, conditions that stress vegetation (e.g., drought or defoliation) can result in outbreaks of insects and disease (Gibson 2008). Bark beetles as a group have great potential to threaten conifers when conditions are conducive to bark beetle survival, host entry, reproduction, and spread (Breshears et al. 2005, Berg et al. 2006, Raffa et al. 2008). Generally, endemic bark beetles are present in low numbers inhabiting dead, dying, and downed trees. For reproductive success adult beetles target older, stressed trees with limited pitch production, but of large enough size to provide phloem and moisture for brood development (Harder 1993, Bleiker et al. 2003). Depending on the species of beetle, larvae or adults over-winter in the cambial layer beneath the bark, and their survival is temperature dependent. An over-abundance of older or stressed trees and consecutive warm winters may result in a bark beetle outbreak where bark beetle populations increase to a point that even healthy trees are affected. Severe windstorms can trigger such outbreaks because beetles reproduce in fresh downed wood and emerge after one to two years in high enough numbers to kill live trees (Furniss and Carolin 1977). Healthy tree defenses reduce beetle populations; however, beetles attacking en masse deplete the resources of vigorous trees.

On the Olympic Peninsula, the most prevalent and damaging bark beetles of subalpine fir are the western balsam bark beetle (*Dryocoetes confusus*), fir engraver (*Scolytus ventralis*), the fir root bark beetle (*Pseudohylesinus granulatus*), and potentially the silver fir beetle (*Pseudohylesinus sericeus*) (Furniss and Carolin 1977). The western balsam bark beetle primarily attacks high-elevation subalpine fir, whereas the fir engraver is more common in other western true firs. Beetle galleries etched in the cambium aid in field identification (Goheen and Willhite 2006) and are easiest to locate on dead trees with sloughing bark. Where entrance holes and boring dust are observed in intact bark, the bark may be cut away from the tree bole to reveal galleries. Bark beetles are responsible for mortality throughout the subalpine fir range (Gibson et al. 1997).

Extended warm, dry conditions stress trees, decrease tree resistance to insects and disease, and cause increases in insect populations because warm climate conditions favor population growth (Weed et al. 2015). In some locations bark beetle life cycles have been accelerated by warm temperatures and longer growing seasons (Logan et al. 2003, Bentz et al. 2010). Bark beetle outbreaks occurring across the western U.S. may be associated with drought conditions, and outbreaks are expected to subside when soil moisture increases or host trees are depleted. Alternatively, cool, wet conditions can also stress trees; late snow melt and short growing season limit tree growth, and fungal pathogens are more likely to spread in persistent, moist conditions (Sturrock et al. 2011).

Harsh conditions can reduce the lifespan of subalpine fir. Trees often succumb to Annosus root disease (*Heterobasidion occidentale* sp. nov.), bark beetles, or mechanical damage at less than 250 years of age (Uchytil 1991). Nevertheless, interactions with disturbance and continued establishment of seedlings beneath the canopy have allowed subalpine fir to maintain a presence in the cold, dry, high-elevation habitats for thousands of years.

Non-native invasive insects and pathogens: novel processes

Non-native invasive insects and pathogens alter vegetation composition by reducing or eliminating host species (Ellison et al. 2005). Balsam woolly adelgid (BWA) may be the greatest non-native insect threat to subalpine fir health on the Olympic Peninsula. It was introduced on nursery stock from Europe in the early 1900s, and over time it has spread and slowly eliminated up to 90% of true fir trees (*Abies* spp.) within some forested stands on the east and west coasts of North America (Witter and Ragenovich 1986, Mitchell and Buffam 2001). Aerial detection surveys (ADS) have tracked the spread of BWA across Oregon and Washington since the 1950s (**Fig. 2.6**) (Ciesla 2006). The insect is about 1 mm long (**Fig. 2.7**) and feeds like an aphid by inserting mouth parts into the tree cambium of the main stem or branches. The process of adelgid feeding alters tree cell growth, restricts transport of water and nutrients to needles, and causes needle drop and swollen branch nodes called “gout” (**Fig. 2.8**) (Balch 1952, Hain et al. 1991). Crown infestations stunt leader growth and result in dense collapsed tree tops (**Fig. 2.9**) (Ragenovich and Mitchell 2006).

White pine blister rust is a disease caused by a non-native pathogen, *Cronartium ribicola*, that has damaged trees on the Olympic Peninsula. It was introduced into western North America in Vancouver B.C. from Eurasia in 1910 on imported white pine seedlings (Maloy 1997, 2001). By the 1950s it was established throughout the west, killing 5-needle pines, and altering forest composition. Research into genetic and endophyte-induced resistance to disease is increasing future management options (Schoettle and Sniezko 2007, Ganley et al. 2008, Kinloch Jr. et al. 2008, King et al. 2010).

Native trees are vulnerable because they lack resistance to non-native insects and disease (Hayes and Ragenovich 2001). Non-native insects and disease typically lack predators, and are spread by many vectors including people, vehicles, firewood, Christmas trees, nursery stock, and wood products. Non-native insects and disease may become problematic for natural resource managers depending on thresholds of introduction, establishment, spread, damage to the host, and extensive host mortality (e.g., With 2002; **Table 2.1**). Predicting the ability of an insect or pathogen to respond to new conditions is complex. Typically, non-native species are contained by geophysical barriers or by early detection and interception. However, integrated pest management actions that attempt to contain insects and disease once they have been introduced, become established, and begin to spread are not always successful, and a few escaped insects and pathogens are able to create extensive damage to tree species with severe consequences to ecosystem structure and dynamics (Ellison et al. 2005).

Land management agencies monitor invasive insects and disease that may be transported to the Olympic Peninsula, including the Asian longhorned beetle (*Anoplophora glabripennis*), sudden oak death (*Phytophthora ramorum*), the emerald ash borer (*Agrilus planipennis*), and the gypsy moth (*Lymantria dispar*). Moreover, some endemic species act like invasive species and cause disease when conditions are altered by climate change or management practices (e.g., fire suppression, planting of trees off-site, species monocultures, and same-age cohorts). Annosus root disease became a management problem when cut tree stumps created an inoculation surface for air-borne spores, and below-ground roots provided a fungal pathway to other trees (Edmonds et al. 2011b). Warming temperatures and an extended growing season have allowed the endemic mountain pine beetle to double its reproductive rate in some areas and kill large numbers of drought-stressed pines in the Rocky Mountains and Canada (Bentz et al. 2010); populations are expected to spread to higher elevations as warmer winters become more common. Plant compositions are likely to change quickly with altered constraints and novel processes.

Interactions: disturbance complexes

The disease spiral conceptual model (Manion 1991) identifies multiple factors that may play a role in tree mortality. Senescence, climate change, soil conditions, or pollution may predispose a tree to disease by increasing susceptibility. Drought, frost, or defoliating insects may compromise physiological processes and further incite tree health decline. Death may result from these factors or from contributing agents such as root disease, beetles, or nematodes that opportunistically feed on weakened trees.

Although a single agent may be implicated in tree mortality, several agents and conditions may be synergistically responsible for the occurrence, extent, and severity of a tree mortality event. Agents that are consistently associated in a progression of tree decline may be grouped into a stress complex, i.e., combinations of biotic and abiotic stressors that compromise the vigor of trees (Ciesla 2006, McKenzie et al. 2009, Ayres et al. 2014). Examples include windthrow/drought/bark beetles (Furniss and Carolin 1977), fire/fungal pathogens/bark beetles (Gara et al. 1985), pollution/susceptibility to disease (e.g., Grulke et al. 1998), and fire suppression/species-composition-structure-change/insects (McCullough et al. 1998). Factors may be sequential and the elimination of one factor may significantly reduce overall damage.

With the introduction of non-native invasive species and climate change, new disturbance complexes may develop. Single invasive agents can be aggressive enough or extreme weather events severe enough to be implicated in a mortality event. However, in some cases a climate factor or new agent simply causes stress, followed by opportunistic, endemic agents that attack and kill the tree. Invasive species and climate change are now common initiators of tree stress and are integral components of novel disturbance complexes (Ayres et al. 2014) (**Fig. 2.10**).

The BWA is an initiator of novel disturbance on the Olympic Peninsula and may be considered a predisposing or inciting factor in a disturbance complex. The BWA creates broad-scale stress to subalpine fir, making trees more susceptible to moisture stress (Hollingsworth and Hain 1994) and endemic bark beetle attack (e.g., Bleiker et al. 2005). Surviving trees are subsequently subject to mechanical damage from wind events that topple trees killed by BWA and beetles. Tree stress, downed trees, and warmer temperatures may further contribute to an increase in bark beetle and BWA activity.

Focal-level pattern: investigation of extensive tree health decline and mortality

Investigation of change

Tree health specialists may begin an investigation of tree mortality by considering whether one or more species are unhealthy. If multiple species are unhealthy, the cause may be an abiotic agent, whereas if a single species is unhealthy, a biotic agent or species susceptibility may be the cause (**Fig. 2.11**). The next step is to assess symptoms, symptom location, and signs on early-affected trees (Goheen and Willhite 2006). However, the investigation may be complicated by the effects of novel constraints, multiple processes, and cross-scale interactions. A multi-scale perspective is needed for a holistic understanding of these processes.

A hierarchical framework provides a foundation for understanding landscape change and tree health at both broad and fine scales in space and time. Broad scale patterns (extent, patch size, location, onset, duration) are indicative of the relationship with constraints, as well as interactions among constraints and processes. Fine scale symptoms verify the presence of active processes. It is the spatial and temporal intersection of process symptoms with broad scale pattern that helps to determine the relative importance of interacting factors responsible for tree health decline.

Annual aerial detection surveys (ADS) conducted by the Washington Department of Natural Resources (WADNR) and the United States Forest Service (USFS) have detected defoliation and tree mortality of subalpine fir on the Olympic Peninsula that may be caused by the BWA. Aerial surveys have a multi-scale advantage because flying over extensive areas at high elevations with human observers for observation, provides an assessment of broad- and fine-scale pattern simultaneously. For example, large patches of forest health decline can be delineated and individual affected tree species can be identified (Ciesla 2006). These surveys provide insights into the agents causing the tree damage but the surveys also have some weaknesses. First, they rely on general tree-level symptoms and common local agents. Accuracy has been estimated at 70% (Johnson et al. 2006) or as low as 50% for some agents (Omdal et al. 2006). Secondly, only new tree damage is documented each year, making it difficult to track temporal patterns, or to quantify progressive decline. Balsam woolly adelgid may follow a slow temporal decline with an increase in area over time that may not be accurately documented in aerial surveys. Validation in the field is also important because subalpine fir may be relatively short-lived and susceptible to many endemic insects and disease, weather events, and climate change (Franklin and Mitchell 1967). Field visits are needed to verify consistent association with BWA.

The hierarchical nature of disturbance dynamics suggests that the following four steps, supported by GIS and statistical methods, are appropriate for systematically investigating forest health decline: (1) document spatial and temporal vegetation change patterns at broad- to meso-scales to identify and differentiate pattern attributes; (2) compare these patterns with the distribution and/or timing of constraints such as climate, landscape features, soil, and vegetation distributions, as well as processes such as abiotic events; (3) document fine scale patterns (e.g., signs and symptoms in field plots) within and outside patches that have changed to aid in the identification of underlying mechanisms; (4) apply independent methods, or test hypotheses based on experiments or data collected from other locations. For step 1, the scale must be large enough to observe spatial boundaries and temporal transitions and cycles. This can be accomplished with Landsat time series data (Kennedy et al. 2010; see Chapter 3), and dendrochronology (see Chapter 5). In steps 2 and 3, change may be detected for an area or time period by focusing on differences across transitions in space, time, and tree health. It is important to note signs and symptoms associated with unhealthy plots or newly declining trees that are not found in healthy plots or trees (see Chapter 4). In step 4, independent methods and data provide complementary information, identify relationships, and offer a means of validation (see Chapters 4 and 5).

On the Olympic Peninsula, punctuated short-term vegetation decline was associated with fire, riparian activity, and wind storm events, which caused indiscriminant damage to all tree species (Chapter 3). However, long-term decline of subalpine fir proceeded relatively slowly, increased in area over time, and was consistently associated with symptoms of the balsam woolly adelgid (Chapter 4).

Alaska yellow cedar decline in Southeast Alaska offers yet another example of a tree health decline pattern (Hennon et al. 2006). Alaska yellow cedar was the only species affected across the 200,000 ha of forest decline; however, health decline boundaries were constrained by topography (below 305 meters elevation), not by tree species distribution. The decline area increased very little after onset in the 1880s but continued to affect new trees within the area after this time. Early decline was consistently associated with damaged roots of shallowly rooted trees growing in saturated or rocky soils, and inconsistently associated with opportunistic insect and disease agents. Hennon et al. (1997, 2006) used a combination of broad-scale spatial and temporal observations of healthy and unhealthy tree distributions and fine-scale mechanisms related to tree physiology and morphology in and out of the decline areas during their research. They concluded that the decline patterns were the result of an interaction between warming climate conditions and tree susceptibility to spring freezing where roots are shallow and unprotected by snow. This landscape pattern differs from the increasing distribution pattern expected in response to an invasive insect or pathogen. Ecosystem dynamics are complex and patterns must be assessed with consideration for the unique boundaries and limitations of both constraints and processes.

ECOLOGICAL, SOCIAL, AND MANAGEMENT IMPLICATIONS

Foundation species loss in Olympic National Park

Foundation species are common and abundant species that "define the structure of a community by creating locally stable conditions for other species, and by modulating and stabilizing fundamental ecosystem processes" (Ellison et al. 2005). Invasive insects and pathogens alter vegetation composition by reducing or eliminating a specific host species. Loss of a foundation species will affect all associated biota that have come to depend on local environmental conditions created by that species.

Land managers on the Olympic Peninsula and other western regions in North America are confronted with the possible loss of subalpine fir due to infestations of the non-native BWA. These areas have already experienced a reduction in 5-needle pines including western white pine and whitebark pine as a result of the non-native disease white pine blister rust. Land managers must consider how the loss of subalpine fir will affect the subalpine ecosystem, which management activities would be effective, acceptable, or feasible, and where information gaps exist regarding specific processes and conditions supported by subalpine fir.

It is difficult to predict how the subalpine forest may change with the loss of subalpine fir. Subalpine fir is unique in high elevation environments in its ability to grow on wind-swept slopes and ridges with deep winter snowpack, rocky soils, and short, dry summers. Subalpine fir plays an important role in soil stabilization and wind-dispersed seeds easily germinate on fire-prepared seedbeds (Uchytil 1991). It is one of few tree species that readily, albeit episodically, is able to move into subalpine meadows, by encroaching from forested edges or creating tree islands that extend outward as trees germinate under the protection of overstory trees (Woodward et al. 1995, Rochefort and Peterson 1996). The encroachment of subalpine fir into meadows has been of concern to the National Park Service because meadows are inhabited by unique biota that are highly valued by park visitors (Rochefort and Peterson 1996). Mitchell and Buffam (2001) observed that BWA has eliminated subalpine fir from some pioneering habitats. The BWA may or may not interfere with meadow encroachment at higher elevations because BWA activity is reduced in colder environments (Balch 1952).

Below treeline subalpine fir can create a closed canopy, and provides browse and habitat across western North America for large and small wildlife, including deer, elk, black bear, snowshoe hare, flying squirrel, red squirrel, porcupine, pine marten, fisher, lynx, and several species of mice, voles, chipmunks, and shrews (Uchytil 1991). Numerous species of birds also nest and feed in subalpine fir forests, including woodpeckers, flycatchers, kinglets, nuthatches, juncos, thrushes, chickadees, crossbills, pine siskins, owls, and grouse. Subalpine fir seeds are an energy-rich food source for small mammals and birds, and grouse feed on fir needles. Subalpine fir forests create varied understory environments that provide multiple ecosystem functions for plants and animals.

Although subalpine fir may comprise 100% of the tree cover in some areas, these stands are often mixed with other conifers. Subalpine fir commonly occurs with mountain hemlock on shaded slopes, with lodgepole pine in dry environments, with Alaska yellow cedar in moist environments, and with Pacific silver fir, Douglas-fir and western hemlock at lower elevations. Associated conifers may grow into areas previously dominated by subalpine fir and may be able to maintain similar functions and provide cover for the wildlife and understory species that are not exclusively dependent on subalpine fir. However, associated conifers may not thrive at high elevations where they are exposed to wind, ice, or snow damage, and may be susceptible to pathogens.

Although many subalpine fir trees may be killed by BWA, there is the chance that more resistant genotypes will survive and eventually establish, perhaps with the aid of common garden breeding programs. BWA-infested fir trees have been observed to wall off adelgid feeding sites with production of wound periderm (Balch 1952) and also to increase production of secondary metabolites that inhibit metamorphosis in other insect species (Puritch and Nijholt 1974). Assuming that resistant forms of subalpine fir exist, survive, and regenerate, increasingly resistant cohorts may develop over time. Subalpine fir may have an adaptive advantage over BWA as well, because it reproduces sexually whereas BWA reproduces asexually. Offspring of BWA are genetically identical (with the exception of mutations) and therefore the population is unable to respond to improved host tree defenses co-evolutionarily with survival of more genetically fit individuals. Although tree mortality has been observed to slow after initially high mortality (Mitchell and Buffam 2001), it was unclear if the remaining live trees were actually resistant; as many as 50% of infested trees appeared to recover from BWA while other previously unaffected trees died several years later.

In the Appalachian Mountains where Fraser fir trees were killed by BWA several decades earlier than fir trees on the west coast, researchers have seen two patterns. The southern-most stands on the edge of the Fraser fir range suffered the greatest damage, and this may be related to greater moisture stress in the south (Hollingsworth and Hain 1994). Secondly, Fraser fir trees have been regenerating successfully at higher elevations (Dale et al. 1991, McManamay et al. 2011, Stehn et al. 2013). Because the Olympic Peninsula is located at the warmer end of the subalpine fir distribution, climate change increases the likelihood that subalpine fir will be confined (with BWA) to local high-elevation or cool-temperature refugia on the Olympic Peninsula. The long-term response of subalpine fir species to BWA is still uncertain.

Adaptive management approach and collaboration

An adaptive management (AM) approach to problem-solving is recommended for situations with a high amount of scientific or social uncertainty (Larson et al. 2013). The adaptive management process, facilitated by a hierarchical framework, offers a range of possible methods that may be used to inform and improve management in an iterative process (Westley 2002, Westgate et al. 2013, Williams and Brown 2013). Learning, monitoring, and using sound experimental designs are principle components of the AM approach. A collaborative team of managers, scientists, and sometimes public stakeholders and policy makers can respond to complex ecological problems with a multi-disciplinary understanding of biology, ecology, climatology, silvicultural prescriptions, public values, and land management policy.

The adaptive management approach is being applied to climate change adaptation on federally protected lands (Joyce et al. 2009, NPS 2010, Halofsky et al. 2011, Peterson et al. 2011, Millar et al. 2014). Land owners across jurisdictions are collaborating with scientists to learn about local climate projections, risks, and develop scale appropriate management solutions (Gregory et al. 2006). The same approach could be applied to address exotic species invasions. Another collaborative group, the National Firewood Task Force (NFTF; http://www.dontmovefirewood.org/) was established in 2009 by individuals from several agencies (National Association of State Foresters, National Park Service, National Plant Board, USDA Animal and Plant Health Inspection Service, and U.S. Forest Service) who perceived the need to address a forest health threat from the transport of invasive pests on firewood where regulations were weak. They created a three part strategy that included outreach, voluntary action, and regulatory action. Voluntary actions have served as a developmental phase for regulations, bridging the time gap before new regulations could be enacted, and offering a means for private industry to participate in the formulation of wood treatment standards, certification, and labeling guidelines. Suggested guidelines for use of local firewood in federal, state, and private campgrounds have also been developed and voluntarily posted at campgrounds. Regulation of interstate and intrastate movement of firewood is a long-term goal that must be addressed at federal and state levels.

Several resources (reports, videos, workbooks) offer guidelines for collaborative action, outlining strategies to address specific management issues that range from invasive pests (NFTF 2010) to climate change adaptation (NPS 2010, Peterson et al. 2011), fuel management, and habitat restoration (Shindler et al., 2011a, 2011b). Details of individual management programs will vary by location, objectives, and resource sensitivities, but guidelines apply to management adaptation efforts in general. The decision regarding the level and timing of public stakeholder involvement in management planning also depends on objectives; public involvement satisfies the objective of broad-scale education and builds a constituency support base.

Long-term solutions to novel ecosystem change lie largely outside the jurisdiction of protected area management, and in the management of processes and constraints that produce greenhouse gas like CO_2 and allow plant and animal species to move across boundaries and into protected land (Evans and Oszaco 2007, Roy et al. 2014). Strategies to address broad-scale ecosystem change often center on education, revision of legislation and policy, and the need for functional working groups and dynamic individuals who are capable of providing vision and leadership (**Fig. 2.12**).

CONCLUSION

The hierarchical framework that I have proposed in this chapter may be used to improve understanding of ecosystem dynamics and management options. The hierarchical perspective encourages simultaneous broad and fine scale investigation of emergent patterns and trends associated with broad-scale constraints and fine-scale mechanisms. Observing interactions among patterns, constraints, and processes at multiple scales, makes it possible to understand the relative importance of multiple factors contributing to landscape change.

Land managers may use the hierarchical framework to anticipate interactive effects of change in constraints and processes and apply principles of adaptive management to achieve short- and long-term ecological goals for protected areas. The framework supports an approach to detect and understand landscape pattern and causal relationships; explore management options that target or replace altered processes or constraints; design research and monitoring programs that further investigate or document key processes and relationships at appropriate scales; identify variables for analysis or as input for predictive models that explore management outcomes; and facilitate information exchange and adaptive management at local and regional scales. The structure provided by the framework can streamline communication and guide management responses to landscape change.

REFERENCES

Agee, J. 1994a. Fire and weather disturbances in terrestrial ecosystems of the eastern Cascades. USDA Forest Service General Technical Report PNW-GTR-32:52.

Agee, J. 1994b. An analysis of catastrophic forest disturbance on the Olympic Peninsula. Unpublished report prepared for Rayonier Inc.

Agee, J. K. 1993. Fire Ecology of Pacific Northwest Forests. Island Press, Washington, D.C.

Agee, J., and L. Smith. 1984. Subalpine tree reestablishment after fire in the Olympic Mountains, Washington. Ecology 65:810–819.

Alexander, R. R., R. C. Shearer, and W. D. Shepperd. 1984. Silvical characteristics of subalpine fir. General Technical Report 115. Rocky Mountain Forest and Range Experimental Station. Fort Collins, Colorado.

Allen, T. F. H., and T. B. Starr. 1982. Hierarchy: Perspectives for Ecological Diversity. University of Chicago Press, Chicago, Illinois.

Antonova, N., C. Thompson, D. Wallin, R. Bryson, S. Clary, A. Hayes, I. Oehler, B. Pickard, M. Ponce-McDermott, and N. Schaller. 2011. Identifying the Causes of Landscape Change in the North Coast and Cascades Network Parks: Final Report for Task Agreement J8W07110003 to PNW CESU Agreement H8W07110001.

Ayres, M. P., J. A. Hicke, B. K. Kerns, D. McKenzie, J. S. Littell, L. E. Band, C. H. Luce, A. S. Weed, and C. L. Raymond. 2014. Disturbance regimes and stressors. Pages 55–92 *in* D. L. Peterson, J. M. Vose, and T. Patel-Weynand, editors. Climate change and United States Forests. Springer Science + Business Media, Dordrecht.

Balch, R. E. 1952. Studies of the Balsam Woolly Aphid, *Adelges piceae* (Ratz.) and Its Effects on Balsam Fir, *Abies balsamea* (L.) Mill. Ottawa, Canada.

Bannister, P., and G. Neuner. 2001. Frost resistance and the distribution of conifers. Pages 3–22 *in* F. J. Bigras and S. J. Colombo, editors. Conifer Cold Hardiness. Kluwer Academic Publishers, Dordrecht.

Bentz, B. J., J. Regniere, M. Hansen, J. L. Hayes, and J. A. Hicke. 2010. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. Bioscience 60:602–613.

Berg, E. E., J. David Henry, C. L. Fastie, A. D. De Volder, and S. M. Matsuoka. 2006. Spruce beetle outbreaks on the Kenai Peninsula, Alaska, and Kluane National Park and Reserve, Yukon Territory: Relationship to summer temperatures and regional differences in disturbance regimes. Forest Ecology and Management 227:219–232.

Bleiker, K. P., B. S. Lindgren, and L. E. Maclauchlan. 2003. Characteristics of subalpine fir susceptible to attack by western balsam bark beetle (Coleoptera: Scolytidae). Canadian Journal of Forest Research 33:1538–1543.

Bleiker, K. P., B. S. Lindgren, and L. E. Maclauchlan. 2005. Resistance of fast- and slow-growing subalpine fir to pheromone-induced attack by western balsam bark beetle (Coleoptera: Scolytinae). Agricultural and Forest Entomology 7:237–244.

Breshears, D. D., N. S. Cobb, P. M. Rich, K. P. Price, C. D. Allen, R. G. Balice, W. H. Romme, J. H. Kastens, M. L. Floyd, J. Belnap, J. J. Anderson, O. B. Myers, and C. W. Meyer. 2005.

Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Sciences of the United States of America 102:15144–8.

Cairns, D. M., C. W. Lafon, J. D. Waldron, M. Tchakerian, R. N. Coulson, K. D. Klepzig, A. G. Birt, and W. Xi. 2008. Simulating the reciprocal interaction of forest landscape structure and southern pine beetle herbivory using LANDIS. Landscape Ecology 23:403–415.

Ciesla, W. M. 2006. Aerial signatures of forest insect and disease damage in the western United States. Forest Health Technology Enterprise Team. FHTET-01-06. USDA Forest Service, Fort Collins, Colorado.

Crawford, R., C. B. Chappell, C. C. Thompson, and F. J. Rocchio. 2009. Vegetation classification of Mount Rainier, North Cascades, and Olympic National Parks plant association descriptions and identification keys. National Parks. National Park Service, Fort Collins, Colorado.

Dale, V. H., R. H. Gardner, and D. L. DeAngelis. 1991. Elevation-mediated effects of balsam woolly adelgid on southern Appalachian spruce-fir forests. Canadian Journal of Forest Research 21:1639–1648.

Dale, V. H., L. A. Joyce, S. Mcnulty, R. P. Neilson, M. P. Ayres, M. D. Flannigan, P. J. Hanson, L. C. Irland, A. E. Lugo, C. J. Peterson, D. Simberloff, F. J. Swanson, B. J. Stocks, and B. M. Wotton. 2001. Climate change and forest disturbances. BioScience 51:723.

Diaz, H. F., M. P. Hoerling, and J. K. Eischeid. 2001. ENSO variability, teleconnections and climate change. International Journal of Climatology 21:1845–1862.

Earle, C. J. 2014. Conifer webpage. http://www.conifers.org/.

Edmonds, R. L., J. K. Agee, and R. I. Gara. 2011a. Fire ecology and fire regimes. Pages 79–112 Forest Health and Protection. Second Edition. Waveland Press, Inc., Long Grove, Illinois.

Edmonds, R. L., J. K. Agee, and R. I. Gara. 2011b. Root diseases. Pages 295–332 Forest Health and Protection. Second Edition. Waveland Press, Inc., Long Grove, Illinois.

Ellison, A. M., M. S. Bank, B. D. Clinton, E. A. Colburn, K. Elliott, C. R. Ford, D. R. Foster, B. D. Kloeppel, J. D. Knoepp, G. M. Lovett, J. Mohan, D. A. Orwig, N. L. Rodenhouse, W. V Sobczak, K. A. Stinson, J. K. Stone, C. M. Swan, J. Thompson, B. Von Holle, and J. R. Webster. 2005. Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Frontiers in Ecology and the Environment 3:479–486.

Estberg, G. N., S. Norris, and K. M. Hutten. 2008. Year-in-advance prediction of lightning activity for the Pacific Northwest. Northwest Science 82:151–157.

Ettl, G. J., and D. L. Peterson. 1995. Growth response of subalpine fir (*Abies lasiocarpa*) to climate in the Olympic Mountains, Washington, USA. Global Change Biology 1:231–241.

Evans, H., and T. Oszaco, editors. 2007. Alien Invasive Species and International Trade. Forest Research Institute, Warsaw.

Fancy, S. G., J. E. Gross, and S. L. Carter. 2009. Monitoring the condition of natural resources in US national parks. Environmental Monitoring Assessment 151:161–174.

Federal Geographic Data Committee. 2008. National Vegetation Classification Standard (Version 2.0). U.S. Geological Survey, Reston, VA.

Fonda, R. W., and L. C. Bliss. 1969. Forest vegetation of the montane and subalpine zones, Olympic Mountains, Washington. Ecological Society of America 39:271–301.

Franklin, J. J. F., and C. Dyrness. 1973. Natural Vegetation of Oregon and Washington. Oregon State University Press, Corvallis, OR.

Franklin, J., and R. Mitchell. 1967. Successional status of subalpine fir in the Cascade Range. U.S. Forest Service Research Paper PNW-46:16.

Furniss, R. L., and V. M. Carolin. 1977. Western Forest Insects. U.S. Government Printing Office, Washington, D.C.

Ganley, R. J., R. A. Sniezko, and G. Newcombe. 2008. Endophyte-mediated resistance against white pine blister rust in *Pinus monticola*. Forest Ecology and Management 255:2751–2760.

Gara, R. I., W. R. Littke, J. K. Agee, D. R. Geiszler, J. D. Stuart, and C. H. Driver. 1985. Influence of fires, fungi and mountain pine beetles on development of a lodgepole pine forest in South-Central Oregon. Pages 153–162 *in* D. M. Baumgartner, R. G. Krebill, J. T. Arnott, and G. F. Weetman, editors. Lodgepole Pine: The Species and Its Management Symposium Proceedings. Washington State University, Pullman, WA.

Gavin, D., and L. Brubaker. 1999. A 6000-year soil pollen record of subalpine meadow vegetation in the Olympic Mountains, Washington, USA. Journal of Ecology 87:106–122.

Gibson, K. 2008. Bark beetle conditions, Northern Region, 2007. USDA Forest Service, Northern Region, Forest Health Protection, Missoula Field Office, Missoula, MT.

Gibson, K., S. Kegley, and B. Oakes. 1997. Western balsam bark beetle activity and flight periodicity in the northern region. Report 97-3. Cooperative Forestry and Forest Health Protection, Northern Region, Missoula, MT and Coeur d'Alene, ID.

Goheen, E. M., and E. A. Willhite. 2006. Field Guide to the Common Diseases and Insect Pests of Oregon and Washington Conifers. R6-NR-FID-PR-01-06. USDA Forest Service, Pacific Northwest Region, Portland, Oregon.

Gregory, R., D. Ohlson, and J. Arvai. 2006. Deconstructing adaptive management: criteria for applications to environmental management. Ecological Applications 16:2411–25.

Grulke, N. E., C. P. Andersen, M. E. Fenn, and P. R. Miller. 1998. Ozone exposure and nitrogen deposition lowers root biomass of ponderosa pine in the San Bernardino Mountains, California. Environmental Pollution 103:63–73.

Hain, F. P., R. G. Hollingsworth, F. H. Arthur, F. Sanchez, and R. K. Ross. 1991. Adelgid host interactions with special reference to the balsam woolly adelgid in North America. *in* Y. N. Baranchikov, W. J. Mattson, F. P. Hain, and T. L. Payne, editors. Forest Insect Guilds: Patterns of Interaction with Host Trees. USDA Forest Service General Technical Report NE-153: 271-287.

Halofsky, J., D. Peterson, and K. O’Halloran. 2011. Adapting to Climate Change at Olympic National Forest and Olympic National Park. USDA Forest Service, Pacific Northwest Research Station. General Technical Report PNW-GTR-844.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2005. Effects of temperature and precipitation variability on snowpack trends in the Western United States. Journal of Climate 18:4545–4561.

Harder, L. 1993. Impact of the western balsam bark beetle, *Dyrocoetes confusus*, at the Sicaous Creek Research Site, and the potential for semiochemical-based management in alternative silviculture systems. M.S. Thesis. Simon Fraser University, Burnaby, British Columbia, Canada.

Hayes, J. L., and I. Ragenovich. 2001. Non-native invasive forest insects of Eastern Oregon and Washington. Northwest Science 75:77–84.

Van Hemert, C., J. M. Pearce, and C. M. Handel. 2014. Wildlife health in a rapidly changing North: focus on avian disease. Frontiers in Ecology and the Environment 12:548–556.

Henderson, J. A., D. H. Peter, R. D. Lesher, and D. C. Shaw. 1989. Forested Plant Associations of the Olympic National Forest. USDA Forest Service, Pacific Northwest Region. R6-ECOL Technical Paper 001-88., Portland, Oregon.

Hennon, P., D. Wittwer, A. Johnson, P. Schaberg, G. Hawley, C. Beier, S. Sink, and G. Juday. 2006. Climate warming, reduced snow, and freezing injury could explain the demise of yellow-cedar in Southeast Alaska, USA. World Resource Review 18:427–450.

Hobbs, R. J., E. Higgs, and J. a. Harris. 2009. Novel ecosystems: implications for conservation and restoration. Trends in Ecology and Evolution 24:599–605.

Holling, C. S. 2001. Understanding the complexity of economic, ecological, and social systems. Ecosystems 4:390–405.

Holling, C. S., and L. Gunderson. 2002. Resilience and adaptive cycles. Pages 25–62 *in* L. Gunderson and C. S. Holling, editors. Panarchy. Island press, Washington, D.C.

Hollingsworth, R. G., and F. P. Hain. 1994. Effect of drought stress and infestation by the balsam woolly adelgid (Homoptera: Adelgidae) on abnormal wood production in Fraser fir. Canadian Journal of Forest Research 24:2295–2297.

Huff, M. H. 1995. Forest age structure and development following wildfires in the western Olympic Mountains, Washington. Ecological Applications 5:471–483.

IPCC. 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. (T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex, and P. M. Midgley, Eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC. 2014. Climate Change 2014: Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. (C. B. Field, V. R. Barros, D. J. Dokken, K. J. Mach, M. D. Mastrandrea, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, L. A. N, M. S, P. R. Mastrandrea, and L. L. White, Eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Johnson, E. W., J. Ross, J. L. Harris, and F. J. Cross. 2006. USDA Forest Service Rocky Mountain Region forest health aerial survey accuracy assessment 2005 report. USDA Forest Service Technical Report R2-06-08, Lakewood, CO.

Joyce, L. a, G. M. Blate, S. G. McNulty, C. I. Millar, S. Moser, R. P. Neilson, and D. L. Peterson. 2009. Managing for multiple resources under climate change: national forests. Environmental Management 44:1022–32.

Kailin, J., and L. Nickey. 2005. Olympic National Park Fire Management Plan. USDI, National Park Service, Port Angeles, WA.

Kennedy, R. E., W. B. Cohen, A. A. Kirschbaum, and E. Haureiter. 2007. Protocol for Landsat-based monitoring of landscape dynamics of North Coast and Cascades Network Parks: 2-G1. U.S. Geological Survey Techniques and Methods, Reston, VA.

Kennedy, R. E., Z. Yang, and W. B. Cohen. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 1. LandTrendr — temporal segmentation algorithms. Remote Sensing of Environment 114:2897–2910.

King, J. N., A. David, D. Noshad, and J. Smith. 2010. A review of genetic approaches to the management of blister rust in white pines. Forest Pathology 40:292–313.

Kinloch Jr., B. B., D. A. Davis, and D. Burton. 2008. Resistance and virulence interactions between two white pine species and blister rust in a 30-year field trial. Tree Genetics & Genomes 4:65–74.

Landres, P., P. Morgan, and F. Swanson. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications 9:1179–1188.

Larson, A. J., R. T. Belote, M. A. Williamson, and G. H. Aplet. 2013. Making monitoring count: project design for active adaptive management. Journal of Forestry 111:348–356.

Levin, S. 1992. The problem of pattern and scale in ecology: the Robert H. MacArthur award lecture. Ecology 73:1943–1967.

Littell, J., E. Oneil, D. McKenzie, and J. Hicke. 2010. Forest ecosystems, disturbance, and climatic change in Washington State, USA. Climatic Change:129–158.

Livingston, R. L., J. E. Dewey, D. P. Beckman, and L. E. Stipe. 2000. Distribution of the balsam woolly adelgid in Idaho. Western Journal of Applied Forestry 15:227–231.

Logan, J. A., J. Régnière, and J. A. Powell. 2003. Assessing the impacts of global warming on forest pest dynamics. Frontiers in Ecology and the Environment 1:130–137.

Maloy, O. C. 1997. White pine blister rust control in North America: a case history. Annual Review of Phytopathology 35:87–109.

Maloy, O. C. 2001. White pine blister rust. Online: Plant Health Progress doi:10.1094/PHP-2001-0924-01-HM.

Manion, P. D. 1991. Tree Disease Concepts. Prentice Hall, Englewood Cliffs, NJ.

Van Mantgem, P. J., and N. L. Stephenson. 2007. Apparent climatically induced increase of tree mortality rates in a temperate forest. Ecology Letters 10:909–16.

Mantua, N. J., S. R. Hare, Y. Zhang, J. M. Wallace, and R. C. Francis. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin of the American Meteorological Society 78:1069–1079.

Mass, C. 2008. The weather of the Pacific Northwest. University of Washington Press, Seattle, WA.

McCullough, D. G., R. a Werner, and D. Neumann. 1998. Fire and insects in northern and boreal forest ecosystems of North America. Annual Review of Entomology 43:107–27.

McIntire, E. J. B., and A. Fajardo. 2009. Beyond description: the active and effective way to infer processes from spatial patterns. Ecology 90:46–56.

McKenzie, D., Z. Gedalof, D. L. Peterson, and P. Mote. 2004. Climatic change, wildfire, and conservation. Conservation Biology 18:890–902.

McKenzie, D., D. L. Peterson, and J. Littell. 2009. Global warming and stress complexes in forests of western North America. Pages 319–337 *in* A. Bytnerowicz, M. J. Arbaugh, A. R. Riebau, and C. Anderson, editors. Wildland Fires and Air Pollution. Elsevier, Amsterdam/London.

McManamay, R. H., L. M. Resler, J. B. Campbell, and R. A. McManamay. 2011. Assessing the impacts of balsam woolly adelgid (*Adelges piceae* Ratz.) and anthropogenic disturbance on the stand structure and mortality of Fraser fir [*Abies fraseri* (Pursh) Poir.] in the Black Mountains, North Carolina. Castanea 76:1–19.

Millar, C. I., C. W. Swanston, and D. L. Peterson. 2014. Adapting to climate change. Pages 183–222 *in* D. L. Peterson, J. M. Vose, and T. Patel-Weynand, editors. Climate Change and United States Forests. Springer Science + Business Media, Dordrecht Heidelberg New York London.

Mitchell, R. G., and P. E. Buffam. 2001. Patterns of long-term balsam woolly adelgid infestations and effects in Oregon and Washington. Western Journal of Applied Forestry 16:121–126.

Mote, P. W. 2003. Trends in snow water equivalent in the Pacific Northwest and their climatic causes. Geophysical Research Letters 30:1–4.

Mote, P. W., and E. P. Salathé. 2010. Future climate in the Pacific Northwest. Climatic Change 102:29–50.

NFTF. 2010. National Firewood Task Force Recommendations. Online: http://www.ct.gov/deep/lib/deep/forestry/nationalfirewoodtaskforcerecommendations.pdf:21.

NPS. 2010. National Park Service climate change response strategy. National Park Service climate change response program. Fort Collins, Colorado.

O'Neill, R. V, A. R. Johnson, and A. W. King. 1989. A hierarchical framework for the analysis of scale. Landscape Ecology 3:193–205.

Omdal, D., M. Kallas-Ricklefs, and J. Moore. 2006. Where has all the root rot gone? Ground-truthing bear and root disease damage in western Washington (a poster). Washington State Department of Natural Resources. http://fhm.fs.fed.us/posters/posters06/bear_root_rot.pdf.

Peterson, D., C. Millar, and L. Joyce. 2011. Responding to climate change in national forests: a guidebook for developing adaptation options. USDA Forest Service, Pacific Northwest Research Station. General Technical Report PNW-GTR-855, Portland, OR.

Peterson, D., E. Schreiner, and N. Buckingham. 1997. Gradients, vegetation and climate: spatial and temporal dynamics in the Olympic Mountains, USA. Global Ecology and Biogeography Letters 6:7–17.

Pickett, S. 1989. Space-for-time substitution as an alternative to long-term studies. Long-term studies in ecology.

Pickford, S. G., G. R. Fahnestock, and R. Ottmar. 1980. Weather, fuel, and lightning fires in Olympic National Park. Northwest Science 54:92–105.

Puritch, G. S., and W. W. Nijholt. 1974. Occurrence of juvabione-related compounds in grand fir and Pacific silver fir infested by balsam woolly adelgid. Canadian Journal of Botany 52:585–587.

Quiring, D., D. Ostaff, L. Hartling, D. Lavigne, K. Moore, and I. DeMerchant. 2008. Temperature and plant hardiness zone influence distribution of balsam woolly adelgid damage in Atlantic Canada. The Forestry Chronicle 84:558–562.

Rabenold, K. N., P. T. Fauth, B. W. Goodner, J. A. Sadowski, and P. G. Parker. 1998. Response of avian communities to disturbance by an exotic insect in spruce-fir forests of the southern Appalachians. Conservation Biology 12:177–189.

Raffa, K., B. Aukema, B. J. Bentz, A. L. Carroll, J. A. Hicke, M. G. Turner, and W. H. Romme. 2008. Cross-scale drivers of natural disturbances prone to anthropogenic amplification: the dynamics of bark beetle eruptions. BioScience 58:501–517.

Ragenovich, I. R., and R. G. Mitchell. 2006. Balsam woolly adelgid. Forest Insect & Disease Leaflet 1182. USDA Forest Service.

Rochefort, R., and D. Peterson. 1996. Temporal and spatial distribution of trees in subalpine meadows of Mount Rainier National Park, Washington, USA. Arctic and Alpine Research 28:52–59.

Roy, B. A., H. M. Alexander, J. Davidson, F. T. Campbell, J. J. Burdon, R. Sniezko, and C. Brasier. 2014. Increasing forest loss worldwide from invasive pests requires new trade regulations. Frontiers in Ecology and the Environment 12:457–465.

Schoettle, A. W., and R. A. Sniezko. 2007. Proactive intervention to sustain high-elevation pine ecosystems threatened by white pine blister rust. Journal of Forest Research 12:327–336.

Seastedt, T. R., R. J. Hobbs, and K. N. Suding. 2008. Management of novel ecosystems: are novel approaches required? Frontiers in Ecology and the Environment 6:547–553.

Shindler, B., R. Gordon, S. McCaffrey, and E. Toman. 2011. Collaborating for healthy forests and communities: A guide for building partnerships among diverse interests. Corvallis, OR, USA.

Simon, H. A. 1974. The organization of complex systems. Pages 3–27 *in* H. H. Pattee, editor. Hierarchy Theory: The Challenge of Complex Systems. Braziller, New York, NY.

Smith, G. F., and N. S. Nicholas. 1998. Patterns of overstory composition in the fir and fir-spruce forests of the Great Smoky Mountains after balsam woolly adelgid infestation. American Midland Naturalist 139:340–352.

Stehn, S. E., M. A. Jenkins, C. R. Webster, and S. Jose. 2013. Regeneration responses to exogenous disturbance gradients in southern Appalachian Picea-Abies forests. Forest Ecology and Management 289:98–105.

Sturrock, R. N., S. J. Frankel, a. V. Brown, P. E. Hennon, J. T. Kliejunas, K. J. Lewis, J. J. Worrall, and a. J. Woods. 2011. Climate change and forest diseases. Plant Pathology 60:133–149.

Swanson, F., J. Jones, D. Wallin, and J. H. Cissel. 1994. Natural variability—implications for ecosystem management. Pages 80–95 *in* M. J. and P. Bourgeron, editor. Ecosystem Management: Principles and Applications, Volume II. Eastside Forest Ecosystem Health Assessment. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Taylor, K., and R. Fonda. 1990. Woody fuel structure and fire in subalpine fir forests, Olympic National Park, Washington. Canadian Journal of Forest Research 20:193–199.

Uchytil, R. J. 1991. *Abies lasiocarpa*. In: Fire Effects Information System, [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer), Available: http://www.fsfed.us/database/feis/ [2015, June 8].

USDA. 2012. Plant hardiness zone map. http://planthardiness.ars.usda.gov.

Vogelmann, J. E., B. Tolk, and Z. Zhu. 2009. Monitoring forest changes in the southwestern United States using multitemporal Landsat data. Remote Sensing of Environment 113:1739–1748.

Weed, A. S., B. J. Bentz, M. P. Ayres, and T. P. Holmes. 2015. Geographically variable response of *Dendroctonus ponderosae* to winter warming in the western United States. Landscape Ecology 30:1075–1093.

West, J. M., S. H. Julius, P. Kareiva, C. Enquist, J. J. Lawler, B. Petersen, A. E. Johnson, and M. R. Shaw. 2009. U.S. natural resources and climate change: concepts and approaches for management adaptation. Environmental Management 44:1001–21.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science 313:940–943.

Westgate, M. J., G. E. Likens, and D. B. Lindenmayer. 2013. Adpative management of biological systems: a review. Biological Conservation 158:1–40.

Westley, F. 2002. The devil in the dynamics: adaptive management on the front lines. Pages 333–360 *in* L. H. Gunderson and C. S. Holling, editors. Panarchy. Island Press, Washington, D.C.

White, P. S., and S. T. A. Pickett. 1985. Natural disturbance and patch dynamics: an introduction. Pages 3–13 *in* S. T. A. Pickett and P. S. White, editors. Natural Disturbance and Patch Dynamics. Academic Press, Inc., San Diego, CA.

Wiens, J. 1989. Spatial scaling in ecology. Functional Ecology 3:385–397.

Williams, B., and E. Brown. 2013. Adaptive management: from more talk to real action. Environmental Management 53:465–479.

With, K. A. 2002. The landscape ecology of invasive spread. Conservation Biology 16:1192–1203.

Witter, J. A., and I. R. Ragenovich. 1986. Regeneration of Fraser fir at Mt. Mitchell, North Carolina, after depredations by the balsam woolly adelgid. Forest Science 32:585–594.

Woodward, A. 1998. Relationships among environmental variables and distribution of tree species at high elevation in the Olympic Mountains. Northwest Science 72:10–22.

Woodward, A., K. M. Hutten, J. R. Boetsch, S. A. Acker, R. M. Rochefort, M. M. Bivin, and L. L. Kurth. 2009. Forest vegetation monitoring protocol for national parks in the North Coast and Cascades Network. Page 228 U.S. Geological Survey Techniques and Methods 2-A8.

Woodward, A., E. Schreiner, and D. Silsbee. 1995. Climate, geography, and tree establishment in subalpine meadows of the Olympic Mountains, Washington, USA. Arctic and Alpine Research 27:217–225.

Woodward, A., C. Torgersen, and J. Chenoweth. 2011. Predicting spread of invasive exotic plants into de-watered reservoirs following dam removal on the Elwha River, Olympic National Park, Washington. USGS Open-File Report 2011-1048.

Wu, J. 1999. Hierarchy and scaling: extrapolating information along a scaling ladder. Canadian Journal of Remote Sensing 25:367–380.

Wu, J., and O. L. Loucks. 1995. From balance of nature to hierarchical patch dynamics: a paradigm shift in ecology. The Quarterly Review of Biology 70:439–466.

Table 2.1. Invasive species progression: thresholds, processes, and management tactics.

Threshold	Mechanism or process	Natural control or management tactic
Introduction	Human-mediated transport Animal hitching	Geophysical barrier, dispersal limitation, host absence; strict import/export policies, early detection, human interception
Establishment	Survival, growth, reproduction	Unfavorable environmental conditions, host defenses, predators; management to eradicate or decrease populations
Spread	Dispersal via wings, wind, water, or animals	Unfavorable environmental conditions, host absence, low host density, limited dispersal capability; integrated pest management (IPM), silvicultural methods, monitoring
Damage to host	Alteration to host nutrient supply, growth, reproduction, defense, and susceptibility to 2^o agents	Host tolerance, environmental conditions; breeding for genetic resistance, monitoring
Regime shift	Extensive mortality, reproductive damage, or depleted resilience/reserve in host	Perpetual restorative maintenance, genetics research, planting resistant host species, adaptive management, monitoring

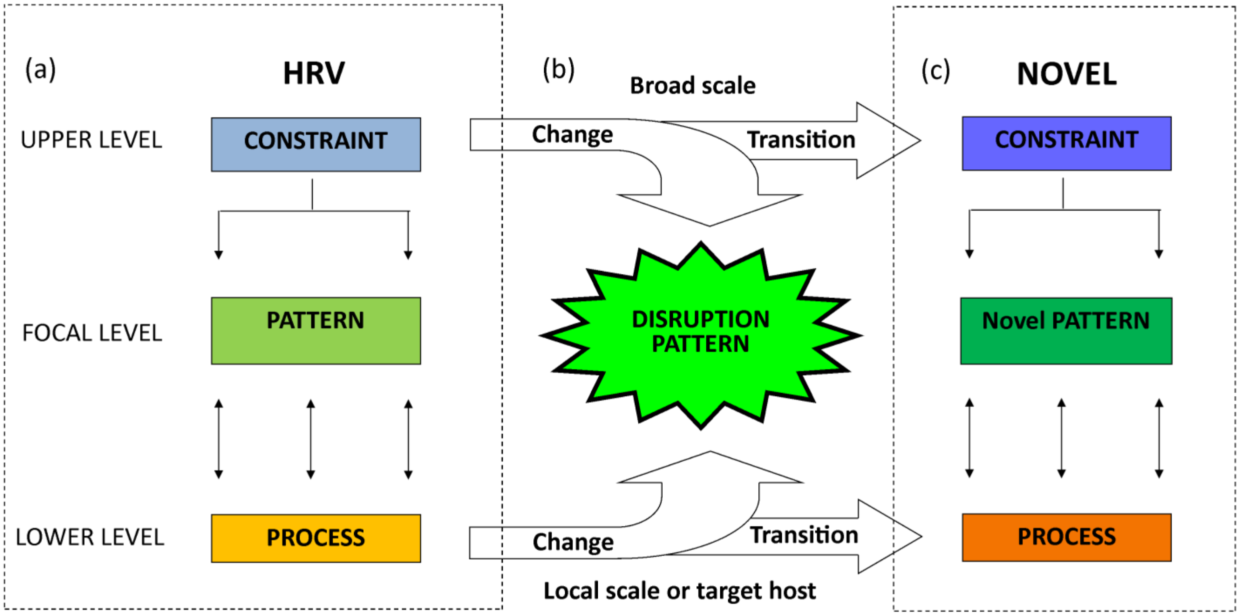


Figure 2.1. A simplified hierarchical patch dynamics model (a) represents the relationship among constraints, patterns, and processes that interact within the historical range of variability (HRV). Altered constraints and processes can initiate top-down or bottom-up system change that may be detected by an early disruption pattern at the focal level (b), and ultimately results in a transition from HRV to a novel state (c).

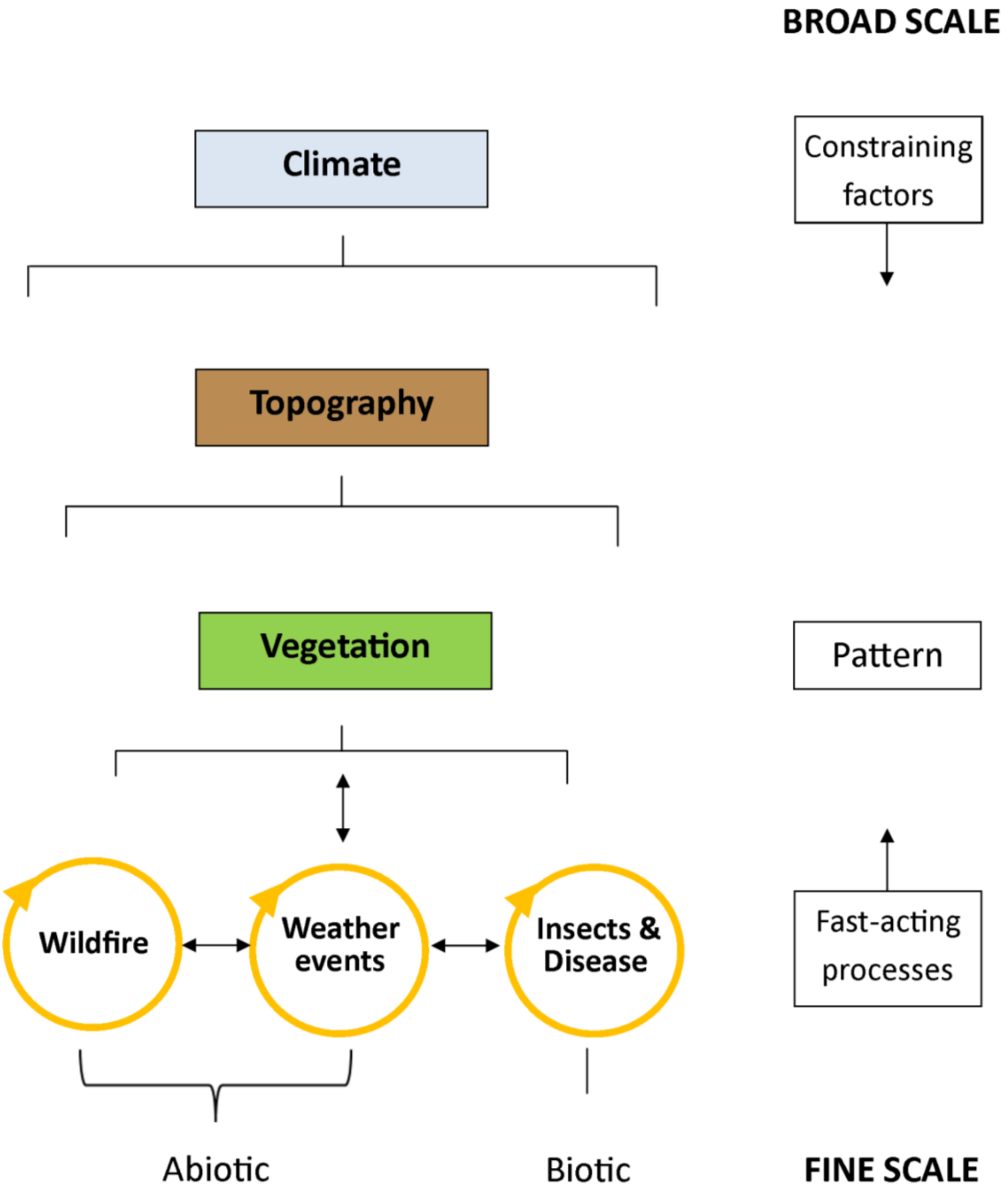


Figure 2.2. Hierarchical patch dynamics model for the Olympic Peninsula. Climate and topography constrain and control patterns in vegetation, the selected focal level of interest. Abiotic and biotic disturbances are fast-acting processes that interact (black arrows) and reoccur (orange arrows) to influence vegetation pattern. Vegetation also constrains disturbance, determining the type that is likely to occur, the extent, severity, seasonality, and return rate.

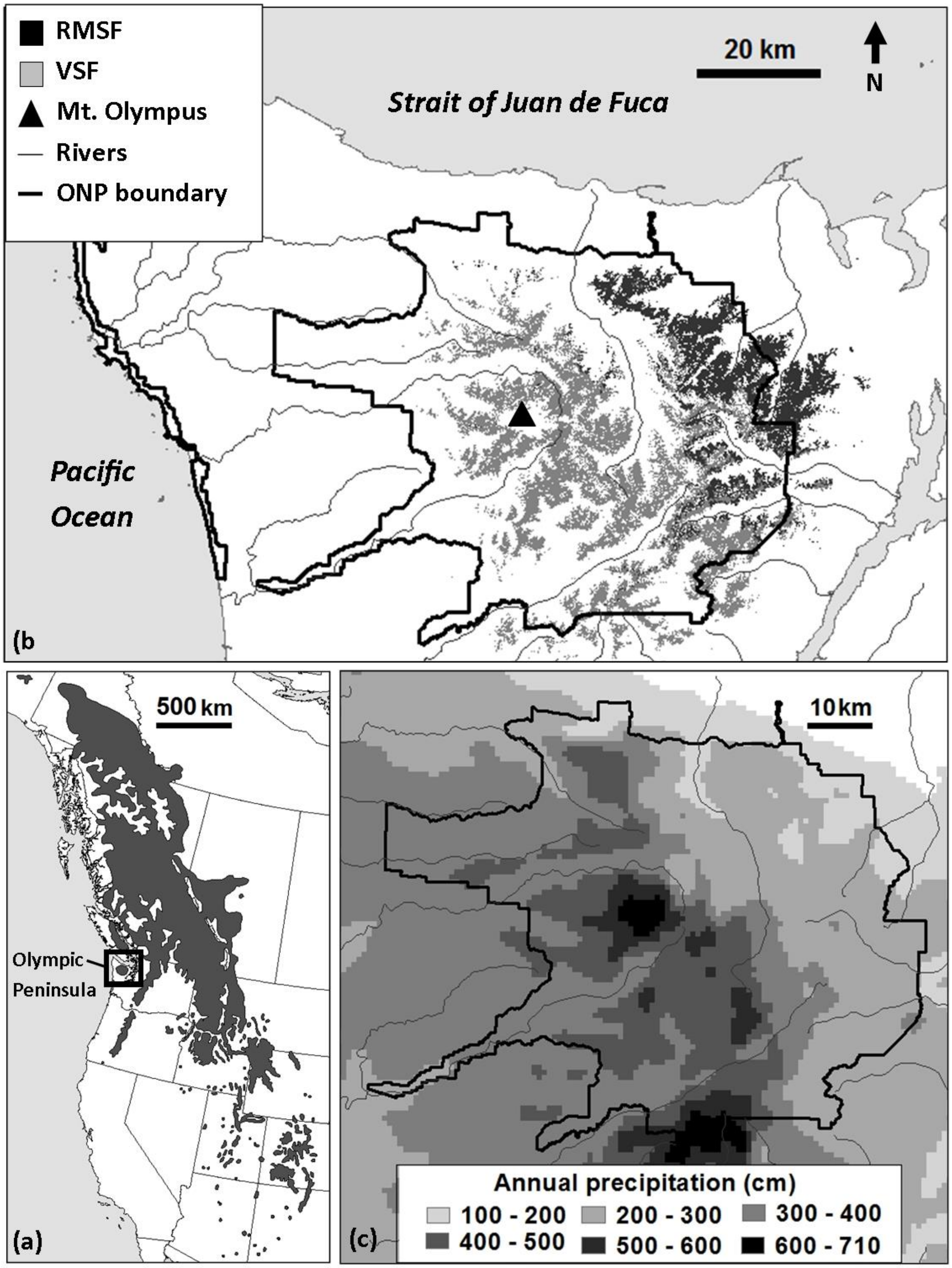


Figure 2.3. (a) Range of subalpine fir in North America. (b) Distribution of subalpine fir forest macro groups on the Olympic Peninsula according to the National Vegetation Classification Standard: RMSF = Rocky Mountain Subalpine & High Montane Conifer Forest, VSF = Vancouverian Subalpine Forest. (c) Precipitation in Olympic National Park.

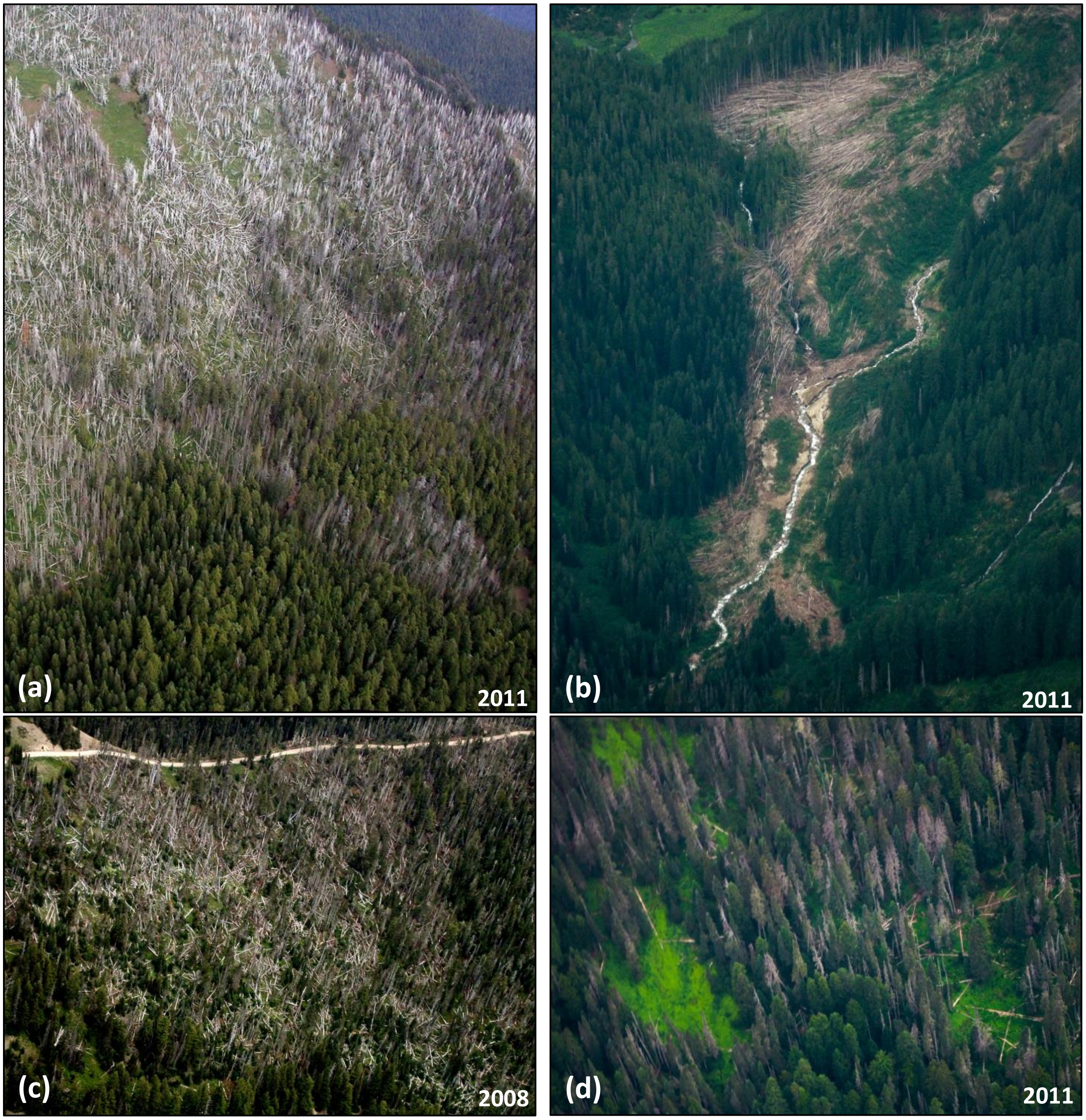


Figure 2.4. Examples of forest disturbance occurring within Olympic National Park include (a) fire (the Griff Fire in 2003), (b) avalanches, (c) insect and wind disturbance complexes (balsam woolly adelgid and western balsam bark beetle, below Obstruction Road, Hurricane Ridge), (d) insects and disease (bark beetles in subalpine fir).

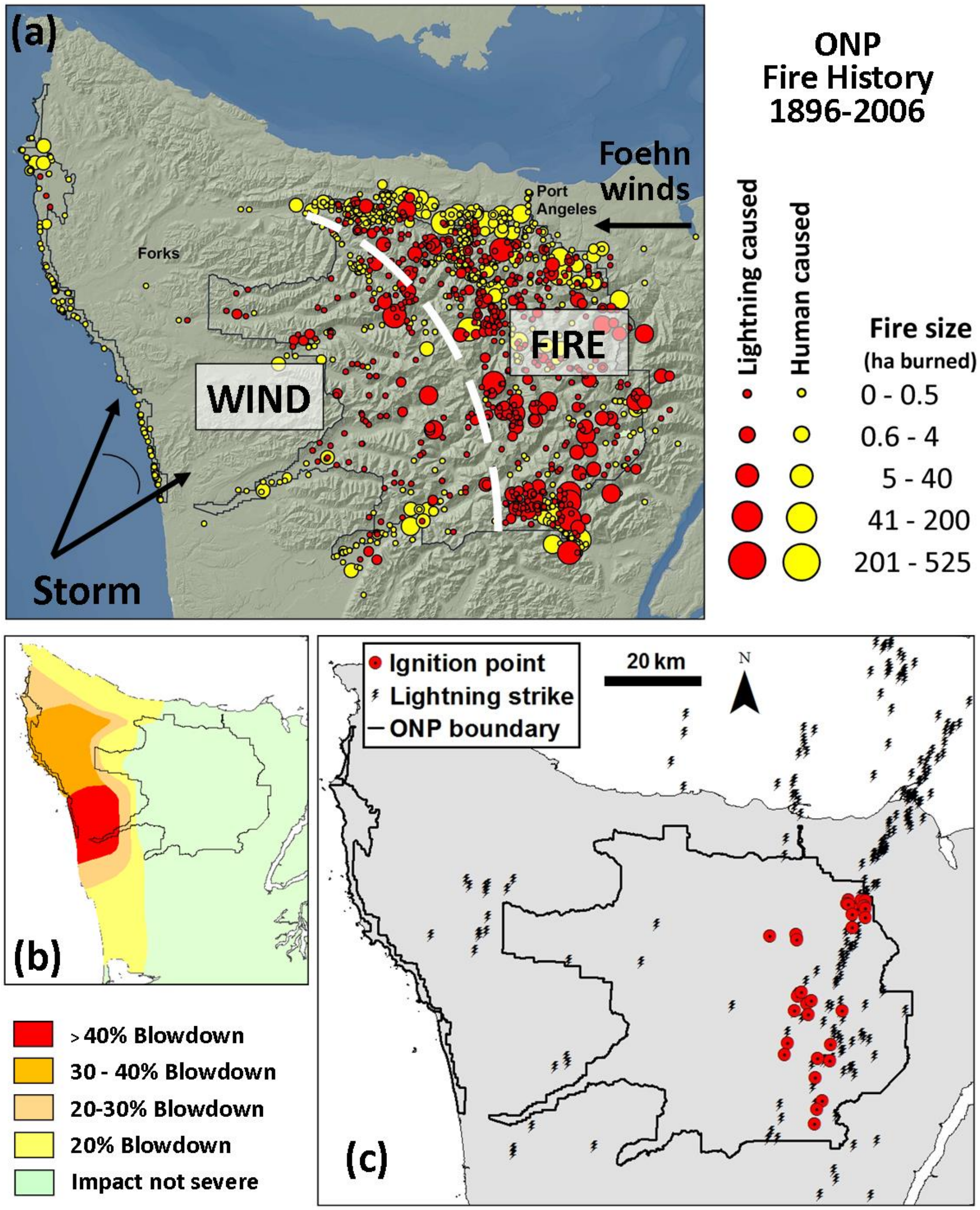


Figure 2.5. (a) Distribution of wind and fire disturbance on the Olympic Peninsula. Wind damage is more severe on the west side and wildfires occur more frequently on the dry east side (white dashed line divides approximate area of influence). Warm foehn winds from the east can fuel fires in E-W running valleys. Fire size classes are represented by circles (fire map by Roger Hoffman, ONP). (b) A cyclonic storm arriving from the SW called "The 1921 Blow" caused extreme wind damage along the west coast (adapted from a graphic by Olympic Natural Resources Center (ONRC)). (c) Late summer thunderstorms such as in August 17, 2008, ignite fires along a commonly N-S path of lightning strikes (fire start and lightning strike locations were provided by the Wildland Fire Management Office of the Olympic National Park (ONP); lightning strike location data was provided to ONP by the Bureau of Land Management (BLM)).

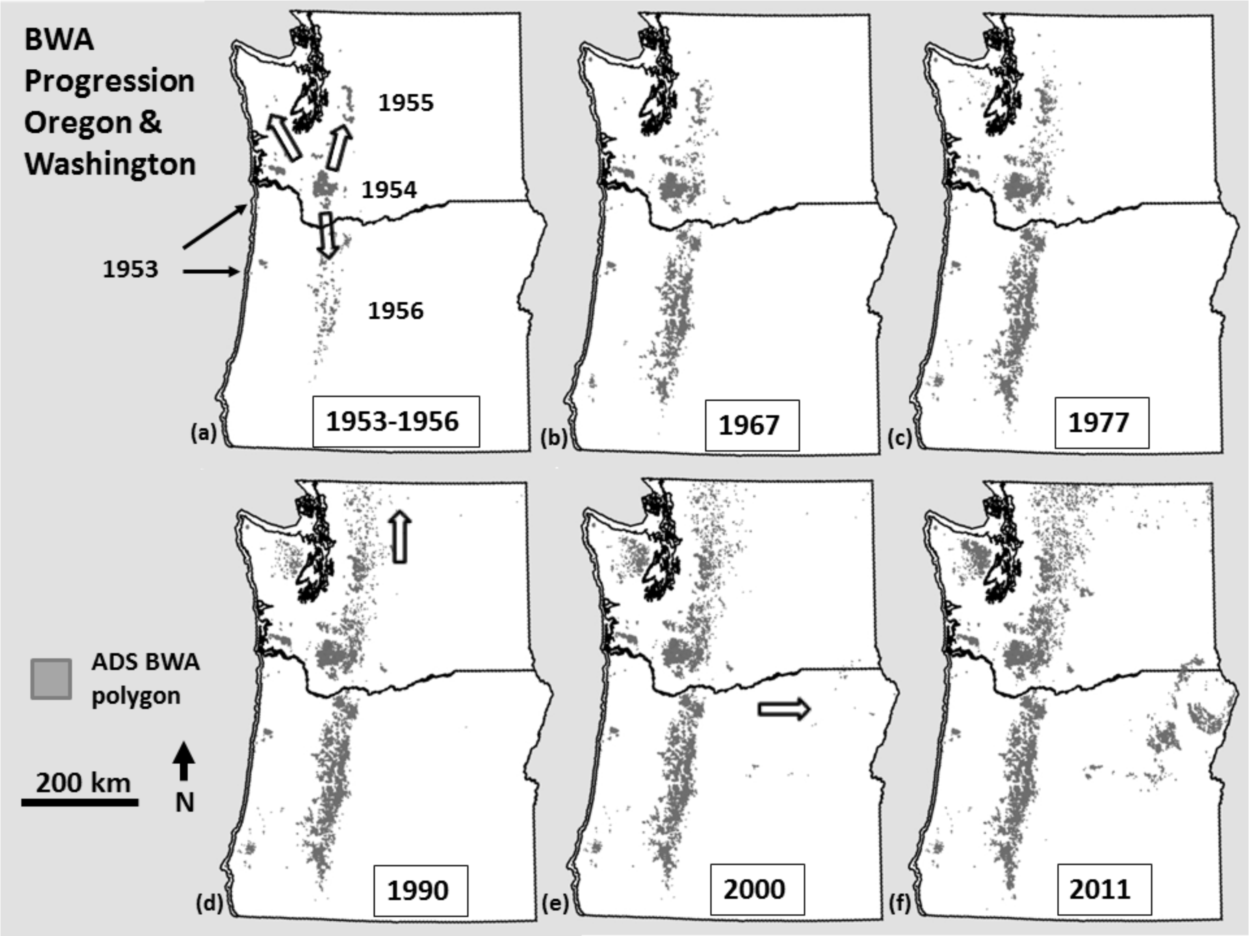


Figure 2.6. Spatio-temporal progression of balsam woolly adelgid (BWA) activity across Oregon and Washington documented by cooperative DNR and USFS aerial detection surveys (ADS), 1953 to 2011. Black arrows show where BWA was first sighted in Oregon (1953) along the coast; white block arrows indicate dominant new direction of movement, and BWA continues spread within infested areas as well.

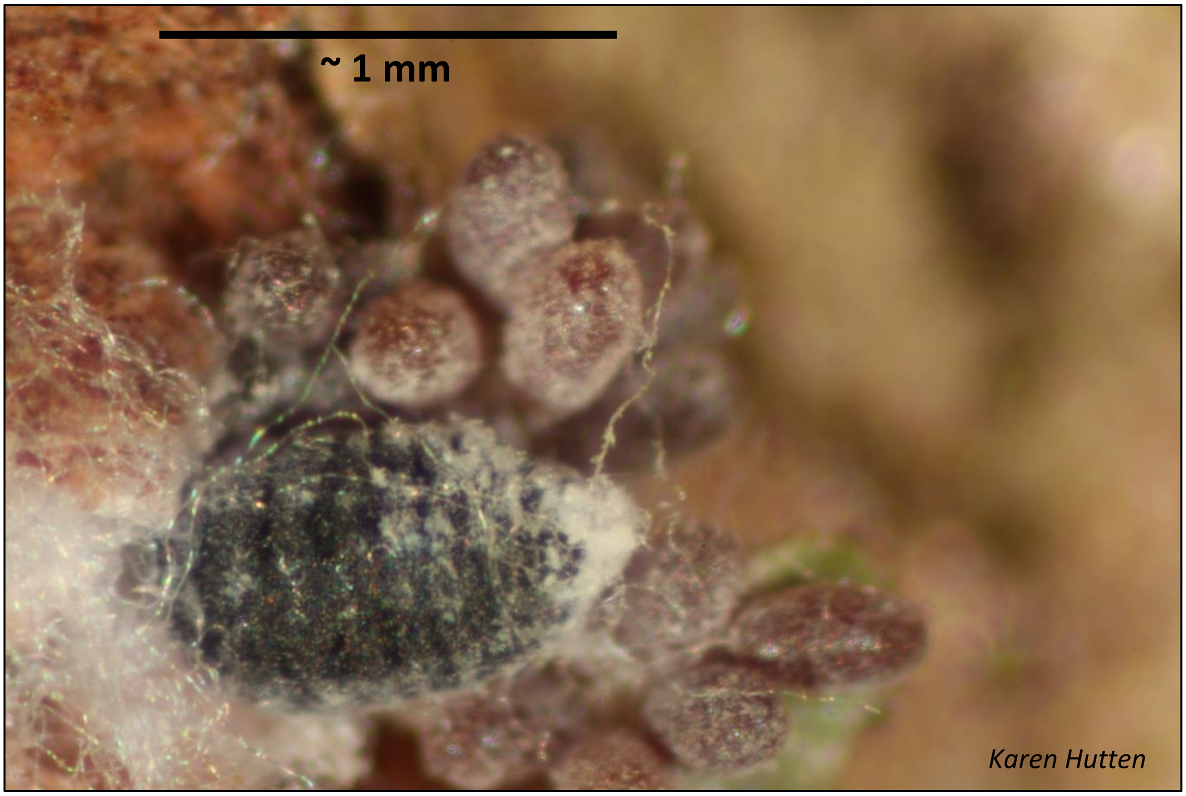


Figure 2.7. Adult female balsam woolly adelgid with eggs collected from the bole of a subalpine fir tree within a sample plot, August 2010.

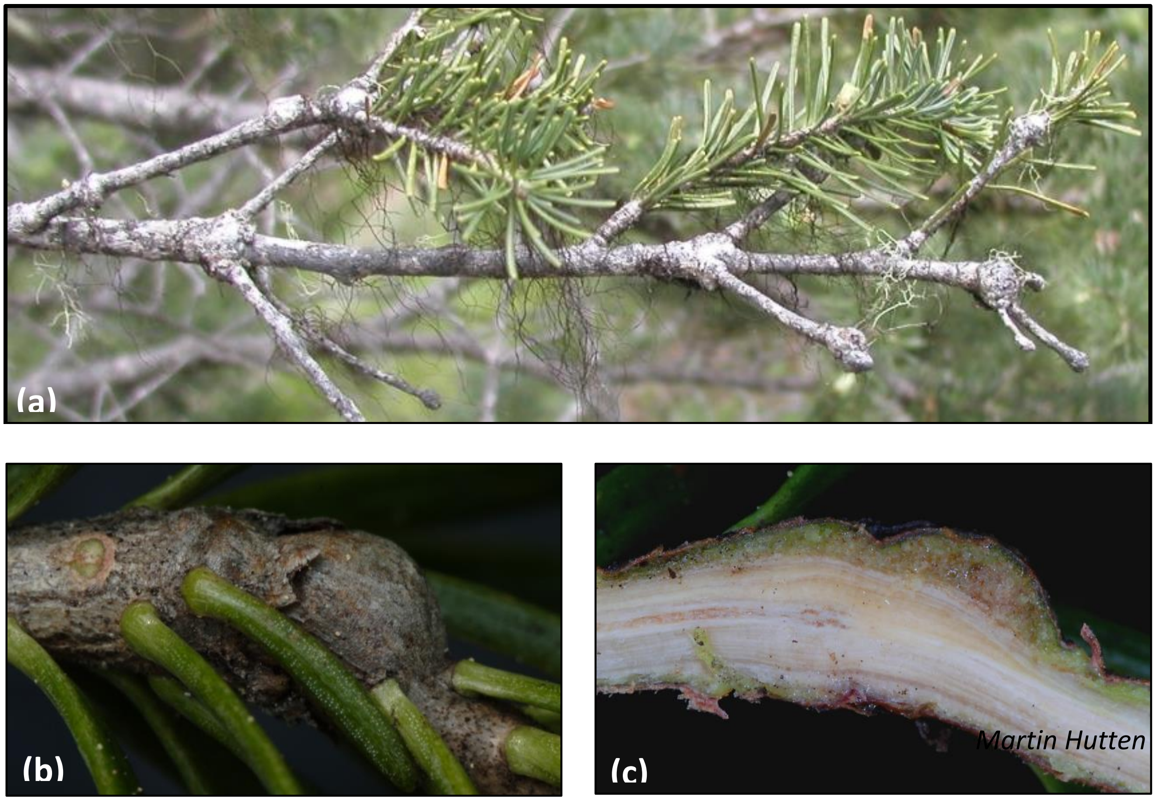


Figure 2.8. BWA-symptomatic branch ‘gouting’ on subalpine fir (a), close up (b), and cross-section (c). Note the dense parenchyma cells in the branch cambium produced by adelgid feeding (c).

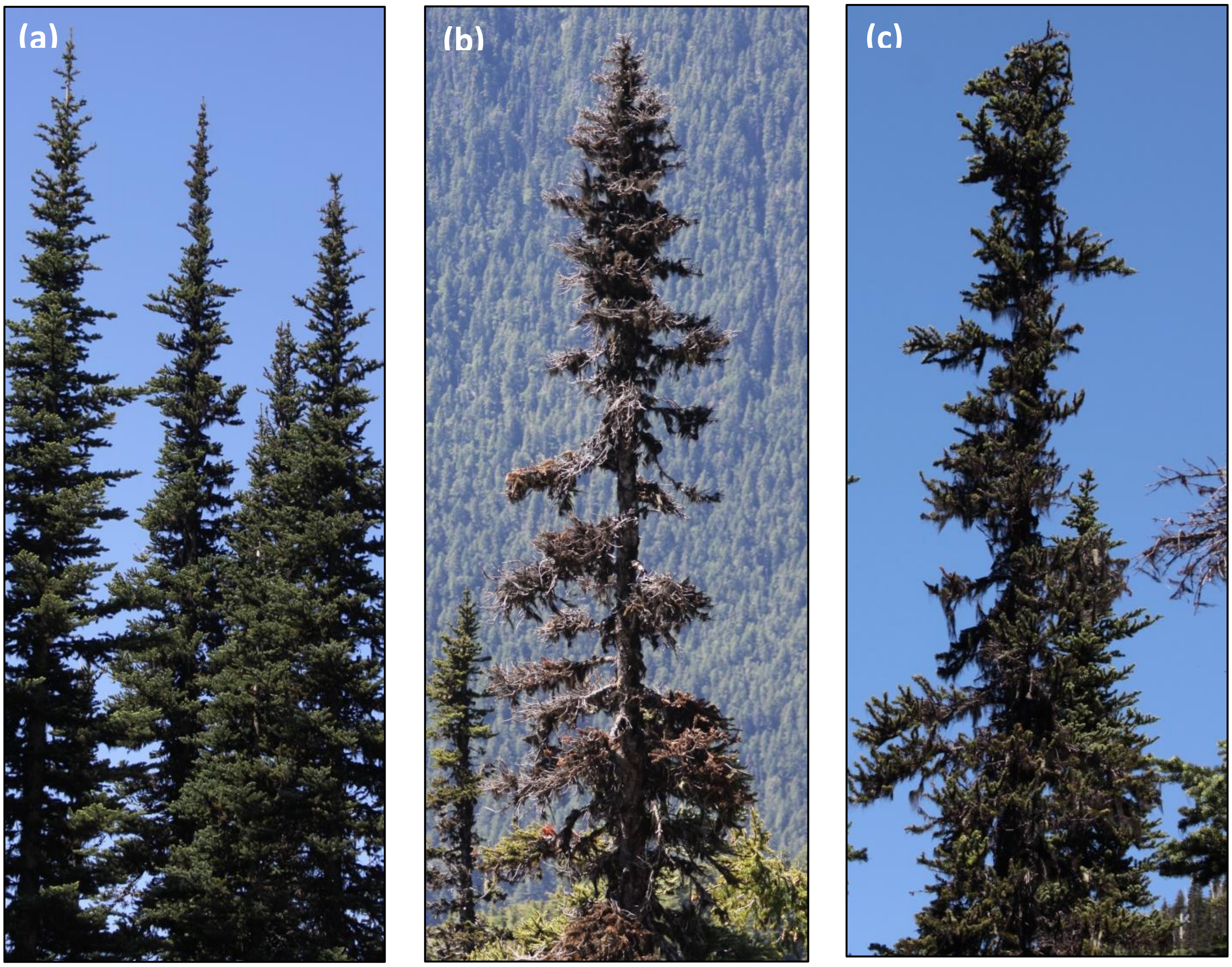


Figure 2.9. Healthy subalpine fir crowns (a), and ‘collapsed’ crowns stunted in growth by balsam woolly adelgid feeding (b, c).

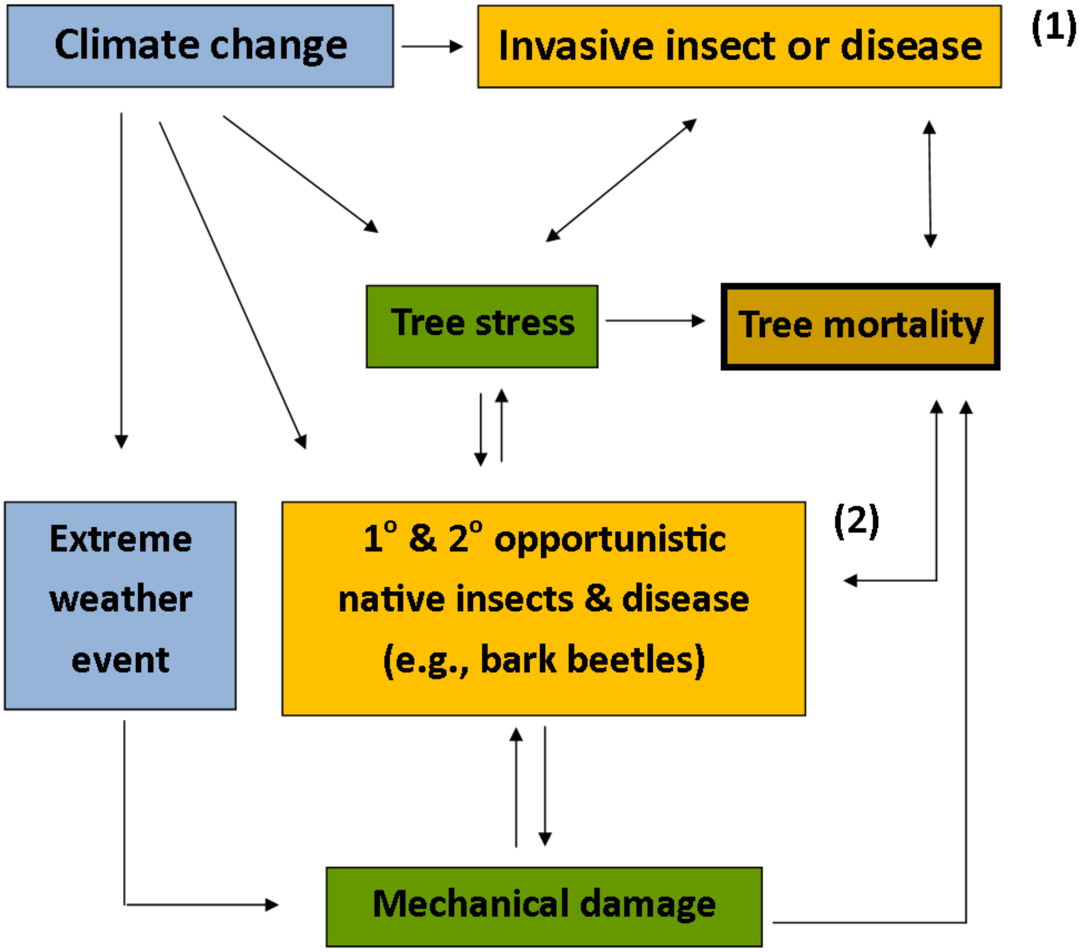


Figure 2.10. Conceptual model of a stress complex triggered by climate change and invasive insects or disease. In the Olympic National Park subalpine fir zone, the invasive insect, balsam woolly adelgid has caused stress to fir trees (1), and bark beetles are opportunistic native insects that have contributed to tree mortality (2). Climate change may simultaneously stress trees and favor insect population growth. Multiple factors and feedback loops create synergistic effects.

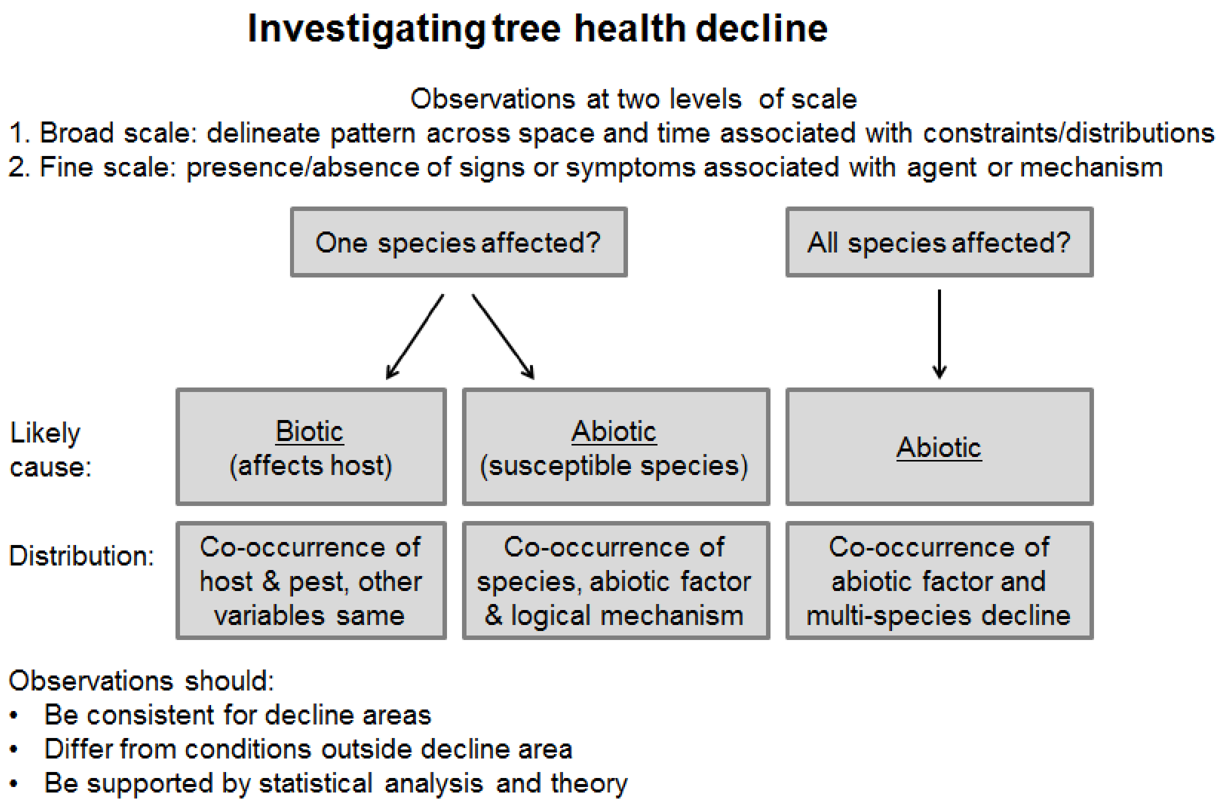


Figure 2.11. Guidelines for investigating tree health decline.

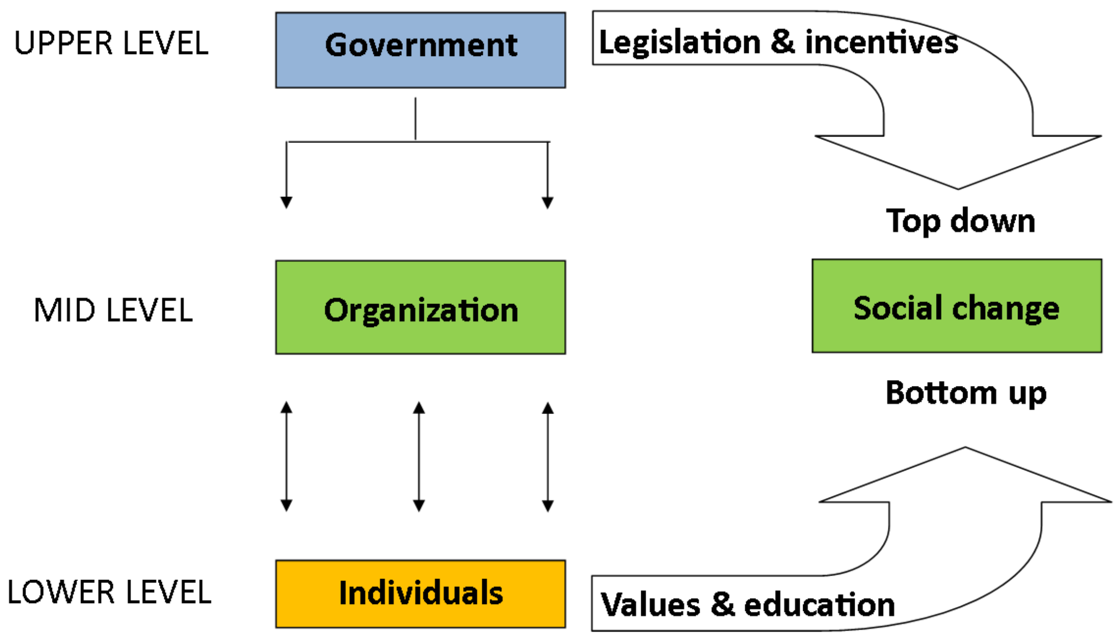


Figure 2.12. Hierarchical patch dynamics model for a social system. The ability of an organization to function, learn, and adapt is dependent both on constraints created by the governing organization, and the values of individuals within the organizations, and their ability to interact constructively and apply new knowledge.

Chapter 3

PATTERNS OF REMOTELY SENSED FOREST HEALTH DECLINE ASSOCIATED WITH DISTURBANCE IN OLYMPIC NATIONAL PARK, WA

SUMMARY

Spectral change detected in 28 years (1985 to 2012) of Landsat imagery processed by LandTrendr was used to investigate patterns of landscape disturbance within the protected forest of Olympic National Park. Normalized Burn Ratio (NBR) was used as the index for change because it is an effective indicator of vegetation health. My objectives were to (1) describe and quantify spatial and temporal patterns of disturbance in forested vegetation of Olympic National park from 1985 to 2012; (2) characterize dominant types of disturbance based on remotely sensed attributes of decline onset, magnitude, and duration; and (3) compare spatiotemporal vegetation disturbance patterns with patterns in climate, storm events, insect activity, and their interactions. I differentiated three types of disturbance based on spatial and temporal attributes of spectral reflectance measured in NBR: very high-, high-, and low-magnitude decline. Very high magnitude decline was caused by abiotic disturbances of fire, avalanches, mass movement, riparian activity, and severe blowdown that affected localized patches. These were represented on LandTrendr disturbance maps as magnitude decline with > 60% relative change in NBR. The remaining disturbance (14-60% relative change in NBR) was widely distributed but more frequent along high elevation slopes. Total area in decline increased in extent from 1994 to 2008 with no increase from 2000 to 2002. When positive change in spectral reflectance, representing vegetation recovery-growth was also considered, onset of change began about 3 years earlier and had the appearance of a cycle ending in 2007 when recovery-growth began to increase. Raw NBR was used to differentiate widespread disturbance into high- and low-magnitude decline types. High-magnitude decline was punctuated and of short duration (1-4 years), distributed across elevations with variable aspect, and temporally related to winter weather events (wind gusts > 95 km/hr, rain, hail, ice, and snow) that commonly occurred during cool years (PDO index values). Most of the decline detected across Olympic National Park was low-magnitude and occurred at high elevation (above 1000 m). Distribution of low-magnitude decline was correlated with the distribution of subalpine fir (*Abies lasiocarpa* (Hook) Nutt. var. *lasiocarpa*) forest ($r = 0.98$, $df = 54$, $p < 0.001$). Low-magnitude decline was also correlated with the elevation ($r = 0.87$, $df = 54$, $p < 0.001$) and aspect ($r = 0.73$, $df = 34$, $p < 0.001$) of aerial detection survey polygons delineating effects of balsam woolly adelgid, an invasive insect that defoliates and kills true fir. Increasing extent of NBR decline was correlated with warm PDO years (PDO index values; $r = 0.57$, $df = 17$, $p < 0.001$) and occurred predominantly on southern slopes indicating the possibility of a climate-biota interaction, because warm temperature favors insect population growth and stresses infested host trees.

INTRODUCTION

Patterns of landscape change in forested areas differ broadly in extent, magnitude, and duration, and the patterns are indicative of causal factors and the scales at which they operate and interact (White and Pickett 1985, Turner 1989). Increased rates of tree mortality in western forests have been attributed to extreme climate factors (Breshears et al. 2005, Westerling et al. 2006, van Mantgem and Stephenson 2007, Allen et al. 2010), invasive insects and disease (Hayes and Ragenovich 2001, Ellison et al. 2005), and interactions with endemic species and altered disturbance regimes (Bentz et al. 2010, Weed et al. 2015).

Within Olympic National Park (ONP), high-magnitude, abrupt changes to vegetation are commonly caused by fire, avalanches, landslides, riparian activity, and occasionally strong mid-latitude cyclones (**Fig. 3.2**). Fire can cause the most extensive change but has been historically variable and linked to climate fluctuations. Three large burning periods in 1308, 1448-1538, and 1668-1701 had fires on the scale of several hundred thousand hectares, e.g., the 1701 fire burned 404,700 ha of Douglas-fir forest (Henderson et al. 1989). In the last century large fires on the Peninsula have burned on the scale of thousands of hectares, the largest being the Sol Duc fire in 1907, burning 12,800 ha, and the Forks fire in 1951, burning 13,355 ha. However, in the past 60 years only two fires have exceeded 400 ha within ONP. The ONP Fire Management Plan aims to limit area burned to below 81 ha per year with a maximum of 526 ha in a peak year (Kailin and Nickey 2005). Fire suppression and control have decreased fire size compared to historic area burned.

Wind from temperate cyclonic storms is a dominant disturbance agent during winter months (Mass 2008) and can cause widespread low- to localized high-magnitude structural damage in forests across the Olympic Peninsula. Storms approach the Oregon and Washington coastline from the southwest and move across flat lowlands and into the mountains. Severe wind gusts accompany storms several times a decade (http://www.climate.washington.edu/stormking/) and cause extensive tree damage approximately every 30 years (e.g., the 1921 and 1962 storms; Agee 1994a).Wind disturbance has the greatest impact on the west side of the Olympic Peninsula, along the coast, and on south-facing slopes (Agee 1994a). However, cyclonic storms originating from the southwest also topple and break trees on north-facing slopes and valleys in the northern part of the Olympic Peninsula, indicating that wind movement is complex and often channeled by topography (Woodward et al. 2011). Wind storms are more common during neutral years of El Niño Southern Oscillation (ENSO) and least common during the El Niño years when the Pacific jet stream splits at the latitude of Washington and moves toward Alaska and

California (Mass 2008). When combined with heavy precipitation, such as occurs during La Niña years, windstorms can be especially damaging to trees because saturated soil loses strength and snow or ice add weight to branches. Avalanches and landslides are likely to contribute to disturbance during cool wet La Niña years as well, compared to years with low precipitation and low snowpack. Wind disturbance interacts synergistically with insects and disease; trees that are structurally weakened by root and stem disease are more likely to break or blow down; downed trees create substrate for bark beetle reproduction (Edmonds et al. 2011b).

Endemic tree pathogens and herbivores coexist with vegetation on the Olympic Peninsula ordinarily causing damage in small scattered patches. Generally only weak and stressed trees are attacked, but on occasion, conditions that stress vegetation (e.g., drought or defoliation) can result in outbreaks of insects and disease. Bark beetles as a group have great potential to threaten conifers when conditions are conducive to bark beetle survival, host entry, reproduction, and spread (Breshears et al. 2005, Berg et al. 2006, Kegley 2006, Raffa et al. 2008). Warm climate conditions favor insect population growth and simultaneously reduce tree defenses when trees become moisture stressed. Alternatively, cool, wet conditions support the spread of fungal pathogens (Sturrock et al. 2011) and can shorten the growing season at high elevation (Ettl and Peterson 1995).

Invasive insects and pathogens have potential to cause extensive tree mortality because they lack native parasites and predators, and hosts lack co-evolved defenses or resistance. Invasive organisms generally target and kill susceptible host species and can eliminate an entire population (Ellison et al. 2005). *Cronartium ribicola*, the cause of white-pine blister rust, has reduced whitebark pine and western white pine populations in Olympic National Park (Bedwell and Childs 1943, McDonald and Hoff 2001). The balsam woolly adelgid (BWA), *Adelges piceae*

Ratzeburg, is a threat to subalpine fir (*Abies lasiocarpa* var. *lasiocarpa*) health on the Olympic Peninsula at this time. Although the insect is only 0.8 mm, a substance in the saliva released during feeding alters tree cell growth and restricts transport of water and nutrients (Balch 1952, Hain et al. 1991). This causes stress and trees become more susceptible to drought (Hollingsworth and Hain 1994) and likely endemic agents as well. A tree can die in a few years, survive with insects for up to 20 years, or recover, depending on duration, severity, and location of infestation, and local environmental conditions (Mitchell and Buffam 2001). The BWA has slowly eliminated up to 79% and 90% of true fir trees (*Abies* spp.) within some forested stands on the west and east coasts of North America, respectively (Witter and Ragenovich 1986, Mitchell and Buffam 2001). Land management agencies continue to be vigilant against a number of key invasive insects and disease that could be transported to the Olympic Peninsula.

The value of remote sensing for investigating, monitoring, and identifying factors involved in landscape change are widely recognized (Bonneau et al. 1999, Wulder et al. 2006, Kennedy et al. 2007a, 2012). One of the challenges has been to detect and quantify gradual and subtle changes, and a second challenge is to differentiate multiple causal factors. Few researchers have been able to fully describe decline trends caused by slow-acting biotic agents and to satisfactorily separate these from other kinds of disturbance (Meigs et al. 2011). Landsat satellite imagery has provided continuous spatial coverage for nearly three decades, a short time period to identify or differentiate long-term patterns or cycles, but a long time period to have a permanent record with such wide spatial scope. Landsat-based Detection of Trends in Disturbance and Recovery (LandTrendr; Kennedy et al. 2010) was developed as a tool for landscape change detection and has been used to quantify harvest impacts (Kennedy et al. 2012), monitor high-magnitude change in protected areas (Antonova et al. 2013), and identify trends in insect defoliation (Meigs et al. 2011). Methods developed to filter spectral noise and capture large concentrated patches of decline often eliminate subtle change or alter patterns that can be biologically significant; I sought to minimize this effect.

I used remotely sensed time series data to investigate landscape patterns of forest disturbance in Olympic National Park by differentiating and assessing high- and low-magnitude change. I developed methods using LandTrendr raster data sets to assess a broad range of vegetation change patterns within a forested protected area. My objectives were to (1) describe and quantify spatial and temporal patterns of disturbance in forested vegetation of Olympic National Park from 1985 to 2012; (2) characterize dominant types of disturbance based on remotely sensed attributes of decline onset, magnitude, and duration; and (3) compare spatiotemporal vegetation disturbance patterns with patterns in climate, storm events, insect activity, and their interactions.

METHODS

Study site

The study site is located on the Olympic Peninsula of Washington State, USA, and encompasses 348,277 ha of Olympic National Park (ONP) where the park intersects Landsat scene 4727 (**Fig 3.1**). This excludes the coastal section of ONP and two small sections of lowland forest on the western side of the Peninsula, but includes the complex of forested mountain ranges that rise to 1,800 m in elevation and higher (Henderson et al. 1989). Montane and subalpine forests are isolated from similar forests in the Cascade Range to the east, and protected from resource extraction within the boundaries of Olympic National Park. Thus disturbance patterns are limited to those caused by biotic and abiotic agents of the Olympic Peninsula.

The climate of the Olympic Peninsula is moderated by three water bodies (Pacific Ocean, Strait of Juan de Fuca, and Puget Sound) and is classified as mild maritime with high winter precipitation and summer drought. Average annual mean temperature for the study area is 6.8 ^{o}C with an average January minimum of -1.5 ^{o}C and a maximum in August of 20.3 ^{o}C (30-year average temperature data extracted to 1000 random points for ONP at 0-1900 m elevation; PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, accessed 23 Apr 2015). Micro-climatic conditions vary due to steep topography and variations in slope and aspect. Orographic precipitation from moisture moving off the Pacific Ocean is released on the west slope of the Olympic Mountains creating drier conditions on the east side of the Olympics. High elevations are cold with low winter soil temperatures, and south-facing slopes are warmer during the summer with relatively high soil temperatures especially in sparsely vegetated stands (Henderson et al. 1989). Annual precipitation ranges from ≥ 500 cm in west-side mountains, to 250 cm in east-side mountains where subalpine fir dominates, and to 41 cm in the northeastern lowlands (PRISM; 30-year annual average for 1971-2000). Adiabatic cooling causes about 60% of winter precipitation to fall as snow at high elevations, thereby creating a moisture reservoir that distributes water to lower elevations during the spring and summer; early snow-melt as a result of climate change may be altering this pattern (Mote 2003, Hamlet et al. 2005). Summer thunderstorms may occur May through August (Estberg et al. 2008) with lightning activity that generally moves south to north across the mountains. Heavy thunderstorm activity occurs during the driest months of late summer (Pickford et al. 1980) when fire risk is moderate to high. From October through February, mid-latitude cyclones produce 65-115 km/h winds that move off the Pacific Ocean from the southwest.

Climate cycles such as El Niño Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO) also affect climate conditions on the west coast of the Olympic Peninsula (Mantua et al. 1997, Diaz et al. 2001, Mass 2008). These cycles are associated with annual to decadal trends in monthly sea-surface temperature and pressure differentials. ENSO completes a cycle every 3-7 years with two alternating climate patterns: (1) El Niño periods that are warm and dry, and (2) La Niña periods that are cool and wet. Transition years are neutral but may be associated with increased storm occurrence. Similar to ENSO, the PDO index registers as monthly warm or cool phases of varying intensity but has a broad cyclic trend of 20 to 30 years that fluctuates between cool, negative phases with high snowpack, and warm, positive phases with low snowpack. When ENSO and PDO are both in a negative or positive phase, effects are compounded.

The forests of Olympic National Park are separated into 3 broad categories for this study based on National Vegetation Classification Standard (NVCS) macrogroups (Federal Geographic Data Committee, 2008): (1) Rocky Mountain Subalpine & High Montane Subalpine Forest (RMSF) at high elevations in the dry northeast; (2) Vancouverian Subalpine Forest (VSF) at high elevations in the wet southwest; and (3) all remaining forest types that I have combined into a "lowland forest" category (**Fig. 3.1**). The NVCS was developed to create a consistent National vegetation classification for federal agencies, and is based on existing vegetation data gathered at local, regional, or national levels as plot data and from scientific literature. The macrogroup is defined by plant communities that have a common set of growth forms and diagnostic plant taxa, and that are similar in geographic region, climate, and disturbance. Several plant alliances and associations may be contained within a macrogroup. Subalpine fir is a dominant tree species in the RMSF macrogroup where it co-occurs with lodgepole pine (*Pinus contorta* var. *latifolia*) on dry slopes and with mountain hemlock (*Tsuga mertensiana*) on north-facing slopes. Mountain hemlock is a dominant species in VSF. In the Pacific Northwest, subalpine fir grows between 1200 m and 1800 m elevation, but is found as low as 600 m along cold stream bottoms (Alexander et al. 1984). For an additional comparison between low-magnitude decline and subalpine fir distribution I added the plant association group series of Henderson (1989).

Soils on the Olympic peninsula are derived from sedimentary rocks, marine basalts, and glacial sediments, and are relatively young and infertile. Inceptisols and entisols are the most common soil orders on the Peninsula and range in depth from exposed bedrock and thin regolith on upper slopes to deep colluvial regolith collecting on lower slopes (Henderson et al. 1989).

Aerial survey of Hurricane Ridge area

I conducted two fixed-wing aerial reconnaissance flights over ONP in July of 2008 and August of 2012. In 2008, I observed poor tree health along high-elevation slopes. The flight in 2012 confirmed that extensive and severe tree health decline had occurred at high elevations though the patterns were less obvious from the ground (**Fig. 3.3**). The United States Forest Service (USFS) and Department of Natural Resources (DNR) conduct aerial detection surveys (ADS) across state and federal forest land every year to detect and monitor insect and disease activity (Ciesla 2006). I used data from ADS to complement the analyses of temporal trends from satellite imagery.

Processing of TM Landsat imagery with LandTrendr

Raster data sets were created by the Laboratory for Applications of Remote Sensing in Ecology (LARSE) at Oregon State University using 28 years (1984-2012) of Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) images for scene 4727 (WRS-2 path 47, row 27). The images were processed with USGS L1T LPGS, converted to surface reflectance, cloud-screened, normalized (MADCAL), and converted to index values (see Kennedy et al. 2007b, 2010, 2012). Image dates were selected to fall primarily in July and August, when clouds are minimal in the region but forest vegetation is maximized. The normalized burn ratio (NBR) was used as the index of spectral reflectance because it effectively detects vegetation change (Cohen et al. 2010). The NBR contrasts near infrared (NIR) and shortwave infrared (SWIR) bands (NBR = (Band 4 – Band 7) / (Band 4 + Band 7)) (Key and Benson 2006). Original NBR values were multiplied by 1000 and rounded to the nearest integer to reduce image bit size. Images have a spatial resolution of 30 m.

The best cloud-free index values were compiled into a source data set as an annual time series image stack. LandTrendr segmentation algorithms were then applied to the annual source data for each pixel, to identify start and end years (vertices) of spectral change, and to fit a straight regression-line through the vertices (i.e., Landtrendr-fitted data). Fitted index values for each vertex and year along a trajectory were used to create a disturbance map layer with three bands that represented attributes of (a) onset year of change, (b) percent magnitude change, and (c) duration of continuous change. Magnitude of change was expressed as a relative percent loss of vegetative cover for a given pixel as modeled from the pre- and post-disturbance NBR values. An annual magnitude change layer was created by applying year-to-year subtraction to fitted NBR values that detected the greatest change trend for each pixel. The result was decline and recovery-growth data sets containing raw annual NBR change values (not percentages), and 28 bands, one for each year.

Spatial processing with ArcGIS

LandTrendr data layers were processed in a geographic information system (GIS) to include only forested pixels within the ONP boundary. The coastal section and two small low-elevation sections at the west edge of the Park are not included in Landsat image scene 4727, and thus were excluded from this landscape assessment (**see Fig. 3.2**). The disturbance map layer was visually calibrated with a fire map layer and aerial photos (provided by Olympic National Park) to identify fire, avalanche, and mass movement patches, and determine the magnitude cutoff of 60% between very high-magnitude localized disturbance (> 60%) and more widespread disturbance (14-60%). The lower limit of 14% was set to minimize the detection of change associated with phenology or other spectral noise rather than chronic stress or mortality of trees. This threshold was determined through analysis of the disturbance magnitude frequency and spatial distribution. Magnitude values below 14% had a highly diffuse spatial pattern unassociated with any particular event and had a frequency too great to represent low-magnitude vegetation loss from biotic and abiotic agents. The resulting spatial data set provided a spatiotemporal record of disturbance in Olympic National Park from 1985-2012.

The annual magnitude change layer was visually calibrated with ADS data to determine the lower and upper limits of broad-scale disturbance at 4-400 NBR. The 4-400 NBR range is similar to the 14-60% disturbance category in the disturbance map layer. This range was further differentiated by magnitude into low- and high-magnitude decline categories. The ADS polygons of insect defoliation guided the setting of low-magnitude decline limits at 4-40 NBR. Excluding NBR values below 4 reduced spectral noise, whereas excluding values above 400 removed the signal from severe abiotic disturbance such as fire and avalanches.

ArcGIS 10.1 (ESRI 2012) was used to extract elevation and aspect data for low- and high-magnitude change. Rose diagrams representing aspect were created in ggplot2 (Wickham 2009).

Statistical analysis

Pearson product-moment correlations (Pearson 1896, Crawley 2005) were calculated in R (R Developent Core Team 2008) to quantify relationships between occurrence of low-magnitude decline each year and the annual average for each of two climate variables: ENSO and PDO. The following values represented climate phase and vegetation response in correlation computations: 1 = increase in extent of vegetation decline; 0 = no increase; 1 = warm climate phase; and 0 = cool climate phase.

Pearson's correlation was also used to quantify topographic (elevation and aspect) relationships between (1) distributions of low-magnitude decline and subalpine fir forest types RMSF (Federal Geographic Data Committee 2008) and Subalpine fir (SF) series (Henderson et al. 1989); (2) distributions of low-magnitude decline and BWA polygons delineated by ADS; and (3) distributions of BWA polygons and subalpine fir forest types RMSF and SF. Topographic correlations were calculated by comparing area (ha) of each variable that occurred within 40 m elevation increments and 10 degree aspect increments. For ease of presentation, the values for low-magnitude decline were from 2003, which was a subsample of the entire time period.

RESULTS

Spatial distribution of NBR spectral decline in ONP forest

Spectral decline in NBR detected with Landsat imagery from 1985 to 2012 was distributed across Olympic National Park (**Fig. 3.2**). Patches of localized very high-magnitude decline (> 60% relative vegetation cover loss) were associated with severe abiotic disturbance such as fire, avalanches, riparian activity, and severe blowdown, and were verified with fire maps and aerial photos. The damaging effects of a single disturbance event can range from low to high in magnitude, but pixels with very high-magnitude change uniquely identified the location and extent of severe abiotic disturbance where a large percentage of vegetation cover had been lost in a short period of time. The patches with very high-magnitude change were described and monitored in NCCN National Parks (Antonova et al. 2013) and are not addressed in this study.

Relative vegetation cover loss of 14 to 60% was widespread in Olympic National Park. Vegetation loss was especially apparent along high-elevation slopes in northeast ONP where the RMSF macrogroup and subalpine fir are dominant (**Fig. 3.2 inset; see also Fig. 3.1**).

Annual change in extent of NBR spectral decline and recovery within forests of ONP

The forested area within ONP experienced an 11-year period of spectral decline (raw NBR values*1000 ranging 4-400) from 1994 to 2008 (**Fig 3.4a**). For eight years prior to this, the extent of decline was consistently low, averaging 3.4% of total area examined. By 2008, 12.5% of the area was in decline, an increase of 9.1%. Decline magnitudes contributing the most area were < 40 units change in NBR.

Spectral recovery and growth (4-400 change in NBR*1000; hereafter recovery/growth) increased abruptly for two years after 1985, 2007, and 2008. Recovery/growth began to slow before spectral decline became evident in 1994 (**Fig 3.4b**). When spectral decline and recovery/growth were considered together (recovery/growth minus decline) a negative trend began approximately in 1990 and a positive trend began in 2008 (**Fig 3.4c**).

NBR spectral decline trends differentiated by magnitude and duration

Spectral decline in NBR differentiated by magnitude and duration of change had different spatio-temporal trends. Low-magnitude decline (4-40 Δ raw NBR*1000) showed a gradual increase from 1994 to 2007 similar to the undifferentiated trend for Olympic National Park.

High-magnitude decline (41-400 Δ raw NBR*1000) was punctuated and often detected after winter storm events if wind gusts reached 95 km/h (at a minimum of three weather stations; **Table 1**; **Fig. 3.1**). When separated by duration using the disturbance map layer, the onset of long-duration ($\geq$ 5 years) spectral decline had consistent new area in decline each year which coincided with the cumulative increase seen in low-magnitude decline (**Fig. 3.5c**). The onset area of short-duration (< 5 years) decline matched the occurrence of high-magnitude decline (**Fig. 3.5b**).

Topographic distribution of high- and low-magnitude NBR spectral decline

Low-magnitude decline consistently dominated elevations above 1000 m and southern aspects each year (**Fig. 3.6**). A comparison of years showed that the area in decline gradually increased over time, especially at high elevation. At lower elevations, low-magnitude decline increased in area very slowly until 2007 and 2008 when mid-elevation decline expanded.

In contrast, distribution of high-magnitude decline had great interannual variability. The area affected became evident across elevations after each winter with severe storm events and then subsided the following year. Dominant aspect varied by year and ranged from northwest to southwest, south, southeast, and east.

Correlations with climate and vegetation constraints and processes

Low-magnitude NBR decline was correlated with short-term warm and cool phases of PDO (**Fig. 3.5a**) after 1993 ($r = 0.57$, df = 17, p = 0.01), however decline was not correlated with ENSO ($r = 0.22$, df = 17, p = 0.37). Low-magnitude NBR decline increased in area during warm phase years of PDO and did not increase during cool phase years. The occurrence of severe winter storms and associated high-magnitude spectral decline may also be related to PDO; storm events often occurred during cool phase years.

The elevation distribution of low-magnitude decline (year 2003) was correlated with elevation distributions of two subalpine fir forest types (**Fig. 3.7a**) RMSF ($r = 0.78$, df = 54, $p < 0.001$) and SF ($r = 0.53$, df = 54, $p < 0.001$), but not with aspect. Low-magnitude decline was detected on southern aspects whereas RMSF dominated north and west aspects (**Fig. 3.7b**) and SF series dominated southwest and north aspects (**Fig. 3.7c**).

Distribution of low-magnitude decline (2003) was strongly correlated with ADS BWA polygons (**Fig. 3.8**) for elevation ($r = 0.87$, df = 54, $p = 0.001$) and aspect ($r = 0.73$, df = 34, $p < 0.001$). Elevation of BWA polygons were similarly correlated with subalpine fir forest types RMSF ($r = 0.98$, df = 54, $p < 0.001$) and SF ($r = 0.82$, df = 54, $p < 0.001$).

DISCUSSION

I found that the total area experiencing decline in NBR increased from 1994 to 2008 or from 1990 to 2007 if recovery-growth was considered. This decline was predominantly associated with low-magnitude biotic activity and appeared to interact with climate, based on an increase in area associated with warm PDO indices. Low-magnitude decline was concentrated at high elevation from the start of the Landsat record in 1985 and associated with subalpine fir distribution and ADS-detection of BWA. Increasing area of decline at mid-elevation in the late 2000s may have represented additional biotic disturbance activity. High-magnitude decline was of short duration with variable distribution and associated with winter weather events, but also increased in area in the late 2000s indicating a possible abiotic x biotic interaction.

Spatial distribution of NBR spectral decline in ONP forest

Landsat spectral decline in NBR was detected across Olympic National Park and separated by magnitude into very high-, high-, and low-magnitude types. A unique set of spatio-temporal attributes was associated with each type and contributed to an understanding of patterngenerating constraints, processes, and interactions. Very high-magnitude decline was localized in patches with shapes, dimensions, and locations that reflected the type of disturbance (e.g., fire, avalanches, riparian activity, or landslides). High- and low-magnitude decline were widely distributed. Low-magnitude decline was more abundant at high elevations for the 28-year time period examined, and the most extensive decline type occurring within Olympic National Park.

Annual change in extent of NBR spectral decline and recovery within forests of ONP

Spectral decline between 4 and 400 NBR units across ONP was suspected to represent forest damage including stem and branch breakage or tree toppling, and pathogen- or insect-caused defoliation and mortality. Annual change in extent of spectral decline was dominated by low-magnitude and is discussed below.

Spectral recovery/growth for ONP showed temporal patterns (**Fig. 3.4b**) that indicated a response to abiotic and biotic disturbance. Recovery/growth is a measure of (1) forest growth and (2) recovery after damage in a forest that is able to positively respond with new vegetative growth (Kennedy et al. 2007b). The abrupt recovery after 1985, 2007, and 2008 may in part be related to a growth response following high-magnitude abiotic disturbance events (unknown event in 1985, severe wind storms in 2006 and 2007). Recovery/growth was not as strong in 2008 and 2009 as it was after 1985 suggesting that the forest may have been less able to positively respond in some areas due to biotic and/or climate stress. A spatial assessment of recovery/growth (e.g., **Figs. 3.2 and 3.6**) in comparison with decline for specific years could lend support to these hypotheses but was not conducted in this research.

Spatio-temporal distribution of decline types and associated constraints and processes

<u>High-magnitude NBR decline associated with winter weather events</u>

High-magnitude spectral decline of NBR within ONP increased abruptly in area and subsided within one to four years after onset. This pattern of short-term high-magnitude decline may be common for abiotic disturbance because abiotic events such as weather and fire terminate quickly and recovery begins once the agent is no longer present. Furthermore, abiotic disturbance generally affects one area simultaneously and does not spread, except from the limited activity of opportunistic biotic agents or wind exposure along disturbance edges (Hennon et al. 2006, Mitchell and Lanquaye-Opoku 2007). Also, abiotic disturbance does not target a single species, except when one species is more susceptible than others, in which case the boundaries may be determined by the intersection of the disturbance and the susceptible species.

High-magnitude NBR spectral decline detected with LandTrendr consistently occurred in the season after winter storm events with wind gusts measuring 95 km per hour or higher. Winter storms have been a repetitive agent of disturbance on the Peninsula. Historically, severe tree damage on the Olympic Peninsula has been caused by high winds accompanied by soil-saturating precipitation and snow- and ice-laden branches (Agee 1994b).

Cyclonic wind patterns likely interact with topography and distributions of precipitation to cause variability in damage extent and aspect each year. Spectral decline was most common on south and southwest aspects in ONP (1991, 1996, 1999, and 2008) because strong cyclonic winds arrive off the Pacific Ocean and move inland from the southwest. Occasionally damage was more prominent along east and north aspects as in 1985, 1999, and 2003. In 1985 short-term high-magnitude decline was extensively distributed across low to mid elevations with northwest aspects (2003 had a similar aspect distribution). Vegetation damage did not follow a winter wind storm with strong gusts, but may have resulted from a hail or ice event (Office of Washington State Climatologist, personal communication, 2013).

Damage to trees from winter weather events can be severe, but total damaged area generally remained below 4000 ha (about 1% of ONP excluding the coastal section), and only occasionally exceeded 8000 ha (**Fig. 3.5**). Strong wind storms in 2006 and 2007 with gusts as high as 140 km/hour and 150 km/hour respectively, caused the extensive high-magnitude decline that was detected in 2007 and 2008. Additional damage in the later 2000's may have been caused by synergistic interactions with biotic agents that increased the susceptibility of trees to toppling and breakage.

Low-magnitude NBR decline associated with BWA and climate

Low-magnitude NBR spectral decline in Olympic National Park increased in area over time from 1994 to 2008 (**Fig. 3.4a**) and was associated with the distribution of subalpine fir forest. This supports the idea that subalpine fir tree health has been predominantly affected by a biotic agent. The pattern of gradual increase in area fits a vegetation response to biotic agents that begin in one area, increase in population, and spread to other areas. Consistent distribution along southern aspects and association with warm PDO index values after 1993 also indicated the possibility of a climate interaction. An interaction between biotic and abiotic agents is further suggested by the increasing extent of high-magnitude damage during short duration disturbance events.

Balsam woolly adelgid is the likely agent of biotic disturbance in subalpine fir forest of ONP based on defoliation of trees along high-elevation slopes and similar patterns observed and verified as BWA in the Cascade Mountains (Mitchell 2001, see also Chapter 4). The first ADS detection of BWA in ONP subalpine fir occurred in 1970 at the east end of Hurricane Ridge; it was also documented SE of ONP on subalpine fir of Olympic National Forest in 1969 (R. Johnsey letter, 1-6-72, DNR memorandum). New locations were mapped in northeast ONP in

1977, but detection increased very little until 1989. After 1989, BWA was detected every year in subalpine fir of ONP. Pacific silver fir is less susceptible to BWA than subalpine fir at high elevations, but both Pacific silver fir and grand fir become infested at low elevations (Johnson and Wright 1957, Johnson et al. 1963) and may have contributed to the more slowly increasing extent of spectral decline at lower elevation.

Landsat imagery indicated that forest condition in ONP was stable and recovering from 1986 to 1993, after possible abiotic disturbance in 1985 (**Fig. 3.4**). However, topographic distribution of low-magnitude decline during this time covered more area above 1000 m than below (**Fig. 3.6**) indicating that biotic agents were already active in high elevation forest as documented by ADS. From 1993 to 2008 the area in decline above 1000 m increased during warm PDO years and on southern exposures, indicating that trees may have been more susceptible during warm dry years.

Warm temperatures can physiologically compound damage to BWA-infested trees. Mitchell and Buffam (2001) noted that three to four years of warmer than average summers in the Cascade Mountains increased the damage to BWA-infested subalpine fir trees at high elevations. The heat accumulation may have allowed more BWA to reach the first instar overwintering stage and improved their survival over winter. Adelgids also alter the cell growth of tree vascular tissue during the feeding process (Balch 1952, Puritch and Petty 1971). Saliva injected into cambial tissue interacts with tree physiology to produce excess parenchyma cells and enlarged, shortened tracheid cells that transport water poorly (Mitchell 1967, Hollingsworth and Hain 1994). The result is reduced xylem pressure potential, disrupted sap flow, and water stress. Infested trees also become inefficient by continuing to produce extra wood during periods of drought, whereas uninfested trees slow production during these times (Hollingsworth and

Hain 1994). Trees in moisture-limited environments thus may succumb more easily to heat, BWA effects, and opportunistic agents. Poor soils can exacerbate drought stress and may play a larger role than aspect (Hollingsworth and Hain 1994, McManamay et al. 2011). Observations in the Cascade Range indicated that relatively dry sites had less mortality than wet sites, but the difference was small (56% vs. 79% mortality; Mitchell and Buffam 2001).

Tree distribution is important to consider when interpreting remotely sensed spatial patterns. Spectral decline detected on south-facing slopes could be caused by heat-loading and tree stress or, alternatively, by the distribution of host species. Although two subalpine forest types are distributed on north aspects (**Fig. 3.7**), it is possible that within ONP, subalpine fir trees are more abundant on south aspects. This would create a pattern of decline that dominates southern aspects simply because the biotic agent targets a species with this distribution. According to Fonda and Bliss (1969), subalpine fir grows (1) on dry, south-facing ridges of west-central Olympics, (2) on most aspects except north in the east-central section, and (3) on north-facing slopes only in the driest eastern section of the mountains. Subalpine fir distribution and heat stress may play roles in the distribution of decline on south aspects.

Implications for subalpine fir health and recovery

The future of subalpine fir in ONP is uncertain because of combined effects of BWA and climate change. Understanding patterns of tree health decline in ONP and comparing this to the effects of BWA infestations in other areas suggests possible outcomes for ONP's subalpine fir in the near future.

The rate of BWA-related tree health decline in ONP was affected by environmental factors, e.g., decline increased during warm climate conditions. The perceived duration of decline varies with method and scale. For the extent of ONP, decline began prior to the start of the TM Landsat time series as evidenced by the 1985 Gaussian-shaped distribution above 1000 m elevation (**Fig. 3.6**). This indicated that biotic disturbance had continued at high elevation for at least 28 years. However, aerial surveys detected the first BWA activity in subalpine fir of ONP in 1970. Thus BWA-related decline has been ongoing for at least 28 years and as many as 43 years for all of ONP by the end of the study period in 2012.

Overall, spectral decline slowed and recovery/growth increased after 2007 in ONP (**Fig 3.4**). This may represent a turning point in the progression of BWA effects. Recovery from insect disturbance generally begins with removal or reduction of the insect or reduction of the susceptible host with obvious feedback to the insect population. The apparent recovery after 2007 could also be a momentary pause due to the onset of temporary cool, wet climate conditions that reduced insect activity and increased the tolerance of trees with compromised vascular systems. Then again, recovery/growth may simply be related to a growth response after severe weather events; further investigation is needed. Consequently, prediction of recovery is premature, but observations of BWA infestations elsewhere suggest that recovery is possible.

In the Appalachian Mountains, BWA infestations of Fraser fir (*Abies fraseri*) have completed a cycle of decline and recovery. Balsam woolly adelgid was detected on Fraser fir in mid-1950 and caused 82-98% mortality of overstory trees 11 years later in stands near Mt. Mitchell (Witter and Ragenovich 1986). Aerial photos of the Black Mountains indicated a 24-year decline (1964-1988; McManamay 2011). Importantly, researchers have reported Fraser fir regeneration, cone production, and a return to dominant and co-dominant status (Adams and Stephenson 2010, McManamay et al. 2011, Stehn et al. 2013). BWA is still present in Fraser fir stands, but is having less impact on the young stands. As trees mature they may become more susceptible. Similar patterns have been observed in the Cascade Range where mortality was initially high, but fir trees were not completely eliminated from their natural habitat (Mitchell 2001).

Management applications

Detecting and differentiating broad scale patterns is an important first step in identifying abiotic and biotic disturbances, their possible interactions, and the role of constraints such as climate. The tools that I employed in this study have broad applicability for forest monitoring and research and may be adapted for use in protected areas outside Olympic National Park. For example, categories of very high-, high-, and low-magnitude spectral may be used to monitor location, extent, and timing of abiotic and biotic disturbance for broad or specific areas, and to identify associations with constraints and processes in an investigation of cause. A range of magnitudes and duration may accompany any event or agent and their interactions, but a dominant set of attributes are associated with disturbance types. Temporal trends for specific disturbance agents and locations may be investigated by employing Landsat time series data within areas identified by ADS, and ADS polygons may be used to train Landsat by identifying attributes associated with specific disturbances. Emphasis may be adapted to the unique needs and disturbances of different protected areas. Limitations may exist for detecting certain agents; for example, spectral change may not be visible for Swiss needle cast (disease caused by the fungus *Phaeocryptopus gaeumannii*) at midsummer and may require the timing of Landsat scenes to be centered in the spring.

The distribution of low-magnitude decline on the Olympic Peninsula provides an example of spatio-temporal patterns associated with an invasive biotic agent (BWA) that is constrained by host species distribution (subalpine fir). The rate of increase, magnitude and duration of response, and return interval vary by agent, and this pattern may increase in complexity with interacting constraints, processes, thresholds, and feedbacks. For example warm, dry conditions may increase insect population growth and stress to host species (Logan et al. 2003), whereas cool, wet conditions may decrease insect activity, but increase the spread of fungal pathogens (Sturrock et al. 2011). Furthermore, one agent may weaken a tree and make it more susceptible to a second agent such as bark beetles. If enough susceptible agents or hosts are present a threshold may be passed that allows the agent to spread and cause widespread damage (Berg et al. 2006). Once a large percentage of hosts are killed, the disturbance agent population declines and may not cause extensive damage again until host and environmental conditions are conducive to an outbreak (Cairns et al. 2008).

Changes in disturbance predominantly related to climate may have distinct patterns. For example, the distribution of declining Alaska yellow cedar (*Callitropsis nootkatensis*) was bound by topography but not by the extent of the species (Hennon et al. 2006). Alaska yellow cedar decline was caused by freezing of shallow roots that had lost snow protection at lower elevations due to climate warming after 1880. Damage occurred abruptly and simultaneously across the entire affected area, and then occurred again during low-snow freeze events. A second example of climate effects occurred with extensive mountain pine beetle outbreaks in the Rocky Mountains. Although the disturbance agent was biotic, the extent and severity of impacts to the host species were related to the influence of a warming climate on beetle survival and reproduction and an abundance of optimal hosts (Bentz et al. 2010). In contrast, patterns of decline in ONP suggest that although climate and weather interacted with BWA, their individual effects on subalpine fir decline were minor in comparison to the effects of the invasive exotic insect.

Conclusion

This study has demonstrated how Landsat imagery data may be used to investigate spatio-temporal patterns and trends in disturbance of forested landscapes. Attributes of spectral change differ with disturbance type; thus magnitude, duration, onset, and distribution may be used to separate types for analysis, especially when aided by aerial detection surveys. Low-magnitude decline is often neglected in Landsat change detection, but I found that it was a major indicator of disturbance likely associated with biotic activity that interacted with climate.

Landsat change detection tools may be developed further by applying them at other locations and comparing results across scales and across broad areas in an exploration of pattern that is related to constraints and processes. Additional bands or indices may be further explored for their ability to resolve signal-to-noise issues and to detect and differentiate change types within the forested landscape, perhaps as a complement to NBR. Landsat tools will become more useful as relationships between pattern and cause are better understood. Importantly, Landsat imagery provides a means to improve change detection using three decades of consistent and objective data that covers entire landscapes at 30 m resolution, and that may be repeatedly revisited as methods are advanced.

The potential for Landsat change detection in monitoring and research has only begun to be explored. With increasing concern for broad-scale effects of climate and invasive species, new approaches for Landsat analysis need to be developed and standardized. The extent and resolution of Landsat imagery is well-suited for detecting, exploring, and interpreting broad-scale causal relationships and complex dynamics. Understanding the relationships among multi-scale factors that create patterns of change will make it possible to predict the risks and potential outcomes of land management applications.

REFERENCES

Adams, H. S., and S. L. Stephenson. 2010. Twenty-five years of succession in the spruce-fir forest on Mount Rogers in southwestern Virginia. Castanea 75:205–210.

Agee, J. 1994. An analysis of catastrophic forest disturbance on the Olympic Peninsula. Unpublished report prepared for Rayonier Inc.

Alexander, R. R., R. C. Shearer, and W. D. Shepperd. 1984. Silvical characteristics of subalpine fir. General Technical Report 115. Rocky Mountain Forest and Range Experimental Station. Fort Collins, Colorado.

Allen, C. D., A. K. Macalady, H. Chenchouni, D. Bachelet, N. McDowell, M. Vennetier, T. Kitzberger, A. Rigling, D. D. Breshears, E. H. (Ted) Hogg, P. Gonzalez, R. Fensham, Z. Zhang, J. Castro, N. Demidova, J.-H. Lim, G. Allard, S. W. Running, A. Semerci, and N. Cobb. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management 259:660–684.

Antonova, N., C. Copass, and S. Clary. 2013. Landsat-based monitoring of landscape dynamics in the North Cascades National Park Service complex: 1985-2009. Natural Resource Data Series NPS/NCCN/NRDS-2013/532. National Park Service, Fort Collins, Colorado.

Balch, R. E. 1952. Studies of the Balsam Woolly Aphid, *Adelges piceae* (Ratz.) and Its Effects on Balsam Fir, *Abies balsamea* (L.) Mill. Ottawa, Canada.

Bedwell, J. L., and T. W. Childs. 1943. Susceptibility of whitebark pine to blister rust in the Pacific Northwest. Journal of Forestry 41:904–912.

Bentz, B. J., J. Regniere, M. Hansen, J. L. Hayes, and J. A. Hicke. 2010. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. Bioscience 60:602–613.

Berg, E. E., J. David Henry, C. L. Fastie, A. D. De Volder, and S. M. Matsuoka. 2006. Spruce beetle outbreaks on the Kenai Peninsula, Alaska, and Kluane National Park and Reserve, Yukon Territory: Relationship to summer temperatures and regional differences in disturbance regimes. Forest Ecology and Management 227:219–232.

Bonneau, L. R., K. S. Shields, and D. L. Civco. 1999. A technique to identify changes in hemlock forest health over space and time using satellite image data. Biological Invasions 1:269–279.

Breshears, D. D., N. S. Cobb, P. M. Rich, K. P. Price, C. D. Allen, R. G. Balice, W. H. Romme, J. H. Kastens, M. L. Floyd, J. Belnap, J. J. Anderson, O. B. Myers, and C. W. Meyer. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Sciences of the United States of America 102:15144–8.

Cairns, D. M., C. W. Lafon, J. D. Waldron, M. Tchakerian, R. N. Coulson, K. D. Klepzig, A. G. Birt, and W. Xi. 2008. Simulating the reciprocal interaction of forest landscape structure and southern pine beetle herbivory using LANDIS. Landscape Ecology 23:403–415.

Ciesla, W. M. 2006. Aerial signatures of forest insect and disease damage in the western United States. Forest Health Technology Enterprise Team. FHTET-01-06. USDA Forest Service, Fort Collins, Colorado.

Cohen, W. B., Z. Yang, and R. Kennedy. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 2. TimeSync - Tools for calibration and validation. Remote Sensing of Environment 114:2911–2924.

Crawley, M. J. 2005. Statistics: An Introduction Using R. John Wiley & Sons, Ltd., West Sussex, England.

Diaz, H. F., M. P. Hoerling, and J. K. Eischeid. 2001. ENSO variability, teleconnections and climate change. International Journal of Climatology 21:1845–1862.

Edmonds, R. L., J. K. Agee, and R. I. Gara. 2011. Root diseases. Pages 295–332 Forest Health and Protection. Second Edition. Waveland Press, Inc., Long Grove, Illinois.

Ellison, A. M., M. S. Bank, B. D. Clinton, E. A. Colburn, K. Elliott, C. R. Ford, D. R. Foster, B. D. Kloeppel, J. D. Knoepp, G. M. Lovett, J. Mohan, D. A. Orwig, N. L. Rodenhouse, W. V Sobczak, K. A. Stinson, J. K. Stone, C. M. Swan, J. Thompson, B. Von Holle, and J. R. Webster. 2005. Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Frontiers in Ecology and the Environment 3:479–486.

ESRI. 2012. ArcGIS Desktop: Release 10.1. Environmental Systems Research Institute, Redlands, CA.

Estberg, G. N., S. Norris, and K. M. Hutten. 2008. Year-in-advance prediction of lightning activity for the Pacific Northwest. Northwest Science 82:151–157.

Ettl, G. J., and D. L. Peterson. 1995. Growth response of subalpine fir (*Abies lasiocarpa*) to climate in the Olympic Mountains, Washington, USA. Global Change Biology 1:231–241.

Federal Geographic Data Committee. 2008. National Vegetation Classification Standard (Version 2.0). U.S. Geological Survey, Reston, VA.

Fonda, R. W., and L. C. Bliss. 1969. Forest vegetation of the montane and subalpine zones, Olympic Mountains, Washington. Ecological Society of America 39:271–301.

Hain, F. P., R. G. Hollingsworth, F. H. Arthur, F. Sanchez, and R. K. Ross. 1991. Adelgid host interactions with special reference to the balsam woolly adelgid in North America. *in* Y. N. Baranchikov, W. J. Mattson, F. P. Hain, and T. L. Payne, editors. Forest Insect Guilds:

Patterns of Interaction with Host Trees. USDA Forest Service General Technical Report NE-153: 271-287.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2005. Effects of temperature and precipitation variability on snowpack trends in the Western United States. Journal of Climate 18:4545–4561.

Hayes, J. L., and I. Ragenovich. 2001. Non-native invasive forest insects of Eastern Oregon and Washington. Northwest Science 75:77–84.

Henderson, J. A., D. H. Peter, R. D. Lesher, and D. C. Shaw. 1989. Forested Plant Associations of the Olympic National Forest. USDA Forest Service, Pacific Northwest Region. R6-ECOL Technical Paper 001-88., Portland, Oregon.

Hennon, P., D. Wittwer, A. Johnson, P. Schaberg, G. Hawley, C. Beier, S. Sink, and G. Juday. 2006. Climate warming, reduced snow, and freezing injury could explain the demise of yellow-cedar in Southeast Alaska, USA. World Resource Review 18:427–450.

Hollingsworth, R. G., and F. P. Hain. 1994. Effect of drought stress and infestation by the balsam woolly adelgid (Homoptera: Adelgidae) on abnormal wood production in Fraser fir. Canadian Journal of Forest Research 24:2295–2297.

Johnson, N. E., R. Mitchell, and K. H. Wright. 1963. Mortality and damage to Pacific silver fir by the balsam woolly aphid in southwestern Washington. Journal of Forestry:854–860.

Johnson, N. E., and K. H. Wright. 1957. The balsam woolly aphid problem in Oregon and Washington, Research paper No. 18. Pacific Northwest Forest & Experient Station, USDA Forest Service, Portland, Oregon.

Kailin, J., and L. Nickey. 2005. Olympic National Park Fire Management Plan. USDI, National Park Service, Port Angeles, WA.

Kegley, S. 2006. Western balsam bark beetle ecology and management. USDA Forest Service Pest Pamphlet.

Kennedy, R. E., W. B. Cohen, A. A. Kirschbaum, and E. Haureiter. 2007a. Protocol for Landsat-based monitoring of landscape dynamics of North Coast and Cascades Network Parks: 2-G1. U.S. Geological Survey Techniques and Methods, Reston, VA.

Kennedy, R. E., W. B. Cohen, and T. A. Schroeder. 2007b. Trajectory-based change detection for automated characterization of forest disturbance dynamics. Remote Sensing of Environment 110:370–386.

Kennedy, R. E., Z. Yang, and W. B. Cohen. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 1. LandTrendr — temporal segmentation algorithms. Remote Sensing of Environment 114:2897–2910.

Kennedy, R. E., Z. Yang, W. B. Cohen, E. Pfaff, J. Braaten, and P. Nelson. 2012. Spatial and temporal patterns of forest disturbance and regrowth within the area of the Northwest Forest Plan. Remote Sensing of Environment 122:117–133.

Key, C. H., and N. C. Benson. 2006. Landscape assessment: ground measure of severity, the composite burn index; and remote sensing of severity, the normalized burn ratio. *in* D. C. Lutes, R. E. Keane, J. F. Caratti, C. H. Key, N. C. Benson, S. Sutherland, and L. J. Gangi, editors. FIREMON: Fire Effects Monitoring and Inventory System. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT.

Logan, J. A., J. Régnière, and J. A. Powell. 2003. Assessing the impacts of global warming on forest pest dynamics. Frontiers in Ecology and the Environment 1:130–137.

Van Mantgem, P. J., and N. L. Stephenson. 2007. Apparent climatically induced increase of tree mortality rates in a temperate forest. Ecology Letters 10:909–16.

Mantua, N. J., S. R. Hare, Y. Zhang, J. M. Wallace, and R. C. Francis. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin of the American Meteorological Society 78:1069–1079.

Mass, C. 2008. The weather of the Pacific Northwest. University of Washington Press, Seattle, WA.

McDonald, G. I., and R. J. Hoff. 2001. Blister rust: an introduced plague. Page 328 *in* D. F. Tomback, S. F. Arno, and R. E. Keane, editors. Whitebark Pine Communities: Ecology and Restoration. Island Press, Washington, D.C.

McManamay, R. H., L. M. Resler, J. B. Campbell, and R. A. McManamay. 2011. Assessing the impacts of balsam woolly adelgid (*Adelges piceae* Ratz.) and anthropogenic disturbance on the stand structure and mortality of Fraser fir [*Abies fraseri* (Pursh) Poir.] in the Black Mountains, North Carolina. Castanea 76:1–19.

Meigs, G. W., R. E. Kennedy, and W. B. Cohen. 2011. A Landsat time series approach to characterize bark beetle and defoliator impacts on tree mortality and surface fuels in conifer forests. Remote Sensing of Environment 115:3707–3718.

Mitchell, R. G. 1967. Abnormal ray tissue in three true firs infested by the balsam woolly aphid. Forest Science 13:327–332.

Mitchell, R. G., and P. E. Buffam. 2001. Patterns of long-term balsam woolly adelgid infestations and effects in Oregon and Washington. Western Journal of Applied Forestry 16:121–126.

Mitchell, S., and N. Lanquaye-Opoku. 2007. Windthrow hazard mapping using GIS, Washington State Trust Lands. Report for Washington State Department of Natural Resources, Olympia, WA.

Mote, P. W. 2003. Trends in snow water equivalent in the Pacific Northwest and their climatic causes. Geophysical Research Letters 30:1–4.

Pearson, K. 1896. Mathematical contributions to the theory of evolution. III. Regression, heredity and panmixia. Philosophical Transactions of the Royal Society of London Ser. A 187:253–318.

Pickford, S. G., G. R. Fahnestock, and R. Ottmar. 1980. Weather, fuel, and lightning fires in Olympic National Park. Northwest Science 54:92–105.

Puritch, G. S., and J. A. Petty. 1971. Effect of balsam woolly aphid, *Adelges piceae* (Ratz.), infestation on the xylem of *Abies grandis* (Doug.) Lindl. Journal of Experimental Botany 22:946–952.

Raffa, K., B. Aukema, B. J. Bentz, A. L. Carroll, J. A. Hicke, M. G. Turner, and W. H. Romme. 2008. Cross-scale drivers of natural disturbances prone to anthropogenic amplification: the dynamics of bark beetle eruptions. BioScience 58:501–517.

Stehn, S. E., M. A. Jenkins, C. R. Webster, and S. Jose. 2013. Regeneration responses to exogenous disturbance gradients in southern Appalachian Picea-Abies forests. Forest Ecology and Management 289:98–105.

Sturrock, R. N., S. J. Frankel, a. V. Brown, P. E. Hennon, J. T. Kliejunas, K. J. Lewis, J. J. Worrall, and a. J. Woods. 2011. Climate change and forest diseases. Plant Pathology 60:133–149.

Team, R. D. C. 2014. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Turner, M. G. 1989. Landscape ecology: the effect of pattern on process. Annual Review of Ecology and Systematics 20:171–197.

Weed, A. S., B. J. Bentz, M. P. Ayres, and T. P. Holmes. 2015. Geographically variable response of Dendroctonus ponderosae to winter warming in the western United States. Landscape Ecology 30:1075–1093.

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, and T. W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science 313:940–943.

White, P. S., and S. T. A. Pickett. 1985. Natural disturbance and patch dynamics: an introduction. Pages 3–13 *in* S. T. A. Pickett and P. S. White, editors. Natural Disturbance and Patch Dynamics. Academic Press, Inc., San Diego, CA.

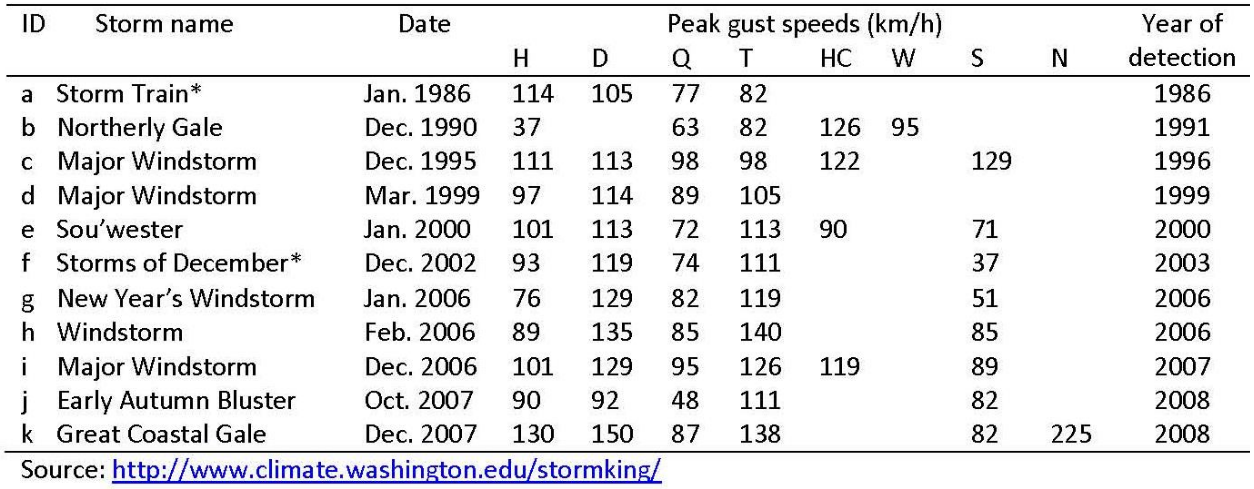
Wickham, H. 2009. ggplot2: Elegant Graphics for Data Analysis. Springer, New York.

Witter, J. A., and I. R. Ragenovich. 1986. Regeneration of Fraser fir at Mt. Mitchell, North Carolina, after depredations by the balsam woolly adelgid. Forest Science 32:585–594.

Woodward, A., C. Torgersen, and J. Chenoweth. 2011. Predicting spread of invasive exotic plants into de-watered reservoirs following dam removal on the Elwha River, Olympic National Park, Washington. USGS Open-File Report 2011-1048.

Wulder, M. A., C. C. Dymond, J. C. White, D. G. Leckie, and A. L. Carroll. 2006. Surveying mountain pine beetle damage of forests: A review of remote sensing opportunities. Forest Ecology and Management 221:27–41.

Table 3.1. Storms of the Olympic Peninsula with a minimum 3-station average peak gust of 95 km/h. Weather stations for wind data collection are depicted in Figure 3.1.



ID	Storm name	Date	Peak gust speeds (km/h)								Year of detection
			H	D	Q	T	HC	W	S	N	
a	Storm Train*	Jan. 1986	114	105	77	82					1986
b	Northerly Gale	Dec. 1990	37		63	82	126	95			1991
c	Major Windstorm	Dec. 1995	111	113	98	98	122		129		1996
d	Major Windstorm	Mar. 1999	97	114	89	105					1999
e	Sou'wester	Jan. 2000	101	113	72	113	90		71		2000
f	Storms of December*	Dec. 2002	93	119	74	111			37		2003
g	New Year's Windstorm	Jan. 2006	76	129	82	119			51		2006
h	Windstorm	Feb. 2006	89	135	85	140			85		2006
i	Major Windstorm	Dec. 2006	101	129	95	126	119		89		2007
j	Early Autumn Bluster	Oct. 2007	90	92	48	111			82		2008
k	Great Coastal Gale	Dec. 2007	130	150	87	138			82	225	2008

Source: http://www.climate.washington.edu/stormking/

*Peak gust speeds for Storm Train represent data from the largest of several storms occurring in the same time period.

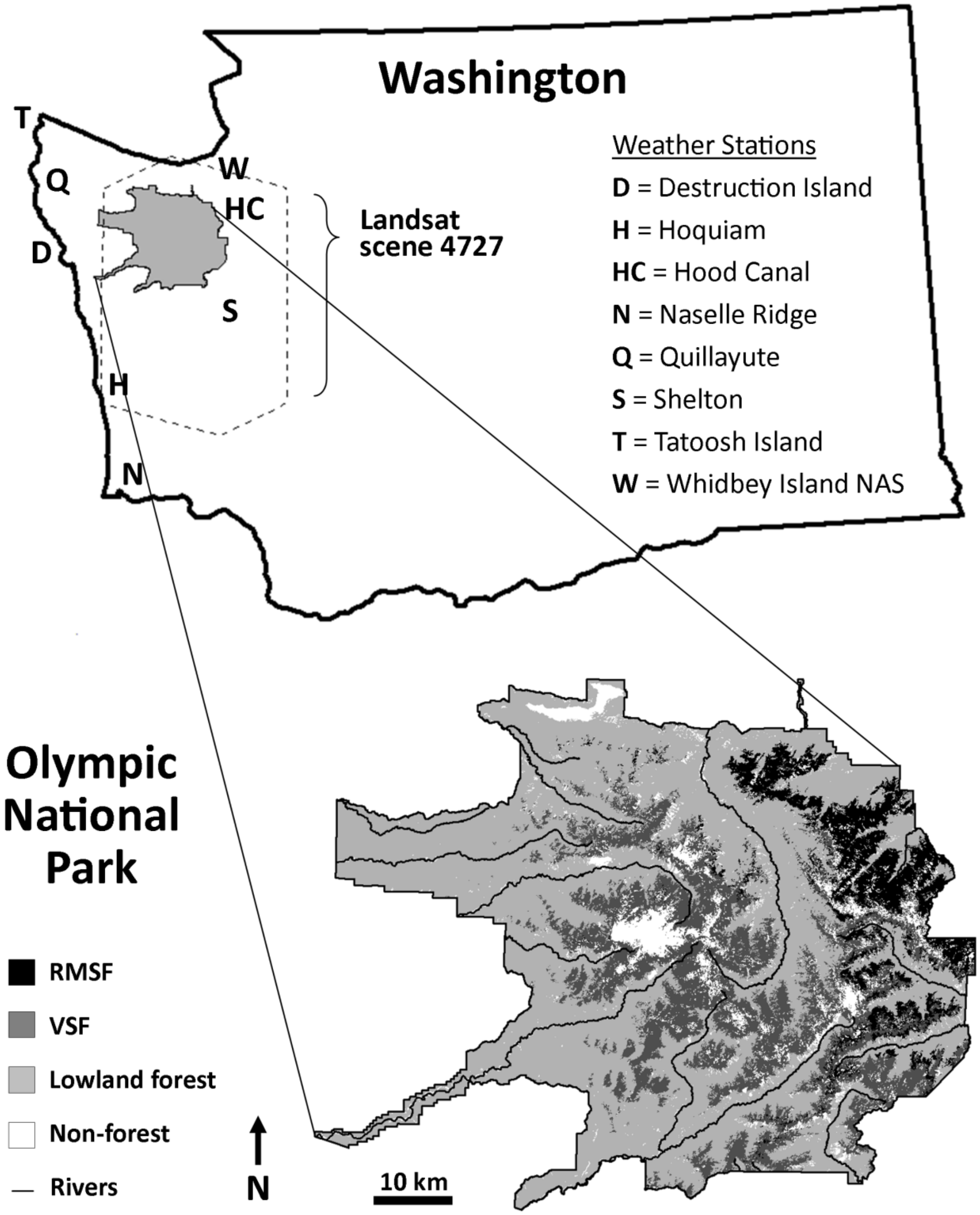


Figure 3.1. Location of Olympic National Park and weather stations in western Washington, USA. Forest vegetation within the boundary of Olympic National Park (lower map) is existing vegetation adapted from the National Vegetation Classification Standard (NVCS) and includes dry, subalpine-fir-dominated Rocky Mountain Subalpine Forest (RMSF) and moist, mountain-hemlock-dominated Vancouverian Subalpine Forest (VSF). Other vegetation macrogroups are simplified here to "Lowland forest" and "Non-forest." Remote sensing area was bound by the intersection of ONP and Landsat scene 4727. Bold letters represent weather station locations where wind data was collected.

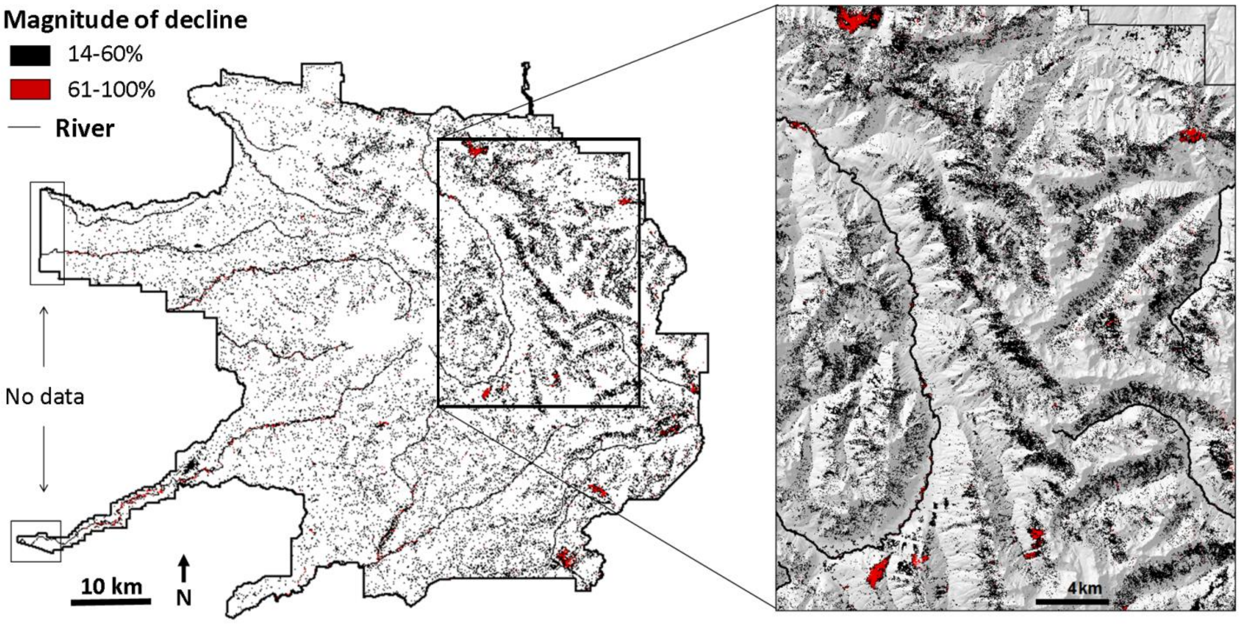


Figure 3.2. Spatial distribution of spectral decline in Olympic National Park identified by LandTrendr for the period 1985-2012. Magnitude (%) is calculated for each pixel from the change in the normalized burn ratio (NBR) index over the decline period divided by pre-disturbance NBR cover values. Types of decline shown include (1) very high-magnitude decline (61-100%; red) which is indicative of fire, extreme blowdown, avalanche, or riparian activity, and (2) low- to high-magnitude decline (14-60%; black) which occurs across Olympic National Park, typically along streams and subalpine slopes (inset).

Figure 3.3. Tree mortality observed along high-elevation ridges of Olympic National Park from fixed-wing aircraft, August 6-7, 2012.

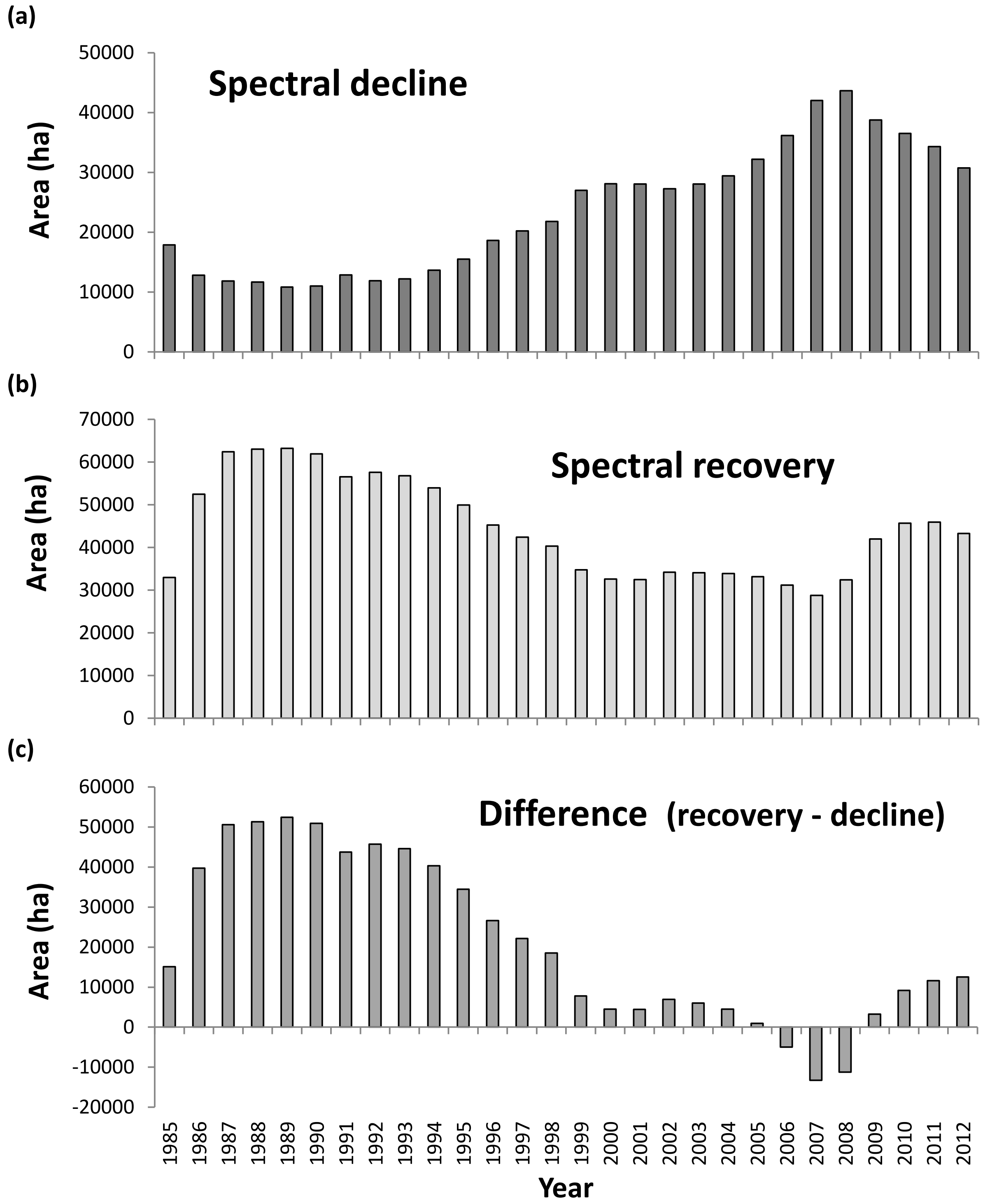


Figure 3.4. Annual spectral change in forested areas of Olympic National Park (ONP) detected by LandTrendr with the normalized burn ratio (NBR) index: (a) total area of spectral decline each year, (b) total area of spectral recovery each year, (c) area in recovery minus area in decline. Totals are for change values (raw NBR*1000) ranging from 4 to 400.

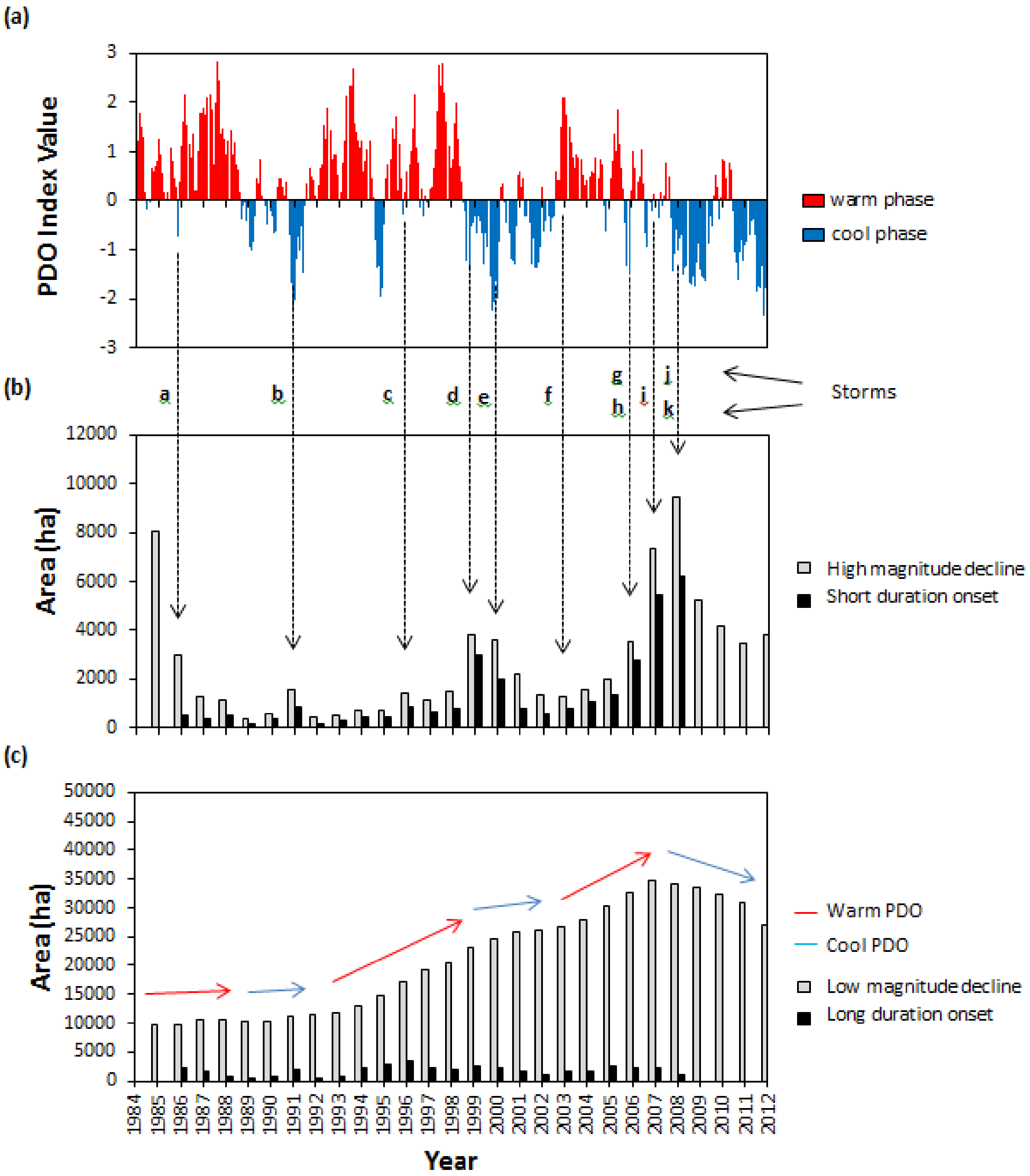
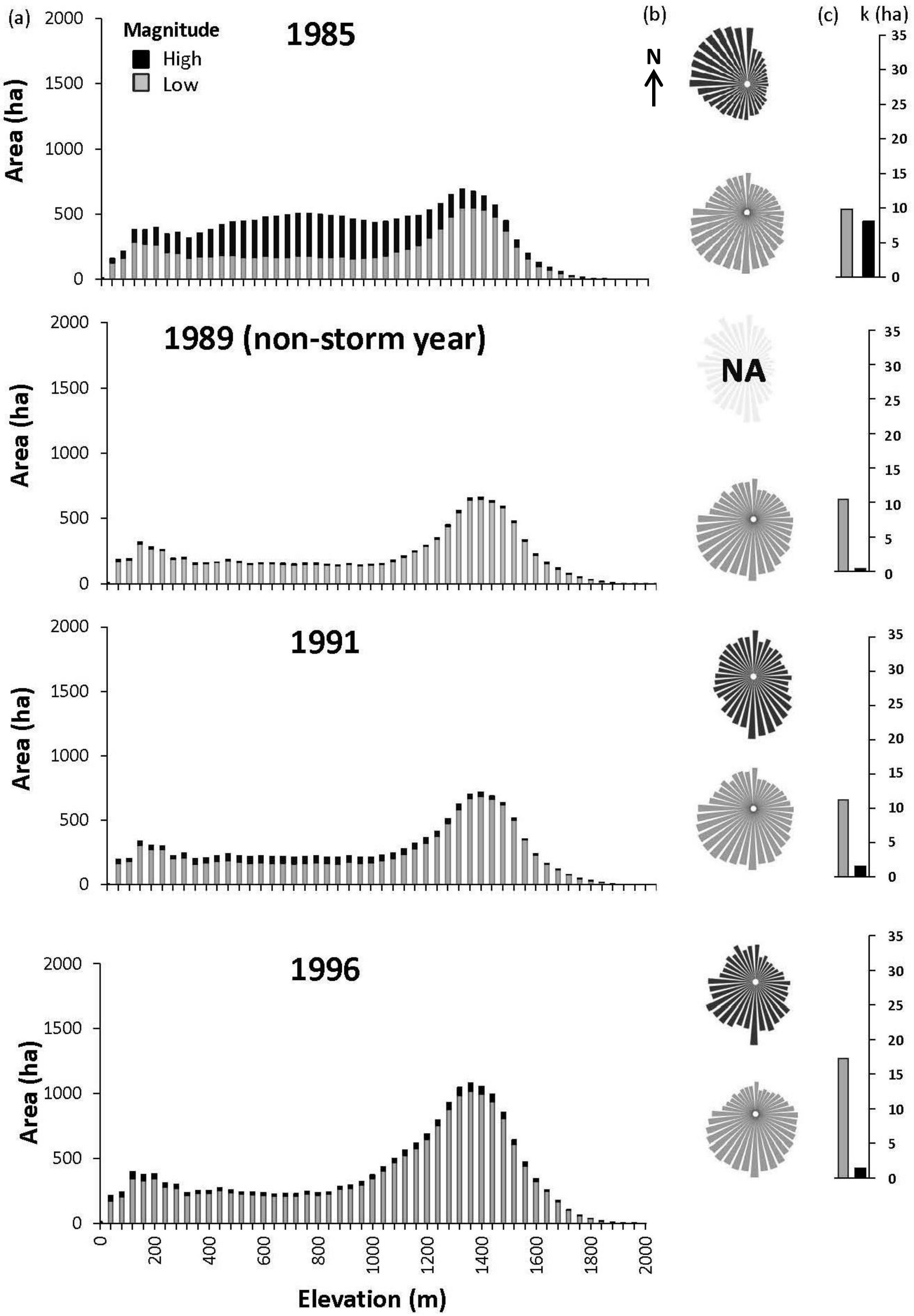


Figure 3.5. (a) Monthly Pacific Decadal Oscillation (PDO) index values in relation to four metrics of forest spectral decline: (b) annual high-magnitude decline and short-duration decline onset, (c) annual low-magnitude decline and long-duration decline onset. Type of onset is identified only for 1986-2008 because 5 years are required to differentiate long- from short-duration decline. Dashed vertical arrows and letter codes in the upper panels relate to storm events (see Table 1).



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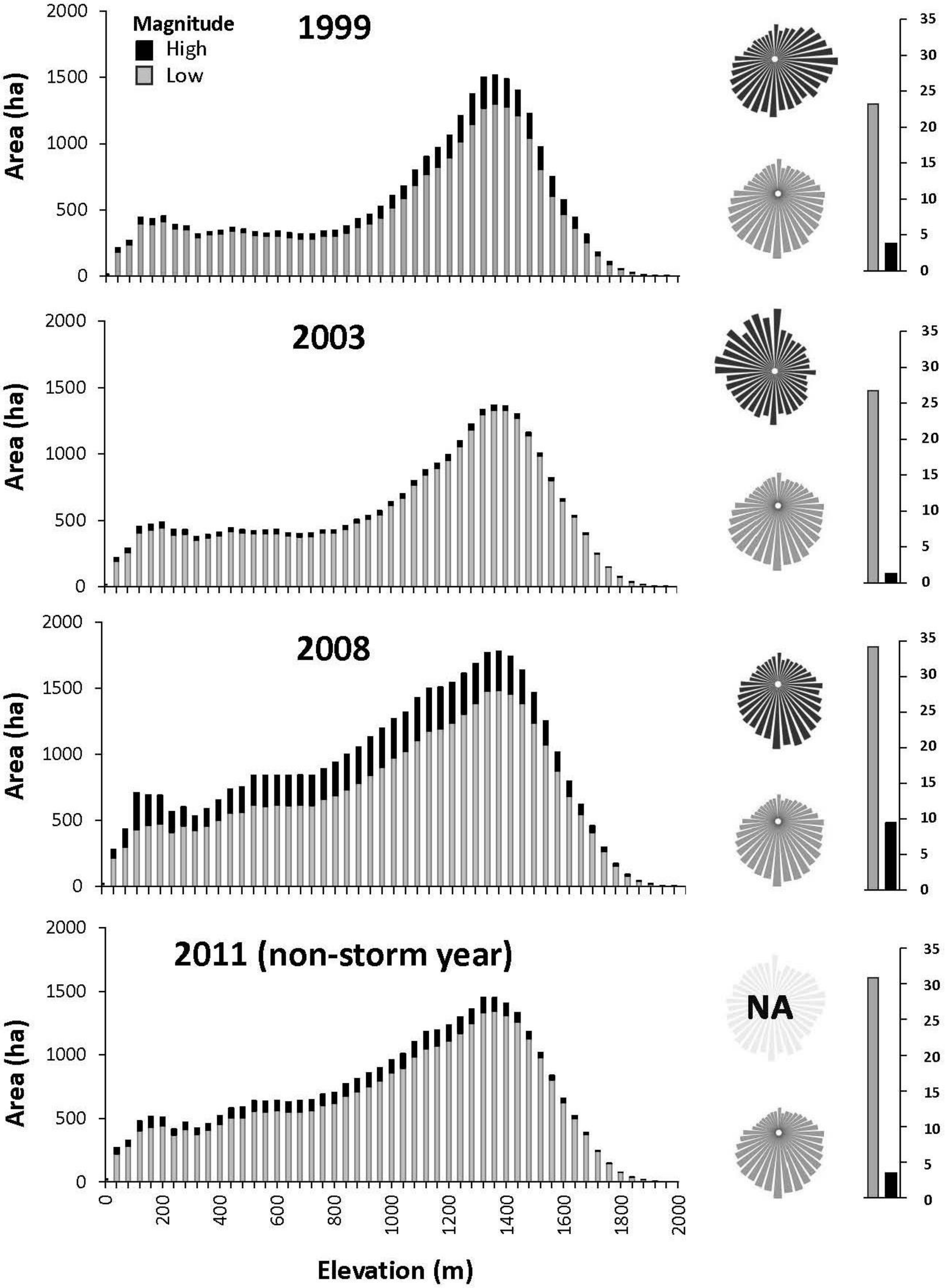


Figure 3.6. Elevation (a) and aspect distribution (b), and total area of low-magnitude (gray) and high-magnitude (black) spectral decline experienced annually (c). Specific years between 1985 and 2011 indicate gradually increasing low-magnitude decline (e.g., insect activity) and punctuated high-magnitude decline associated with winter storm events. NA = not applicable (non-storm year; high magnitude decline is minimal and not likely related to a particular storm).

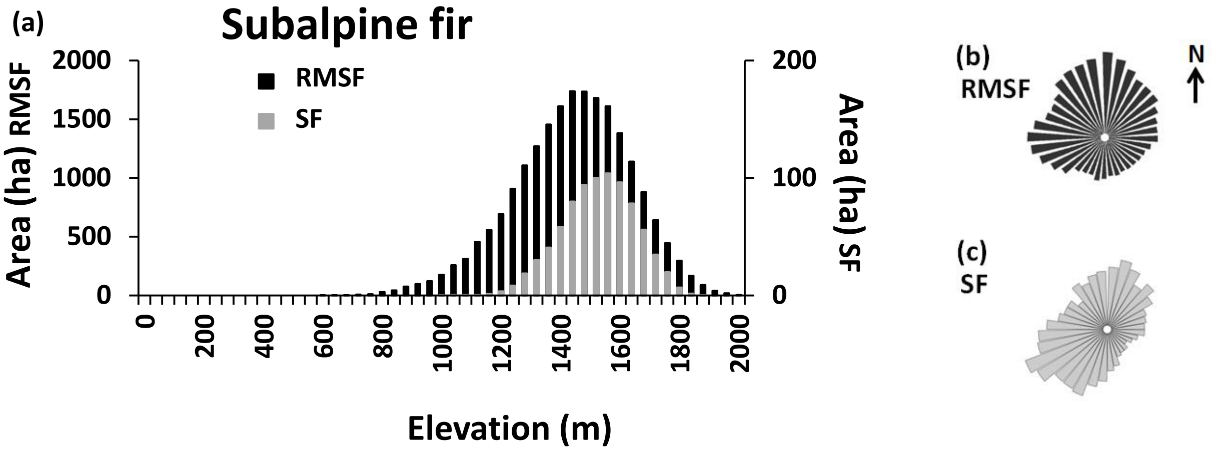


Figure 3.7. Elevation (a) distribution of the Rocky Mountain Subalpine Forest macrogroup (RMSF; black), and Subalpine fir plant association series (SF; gray) within Olympic National Park. Slope aspect distribution for (b) RMSF and (c) SF.

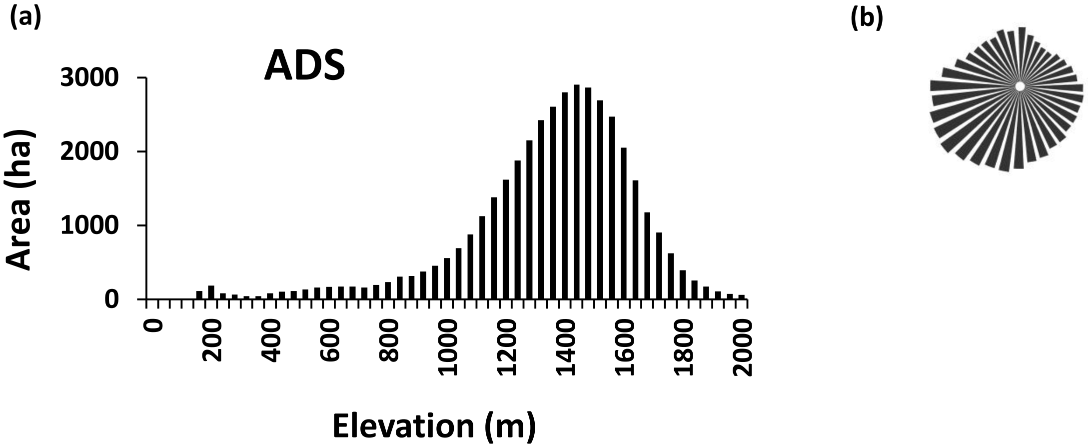


Figure 3.8. Elevation (a) and aspect (b) distribution of USFS/DNR aerial detection surveys (ADS) defoliation and mortality polygons attributed to balsam woolly adelgid (BWA) infestations within Olympic National Park 1955 to 2012.

Chapter 4

DISTRIBUTION OF BALSAM WOOLLY ADELGID AND RELATIONSHIP TO DECLINING HEALTH OF SUBALPINE FIR ON THE OLYMPIC PENINSULA, WA

SUMMARY

The balsam woolly adelgid (BWA), *Adelges piceae*, is an invasive insect that was introduced to North America from Europe in the early 1900s. It feeds on and weakens or kills true fir (*Abies* spp.) and has altered the composition and structure of conifer stands in the Appalachian Mountains and the Cascade Range, USA. The BWA was first detected in subalpine fir of the Olympic Peninsula in 1969, near Mt. Washington, Olympic National Forest. In this study, I investigated the distribution of BWA and the condition of subalpine fir (*Abies lasiocarpa* var. *lasiocarpa*) on the Olympic Peninsula in Washington, USA. I used Landsat imagery and field sampling to compare spectral reflectance, tree mortality, and tree health for plots within subalpine fir forest in and out of BWA-infested stands. I found that the BWA signs and symptoms were widely distributed across subalpine fir forest of the Olympic Peninsula occurring in 90% of subalpine fir sample plots. Landsat reflectance measured as normalized burn ratio (NBR) for plot locations showed a decline trend that became evident in the 1990s and a mean magnitude change of 114 NBR units for subalpine fir with BWA symptoms. In contrast plots with no BWA symptoms or with subalpine fir absent had relatively stable trajectories, and magnitude change did not differ from zero. Basal area (BA) of dead subalpine fir (decay classes 0-2) sampled once during 2010 and 2011 ranged widely in symptomatic plots with a mean of 11.4 m^2ha^{-1} dead trees, whereas BA of dead subalpine fir in non-symptomatic plots did not differ significantly from zero. Tree health based on percentage of dead branches differed among (1)

non-subalpine fir conifers, (2) subalpine fir with no BWA symptoms, and (3) subalpine fir with BWA symptoms ($p < 0.001$; ANOVA) with 13-14% more unhealthy trees in the symptomatic group than in non-symptomatic groups. There was a significant linear relationship between change in magnitude (NBR) and basal area of dead and unhealthy trees ($R^2 = 0.36$, $F_{1,53} = 29.79$, $P < 0.0001$). My results support the hypothesis that BWA infestation is the mechanism causing decline in spectral reflectance (NBR), increased tree mortality, and poor tree health in subalpine forest of the Olympic Peninsula. The correlation between spectral reflectance and basal area of dead and unhealthy trees can be used to estimate the condition of trees within subalpine fir of the Olympic Peninsula.

INTRODUCTION

The balsam woolly adelgid (BWA), *Adelges piceae* (Ratzeburg) (Homoptera: Adelgidae), is an invasive insect that was introduced to North America from Europe in the early 1900s (Johnson and Wright 1957). It feeds on and kills true fir (*Abies* spp.) and has altered the composition and structure of conifer stands in the Appalachian Mountains (Witter and Ragenovich 1986) and the Cascade Range, USA (Mitchell and Buffam 2001). Throughout the Pacific Northwest, BWA has developed a widespread patchy distribution. It moved relatively rapidly across Oregon after the first ground detection in 1930 (Keen 1952), and then into Washington where it was detected near Mt. St Helens in 1952 (Johnson and Wright 1957). The BWA reached subalpine fir (*Abies lasiocarpa* var. *lasiocarpa*) of the Olympic Peninsula, Washington, in 1969, near Mt. Washington of Olympic National Forest. Aerial detection surveys (ADS) now document a wide distribution in subalpine fir across the Olympic Peninsula (USFS et al. 2013). The BWA was observed in Coeur d'Alene, Idaho in 1983 and in Eastern Oregon in 1998 (Overhulser et al. 2004b). Overhulser (2004) noted that BWA has been able to spread across discontinuous fir forests separated by high desert, agricultural land, and stands of non-host conifers, and reached isolated fir trees in predominantly non-host stands. Yet, approximately 50% of fir plots in Oregon still lacked symptoms of BWA nearly 80 years after first detection, not because of tree resistance, but because of the pattern of spread (Overhulser et al. 2004b). Although the insect is wingless, first instar nymphs (crawlers) and eggs are able to disperse over many kilometers via wind or animals (McClure 1990, Balch 1952). This creates localized infestations that increase within a stand and continue to spread between stands.

Damage to trees begins during the feeding process when BWA saliva interacts with host tree vascular tissues, changing the growth of host tree vascular tissue. The BWA inserts long narrow stylets between epidermal cells and into the cortical parenchyma to extract stored nutrients (Balch 1952). In response, the tree grows additional parenchyma cells and short tracheid cells with thick walls (Doerksen and Mitchell 1965). The vascular system becomes inefficient and causes physiological drought stress to the tree that can be increased by environmental conditions (Hollingsworth and Hain 1994).

Ground-based surveys that verify the presence of BWA and document the condition of trees relative to BWA presence or absence aid in the understanding of BWA effects. These surveys also document spatial distribution of BWA and mortality that can be related to patterns in satellite imagery and increase the ability of satellite imagery to provide inference. My objectives were to (1) use ground-based sampling to determine the distribution of BWA signs and symptoms across the Olympic Peninsula, (2) demonstrate the association between the presence of BWA symptoms and subalpine fir tree health and mortality, and (3) determine the relationship between change in spectral reflectance and basal area of dead and unhealthy trees in subalpine fir forest. This research complements Landsat assessment of broad-scale change in

Olympic National Park (ONP) (Chapter 3) by identifying the fine-scale mechanism associated with satellite-derived decline, and also quantifying the relationship between spectral decline and tree health and mortality.

METHODS

Study site

During the summers of 2010 and 2011, I documented BWA in subalpine fir forest (PMR 1996) across ONP and the adjacent Olympic National Forest in Washington (**Fig. 4.1**). Subalpine fir is a dominant species in the study area and co-occurs with lodgepole pine (*Pinus contorta* var. *latifolia*) on dry slopes, mountain hemlock (*Tsuga mertensiana*) on north-facing slopes, Pacific silver fir (*Abies amabilis*) Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) and western hemlock (*Tsuga heterophylla*) at mid elevations, Alaska yellow cedar (*Callitropsis nootkatensis*) on dry sites and along high-elevation riparian areas and avalanche chutes, and very rarely Engelmann spruce (*Picea engelmannii*) in moist areas. Understory shrub species included white rhododendron (*Rhododendron alba*), big huckleberry (*Vaccinium membranaceum*), Alaska huckleberry (*Vaccinium alaskaense*), blueleaf huckleberry (*Vaccinium deliciosum*), bear grass (*Xerophyllum tenax*), Oregon grape (*Mahonia nervosa*), valerian (*Valeriana sitchensis*), and subalpine lupine (*Lupinus arcticus*).

The Olympic Peninsula has a mild maritime climate with high winter precipitation and summer dry period. Average annual mean temperature for the study area is 5.5 ^{o}C with a minimum of -2.8 ^{o}C and a maximum in August of 18.9 ^{o}C (30-year average temperature data extracted to 500 random points for ONP at 960-1900 m elevation; PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, accessed 23 Apr 2015). Micro-climatic conditions vary due to steep topography and variations in slope and aspect. The Olympic

Peninsula has a strong orographic precipitation gradient where moisture moving off the Pacific Ocean is released on the west slope of high-elevation ridges creating dry almost continental conditions on the east-side (Henderson et al. 1989). Annual precipitation within the subalpine fir vegetation type ranges from 500 cm or more in the west to 250 cm on the east side of the Peninsula (PRISM Climate Group). Due to adiabatic cooling, about 60% of winter precipitation falls as snow at high elevations and creates a moisture reservoir that distributes water to lower elevations during the spring and summer. A pattern of increasingly early snow-melt in recent years may be the result of climate change (Mote 2003, Hamlet et al. 2005). Soils on the Olympic Peninsula are derived from sedimentary rocks, marine basalts, and glacial sediments, and are relatively young and infertile. Inceptisols and entisols are the most common soil orders and depth ranges from exposed bedrock and thin regolith on upper slopes to deep colluvial regolith collecting on lower slopes (Henderson et al. 1989).

Study design

Subalpine fir was the dominant forest type in the study area (PMR 1996). Pacific Meridian Resources created maps of existing vegetation using Landsat Thematic Mapper (TM) satellite imagery and field data. I placed random sampling points within 750 m from established trails and routes in subalpine forests with at least 25% tree canopy cover. Additional plot selection criteria included a minimum distance of 50 m between sampling points and safe access. To ensure representation of both infested and uninfested forests, randomization was stratified to generate points in each of three categories: (1) ADS BWA polygons, (2) forest stress and disturbance polygons identified by “Landsat-based detection of trends in disturbance and recovery” (LandTrendr), and (3) forest without remotely detected stress or disturbance. The stratification was applied to each of several sampling areas which were determined by accessibility for week-long sampling periods. I sampled 104 plots at 10 locations (**Fig. 4.1**), but some categories were under-represented in a sampling period if they were unavailable or inaccessible due to steep topography.

LandTrendr layers and change in magnitude

LandTrendr raster data sets of satellite imagery were used for observing change in spectral reflectance of forest vegetation over time (Kennedy et al. 2010). The raster data sets were created from a 28-year time series (1984-2012) of Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) images for scene 4727 (WRS-2 path 47, row 27) with a spatial resolution of 30 x 30 m. Image dates were primarily from July and August when clouds were minimal in the region but forest vegetation was maximized. Images were processed with USGS L1T LPGS, converted to surface reflectance, cloud-screened, normalized (MADCAL), and converted to normalized burn ratio (NBR) for observing vegetation relevant values (see Kennedy et al. 2007, 2010, 2012). The NBR is an effective index for detecting change in forest vegetation (Cohen et al. 2010); the index is calculated from near infrared (NIR) and shortwave infrared (SWIR) bands (NBR = (Band 4 – Band 7) / (Band 4 + Band 7)) (Key and Benson 2006). The original NBR values were multiplied by 1000 and rounded to the nearest integer to reduce image bit size. They are hereafter referred to as NBR or NBR units.

For this research I used (1) unfitted reflectance values for each year and pixel to observe raw time series averaged for three species-symptom categories, and (2) reflectance values that had been fitted using a change point segmentation process to identify long-term trends and calculate magnitude change. Before applying the segmentation process, the original spectral data were filtered to remove anomalous points generally associated with clouds or cloud shadows that were not properly screened during image pre-processing. Anomalies were identified by an algorithm that searches for data points that are unusually different then their immediate neighbors in time. The identifying attribute for these points is the ratio between the difference in spectral value of the neighbors around a given point and the difference between the target point and each neighbor. A threshold rule was set and each point that violated the rule was considered anomalous and given the mean value of immediate neighbors. This initial spectral filter helped ensure that the spectral temporal fitting procedure “ecp” did not falsely identify anomalous points as trajectory change points.

Segmentation of image pixel spectral time series was performed using the R (R Development Core Team 2014) package “ecp” (James and Matteson 2013, Matteson and James 2014) for estimating change points. The ecp package performs multiple change point analysis of multivariate time series data. The divisive approach in the software was used to automatically identify change-points in the image pixel time series sets. The analysis identified the years where trends in spectral trajectory showed a robust, consistent change. This information was then used to reconstruct the spectral time series into linear segments by interpolating the values between the identified breakpoints. This procedure simplified the spectral signal into a noise-free profile that represented the dominate decline trend. From these data, the magnitude and length of each spectral-temporal segment was calculated.

Field data collection

Within each plot I recorded geographic coordinates, elevation, plant association (Crawford et al. 2009), stand height and age, slope, aspect, evidence and severity class of biotic and abiotic damage, general stand health, and presence/absence of subalpine fir > 12.7 cm dbh. For all plots containing subalpine fir > 12.7 cm dbh, tree data were collected including species, diameter at 1.37 m above ground, status (dead or alive), crown class (e.g., overtopped, intermediate, co-dominant), cone-bearing status, and signs and symptoms of insects, disease, and mechanical damage with damage location and percent damage. The number of live seedlings and saplings (> 15 cm tall and < 12.7 cm dbh) was noted for each species, and the number with BWA sign or symptoms was also noted.

Each tree was examined for balsam woolly adelgid signs and symptoms. Insect observations were recorded as BWA present or undetected for each tree. Insects were visible if they had produced wax threads and they were differentiated from small lichens with a hand lens as needed. Symptomatic enlargement of branch nodes (gout) indicative of BWA feeding was recorded as present or absent for each tree. Plots that had BWA insects but lacked gout symptoms did not show health decline and were presumed to be recently infested. Bark beetle activity was determined and identified by presence of diagnostic galleries (Goheen and Willhite 2006).

Live trees were given a health class rating of 1 (0-5% dead branches), 2 (6-50% dead branches), and 3 (51-99% dead branches). Dead trees were noted if standing or fallen and given a decay class rating of 0 (needles brown but present on branches), 1 (needles gone and fine branches present), 2 (most fine branches gone and bark still present but detached in places), 3 (secondary branches mostly gone, branch stubs solidly attached, and bark mostly gone), 4 (bark gone, branch stubs pull out, and stem decaying but still holding form), or 5 (well decayed and snag or log not holding form).

Statistical analysis

Plots were sorted into three groups by tree species and presence of BWA gout symptoms: (1) no subalpine fir, (2) subalpine fir without gout, and (3) subalpine fir with gout. Spectral time series and change in spectral-reflectance magnitude were compared among the three groups. A student’s t-test (Crawley 2005) was conducted in R (R Developent Core Team 2014) to determine if magnitude change in NBR for each group differed significantly from zero (p-value for significance = 0.05). Difference in variance precluded use of analysis of variance (ANOVA) or paired t-tests.

Basal area of dead subalpine fir trees was compared between plots in two groups sorted by BWA gout symptoms: (1) subalpine fir without gout, and (2) subalpine fir with gout. Only dead trees in decay classes 0 through 2 were considered because they represent recent mortality. Several trees in each decay class 0-2 showed gout symptoms on remaining branches, and age of death for trees of decay class 2 was within the time range of BWA introduction (K. Hutten, unpublished data). A student’s t-test was conducted in R to determine if basal area of dead trees for each group differed significantly from zero (p-value for significance = 0.05). Difference in variance among groups precluded use of ANOVA or paired t-tests. For both spectral change and mortality analyses, plots in any category that showed evidence of severe wind damage (i.e., abrupt change in spectral reflectance, see Chapter 3) were removed from the analysis.

Live tree health (classes 1-3) was assessed for individual trees and tallied as class counts and percentages among three groups: (1) non-subalpine fir conifer species; (2) subalpine fir with no gout symptoms; and (3) subalpine fir with gout symptoms. Pearson’s chi-square test was conducted in R to test for differences in proportions of tree health classes among groups.

Simple linear regression was used to determine the relationship between spectral magnitude change (NBR*1000) and basal area (m^2ha^{-1}) of dead (decay classes 0-2) and unhealthy trees (health classes 2-3). Dead trees included all tree-sized conifer species (as observed in satellite imagery), and unhealthy trees included only subalpine fir which was the predominant species experiencing health decline in study plots.

RESULTS

Distribution of balsam woolly adelgid in Olympic National Park

The balsam woolly adelgid is widely distributed across the subalpine fir region of the Olympic Peninsula (**Fig. 4.1**). Of 104 plots established, subalpine fir grew in 71 plots. Regarding plots with subalpine fir, signs or symptoms of BWA were documented in 90% of plots, 79% of plots had gout symptoms; 46% had observable signs of BWA; 7% had signs of BWA but no gout symptoms; and 10% were uninfested with BWA.

Spectral reflectance and magnitude change among plots grouped by presence of subalpine fir and BWA symptoms

Time series of spectral reflectance (NBR) for plots differentiated into three groups based on species and BWA symptoms showed that plots (1) without subalpine fir and (2) with subalpine fir but without gout remained relatively stable through time in comparison with (3) those having subalpine fir and gout (**Fig. 4.2**). Spectral reflectance for plots with subalpine fir and gout began to differ from the other groups in 1994, and a consistent decline became apparent after 1998.

Spectral magnitude change (NBR*1000) calculated for the three categories was not different from zero for plots without subalpine fir ($t = -1.01$, $df = 24$, $p = 0.32$) or plots with subalpine fir and without gout ($t = -0.90$, $df = 9$, $p = 0.39$) (**Fig. 4.3**). For plots with subalpine fir and gout, change in magnitude ranged widely but was greater than zero with a mean change of 114 ($t = -9.57$, $df = 45$, $p < 0.0001$).

Tree mortality and tree health among groups and related to spectral magnitude change

Basal area of dead subalpine fir (decay classes 0-2) in plots without gout symptoms did not differ from zero ($t = 1.35$, $df = 6$, $p = 0.23$). Basal area of dead subalpine fir in plots with gout symptoms ranged widely but was greater than zero ($t = 6.40$, $df = 41$, $p < 0.0001$) with a mean of 11.4 m^2ha^{-1} (**Fig. 4.4**). Non-symptomatic subalpine fir plots had 4% mortality, and symptomatic plots had 27% when proportion of dead to total (live and dead) subalpine fir basal area was considered.

Pearson's chi-square test for difference indicated that tree health class proportions differed among species-symptom groups separated by presence or absence of subalpine fir or BWA symptoms ($p < 0.001$) (**Table 4.1**). Non-subalpine fir conifer species and subalpine fir without gout had similar proportions of healthy (class 1) and unhealthy (classes 2 and 3) trees with the majority of trees in health class 1 (61-62%). Subalpine fir trees with gout had less than 50% healthy trees and more unhealthy trees than the other two groups by 12-14%.

There was a significant linear relationship between the response variable: change in magnitude (NBR units), and the predictor variable: basal area of dead and unhealthy trees ($R^2 = 0.36$, $F_{1,53} = 29.79$, $p < 0.0001$) (**Fig. 4.5**).

DISCUSSION

Distribution of balsam woolly adelgid in subalpine fir of the Olympic Peninsula

Since its introduction to subalpine fir of the Olympic Peninsula in 1969, BWA has spread across the distributional extent of subalpine fir. Field surveys detected few uninfested stands despite the difficulty of detecting incipient BWA populations. The BWA will likely spread to uninfested areas in time. Aerial detection surveys (ADS) reported low-level severity defoliation in areas of subalpine fir followed by patches of increasing severity, extent, or spatial frequency.

With successful establishment at each new location BWA spreads outward to surrounding trees and locally intensifies over several years.

Field validation of BWA presence across the Olympic Peninsula confirmed ADS results. I also observed signs and symptoms in areas not documented by ADS that are likely to become more evident in future surveys.

Spectral reflectance and magnitude change among plots grouped by presence of subalpine fir and BWA symptoms

Spectral reflectance and magnitude change in NBR in plots that lacked BWA symptoms or were dominated by species other than subalpine fir were stable or close to zero, indicating a healthy forest canopy. In contrast, symptomatic subalpine fir plots showed a wide range of spectral change that represented a decline in health overall, although some plots showed growth. The spectral response detected with Landsat imagery may depend on the proportion of subalpine fir in the area, or the duration and severity of infestation. Differences in severity have been related to tree susceptibility and duration of infestation and are discussed in relation to tree mortality below.

Single-year declines in spectral reflectance may be a response to winter weather events that are associated with short-term decreases in NBR (see Chapter 3). Other interannual variability is likely noise that resulted from seasonal fluctuations in vegetation growth (phenology), differences in atmospheric moisture, sun angle, or issues related to geometric registration (alignment of imagery) (Kennedy et al. 2007b, 2010, 2012). However, the spectral response in symptomatic subalpine fir was different because NBR did not recover completely after each short-term decrease, resulting in an overall long-term decline that became evident sometime in the 1990s. The spectral time series for symptomatic and non-symptomatic subalpine fir began to diverge as early as 1994, perhaps even as early as 1987 with less certainty. Although BWA was detected at low levels in subalpine fir of ONP in 1970, the effects were not widely distributed in aerial surveys until 1989, and likely with a patchy distribution. Because the decline of tree health in BWA branch-infested trees occurs over several years, the change is not immediately evident in satellite imagery. Insects may need to reach a threshold population and level of activity before change in satellite imagery surpasses spectral noise. Infested trees are physiologically stressed by altered cell growth and decreased vascular function (Mitchell 1967, Hain et al 1991), thus branch-infested trees experience subtle needle loss that appears as reducee reflectance but not as abrupt decline. Additional periodic stresses may increase needle loss and detection in satellite imagery, e.g., tree damage, late snow melt, or drought that reduce the ability of the tree to compensate for reduced vascular capacity with the growth of new tissue.

Tree mortality and tree health among plots grouped by presence of subalpine fir and BWA symptoms

Field data relating tree health and mortality to presence of BWA symptoms provided a more direct indication of the effect of BWA on subalpine fir than could be determined from remotely sensed data. Where BWA symptoms were not observed the trees were healthier (**Table 4.1**) and tree mortality was relatively low (**Fig. 4.4**). Sample plots with BWA symptoms had 11.4 m^2ha^{-1} dead subalpine fir in decay classes 0-2, and were estimated to represent trees that died during the BWA infestation time period. This is equivalent to 180 trees ha^{-1} with average dbh for the study area of 28.4 cm. However, the range in BA of mortality was large (0-41 m^2ha^{-1} dead subalpine fir) possibly because of differences in susceptibility or duration of infestation. I conducted exploratory analyses that indicated mortality may be greater in older, less dense, and faster growing stands (K. Hutten unpublished data). Aspect and elevation did not appear to have an influence but this study was not designed to test this. The relative importance of these and other factors may change with location and time period.

Other studies have shown that elevation (lower temperature), poor or shallow soil (lower moisture) and access to cortical tissue via bark fissures (i.e., older and faster growing trees) may be three important susceptibility factors. For example, Mitchell and Buffam (2001) observed that trees were more likely to die quickly at lower elevations than higher elevations, and they hypothesized that this was because colder temperatures limited insect development and survival. Quiring et al. (2008) found that BWA did not occur where January mean temperature was below −11 °C in Atlantic, Canada. However, on the Olympic Peninsula, January mean minimum temperature is − 6 °C at the highest elevation (30-year average temperature data; PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, accessed 23 Apr 2015).

In Fraser fir forests, infested trees were more stressed and mortality greater where moisture was limited by poor soils, such as in the Black Mountains and the Great Smokey Mountains of North Carolina (Hollingsworth and Hain 1994). Trees growing at Mount Rogers, Virginia where soils were deeper experienced less mortality initially; however, many trees died over time (Adams and Stephenson 2010) indicating that mortality may have been delayed. McManamay et al. (2011) found that larger Fraser fir trees were more likely to die from BWA than smaller trees possibly because the bark of young trees is smooth and difficult for BWA to penetrate. Although Fraser fir mortality was greater at higher than lower elevations in the Black Mountains, the trees there were also larger (McManamay et al. 2011). Dense stands would be less susceptible to BWA if they also happen to include younger trees with smooth resistant bark.

Duration of infestation may explain differences in mortality. BWA may spread, establish, and affect tree health slowly within an infested stand. Trees on edges of forest openings may intercept wind or animals carrying BWA and be the first to become infested but these patterns are difficult to determine with sampling at a single point in time or in stands that are already infested. In areas with stem infestations researchers have examined tree cores to identify the year that BWA began feeding on a tree by the occurrence of dense, red cells called rotholz that form in response to contents in BWA saliva (e.g., Kanoti 2006). Where stem infestations are not common (e.g., the OP), estimates of duration may be determined from the length of spectral change trends detected in satellite imagery. Spectral trends would likely be shorter than the actual period of BWA infestation because satellite imagery detects visible change in vegetation rather than presence of BWA (see Chapter 5).

Relationship between spectral decline magnitude and subalpine fir health and mortality

The linear relationship between spectral change in NBR and tree condition (mortality and poor health) offers a means to estimate tree condition in other subalpine fir stands of the Olympic Peninsula (**Fig. 4.5**). This may aid monitoring efforts. Basal area of trees was used because it is more comparable to satellite imagery than number of trees which varies in spatial coverage depending on tree size. Average diameter may be used to estimate the number of dead trees. Dead and unhealthy trees were included in this assessment because both affect change in spectral reflectance. Many unhealthy trees are expected to eventually die from BWA infestation-related stress (~50% in Mitchell, 2001). Similarly, Meigs et al. (2011) found a positive linear relationship between magnitude change (NBR units) and dead standing dead trees in the Oregon Cascade Range (USA) with $R^2 = 0.40$.

Basal area of dead (decay class 0-2) and unhealthy (health class 2-3) trees explained 36% of the variance in spectral magnitude change (NBR units). The low correlation between change in magnitude and tree condition may be caused by three factors. First, heterogeneity of forest structure inherently affects the way that dead and dying trees are detected by satellite imagery. Death of dominant trees is more easily detected than death of intermediate or overtopped trees. Second, decay classes 0 through 2 provided the best correlation and estimate of trees that have died since BWA introduction which is reflected in recent NBR decline, but some decay class 2 trees may have died prior to Landsat coverage. Third, field measurements of dead trees in 10-m radius plots sampled a small area relative to Landsat resolution of 30 x 30 m pixels and the sampled area may cover parts of more than one pixel.

Complementary use of Landsat imagery and ADS

The results of this study indicate that BWA is the primary cause of decline in the health of subalpine fir trees on the Olympic Peninsula. This confirms ADS observations and informs broad-scale spectral change patterns detected with Landsat imagery (Chapter 3). However, the field work portion was time and labor intensive and many areas were not accessible due to remoteness and steep terrain. It is possible to simultaneously increase the spatial and temporal frequency and scope of disturbance research and field visitation efficiency by employing and integrating the use of Landsat imagery and ADS in innovative ways. These complementary methods may be combined to describe patterns of progression and impact associated with specific disturbance agents. Field sampling may be strategically targeted to validate remotely sensed observations.

Landsat and ADS each have strengths and limitations. Landsat imagery is available from 1972 (Multispectral Scanner) and 1984 (TM and ETM+) and gives researchers the ability to revisit time-sensitive data, check assumptions, make new observations and corrections, and fine-tune change detection methods. However, Landsat-based change detection does not differentiate among disturbance agents unless attributes of magnitude, duration, and distribution (e.g.

elevation and vegetation type) are known. In contrast, ADS uses high resolution, real-time methods (human eyesight) to differentiate disturbance patterns by identifying tree species and aerial signatures of different disturbance agents, and effectively detecting early onset patterns. However, documentation on maps does not retain the detail seen during the survey flight and these details cannot be revisited to check the information. Accurate historic trends or time series are difficult to produce when (1) aerial signatures (Ciesla 2006) for new agents are not recognized in early years, (2) timing of symptom expression does not match the flight date, or (3) a subtle or gradual progression of decline is not consistently recorded.

A combined approach uses ADS polygons to identify areas of known disturbance within Landsat imagery. Then spectral attributes (magnitude, onset and duration of decline, and onset of recovery) can be determined with Landsat imagery (e.g., LandTrendr) and used to identify temporal trends for a given area or disturbance agent. Understanding trends and cycles of decline and recovery associated with a particular agent is useful for investigating ecological relationships and dynamics, assessing impacts, determining management actions, and monitoring for change. Attributes identified for a particular disturbance agent also may be applied to enhance spatial analysis with Landsat imagery. This combined approach using ADS and Landsat imagery may be very powerful for monitoring spatial patterns of landscape disturbance but it has not yet been widely applied (Meigs et al. 2011).

CONCLUSION

Comparisons of species-symptom groups showed that plots and trees with BWA gout symptoms consistently had higher mortality and poorer tree health than plots and trees without gout or without subalpine fir. These results support the hypothesis that BWA infestation is the likely mechanism for increased subalpine fir mortality and poor tree health detected in field surveys and in Landsat imagery. Furthermore, change in spectral magnitude (NBR) may be used to estimate the basal area of dead and unhealthy trees in subalpine fir forest of the Olympic Peninsula. Subalpine fir is expected to continue to decline in health, with potential for recovery in some areas (e.g., Mitchell and Buffam 2001).

REFERENCES

Adams, H. S., and S. L. Stephenson. 2010. Twenty-five years of succession in the spruce-fir forest on Mount Rogers in southwestern Virginia. Castanea 75:205–210.

Balch, R. E. 1952. Studies of the Balsam Woolly Aphid, *Adelges piceae* (Ratz.) and Its Effects on Balsam Fir, *Abies balsamea* (L.) Mill. Ottawa, Canada.

Ciesla, W. M. 2006. Aerial signatures of forest insect and disease damage in the western United States. Forest Health Technology Enterprise Team. FHTET-01-06. USDA Forest Service, Fort Collins, Colorado.

Cohen, W. B., Z. Yang, and R. Kennedy. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 2. TimeSync - Tools for calibration and validation. Remote Sensing of Environment 114:2911–2924.

Crawford, R., C. B. Chappell, C. C. Thompson, and F. J. Rocchio. 2009. Vegetation classification of Mount Rainier, North Cascades, and Olympic National Parks plant association descriptions and identification keys. National Parks. National Park Service, Fort Collins, Colorado.

Crawley, M. J. 2005. Statistics: An Introduction Using R. John Wiley & Sons, Ltd., West Sussex, England.

Doerksen, A. H., and R. G. Mitchell. 1965. Effects of the balsam woolly aphid upon wood anatomy of some western true firs. Forest Science 11:181–188.

Goheen, E. M., and E. A. Willhite. 2006. Field Guide to the Common Diseases and Insect Pests of Oregon and Washington Conifers. R6-NR-FID-PR-01-06. USDA Forest Service, Pacific Northwest Region, Portland, Oregon.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2005. Effects of temperature and precipitation variability on snowpack trends in the Western United States. Journal of Climate 18:4545–4561.

Henderson, J. A., D. H. Peter, R. D. Lesher, and D. C. Shaw. 1989. Forested Plant Associations of the Olympic National Forest. USDA Forest Service, Pacific Northwest Region. R6-ECOL Technical Paper 001-88., Portland, Oregon.

Hollingsworth, R. G., and F. P. Hain. 1994. Effect of drought stress and infestation by the balsam woolly adelgid (Homoptera: Adelgidae) on abnormal wood production in Fraser fir. Canadian Journal of Forest Research 24:2295–2297.

James, N. A., and D. S. Matteson. 2013. ecp: An R package for nonparametric multiple change point analysis of multivariate data. published online arXiv:1309:1–31.

Johnson, N. E., and K. H. Wright. 1957. The balsam woolly aphid problem in Oregon and Washington, Research paper No. 18. Pacific Northwest Forest & Experient Station, USDA Forest Service, Portland, Oregon.

Kanoti, A. M. 2006. Relationship between balsam woolly adelgid damage, radial growth, climate and stand characteristics in eastern Maine. M.S. Thesis. Paper 434. University of Maine, Orono, Maine.

Kennedy, R. E., W. B. Cohen, and T. A. Schroeder. 2007. Trajectory-based change detection for automated characterization of forest disturbance dynamics. Remote Sensing of Environment 110:370–386.

Kennedy, R. E., Z. Yang, and W. B. Cohen. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 1. LandTrendr — temporal segmentation algorithms. Remote Sensing of Environment 114:2897–2910.

Kennedy, R. E., Z. Yang, W. B. Cohen, E. Pfaff, J. Braaten, and P. Nelson. 2012. Spatial and temporal patterns of forest disturbance and regrowth within the area of the Northwest Forest Plan. Remote Sensing of Environment 122:117–133.

Key, C. H., and N. C. Benson. 2006. Landscape assessment: ground measure of severity, the composite burn index; and remote sensing of severity, the normalized burn ratio. *in* D. C. Lutes, R. E. Keane, J. F. Caratti, C. H. Key, N. C. Benson, S. Sutherland, and L. J. Gangi, editors. FIREMON: Fire Effects Monitoring and Inventory System. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT.

Matteson, D. S., and N. A. James. 2014. A nonparametric approach for multiple change point analysis of multivariate data. Journal of the American Statistical Association 109:334–345.

McManamay, R. H., L. M. Resler, J. B. Campbell, and R. A. McManamay. 2011. Assessing the impacts of balsam woolly adelgid (*Adelges piceae* Ratz.) and anthropogenic disturbance on the stand structure and mortality of Fraser fir [*Abies fraseri* (Pursh) Poir.] in the Black Mountains, North Carolina. Castanea 76:1–19.

Meigs, G. W., R. E. Kennedy, and W. B. Cohen. 2011. A Landsat time series approach to characterize bark beetle and defoliator impacts on tree mortality and surface fuels in conifer forests. Remote Sensing of Environment 115:3707–3718.

Mitchell, R. G., and P. E. Buffam. 2001. Patterns of long-term balsam woolly adelgid infestations and effects in Oregon and Washington. Western Journal of Applied Forestry 16:121–126.

Mote, P. W. 2003. Trends in snow water equivalent in the Pacific Northwest and their climatic causes. Geophysical Research Letters 30:1–4.

Overhulser, D. L., I. R. Ragenovich, M. McWilliams, and E. A. Willhite. 2004. Balsam woolly adelgid occurrence on true fir in Oregon. Oregon Department of Forestry, Salem, OR.

PMR. 1996. Landforms in Olympic National Park (polygon) - Landform classification and mapping - National Park Service Pacific Northwest Region vegetation and landform database Development Study. Pacific Meridian Resources, Portland, Oregon.

Quiring, D., D. Ostaff, L. Hartling, D. Lavigne, K. Moore, and I. DeMerchant. 2008. Temperature and plant hardiness zone influence distribution of balsam woolly adelgid damage in Atlantic Canada. The Forestry Chronicle 84:558–562.

Team, R. D. C. 2014. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

USFS, WDNR, and ODF. 2013. Annual forest insect and disease aerial detection survey. 1955-2012. Forest Health and Protection, State and Private Forestry, Pacific Northwest Region, USDA Forest Service; Washington Department of Natural Resources, Resource Protection Division, Forest Health; Oregon Department of Forestry.

Witter, J. A., and I. R. Ragenovich. 1986. Regeneration of Fraser fir at Mt. Mitchell, North Carolina, after depredations by the balsam woolly adelgid. Forest Science 32:585–594.

Table 4.1. Health classes of live trees in three groups within subalpine fir plots: (1) conifer species excluding subalpine fir (Non SF); (2) subalpine fir without gout symptoms (SF no G); and (3) subalpine fir with gout symptoms (SF with G). Number and percent of trees within each health class are listed. Tree health was significantly different among groups (p < 0.001).

	Number (proportion) of live trees in each defoliated branch class			
	Healthy	← Unhealthy →		
Live tree group	Class 1 (0-5%)	Class 2 (6-50%)	Class 3 (51 to < 100%)	Total
Non SF	267 (0.62)	135 (0.31)	32 (0.7)	434
SF no G	99 (0.61)	49 (0.31)	15 (0.9)	163
SF with G	139 (0.48)	112 (0.39)	38 (0.13)	289

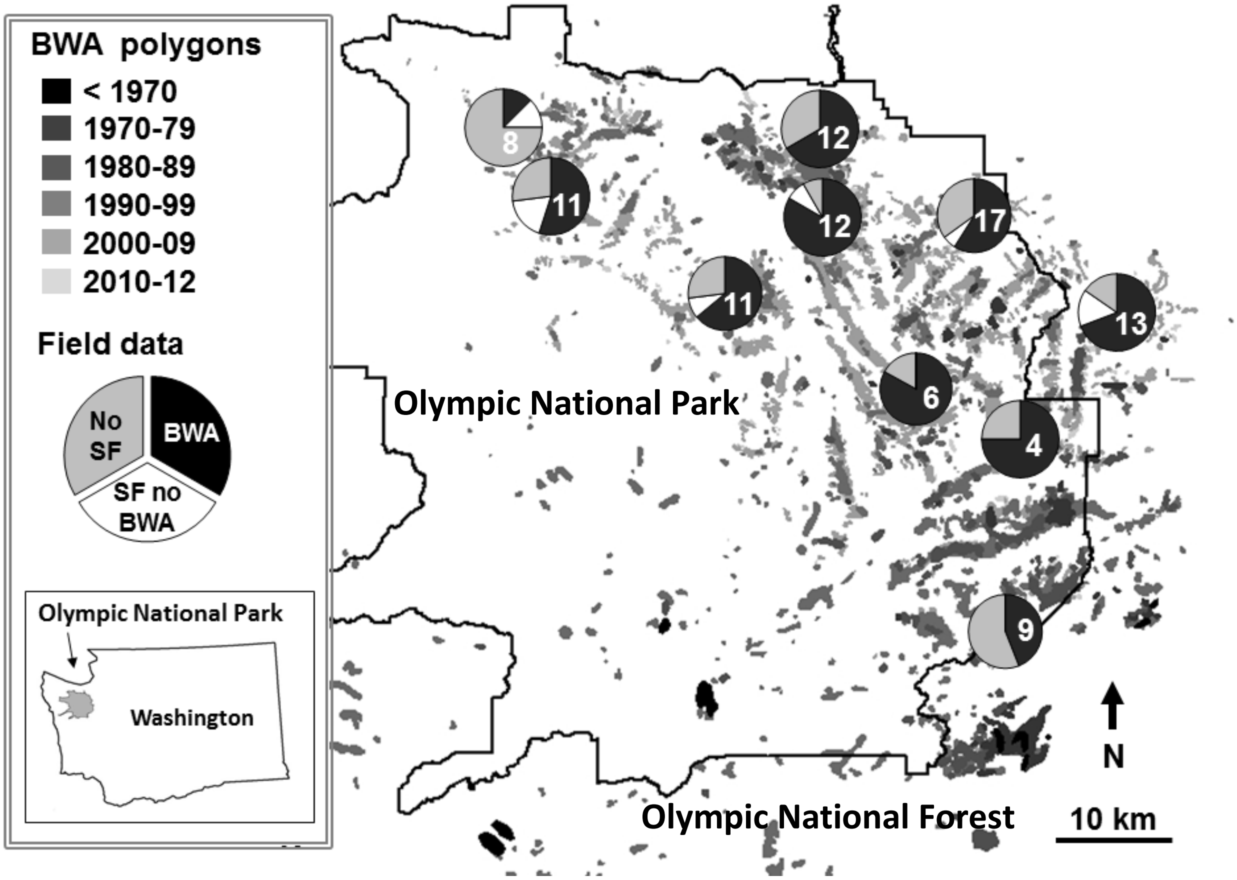


Figure 4.1. Progression and distribution of balsam woolly adelgid (BWA) infestations across the Olympic Peninsula based on aerial detection survey (ADS) polygons and field-sampled plots. Map shading indicates period of ADS detection. Polygons concentrated in east ONP represent infested subalpine fir on high elevation ridges. Polygons in the southwest may represent low-elevation infestations on Pacific silver fir or grand fir. Pie graphs indicate the proportion of field-sampled plots at each site that have (1) subalpine fir (SF) with BWA signs or symptoms (black), (2) SF with no BWA signs or symptoms (white), and (3) conifer species but no subalpine fir or BWA (gray). Number of plots sampled at each site is indicated on pie graphs.

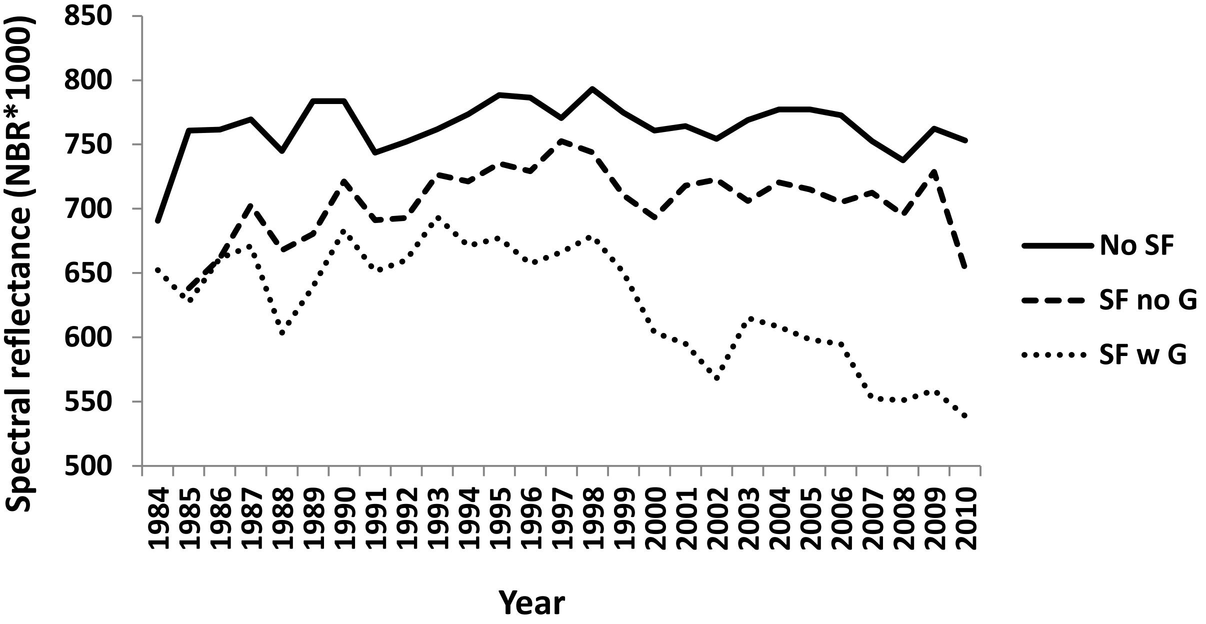


Figure 4.2. Average spectral reflectance for three forest categories from 1984 to 2010: (1) plots lacking subalpine fir and balsam wooly adelgid (BWA; No SF; n = 25); (2) plots with subalpine fir and no BWA gout symptoms (SF no G; n = 10); and (3) plots with subalpine fir and BWA gout symptoms (SF with G; n = 46). Spectral values are normalized burn ratio (NBR) index values multiplied by 1000.

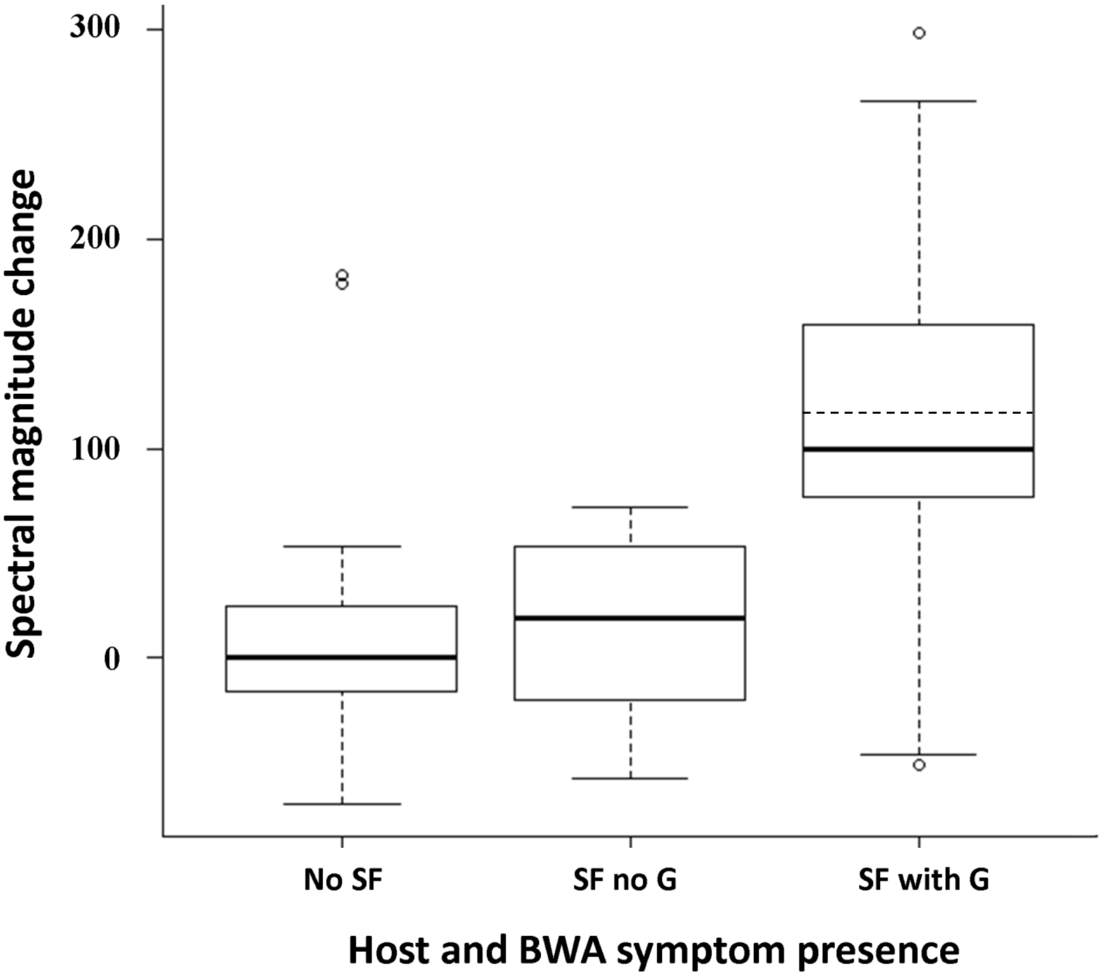


Figure 4.3. Spectral magnitude change (MC) in normalized burn ratio (NBR*1000) calculated for decline trends at each plot for the Landsat time period 1984-2010, and compared among three coniferous forest categories: (1) no subalpine fir and no balsam woolly adelgid (BWA; No SF; n = 25); (2) subalpine fir with no BWA gout symptoms (SF no G; n = 10); and (3) subalpine fir with BWA gout symptoms (SF with G; n = 46). For each box plot the bold center line represents the median value of y, the dashed center line represents the mean, the box represents the middle 50% of the data with lower and upper margins marking the 25th and 75th percentiles, whiskers indicate maximum and minimum values within 1.5 times the interquartile range, and hollow dots represent values greater than 1.5 times the interquartile range. Mean MC was not significantly different from zero for No SF (p = 0.32; student's t-test) and SF no G (p = 0.39). Mean MC was significantly greater than zero for SF with G (mean MC = 114; P < 0.0001).

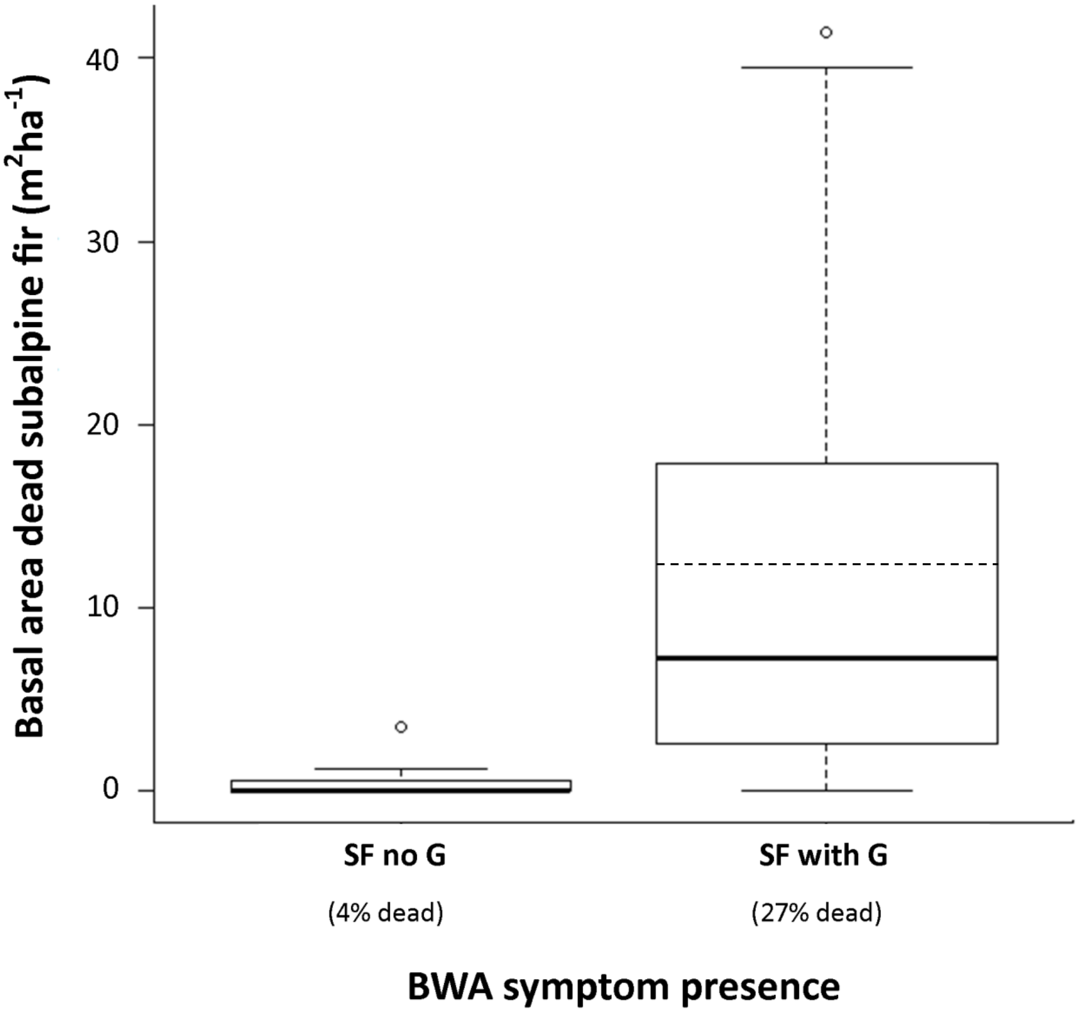


Figure 4.4. Basal area dead subalpine fir (SF) between plots without (SF no G; n = 7) and with (SF with G; n = 42) observed gout symptoms. The average percentage of dead subalpine fir, relative to total live and dead, is in parentheses below the x-axis. Box plot center lines represent median values, dashed center line represents the mean, lower and upper box margins mark the 25th and 75th percentiles, whiskers indicate maximum and minimum values within 1.5 times the interquartile range (~ 2 standard deviations), and hollow dots represent values greater than 1.5 times the interquartile range. Basal area of dead SF for category 'SF no G' was not different from zero ($t = 1.35$, $df = 6$, $p = 0.23$); whereas basal area of dead SF for category 'SF with G' was significantly greater than zero ($t = 6.3979$, $df = 41$, $p < 0.0001$).

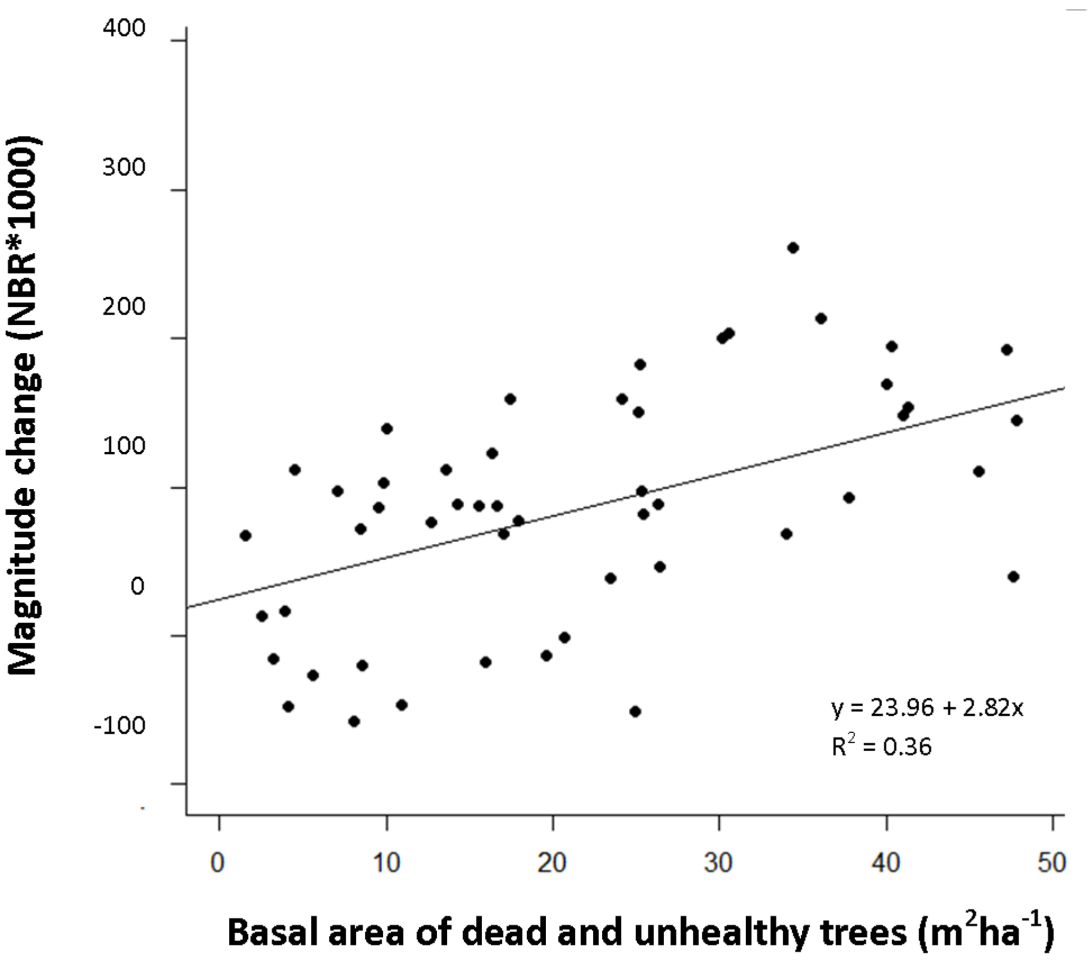


Figure 4.5. Linear relationship between change in magnitude and basal area of dead and unhealthy trees ($R^2 = 0.36$, $F_{1,53} = 29.79$, $P < 0.0001$). Dead trees included all conifer species as would be observed with satellite imagery; unhealthy trees included assessment of subalpine fir only, the predominant declining species. Change in magnitude is expressed as the normalized burn ratio (NBR) multiplied by 1000.

Chapter 5

GROWTH, MORTALITY, AND SPECTRAL SIGNATURE IN A BALSAM WOOLLY ADELGID-INFESTED SUBALPINE FIR FOREST IN OLYMPIC NATIONAL PARK, WA

SUMMARY

The balsam woolly adelgid (BWA), *Adelges piceae* Ratzeburg, has killed true fir (*Abies* spp.) on both east and west coasts of North America since its introduction in the early 1900s, and was first detected in subalpine fir (*Abies lasiocarpa* var. *lasiocarpa*) forests of Olympic National Park (ONP), Washington, USA, during aerial detection surveys in 1970. Studies have documented the adverse effects of BWA on tree vascular systems preceding mortality, but few have investigated tree growth response or stand growth patterns in BWA-infested forests. I used dendrochronological methods to assess subalpine fir growth patterns from 1900 to 2008 in a BWA-infested forest along Hurricane Ridge in ONP. My objectives were to (1) characterize tree condition and growth of BWA-infested subalpine fir trees, comparing infested stand growth at Hurricane Ridge to growth in an uninfested stand in Glacier National Park (GNP); (2) identify a signal of tree-growth stress associated with BWA infestation, comparing results with Landsat spectral reflectance time series and aerial detection surveys, and (3) quantify mortality associated with BWA. Growth response of individual BWA-infested trees was complex because of long-term gradual progression of BWA infestations. Many infested trees experienced growth decline, whereas others showed growth release, possibly due to an increase in available resources following defoliation and mortality of adjacent trees. Pearson's Chi-square was used to test for growth difference between two time periods, approximately pre- and post-BWA, and was found to be significantly different for subalpine fir at Hurricane Ridge ($p < 0.001$) but not for uninfested subalpine fir in GNP ($p = 0.155$). The increase in growth variability at Hurricane

Ridge was observed as an increase in growth standard deviation that began in 1993. This growth response was comparable to changes detected in 1999, in Landsat-derived spectral reflectance (using the normalized burn ratio index; NBR), and in 1989, with aerial detection surveys (ADS). I estimated that 8 to 25 m^2ha^{-1} of basal area or 19 to 53% of subalpine fir trees had died within Hurricane Ridge sample plots over the BWA infestation period. The BWA infestation at Hurricane Ridge resulted in decreased and increased tree growth that were detected as an increase in the standard deviation for the stand, and signaled the onset of BWA effects on stand growth.

INTRODUCTION

Damage to true fir (*Abies* spp.) by balsam woolly adelgid (BWA), *Adelges piceae* Ratzeburg, has been well documented in Washington and Oregon (Johnson et al. 1963, Mitchell and Buffam 2001, Overhulser et al. 2004b), but few studies have used dendrochronological methods to describe tree growth patterns in adelgid-infested forests (Kanoti 2006, Rentch 2009). Within Olympic National Park (ONP),Washington, USA, defoliation and mortality of subalpine fir has been detected across high-elevation ridges by aerial detection surveys (ADS) (Ciesla 2006) since 1970, and as low-magnitude spectral change in Landsat imagery (see Chapter 3). Presence has also been verified in ground-based surveys (Chapter 4). I expected effects of BWA to be evident in patterns of tree growth, tree mortality, and Landsat spectral time series for an infested stand on Hurricane Ridge, ONP.

Dendrochronology is used to observe tree growth responses to insect and disease agents (Swetnam 1985, Veblen et al. 1991, 1994, DelPrato 1999, Zhang et al. 1999, Eisenhart and Veblen 2000), climate variability (Ettl and Peterson 1995), and stand dynamics (Antos et al. 2008). Tree-ring patterns are an indication of tree health, stress, or suppression and can be used to understand environmental relationships and predict mortality (Bigler et al. 2004). Tree-ring chronologies also offer a means to detect past outbreaks. Knowledge of growth decline signatures (e.g., typical duration and rate of decline) aids in the detection of outbreaks for specific agents. Holmes and Swetnam (1996) developed a methodology and software (OUTBREAK) to detect episodes of tree defoliation by comparing tree ring chronologies of host species to non-host species. They assumed that the species would respond similarly to climate, but that host trees would decline in growth relative to non-host species during periods of defoliation. The method allows adjustment of a decline window so that an agent with known cycle duration can be targeted for detection (e.g., 3-4 years for Douglas-fir tussock moth). The program OUTBREAK works well for endemic defoliators, but not for invasive species that have been recently introduced and for which a cycle is not yet established. A second program (JOLTS) was designed by R.L. Holmes (Laboratory of Tree-Ring Research, University of Arizona; unpublished software) to detect insect outbreaks by observing growth release in healthy trees growing adjacent to declining trees. These concepts of growth decline and growth release associated with insect infestations were applied in this chapter to find a parsimonious method for detecting stand growth change associated with the BWA.

My objectives were to (1) characterize tree condition and growth of BWA-infested subalpine fir trees, comparing infested stand growth at Hurricane Ridge, ONP to growth in an uninfested stand in Glacier National Park (GNP); (2) identify a signal of tree-growth stress associated with BWA infestation, comparing results with Landsat spectral reflectance time series and aerial detection surveys, and (3) quantify mortality associated with BWA. I used two independent methods (dendrochronology and satellite imagery) to create time series that spanned the temporal extent of recent changes in subalpine fir health. Tree growth patterns were related to spectral change in Landsat imagery, ADS data, and field plot attributes of tree mortality and symptoms of insects and disease.

METHODS

Study site

I conducted this study in a 40 ha section of BWA-infested subalpine fir forest on a southwest-facing slope (4655440 m E, 5311875 m N, Datum: nad83, Zone: 10N) of Hurricane Ridge, ONP, WA, USA (**Fig. 5.1**). The study area is in the Rocky Mountain Subalpine & High Montane Subalpine Forest macrogroup (RMSF), an existing vegetation classification that commonly occurs on relatively dry high-elevation ridges and includes subalpine fir as a dominant tree species (Federal Geographic Data Committee 2008). Subalpine fir co-occurs on Hurricane Ridge with mountain hemlock (*Tsuga mertensiana*), Pacific silver fir (*Abies amabilis*), Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*), western hemlock (*Tsuga heterophylla*), and western white pine (*Pinus monticola*). Dominant understory species include white rhododendron (*Rhododendron alba*), big huckleberry (*Vaccinium membranaceum*), Oregon grape (*Mahonia nervosa*), vanilla leaf (*Achlys triphylla*), queen cup (*clintonia uniflora*), and subalpine lupine (*Lupinus arcticus*). Soils are derived from sedimentary rocks, marine basalts, and glacial sediments, and are relatively young and infertile with depths ranging from exposed bedrock and thin regolith on upper slopes to deep colluvial regolith on lower slopes (Henderson et al. 1989).

The climate is mild maritime with high winter precipitation and summer drought but micro-climatic conditions vary due to steep topography (Henderson et al. 1989). At high elevations winter temperatures are cold with an average minimum January temperature of −2.8 °C and the potential for growth-limiting snowpack. Summers are warm, especially on south-facing slopes, with an average high temperature in August of 18.9 °C (30-year average temperature data extracted to 500 random points for ONP at 960-1900 m elevation; PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, accessed 23 Apr 2015). An orographic effect causes the release of $\geq$ 500 cm precipitation on the west slope of the Olympic Mountains and 250 cm in east-side mountains where subalpine fir is dominant. About 60% of winter precipitation falls as snow at high elevations, creating a moisture reservoir that distributes water to lower elevations during the spring and summer. Subalpine fir growth can be negatively correlated with winter precipitation, because a deep snowpack shortens the growing season (Ettl and Peterson 1995). With climate change, subalpine fir growth may increase at high elevations where moisture is not limiting (Zolbrod and Peterson 1999, Nakawatase and Peterson 2006).

Subalpine fir is subject to disturbance by endemic insects and pathogens including western balsam bark beetle (*Dryocoetes confusus*), fir engraver (*Scolytus ventralis*), fir root bark beetle (*Pseudohylesinus granulatus*), potentially silver fir beetle (*Pseudohylesinus sericeus*), and Annosus root disease (*Heterobasidion occidentale* sp. nov.). Wind and snow are dominant abiotic disturbance agents causing stem and branch breakage, and stand-replacing fires have occurred in the area historically, but return intervals are 100 to 275 years or more (Agee 1993, Edmonds et al. 2011a). Both high and low snowpack can stress subalpine fir by shortening the growing season or causing late summer drought, respectively (Peterson et al. 2002). Subalpine fir forest is adapted to these disturbance regimes, but BWA may have changed this dynamic by increasing physiological stress and making trees more susceptible to endemic agents and extreme weather events (e.g., Mitchell et al. 2001).

Aerial detection survey of Hurricane Ridge area

Aerial detection surveys provide a history of BWA activity for the study area at Hurricane Ridge. Surveys are flown each year by the U.S. Forest Service (USFS) and Washington State Department of Natural Resources (DNR). Defoliation is mapped and rated as low, medium, or high (L, M, or H) severity. Where mortality is observed, a numeric rating indicates the number of or trees per ha (TPH) that have died (trees per area (TPA) used by ADS). BWA infestations were initially detected in aerial surveys by identifying fading foliage and branch flagging in fir trees. This was most effective for detecting stem-infested trees but not branch infestations which can cause slow needle drop and inconspicuous mortality (Overhulser et al. 2004a). Detection of BWA improved in 2000 when surveyors learned that abundant dark lichens (*Bryoria* spp.) in tree tops indicated BWA-caused defoliation. Documentation of BWA activity may therefore be discontinuous before 2000, also because surveys were semi-annual until 1989, and because surveyors attempted to mark only areas with evidence of new decline (Ciesla 2006).

The first ADS observation of BWA activity in the Hurricane Ridge area occurred in 1970, west of Hurricane Hill and 5 km northwest of the sample plots (**Fig. 5.1**, **Table 5.1**). Severity was low (0.03 TPH) and observed in three patches that totaled 121 ha. Seven years later in 1977 low severity BWA impacts (0.11 and 0.16 TPH) were detected closer to the sample area, 537 m south of the southernmost plot, and 689 m west of the northernmost plot. Hurricane Hill experienced new low severity impacts in 1977 and 1980 as well. In 1980, low severity BWA activity was detected at the base of the Klahane Ridge trail, 2.4 km to the northeast of sampling plots. Then in 1989, low severity BWA was detected for the first time in the sample area encompassing the two northern plots (1 and 2), and another new infestation patch was seen 1.5 km to the east. The following year, BWA spread to include plots 1, 2 and 3, and was detected within 20 m and 100 m of plots 5 and 6, as well as in a patch 270 m south of the southern-most plot (plot 4). In 1991 the entire sampling area along Hurricane Ridge increased to medium severity. Two small additional infestations were detected downslope of the sampling area in 1993. Although BWA detection continued to spread to surrounding areas, no polygons were drawn in the sampling area again until 2002, this time along the upper part of the ridge above the sampling plots, including plots 3 and 8. Similarly, no further activity was noted until 2006 when a polygon encompassed the four northern plots. From 2007 through 2012 BWA was detected along the sampled slope in patches of various configurations that included all plots one or more times.

LandTrendr analysis of spectral decline in NBR

For analysis of spectral decline I used two "Landsat-based detection of trends in disturbance and recovery" (LandTrendr) products (Kennedy et al. 2010): (1) a disturbance map layer created from LandTrendr algorithm-fitted trajectories to determine decline locations and magnitude change as percent vegetation loss for each plot, and (2) unfitted reflectance values for each year and pixel to graph time series. The LandTrendr raster data sets were acquired from the Laboratory for Applications of Remote Sensing in Ecology (LARSE) at Oregon State University. The raster data sets were created from a 28-year time series (1984-2012) of Landsat Thematic Mapper and Enhanced Thematic Mapper Plus images for scene 4727 (WRS-2 path 47, row 27) with a spatial resolution of 30 x 30 m. Image dates were primarily from July and August when clouds were minimal in the region but forest vegetation was maximized. Images were processed with USGS L1T LPGS, converted to surface reflectance, cloud-screened, normalized (MADCAL), and converted to Normalized Burn Ration (NBR) for observing vegetation relevant values (see Kennedy et al. 2007b, 2010, 2012). The NBR is an effective index for detecting change in forest vegetation (Cohen et al. 2010); the index is calculated from near infrared (NIR) and shortwave infrared (SWIR) bands (NBR = (Band 4 – Band 7) / (Band 4 + Band 7)) (Key and Benson 2006). The original NBR values were multiplied by 1000 and rounded to the nearest integer to reduce image bit size. They are hereafter referred to as NBR or NBR units. LandTrendr segmentation algorithms were applied to annual source data for each pixel, to identify start and end years (vertices) of spectral change, and to fit a straight regression-line through the vertices. Fitted NBR values for each vertex and year along a trajectory were used to create the disturbance map layer with three bands that represented attributes of (a) onset year of change, (b) percent magnitude change, and (c) duration of continuous change. Magnitude of change was expressed as a relative percent loss of vegetative cover for a given pixel as modeled from the pre- and post-disturbance NBR values.

Increment core and plot data collection

Eight 18-m radius plots were established on a southwest-facing slope along Hurricane Ridge, ONP. The plots were placed across 40 hectares within a 2.5 km length of slope and an elevation range of 1360-1460 m (**Fig. 5.1**). The plots proceeded from northwest to southeast in this order 1, 2, 3, 8, 5, 6, 7, 4; plot numbers 4 and 8 were switched from the first field visit and the order was kept to avoid confusion of labeled cores and data sheets.

At each plot I collected tree cores and data on trees, environment, and disturbance. Tree cores were sampled from the upper half of each plot. Fifty percent of a tree base was required to stand or lie within the half plot to be sampled. An increment borer was used to extract cores at 1.37 m from the ground and parallel to the slope contour. Cores were extracted from all conifer species, live and dead, and > 12.7 cm diameter at breast height (dbh). Each core was labelled with date, plot, species, and tree number. Cores were then transported to the School of Environmental and Forest Sciences at the University of Washington for storage and analysis.

Tree data collected from the entire circular plot included tree number, species, status (dead or alive), dbh, crown class, health, cone-bearing status, and signs and symptoms of insects and disease, and mechanical damage with damage location and percent damage. Dead trees were noted if standing or fallen and given a decay class for the following criteria: 0 = needles brown but present on branches; 1 = needles gone, fine branches present; 2 = most fine branches gone, bark still present but detached in places; 3 = secondary branches mostly gone, branch stubs solidly attached, bark mostly gone; 4 = bark gone; branch stubs pull out, stem decaying but still holding form; 5 = well decayed and snag or log not holding form (adapted from Woodall and Monleon 2008). Dead trees in decay classes 0-2 showed symptoms of gout and were classified together as “recently dead” and assumed to be among the dead cohort most affected by BWA. A BWA-infection class was assigned to each tree based on percent of branches showing gout symptoms: 0 = 0%; 1 = 1-5%; 2 = 6-25%; 3 = 26-50%; 4 = 51-75%; 5 = 76-95%; 6 = 96-100%. Seedling and sapling data on species, status, and symptoms were recorded for all individuals within a 7.3 m radius subplot.

To make a comparison in tree growth with an uninfested stand I obtained tree ring data from the international tree ring data bank (http://www.ncdc.noaa.gov/data-access/paleoclimatology-data/datasets/tree-ring) for a subalpine fir stand in Glacier National Park, Montana (Bekker et al., accessed on April 8, 2015, https://www.ncdc.noaa.gov/paleo/study/5993). The GNP tree ring data were selected because the last year of the time series was 2006, which was comparable to my data (end year 2008), and because BWA had not been detected in GNP by their sample end date (USFS and DNR 2013).

Tree core data from the Olympic Mountains or Cascade Range were not available from the tree ring data bank after 1990-1992.

Crossdating and standardization of cores

Cores were mounted and sanded, and ring widths were measured to the nearest 0.01 mm using a Velmex incremental measuring machine and J2X software. Ring widths were visually cross-dated and verified using the computer program COFECHA (Holmes 1983). COFECHA (Holmes 1983) detects measurement and cross-dating errors by computing correlation coefficients between overlapping 50-year segments from individual series (Eisenhart and Veblen 2000). Cores were removed from the data set that did not meet the Pearson (parametric, quantitative) critical correlation 99% confidence level. Cores in the final data set totaled 148. The last year of growth in dead trees was determined by visual cross-dating with live trees, and verification with COFECHA. Individuals that could not be confidently crossdated because of unmatching rings sequences, and highly weathered or broken cores, were excluded from analysis.

Chronologies were standardized using the dplR packet in R (Bunn 2008). Because the objective in this study was to identify long-term growth reductions associated with trees infested with BWA, the Modified Negative Exponent (Fritts 2001) was selected as the standardization method. Ring width decreases with increasing age and bole circumference of the tree and this produces a downward trend in ring width. Standardization allows trees of different age and size to be used together in analysis (Fritts and Swetnam 1989). An algorithm removes the intrinsic nonlinear decrease in ring width associated with the increase in stem circumference over time without removing effects of disturbance. My detection method was robust to different standardization methods (e.g., horizontal line) but the Modified Negative Exponent was the logical standardization for this research because it removed the growth factor without reducing, exaggerating, or distorting the growth signal in later years. Use of basal area increment (BAI) is another way to remove the tree-age growth effect in dplR. Ring-width (mm) is converted to ring-area (mm^2) based on the diameter of the tree and the width of each ring moving towards the pith of the tree. I used BAI to standardize tree cores for comparing ten-year growth averages among gout classes (e.g., DelPrato 1999).

Analysis of tree cores

Ten-year (1998-2008) average growth BAI mm^2 was compared between trees grouped by gout class in to identify recent growth differences among groups with different levels of infestation (e.g., DelPrato 1999). Unequal variances precluded use of analysis of variance (ANOVA) but a graphical representation of the data was assessed. The mean and standard deviation of tree-ring growth were calculated from all cores for each year and presented as a time series from 1900 to 2008 to identify a signal of BWA-related growth change. Pearson’s Chi-square statistical analysis was used to compare average growth as counts within growth categories for time periods before and after ADS detection of BWA at the infested study site for (a) all trees (predominantly subalpine fir); (b) subalpine fir only; (c) non-subalpine fir species; and (d) for an uninfested subalpine fir stand in GNP, Montana.

RESULTS

Condition and growth of subalpine fir

Plot characteristics

Co-dominant trees at all plots were older than 100 years. Plots had at least 20% subalpine fir, and ranged in tree density from 226 to 717 stems ha^{-1} with average diameters of 26.7 to 46.4

cm (**Table 5.2**). Gout symptoms were detected on all size classes of live subalpine fir (seedling, sapling, and tree) and on dead subalpine fir decay classes 0 through 2. Cones were produced on tree-sized subalpine fir in all plots, and abundant subalpine fir reproduction occurred in all plots except plot 3 which was regenerating in Pacific silver fir. Percent vegetation loss indicated by spectral change magnitude (NBR) was low for plots with low basal area of subalpine fir (plot 5), having few individuals with gout symptoms (plot 8), or both (plot 3). The four plots with the highest percentages of gouted trees had the greatest % vegetation loss, with the exception of plot 5 which had 75% gouted trees but only 8% of the conifer basal area was subalpine fir.

Growth of subalpine fir infested with BWA

The growth pattern of BWA infested trees was variable. Trees with heavy branch infestations showed slowly declining growth but some trees showed no decline (tree 4-11; **Fig. 5.2a**) or experienced sporadic growth release (trees 1-30 and 1-32). Some trees maintained stable growth and then died suddenly with evidence of mechanical damage (broken stem or treefall damage; trees 7-17 and 5-14; **Fig. 5.2b**).

Attempts to associate growth differences with degrees of gout by comparing recent 10-year BAI (mm^2) growth averages among gout and health classes showed that growth was too variable within these groups to detect significant differences. However, a declining trend is visible with increasing gout severity (**Fig. 5.3**). Variability may be related to inaccurate ground-level estimates in health and gout, or multiple other factors including competition or mortality of adjacent trees, duration of infestation, and micro-environmental conditions.

Detecting a signal for BWA-induced forest stress

Stand growth

Mean tree growth for Hurricane Ridge sample plots was fairly stable from 1900 through

2008 except for a low growth period from 1916 to 1934 and a peak in mean growth in 1955 (**Fig. 5.4a**). Growth variability, represented by standard deviation, remained relatively stable until 1993, with lower values also from approximately 1916 through 1934 and a peak in 1955 (55% increase). However, in 1993 standard deviation increased by 65% compared with earlier years (mean from 1900-1980) and continued to increase for the next 11 years (**Fig. 5.4b**). The standard deviation peaked in 2004 at 144% and remained high through 2008. This unique signal is assumed to be the result of stress from BWA infestation.

Pearson's Chi-squared test showed tree growth to be significantly different between pre- and post-BWA infestation periods (1970-1989 and 1990-2008; $p < 0.001$) in Hurricane Ridge subalpine forest. Trees counts for three growth classes: greater than mean growth (> mean + 20%), near mean growth (within ± 20%), and less than mean growth (< mean - 20%), were totaled and compared for each time period (**Table. 5.3**). When separated by species, the number of subalpine fir with lower than mean growth increased by 23%, and the number with greater than mean growth increased by 4% ($p < 0.001$), whereas non-subalpine fir species showed no significant difference ($p = 0.155$). Tree growth in an uninfested subalpine fir forest in GNP, also showed no significant difference for the same time periods ($p = 0.095$). The ONP and GNP stands are both mature subalpine fir, with the oldest cored tree at 272 and 240 years respectively.

Spectral reflectance

Average plot spectral decline (NBR) became apparent by 1999 (**Fig. 5.5**). Magnitude of response in each plot was influenced by the relative basal area of subalpine fir and number of trees with BWA symptoms as mentioned under plot characteristics above. Difference in spectral reflectance among individual plots began to increase around 1999 as well.

Tree health and mortality

Basal area of recent subalpine fir mortality (dead trees in decay classes 0-2) ranged from 8 to 25 m^2 ha^{-1} (**Table 5.4**). Plots with the greatest dead tree basal area generally had more gouted trees and higher percentage of subalpine fir. Up to 53% (37% average) of original live subalpine fir basal area (trees > 12.7 cm dbh) died during approximately 19 years (ADS) of infestation within the Hurricane Ridge study area. Percent vegetation loss ranged from 0% (plot five) to 56% with an average of 26% vegetation loss. Vegetation loss is visible in aerial photographs from 1976 to 2009 (**Fig. 5.6**) and is most pronounced near plots 1 and 2 and further northeast. Plots 1 and 2 were within the first ADS BWA polygons detected in the study area in 1989.

DISCUSSION

Condition and growth of subalpine fir

Many trees infested with BWA showed a gradual decrease in growth (**Fig. 5.2** Trees 4-08, 4-16, 4-60, 5-18) but this pattern was highly variable and likely related to individual tree responses to infestation level, infestation location on the tree, and duration of BWA presence, as well as resource availability, and presence of other disturbance agents. Stem infestations are not common on the Olympic Peninsula but can kill a tree in 2-3 years, whereas trees with branch infestations can live for many years (Ragenovich and Mitchell 2006). Competition for resources likely limits growth, whereas abundant resources may sustain growth of branch-infested trees and allow them to live longer than they would with environmental stress. Death of adjacent trees may even cause growth release in neighboring branch-infested trees that suddenly gain access to water, nutrients, and light (e.g., Trees 1-04, 1-30, 1-32). Alternatively, tree growth may be abruptly terminated by wind events that break stems (e.g., Trees 7-17, 5-14).

Opportunistic bark beetles or stem and root pathogens may shorten the growth response and decrease the time to tree mortality as well. Western balsam bark beetle (WBBB) (*Dyocoetes confusus*) was documented along Hurricane Ridge in 2010 and 2011 by ADS, and I confirmed its presence in 2012 with baited Lundgren funnel traps (trap # WB 2 and NM 2 near study area) (**Fig. 5.7**). WBBB may target trees weakened by BWA (Furniss and Carolin 1977, Carlson 2013), although their initial occurrence in a BWA-infested stand may also be coincidental. Diagnostic radiating beetle galleries of WBBB were observed on 12% of dead subalpine fir trees across the study area (all plots except plot 2), predominantly on decay classes 2-4 that have sloughing bark and allow easy gallery detection (23% in class 2, 46% in class 3, 27% in class 4). Only two trees had both galleries and BWA gout symptoms. By definition, decay class 3 and older lack fine branches that show gout symptoms, and most trees in this decay class died before BWA was detected (ages of cored trees with WBBB are 1961, 1969, 1975, 1983, 1986, and 1992). This indicates that WBBB was already present in the area when BWA defoliation was detected by ADS. Galleries of fir engraver (*Scolytus ventralis*), fir root bark beetle (*Pseudohylesinus granulatus*) and signs of decay fungi (*Armillaria* and *Heterobasidion occidentalis*) were occasionally noted in field plots as well. Fir engraver was documented in the study area by ADS in 2002.

The potential for interactions between chronic BWA infestations and endemic disturbance agents is high. In the Cascade Range a BWA-infested stand experienced 40% subalpine fir mortality, and later was infested with western spruce budworm that killed the remaining 60% of subalpine fir (Mitchell and Buffam 2001). In the Chilliwack unit of North Cascades National Park, WBBB and BWA were treated as a complex and estimated to have killed 3,200 subalpine fir trees and damaged 1635 ha from 1993 to 2012 (Carlson 2013).

Additional interactions among temperature and moisture conditions, adelgid population growth, and tree health may create complex tree responses that are challenging to typify. Nevertheless, the high variability in growth response for the infested study area along Hurricane Ridge created in a signal that may be unique to BWA infestations.

Detecting a signal for BWA-induced forest stress

Stand growth became increasingly variable beginning in 1993 with a 65% increase in standard deviation that increased or remained high until the end year in 2008 (**Figure 5.4b**). A similar growth pattern was not seen elsewhere in the chronology extending from 1900 to 2008. This broad time period provided a temporal perspective that included more than 50 years before BWA arrived on the Olympic Peninsula and over 100 years for observing growth variability on Hurricane Ridge.

The increase in standard deviation can be explained by species-specific stress that causes growth decline and mortality in affected host trees while adjacent trees respond with growth release due to an increase in resource availability when neighboring trees die. Interestingly, mean tree ring growth remained fairly stable throughout the period before and after BWA introduction (**Fig. 5.4a**). I expected to see a decrease in mean growth, but because BWA progresses slowly within a stand, affecting a few trees at a time, less affected trees are able to respond with growth. The stable mean growth may reflect a constant productivity level for this site.

Landsat spectral reflectance (NBR) averaged for the eight plots showed a decline trend that indicated a loss of vegetation. The decline was pronounced in 1999, several years after the increase in standard deviation of growth, but gradual decline may have begun earlier (**Fig. 5.5a**). Spectral change relates to visible change in vegetation cover such as defoliation and mortality. It makes sense that change in satellite imagery would become visible after the onset of tree growth change. Two of the eight plots (plots 3 and 5) did not show decline, and this is reasonable considering their low percentages of subalpine fir basal area (17% and 8%) or less severe or possibly more recent infestations (**Table 5.2**).

The decline trend in spectral reflectance is not a straight line and likely reflects interactions with weather events such as the late snow melt in August of 1999 and winter wind storms of 2006 and 2007. Late snow melt shortens the growing season for subalpine vegetation, reducing tree growth for one to two years with possible effects to tree health (Antos et al. 2008). Winter storms damage forest vegetation but forests generally recover spectrally within a year or two (chapter 3). The extreme response and continued decline suggests a synergistic interaction between stress caused by BWA and weather events.

Aerial detection surveys detected moderate levels of BWA defoliation across the sample area in 1991 (**Table 5.3**). This is approximately 2 years before effects were evident in a stand growth response, and approximately 8 years before change was detected by satellite imagery. Although the onset years are not the same, this can be explained by differences in detection method, scope, resolution, and the response variable observed. Importantly, all three change detection methods identified a unidirectional trend for the same general time period that can be related to forest vegetation stress, and that is likely the result of a BWA infestation interacting with other endemic agents and processes.

Tree health and mortality at Hurricane Ridge

According to ADS, BWA had been active in subalpine fir of the Hurricane Ridge study area for approximately 19 years (1989-2008) by the end date of the core sample. The number of years BWA had been present at a sample plot or individual tree likely varied. The rate of population growth, spread, and impact within any forest varies with temperature, wind, topographic, and micro-environmental conditions (Greenbank 1970, McClure 1990, Dale et al. 1991).

I estimated that 8 to 25 m^2ha^{-1} basal area or 19 to 53% of live subalpine fir had died within sample plots as a result of BWA. This was calculated from the total basal area of dead trees in decay classes 0-2 (percentage was calculated from basal area dead relative to the total live and dead). Decay class 0-2 was the best estimate of trees that had died during the time period of BWA infestation and assumes that all dead trees of decay class 0-2 died from BWA. This is a reasonable estimate based on the year of death determined in tree cores for Hurricane Ridge.

In comparison, long-term BWA study plots in the Oregon and Washington Cascade Range experienced 7 to 79% subalpine fir mortality (tally of stem counts > 2.54 cm dbh) over a 19 to 38 year study period (Mitchell and Buffam 2001). The high range in mortality was associated with differences in elevation and moisture. A low-elevation (1280 m), mesic site lost 60% of subalpine fir trees in the first 8 years and 19% in the following 18 years for a total of 79%. A high-elevation (1830 m) stand near timberline lost only 7% subalpine fir in 33 years, and damage was considered insignificant and ephemeral. As mentioned earlier, one stand experienced 40% mortality in19 years, but lost the remaining 60% during a subsequent spruce budworm infestation. Most plots continued to show sporadic signs of adelgid presence and continued tree mortality. However, many trees also survived stem infestation, 41-69% in five of seven plots. After a high initial mortality of susceptible trees, new damage was slow to accumulate. Although the mortality percentages from this study were calculated from stem counts of saplings and trees in long-term study plots, and are not directly comparable to my study, the percentages give an indication of the relative impact BWA has had on subalpine fir.

Conclusion and future of subalpine fir on Hurricane Ridge

Research in the Cascade Range and Appalachian Mountains has shown that once BWA is introduced to a forest, it maintains a presence and continues to affect the fir tree hosts, with the exception of very cold and high elevation sites (Mitchell and Buffam 2001). Nevertheless, rates of mortality decrease after the initial infestation period and some trees are able to recover. At Hurricane Ridge, subalpine fir trees were producing cones in each plot and regeneration was strong in all but one plot that was dominated by Pacific silver fir. Gout symptoms were noted on all age classes; however, saplings may be less susceptible, allowing them to survive and replace older subalpine fir trees that are killed by BWA (Ragenovich and Mitchell 2006). Hurricane Ridge sample plots averaged 1400 m elevation but subalpine fir also grows above 1600 m along the ridge; temperatures are not cold enough to eliminate BWA from the Olympic Peninsula (Quiring et al. 2008) but may hinder BWA population growth at the higher elevations. Thus, after an initial wave of mortality the rate of damage may decrease and be followed by forest regeneration. This has been the case in the Appalachian Mountains where mortality averaged 82, 98, and 95% in 3 compositional types of Fraser fir forest (Witter and Ragenovich 1986) and stands have since regenerated in Fraser fir (Adams and Stephenson 2010, McManamay et al. 2011, Stehn et al. 2013). There is speculation that Fraser fir forests will remain in a young age class if trees are killed by BWA when they mature, but trees are expected to reach cone-bearing age (Potter et al. 2005). The effect that chronic BWA infestations has on susceptible fir species over long time periods is uncertain because of the potential for interaction with endemic agents and other disturbances. Trees already stressed by BWA may not tolerate additional stress, and they provide a large susceptible resource for opportunistic insects and pathogens.

There are currently no viable management options for treating forests of protected areas that are infested with BWA. Biocontrol efforts that began in 1957 with the introduction of over 25 predatory species to North America from Europe and Asia have not successfully contained BWA (Mitchell and Wright 1967, Zilahi-balogh et al. 2002). Pesticide application is not effective or desirable for broad-scale application; however, products may be applied to protect individual trees with cultural significance.

REFERENCES

Adams, H. S., and S. L. Stephenson. 2010. Twenty-five years of succession in the spruce-fir forest on Mount Rogers in southwestern Virginia. Castanea 75:205–210.

Agee, J. K. 1993. Fire Ecology of Pacific Northwest Forests. Island Press, Washington, D.C.

Antos, J. A., R. Parish, and G. D. Nigh. 2008. Growth patterns prior to mortality of mature *Abies lasiocarpa* in old-growth subalpine forests of southern British Columbia. Forest Ecology and Management 255:1568–1574.

Bigler, C., J. Gričar, H. Bugmann, and K. Čufar. 2004. Growth patterns as indicators of impending tree death in silver fir. Forest Ecology and Management 199:183–190.

Bunn, A. G. 2008. A dendrochronology program library in R (dplR). Dendrochronologia 26:115–124.

Carlson, D. 2013. Twenty-year forest insect damage pattern and risk assessment, North Cascades National Park. USDA Forest Service, Wenatchee, WA.

Ciesla, W. M. 2006. Aerial signatures of forest insect and disease damage in the western United States. Forest Health Technology Enterprise Team. FHTET-01-06. USDA Forest Service, Fort Collins, Colorado.

Cohen, W. B., Z. Yang, and R. Kennedy. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 2. TimeSync - Tools for calibration and validation. Remote Sensing of Environment 114:2911–2924.

Dale, V. H., R. H. Gardner, and D. L. DeAngelis. 1991. Elevation-mediated effects of balsam woolly adelgid on southern Appalachian spruce-fir forests. Canadian Journal of Forest Research 21:1639–1648.

DelPrato, P. 1999. Response of whitebark pine (*Pinus albicaulis*) to white pine blister rust (*Cronartium ribicola*) in Mount Rainier National Park, Washington. M.S. Thesis. St. Joseph's University, Philadelphia, PA.

Edmonds, R. L., J. K. Agee, and R. I. Gara. 2011. Fire ecology and fire regimes. Pages 79–112 Forest Health and Protection. Second Edition. Waveland Press, Inc., Long Grove, Illinois.

Eisenhart, K. S., and T. T. Veblen. 2000. Dendroecological detection of spruce bark beetle outbreaks in northwestern Colorado. Canadian Journal of Forest Research 30:1788–1798.

Ettl, G. J., and D. L. Peterson. 1995. Growth response of subalpine fir (*Abies lasiocarpa*) to climate in the Olympic Mountains, Washington, USA. Global Change Biology 1:231–241.

Federal Geographic Data Committee. 2008. National Vegetation Classification Standard (Version 2.0). U.S. Geological Survey, Reston, VA.

Fritts, H. C. 2001. Tree Rings and Climate. Blackburn Press, Caldwell, NJ.

Fritts, H. C., and T. W. Swetnam. 1989. Dendroecology: a tool for evaluating variations in past and present forest environments. Advances in Ecological Research 19:111–188.

Furniss, R. L., and V. M. Carolin. 1977. Western Forest Insects. U.S. Government Printing Office, Washington, D.C.

Greenbank, D. O. 1970. Climate and the ecology of the balsam woolly aphid. The Canadian Entomologist 102:546–579.

Henderson, J. A., D. H. Peter, R. D. Lesher, and D. C. Shaw. 1989. Forested Plant Associations of the Olympic National Forest. USDA Forest Service, Pacific Northwest Region. R6-ECOL Technical Paper 001-88., Portland, Oregon.

Holmes, R. L. 1983. Computer-assisted quality control in tree-ring dating and measurement. Tree-ring Bulletin 43:69–78.

Holmes, R. L., and T. W. Swetnam. 1996. Program OUTBREAK User's Manual. Laboratory of Tree-Ring Research, University of Arizona, Tucson, AZ.

Johnson, N. E., R. Mitchell, and K. H. Wright. 1963. Mortality and damage to Pacific silver fir by the balsam woolly aphid in southwestern Washington. Journal of Forestry:854–860.

Kanoti, A. M. 2006. Relationship between balsam woolly adelgid damage, radial growth, climate and stand characteristics in eastern Maine. M.S. Thesis. Paper 434. University of Maine, Orono, Maine.

Kennedy, R. E., W. B. Cohen, and T. A. Schroeder. 2007. Trajectory-based change detection for automated characterization of forest disturbance dynamics. Remote Sensing of Environment 110:370–386.

Kennedy, R. E., Z. Yang, and W. B. Cohen. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 1. LandTrendr — temporal segmentation algorithms. Remote Sensing of Environment 114:2897–2910.

Kennedy, R. E., Z. Yang, W. B. Cohen, E. Pfaff, J. Braaten, and P. Nelson. 2012. Spatial and temporal patterns of forest disturbance and regrowth within the area of the Northwest Forest Plan. Remote Sensing of Environment 122:117–133.

Key, C. H., and N. C. Benson. 2006. Landscape assessment: ground measure of severity, the composite burn index; and remote sensing of severity, the normalized burn ratio. *in* D. C. Lutes, R. E. Keane, J. F. Caratti, C. H. Key, N. C. Benson, S. Sutherland, and L. J. Gangi, editors. FIREMON: Fire Effects Monitoring and Inventory System. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT.

McClure, M. S. 1990. Role of wind, birds, deer, and humans in the dispersal of hemlock woolly adelgid (Homoptera: adelgidae). Environmental Entomology 19:36–43.

McManamay, R. H., L. M. Resler, J. B. Campbell, and R. A. McManamay. 2011. Assessing the impacts of balsam woolly adelgid (*Adelges piceae* Ratz.) and anthropogenic disturbance on the stand structure and mortality of Fraser fir [*Abies fraseri* (Pursh) Poir.] in the Black Mountains, North Carolina. Castanea 76:1–19.

Mitchell, R. G., and P. E. Buffam. 2001. Patterns of long-term balsam woolly adelgid infestations and effects in Oregon and Washington. Western Journal of Applied Forestry 16:121–126.

Mitchell, R. G., and K. H. Wright. 1967. Foreign predator introductions for control of the balsam woolly aphid in the Pacific Northwest. Journal of Economic Entomology 60:140–147.

Nakawatase, J. M., and D. L. Peterson. 2006. Spatial variability in forest growth – climate relationships in the Olympic Mountains , Washington. Canadian Journal of Forest Research 36:77–91.

Overhulser, D., L. Geiser, A. Kanaskie, I. Ragenovich, and M. Mcwilliams. 2004a. Aerial detection of balsam woolly adelgid infestations on subalpine fir using lichen cover. USDA Forest Service, Pacific Northwest Region, and Oregon Department of Forestry.

Overhulser, D. L., I. R. Ragenovich, M. McWilliams, and E. A. Willhite. 2004b. Balsam woolly adelgid occurrence on true fir in Oregon. Oregon Department of Forestry, Salem, OR.

Peterson, D. W., D. L. Peterson, and G. J. Ettl. 2002. Growth responses of subalpine fir to climatic variability in the Pacific Northwest. Canadian Journal of Forest Research 32:1503–1517.

Potter, K. M., J. Frampton, and J. Sidebottom. 2005. Impacts of balsam woolly adelgid on the southern Appalachian spruce-fir ecosystem and the North Carolina Christmas tree industry. Third Symposium on Hemlock Woolly Adelgid in the Eastern United States, Asheville, NC Feb. 1-3.

Quiring, D., D. Ostaff, L. Hartling, D. Lavigne, K. Moore, and I. DeMerchant. 2008. Temperature and plant hardiness zone influence distribution of balsam woolly adelgid damage in Atlantic Canada. The Forestry Chronicle 84:558–562.

Ragenovich, I. R., and R. G. Mitchell. 2006. Balsam woolly adelgid. Forest Insect & Disease Leaflet 1182. USDA Forest Service.

Stehn, S. E., M. A. Jenkins, C. R. Webster, and S. Jose. 2013. Regeneration responses to exogenous disturbance gradients in southern Appalachian Picea-Abies forests. Forest Ecology and Management 289:98–105.

Swetnam, T. W. 1985. Radial growth losses in Douglas-fir and white fir caused by western spruce budworm in northern New Mexico: 1700 to 1983. USDA Forest Service, Southwestern Region, Forest Pest Management Report R-3 86-2.

USFS, and DNR. 2013. Montana forest insect and disease conditions and program highlights 2013. Forest Health Protection Report 14-2. (C. Hayes, Ed.). Forest Service Region 1, Northern Region, Montana Department of Natural Resources and Conservation Forestry Division.

Veblen, T. T., K. S. Hadley, E. M. Nel, T. Kitzberger, M. Reid, and R. Villalba. 1994. Disturbance regime and disturbance interactions in a Rocky Mountain subalpine forest. The Journal of Ecology 82:125–135.

Veblen, T. T., K. S. Hadley, and M. S. Reid. 1991. Disturbance and stand development of a Colorado subalpine forest. Journal of Biogeography 18:707–716.

Witter, J. A., and I. R. Ragenovich. 1986. Regeneration of Fraser fir at Mt. Mitchell, North Carolina, after depredations by the balsam woolly adelgid. Forest Science 32:585–594.

Woodall, C. W., and V. J. Monleon. 2008. Sampling protocol, estimation, and analysis procedures for the down woody materials indicator of the FIA program. USDA Forest Service, Northern Research Station. General Technical Report NRS-22, Newtown Square, PA.

Zhang, Q., R. I. Alfaro, and R. J. Hebda. 1999. Dendroecological studies of tree growth, climate and spruce beetle outbreaks in Central British Columbia, Canada. Forest Ecology and Management 121:215–225.

Zilahi-balogh, G., L. T. Kok, and S. M. Salom. 2002. A review of world-wide biological control efforts for the family Adelgidae. Pages 129–140 *in* B. Onken, R. Reardon, and J. Lashomb, editors. Proceedings: Hemlock Woolly Adelgid in the Eastern United States Symposium. Rutgers University, East Brunswick, NJ.

Zolbrod, A. N., and D. L. Peterson. 1999. Response of high-elevation forests in the Olympic Mountains to climatic change. Canadian Journal of Forest Research 29:1966–1978.

Table 5.1. Aerial detection survey (ADS) data for balsam woolly adelgid (BWA) activity at Hurricane Ridge.

Year	Severity	ADS detection of BWA activity
1970	0.03 TPH	First sighting on Hurricane Ridge, 5 km NW of sample plots
1977	0.11, 0.16 TPH	Within 1 km N and S of sample plots
1980	L	2.4 km NE of sample plots, base of Klahane Ridge Trail
1989	L	At sample plots 1 and 2; also 1.5 km to the east of sample plots
1990	L	At sample plots 1, 2, and 3; nearing plots 4, 5, and 6
1991	M	Entire sample area; increased to medium severity
1993	0.28-0.81 TPH	Two new polygons on lower slope below sample plots
2002	M	At sample plots 3 and 8; and upper part of ridge above sample plots
2006	H	Across 4 northern plots (1,2,3,8)
2007-2012	M-H	Continued detection within sample area; patches vary in size, location, and coverage of plots.

Mortality: TPH = trees per hectare

Defoliation levels: L = low, M = medium, H = high

Table 5.2. Site characteristics, live conifer abundance, and dimensions of subalpine fir (trees > 12.7 cm dbh) at eight plots along Hurricane Ridge, Olympic National Park, WA.

Sample plot	Elev. (m)	Aspect (degrees)	Slope (degrees)	Tree density (stems ha^{-1})	Basal area (m^2ha^{-1})	Dominant live tree species*	% Subalpine fir (stems)	Height (m)	Age** (years)	Mean dbh (cm)
1	1425	226	19	422	49.4	subalpine fir	93	21.9	170	27.8
2	1452	200	19	226	32.2	subalpine fir	100	23.3	226	40.5
3	1409	198	24	678	62.7	Pac. silver fir	20	25.1	152	29.9
4	1388	232	22	717	50.4	subalpine fir	67	23.1	227	26.7
5	1376	191	32	334	88.4	subalpine fir	35	28.0	144	28.9
6	1450	276	6	245	50.1	subalpine fir	100	27.0	206	46.4
7	1394	178	35	285	87.4	subalpine fir	83	26.2	119	32.7
8	1443	172	31	589	79.1	subalpine fir	92	21.4	181	29.0

* Conifer species present in sampling plots in order of abundance are subalpine fir, Pacific silver fir, Douglas-fir, western hemlock, mountain hemlock, and western white pine.

**Age based on raw core ring count at dbh from average of two live codominant subalpine fir trees per plot.

Table 5.3. Percentage of trees in each of three growth classes for two time periods representing pre- and post-balsam woolly adelgid (BWA) introduction based on aerial detection surveys (ADS). At a significance level of α = 0.05, growth is significantly different between pre- and post-BWA time periods for the ONP study area overall and subalpine fir trees considered separately. Non-subalpine fir conifers and subalpine fir in the uninfested GNP stand are do not differ between time periods. Low severity BWA-related defoliation was detected in the ONP study area by ADS in 1989 and 1990, and detected across the entire study area at moderate severity by 1991.

	Tree growth class	Percentage of trees in each class		
		1970-1989	1990-2008	p-value
Olympic National Park	> 20%	14	21	< 0.001
	Mean ± 20%	74	46	
	< 20%	12	32	
Subalpine fir	> 20%	18	22	< 0.001
	Mean ± 20%	70	43	
	< 20%	12	35	
Non-Subalpine fir	> 20%	5	16	0.155
	Mean ± 20%	95	74	
	< 20%	0	11	
Glacier National Park	> 20%	15	24	0.095
	Mean ± 20%	66	49	
	< 20%	19	27	

Table 5.4. Condition of subalpine fir (SF) trees, saplings, and seedlings at eight plots along Hurricane Ridge, Olympic National Park. WA, including basal area of live and dead (decay classes 0-2) subalpine fir trees, percentage of total tree basal area that are subalpine fir, percentage of subalpine fir showing gout symptoms in each size class, and remotely sensed vegetation loss determined by LandTrendr with the normalized burn ratio (NBR). The NBR is an index of spectral reflectance used for detecting change in forest vegetation. It is calculated by combining near infrared (Band 4) and shortwave infrared (Band 7) bands in a ratio: Band 4 – Band 7/Band 4 + Band 7. Vegetation loss is a relative percentage calculated from pre- and post-disturbance NBR values.

	Live and dead SF > 12.7 cm DBH			Individual SF with gout			NBR decline magnitude
Plot	Live (m^2ha^{-1})	DC 0-2 (m^2ha^{-1})	Percent live BA SF versus all conifers	Live t (%)	Live sp, s (%)	DC 0-2 t (%)	1984-2008 (% vegetation loss)
1	29	23	58	88	95	43	35
2	32	18	100	61	33	36	56
3	11	11	17	0	0	13*	12
4	29	15	58	26	16	47	11
5	7	8	8	75	17	50	0
6	50	25	100	92	56	38	50
7	22	8	25	100	78	67	26
8	39	9	49	13	9	13	21

Abbreviations: DC = decay class; S = seedling; SP = sapling; T = tree (>12.7 cm dbh); DBH = diameter at breast height (1.37 m)

*Gouting only found on one dead tree in plot 3

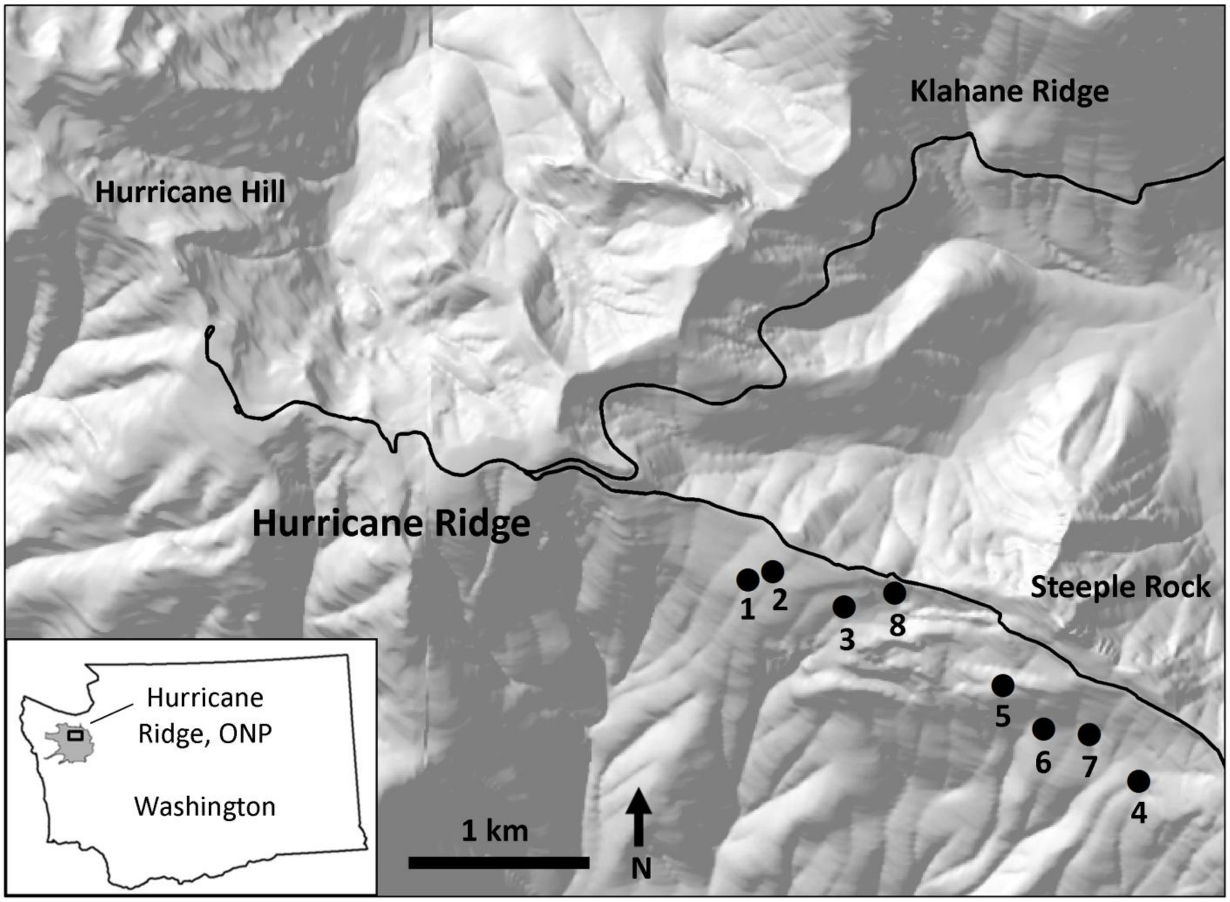
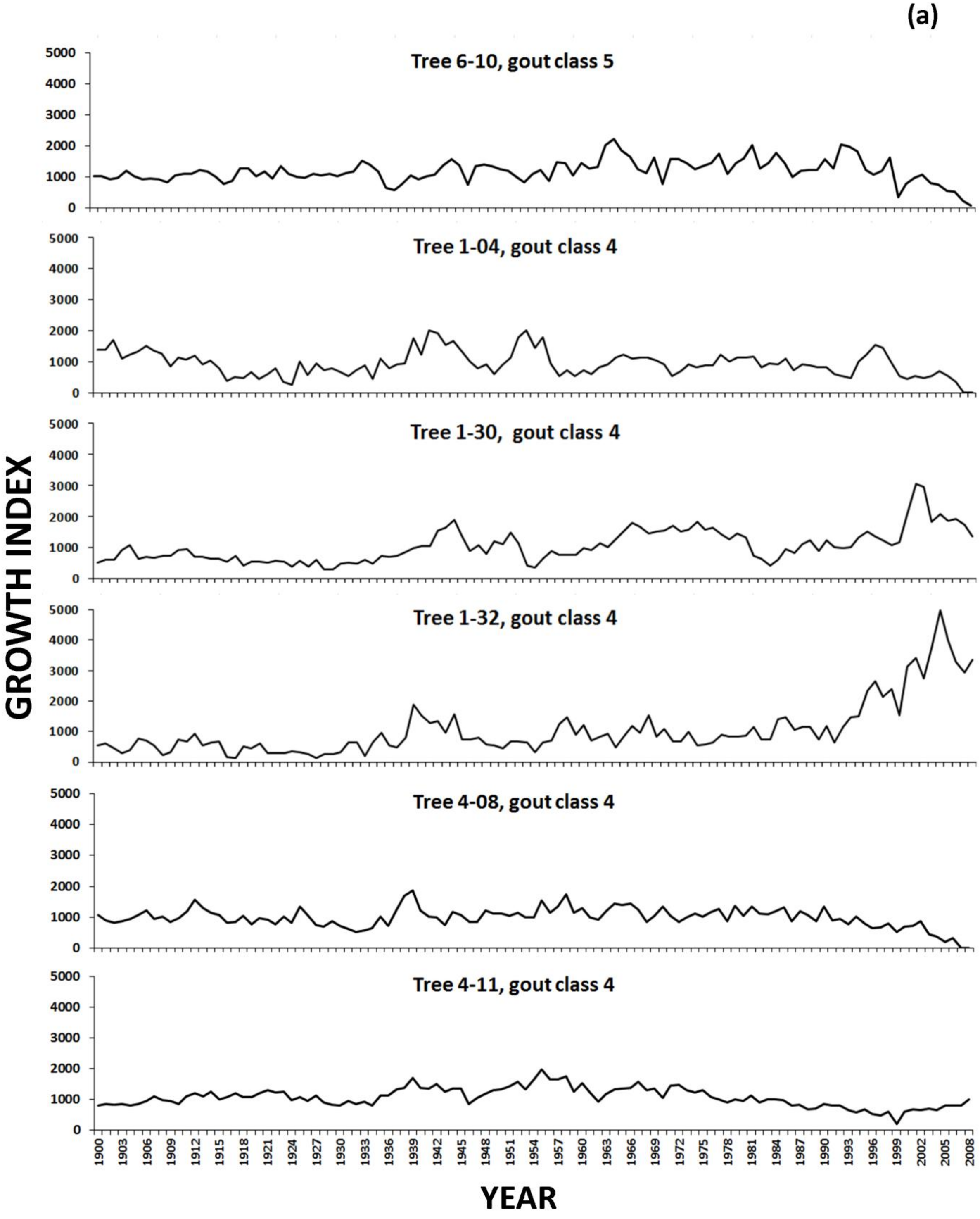


Figure 5.1. Locations of eight dendrochronology plots (solid black dots) along Hurricane Ridge, Olympic National Park, Washington, USA. Black lines represent Hurricane Ridge Road and Obstruction Road (above plots). This hillshade depiction of topography is derived from a 10-m digital elevation model.



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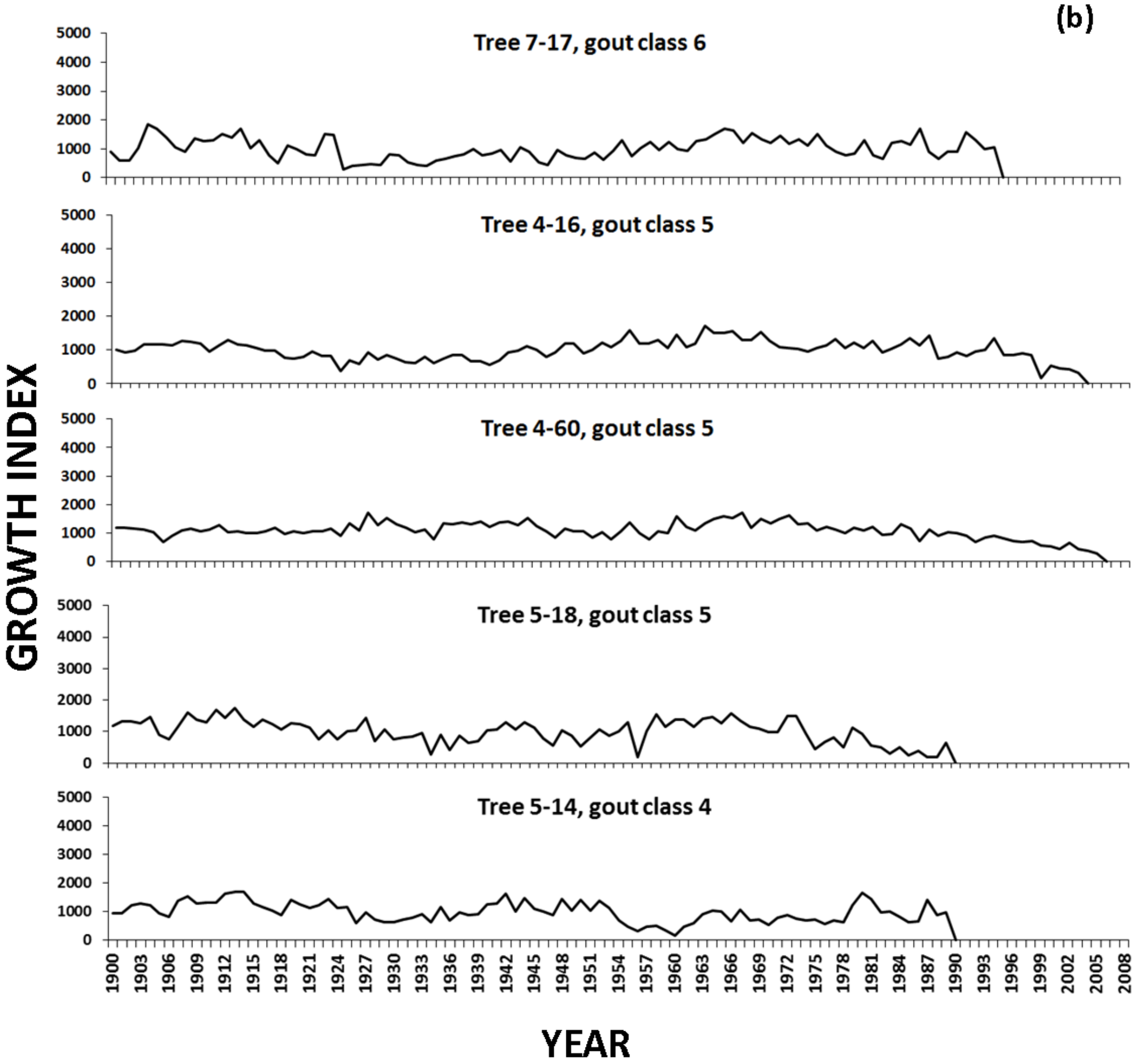

Figure 5.2. Growth time series from 1900 to 2008 for individual subalpine fir trees with high gout coverage grouped in two groups: (a) live, and (b) dead; cores were detrended with negative exponent. Trees in classes 4, 5, and 6 have balsam-woolly-adelgid-induced gout symptoms on branch nodes covering 51-75%, 76-95%, and 96-100% of the tree, respectively, in ground-based estimates. Trees are identified with the plot and tree number ([plot number]-[tree number]).

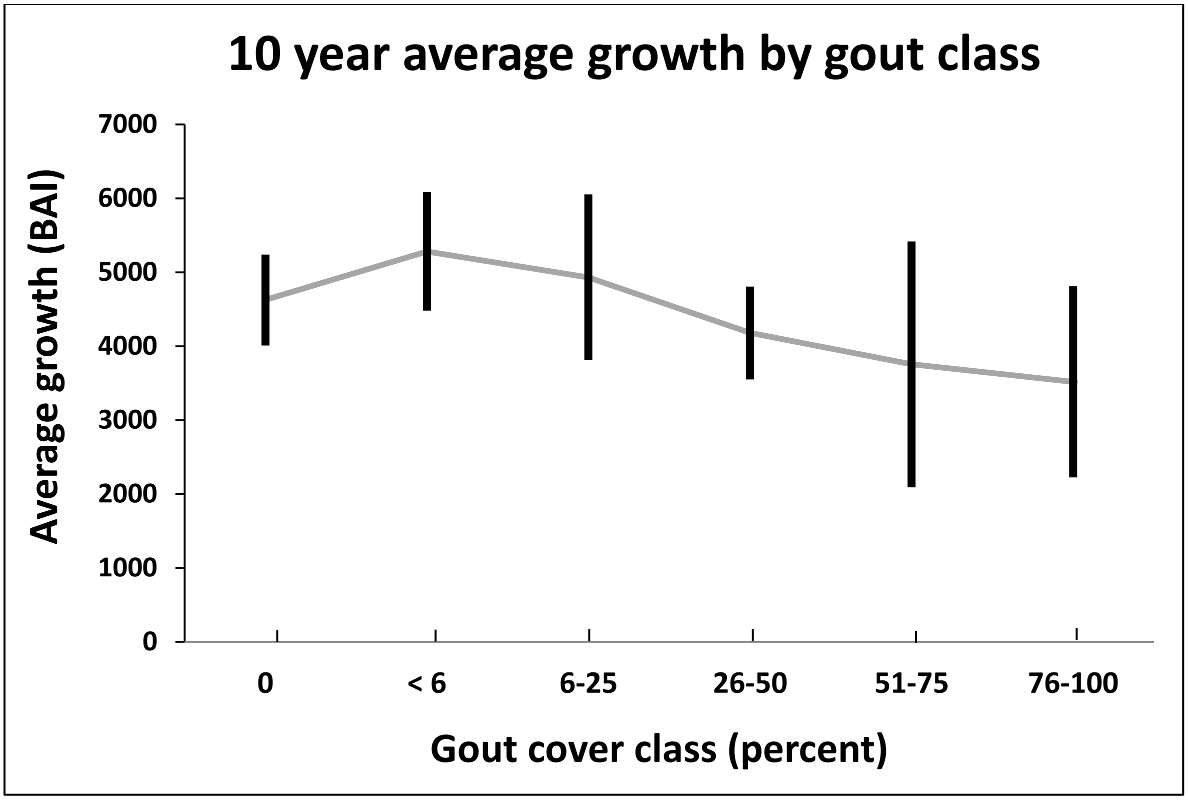

Figure 5.3. Average 10-year basal area increment (BAI) mm^2 growth in subalpine fir separated by gout cover class. Growth appears to decrease as gout cover increases. Black vertical lines represent the range in values for each gout class. The gray line indicates the trend of mean values. Due to high growth variability within gout classes growth is not significantly different between classes.

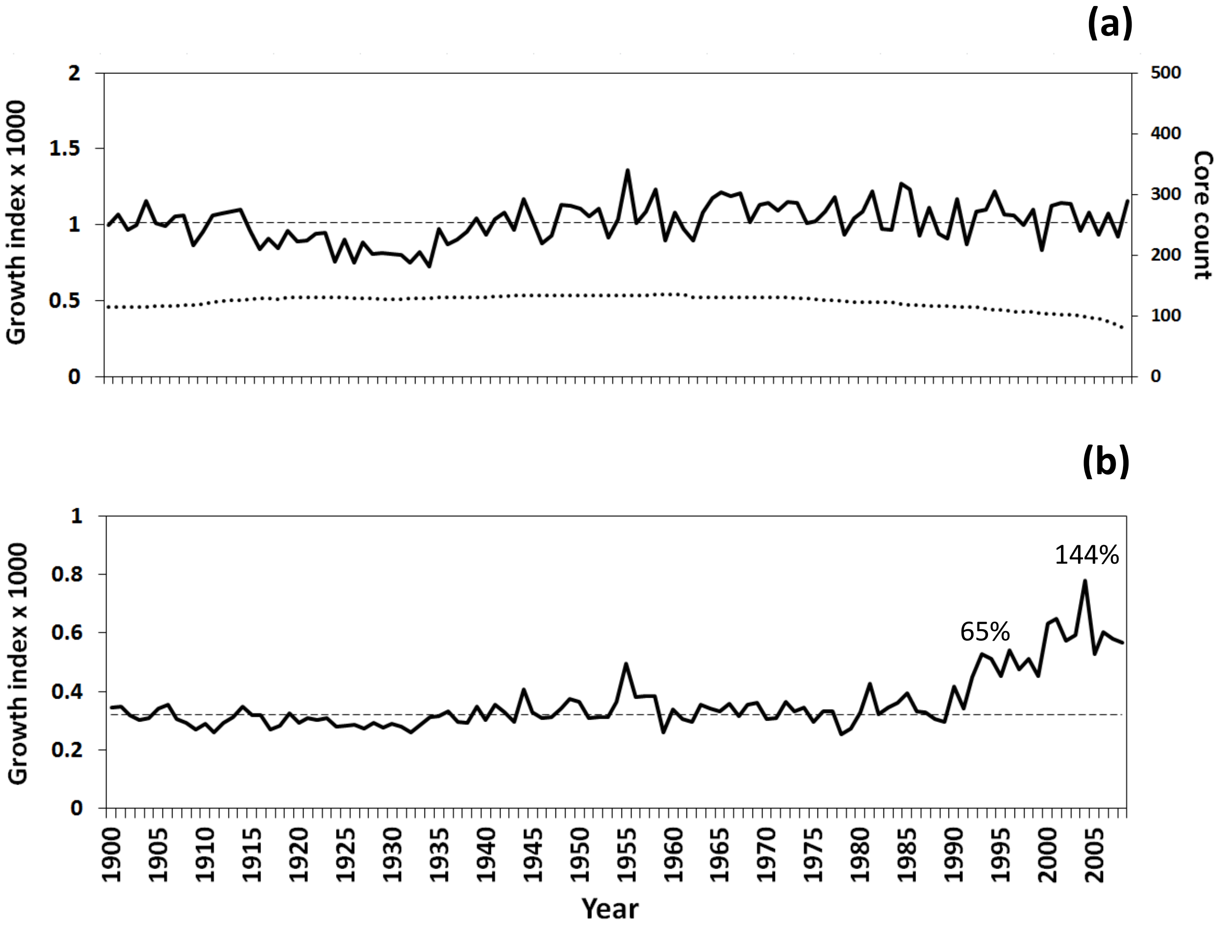

Figure 5.4. Mean tree growth (a) from 1900 to 2008 (solid black line) for all species (80% subalpine fir; 10% Pacific silver fir; and 10% other or unknown), live and dead, in all eight plots; tree core count for 148 cores (dotted line); and average mean growth across all years (dashed horizontal line) for reference. Standard deviation of tree growth (b) for 148 cores (solid black line); and average of standard deviation from 1900 to 1980 (dashed horizontal line) for reference. Forty-one percent of the trees had BWA-induced gout symptoms. Standard deviation increased by 65% in 1993 relative to the average standard deviation, and by 144% in 2004.

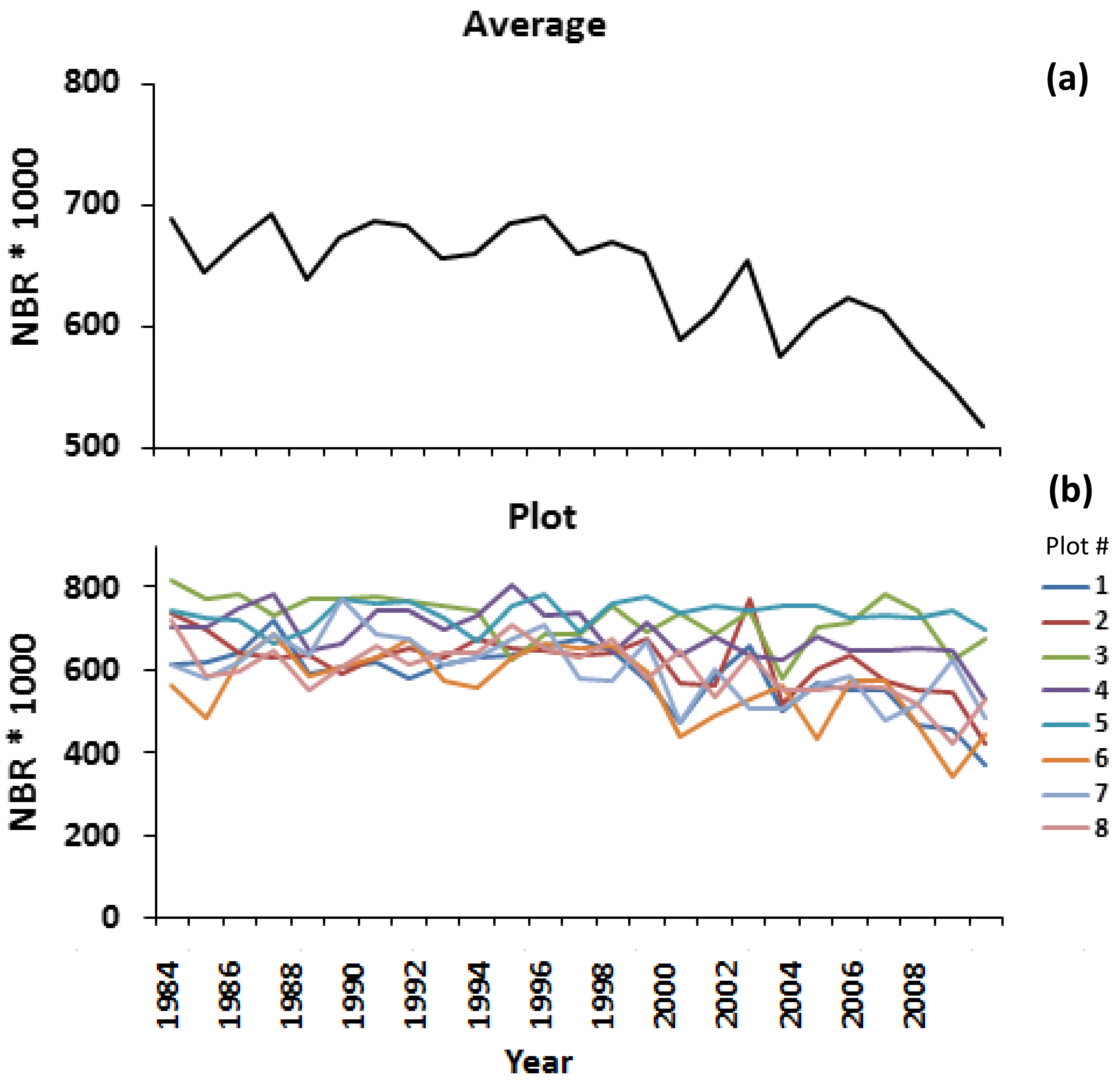

Figure 5.5. Time series of spectral reflectance measured with the normalized burn ratio (NBR) index (a) averaged across eight plots and (b) separately by plot. The NBR is an index of spectral reflectance used for detecting change in forest vegetation. It is calculated by combining near infrared (Band 4) and shortwave infrared (Band 7) bands in a ratio: Band 4 – Band – 7/Band 4 + Band 7. The onset of decline and a divergence among plots is evident after 1998. Plots 3 and 5 (green and blue lines) have more stable trajectories than the other plots, and also have the smallest percentage and basal area of subalpine fir; plot 3 also has low gout occurrence.

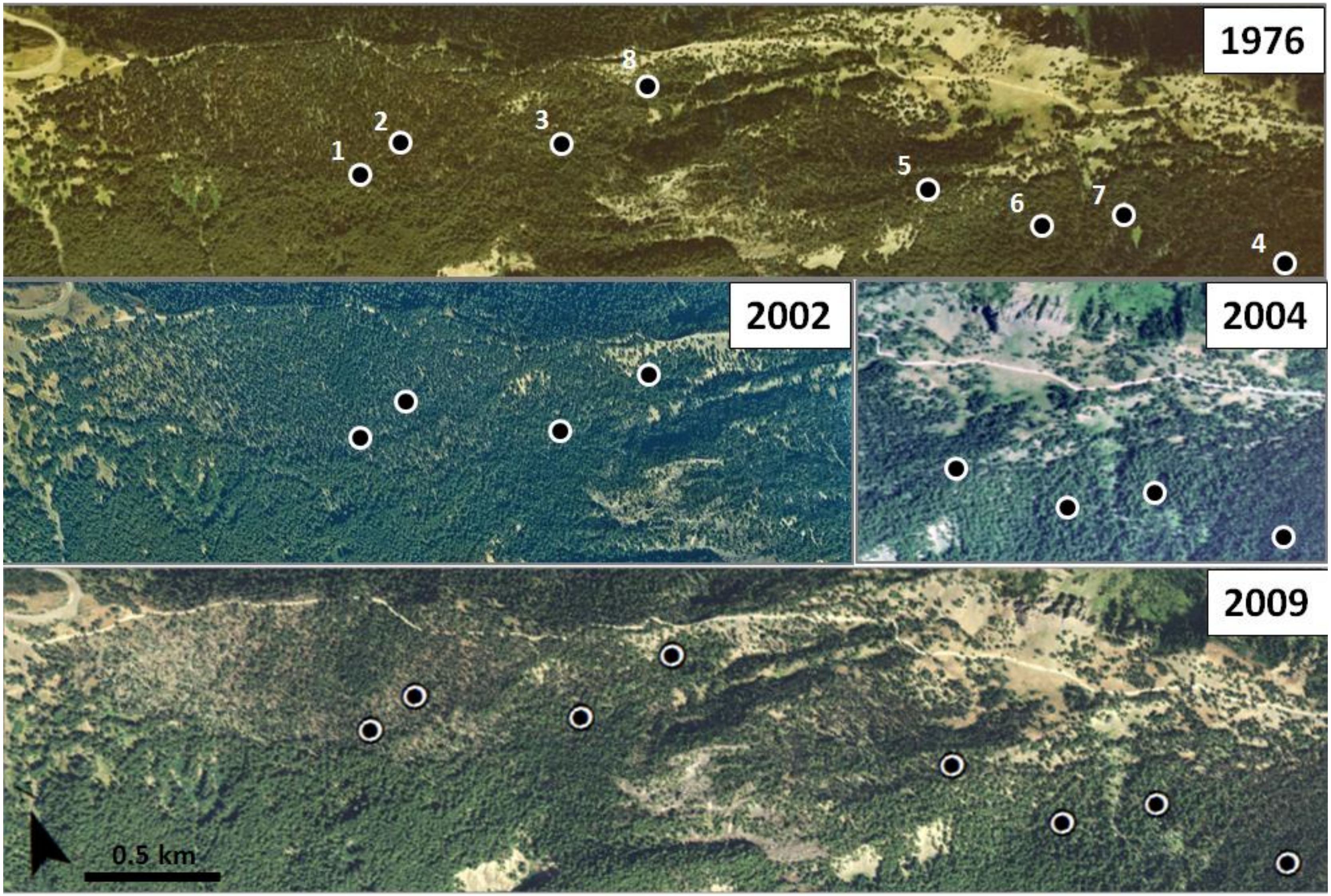

Figure 5.6. Aerial imagery showing vegetation change of the study area in three panels from 1976 to 2011. Circles mark sampling plot locations which are numbered in top panel. Change is more evident near plots 1 and 2.

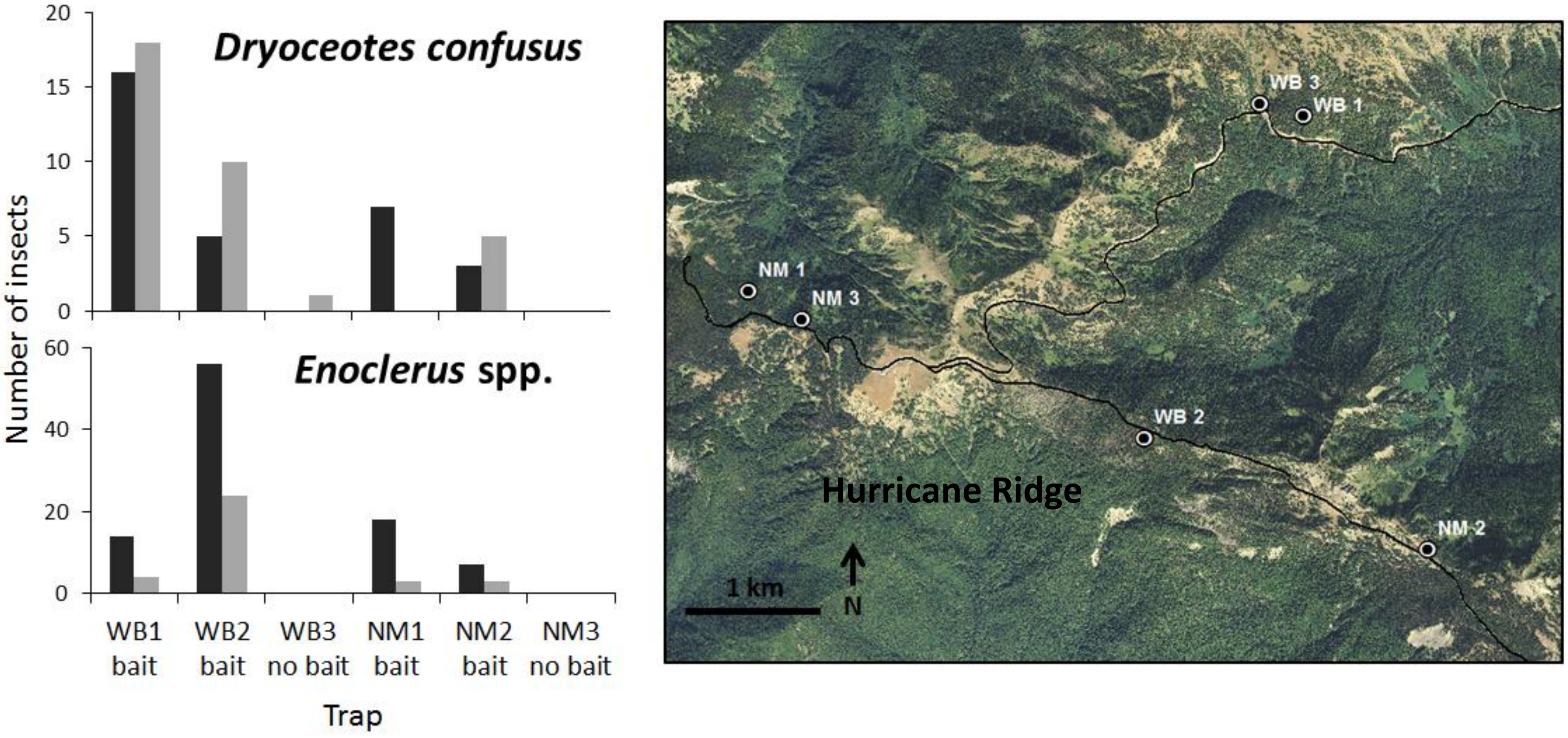

Figure 5.7. Collections from Lundgren funnel traps (blue dots) that were deployed during late summer of 2012 to verify presence of western balsam bark beetle (WBBB: *Dryoceotes confusus*) in the Hurricane Ridge area. Incidental collections of the bark beetle predator, *Enoclerus* spp. were also recorded. Traps were set on August 11 and collected on August 20 (black bars) and September 16 (gray bars) of 2012. Traps labeled WB were located in areas with beetle galleries; traps labeled NM were located in forest without evident bark beetle galleries.

Chapter 6

CONCLUSION AND MANAGEMENT CONSIDERATIONS

OVERVIEW

The balsam woolly adelgid (BWA), *Adelges piceae* Ratzeburg, has been the single most important agent of disturbance on the Olympic Peninsula in the last 20 or more years, predisposing subalpine fir trees to chronic and long-term health decline. Although this decline has been gradual and of low-magnitude, it has covered more area each year than the high-magnitude episodic disturbances caused by wind or fire. The scale and severity of damage that an invasive insect can have on forested ecosystems has been demonstrated by many studies (Hayes and Ragenovich 2001, Orwig 2002, Ellison et al. 2005) and the balsam woolly adelgid is another example.

For this dissertation, I developed a hierarchical framework to provide context for the evaluation of forest health decline in Olympic National Park (ONP) by identifying constraints and processes that influence landscape change patterns. This hierarchical approach involves the use of broad- to fine-scale methods to detect, interpret, and compare emergent patterns and mechanisms, and to extend spatial and temporal perspectives. Combined use of remote sensing, field research, and dendrochronology provided a more complete assessment of the spatial and temporal patterns and dynamics of change than would have been possible with one method alone. It also provided multiple lines of evidence and improved my ability to assess the relative importance of multiple factors impacting tree health.

FINDINGS

Observations at broad spatial and temporal scales via satellite imagery allowed a simultaneous investigation of the influence of climate, topography, vegetation, and abiotic and biotic processes on patterns of vegetation change (**Fig. 6.1**). Landsat time series data provided continuous and objective park-wide detection of vegetation change for 28 years with the normalized burn ratio (NBR) as the index of spectral reflectance (Kennedy et al. 2010). The temporal scale was long enough to detect a pattern of vegetation disturbance in ONP that was relatively stable until 1994 and then increased until 2008, after which it showed gradual signs of recovery.

Different patterns emerged when NBR decline was separated into high- and low-magnitude types representing abiotic and biotic disturbance, respectively. Low-magnitude disturbance covered the most area and was responsible for the gradually increasing pattern characteristic of a progressive biotic invasion. This was the dominant pattern observed in total vegetation decline for ONP. In contrast, high-magnitude decline was episodic and associated with winter weather or wind storm events with gusts > 95 km/hr.

A possible interaction between climate and invasion effects became evident when monthly indices for the Pacific decadal oscillation were compared with low-magnitude disturbance. The extent of disturbance increased during warm years. However, disturbance did not increase during warm years prior to 1994. This suggests that a biotic threshold, such as a critical population load, distribution, or host stress level was reached in 1994.

Topographic distribution of low-magnitude NBR decline showed several emergent patterns including a Gaussian distribution of decline at high elevation that matched the distribution of subalpine fir forest and aerial-detected symptoms of BWA. Low-magnitude disturbance consistently occurred on southern aspects, and this indicated an interaction with temperature or may have resulted from subalpine fir occurring more frequently on south-facing slopes. Disturbance was already prevalent at high elevation beginning in 1985 but covered less area than in later years, suggesting that the agent was already present by 1985 or before. These patterns concurred with expected patterns for BWA infestation. They were also consistent with aerial detection surveys that documented the first detection of BWA in ONP subalpine fir forest in 1970, with widespread coverage by 1989, and continued spread thereafter (USFS et al. 2013). The expansion of NBR decline into mid- and low-elevations that began around 2004 may have been related to bark beetle activity and spread of BWA into Pacific silver fir because both species were documented in ADS and field notes. High-magnitude topographic decline patterns were different each year as would be expected for individual short-term events. Similarities among patterns may be related to typical storm progression such as cyclonic wind storms arriving from the southwest that are channeled by topography.

My observations of signs and symptoms of fine-scale disturbance provided evidence for an association between BWA activity and declines in subalpine fir health. Tree health, mortality, and insect and disease were documented within and outside of affected areas in study plots distributed across the extent of subalpine fir on the Olympic Peninsula. Presence of BWA signs or symptoms was confirmed in 90% of plots with subalpine fir trees. Furthermore, live trees with BWA symptoms were less healthy than trees without BWA symptoms, and plots with BWA showed increased mortality and magnitude change, whereas plots without BWA did not. The mechanism causing tree health decline in BWA-infested trees has been well documented (Balch 1952, Mitchell 1967, Puritch 1971, Puritch and Petty 1971, Hollingsworth and Hain 1994). Other insect and disease symptoms were not consistently associated with affected areas but may contribute to tree mortality, most notably the western balsam bark beetle. Estimates of tree mortality (basal area of decay classes 0-2) and unhealthy subalpine fir were related to change in NBR, and this suggests that satellite imagery may be used to estimate tree condition for other areas with subalpine fir health decline.

My dendrochronology results from Hurricane Ridge, ONP provided fine-scale information on tree growth dynamics within a BWA-infested area, and extended the temporal observation of pattern to > 100 years. Mean growth of BWA-symptomatic trees decreased relative to non-symptomatic or lightly symptomatic trees but the difference was insignificant due to high variability. This variability may result from different infestation durations, a growth-release response in trees adjacent to dying trees, or a sudden growth increase where BWA may be infesting the tree bole as opposed to branch infestation, which is more common on the Olympic Peninsula. Increased variability in growth is an indicator of change, and I found a clear signal in the time series of growth standard deviation that began in 1993 and continued to the end of the sample period in 2008. This signal was unique over the entire 108-year time series. A comparison of change detection methods at Hurricane Ridge showed general agreement among aerial detection surveys (onset 1991), change in growth pattern (onset 1993), and decline in NBR (onset between 1994 and 1998). The onset year was different for each method because of differences among variables and in the scale of measurement. The approach used in this dissertation was facilitated by the hierarchical framework that guided research design, investigation, and interpretation of vegetation change patterns.

MANAGEMENT CONTEXT

Few management options are currently available for controlling BWA in natural forests. Biocontrol, chemical management, silvicultural methods, genetic resistance, and conservation are typical of management tactics that have been applied or proposed. Biocontrol efforts of BWA began in 1957 with the introduction of over 25 predatory species to North America from Europe and Asia (Mitchell and Wright 1967, Zilahi-balogh et al. 2002). Research on native predators was conducted simultaneously on species of mites, lacewings, flies, and beetles likely to feed on adelgids (Mitchell 1962). Although several non-native predators were established, and native species were identified (Humble 1994), none of these species have had an impact on BWA populations (Wallace and Hain 2002, Zilahi-balogh et al. 2002). These attempts at biological control failed because the predator species were generalist feeders, did not target the most strategic phase to reduce populations, or could not survive cold temperatures at higher elevations. More recent biocontrol research has focused on hemlock woolly adelgid with possible applications for BWA infestations at lower elevations (Potter et al. 2005, Jetton et al. 2011, Marten and Moore 2011).

Pesticides may be used to manage BWA in Christmas tree plantations, but are not feasible for subalpine forests due to cost, effort, and ecological impact. Each tree must be saturated with the chemical to be effective. Even in small plantations pesticides must be properly timed and applied every few years. Some insecticides have caused outbreaks of mites or endangered aquatic life (Potter et al. 2005).

Silvicultural methods of thinning to reduce spread or improve tree vigor have not been effective against BWA because it spreads readily across forests and increases on vigorous trees (Mitchell and Buffam 2001). Salvage logging and shorter rotations are suggested for forests managed for timber (Kanoti 2006), and planting non-host tree species can be used to reduce the risks of infestation (Ragenovich and Mitchell 2006). The balsam woolly adelgid is a non-quarantined, unregulated invasive species, and it is recommended that the potential spread of this pest be considered when processing or transporting infested wood (http://www.firewood.ca.gov/docs/bmps/arboristbmpspub2.pdf).

More resistant host tree species may evolve over time or be propagated with the aid of common garden breeding programs. BWA-infested fir trees have been observed to resist adelgid feeding by producing wound periderm (Balch 1952); trees may also to increase production of secondary metabolites that inhibit metamorphosis in insect species (Puritch and Nijholt 1974). Assuming that resistant forms of subalpine fir exist, survive, and regenerate, increasingly resistant subpopulations of this species may develop. Subalpine fir may have an adaptive advantage, because it reproduces sexually whereas BWA reproduces asexually. Offspring of BWA are genetically identical (except when mutations occur); therefore, the population may be unable to adapt to improved host tree defenses. Mitchell and Buffam (2001) observed that tree mortality often slowed after initially high mortality; however, it was unclear if the remaining live trees were resistant. As many as 50% of infested trees appeared to recover from BWA whereas other previously unaffected trees died several years later.

In the Appalachian Mountains where Fraser fir trees were killed by BWA several decades earlier than fir trees in the Pacific Northwest, researchers have seen two patterns. Firstly, the southern-most stands on the edge of the Fraser fir range suffered the greatest damage, and this may be related to greater moisture stress in the south (Hollingsworth and Hain 1994). Secondly, Fraser fir trees have been regenerating successfully at higher elevations (Dale et al. 1991, McManamay et al. 2011, Stehn et al. 2013). Because the Olympic Peninsula is located at the warmer end of the subalpine fir distribution, climate change increases the likelihood that subalpine fir will be confined (with BWA) to local high-elevation or cool-temperature refugia. Thus the long-term response of subalpine fir species to BWA is still uncertain.

Introduced exotic insect pests and pathogens have increasingly affected natural ecosystems within protected areas (De Poorter et al. 2007). The prevention of species introductions is the most effective management approach. The second most effective approach is early detection and eradication or containment (Tu 2009). However, it can be difficult to predict which species will become problematic, or when and where they will arrive. Furthermore, protection efforts and regulatory actions are typically directed toward tree species of economic importance. Balsam woolly adelgid is transported without restriction within and across state boundaries on Christmas trees, timber, and nursery stock (e.g., Jacobi et al. 2011), and these actions increase the speed and extent of invasion. Quarantines limit the movement of other exotic insects and pathogens such as the European gypsy moth (introduced in 1869), the emerald ash borer (introduced in 2002) (see http://www.aphis.usda.gov/), and sudden oak death (introduced around 1995) (Kliejunas 2010), but quarantines have not been applied to BWA, probably because of its gradual effect on tree species that are not economically important. The long-term impacts of BWA may provide insights into new and potentially unpredictable exotic agents that need further study. Tu (2009) proposed that impacts of invasive species need to be addressed at local, organizational, and national and international policy levels. Stricter regulations on the movement and importation of wood and nursery products may be the most effective approach for reducing the spread of exotic species (Roy et al. 2014).

Solving ecological problems caused by invasive species will require the development of effective land management strategies that involve resource manager and scientists with different areas of expertise at multiple levels of organization in a process of continuous learning, feedback, re-evaluation, and adaptation (e.g., Westley 2002). Effective management of protected areas is most effective when it proceeds collaboratively and iteratively, where science and management inform each other in an ongoing manner as new knowledge is gained and new challenges arise (Cole 2004).

REFERENCES

Balch, R. E. 1952. Studies of the Balsam Woolly Aphid, *Adelges piceae* (Ratz.) and Its Effects on Balsam Fir, *Abies balsamea* (L.) Mill. Ottawa, Canada.

Cole, D. N. 2004. Monitoring and management of recreation in protected areas: the contributions and limitations of science. Working Papers of the Finnish Forest Research Institute 2:10–17.

Dale, V. H., R. H. Gardner, and D. L. DeAngelis. 1991. Elevation-mediated effects of balsam woolly adelgid on southern Appalachian spruce-fir forests. Canadian Journal of Forest Research 21:1639–1648.

Ellison, A. M., M. S. Bank, B. D. Clinton, E. A. Colburn, K. Elliott, C. R. Ford, D. R. Foster, B. D. Kloeppel, J. D. Knoepp, G. M. Lovett, J. Mohan, D. A. Orwig, N. L. Rodenhouse, W. V Sobczak, K. A. Stinson, J. K. Stone, C. M. Swan, J. Thompson, B. Von Holle, and J. R. Webster. 2005. Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Frontiers in Ecology and the Environment 3:479–486.

Hayes, J. L., and I. Ragenovich. 2001. Non-native invasive forest insects of Eastern Oregon and Washington. Northwest Science 75:77–84.

Hollingsworth, R. G., and F. P. Hain. 1994. Effect of drought stress and infestation by the balsam woolly adelgid (Homoptera: Adelgidae) on abnormal wood production in Fraser fir. Canadian Journal of Forest Research 24:2295–2297.

Humble, L. M. 1994. Recovery of additional exotic predators of balsam woolly adelgid, *Adelges piceae* (Ratzeburg)(Homoptera: adelgidae), in British Columbia. Canadian Medical Association Journal 126:1101–1103.

Jacobi, W. R., B. A. Goodrich, and C. M. Cleaver. 2011. Firewood transport by national and state park campers: a risk for native or exotic tree pest movement. Arboriculture and Urban Forestry 37:126–138.

Jetton, R. M., J. F. Monahan, and F. P. Hain. 2011. Laboratory studies of feeding and oviposition preference, developmental performance, and survival of the predatory beetle, Sasajiscymnus tsugae on diets of the woolly adelgids, *Adelges tsugae* and *Adelges piceae*. Journal of Insect Science 11:1–14.

Kanoti, A. M. 2006. Relationship between balsam woolly adelgid damage, radial growth, climate and stand characteristics in eastern Maine. M.S. Thesis. Paper 434. University of Maine, Orono, Maine.

Kennedy, R. E., Z. Yang, and W. B. Cohen. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 1. LandTrendr — temporal segmentation algorithms. Remote Sensing of Environment 114:2897–2910.

Kliejunas, J. T. 2010. Sudden oak death and Phytophthora ramorum: a summary of the literature. 2010th edition. USDA Forest Service, Pacific Southwest Research Station. General Technical Report PSW-GTR-234., Albany, CA.

Marten, A. L., and C. C. Moore. 2011. An options based bioeconomic model for biological and chemical control of invasive species. Ecological Economics 70:2050–2061.

McManamay, R. H., L. M. Resler, J. B. Campbell, and R. A. McManamay. 2011. Assessing the impacts of balsam woolly adelgid (*Adelges piceae* Ratz.) and anthropogenic disturbance on the stand structure and mortality of Fraser fir [*Abies fraseri* (Pursh) Poir.] in the Black Mountains, North Carolina. Castanea 76:1–19.

Mitchell, R. G. 1962. Balsam woolly aphid predators native to Oregon and Washington.

Mitchell, R. G. 1967. Abnormal ray tissue in three true firs infested by the balsam woolly aphid. Forest Science 13:327–332.

Mitchell, R. G., and P. E. Buffam. 2001. Patterns of long-term balsam woolly adelgid infestations and effects in Oregon and Washington. Western Journal of Applied Forestry 16:121–126.

Mitchell, R. G., and K. H. Wright. 1967. Foreign predator introductions for control of the balsam woolly aphid in the Pacific Northwest. Journal of Economic Entomology 60:140–147.

Orwig, D. 2002. Ecosystem to regional impacts of introduced pests and pathogens: historical context, questions and issues. Journal of Biogeography:1471–1474.

De Poorter, M., S. Pagad, and M. I. Ulla. 2007. Invasive alien species and protected areas: a scoping report. Produced for the World Bank as a contribution to the Global Invasive Species Programme (GISP) 1:1–94.

Potter, K. M., J. Frampton, and J. Sidebottom. 2005. Impacts of balsam woolly adelgid on the southern Appalachian spruce-fir ecosystem and the North Carolina Christmas tree industry. Third Symposium on Hemlock Woolly Adelgid in the Eastern United States, Asheville, NC Feb. 1-3.

Puritch, G. S. 1971. Water permeability of the wood of grand fir. Journal of Experimental Botany 22:936–945.

Puritch, G. S., and W. W. Nijholt. 1974. Occurrence of juvabione-related compounds in grand fir and Pacific silver fir infested by balsam woolly adelgid. Canadian Journal of Botany 52:585–587.

Puritch, G. S., and J. A. Petty. 1971. Effect of balsam woolly aphid, *Adelges piceae* (Ratz.), infestation on the xylem of *Abies grandis* (Doug.) Lindl. Journal of Experimental Botany 22:946–952.

Ragenovich, I. R., and R. G. Mitchell. 2006. Balsam woolly adelgid. Forest Insect & Disease Leaflet 1182. USDA Forest Service.

Roy, B. A., H. M. Alexander, J. Davidson, F. T. Campbell, J. J. Burdon, R. Sniezko, and C. Brasier. 2014. Increasing forest loss worldwide from invasive pests requires new trade regulations. Frontiers in Ecology and the Environment 12:457–465.

Stehn, S. E., M. A. Jenkins, C. R. Webster, and S. Jose. 2013. Regeneration responses to exogenous disturbance gradients in southern Appalachian *Picea-Abies* forests. Forest Ecology and Management 289:98–105.

Tu, M. 2009. Assessing and managing invasive species within protected areas. *in* J. Ervin, editor. Protected Area Quick Guide Series. The Nature Conservancy, Arlington, VA.

USFS, WDNR, and ODF. 2013. Annual forest insect and disease aerial detection survey. 1955-2012. Forest Health and Protection, State and Private Forestry, Pacific Northwest Region, USDA Forest Service; Washington Department of Natural Resources, Resource Protection Division, Forest Health; Oregon Department of Forestry.

Wallace, M. S., and F. P. Hain. 2002. Summary of native and established predators of the hemlock woolly adelgid and the balsam woolly adelgid (Homoptera: Adelgidae) in the southeastern United States. Pages 141–149 *in* B. Onken, R. Reardon, and J. Lashomb, editors. Proceedings: Hemlock Woolly Adelgid in the Eastern United States Symposium. Rutgers University, East Brunswick, NJ.

Westley, F. 2002. The devil in the dynamics: adaptive management on the front lines. Pages 333–360 *in* L. H. Gunderson and C. S. Holling, editors. Panarchy. Island Press, Washington, D.C.

Zilahi-balogh, G., L. T. Kok, and S. M. Salom. 2002. A review of world-wide biological control efforts for the family Adelgidae. Pages 129–140 *in* B. Onken, R. Reardon, and J. Lashomb, editors. Proceedings: Hemlock Woolly Adelgid in the Eastern United States Symposium. Rutgers University, East Brunswick, NJ.

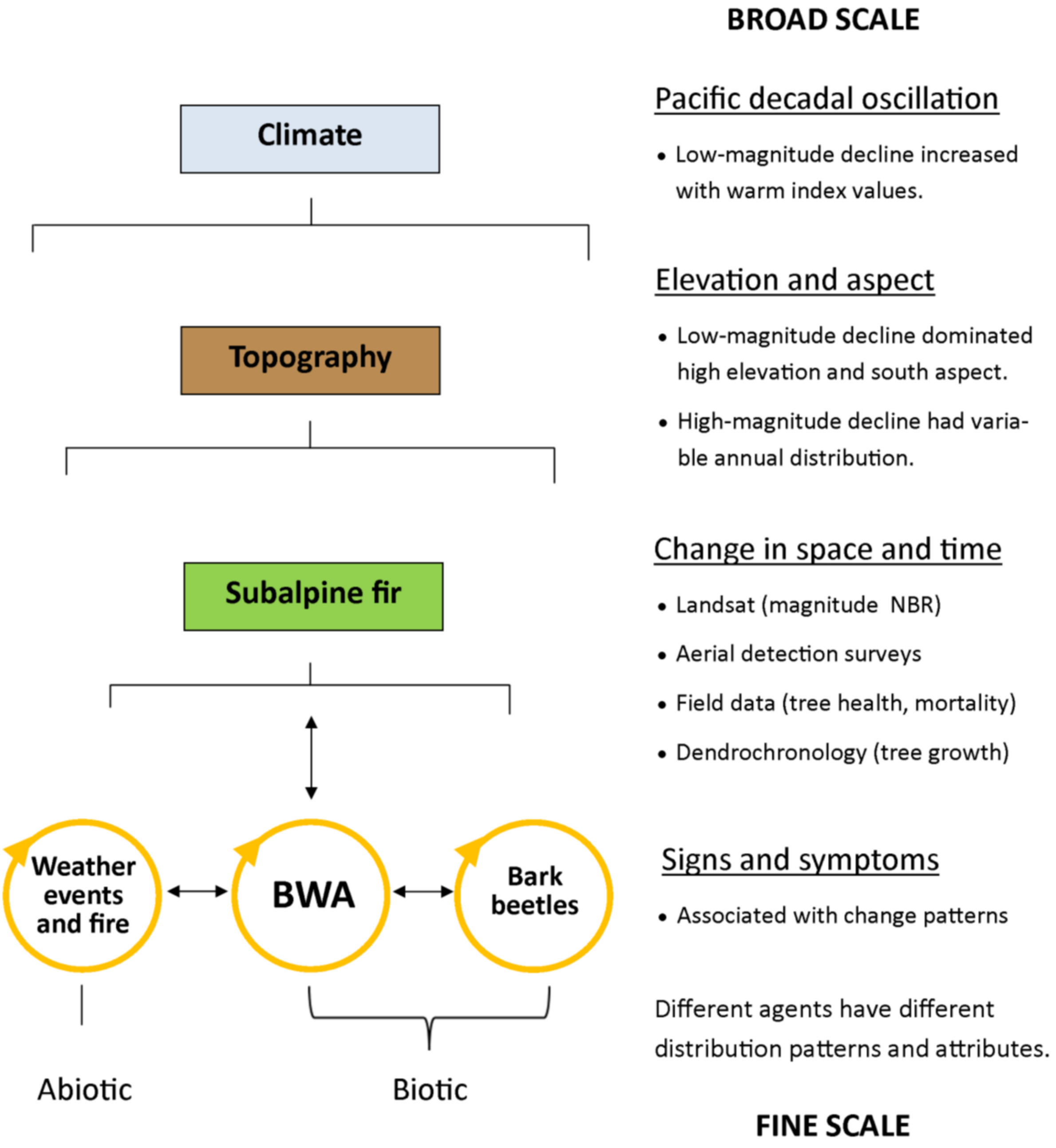

Figure 6.1. Hierarchical model for the Olympic Peninsula representing ecosystem dynamics detected in this research. Patterns of change in subalpine fir forest were detected with Landsat, aerial detection surveys, field data, and dendrochronology and compared to Pacific decadal oscillation index values and signs and symptoms of abiotic and biotic disturbance agents.