National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science

Natural Resource Condition Assessment

Olympic National Park

Natural Resource Report NPS/OLYM/NRR—2018/1826

ON THIS PAGE
Photograph of the Hoh Rainforest in Olympic National Park
Photograph courtesy of NPS

ON THE COVER
Photograph of the Olympic Mountains
Photograph courtesy of NPS/Dave Turner

Natural Resource Condition Assessment

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Natural Resource Report NPS/OLYM/NRR—2018/1826

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November 2018

U.S. Department of the Interior
National Park Service
Natural Resource Stewardship and Science
Fort Collins, Colorado

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Please cite this publication as:

McCaffery, R., and K. Jenkins (Eds.). 2018. Natural resource condition assessment: Olympic National Park. Natural Resource Report NPS/OLYM/NRR—2018/1826. National Park Service, Fort Collins, Colorado.

Please cite contributed chapters as (for example):

McCaffery, R., A. Woodward, K. Jenkins, and P. Haggerty. 2018. Ch. 2: Introduction and resource setting. Pages 4-32 *in* R. McCaffery and K. Jenkins, editors. Natural resource condition assessment: Olympic National Park. Natural Resource Report NPS/OLYM/NRR—2018/1826. National Park Service, Fort Collins, Colorado.

NPS 149/149534, November 2018

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Appendices

Executive Summary

The Natural Resource Assessment Program aims to document condition and trends of selected park resources while identifying emerging issues and information needs. This information is intended to serve as a platform for natural resource managers to use in developing future resource stewardship priorities and planning.

Northern Spotted Owl in OLYM. Photograph courtesy of NPS/Mandy Holmgren.

Olympic National Park (OLYM) on Washington's Olympic Peninsula protects remarkable examples of several Pacific Northwestern ecosystems, including the glacier-capped Olympic Mountains, old-growth temperate rainforests, pristine river systems, and wild Pacific coastline and islands. The park provides habitat for numerous plants and animals, including at least 16 animal and eight plant taxa endemic to the Olympic Peninsula. The park's lakes and rivers support over 70 stocks of Pacific salmonids and 29 native freshwater fish species. The rocky intertidal community is one of the most complex and diverse shorelines in the United States.

This assessment was concentrated on 12 focal resources in OLYM, each aligned with one of five guiding questions (Table E-1) of primary interest to park managers:

Table E-1. Guiding questions of primary interest to park managers.

Guiding Questions	Focal Resources
I. What are the effects of the loss of top predators on ecosystem health?	• Roosevelt Elk • Riparian Vegetation • River Geomorphology
II. What are the effects of resource harvest for human use on ecosystem function and sustainability of the resource?	• Pacific Salmonids • Razor Clams
III. What are the population trends and ecological impacts of key declining and/or sensitive species or communities?	• Olympic Marmots • Sea Stars • Amphibians
IV. What species or ecosystem functions are threatened by non-native or invasive species?	• Barred Owl (Northern Spotted Owl) • Balsam Woolly Adelgid (Subalpine Fir) • Invasive Non-native Plants (Native Plants)
V. What are the ecosystem effects of airborne contaminants?	• Air Quality and Contaminants

Although we conducted in-depth assessments on a limited number of resources, the general condition of several other physical and biological components of OLYM ecosystems is described in Chapter 2, along with an overview of park history and ecology. In Chapter 3, we describe how we chose our focal resources and measures, as well as our protocol for conducting the assessment. In Chapter 4, we provide a detailed assessment of each resource, with a summary of condition and trends. In Chapter 5, we further interpret and discuss the implications of each focal resource status, highlighting future areas for monitoring and research. We summarize the general findings from our assessments below.

E.1. What are the effects of the loss of top predators on ecosystem health?

The gray wolf (*Canis lupus*) is the only native top predator not currently present in OLYM. Our analysis centered on whether the loss of wolves has resulted in an overabundance of Roosevelt elk (*Cervus elaphus roosevelti*, a primary prey of wolves), and whether elk have affected the functional integrity of riparian vegetation and river system processes through trampling or browsing. Elk populations have left a clear imprint on riparian vegetation conditions of the west-side river valleys for over 100 years. However, historical increase in elk populations following wolf eradication also coincided with reduction in cougar populations, cessation of hunting, and increased livestock grazing, making it impossible to assign a single factor affecting elk populations, vegetation, and river dynamics. Counts of elk have declined during recent decades in limited areas where they have been monitored in OLYM.

Understory vegetation of riverine forests appears to be resilient to changes in elk abundance, with the possible exception of black cottonwood (*Populus balsamifera*) and bigleaf maple (*Acer macrophyllum*) recruitment. West-side river channels have widened over recent decades in response to increased hydrologic flows. Braidedness of the Hoh River has increased, ostensibly in response to increased sediment delivery associated with glacial retreat at the headwaters. We saw no evidence that the abundance of elk (whether affected by the extirpation of wolves or other factors) caused a fundamental shift in river channel dynamics either historically or in the present. All told, park riparian systems appear to be under the continual dynamic influences of natural disturbance regimes driven by herbivory, wind and fire, physical forcing of hydrologic flows, and perhaps also human land-use changes near the park boundary. At present, we are most concerned by the diminished recruitment of palatable riparian tree species, which may be an indirect effect of the absence of wolves.

We rated elk populations, riparian vegetation, and river systems as all warranting a moderate level of concern due primarily to the gradual decline in elk numbers, the lack of recruitment of cottonwood, and increased widening of the rivers. Disease and warming climate pose two distinct threats to the continued health of OLYM elk herds. Changing weather patterns, glacial retreat at the river's headwaters, and higher peak flows during winter are primary concerns for the integrity of riverine ecosystems in the park. The addition of wolves would likely change elk populations, herbivory patterns, and composition of riparian forests, with potential beneficial effects on other populations of wildlife. We are less certain that these changes would ameliorate the effects of projected increased winter flooding, glacial retreat, and sediment delivery on river channel dynamics. Expanded monitoring of elk populations, vegetation, and river geomorphology in lowland forests would position the park to better understand the potential effects of wolf reintroduction, disease, and changing climate on elk-vegetation-river processes.

E.2. What are the effects of resource harvest for human use on ecosystem function and sustainability of the resource?

We assessed population condition and the effects of human exploitation in 17 Pacific salmonid stocks found in five OLYM rivers and in razor clams (*Siliqua patula*), a highly sought shellfish at Kalaloch beach. We compiled harvest levels, escapement goals, and hatchery releases relative to historical information to assess current health and status of OLYM salmon stocks. Two-thirds of the 17 stocks assessed had some level of concern. The sustainability of spring and summer-run Chinook salmon (*Oncorhynchus tshawytscha*) in coastal rivers is of utmost concern. We identified salmon stocks where there are no established escapement goals yet harvest occurs annually (i.e., Sol Duc summer coho [*O. kisutch*], Quinault stocks), stocks where run sizes consistently missed escapement goals (e.g., spring Chinook), others that consistently met management goals (e.g., fall coho), and stocks that are particularly dependent upon OLYM waters for spawning and rearing (e.g., summer coho). We suggest that both harvest and hatcheries have the potential to affect the sustainability of Pacific salmonid resources in OLYM. We discuss opportunities available from new technologies and through continued cooperation among OLYM, State of Washington, and Treaty Tribes to expand monitoring and assessment to improve management.

Human harvest does not appear to be the primary factor currently affecting the sustainability of razor clams on OLYM beaches. The primary stressor to this population is the bacterial gill pathogen Nuclear Inclusion X (NIX), with other natural stressors potentially also affecting population dynamics. The condition of razor clams on OLYM beaches warrants a classification of high concern due to declining population size, average adult size, and size-class distribution. More targeted studies of razor clam population dynamics and its drivers, the NIX pathogen, and other potential stressors would help us understand how and when this population can be sustainably harvested.

E.3. What are the population trends and ecological impacts of key declining and/or sensitive species or communities?

We examined condition and trends of three species or taxonomic groups representing alpine, aquatic, and coastal ecosystems. Analyses of historical and recent surveys of Olympic marmot (*Marmota olympus*) colonies in high-elevation subalpine meadows confirmed the historical decline in marmot distribution, relative stability in marmot colonies over recent years in much of the park, and ongoing declines in the park's southwestern quadrant. We rated concern for the Olympic marmot as high due to past declines, concern over recent contractions of range, and potential threats from coyote predation, changing snowpack, and tree encroachment. Greater understanding of relationships between snow, tree encroachment, and coyote predation would help to clarify the utility of adaptive management approaches that have been considered as possible options for abating declining marmot populations.

Amphibian taxa are declining worldwide, and represent a sensitive community in OLYM's ecosystems. Past amphibian inventories point to wide distribution of many species of amphibians in the park, but trend of most species is unknown, including species of regional concern such as the Western toad (*Anaxyrus boreas*). Focal studies of a Cascades frog (*Rana cascadae*) population show a highly variable population size that warrants moderate concern. Ongoing threats to pond-breeding amphibians include climate change, non-native fishes, and disease. There is particular concern that reduced snowpack, warming temperatures, and non-native fish may interact to negatively impact pond-breeding amphibians in a warming climate. Enhanced monitoring of water temperature, disease, and changes in amphibian distributions would help determine the conservation status of the park's vulnerable amphibian populations.

Sea stars are key species in intertidal communities. In 2013, an outbreak of sea star wasting disease (SSWD) began causing mass mortalities of these species, particularly the ochre sea star (*Pisaster ochraceus*). We rated the concern over SSWD in OLYM sea star communities as high because *Pisaster* has declined in all monitoring plots and continues to be affected by SSWD, the prospect for species recovery is poorly understood, and warming ocean temperatures may pose recurring threats by SSWD. Other species have shown symptoms of SSWD, but population-level effects in those species are unknown. Continued monitoring of sea stars in OLYM is important for understanding the long-term effects of disease on these key species in the rocky intertidal community.

E.4. What species or ecosystem functions are threatened by non-native or invasive species?

For this question, we examined three very different invasive species or species groups, each with unique challenges to monitoring, control, and management: barred owls (*Strix varia*), balsam woolly adelgid (*Adelges piceae*), and invasive non-native plant species as a group. The barred owl has slowly displaced the northern spotted owl (*Strix occidentalis caurina*) in OLYM following its range expansion across western North America. Concern over the impact of barred owls on northern spotted owls is high due to the rapid decrease of spotted owl territory occupancy, increased occupancy of invasive barred owls, and increased local extinction rates in northern spotted owls since the late 1990s. At present the abundance of breeding pairs of spotted owls is only a small fraction of historical levels. Future research and monitoring could allow better estimation of barred owl distribution in OLYM, measure impacts of barred owl invasion on wildlife communities, and explore efficacy of potential management actions.

The balsam woolly adelgid (BWA), a European insect species that slowly kills fir (*Abies* spp.) trees over multiple years, was first detected in OLYM in about 1970 and is now widespread. Balsam woolly adelgid infestation in OLYM is of moderate concern, having shown increased severity and rate of spread up to 2007 and more recent stabilization. Although many trees have died, little is known of the consequences of large-scale subalpine fir (*Abies lasiocarpa*) mortality on subalpine communities, community recovery, and ecosystem functions. We identify areas of research to better identify genetic or environmental conditions that deter the incidence or severity of BWA, and to improve understanding of community and ecosystem responses to subalpine fir mortality.

The presence and potential impacts of invasive non-native plant species on the Olympic Peninsula have been recognized for over a century, yet we still know relatively little about the distribution and abundance of the vast majority of the non-native plant species found in OLYM. We documented a continuous increase in the number of invasive non-native plants in OLYM over time, with the highest establishment in developed areas and along roads. We rated the concern over invasive non-native plant species as significant based on the increasing threat and limited resources available for control efforts. Development of new monitoring and assessment methods can help better prioritize the limited resources available for control efforts and monitoring species responses to management. Additional research would help improve understanding the effects of invasive plant species on ecosystem functions, their role in various park ecosystems, and responses of native and non-native plant species to changing climates.

E.5. What are the ecosystem effects of airborne contaminants?

Air quality and associated contaminants affect core values of all national parks, including visitor experience and the health and integrity of its ecosystems. We examined trends in visibility, ozone, nitrogen and sulfur deposition, mercury, and other bioaccumulative toxins. We found variable resource concerns for different measures of air quality: no concerns over the effects of ozone on human or vegetation health; moderate concern over visibility; severe concerns regarding the effects of nitrogen and sulfur deposition; and moderate concern over mercury exposure levels. Continued monitoring of these pollutants could include studies to better understand how OLYM's ecosystems are affected by these stressors. While future trends and dominant sources of air pollution are unknown, OLYM can play a role in understanding how these contaminants impact the protected landscapes that we value.

Collectively, our assessment depicts complex systems with numerous interacting factors affecting park resources, communities, and ecosystems. While enlightening in some places, our analyses have also revealed processes underway (e.g., due to ongoing climate change) that may be largely out of our control to address or manage in meaningful ways. In other places, OLYM may have difficult decisions to make in allocating resources to better understand and manage resource conditions.

Syntheses and analyses provided in this assessment aim to further the discussion of resource management priorities and viable stewardship strategies for the future.

Acknowledgments

This project was funded primarily through an interagency agreement between the National Park Service (NPS) Natural Resource Condition Assessment (NRCA) Program and the U.S. Geological Survey's (USGS) Forest and Rangeland Ecosystem Science Center (FRESC). Funding received from the NPS NRCA Program, however, purchased only a small fraction of expertise contributing to this report. We could not have produced this report without the generous collaboration of the entire Olympic National Park (OLYM) Resources Management staff and the authors.

We are thankful for the scientific contributions and time provided by authors from: USGS (particularly FRESC, Western Fisheries Research Center, and Pacific Coastal and Marine Science Center), NPS (OLYM and Pacific West Region), National Oceanic and Atmospheric Administration (Northwest Fisheries Science Center), U.S. Bureau of Reclamation (Sedimentation and River Hydraulics Group), Oregon State University, Simon Fraser University, U.S. Forest Service, and USGS-Washington Water Science Center. We extend our thanks to Louise Johnson (Chief of Resources Management) and Jerry Freilich (Research Coordinator) at OLYM for providing a vision and ideas for this NRCA; to Jerry Freilich, Janet Coles, Catharine Copass for providing administrative support; and to Marsha Davis and Cathy Schwemm for providing programmatic direction during the preparation of this document.

We would like to thank a number of people who assisted us in compiling information for this report. Our thanks go out to Randy Cooper, Mike Gross, and David Low (Washington Department of Fish and Wildlife), Tyler Jurasin (Quinault Indian Nation), and Anna Geffre (OLYM) for helpful assistance in compiling the fisheries records presented in Chapter 4.2.1. We thank Carrie Cook-Tabor (U.S. Fish and Wildlife Service) for assistance compiling hatchery release data and Bill McMillan for providing historical canning records used in the salmon section. We thank Katherin Vittum for her assistance in compiling fisheries data into a relational database. We appreciate GIS support and mapping support and products provided by Patti Haggerty and Josh Logan (USGS), Roger Hoffman, (OLYM), and Katherine Beirne (NPS-North Coast and Cascades Network). We are grateful to Matt Dubeau (OLYM) for providing access to many key historical documents from the OLYM library and Casey Woodley (USGS-FRESC) for bibliographic assistance. The report benefitted from many valuable discussions regarding resource conditions in OLYM with Robert Beschta, Gordon Grant (Oregon State University), Josh Latterell (King County, Seattle, Washington), Joel Sankey and Chris Konrad (USGS), Andy Ritchie, Joshua Chenowith, and John Riedel (NPS), Jerry Franklin (University of Washington) and Tim Abbe (Natural Systems Design).

Lastly we appreciate the quality reviews of this manuscript provided by a large cast of experts, including (alphabetical order): Steve Acker, Erik Beever, Katherine Beirne, Bruce Bury, Catharine Copass, Janet Coles, Marsha Davis, Cheryl Decker, Katie Dugger, Colleen Flanagan Pritz, Irina Irvine, Drew Harvell, Brock Hoenes, Roger Hoffman, Blake Hossack, Douglas Houston, Kristin Jaeger, Louise Johnson, Terrie Klinger, Glen Kohler, Chris Magirl, Melissa Miner, Reginald Reisenbichler, Judy Rocchio, Regina Rocheforte, Cathy Schwemm, Pat Shaffroth, Peter Singleton, Stan Sovern, Ksienya Taylor, and Dirk Van Vuren. The authors take full responsibility for all interpretations in this report, but greatly appreciate the reviewers' many ideas and suggestions that have improved the final report.

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Acronyms

Table E-3. List of Acronyms.

Acronym	Definition
ADS	Aerial Detection Survey
AIC	Akaike's Information Criterion
GIS	Geographic Information System
ARD	NPS Air Resources Division
ARMI	Amphibian Research and Monitoring Initiative
BLM	US Bureau of Land Management
BWA	Balsam Woolly Adelgid
DNR	Department of Natural Resources
ENSO	El Niño Southern Oscillation
EPMT	Exotic Plant Management Teams
FRESC	Forest and Rangeland Ecosystem Science Center
GLO	General Land Office
GNN	Gradient Nearest Neighbor
GPS	Global Positioning System
I&M	Inventory and Monitoring
IDW	Inverse Distance Weighted
IHNV	Infectious Hematopoietic Necrosis Virus
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
LandTrendr	Landsat-based Detection of Trends in Disturbance and Recovery
LEKT	Lower Elwha Klallam Tribe
LEMMA	Landscape Ecology, Modeling, Mapping and Analysis
LiDAR	Light Detection and Ranging
MARINe	Mulit-agency Rocky Intertidal Network
MSY	Maximum Sustained Yield
NADP	National Atmospheric Deposition Program
NBR	Normalized Burn Ratio
NCCN	North Coast and Cascades Network
NDBC	National Data Buoy Center
NRCA	Natural Resource Condition Assessment
NIX	Nuclear Inclusion X
NOAA	National Oceanic and Atmospheric Administration
NPS	National Park Service
OA	Ocean Acidification

Table E-3 (continued). List of Acronyms.

Acronym	Definition
OLYM	Olympic National Park
ONF	Olympic National Forest
PDO	Pacific Decadal Oscillation
PIT	Passive Integrated Transponder
QDNR	Quinault Department of Natural Resources
QIN	Quinault Indian Nation
RMIS	Regional Mark Information System
RPRS	Research Permit and Reporting System
SASSI	Salmon and Steelhead Stock Inventory
SD	Standard Deviation
SFU	Simon Fraser University
SSWD	Sea Star Wasting Disease
UNESCO	United Nations Educational, Scientific, and Cultural Organization
USDA	US Department of Agriculture
USDI	US Department of the Interior
EPA	US Environmental Protection Agency
USFS	US Forest Service
USFWS	US Fish and Wildlife Service
USGS	US Geological Survey
WACAP	Western Airborne Contaminants Assessment Project
WDFW	Washington Department of Fish and Wildlife
WDG	Washington Department of Game

Chapter 1. NRCA Background Information

Natural Resource Condition Assessments (NRCAs) evaluate current conditions for a subset of natural resources and resource indicators in national park units, hereafter "parks." NRCAs also report on trends in resource condition (when possible), identify critical data gaps, and characterize a general level of confidence for study findings. The resources and indicators emphasized in a given project depend on the park's resource setting, status of resource stewardship planning and science in identifying high-priority indicators, and availability of data and expertise to assess current conditions for a variety of potential study resources and indicators.

NRCAs represent a relatively new approach to assessing and reporting on park resource conditions. They are meant to complement—not replace—traditional issue-and threat-based resource assessments. As distinguishing characteristics, all NRCAs:

- Are multi-disciplinary in scope;[1]
- Employ hierarchical indicator frameworks;[2]
- Identify or develop reference conditions/values for comparison against current conditions;[3]
- Emphasize spatial evaluation of conditions and GIS (map) products; [4]
- Summarize key findings by park areas; and [5]
- Follow national NRCA guidelines and standards for study design and reporting products.

NRCAs Strive to Provide…

- Credible condition reporting for a subset of important park natural resources and indicators
- Useful condition summaries by broader resource categories or topics, and by park areas

Although the primary objective of NRCAs is to report on current conditions relative to logical forms of reference conditions and values, NRCAs also report on trends, when appropriate (i.e., when the underlying data and methods support such reporting), as well as influences on resource conditions. These influences may include past activities or conditions that provide a helpful context for

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[1] The breadth of natural resources and number/type of indicators evaluated will vary by park.

[2] Frameworks help guide a multi-disciplinary selection of indicators and subsequent "roll up" and reporting of data for measures ⇨ conditions for indicators ⇨ condition summaries by broader topics and park areas.

[3] NRCAs must consider ecologically-based reference conditions, must also consider applicable legal and regulatory standards, and can consider other management-specified condition objectives or targets; each study indicator can be evaluated against one or more types of logical reference conditions. Reference values can be expressed in qualitative to quantitative terms, as a single value or range of values; they represent desirable resource conditions or, alternatively, condition states that we wish to avoid or that require a follow-up response (e.g., ecological thresholds or management "triggers").

[4] As possible and appropriate, NRCAs describe condition gradients or differences across a park for important natural resources and study indicators through a set of GIS coverages and map products.

[5] In addition to reporting on indicator-level conditions, investigators are asked to take a bigger picture (more holistic) view and summarize overall findings and provide suggestions to managers on an area-by-area basis: 1) by park ecosystem/habitat types or watersheds, and 2) for other park areas as requested.

understanding current conditions, and/or present-day threats and stressors that are best interpreted at park, watershed, or landscape scales (though NRCAs do not report on condition status for land areas and natural resources beyond park boundaries). Intensive cause-and-effect analyses of threats and stressors, and development of detailed treatment options, are outside the scope of NRCAs.
Due to their modest funding, relatively quick timeframe for completion, and reliance on existing data and information, NRCAs are not intended to be exhaustive. Their methodology typically involves an informal synthesis of scientific data and information from multiple and diverse sources. Level of rigor and statistical repeatability will vary by resource or indicator, reflecting differences in existing data and knowledge bases across the varied study components.

The credibility of NRCA results is derived from the data, methods, and reference values used in the project work, which are designed to be appropriate for the stated purpose of the project, as well as adequately documented. For each study indicator for which current condition or trend is reported, we will identify critical data gaps and describe the level of confidence in at least qualitative terms. Involvement of park staff and National Park Service (NPS) subject-matter experts at critical points during the project timeline is also important. These staff will be asked to assist with the selection of study indicators; recommend data sets, methods, and reference conditions and values; and help provide a multi-disciplinary review of draft study findings and products.

NRCAs can yield new insights about current park resource conditions, but, in many cases, their greatest value may be the development of useful documentation regarding known or suspected resource conditions within parks. Reporting products can help park managers as they think about near-term workload priorities, frame data and study needs for important park resources, and communicate messages about current park resource conditions to various audiences. A successful NRCA delivers science-based information that is both credible and has practical uses for a variety of park decision making, planning, and partnership activities.

Important NRCA Success Factors

- Obtaining good input from park staff and other NPS subject-matter experts at critical points in the project timeline
- Using study frameworks that accommodate meaningful condition reporting at multiple levels (measures ⇨ indicators ⇨ broader resource topics and park areas)
- Building credibility by clearly documenting the data and methods used, critical data gaps, and level of confidence for indicator-level condition findings

However, it is important to note that NRCAs do not establish management targets for study indicators. That process must occur through park planning and management activities. What an NRCA can do is deliver science-based information that will assist park managers in their ongoing, long-term efforts to describe and quantify a park’s desired resource conditions and management targets. In the near term, NRCA findings assist strategic park resource planning[6] and help parks to report on government accountability measures.[7] In addition, although in-depth analysis of the effects of climate change on park natural resources is outside the scope of NRCAs, the condition analyses and data sets developed for NRCAs will be useful for park-level climate-change studies and planning efforts.

NRCAs also provide a useful complement to rigorous NPS science support programs, such as the NPS Natural Resources Inventory & Monitoring (I&M) Program.[8] For example, NRCAs can provide current condition estimates and help establish reference conditions, or baseline values, for some of a park's vital signs monitoring indicators. They can also draw upon non-NPS data to help evaluate current conditions for those same vital signs. In some cases, I&M data sets are incorporated into NRCA analyses and reporting products.

NRCA Reporting Products…

Provide a credible, snapshot-in-time evaluation for a subset of important park natural resources and indicators, to help park managers:

- Direct limited staff and funding resources to park areas and natural resources that represent high need and/or high opportunity situations **(near-term operational planning and management)**
- Improve understanding and quantification for desired conditions for the park's "fundamental" and "other important" natural resources and values **(longer-term strategic planning)**
- Communicate succinct messages regarding current resource conditions to government program managers, to Congress, and to the general public **("resource condition status" reporting)**

Over the next several years, the NPS plans to fund an NRCA project for each of the approximately 270 parks served by the NPS I&M Program. For more information visit the NRCA Program website.

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[6] An NRCA can be useful during the development of a park's Resource Stewardship Strategy (RSS) and can also be tailored to act as a post-RSS project.

[7] While accountability reporting measures are subject to change, the spatial and reference-based condition data provided by NRCAs will be useful for most forms of "resource condition status" reporting as may be required by the NPS, the Department of the Interior, or the Office of Management and Budget.

[8] The I&M program consists of 32 networks nationwide that are implementing "vital signs" monitoring in order to assess the condition of park ecosystems and develop a stronger scientific basis for stewardship and management of natural resources across the National Park System. "Vital signs" are a subset of physical, chemical, and biological elements and processes of park ecosystems that are selected to represent the overall health or condition of park resources, known or hypothesized effects of stressors, or elements that have important human values.

Chapter 2. Introduction and Resource Setting

Rebecca McCaffery, Andrea Woodward, Kurt Jenkins, and Patricia Haggerty

2.1. Introduction

2.1.1. Background and Geographic Setting

Olympic National Park is located in the Olympic Mountains physiographic province on the Olympic Peninsula of Washington, encompassing 373,383 ha (922,650 ac) in the interior and coastal regions of Washington's Olympic Peninsula (Figure 2.1.1). The Olympic Peninsula, on the northwestern tip of Washington State, is bounded by the Hood Canal to the east, the Strait of Juan de Fuca to the north, and the Pacific Ocean to the west. The core of the park is largely bounded on the west, south and east by 260,382 ha (643,419 ac) of National Forest lands, of which 35,720 ha (88,265 ac) on the eastern and southern borders of the park are designated wilderness (Figure 2.1.2). Additionally, a significant portion of the western boundary of the park abuts Washington Department of Natural Resources (DNR) land, including the Olympic State Experimental Forest, which encompasses about 109,265 ha (270,000 ac) of state trust lands for the purposes of learning how to better integrate timber harvest and ecosystem values (e.g., native wildlife species and biological diversity). The southwestern and coastal portions of the park are bounded by a combination of State, Tribal, and private land (Figure 2.1.2). The administrative headquarters of OLYM is located in Port Angeles, WA, which lies 132 km (82 mi) west of Seattle.

Olympic Mountains. Photograph courtesy of NPS/Bill Baccus.

Figure 2.1.1. Geographical setting of Olympic National Park. Source: boundary, NPS; basemap, ESRI 2008 North America EarthSat NaturalVue Global Landsat Mosaic.

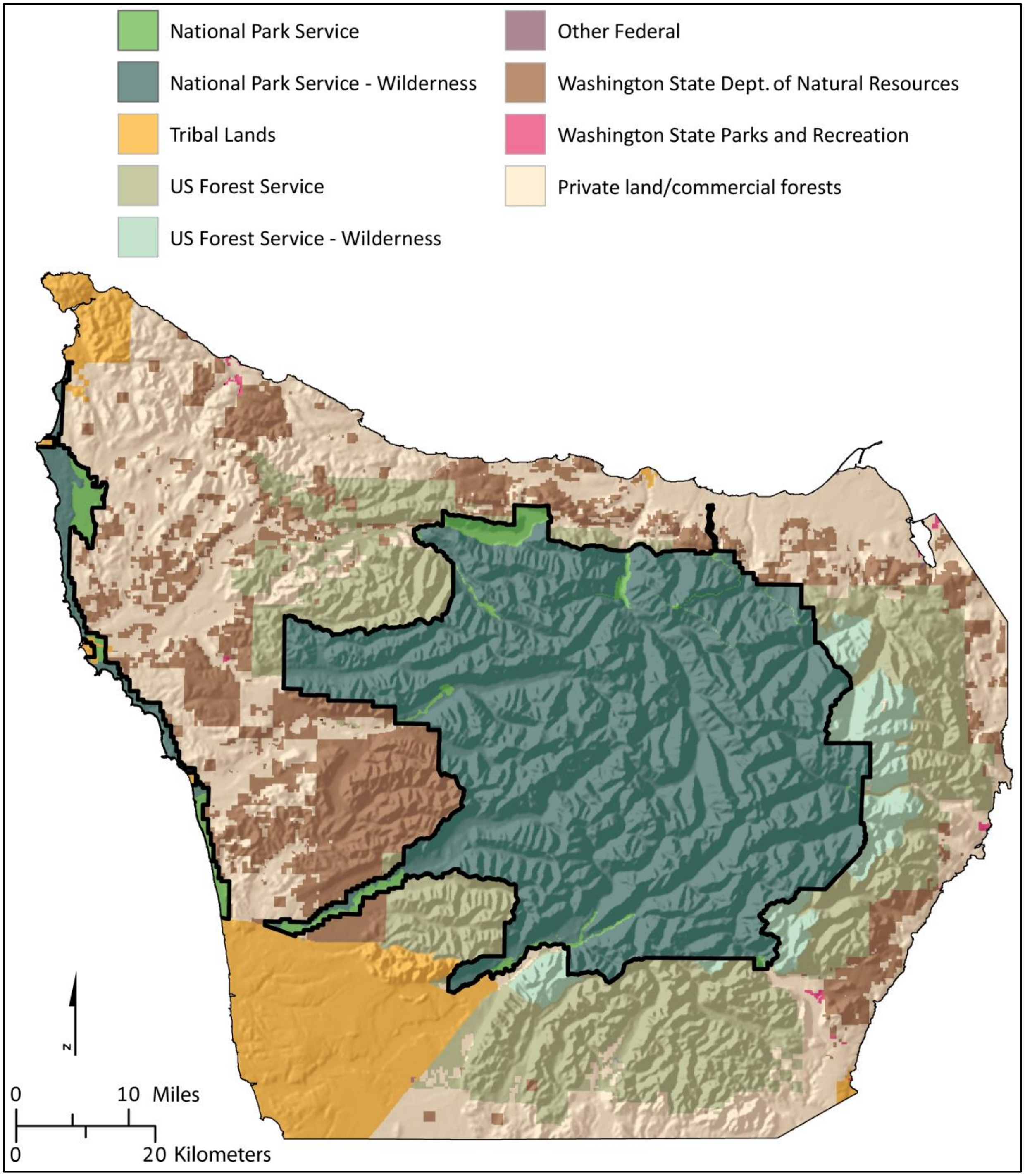

Figure 2.1.2. Olympic National Park and adjacent land ownership. Source: NPS.

Olympic National Park encompasses three distinct ecosystems and includes one of the largest wilderness areas in the contiguous United States. The park contains the glacier-capped Olympic Mountains, old-growth temperate rainforest, and over 103.5 km (64.3 mi) of wild Pacific coast and islands. These ecosystems protect unique habitats and life forms. For example, at least 16 animal and eight plant taxa are endemic to the Olympic Peninsula. OLYM contains some of the last remaining undisturbed and connected aquatic ecosystems for important anadromous fish species. The park has 11 major river basins and 646 high mountain lakes, and two large lowland lakes. These aquatic habitats support over 70 unique stocks of Pacific salmonids, 29 native freshwater fish species, and one endemic fish species. Olympic National Park protects some of the best remaining old-growth temperate rainforest stands in the United States, which provide habitat for countless plants and animals. The rocky intertidal community protected by OLYM is considered to be one of the most complex and diverse shoreline communities in the United States. Olympic National Park also protects important cultural resources, which include over 650 archeological sites, hundreds of ethnographic sites, 31 cultural landscapes, and 16 historic districts.

2.1.2. Park History and Enabling Legislation

President Franklin Roosevelt signed the bill creating OLYM on June 29, 1938 (35 Stat. 2247). The purpose of the enabling legislation was to

> *preserve for the benefit, use, and enjoyment of the people, the finest sample of primeval forests of Sitka spruce, western hemlock, Douglas fir, and western red cedar in the entire United States; to provide suitable winter range and permanent protection for the herds of native Roosevelt elk and other wildlife indigenous to the area; to conserve and render available to the people, for recreational use, this outstanding mountainous country, containing numerous glaciers and perpetual snow fields, and a portion of the surrounding verdant forests together with a narrow strip along the beautiful Washington coast.*

The area that is now OLYM has a rich history of land use and protection on the Olympic Peninsula that reflects the competing interests of groups intent on capitalizing on the wealth of natural resources present on the peninsula, and groups seeking to protect the Olympic Peninsula's ecosystems from human exploitation (Lien 2000). The first protection of Olympic forests came with the designation of portions of the peninsula as the Olympic Forest Preserve in 1897 by President Grover Cleveland. Over 809,372 ha (2 million ac) of Olympic forests were protected from development with this designation, which encompassed the mountains as well as significant portions of the west-side forests (Figure 2.1.3a). However, this acreage was quickly diminished under pressure by timber and settlement interests during the McKinley administration, when over 303,514 ha (750,000 ac) were removed from the reserve, mostly in the west-side forests (Lien 2000). In 1909, under the urging of Tacoma Congressman W.E. Humphreys, President Theodore Roosevelt used the authority of the 1909 Antiquities Act to set aside 248,882 ha (615,000 ac) of the Olympic Mountains as the Mt. Olympus National Monument (Figure 2.1.3b). Again, this protection was subsequently reduced under President Woodrow Wilson, who removed 68,797 ha (170,000 ac) of forest from the Monument in 1915, succumbing to timber interests (Figure 2.1.3c). Throughout World War I and the 1920s, timber harvest continued to intensify, as did a movement by various local and regional conservation groups to create a national park (Lien 2000).

The pressure to create the park intensified in the 1930s, and a new effort was made when Franklin D. Roosevelt became president in 1933. After visiting the peninsula in 1937, where popular support for the creation of a park was widespread, President Roosevelt signed the bill creating Olympic National Park (Figure 2.1.3d). The park initially encompassed 258,303 ha (638,280 ac), but provisions were made in the House Report to expand the park to up to 363,526 ha (898,292 ac), which would include the narrow strip along the Washington coast. The first expansion occurred in 1943, with notable expansions along west-side rivers (Figure 2.1.3e). In 1953, most of the park’s current coastal strip was added under President Dwight Eisenhower (Figure 2.1.3f). In 1976, the park was further expanded by Congress extending the coastal strip northward and including a narrow strip on the east shore of Lake Ozette (Figure 2.1.3g).

As an international recognition of the unique biological and cultural resources of this site, OLYM was designated an International Biosphere Reserve in 1976 and World Heritage Site in 1981, both honorary designations by United National Educational, Scientific, and Cultural Organization (UNESCO). In 1988, Congress passed a bill that designated approximately 95% of the land in OLYM as Wilderness as defined in the 1964 Wilderness Act. Three smaller boundary changes (not pictured) occurred on the east side of the park in 1986 and in the coastal strip in 2010 and 2012. The park currently encompasses 373,383 ha (922,650 ac; Figure 2.1.3h).

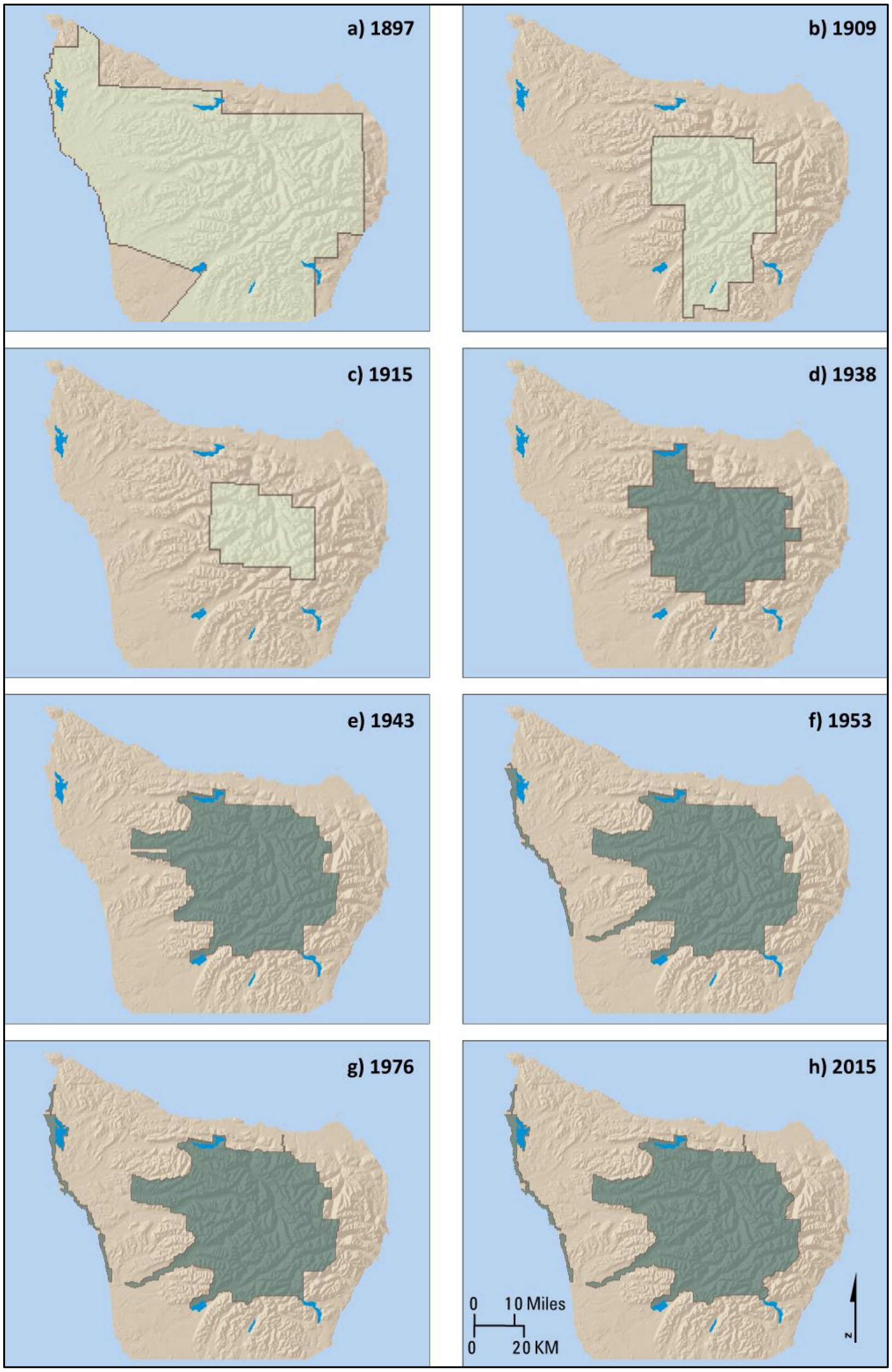

Figure 2.1.3. Significant boundary changes for the (a) Olympic Forest Preserve, (b-c) Mt. Olympus National Monument, and (d-h) Olympic National Park from 1897 to the present. Pale green indicates protected designations prior to the establishment of the park, and dark green represents boundary changes once the national park was established. See text for descriptions of each boundary change. Source: Roger Hoffman, NPS.

2.1.3. Visitation Statistics

Visitation to the park has grown over time but leveled off in recent decades. Peak visitation occurred in 1997, when over 3.8 million visitors came to the park (Figure 2.1.4). In the past ten years visitation has declined somewhat from that peak, with numbers of annual visitors fluctuating between approximately 2.8 and 3.2 million people (Figure 2.1.4). In 2015, the most visited area in the park was Lake Crescent, which was followed by the park's coastal areas (Mora, Kalaloch, and Ozette districts combined). Of total visitors, only 11.4% stayed overnight, and 23.9% of those visitors stayed in the backcountry (i.e., wilderness). The remaining visitors used concession lodging and camping or front country camp sites. The proportion of visitors to different areas of the park, and the proportion of visitors using the front versus backcountry has been relatively similar across recent years.

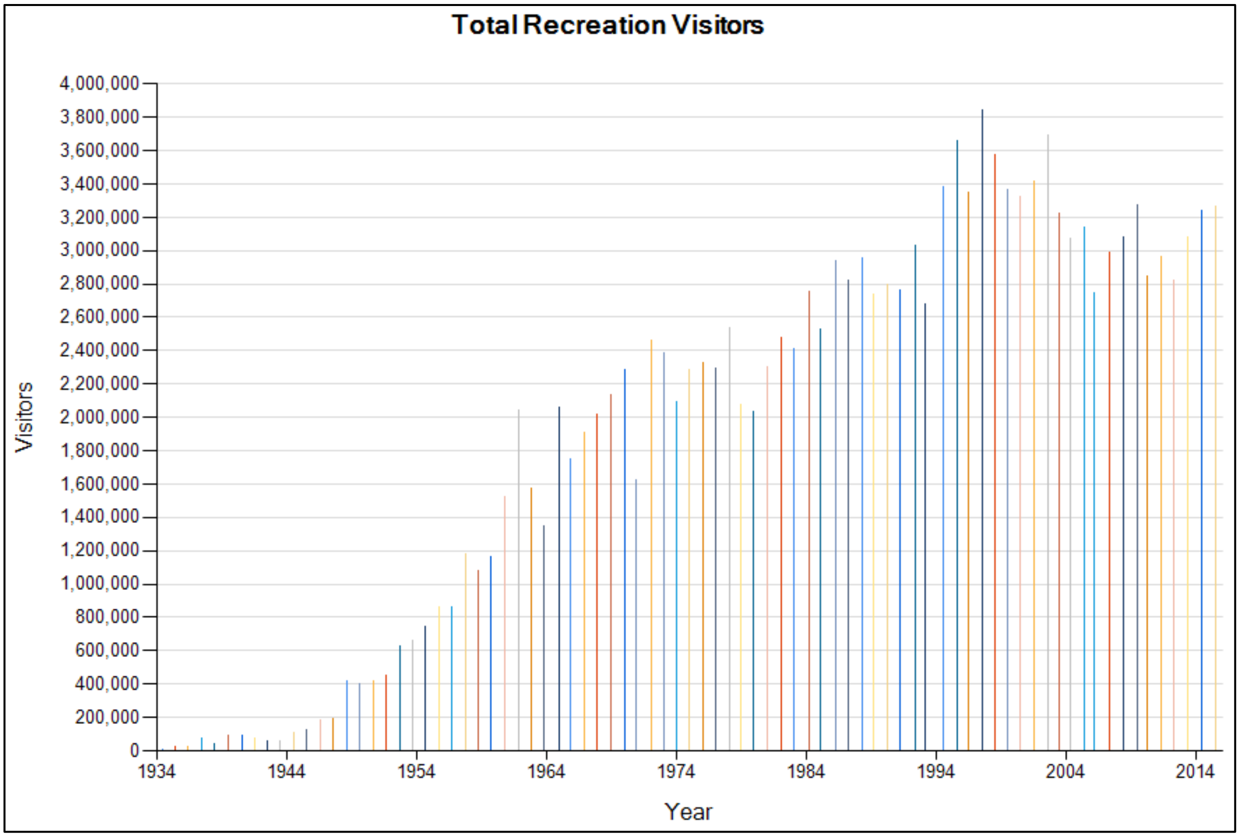

Figure 2.1.4. Visitation summary for Olympic National Park from park establishment to the present. Source: NPS.

2.2. Climate

Olympic National Park's climate is moderate, with dry summers and wet winters. Most rainfall occurs between October and March, with less than 5% of annual rainfall typically occurring in July and August. Winter precipitation mostly falls as rain below 300 m (984 ft) elevation; as a mixture of rain and snow from 300 – 750 m (984 – 2460 ft) elevation; and as snow at higher elevations. The peninsula is characterized by a pronounced rainfall gradient, with extremely wet conditions on the south and west sides of the park and relatively dry conditions on the north and east side of the park.

Moist air coming from the Pacific Ocean is blocked by the Olympic Mountains, leading to heavy precipitation on the west side of the peninsula that increases with elevation, culminating with a striking 5 m (16.4 ft) or more of precipitation falling annually on Mt. Olympus. By contrast, the northeast part of the peninsula receives less than a tenth of that rainfall annually (Peterson et al. 1997, Halofsky et al. 2011b, Gavin and Brubaker 2015).

While the general climate description above provides an overview of annual and spatial variations in rainfall and temperature, systematic climate change in OLYM is also occurring, and requires ongoing attention. From alpine ecosystems to the coastlines, OLYM's natural resources have been and will be impacted by climate change. The effects of climate change will permeate all of the resource conditions we examine in this assessment. Here, we briefly describe projections for climate change in OLYM and expected responses of these current and ongoing changes.

Monahan and Fisichelli (2014) recently analyzed a century of weather data from OLYM and show that in the past 10-30 years, two temperature variables (minimum temperature of the coldest month and mean temperature of the wettest month) show warming deviations that are outside the historic range of variability measured from 1901-2012. These results demonstrate that climate in the park has shown a warming trend, particularly during winter.

Climate model predictions and expected future climatic conditions in the park broadly show trends of increasing temperature, with no significant trend in overall precipitation (Mote and Salathé 2010, Salathé et al. 2010). Projected increases in temperature could reach 1.6 to 5.4 °C (2.3 to 9.7 °F) by the 2080s in the Pacific Northwest (Mote and Salathé 2010). Warming is expected to occur during all seasons but be especially pronounced in the summer. Summer potential evapotranspiration (related to fuel moisture and tree stress) is expected to increase, with larger increases in low-elevation forests on the drier northeastern portion of the peninsula. While overall precipitation is not expected to change significantly, models suggest wetter winters and drier summers. On the Olympic Peninsula, winter precipitation may increase on average from 4.5 to 5 percent, depending on location. Precipitation intensity is projected to increase on the western side of the peninsula (Littell 2011).

Expected impacts of current and ongoing warming include decreases in snowpack (Mote 2003, Hamlet et al. 2005, Mote et al. 2005, Barnett et al. 2008), earlier spring snowmelt (Stewart et al. 2005, Hamlet et al. 2007), and more precipitation falling as rain rather than snow during the winter (Elsner et al. 2010). These changes in precipitation will have a direct effect on streamflow (timing and quantity), frequency and magnitude of flood events, and stream temperatures (Elsner et al. 2010). Changes in these stream characteristics will impact stream habitat for aquatic species such as fish (Mantua et al. 2011). Warming temperatures will also affect vegetation on the peninsula, and may include the following: increased invasion by exotic species; greater incidence of disease and pests; higher mortality and lower regeneration in forests; increased forest drought stress and lower productivity at low elevations; and altered ecosystem structure and function (Halofsky et al. 2011a). Finally, a changing climate in OLYM is expected to directly affect wildlife species that are adapted to specific habitat and climate characteristics (Halofsky et al. 2011c). Changing habitat conditions could cause physiological stress and increased mortality in many species, especially if they are unable to move to more favorable habitats due to restricted mobility, reduced connectivity, and loss of suitable habitat. We discuss specific climate-related stressors, challenges, and data gaps throughout the resource assessment (Chapter 4) and in the discussion (Chapter 5).

2.3. Natural Resources

In this section we describe the major natural resources of the park. We include descriptions of key physical components of the park, such as glaciers and freshwater systems. We then provide information on the biological components of the park, including ecological zones and vegetation, and an overview of the biological diversity in the park. We take special note of threatened, endemic, and extirpated taxa in the park, highlighting the unique, insular nature of the Olympic Peninsula and the resulting impacts on biological diversity. Finally, we provide an overview of the stressors affecting park resources.

2.3.1. Physical Components

Air

Because of its size and date of establishment, OLYM is classified as a Class I Area under the Clean Air Act, with the highest requirements for air quality. Overall, air quality in the park is considered moderate. The east side of the park can be affected by the high population density, industrial areas, and associated pollution found in the Puget Sound region. Furthermore, air quality in the park can be affected by air masses moving across the Pacific Ocean from Asia. In collaboration with the NPS Air Resources Division (ARD), OLYM is monitoring a diversity of air quality metrics as well as their impacts on park ecosystems. The condition assessment of OLYM's air resources is found in Chapter 4.5.1.

Geology

The Olympic Mountains are formed of basalts and sedimentary rocks (sandstones, shales, and conglomerates) that accumulated on the sea floor at the juncture of offshore oceanic and continental plates during the Eocene to mid-Miocene epochs (20 – 58 mya). The older Eocene basalts now comprise the Crescent Formation, a perimeter of basaltic rock along the northern, eastern, and southern Olympic Mountains. The horseshoe shape of the Crescent Formation likely formed as the oceanic plate pushed the basaltic mass into a corner of land masses between Vancouver Island and the continent (Gavin and Brubaker 2015). Continued subduction of the oceanic plate against the peripheral Crescent Formation resulted in the uplift, and folding of younger marine sediments that form the core of the Olympic Mountains today. The Olympic Mountains began their uplift during the Miocene epoch (7 – 26 mya) and may still be rising (Tabor 1987). The uplifted mountains began to intercept moisture from the Pacific Ocean, leading to rapid erosion and steep slopes. The mountains were further shaped by glacial events that occurred during the Pleistocene epoch (about 1.8 million to 12,000 years ago).

Glaciers

Olympic National Park is a land shaped by glaciers. During the Pleistocene epoch, up to six major glaciations impacted the Olympic Peninsula, with continental ice sheets moving southward from western Canada and then retreating. These glaciers carved out and filled the Strait of Juan de Fuca and Puget Sound to 914 m (3,000 ft) elevation, isolating the Olympic Peninsula from other landmasses. Valley glaciers originating within the Olympic Mountains carved deep valleys that extended from the interior of the park. These valleys are broadest on the southwest side of the peninsula, and are narrower on the south and east sides of the park. In modern times, glaciers have retreated to the high-elevation basins of the park.

Current glacier status was recently assessed by Riedel et al. (2015). After comparing current conditions to conditions in 1980, authors found the 266 glaciers present in the park in 1980 have been reduced to 184 at present, with a total cover of 30.20 ± 0.95 km^2 (11.66 ± 0.37 mi^2). This loss of 82 glaciers represents a 34% decline in aerial extent over the time period totaling 15.7 km^2 (6.06 mi^2). Of the 184 remaining glaciers, 120 are less than 0.1 km^2 (0.04 mi^2), and only four are >1 km^2 (0.39 mi^2): Blue, Hoh, White, and Humes. In 1980, there were eight glaciers >1 km^2. The remaining large glaciers are all found on Mt. Olympus. The rate of glacier loss over the 1980-2009 time period was 540 m/yr (1772 ft/yr) across the peninsula, representing an acceleration over the 260 m/yr (853 ft/yr) loss measured from 1900-1980. In general, east- and south-facing glaciers have lost more area, so aspect and distance inland from Pacific moisture have a large influence on glacial response. Glaciers that are higher in elevation and on the wet, windward side of the Olympic Mountains have experienced less loss.

Overall, there is a higher rate of loss of glacial extent in the Olympic Mountains compared to further inland (British Columbia and the North Cascades), with a comparable loss to Vancouver Island. This indicates a greater sensitivity to changes in temperature in the coastal ranges compared to inland, which influence spring ablation and early fall accumulation. The loss in glaciers over the past 30 years appears to be a result of mean temperature increases, and illustrates how sensitive these relatively small, thin, and low-elevation glaciers are to climate change. Riedel et al. (2015) expect this trend to continue, with further reductions in the size and numbers of glaciers in OLYM following future increases in mean temperature.

Water Resources

Olympic National Park contains the headwaters of 11 major rivers that radiate from the core of the Olympic Mountains (Figure 2.3.1; Table 2.3.1). These rivers include the Skokomish, Duckabush, Dosewallips, Dungeness, Morse, Elwha, Sol Duc/Quillayute, Bogachiel, Hoh, Queets, and Quinault. Portions of two other watersheds, the Hamma Hamma and Calawah, also lie within the park, and there are parts of five other watersheds that flow through the park’s coastal strip. These dynamic rivers are a quintessential component of OLYM, providing habitat for numerous fish species and other aquatic taxa. Glaciers play an important role in streamflow (especially late-summer streamflows) on certain rivers, and the dynamics of these glacier-fed rivers will continue to change with the accelerated melting of glaciers.

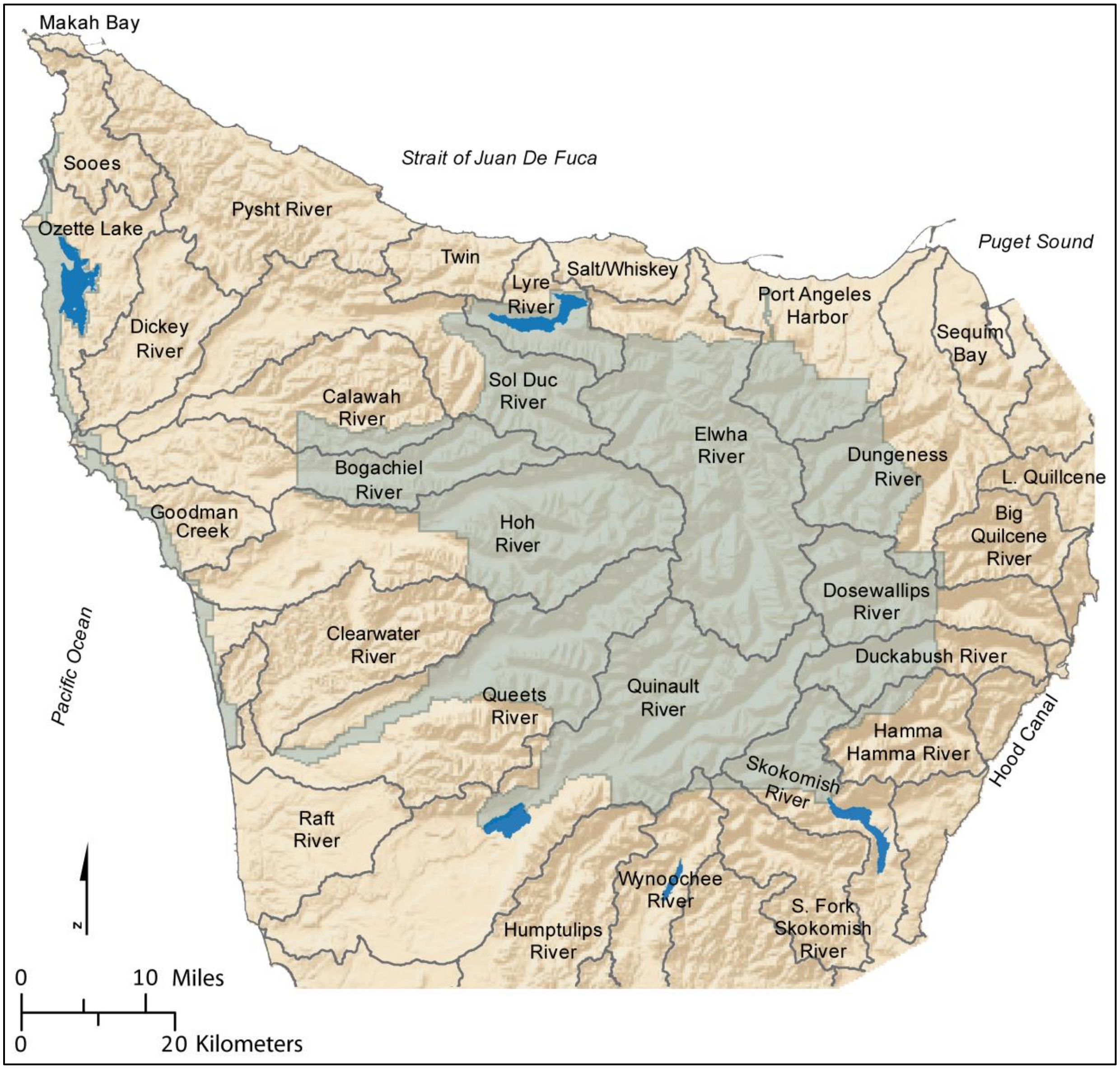

Figure 2.3.1. Major watersheds of Olympic National Park. Source: Watershed Boundary Dataset (WBD), HUC 10 (coordinated effort between the U.S. Department of Agriculture-Natural Resources Conservation Service, U.S. Geological Survey, and the U.S. Environmental Protection Agency (USEPA). The WBD was created from a variety of sources from each state and aggregated into a standard national layer. Source: http://datagateway.nrcs.usda.gov, accessed 25 October 2017).

Table 2.3.1. Area of all watersheds with >10 km^2 area in Olympic National Park. Source: Watershed Boundary Dataset (WBD), http://datagateway.nrcs.usda.gov, accessed 25 October 2017).

Watershed Names	Area Within OLYM (km^2)	Area Outside OLYM (km^2)	Total Area (km^2)
Bogachiel River	212.09	183.42	395.51
Calawah River	66.73	284.94	351.67
Dosewallips River	190.48	107.97	298.45
Duckabush River	116.15	81.66	197.81
Dungeness River	192.21	321.48	513.69
Elwha River	728.21	104.05	832.25
Goodman Creek	44.78	173.41	218.18
Hoh River	445.63	325.34	770.97
Hamma Hamma River	18.32	199.01	217.34
Lyre River	122.32	51.45	173.76
Morse Creek - Port Angeles Harbor	144.76	271.34	416.10
Ozette Lake	97.57	206.73	304.30
Queets River	388.92	380.61	769.54
Quinault River	567.06	556.42	1,123.48
Skokomish River	127.52	238.60	366.12
Sol Duc River-Quillayute River	194.03	409.42	603.45

In addition to major rivers, OLYM has two large low-elevation lakes (Crescent and Ozette), 646 mountain lakes, and about 6,438 km (4,000 mi) of smaller rivers and streams. Lake Crescent is an inland lake that contains endemic fish due to its isolation. Lake Ozette is coastal, with an outlet to the ocean and anadromous fish use. It also contains an important endemic species of fish.

2.3.2. Biological Components

The resource descriptions that follow provide a brief overview of the biological diversity and breadth of the park's natural resources, while highlighting areas that will be considered in greater detail later in the resource assessment.

Ecological Zones

Olympic National Park and the peninsula are characterized by eight potential vegetation zones (i.e., presumed climax vegetation) described in Henderson et al. (2011): Sitka spruce (*Picea sitchensis*), Douglas fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), Pacific silver fir (*Abies amabilis*), mountain hemlock (*Tsuga mertensiana*), subalpine fir (*Abies lasiocarpa*), parkland, and alpine (Figure 2.3.2; Table 2.3.2). The distribution and extent of these zones are derived from gradient-analysis-based models of boundaries between presumed climax vegetation distributions.

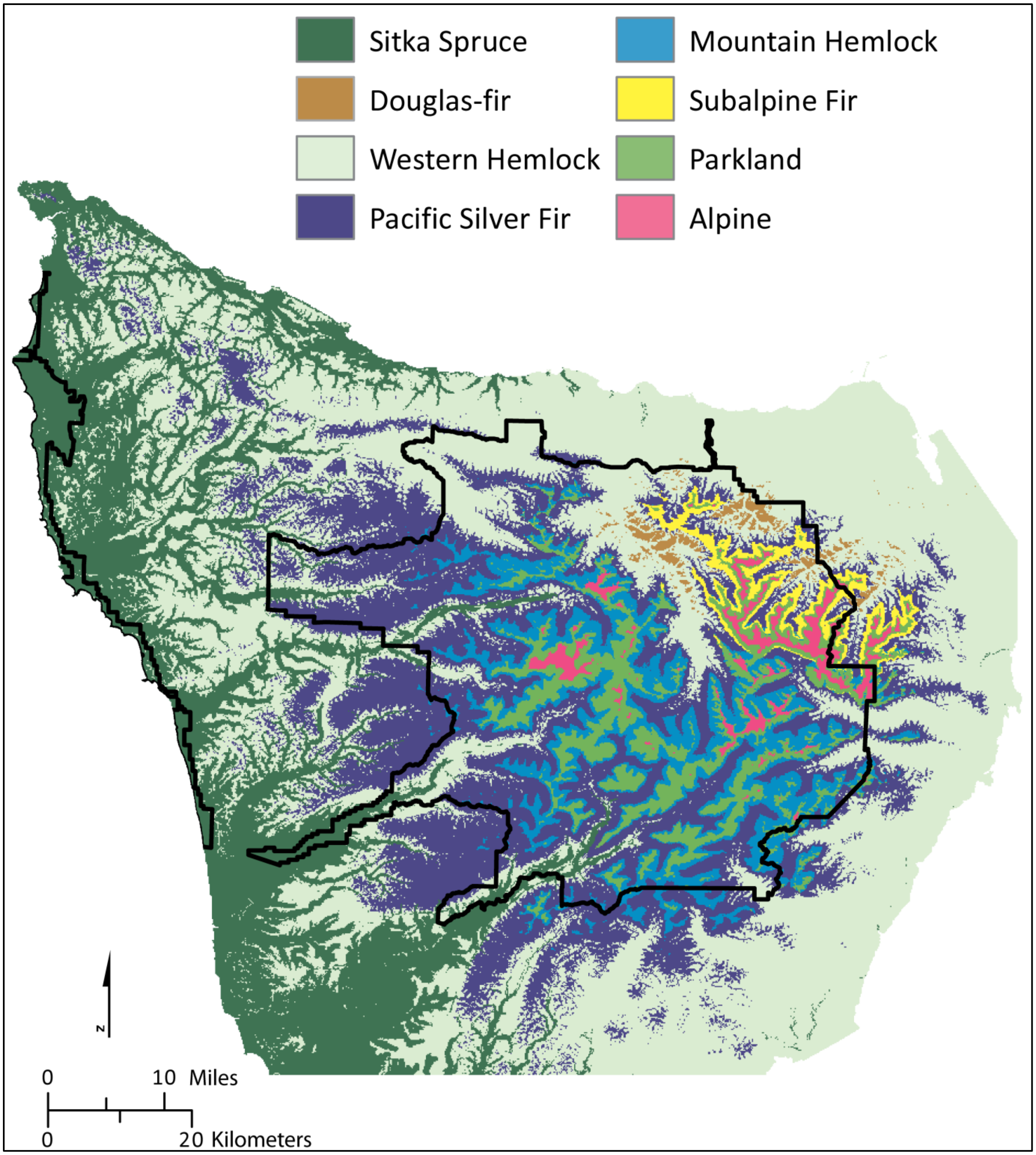

Figure 2.3.2. Eight classified potential vegetation zones found on the Olympic Peninsula. Source: Henderson et al. 2011.

Table 2.3.2. Area of each potential climax vegetation zone (Henderson et al. 2011) in Olympic National Park with elevation, precipitation, temperature, and distribution characteristics for each zone (modified from Gavin and Brubaker 2015).

Vegetation Zone	Area (ha)	Approximate Elevation Range (m)	Mean Annual Precipitation (mm)	Mean January/July Temperature (ºC)	Distribution
Sitka spruce	29,911	0 – 250	2,700	4.3/15.6	Low-elevation coastal
Douglas fir	5,707	500 – 1,200	1,900	0.5/15.0	Narrow distribution in the driest portions of the western hemlock zone on south-facing slopes
Western hemlock	77,288	0 – 1,200	2,200	3.7/6.2	Broad distribution in drier NE, narrow band in west
Pacific silver fir	121,050	300 – 1,400	3,500	2.0/15.1	Mid-elevation, above western hemlock and below mountain hemlock
Mountain hemlock	63,556	800 – 1,500	3,500	-0.2/13.1	Highest forested zone in the west, with some patches in the drier east
Subalpine fir	10,623	1,300 – 1,800	2,300	-2.6/12.2	Highest forested zone in the rain shadow climate of the northeast
Parkland	49,845	1,100 – 1,900	2,300	-1.5/12.1	Occurs throughout high elevations on the peninsula
Alpine	10,987	1,700 – 2,000	–	-3.5/NA	Highest portions of the peninsula

The boundaries of OLYM are clearly identifiable from space, offering the best evidence that OLYM is an ecological island of structurally complex forested ecosystems within a sea of managed forests outside the park. This distinction can easily be seen using the Landscape Ecology, Modeling, Mapping and Analysis (LEMMA) project of the U. S. Forest Service (USFS) and Oregon State University (Ohmann and Gregory 2002; Figure 2.3.3). The LEMMA project defines forest structural conditions within 30-m^2 pixels, which are shown here for OLYM and the peninsula (Figure 2.3.3; Table 2.3.3), using direct gradient analysis and nearest neighbor imputation (GNN).

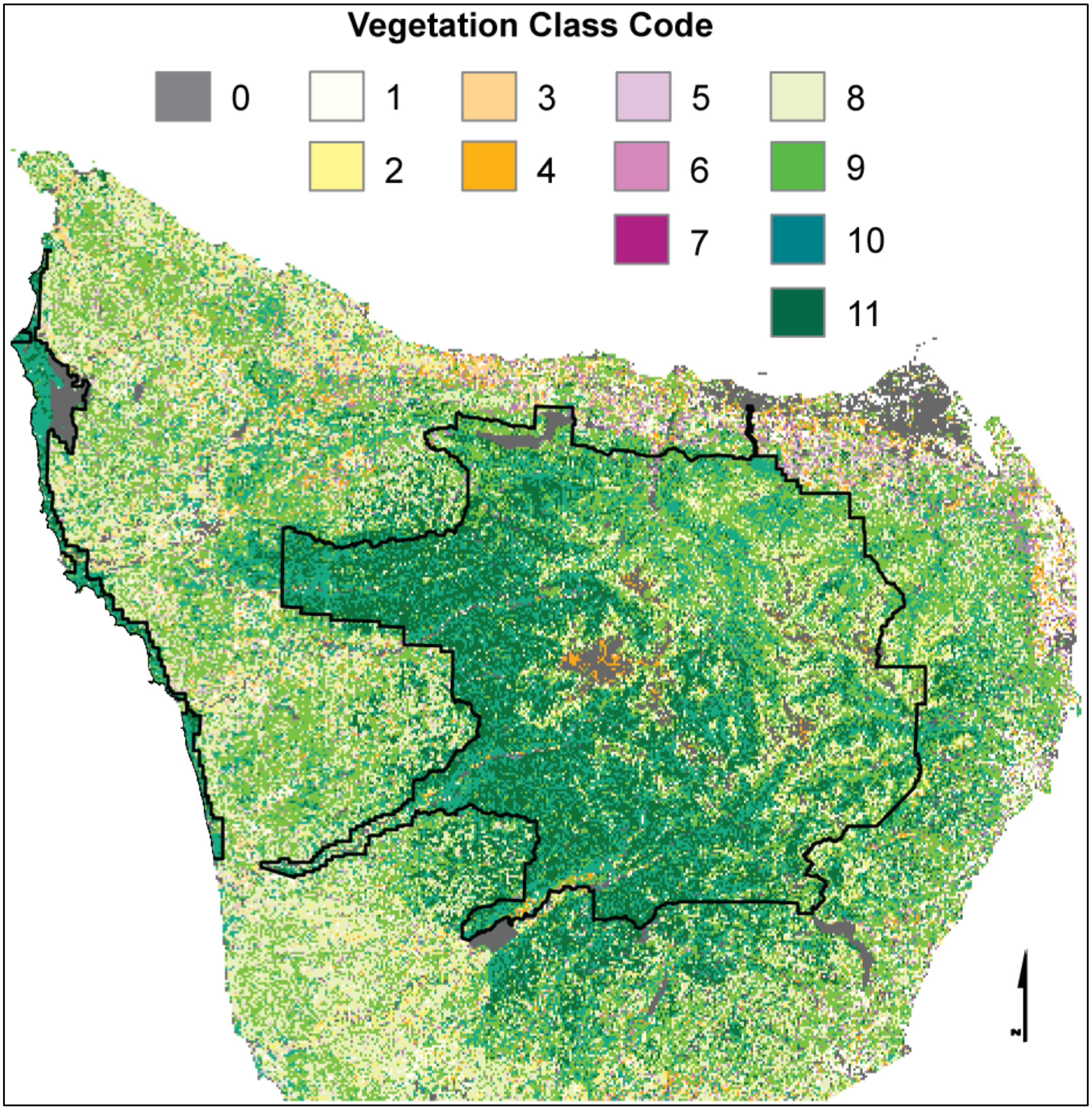

Figure 2.3.3. Gradient nearest neighbor (GNN) vegetation classes found on the Olympic Peninsula. See Table 2.3.3 for definition of vegetation classes. Source: Ohmann and Gregory 2002, with data from the Northwest Forest Plan Effectiveness Monitoring GNN dataset (2012).

Table 2.3.3. Description of the 11 GNN vegetation classes found on the Olympic Peninsula (Figure 2.3.2), with a summary of the area of each of these classes within Olympic National Park (GNN dataset 2012).

Forest Structural Condition Class	Area (ha)	Canopy Cover of All Live Trees (%)	Proportion of Live Tree Basal Area Represented by Hardwoods	Quadratic Mean Diameter of Dominant Trees (cm)
0 – Ice, water agricultural, or exurban	31,682	–	–	–
1 – Sparse	4,185	<10	N/A	N/A
2 – Open	26,229	10-39	N/A	N/A
3 – Broadleaf, sapling/pole, moderate/closed canopy	256	≥40	≥0.65	<25
4 – Broadleaf, small/medium/large, moderate/closed canopy	1,197	≥40	≥0.65	>25
5 – Mixed, sapling/pole, moderate/closed canopy	316	≥40	0.20 – 0.64	<25
6 – Mixed, small/medium, moderate/closed	1,487	≥40	0.20 – 0.64	25 – 50
7 – Mixed, large/giant, moderate/closed canopy	156	≥40	0.20 – 0.64	>50
8 – Conifer, sapling/pole, moderate/closed	33,143	≥40	<0.20	<25
9 – Conifer, small/medium, moderate/closed canopy	69,500	≥40	<0.20	25 – 50
10 – Conifer, large, moderate/closed	100,343	≥40	<0.20	50 – 75
11 – Conifer, giant, moderate/closed canopy	100,926	≥40	<0.20	>75

Natural Disturbance and Landscape Change

While OLYM's major ecosystems can be characterized by its dominant and potential vegetation structure, the landscape certainly has not remained static over time. The composition, structure and function of forest ecosystems are shaped by disturbance events (Dale et al. 2001) that range in scale from extensive mortality over large areas (e.g., fire) to small patches (e.g., local root rot pockets), or the widespread decline of individual species (e.g., insect infestation). Disturbance events in the Pacific Northwest include fires, wind storms, ice storms, avalanches, drought, landslides, floods, insect and pathogen outbreaks, and introduction of exotic species (Spies and Franklin 1989). We present the distribution and nature of natural disturbance and landscape change throughout the Olympic Peninsula using two datasets: (1) Aerial Detection Survey (ADS) data collected by the U.S. Department of Agriculture (USDA) Forest Service and the Washington DNR, and (2) models and products of the NPS North Coast and Cascades Network (NCCN) landscape change monitoring program.

Long-term patterns of natural disturbance on the peninsula are shown using ADS data that have been collected annually since 1949 (Figure 2.3.4). These data describe the location of forest insect-caused damage, disease symptoms, weather-related damage, and other forest health stressors (Johnson and Wittwer 2008). Using fixed-wing aircraft flying relatively low and slow, observers evaluate a swath of land out of each side of the airplane and sketch the location of disturbances on geo-referenced topographic maps. Identification of disturbance agents is based on seeing pest-specific damage “signatures” consisting of foliage color, canopy texture, tree species identity, and season (McConnell et al. 2000). Creating disturbance maps using this sketch-mapping method is highly subjective and therefore variable among observers (Klein et al. 1983), and is not effective at detecting root disease, dwarf mistletoe, understory mortality, or minor defoliation. Moreover, methods have changed over the 70 years of data collection as electronic tools have become available such that later maps are more accurate and have finer-scale polygons than older maps (see Figure 2.3.4). Consequently, the data are best used for demonstrating trends rather than precisely identifying affected areas (Johnson and Wittwer 2008).

A summary of the area affected by various disturbance agents (Table 2.3.4) shows that beetles, primarily mountain pine beetle (*Dendroctonus ponderosae*) and silver fir beetle (*Pseudohylesinus sericeus*) is the most prevalent disturbance agent in terms of affected area in OLYM. The decline in mountain pine beetle through the time series, and its near absence since 1993, may reflect the decline in western white pine abundance due to logging, fire suppression, and white pine blister rust (*Cronartium ribicola*) since the mid-1900s (Graham 1990).The douglas-fir beetle (*Dendroctonus pseudotsugae*) is an additional minor component of the beetle-caused mortality of trees, but is so small that it was not mapped. Dying hemlock complex is the second-most prevalent disturbance agent in the park. Dying hemlock complex refers to the interaction of a number of disease and environmental conditions that result in hemlock mortality, where an individual agent cannot be identified.

Disturbance recorded as bear damage is extensive throughout the peninsula, but is rare within the park (Table 2.3.4; hence, bear damage is also not mapped in Figure 2.3.4). Bear damage occurs primarily in second-growth forest following logging because bears feed on the cambium of young trees, especially in early spring. Damage to young trees outside of the park tends to be coded as bear damage but causes such as root disease or other herbivores are also possible and have been verified in ground checks.

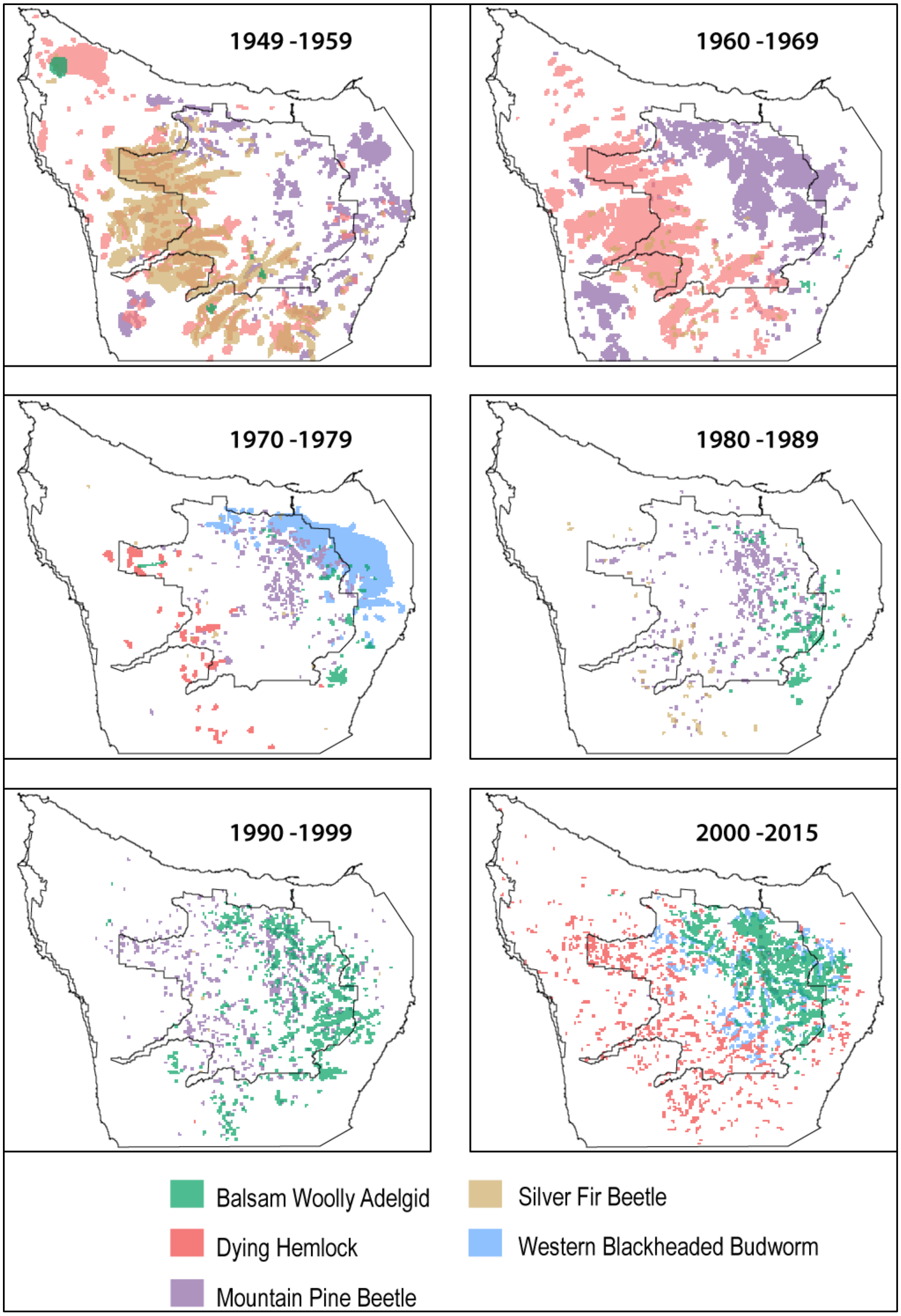

Figure 2.3.4. Spatial distribution of five disturbance agents on the Olympic Peninsula summarized over six time intervals from 1949-2015. Source: USDA Forest Service Pacific Northwest Region Aerial Detection Surveys.

Table 2.3.4. Area affected by various disturbance agents on the Olympic Peninsula, 1949-2015. Source: USDA Forest Service Pacific Northwest Region Aerial Detection Surveys.

Disturbance Agent	Affected Area (ha)	Affected Area Peninsula (%)	Affected Area OLYM (%)
Beetles	281,504	23.2	55.7
Other insects	103,863	8.8	19.8
Dying hemlock complex	276,786	22.8	22.2
Bear	521,495	43.0	1.2
Physical	5,384	0.4	0.8
Other pests and diseases	22,031	1.7	1.7

The 66-year ADS record provides perspective on the spatial and temporal dynamics of disturbance events and agents. Large catastrophic events such as fire and windthrow, which have occurred historically (Henderson et al. 1989), were not significant during this period. The ADS does not effectively map small windthrow events because individual or small groups of fallen trees are difficult to map from the air. The dominant beetle species were prominent early in the record, primarily in the south and west (silver fir beetle) or north and east (mountain pine beetle) but are both mostly absent after 2000 (Figure 2.3.4). This contrasts with the observation of increasing mountain pine beetle in recent decades in other areas of the western United States (Logan et al. 2003, Carroll et al. 2004). Western black-headed budworm (*Acleris gloverana*), the most widespread “other insect”, and dying hemlock complex appear to wax and wane. Balsam woolly adelgid is an introduced species that originated in the southeastern Olympic Peninsula, then spread northward first along the eastern mountains and later to more western parts of OLYM where BWA damage may now be subsiding. The distribution of this non-native species in relation to subalpine fir communities is assessed in Chapter 4.4.2.

We also examined landscape changes from 1985 to 2010 throughout the Olympic Peninsula as determined by the NCCN landscape change monitoring program (Figure 2.3.5, Copass et al. 2016). This program used the Landsat-based Detection of Trends in Disturbance and Recovery (LandTrendr) algorithm to detect rapid (occurring over <4 years) landscape changes resulting from avalanche, forest clearing, human developments, fire, mass movements, progressive defoliation (i.e., defoliation resulting from forest insects or disease), riparian flooding, or tree toppling (i.e., windthrow events, Table 2.3.5). The predominant disturbance outside the park was clearing while both fire and tree topping were important inside the park. Landscape changes both within and outside the park will affect OLYM’s natural resources in different ways. We address the effects of various landscape changes on key resources throughout the condition assessment.

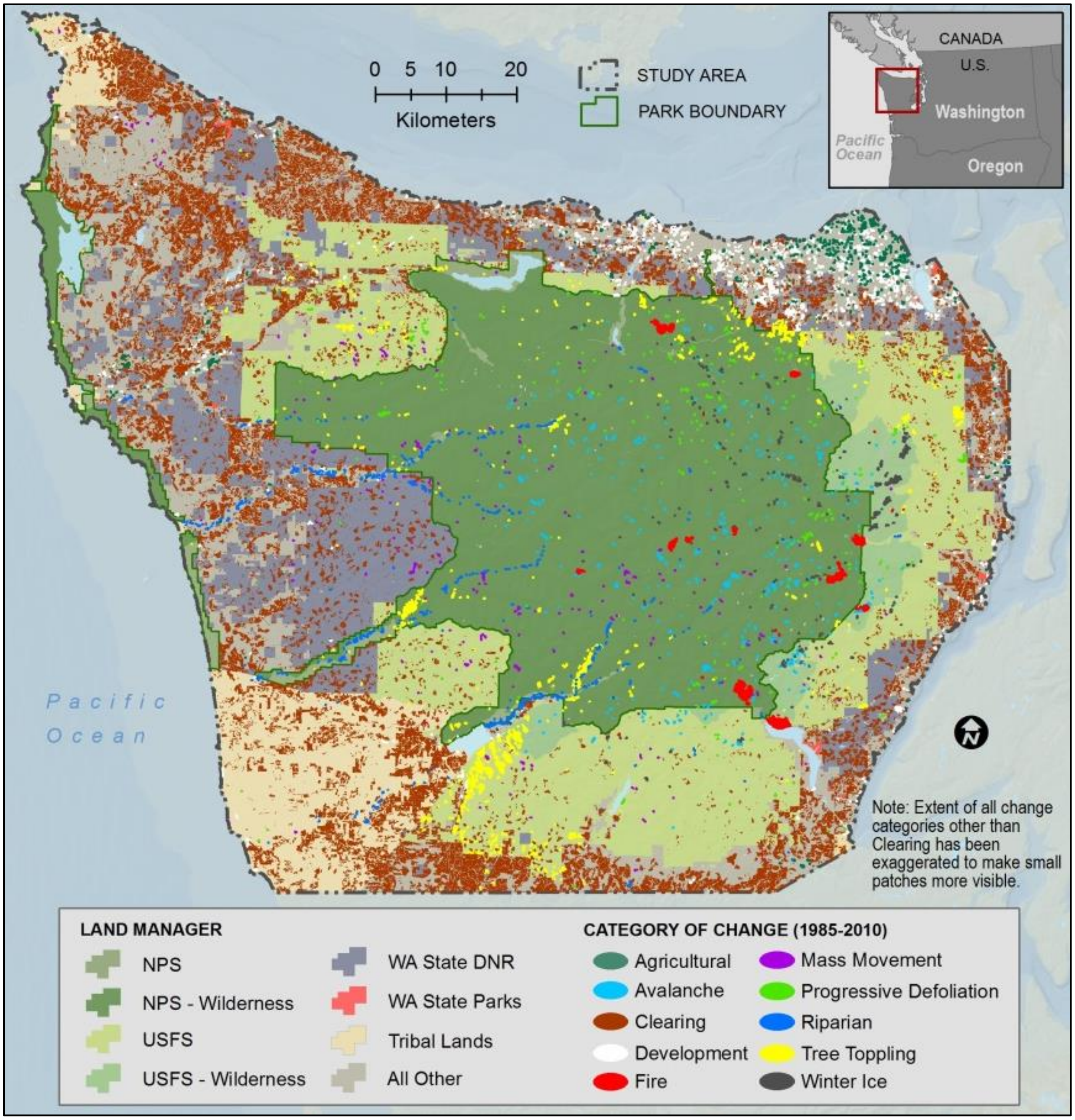

Figure 2.3.5. Landscape change in Olympic National Park from 1985-2010. Source: NPS.

Table 2.3.5. Area affected by various disturbance agents on the Olympic Peninsula, 1985-2010. Source: Copass et al. 2016.

Disturbance Agent	**Affected Area (ha)**	**Area Peninsula (%)**	**Area OLYM (%)**
Avalanche	714	0.06	0.16
Clearing	150,327	13.12	0.00
Development	2,463	0.21	0.00
Fire	1,371	0.12	0.28
Mass movement	341	0.03	0.04
Progressive defoliation	381	0.03	0.06
Riparian	878	0.08	0.12
Tree topping	2,531	0.22	0.26
Winter ice	622	0.05	0.05
Total	**159,630**	**13.93**	**0.97**

Terrestrial and Aquatic Fauna

Olympic National Park contains a unique and diverse biota, which has been shaped by the geologic and glacial history of the Olympic Peninsula. Within the park, OLYM has documented 65 mammal species, 258 bird species, 92 fish species, 13 amphibian species, and five reptile species (Appendix A).

The mammals of OLYM are diverse, and include notable widespread, endemic, extirpated, reintroduced, and invasive species. One of the most iconic and wide-ranging mammal species in OLYM – and indeed, one of the reasons OLYM was formed – is the Roosevelt elk. Roosevelt elk populations have been monitored periodically, with more quantitative analyses of population size being conducted in recent decades. Elk population dynamics are assessed in Chapter 4.1.1. The Pacific fisher (*Pekania pennanti*) was reintroduced in OLYM in 2008, and monitoring of the species is ongoing as populations begin to establish and spread over the peninsula. Surveys are currently underway for the Pacific marten (*Martes caurina*), another rare mesocarnivore in the peninsula forests.

One notable non-native mammal species is the mountain goat (*Oreamnos americanus*), which threatens fragile subalpine plant communities (Houston et al. 1994). Twelve mountain goats were introduced to the Olympic Mountains during the 1920s prior to the park's establishment, and subsequently increased in numbers and expanded throughout the Olympic Range. In 1983, OLYM conducted the first aerial mountain goat survey in the Olympic Mountains, returning an estimate of over 1,000 mountain goats (Houston et al. 1986). Concerns over the environmental impacts caused by high numbers of mountain goats led to management efforts to reduce numbers through live capture, removal to other understocked parts of the species native range, and legal hunting outside the park (Houston et al. 1994). Mountain goat populations were reduced to less than 400 animals by 1990 and remained relatively stable, ranging between about 300-400 during the 1990s and early 2000s. The most recent population survey in 2016, however, indicated that the population had grown at an average rate of 6% annually since 2004 to an estimated population of 623±43 (standard error of the estimate; Jenkins et al. 2016), fueling renewed interest in managing the population. The NPS has prepared a draft Environmental Impact Statement that explores management issues associated with non-native mountain goats on the Olympic Peninsula and alternative management options (https://parkplanning.nps.gov/projectHome.cfm?projectId=49246; accessed 19 September 2017). Consequently, issues associated with non-native mountain goat population growth and management are not considered further in this assessment.

The landbird community in Olympic National Park has been monitored since 2005 to detect trends in avian populations over time (Ray et al. 2017). Population densities pooled across three parks in the region remained stable or increased for most of the 39 species studied over a nine-year period. Certain species of particular interest have been more extensively studied, such as the northern spotted owl and the marbled murrelet (*Brachyramphus marmoratus*), both of which are listed as threatened species under the federal Endangered Species Act (USFWS 1990, USFWS 1992). Ongoing monitoring of the northern spotted owl in relation to increasing populations of the invasive barred owl is discussed in detail in Chapter 4.4.1.

Olympic National Park has 31 native freshwater fish species, and 70 unique salmonid populations, or stocks. The seven anadromous fish species (i.e., whose life history includes both freshwater and marine stages) are bull trout (*Salvelinus confluentus*), Chinook salmon, chum salmon (*Onchorhynchus keta*), Coho salmon, pink salmon (*O. gorbuscha*), sockeye salmon (*O. nerka*), and steelhead (*O. mykiss*). Olympic National Park's salmonids play a critical role in the ecosystem function of the peninsula's rivers, and represent key cultural and economic resources for the communities of the Olympic Peninsula. There are 7 nonnative fish species in OLYM. Status and trends of key fish resources are considered in Chapter 4.2.1.

Amphibian and reptile distributions have been documented throughout the park. Amphibians are a group of interest due to ongoing global declines. We consider amphibians in greater detail in Chapter 4.3.3.

Despite their key roles in ecosystem processes, we know relatively little about the invertebrate species inhabiting OLYM, including basic species lists. In addition to terrestrial and freshwater invertebrate diversity, OLYM hosts an incredible marine intertidal invertebrate diversity on the coast. In this report, we assess the status of a harvested razor clam population in Chapter 4.2.2, which has experienced declines. We also examine the status of the sea star community in Chapter 4.3.2, which has been several impacted by disease in recent years.

Plants

There are over 1,400 vascular plant taxa present in OLYM, with a high concentration of rare and endemic taxa (Buckingham et al. 1995). In fact, the Olympic Peninsula contains 28% of the plants identified as "rare" by the Washington Natural Heritage program while occupying only 9% of the total area of the state. This represents the highest concentration of rare plants in the state. The wide variety of habitats, the huge precipitation gradient, and the elevational diversity of the Olympic Peninsula have all contributed to the plant diversity found on the peninsula and in the park.

Non-vascular plants (mosses, liverworts, hornworts, and lichens) make up a significant portion of the biomass and annual production in temperate rainforests (Vitt et al. 1988) and are known to play important ecological roles in hydrology (Boucher and Nash 1990, Norris 1990) and nutrient cycling (Nadkarni 1985) in these ecosystems. Because of its tremendous habitat complexity, relatively pristine condition, and moderate maritime climate, OLYM has an astonishing diversity of non-vascular plants across the entire elevation range. More than 1,280 non-vascular species have been catalogued from the Olympic Peninsula (Hutten et al. 2005), and species new to the area are still being discovered. More than 220 bryophytes and lichens found on the Olympic Peninsula are listed as rare in the Washington and Oregon Heritage Plans (Oregon Natural Heritage Information Center 2004) and in former Northwest Forest Plan Survey and Manage documents. Several species that may be rare in other places appear to be more common on the Olympic Peninsula, making it a potential refuge for threatened species.

Across the whole peninsula, there are 333 known exotic species, which represent about 25% of the flora (Buckingham et al. 1995). There are 313 known exotic species in OLYM. Most infestations of exotic species are confined to the front-country, but at least 70 species occur in park wilderness. These exotics represent a threat to imperiled plant species and to ecosystem function and integrity. We consider the role of invasive plant species in greater depth in Chapter 4.4.3.

Endemism and Disjunction

The unique plant and animal taxa seen only on the Olympic Peninsula, which are called endemic species, are a defining feature of the park’s biological diversity (Gavin and Brubaker 2015, Gavin and University of Oregon 2015). The ice sheets that advanced and retreated throughout the Pleistocene effectively isolated the Olympic Peninsula for several millennia, creating ice-age refugia on the Olympic Mountains during that period (Gavin and Brubaker 2015). Twenty-nine plant and animal taxa are known to be endemic to the Olympic Peninsula (Table 2.3.6), with 13 species restricted to high-elevation habitats in the north or northeast parts of the park and other species restricted to streams and low-elevation shaded sites on the southwestern part of the peninsula. There are likely many more endemic species and lineages on the peninsula, for example in the poorly studied arthropods. Additionally, many species that are not fully endemic to the peninsula show distinct genetic lineages, such as certain amphibian species. One endemic species of considerable interest in OLYM is the Olympic marmot, which is assessed in Chapter 4.3.1.

Table 2.3.6. Known endemic species or subspecies on the Olympic Peninsula (adapted from Gavin and Brubaker 2015).

Taxonomic Category	Scientific Name	Common Name	Distribution
Amphibians	*Rhyacotriton olympicus*	Olympic torrent salamander	Steep gradient streams, lowlands to montane
Mammals	*Marmota olympus*	Olympic marmot	Subalpine, open meadows (throughout)
	Tamias amoenus caurinus	Olympic yellow-pine chipmunk	Subalpine, forest, and parkland (north and northeast)
	Scapanus orarius (Olympic clade)	Coast mole	Subalpine (northeast and southwest)
	Thomomys mazama melanops	Olympic Mazama pocket gopher	Meadows and young forest (north and northeast)
	Mustela ermine olympica	Olympic ermine	Throughout
Fish	*Novumbra hubbsi*	Olympic mud minnow	Low gradient rivers, muddy sediment (south)
Opiliones (harvestmen)	*Acuclavella makah*	–	Perennial headwater stream banks, woody debris
Orthoptera (grasshoppers)	*Niquallia olympica*	Olympic grasshopper	Scree sites in subalpine to alpine
Lepidoptera (butterflies and moths)	*Hesperia comma hurlbirti*	Hulbirt's skipper	Subalpine to alpine
	Oeneis chryxus valerata	Olympic arctic	Subalpine to alpine (northeast)
Coleoptera (beetles)	*Bryelmis rivularis*	Riffle beetle	Streams 3 – 6 m wide, woody debris (west and NW Oregon)
	Cicindela bellissima frechini	Pacific coast tiger beetle	Sand dunes and deflation plains (west)
	Nebria acuta quileute	Quileute gazelle beetle	River banks at mid-elevation (north)
	Nebria danmanni	Mann's gazelle beetle	Montane to subalpine (northeast)
Diplopoda (millipedes)	*Leonardesmus injucundus*	–	Litter and soil in dense forest (south)
	Mircolympia echina	–	Alder litter in riparian forest (west)
	Tubaphe levii	Olympic Peninsula millipede	Lowlands (west)

Table 2.3.6 (continued). Known endemic species or subspecies on the Olympic Peninsula (adapted from Gavin and Brubaker 2015).

Taxonomic Category	Scientific Name	Common Name	Distribution
Mollusks	*Hemphillia burringtoni*	Arionid jumping slug	Dense riparian forest (possibly also in the Cascade range & Vancouver island)
Plants	*Astragalus cottonii*	Olympic Mountain milkvetch	Open subalpine sites (northeastern mountains)
	Campanula piperi	Piper’s bellflower	Rocky montane to alpine sites (northeastern to central)
	Corallorhiza maculata var.*ozettensis*	Spotted coral root	Lowland partial shade sites (northwest)
	Erigeron flettii	Flett’s fleabane	Open subalpine to alpine sites (northeast to south)
	Erigeron peregrinus ssp. *peregrinus* var. *thompsonii*	Thompson’s wandering fleabane	Lowland bog sites (southwest)
	Erythronium quinaultensis	Quinault fawn lily	Open or partially open lowland sites (southwest)
	Petrophytum hendersonii	Olympic rockmat	Rocky montane to alpine sites (north, east, and south)
	Senecio neowebsteri	Olympic Mountain groundsel	Scree subalpine to alpine sites (north, northeast and central)
	Synthyris pinnatifida var. *lanuginosa*	Olympic Mountain synthyris	Scree subalpine to alpine sites (north, northeast and central)
	Viola flettii	Flett’s violet	Rocky subalpine to alpine sites (north, northeast, and central)

At least 13 species common in the Cascade Mountains and beyond were not found historically on the peninsula, providing additional evidence of a long-term barrier to dispersal between the peninsula and surrounding areas. Notable species that have not colonized the peninsula include grizzly bears (*Ursus arctos*), wolverine (*Gulo gulo*), lynx (*Lynx canadensis*), water vole (*Microtus richardsoni*), golden-mantled ground squirrel (*Callospermophilus lateralis*), northern bog lemming (*Synaptomys borealis*), pika (*Ochotona princeps*), and mountain sheep (*Ovis canadensis*; Gavin and Brubaker 2015). Red foxes (*Vulpes vulpes*) and mountain goats were introduced by humans in the 20th century, and coyotes (*Canis latrans*), porcupines (*Erethizon dorsatum*), and opossums (*Didelphis virginiana*) have also now moved into the area. Absent tree species include noble fir (*Abies procera*), ponderosa pine (*Pinus ponderosa*), subalpine larch (*Larix lyallii*), western larch (*Larix occidentalis*), and western juniper (*Juniperus occidentalis*; Houston et al. 1994).

Finally, many taxa found on the peninsula and in the park are geographically disjunct from the remainder of their populations, or their closest sister taxa, often by hundreds of kilometers. This also suggests the existence of a Pleistocene refugium. For example, there are taxa found in OLYM that are most closely associated with species in the Yukon and Alaska, or in the interior Rocky Mountains, yet which are absent from the Cascade Mountains.

Threatened, Endangered, and Sensitive Species

Olympic National Park contains a number of animal taxa that are listed by federal agencies or by the State of Washington as threatened, endangered, or sensitive (Appendix B). Federally threatened species and salmonid stocks include the following: northern spotted owl, marbled murrelet, bull trout, Puget Sound steelhead (*O. mykiss*), Lake Ozette sockeye (*O. nerka*), Puget Sound chinook (*O. tshawytscha*), and Hood Canal summer chum (*O. keta*). The gray wolf, which is currently not found on the Olympic Peninsula, is listed as endangered at both the federal and state level. The Pacific fisher, a distinct population segment that ranges from British Columbia to California, is a state endangered species in Washington, but was recently denied federal listing as a threatened species. In total, 20 species and three salmonid stocks are listed at either the state or federal level.

Although not listed at the state or federal level, Pacific martens are another species that appear to have declined on the peninsula over the past century (Zielinski et al. 2001). The scarcity of martens found in recent surveys has prompted a shift in the NatureServe designation (NatureServe 2015), where the Pacific marten populations in Washington and Oregon (i.e., those found on the Olympic Peninsula) are now listed separately from the interior populations, and are designated as “G4G5T1”, which means that they are critically imperiled at the population level. In contrast, the inland population is listed as secure.

There are no federally listed plant taxa, but the whitebark pine (*Pinus albicaulis*) is a candidate for listing due to ongoing declines associated primarily with white pine blister rust and bark beetle infestations (USFWS 2011a). The U.S. Fish and Wildlife Service (USFWS) published a 90-day review of a petition to list Alaska yellow cedar (*Callitropsis [Chamaecyparis] nootkatensis*) as threatened and found that the petition presented substantial scientific information indicating that listing may be warranted. A more detailed (“12-month”) review by USFWS is underway; this review will determine whether listing is warranted. Of the endemic plant taxa, Quinault fawn lily and Thompson’s wandering fleabane are categorized as state sensitive species, and Olympic mountain milkvetch is listed as a state threatened species and a federal species of concern.

Extirpated Species

The gray wolf was a key top predator in the Olympic Peninsula’s ecosystems until it succumbed to the pressure of expanding human settlement and persecution in the 1920s. It is presumed that as the most significant top predator in OLYM, the wolf historically played a key role in structuring ecological communities throughout the park. The wolf is hypothesized to have controlled Roosevelt elk populations, which could have led to trophic cascades affecting riparian plant communities and physical attributes of streams and rivers (Beschta and Ripple 2008). Furthermore, the extirpation of the wolf is hypothesized to have led to increases in coyote populations, which were previously absent from OLYM (Witczuk et al. 2013). We explore the potential ecological consequences of the loss of this top predator in Chapters 4.1 – 4.3.

2.3.3. Resource Issues Overview

Olympic National Park ecosystems are exposed to a variety of stressors, reflecting changes in various global, regional and local forces that drive ecosystem processes (Figure 2.3.6). Ecosystem stressors may be defined as foreign or uncharacteristically variable physical, chemical, or biological perturbations that have the capacity to alter natural ecosystem components, patterns or processes (Barrett et al. 1976). Identifying and mitigating to the extent possible the effects of stressors that degrade natural ecosystem processes or specific resources within the park is a high priority of the NPS.

Ecological stressors and resource issues received considerable scrutiny recently during the development of a long-term ecological monitoring program for OLYM (Jenkins et al. 2003) and similarly, within parks of the NCCN (NCCN; Weber et al. 2009). Changes in climate, pollution, land uses, exotic species, and direct human activities are all key forces that have the potential to impact park resources via numerous ecological stressors (Figure 2.3.6). Stressors can in turn interact to impact a variety of key resources, which are found across marine/coastal, freshwater, and terrestrial ecosystems. Potential responses related to loss of apex predators, harvesting of native fish and wildlife, decline of key species groups, the rise of exotic species, and the influx of contaminants were considered so important that they form the core of this assessment, as described in Chapter 3.

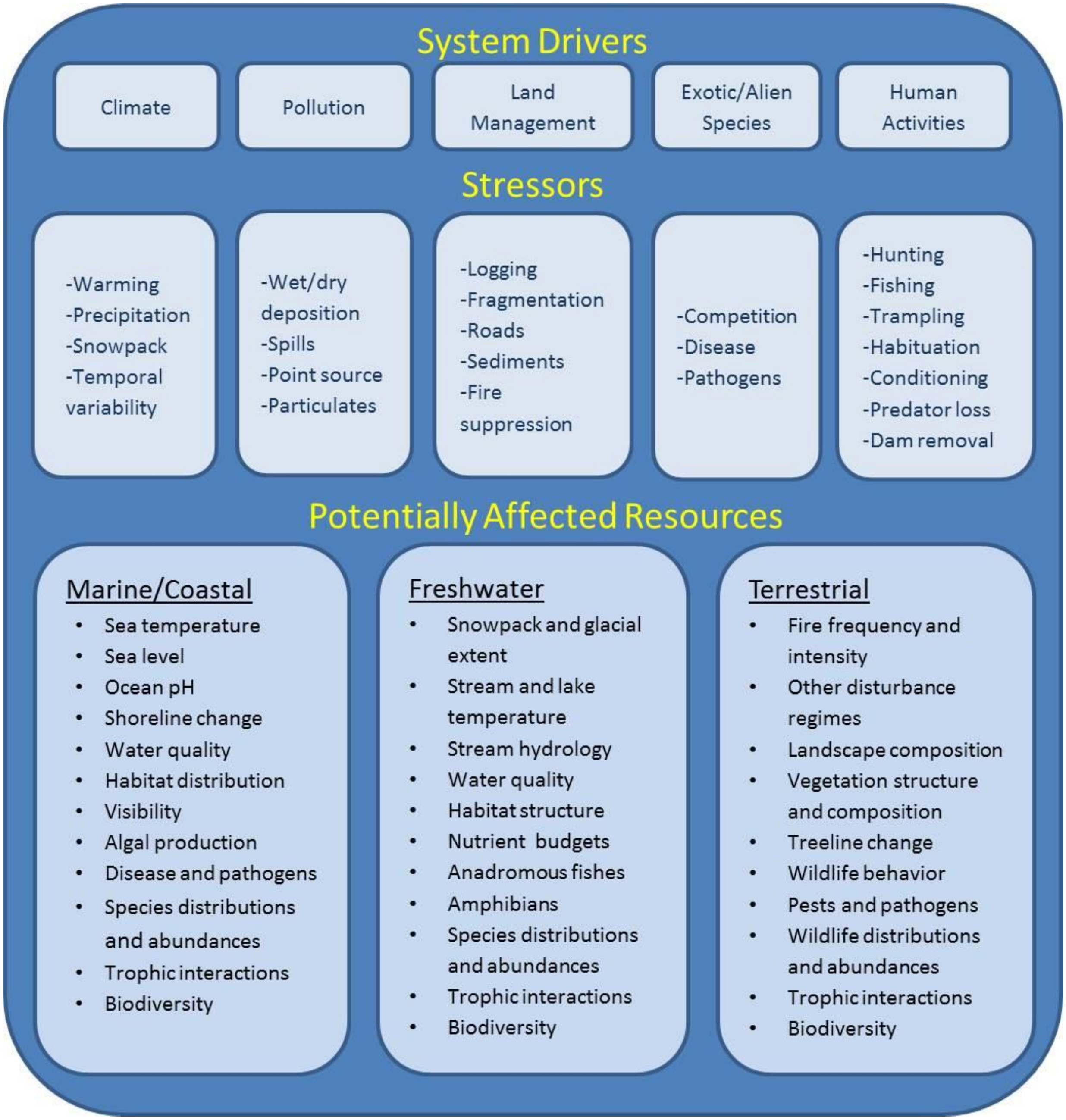

Figure 2.3.6. Conceptual model of ecosystem drivers and stressors and a sample of potential ecological responses in marine/coastal, freshwater, and terrestrial ecosystems of Olympic National Park. Additional detail is provided in Jenkins et al. (2003) and Weber et al. (2009).

2.4. Resource Stewardship

2.4.1. Management Directives and Planning Guidance

The NPS Organic Act (39 Stat. 535) provides the fundamental direction for managing national park lands in accordance with its mission of conserving "*the scenery and the natural and historic objects and the wild life therein and to provide for the enjoyment of the same in such manner and by such means as will leave them unimpaired for the enjoyment of future generations*." The NPS strives to achieve this mission in accordance with NPS statutes such as the National Parks Omnibus Management Act of 1998, other general laws such as the Clean Air Act, the Clean Water Act, the Endangered Species Act, the National Environmental Policy Act and Wilderness Act, as well as with all applicable federal regulations and agency orders.

Several milestones chart the evolution of natural resource policies of the NPS. In 1963, recommendations of an advisory committee chaired by A. Starker Leopold set the tone for decades of resource management policy of the NPS (Leopold et al. 1963). The Leopold Committee, charged with determining the goals and principles for wildlife management in the national parks, recommended that "*biotic associations within each park be maintained or where necessary recreated as nearly as possible in the condition that prevailed*" before the arrival of Europeans on the continent. The committee concluded that "*a national park should present a vignette of primitive America.*"

In 2006, the NPS revised management policies that interpret laws, establish frameworks, and provide specific direction for management decisions in national parks (NPS 2006). The updated policies placed emphasis on preserving not just individual species, features, and plant and animal communities, but to "*maintain all the components and processes of naturally evolving park ecosystems, including the natural abundance, diversity, and genetic and ecological integrity of the plant and animal species native to those ecosystems.*" These policies recognize natural change as an integral component of preserving naturally functioning ecosystems.

Today, the NPS management directives and guidance continue to evolve and mature. Recently a new advisory committee convened to revisit the "Leopold" document and provide a vision for the next century of management in the national parks (Colwell et al. 2012). The committee recognized that parks are now facing environmental changes of increasing complexity at an accelerating rate amidst growing uncertainties. The advisory panel recommended that the "*overarching goal of NPS resources management should be to steward NPS resources for continuous change that is not yet fully understood in order to preserve ecological integrity and cultural and historical authenticity, provide visitors with transformative experiences, and form the core of a national conservation land- and seascape.*"

In 2010, OLYM published an updated general management plan (GMP), which serves as a general construct for decision-making in managing the park, including the development of stewardship documents (NPS 2010). Specifically, the GMP defines desired resource conditions within the park, setting endpoints for management planning.

2.4.2. Status of Supporting Science

Olympic National Park is an active site for research in diverse disciplines. In addition to work conducted by the park's own staff in both cultural and natural resources, a large number of other researchers work in the park each year. These include university researchers from around the world as well as scientists from local, state, tribal, and national agencies. Since 1996, the U.S. Geological Survey (USGS) Forest and Rangeland Ecosystem Science Center (FRESC) has maintained a biological field station at OLYM dedicated to providing research in support of sound science-based decision-making in national parks (with a focus on OLYM). Collaborative research undertaken by the USGS and NPS has focused primarily in fields of wildlife, fisheries, forest, and landscape ecology, as well as working with park staff in developing resource inventory and monitoring programs. The park's staff and ecologists with the NCCN perform inventory and monitoring of key resources through the NCCN Inventory and Monitoring Program (Table 2.4.1). Park staff members also conduct surveys needed to monitor permitted resources (e.g., fish populations) and scientific work related to NPS restoration (e.g., Elwha River restoration). By park policy, no permits are issued for work done by NPS staff (and USGS collaborations with NPS staff). Outside researchers are issued permits after peer review and review by the park.

Table 2.4.1. North Coast and Cascades Network natural resource monitoring programs in Olympic National Park. Source: NPS.

Program (Resource)	Protocol
Alpine/Subalpine Vegetation	Rochefort et al. (2012)
Climate	Lofgren et al. (2010)
Elk	Griffin et al. (2012)
Fish Assemblages	Brenkman and Connolly (2008)
Forest Vegetation	Acker et al. (2010)
Intertidal	Fradkin and Boetsch,(2012)
Landbirds	Siegel et al. (2007)
Landscape Change	Antonova et al. (2012)
Large Lowland Lakes	Fradkin (2013)
Mountain Lakes	Glesne et al. (2012)
Water Quality	Rawhouser et al. (2012)

The number of research permits issued annually by the park has increased since permitting was initiated in 1990. During the 1990s about 12-25 research permits were issued annually, whereas since 2006, the number of permits issued annually has approached or exceeded 70. The Omnibus Parks Management Act of 1998, which challenged the parks to openly encourage research, may have led to some of the increase in research that has occurred in the park since 2000. Implementation of a national computerized Research Permit and Reporting System (RPRS) in 2002 made it easier to track and administer applications consistently, and also likely influenced the increased permitting seen in recent years.

The diversity of OLYM research topics is especially broad because the park's resources range from alpine glaciers and rainforest to wild rocky beaches. Permitted research encompasses work in both the physical and biological sciences. Examples of recent permits in the physical sciences include studies of marine hydrothermal vent deposits, and a project monitoring tectonic processes and seismicity of the North Cascadia subduction zone. Biological research ranges from studies of individual species to projects that examine the ecology of broader communities or entire ecosystems.

Recent permits have included studies of native freshwater mussels, ice worms, marbled murrelets, Olympic marmots, bumblebee pollinator communities, songbird communities, and fungal sporocarps. Outside researchers thus contribute enormously to the collective understanding of OLYM resources.

Lake Crescent. Photograph courtesy of NPS/Bill Baccus.

Chapter 3. Study Scoping and Design

Kurt Jenkins and Rebecca McCaffery

3.1. Preliminary Scoping

The conceptual framework, guiding questions, and focal resources for this NRCA were generated during two scoping meetings. Prior to USGS involvement, the park's staff held an initial scoping meeting with NPS NRCA program leaders to outline general themes and priority topics for this NRCA. Park staff identified a list of focal resources for potential assessment (Appendix C), which were subsequently distilled into a shorter list of top priorities. These resources were selected based on their alignment with three guiding questions the park staff established as the foundation for this NRCA. These questions were presented to potential bidders as the basis for developing competitive proposals:

1. What is the current health of the Olympic ecosystem and in what ways does it differ from the way it was before European settlement?
2. What are the ecosystem effects of harvest, poaching, or decline of sensitive species?
3. What species or ecosystem functions are threatened by non-native invaders?

The USGS, in their funding proposal, subsequently refined these three questions on the basis of supporting materials provided to all potential bidders in the Statement of Work. Supporting materials clarified that Question 1 centered on assessing the condition of resources affected by the loss of wolves as the apex predator in Olympic National Park. Other supporting materials highlighted the need to address the condition of air resources within the park. Consequently, the USGS team morphed the three guiding questions into the following five, which have persisted as the guiding questions driving subsequent development of this NRCA

1. What are the effects of the loss of top predators on ecosystem health?
2. What are the effects of resource harvesting for human use on ecosystem function and sustainability of the resource?
3. What are the population trends and ecological impacts of key declining and/or sensitive species or communities?
4. What species or ecosystem functions are threatened by non-native or invasive species?
5. What are the ecosystem effects of airborne contaminants?

Following the funding award to USGS, members of the USGS assessment team held a second scoping meeting on 16 October 2014 with OLYM staff to establish a common vision of the proposed NRCA in OLYM. Specific objectives of the meeting were to review, confirm, and refine the guiding questions of the NRCA and specific focal topics under each, as well as to identify site-specific data and resources available for assessing each focal resource. For the latter objective, we identified

USGS and NPS team members who could make data available and provide expertise for each resource assessment. Seventeen individuals (two by telephone) attended the meeting, including seven USGS representatives from three USGS Science Centers, nine representatives from OLYM and one from the NPS ARD (Appendix D). Park representatives emphasized that their focus on guiding resource questions was driven by national programmatic direction suggesting that the chosen indicators or measures of resource condition should scale up to provide a larger view of important ecosystem characteristics and management issues.

Participants collectively reviewed each of the five guiding questions and a preliminary list of focal resources that fell within the scope of each question. The merits and disadvantages of including various resources under the banner of each question were discussed while considering the urgency of resource issues, the availability of data and resources to provide a substantial assessment useful to resource planning, and redundancy of certain topics with other assessments or management reviews. For example, we concluded that the impacts of climate change on numerous park resources are being explored at national, regional (e.g., Safeeq et al. 2014, Lee et al. 2015), and local levels (Halofsky et al. 2011b), and thus would not comprise a major focus of this assessment. Similarly, assessment of subalpine and alpine ecosystem condition in relation to non-native mountain goats has already been completed in the 2017 draft Environmental Impact Statement, which outlines different management strategies for the mountain goat population found in OLYM and surrounding national forest lands (https://parkplanning.nps.gov/document.cfm?parkID=329&projectID=49246&documentID=77644, accessed 30 October 2017). We also identified resources that were not on the initial list of focal resources that are likely to have far-reaching influences within park ecosystems. For example, we assessed the impact of non-native balsam woolly adelgid beetles on subalpine fir forests, and sea star communities affected by a severe outbreak of sea star wasting disease.

By the end of the meeting, USGS and NPS partners agreed on twelve focal resources to be assessed under the banner of the previously identified guiding questions. Both sets of partners understood that aligning this NRCA along the axes of the guiding resource questions facing park managers today represented a deviation from the more common broad-based assessment of all park resources. NPS participants at the meeting, however, affirmed their belief that this structure would provide the most immediately useful assessment of resource conditions in the park.

3.2. Study Design

3.2.1. Indicator Framework, Focal Study Resources and Indicators

The guiding questions presented by OLYM laid the foundation for identifying focal resources of this assessment (as described in the previous section). For each focal resource, we identified specific measures of resource condition that could provide an ecosystem perspective of the key issues while keeping with park goals. We were guided by a hierarchical framework for selecting key measures by considering measures of resource composition, structure and function at landscape, community, and population levels of ecological organization relevant to each of the key resource issues (*sensu* Noss 1990).

This process, accomplished through many discussions with park biologists, yielded the final list of focal resources and the specific ecological measures that comprise this assessment (Table 3.2.1).While these resources and measures do not encompass all of the aquatic, marine, and terrestrial biodiversity of the park and its ecosystems, they are representative of issues of concern in OLYM related to ecosystem integrity and maintenance of biodiversity. This assortment also represents both biotic and abiotic aspects of each of the park's major ecosystems, including coastal, freshwater aquatic, and both low-elevation temperate and high-elevation subalpine ecosystems. The indicators represent an amalgam of system processes and components that embrace substantial taxonomic diversity, including mammal, bird, amphibian, fish, invertebrate, and plant resources.

Table 3.2.1. Focal Olympic National Park resources and their assessment categories.

Guiding Question	Focal Resource	Assessment Measures
Guiding question 1: What are the effects of the loss of top predators on ecosystem health?	Roosevelt Elk	• Temporal and spatial trends in relative abundance (winter range) • Temporal trends in sex and age composition (migratory herds)
	Riparian Vegetation	• Temporal changes in percent vegetation cover of elk browse species • Temporal changes in recruitment of selected overstory trees in established floodplains and fluvial terraces • Percent conifer vegetation along the active river channel that could become key members in debris jams • Valley floor patch composition along the major west-side rivers systems
	River Geomorphology	• For four major OLYM rivers, temporal changes in: • Width of the active flow zone • Channel braiding index
Guiding question 2: What are the effects of resource harvest for human use on ecosystem function and sustainability of the resource?	Pacific Salmonids	• For each of 17 salmonid stocks found in each of five river systems: • Trends in escapement and terminal run size • Harvest rates and the annual number of hatchery and wild salmonids harvested • Annual number of hatchery releases
	Razor Clams	• For the focal population on Kalaloch beach: • Annual population size • Size-class distribution over time • Average adult size over time

Table 3.2.1 (continued). Focal Olympic National Park resources and their assessment categories.

Guiding Question	**Focal Resource**	**Assessment Measures**
Guiding question 3: What are the population trends and ecological impacts of key declining and/or sensitive species or communities?	Olympic Marmots	• Long-term occupancy of select colonies • Contemporary park-wide occupancy • Historical and current demography (survival and reproduction)
	Sea Stars	• For *Pisaster orchraceus*: • Relative abundance at four monitoring sites • Size frequency over time at four monitoring sites • Monthly prevalence of sea star wasting disease (SSWD) at a focal site • For the sea star community: • Population trends of two other species • Incidence of SSWD in all OLYM sea stars
	Amphibians	• Species diversity and distribution throughout the park • Demography of a focal Cascades frog population
Guiding question 4: What species or ecosystem functions are threatened by non-native or invasive species?	Barred Owl (Northern Spotted Owl)	• Temporal trend in occupancy of northern spotted owls at historical territories, and occupancy of barred owls at these same territories • Changes in local extinction and colonization rates of spotted owls and barred owls over time Consequences of invading barred owls on territory occupancy dynamics of spotted owls
	Balsam Woolly Adelgid (Subalpine Fir)	• Distribution of BWA-infested subalpine fir forest over time • Extent of subalpine fir forest decline over time • Basal area of dead and dying trees
	Invasive Non-native Plants (Native Plants)	• Spatial distribution of invasive non-native plant species • Rate of change of non-native or invasive non-native species abundance in areas having repeated surveys • Status of invasive non-native species establishment in the dewatered reservoirs following dam removal on the Elwha River
Guiding question 5: What are the ecosystem effects of airborne contaminants?	Air Quality and Contaminants	• Visibility • Ozone effects on human health and vegetation health • Nitrogen and sulfur wet deposition • Mercury concentration in focal biota • Summary of other contaminants

We note that the resources selected for each guiding question could also, in many cases, pertain to other guiding questions. For example, northern spotted owls and razor clams represent sensitive and declining resources in the park, but their statuses are covered in the invasive species (where spotted owls are impacted by barred owls) and harvested species guiding questions, respectively. As another example, Olympic marmots represent a species that may indirectly be affected by the loss of wolves (guiding question 1), but they are assessed in the sensitive and declining species guiding question. Throughout the report, we highlight these overlaps in topics and resources.

3.2.2. General Approach and Methods

We maintained frequent communication with NPS resource experts to identify relevant data sets and literature available for assessing historical and current conditions and trends of each focal resource. These materials often included (1) existing databases that were largely suitable for analysis in their current conditions or with minor revision; (2) geospatial data; and (3) published and unpublished reports containing previously completed analyses and interpretations of historical or present resource conditions and trends. For certain resources, park biologists played a primary role in the assembly of data and assessment of the resource (e.g., razor clams, salmon, and sea stars). Assessments were made either for the resource as a whole, or for the resource within specific watersheds, vegetation types, or other ecological designations as spatial grain of resource data permitted.

Specific methods and approaches for assessing resource condition varied considerably among resources, reflecting the wide variety in the types and qualities of data and written materials available. Each resource assessment, however, followed the same general structure: (1) Introduction, (2) Approach and Methods, (3) Reference Conditions, (4) Results and Assessment (5) Emerging Issues, (6) Information Needs/Gaps, and (7) Resource Summary. First, we introduced the resource, highlighting its ecological and management importance, any factors suspected of affecting resource condition, and its relationship to the specific metrics or measurements used in the assessment. The approach and methods of analysis were described by first identifying the specific measures of resource condition and then briefly describing field data collection and analysis methods. Detailed descriptions of analytical methods were referenced in appendices. For each resource we identified a reference condition to use as a benchmark for condition assessment.

To identify reference conditions for each resource, we relied on the reference standards elaborated by Stoddard et al. (2006). These include: (1) historical condition (conditions preceding human disturbances), (2) minimally or least disturbed condition (the least-disturbed conditions found today), and (3) best attainable condition (goals established by best possible management practices). However, we occasionally had to present reference conditions outside these categories due to limited historical data records or other lack of information. For example, we rarely had data on conditions prior to human disturbance. Hence, we frequently relied on historical information to identify less disturbed conditions compared to the present, or a range of historical variability that provided a reference standard.

In the results and assessment section, we interpreted resource conditions relative to the reference conditions based on the qualitative or quantitative data presented, examined trends in resource status, and identified the important limitations or caveats associated with these interpretations. Subsequent sections on emerging issues and data needs provided information that park managers can use to prepare plans to understand and/or manage resource conditions in the future. We included further discussion of the results, emerging issues, and data needs for each focal resource in Chapter 5.

In the resource summary, we encapsulated our assessment of current condition and trend of each resource, using a combination of text and standardized NRCA coding symbols (Table 3.2.2). We first assessed the *status* of each measure for a particular resource (Table 3.2.1). For individual measures, trend (i.e., arrow) in the measure was not generally delineated in the symbology, but was described where appropriate in a short rationale sentence. In the rationale, we would then verbally describe that the invasive species is increasing in spatial extent.

Table 3.2.2. Symbol definition used in resource condition summaries in Chapters 4 and 5.

Condition Status		**Trend in Condition**		**Confidence in Assessment**	
🟢	Resource is in Good Condition	⇧	Condition is Improving	◯	High
🟡	Resource warrants Moderate Concern	⬄	Condition is Unchanging	○	Medium
🔴	Resource warrants Significant Concern	⇩	Condition is Deteriorating	◌	Low

After assessing each individual measure for a resource, we used those results to determine a summary of condition status, trend in condition, and our confidence in the assessment for the whole resource. If we had no information on the trend in condition, we did not include an arrow. For example, in some assessments we saw changes in the resource over time, but did not believe those changes represented “improvements” or “deterioration” in the condition. We also included a brief rationale for our summary assessment, which explained the measures and results leading to that designation. A sample table structure is provided in Table 3.2.3. For certain resources with complicated sets of measures that are distinct in geographical extent, species, or ecological indicators (e.g., salmon and air quality), we reported the summary condition as “variable” and referred the reader to the condition identified for individual measures. Finally, for the invasive species/communities we assessed, we include two symbols for the overall condition and trend (Chapters 4.4.1 – 4.4.3). The first symbol shows the level of concern for the invasive species/community and the confidence in the assessment overall, without indicating a trend. The second shows the condition of the impacted native species/community, with a trend arrow.

Table 3.2.3. Sample table structure used in resource condition summaries in Chapter 4.

Resource/Measure	**Condition Status and Trend**	**Rationale**
Measure 1	*Status bubble for measure 1*	*Rationale for measure 1*
Measure 2	*Status bubble for measure 2*	*Rationale for measure 2*
Measure 3	*Status bubble for measure 3*	*Rationale for measure 3*
Resource 1	*Overall condition and trend bubble*	*Overall rationale*

Shelf fungi. Photograph courtesy of NPS.

Chapter 4. Natural Resource Conditions

4.1. What are the effects of the loss of top predators on ecosystem health?

Historically, wolves, cougars (*Puma concolor*), and black bears (*Ursus americanus*) were the top mammalian predators on the Olympic Peninsula. An estimated 115 wolves lived on the peninsula during the early 1900s, and an estimated 180 cougars and 1300 black bears inhabited the Olympic Forest Reserve from 1910-1918 (Scheffer 1995). Early pioneers and settlers of the peninsula persecuted wolves and cougars during the first half of the 1900s, motivated by bounties paid from 1905 to 1960 and by concerns for personal safety and protection of wild game populations and livestock. Wolves were widely believed to have been eliminated on the peninsula by the 1920s (Scheffer 1995). Cougars persisted throughout the bounty era, although their numbers were undoubtedly reduced. Bears were not classified as predators or controlled by bounty laws during the first half of the 20th century, although harvests were unrestricted. Beginning in the 1950s, however, bears were classified as predators in western Washington and hundreds were killed each year until the early 1980s in targeted control efforts designed to reduce feeding damage on commercially valuable trees (Tirhi 1996). Despite efforts to reduce both cougars and black bears historically, current populations of both species remain relatively healthy throughout Washington (WDFW 2015), and the wolf is the only top predator missing from the OLYM ecosystem.

The loss of wolves from the Olympic Peninsula could have resulted in diverse changes to its ecosystems. Wolves are often considered keystone species by virtue of their outsized effects on ecological communities and ecosystems (Ripple and Beschta 2004, Eisenberg 2010). Specifically, the loss of a top predator such as the wolf can trigger what is known as a trophic cascade (Paine 1980), where ecological effects of predator loss cascade through multiple levels of the food chain to fundamentally alter ecosystems (Schmitz et al. 2000, Prugh et al. 2009, Terborgh and Estes 2010, Estes et al. 2011). Wolves are widely known to limit populations and affect behavior and distribution of their primary ungulate prey (Leopold 1943, Gasaway et al. 1992, Winnie and Creel 2007, Gower et al. 2009). This effect on ungulate populations is important, because herbivore abundance and distribution play a key role in shaping the structure and function of vegetation across many levels of ecological organization (McNaughton 1985, Pastor and Naiman 1992, McLaren and Peterson 1994, Olff and Ritchie 1998, Singer et al. 2003). Changes in herbivore numbers as a result of loss or reduction of top predators has played a key role affecting the establishment of several palatable tree and shrub species in several U.S. and Canadian national parks, notably trembling aspen (*Populus tremuloides*) as well as other species of cottonwoods (*Populus* spp.) and willows (*Salix* spp.) (Hebblewhite et al. 2005, Beschta and Ripple 2009). Changes in plant community structure and composition initiated by a predator decline has been linked to changes in community composition and population size of certain taxa including butterflies, reptiles and amphibians (Ripple and Beschta 2006), birds (Berger et al. 2001, Hebblewhite et al. 2005), and beavers (Hebblewhite et al. 2005), as well as changes in geomorphic characteristics of river systems (Beschta and Ripple 2008, 2012b).

Wolves and other top predators can also affect densities or distributions of other predator species in the community with whom they compete. Populations of cougars have been shown to decline in numbers, shift distributions, and alter prey selection in response to an influx of wolves (Kortello et al. 2007, Bartnick et al. 2013, Elbroch et al. 2015). Further, the elimination of wolves may lead to increased densities of smaller carnivores with whom they also compete—a process known as mesocarnivore (i.e., mid-sized carnivore) release (Prugh et al. 2009, Ritchie and Johnson 2009). Mesocarnivores tend to be more abundant than top-level predators and some are more general in their feeding habits than wolves, potentially eliciting changes in populations of smaller prey that are not normally impacted to the same degree by wolves. A change in the abundance of coyotes, for example, has been linked to changes in populations of small mammals or birds (Crooks and Soulé 1999, Miller et al. 2012).

The recognition of critical linkages between top predators and myriad components of naturally functioning ecosystems has ignited concern over the ‘trophic downgrading” of ecological systems around the globe (Estes et al. 2011). In the case of OLYM ecosystems, as in other parts of western North America, Beschta and Ripple (2008) postulated that overabundant elk populations resulting from the loss of wolves initiated a trophic cascade involving the reduction of important riparian tree and shrub species and disruption of key riverine processes. They concluded that rivers widened and changed from primarily single-threaded to braided morphology following the increase in elk population during the early 20th century. Moreover, they speculated that widening of river channels and the loss of key tree species have disrupted the connectivity of floodplain forests and the delivery of large wood to rivers, with potential additional impacts to river channel dynamics and fish habitat.

In the following three sections, we examine the effects of wolf extirpation on the health of low-elevation floodplain and riverine ecosystems in OLYM. Ecosystem health, the endpoint of our assessment, has been defined in myriad ways (Costanza and Mageau 1999). To guide our assessment, we adopted the definition of a healthy ecosystem as one that is sustainable by virtue of maintaining its organization (i.e., structural diversity), vigor (i.e., primary productivity), and resilience to change (Rapport et al. 1998, Costanza and Mageau 1999). We recognize that ecosystems are comprised of many moving parts and that maintaining a redundancy of key ecological functions is a characteristic of sustaining healthy ecosystems in the face of change (Marcot and Vander Heyden 2001).

In evaluating ecosystem health in response to wolf eradication, we focused on three resources of particular interest to the park that may have been affected by the loss of wolves—Roosevelt elk, riparian vegetation along low elevation west-side rivers, and river geomorphology. Each of these resources is important in its own right, but together they comprise the principal elements of the hypothesized trophic cascade that may have accompanied the loss of wolves. For each of these resources we scoured historical accounts for clues suggesting early ecosystem conditions. We searched for unanalyzed data sets, aerial photographic records, or previously published literature that shed light on resource status and condition. Our assessment is guided by lessons learned from the Greater Yellowstone Ecosystem: multi-causal systems are rarely explained by a single explanation having to do with wolves (Mech 2012, Marris 2014, Peterson et al. 2014). Hence, we question the causes of observed changes in the selected resources, explore whether changes seem out of the bounds of natural variability, and determine whether they signal a threat to the sustainability of low-elevation riverine and forest ecosystems.

4.1.1. Roosevelt Elk

Kurt Jenkins and Patricia Happe

Introduction

As key components of lowland and mountain ecosystems in the Pacific Northwest, elk are tightly woven into the ecological, historical, and cultural fabrics of OLYM. Part of what would later become OLYM was first set aside as Mount Olympus National Monument to protect remnant herds of Roosevelt elk after the subspecies had been eliminated from most of its historical range and decimated in parts of the Olympic Peninsula. This protection was later expanded when the national park was formed to include a greater measure of the elk's low-elevation forested winter range (Figure 2.1.3). Today, OLYM is year-round or seasonal home to the largest relatively undisturbed population of this unique coastal form of elk throughout its range. Roosevelt elk populations protected within the park, where natural processes prevail over anthropogenic disturbances, display naturally evolved behaviors, social systems, and population characteristics found nowhere else (Jenkins and Starkey 1982, Houston et al. 1990). The sight of elk amidst a towering forest, on a river bar, or grazing in a subalpine meadow remains a touchstone for many visitors to the park.

Roosevelt Elk in the Quinault Valley of Olympic National Park. Photograph courtesy of Chuck and Grace Bartlett.

Roosevelt elk and Columbian black-tailed deer (*Odocoileus hemionus columbianus*) are the two native ungulate species (herbivorous hooved mammals) found on the Olympic Peninsula. Elk and deer are important players within the park's ecosystems, fueling higher trophic levels of predators and scavengers, while also shaping the structure and composition of plant communities through their grazing and browsing (Happe 1993, Woodward et al. 1994, Schreiner et al. 1996). Both species are important prey for cougars and black bears (Raedeke et al. 2002:462, Murphie and McCoy 2015), which appear to be thriving in the park based on the frequency of reported sightings (NPS, unpublished data). Carcasses of ungulates dying from natural causes provide winter and spring food for a diverse community of mammalian and avian carnivores and insectivores, as has been demonstrated in Yellowstone National Park (Houston 1978). The pronounced effect of elk and deer on vegetation continues to generate debate on whether overbrowsing has caused long-lasting damage to park ecosystems (explored further in Chapters 4.2 and 4.3). Although elk and deer are both important cogs in the park's food webs, we focused this analysis on the status and trends of Roosevelt elk due to the historical importance of elk to the park, the comparatively abundant information on Roosevelt elk, their numerical superiority in the temperate rainforests (Leslie et al. 1984), and consequently their dominant influence on lowland forest and riverine ecosystems.

Roosevelt elk are found throughout most of OLYM although they demonstrate considerable variability in both temporal and spatial abundance patterns due to their seasonal movements. Elk are most abundant in the western rainforest valleys and are generally least abundant in the much drier northeast region of the park (Houston et al. 1990). Two life-history patterns describe endpoints in a range of seasonal movement patterns exhibited by elk in OLYM: year-round resident (i.e., non-migratory) elk inhabit low-elevation home ranges throughout the year, whereas migratory herds may travel up to 30 km seasonally between high-elevation summer ranges and winter ranges in adjoining lowland valleys (Figure 4.1.1). Recent telemetry studies indicate there are many variations in seasonal movement patterns between migratory and resident life histories (NPS, unpublished data). Typically migrants spend the winter as far up valley as snow permits, and move to the subalpine zone during early summer following snowmelt and the onset of herbaceous vegetation growth at higher elevations. Other elk move down valley during the winter and up valley during the summer but never migrate to subalpine summer ranges. Other primarily low-elevation herds may exhibit pronounced short-distance altitudinal movements within a season. The distinction between short-term altitudinal movements and long-distance migrations are often blurred, and individual elk may exhibit different migratory strategies over time (NPS, unpublished data).

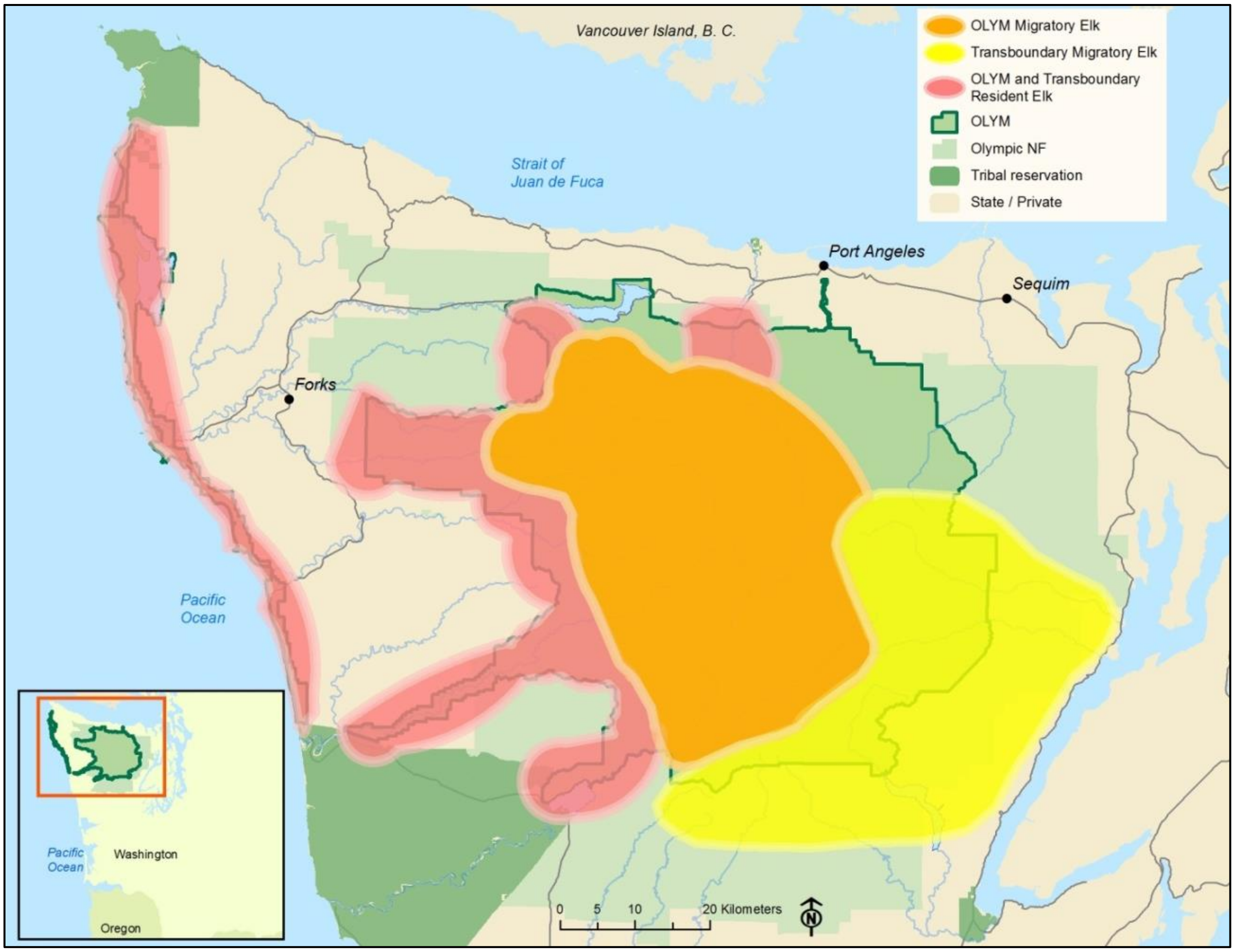

Figure 4.1.1. General distribution of migratory and resident (non-migratory) elk herds in Olympic National Park. Migratory herds that summer in the park and winter outside the park are identified as transboundary migratory herds. The shaded boundary on the periphery of each distribution is meant to indicate that distributions are approximations only. Herd movement patterns are variable and may overlap in many areas. Source: NPS.

There are many complex processes that could influence the park's elk, their habitat, and elk-habitat relationships in the future (Houston et al. 1990). Olympic National Park is not a complete ecological unit for many elk herds: several migratory herds, particularly on the eastern and southern ranges (Figure 4.1.1), leave the park during the winter where they experience changing patterns of land use, exurban development, predator populations, and hunting exposure. Moreover, there are resident elk herds, predominantly in the park's western valleys, that live on or near the park boundary and are also influenced by adjacent human pressures. In addition to the ongoing influence of human activity outside the park, there has been a recurring interest in actively reintroducing wolves to OLYM (Ratti et al. 2004). Wolves that recently colonized Washington State also have the potential to recolonize the peninsula naturally (Wiles et al. 2011). The return of wolves to the Olympic Peninsula would affect the dynamic relationships that exist among populations of both predators and their prey, including elk. Further, as climate changes in the Pacific Northwest, elk populations will be influenced by shifts in temperature, precipitation, and snowpack (Salathé et al. 2009); forest disturbance regime (Dale et al. 2001, Westerling et al. 2006); and vegetation (Zolbrod and Peterson 1999). Lastly, the increased prevalence and proliferation of diseases is a growing concern of wildlife managers nationally (Daszak et al. 2000, Angers et al. 2006). The recent identification of bacterial hoof infections among elk in southwestern Washington poses a significant and immediate threat to elk throughout the region (Han and Mansfield 2014). In this assessment, we examine historical and recent trends in OLYM's elk populations in relation to past, current, and potential future drivers of elk population change.

Approach and Methods

We assessed two measures of Roosevelt elk population condition within the park: (1) temporal and spatial trends in the relative abundance of elk within and among key winter ranges of the park, and (2) temporal trends in sex and age composition of migratory elk herds on their high-elevation summer ranges. Trends in abundance reflect the overall status of elk on selected primary winter ranges where both resident and migratory elk herds intermix during winter and spring. Trends in sex and age composition of elk on their summer range signal changes in population productivity (age composition) or sex-specific survival rates (sex composition) of the migratory population component.

Relative Abundance

We assessed long-term trends in relative abundance of elk by examining (1) the historical record, (2) recent aerial counts of elk in selected winter ranges of the park, and (3) extensive ground-based surveys of elk fecal pellet groups throughout all the primary winter ranges within the park.

The writings of early pioneers, foresters, and biologists on the Olympic Peninsula provided general information on historical numbers of elk on the Olympic Peninsula and within OLYM. Unfortunately, changing administrative boundaries of the forest reserve, monument, and the early park introduced uncertainty in the interpretation of long-term trends of elk populations within the current park boundary. Early observers rarely reported the spatial extent of their observations and whether population estimates included migratory elk that leave the park during winter or year-round residents only. In an attempt to minimize these uncertainties, we estimated trends in the park by summing historical counts provided for the primary winter ranges that lie mostly within the current park boundary, including the Elwha, Sol Duc, Calawah, Bogachiel, Hoh, Queets, and Quinault Valleys (Houston et al. 1990).

Against this historical backdrop, we examined recent trends in numbers of elk counted during helicopter surveys within trend-count areas delineated on three of the park's primary elk winter ranges—the Hoh, South Fork Hoh, and Queets Valleys (Figure 4.1.2; Houston et al. 1987, Jenkins et al. 2015). Trend-count areas comprised the valley floor (defined as the relatively flat glacial and fluvial terraces and floodplains; (Fonda 1974, Latterell et al. 2006). With the exception of the South Fork Hoh area, which was already quite small compared to the other two, areas adjacent to the park boundary were excluded from the trend-count areas when they were first delineated (i.e., including the entire Queets corridor; Figure 4.1.2). Helicopter surveys were conducted periodically from 1985 to 2010 by a pilot and three observers during early spring mornings when elk are drawn to open-canopied forests to feed on the first greening vegetation (Jenkins and Starkey 1984, Houston et al. 1987). We surveyed from a safe altitude above tree level at an average airspeed of 55 km/hr along parallel flight lines spaced about 200-300 m apart. Experience suggested that this speed and spacing allowed efficient and complete survey coverage of the trend-count areas while permitting easy identification of duplicate observations from adjacent transects. We assessed the effects of several environmental variables on elk counts, including survey date, minimum temperature on the morning of the survey, and the cumulative number of growing degree days (>40°F) preceding the survey (measured from 1 February; Jenkins et al. 2015). We assessed trends in the raw numbers of elk counted while adjusting for effects of those same environmental variables (see Jenkins et al. 2015 for additional details on analytical methods). We computed the annual rate of population growth ($\lambda = e^r$), where e is the base of the natural logarithm and *r* is the slope of the natural log-transformed counts plotted against time (Mills 2013:81).

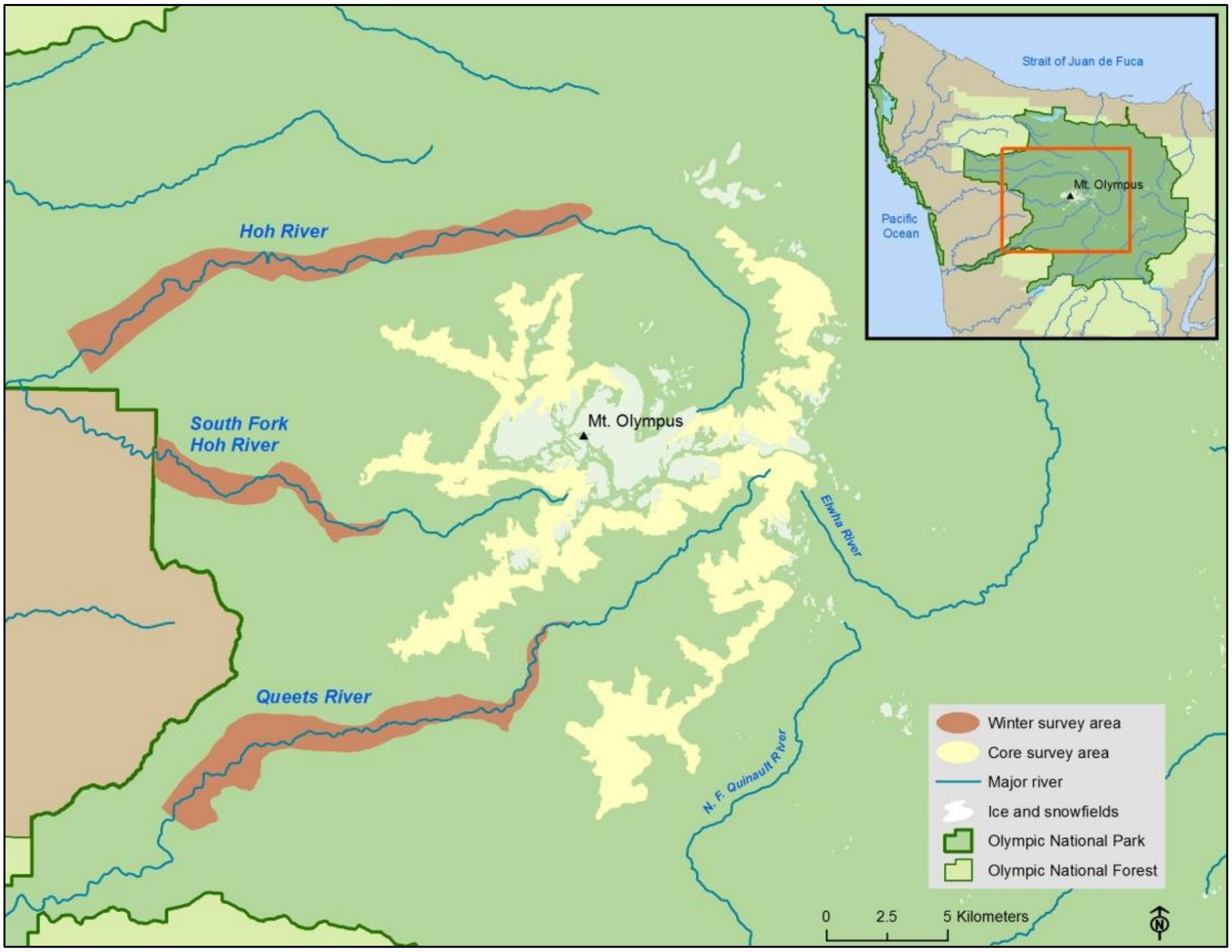

Figure 4.1.2. Boundaries of winter trend-count survey areas corresponding with key winter ranges of elk along the Hoh, South Fork Hoh, and Queets Rivers. The core survey area was surveyed during late summer and generally corresponds with summer ranges of elk migrating from the winter survey areas. The Hoh, South Fork Hoh, and Queets trend-count areas are approximately 27, 11, and 24 km^2, respectively, whereas the core summer range survey area is approximately 100 km^2. Source: NPS.

We inferred differences in the relative abundances of elk among key winter ranges of the park based on previous studies of elk fecal pellet group densities (Jenkins and Manly 2008). We estimated pellet group densities from observations made along a random sample of line transects distributed throughout low-elevation winter ranges (<600m in elevation [about 2,000 ft]) during two winters from fall 2001 to spring 2003 (Jenkins and Manly 2008). Pellet-group densities reflect spatial patterns of elk densities only if defecation rate is constant, and elk pellet groups are equally visible and decay at similar rates across environmental gradients. To better standardize counts made among various regions of the park, we developed and applied statistical models to account for variations in pellet group detectability and decay rates related to different observers, vegetation, and precipitation gradients (Jenkins and Manly 2008).

Sex and Age Composition

Ratios of males and young of the year to adult female elk were measured previously during: (1) ground counts of elk made on a key summer range of elk on the High Divide during the late 1970s (where upper reaches of the Sol Duc, Elwha, Bogachiel and Hoh Rivers come together; Smith 1980) (2) helicopter-based composition counts of elk on summer ranges in the upper Quinault to Elwha basins conducted from 1984-86 (NPS, unpublished data), and (3) recent helicopter surveys from 2008-2015 (Jenkins et al. 2015). The 2008-2015 summer surveys were conducted over a subset of the area covered by the previous 1984-86 surveys, corresponding with portions of the upper Hoh, South Fork Hoh and Queets Valleys (i.e., summer ranges aligned with the spring trend-count areas; Figure 4.1.2). Because ratio data are highly influenced by small samples, incomplete counts, or seasonal variation in sex and age distributions of elk, we limited the analyses to reliable counts of >100 elk obtained during late summer (generally mid-August to early October) when males and females with young are aggregated prior to and during the fall mating season. Observers classified and counted all elk seen in the following sex and age categories: adult females (cows), young of the year (calves), yearling males (spike bulls), subadult males (2-4 antler tines on a side) and adult males (5 or more tines on a side). However, during the 1984-86 surveys, subadult and adult males were classified together as adult males. From 2008-2015, we employed a double-observer sightability model to adjust all raw counts of elk seen from the helicopter for detection biases (Griffin et al. 2013). Although the correction for detection biases had minimal effect on population composition ratios, we present both the corrected and uncorrected estimates of population composition derived from the 2008-2015 surveys so that composition ratios are directly comparable to the earlier uncorrected estimates.

Reference Conditions

We interpreted recent population trends relative to the historical range in elk numbers. Elk numbers within this historical range of variability, preferably not at the extreme values when elk were considered to be at dangerously low (early 1900s) or problematically high levels (1920s-1930s), were considered acceptable.

We interpreted the sex and age composition of park elk herds relative to hunted populations outside the park and other unhunted herds of elk in western North America. Mature elk generally comprise >35 males per 100 cows in unhunted naturally regulated populations (Flook 1970, Houston 1982). Further, because selective male hunting may skew sex ratios sufficiently to affect breeding efficiency in some heavily hunted populations (Noyes et al. 1996), Washington State strives to maintain 15-35 bulls:100 cows measured before the hunting season or 12-20 bulls after the hunting season in areas managed for general hunting (WDFW 2015). Based on these considerations, we determined that optimum bull:cow ratios in a protected national park such as OLYM should exceed 35 bulls:100 cows.

Results and Assessment

Relative Abundance

Historical Context: Long-term Trends in Abundance

Pre-EuroAmerican Settlement

Few early writers ventured to guess at the numbers of elk present on the Olympic Peninsula before EuroAmerican settlement, but all agreed that elk were abundant at that time (Appendix E). Suckley and Cooper (1860) reported that elk were "*especially abundant*" on the northern slope of the Olympic Range following their exploration of a route for a transcontinental railroad in the mid-1850s. Gilman (1896), one of the first explorers of the western Olympics in 1889-90, remarked that subalpine meadows were "*favorite ranges for large numbers of the elk that are common all over the peninsula*." M. P. Skinner (1933b), a field naturalist commissioned by the Boone and Crockett club in 1933 to investigate elk on the Olympic Peninsula, speculated that "*probably there were as many as 25,000; possibly as many as 40,000 elk [prior to EuroAmerican settlement]*."

EuroAmerican Settlement and Early Exploitation

Early U.S. Forest Service reports indicated that elk populations declined during the late 1800s as settlers inhabited the peninsula. Morgenroth (1909), the first ranger of the Olympic National Forest Reserve, reported that elk were abundant on the peninsula until about 1890, but were all but exterminated by about 1905 on the north and east sides of the mountains. He reported that "*on the west and parts of the south side, [elk] were still plentiful, this part of the country being too densely forested and inaccessible for the average hunter.*" At the low point in the early 1900s, following decades of unregulated exploitation, there were believed to be only 1,800-2,000 elk remaining on the Olympic Peninsula (Hanson 1905, Morgenroth 1909; Figure 4.1.2).

Market exploitation likely played a key role in the depletion of elk in the more accessible areas. In addition to subsistence uses of elk by Native Americans and early settlers, meat, hides and antlers of elk were also sold. There are trading records itemizing elk and deer hides and meats leaving the peninsula, many bound to Seattle or Hudson Bay markets on Vancouver Island; hides were listed as outgoing cargo from the docks at Dungeness Bay as late as 1902 (Schultz 1996).

The growing demand for elk teeth as a jewelry item during the late 19th and early 20th century also likely played one of the more significant roles in the demise of elk throughout the region. Elk produce an ivory-toned canine tooth on the upper jaw (the "eye" tooth) that was fashionable to wear as a watch-chain ornament in a gold setting. Ironically, this trend was first popularized by members of the Benevolent and Protective Order of Elks. There were reports during this era of tooth hunters slaughtering many elk only to retrieve the valuable canine teeth while leaving useable meat where it lay (Johnson 1923).

Early Protection

Overexploitation significantly diminished Roosevelt elk populations throughout the subspecies range by the early 1900s, leaving the most significant remnant populations in inaccessible parts of Washington's Olympic Peninsula and coastal northwestern California (Murie 1951, Harper et al. 1967, Harper 1985). In the early 20th century comprehensive conservation measures were put in place to stem the decline of the last great herds of Roosevelt elk in Washington (Table 4.1.1). In 1905, the newly created Washington Department of Fisheries and Game initiated a bounty system to stimulate hunting of wolves and cougars, while also placing a 10-year moratorium on all elk hunting. Wolves were ultimately extirpated from the peninsula by the early 1920s, while cougar populations persisted despite heavy persecution. The hunting moratorium ultimately lasted 28 years until hunting seasons were renewed in 1933 and from 1936-38, just prior to OLYM's establishment. Since the creation of the park, there has been no legal hunting within its boundaries.

Table 4.1.1. Chronology of selected conservation measures aimed at benefiting and protecting Roosevelt elk on the Olympic Peninsula and other milestone events, 1897-1938. Source: Skinner 1933, Schwartz 1939, Schultz 1996, Scheffer 1995.

Year	Events
1897	• Olympic Forest Reserve created • State fish commissioner appointed as first state game warden—no funding appropriated • First game laws enacted by counties • First county game wardens
1903	• Washington Department of Fisheries and Game created • First game licenses established and required • State set first bag limit of one male elk
1905	• State legislature established predator bounty • State moratorium established on all elk hunting
1909	• Mount Olympus National Monument established
1914	• State extended moratorium on elk hunting
1920	• Last verified wolf shot on the Olympic Peninsula
1933	• First elk hunt conducted on the peninsula in 28 years (157 elk shot) • Authority for managing Mt. Olympus National Monument transferred to National Park Service
1936-38	• Hunting seasons reopened after 3-year hiatus(1187 elk shot, including over 500 in the Hoh Valley)
1938	• Olympic National Park created

Elk populations responded quickly to protective measures, including the creation of Mt. Olympus National Monument and reduction of top predators in the early twentieth century (Figure 4.1.3). Morgenroth (1991) reported: "*Since the protection of these animals they have increased very well, some even coming back to their old stamping grounds on the north and east sides. A fair estimate of these animals, now in the Olympics would be about three thousand and by 1915, no doubt, they will be very plentiful again.*" Indeed, by 1917, elk population estimates for the peninsula averaged approximately 7,000. Cecil (2011) reported that elk herds increased by an annual rate of about 20% per year from 1908-11.

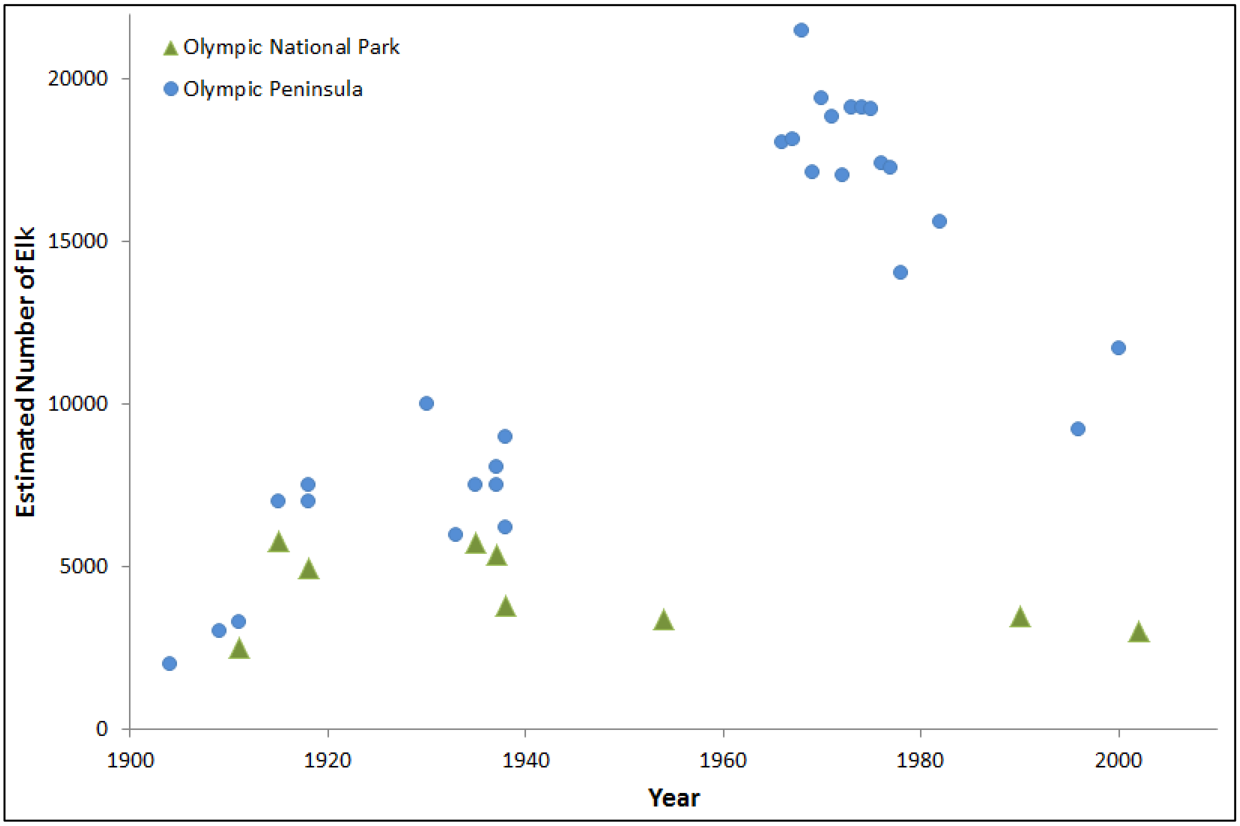

Figure 4.1.3. Estimated number of elk on the Olympic Peninsula (1905-2000) and on key winter ranges in Olympic National Park. Compilations for OLYM combine estimates for the primary winter ranges that lie mostly within the current park boundary including the Elwha, Sol Duc, Calawah, Bogachiel, Hoh, Queets and Quinault Valleys. Peninsula-wide estimates for the period (1966-2000) were derived by combining estimates for non-park lands (WDFW annual reports summarized by Tabor and Raedeke (1980) and WDFW (2004) with interpolated park estimates. Data sources: Morgenroth (1909), Lovejoy (1911), Fromme (1915), Bailey (1918), Riley (1918), Kavanagh (1930), Skinner (1933b), (Murie 1935b), NPS (1937), Pautzke et al. (1939), Schwartz (1939), Newman (1954), Houston et al. (1990); we used the midpoint of range provided), and Jenkins and Manly (2008).

Following the hunting moratorium, the elk population appeared to rebound during the mid-1910s to mid-1930s (Figure 4.1.3). Population estimates for the peninsula ranged as widely as 6,000-10,000 elk during the early 1930s. Murie (1935b), who investigated elk for the NPS, suggested the middle ground of 7-8,000 may be "*nearly correct*", while recognizing the uncertainty of these early estimates. Elk populations were often considered to be overabundant, particularly in the west-side drainages, as a result of nearly three decades of protection and predator control (Murie 1935b, Schwartz 1939). Growing concern over range conditions in the western rainforest valleys led to the reopening of long-suspended hunting seasons in 1933, and from 1936-38. Interestingly, concern over burgeoning elk populations did little to quell the spirited war waged against cougars, despite reasoned pleas coming from several prominent biologists of the era, who reminded managers and the public that maintaining healthy cougar populations would be key to solving the elk browse "problem" (Murie 1935a, Sumner 1938).

Olympic National Park Era

Elk populations may have been reduced just before the park was created as a result of the renewed elk harvests (Schwartz 1939). Schwartz (1939) estimated the peninsula's elk population at 6,225. OLYM staff estimated the park's elk population as 4,500-4,600 during the mid-1940s, although there is considerable uncertainty surrounding those early estimates. The first comprehensive ground surveys of elk were conducted by the NPS in the 1950s (Newman 1953, 1954, 1958). Newman estimated there were approximately 2,650 elk in OLYM in 1953, 4,025 in 1954, and nearly 5,000 elk in 1958 following two favorable winters. It is unlikely that the population actually grew by approximately 35% in a single year from 1953-54, and far more likely that Newman revised his estimate as he spent more time in the field. Following four years of research, Newman concluded that the "*elk population in Olympic National Park is rather stable and it seems that some contributing factors are: a mature forest, rapid and regular seasonal growth of forage plants, even pressure from predators, and natural die-off.*" By all indications, the 1940s and 1950s were decades of relative stability in elk numbers. Houston cautiously estimated numbers as about 3,000-4,000 elk wintering in the park during the late 1980s based on extrapolations from early helicopter surveys. Jenkins and Manly (2008) estimated the park's overwintering population as about 3,000 (excluding migratory herds that leave the park) based on fecal pellet counts used to extrapolate elk densities estimated from helicopter surveys conducted previously (Houston et al. 1987) in a subset of the winter ranges. These crude estimates suggest a slight decline in elk numbers between the 1950s and 1980s, when the park began conducting limited aerial surveys of elk.

The picture outside the park was far different than inside during the second half of the 20th century. Elk populations outside the park grew following the rapid rise of industrial logging on the peninsula from the early 1950s through the late 1980s (Figure 4.1.3). Elk numbers appear to have reached a new high on the peninsula (park and non-park areas combined) of approximately 16-20,000 elk during the late 1960s and 1970s. We derived this estimate by combining WDFW estimates for non-park lands, which generally ranged from 12,000-16,000 elk during this era outside the park (Tabor and Raedeke 1980) with about 4,000 elk for park lands. The large increase in elk on non-park lands corresponded with peak harvests of old-growth forests outside the park, and enhanced herbaceous and woody forage production following logging (Tabor and Raedeke 1980). After the slowdown of timber harvests on the peninsula during the 1990s and 2000s, elk numbers declined to 9,000-12,000 elk, likely due in part to the advancing age of clearcuts and reduced forage production and dietary nutrients associated with regenerating second-growth forests (Jenkins and Starkey 1996, Cook et al. 2016). The emphasis on harvesting female elk during the 1980s may also have played a role in the decline of elk outside the park during that period (WDFW 2004).

Recent Trends in Relative Abundance of Elk on Selected Winter Ranges

Houston et al. (1987) estimated that densities of elk averaged about 11, 12, and 16 elk per km^2 in the Hoh, South Fork Hoh and Queets Valleys, respectively. These estimates were derived from surveys conducted when elk were seasonally concentrated on the valley floors during early spring green up, and must be considered annual maxima. Raw counts of elk observed during aerial surveys, however, declined in the Hoh, South Fork Hoh, and Queets Valleys from 1985 to 2010 (Figure 4.1.4). Counts diminished by over 60% during this 25-year interval in the South Fork Hoh from an average count of about 90 elk in the mid-1980s to about 34 elk in 2010. Counts diminished by about 18% (from about 196 to 160) in the main stem of the Hoh Valley and by about 22% (from about 250 to 195) in the Queets Valley during the same time interval. The annual rate of decline averaged 0.8%, 3.9%, and 1% in the Hoh, South Fork Hoh, and Queets, respectively (Jenkins et al. 2015). Although rates of decline were relatively small in the Hoh and Queets Valley, all declines were statistically significant at the 0.05 confidence level (Jenkins et al. 2015).

Any inference from these trend counts that herds have declined requires the assumption that aerial detection probabilities and distributions of elk inside and out of the trend-count areas remained constant during the period of comparison. We attempted to control for the effects of late winter weather and phenological conditions on the distribution of elk within the trend-count areas using statistical modeling methods, but we cannot be sure that variations in survey conditions did not contribute to some of the apparent decline. Vegetation surveys suggest that there has been an increase in the cover of deciduous browse species on the South Fork Hoh winter range and an increase in the number of young western hemlock and Sitka spruce trees coincident with the apparent population decline (Chapter 4.1.2). We believe that changes in cover of deciduous shrub species or increases in the number of tree seedlings and saplings did not obscure elk appreciably during winter. Declines in aerial survey counts of elk shown from aerial survey results corroborated the slow declines apparent from parkwide estimates presented earlier (Figure 4.1.3).

Explanations for why counts and presumably populations of elk declined at a greater rate in the South Fork Hoh than the adjacent drainages remain speculative. We hypothesize that the greater population decline in the South Fork may relate to its proximity to the park boundary (Figure 4.1.2). The survey area in the South Fork Hoh abuts the park boundary, whereas there is a buffer between the park boundary and the survey zones in the Hoh and Queets (Figure 4.1.2). Previous studies suggest that movements of individual elk herds in the western Olympic valleys span approximately 5 km of river valley throughout the year (Jenkins and Starkey 1984), so elk likely range both within and outside the park up to 5 km from the boundary. We speculate that clearcut logging practices near the park boundary in the South Fork Hoh, which benefited elk populations broadly throughout the peninsula from the 1960s-80s, may have elevated the numbers of elk using the South Fork Hoh near the park boundary during the 1980s. Forest succession in the absence of new cutting since the mid-1990s likely diminished range capacity for these boundary herds. Proximity of elk to the park’s boundary may also increase susceptibility to both legal and illegal harvesting.

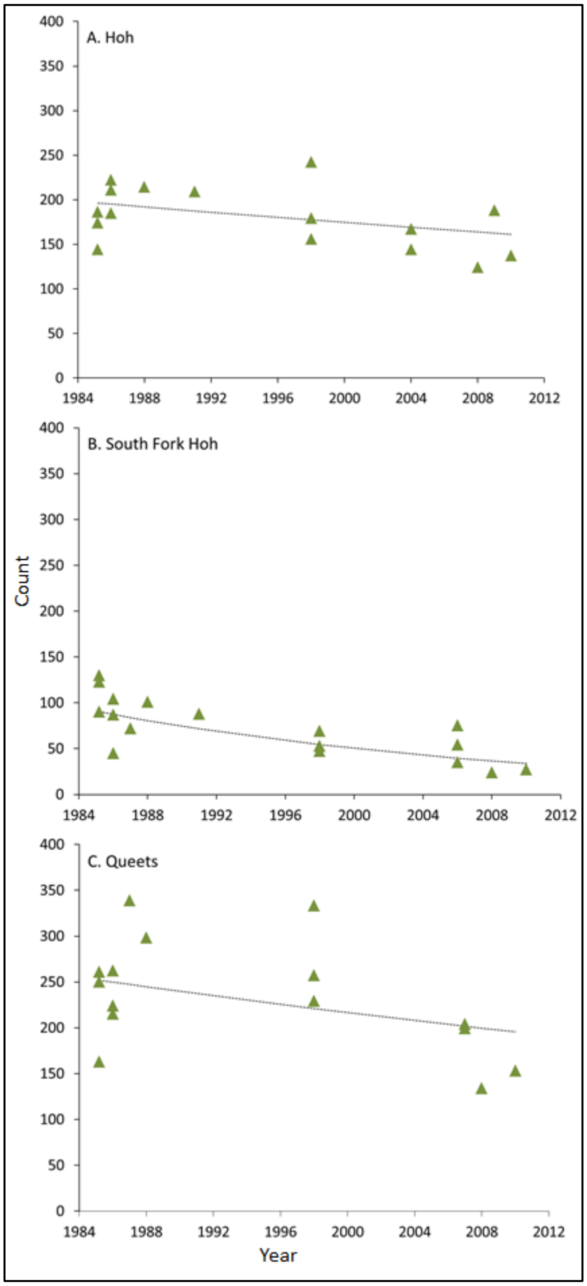

Figure 4.1.4. Trends in the number of elk counted during early-spring surveys in the (A) Hoh, (B) South Fork Hoh, and (C) Queets Valleys, 1985-2010. Source: NPS.

We also speculate that changes in predator densities mediated by changes in hunting seasons and methods outside the park could have affected boundary elk herds disproportionately. Although we have no information on cougar and black bear densities, black bears were managed as a forest pest from the 1950s through the 1970s (no harvest restrictions on the peninsula plus special control hunts; Tirhi 1996), which may have suppressed predation on elk calves near the boundary. Further, bounty payments for cougars were eliminated in 1961 and legal hunting methods for black bears and cougars were restricted in 1997, which also may have affected predator densities and predation pressures locally. Predator trends, however, and their influence on park elk populations are unknown.

Relative Abundance among Winter Ranges

Surveys of pellet group densities corroborated results from previous aerial surveys that densities of elk tended to be highest in the Queets Valley. Overlapping standard errors of the estimated pellet densities, however, indicated that differences in pellet group densities among the Queets, Hoh, Quinault and Elwha Valleys were not statistically significant. Pellet group densities tended to be lower in the Sol Duc and Bogachiel Valleys than in the Queets, Hoh, Quinault, and Elwha Valleys. Pellet group densities in the Dosewallips, Duckabush, and Skokomish Valleys were intermediate compared to the other ranges examined (although winter ranges within the park were very limited in these east-side drainages; Table 4.1.2).

Table 4.1.2. Mean densities of elk (elk/km^2) during early spring green-up and mean densities of elk pellet groups (pellet groups/60 m^2, with standard errors[SE]) at the end of winter measured in individual drainages of Olympic National Park from 1984-86 (Houston et al. 1987), and 2001-2002 (Jenkins and Manly 2008), respectively.

	Elk (Houston et al. 1987)		**Pellet Groups (Jenkins and Manly 2008)**		
Drainages	**Area Surveyed (km^2)[1]**	**Mean Density (elk/km^2)**	**Area Surveyed (km^2)[2]**	**Mean Density (groups/60 m^2)**	**SE**
Elwha	–	–	64.6	1.6	0.4
Sol Duc	–	–	52.3	0.3	0.2
Bogachiel	–	–	118.4	0.4	0.6
Hoh	23.4	10.8	93.9	1.7	0.6
South Fork Hoh	9.5	11.7	–	–	–
Queets	19.5	16.0	144.3	2.4	0.7
Quinault	–	–	136.5	2.0	0.4
Dosewallips/ Duckabush/ Skokomish	–	–	23.8	1.2	0.4

[1] Area corresponds to valley floor within a prescribed survey unit (Houston et al. 1987).

[2] Area corresponds to entire area <600 m (2000 ft) elevation within the drainage.

Broad patterns of variation in elk density among the primary regions of the park reflected patterns similar to those observed by Schwartz (1939) and Newman (1954) in past decades, all substantiating that the large majority of the park's elk herds occupied the west-side rainforest valleys. Although densities of elk have tended to be highest in the Queets Valley since the 1980s, ground surveys conducted by Schwartz (1939) and Newman (1954) suggested that elk may have reached peak densities decades ago in the Hoh Valley. Surveys of browsing intensity conducted in 1980 in the Queets, Quinault, and Bogachiel Valleys, however, corroborated the same differences among these watersheds as those derived from recent pellet group surveys: browsing pressure was greatest in the

Queets Valley and lowest in the Bogachiel Valley (Jenkins 1981). Jenkins (1981) reasoned that browsing pressure likely reflected differences in elk density, perhaps as mediated by the proportion of floodplain habitats found in the various valleys and proximity to the most extensive and productive subalpine summer ranges.

Sex and Age Composition

Elk population ratios measured from 1976 to the present generally ranged from about 20-40 young of the year (calves), 4-20 yearling males (spikes), and 30-60+ total males per 100 adult females (cows) (Figure 4.1.5). Variation in calf ratios could reflect unmeasured changes in calf production or early survival of calves from parturition to late summer, particularly during the 1980s when calf counts were lowest. However, we saw no sustained or long-term trend in the proportion of calves counted on key summer ranges of the park, suggesting no discernible changes in either productivity or early survival over recent decades. Calf ratios among migratory herds measured in OLYM were similar to but slightly less overall than what was measured for managed elk herds during a comparable time span on managed lands outside the park (41 calves:100 adult females; WDFW 2004). Our information on calf production over time was incomplete for low-elevation resident herds, but previous studies suggested that calf ratios may be higher among the migratory herds compared to low-elevation resident herds (Jenkins 1981).

With the exception of yearling males, ratios of males to adult females appear to have increased since the 1970s (Figure 4.1.5). This apparent trend, however, may reflect variation in survey methods used over the years rather than a true change in sex ratio. Recent aerial surveys conducted from 2008-2015 were flown at considerably greater search intensities than surveys conducted in the mid-1980s. We suspect that the more rapid aerial surveys and ground surveys would not have as detected many lone or small groups of males in denser vegetation or at lower elevations and not closely associated with the larger and more easily detected calf and cow groups. Because we see no compelling explanation for an increase in the ratios of males in the population, we assert that the modal counts of about 55-60 males:100 females seen in recent years likely represent the most accurate estimates of sex ratio of elk in the core of OLYM. The outlier observation of over 80 males:100 females was associated with a very hot and dry summer when the majority of calf:cow bands appeared to have been at lower elevations beyond the survey boundaries (NPS, unpublished data).

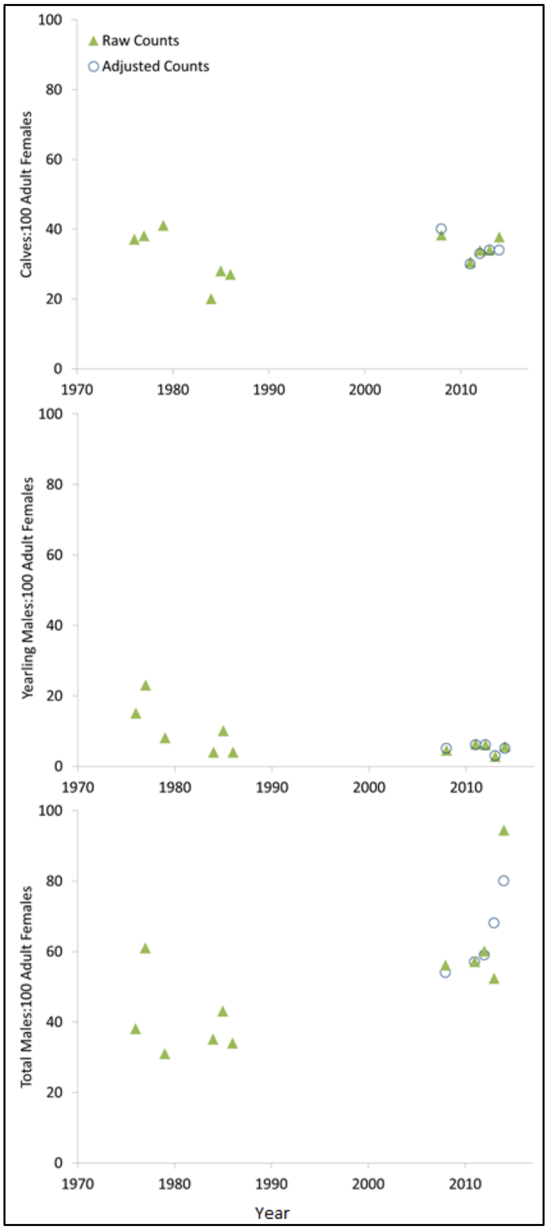

Figure 4.1.5. Ratios of calves, yearling males, and total males (all age classes of males combined) per 100 adult female elk on key summer ranges of Olympic National Park, August-September 1976-2014. Ratios corresponding to data collected from 2008-14 were computed from raw counts as well as from counts adjusted for detection biases based on double-observer sightability models (Griffin et al. 2013), demonstrating that there was generally minimal detection bias associated with ratio estimation. Source: NPS.

For comparison, ratios of spike bulls and total males on lands outside the park (open to hunting) averaged approximately 15 spikes and 24 total males:100 adult females from 1976-2000 (WDFW 2004). The ratio of spike bulls in populations outside the park may be comparable to what was seen in the 1970s inside the park, but appears higher than what has been recorded more recently (Figure 4.1.5). It is difficult to interpret the potential decrease in the ratio of yearling males seen in the park recently, but the change suggests that survival of males (if not both sexes) over their first winter may have declined inside the park during recent years, based on the assumption that calf survival is generally more variable than that of adult females (Gaillard et al. 1998).

Bull to cow ratios measured inside OLYM were appreciably higher than those measured outside the park during comparable years (24:100; WDFW 2004), no doubt reflecting the selective harvesting of males outside the park. Indeed, the ratios of males to females inside the park, averaging about 60:100, is among the highest recorded anywhere, including many other non-hunted populations of Rocky Mountain elk, such as in Yellowstone National Park (average 1971-1979=37 males:100 adult females; Houston 1982) or in the Canadian Rocky Mountain parks during a culling in the 1960s (37 males:100 females; Flook 1970). The high ratio of bulls observed in OLYM indicate that social mating systems and genetic integrity of elk have not been affected by hunting on adjoining lands, at least among the core migratory herds monitored.

Uses and Limitations

The existing data provide useful glimpses into the population ecology of Roosevelt elk in the relative absence of sport hunting and anthropogenic disturbances. We do not know the causes of the very gradual decline in the numbers of elk counted in the primary OLYM winter ranges. The gradual decline of counts within two of the low-elevation trend-count areas supports the conclusion expressed by biologists at OLYM for several decades that nutrient constraints and predation by cougars and black bears appear to limit populations in the absence of wolves and active management (Newman 1958). The more substantial decline in elk counts in the South Fork Hoh focuses attention on the potential effects of adjoining land use or predator management practices outside the park, a situation that warrants continued attention. Moreover, our interpretations of recent population trends of elk have also provided useful insights for interpreting potential impacts of elk on important ecosystem characteristics and processes (see Chapters 4.1.2 and 4.1.3).

The high bull:cow ratios observed within OLYM provide assurance that naturally evolved mating systems have been preserved effectively for several decades, at least among migratory herds within the park's interior. Although calf ratios of migratory herds may be lower than those on adjacent managed forest lands where they are hunted, there are no indications of nutritional deficiencies manifested in especially low productivity among the migratory herds. Limited and incomplete sampling of sex and age composition among low-elevation resident elk prevented us from examining calf ratios in low-elevation resident herds, although existing evidence indicates that productivity and nutrient status is lower among the low-elevation resident herds than migratory herds (Jenkins 1981, Leslie et al. 1984, Houston et al. 1990). This is to be expected among resident elk herds that rely on relatively low forage qualities characteristic of forested ecosystems (Leslie et al. 1984, Cook et al. 2016).

Inference on the current status of Roosevelt elk populations has been restricted by the limited scope of current and past elk monitoring programs. For many years elk monitoring has been constrained to narrow strips of optimum habitat during time windows when elk congregate in these habitats seasonally (i.e., floodplain habitats during late spring, and subalpine habitats during late summer). Although we have attempted to control for the effects of weather variables that influence the distribution of elk within the survey areas, we cannot rule out the possibility that results have been influenced by unmeasured effects of weather variability and plant phenology on elk distribution. Further, the limited spatial extent of sampling does not permit a full appraisal of factors that may independently affect migratory and resident components of the population or individual segments of the population.

Emerging Issues

Wolf Restoration

Although there is no current proposal to reintroduce wolves in OLYM, the idea of returning wolves, which was first voiced soon after wolves were eradicated (Murie 1935a), has continued to generate interest, discussion and controversy on the Olympic Peninsula (Dratch et al. 1975, Allen et al. 1981, Ratti et al. 2004). The feasibility of restoring wolves was first studied by a student group from Evergreen State College during the 1970s (Dratch et al. 1975). The group concluded that the Olympic Peninsula might support approximately 40-60 wolves. In 1997, a more formal and complete study concluded that establishment of a marginally viable population of wolves (estimated at about 56 wolves) was biologically feasible (Ratti et al. 2004). However, rancorous public discourse and political intervention soon ended any further discussion of potential wolf reintroduction.

By the mid-to-late 2000s, however, wolves had naturally recolonized parts of Washington from Canada and Idaho. At this writing there are 20 confirmed wolf packs in eastern (primarily northeastern) Washington and in the North Cascades Range (http://wdfw.wa.gov/conservation/gray_wolf/, accessed 6 September 2017). It is not known how readily dispersing wolves might cross barriers presented by major interstate freeways and urban corridors to reach the Olympic Peninsula—although some State wolf managers do not rule out that possibility. Simulation exercises suggest that if wolves were restored to the park, the added predation from wolves would likely reduce the park's elk herds by 5-35% (Fieberg and Jenkins 2005). This range of uncertainty reflects many poorly understood components of this predator/prey system, including the rate at which wolves kill elk, the extent to which different sources of mortality compensate one another (i.e., wolves and cougar predation, undernourishment), and the effect of lowered elk density on demographic traits of the population (Fieberg and Jenkins 2005).

Currently, the possibility of wolves influencing park elk populations as a result of either active reintroduction or natural recolonization seems quite low. Over the longer term, however, the return of wolves to OLYM is plausible. As recent research from Yellowstone National Park and elsewhere has informed, returning an apex predator such as the wolf could affect numbers and behavior of elk (White and Garrott 2005, Gower et al. 2009), with potential direct and indirect effects on other carnivore populations (Kortello et al. 2007), and trophic levels (Berger et al. 2001, Painter et al. 2015).

Disease

Digital dermatitis, more simply known as hoof disease, is a bacterial (treponeme) infection that causes severely deformed (even missing) hooves in ungulates (Han and Mansfield 2014). Although this disease has been prevalent in the livestock industry for some decades, it had not been reported in wildlife until it was documented in elk in western Washington. In the late 1990s, limping elk were reported in Cowlitz county (southeast of Puget Sound), and then throughout a broad swath of southwestern Washington (Figure 4.1.6). Observations of limping and dead elk with hoof deformities have been reported as far north as the Skagit River on the eastern Sound and the Dosewallips River adjacent to OLYM along the Hood Canal and in the Quinault Valley (Figure 4.1.6). Demographic consequences of hoof disease on reproduction and survival of elk are not known, and currently are the subject of intensive research by the WDFW. Basic principles, however, suggest that any ailment that affects locomotion in elk will have implications for foraging efficiency and predator evasion. As there is no practical cure for treating this disease in free-ranging wildlife, the emergence of hoof disease in such proximity to OLYM poses an immediate concern.

Changing Climate

A warming climate is expected to reduce snowpack and promote earlier melt out in the Pacific Northwest (Mote and Salathé 2010), with potential effects on forage phenology, timing of seasonal migrations, and demographic traits of elk. Migratory ungulates generally time their seasonal uphill movements to correspond with an advancing wave of green forage production that follows behind the snow's disappearance (Sawyer and Kauffman 2011, Middleton et al. 2013). Snowmelt at high elevations during summer influences the availability of nutritious forage during the time of the year when energy demands of ungulates are elevated in support of lactation and growth. Summer drought has been linked to decreased recruitment of young elk (Griffin et al. 2011, Middleton et al. 2013) and moose (Monteith et al. 2015), as well as to reduced survival or growth of mountain sheep and goats (Pettorelli et al. 2007). Although direct responses of ungulate distribution and demography to climate variations have not been studied in the Olympic Mountains, elk density observed on subalpine meadows was very low during a recent drought, suggesting unfavorable nutritional conditions in the subalpine zone (Happe et al. 2016). We speculate that poor forage conditions associated with hot summer weather and early snow melt on subalpine summer ranges could affect body condition of elk during late summer, with potential effects on reproduction or overwinter survival in the future.

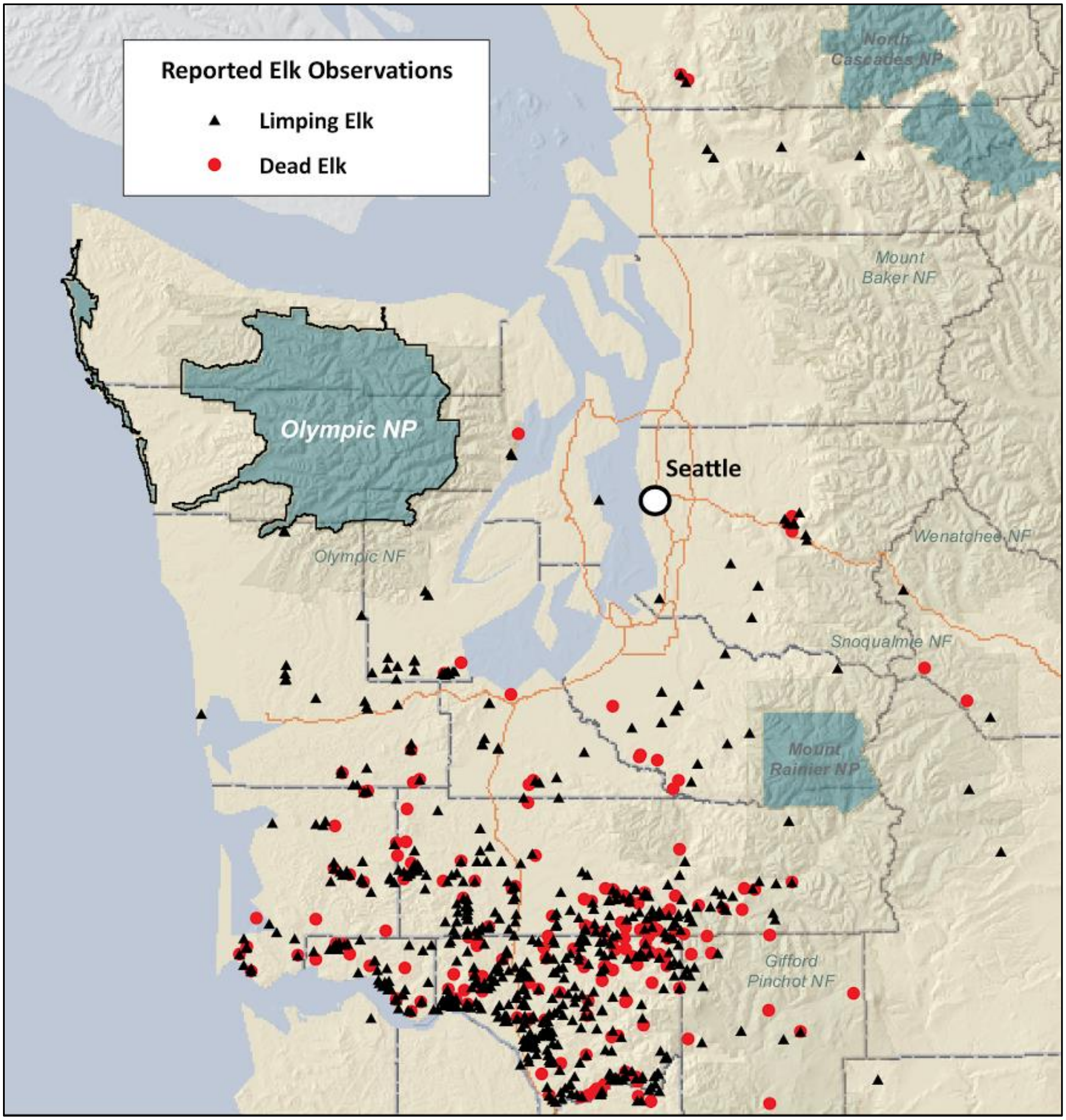

Figure 4.1.6. Publically reported (i.e., unconfirmed) cases of limping elk (black triangles) and dead elk with hoof deformities (red circles) indicating the approximate recent distribution of hoof disease in Washington, 2012-2015. Source: Washington Department of Fish and Wildlife (http://wdfw.wa.gov/conservation/health/hoof_disease/, accessed 18 Aug 2016). Data published with permission of WDFW.

Information Needs/Gaps

Population monitoring

Elk population monitoring has been limited by funding, logistical, and operational constraints to relatively small areas of the park and segments of the elk population. Changing climate or predation pressures in the future could influence distribution patterns of elk, which could mask or obscure changes in population status if monitoring does not cover a sufficient spatial extent. Development of a more comprehensive spatially extensive monitoring program would provide more robust inference on elk population trends in the face of changing climate, predation, or disease issues. Increasing the temporal replication of trend-count surveys within a season would help to reduce the effects of anomalous weather on annual surveys, but would not shield against erroneous inferences that could result from long-term changing climate or predation effects on elk distributions.

There is much to learn about variations in the status and trends of elk herds in relation to the park boundary (including adjacent land use, predator management, and hunting seasons), and life history patterns of elk (i.e., migratory versus resident status). Little is known of trends in migratory elk herds that leave the park during winter on the east and south sides of the park. These segments of the population may be the most susceptible to changing land-use or predator management practices outside the park. Further, long-term monitoring has not sufficiently covered elk affected by the removal of two hydroelectric dams in the Elwha Valley and subsequent restoration of >300 ha of floodplain winter range. Opportunities may exist for developing or enhancing cooperative elk monitoring programs involving State or Tribal partners to assure persistence of migratory herds leaving the park during winter.

Nutritional status

Improved information on seasonal and annual variation in nutritional condition of individual elk, reproduction and survival of females, calf production, and cause-specific rates of mortality would greatly improve understanding of natural regulation of the OLYM elk population and mechanisms of elk population change over time, particularly as related to changing climate, disease, or predator influences.

Distribution and movements

Studies from Rocky Mountains highlight the value of collecting fine-scale data on the movements of individually collared elk to understand the interplay between elk behavior and distribution and changing predator communities, climate, or plant phenology (Mao et al. 2005, Middleton et al. 2013). From 2008-2015, we radio-tracked a sample of migratory and resident elk throughout the park, as part of an effort to measure aerial survey detection biases (Jenkins et al. 2015). While analysis of elk movement patterns was outside the scope of this assessment, these data will provide a useful baseline for evaluating long-term effects of changing climate or predator assemblages in the future.

Predation effects

Understanding the current role of predation by cougars and black bears on elk would be useful for assessing natural regulation and population trends of elk in relation to nutritional limitations. Information on the current abundance, distribution, and predation patterns of cougars and black bears, as well as cause-specific mortality rates of elk would provide critically important baseline information needed to assess the effects of potential wolf restoration on elk populations in the future.

Potential disease impacts

Elk in OLYM face an imminent threat from non-native treponeme bacteria that causes hoof disease. Enhanced monitoring of population abundance and demographic processes would help to evaluate and manage potential disease effects on park elk. Establishing a baseline of current disease and parasite incidence would be useful for detecting and interpreting changes in the future health of park elk.

Resource Summary

We rated condition of elk population as warranting moderate concern, declining, with low confidence (Table 4.1.3). The population measured within three reference winter range areas appears to be within the range of acceptable standards we established previously (Section 4.1.1)—that is, lower than when there was widespread concern about overpopulation during the 1920s and 1930s, but not so low as to create concern about long-term viability of the population. We have low confidence in this assessment because the park has monitored trend in only a small fraction of the population, and the potential influences of climate effects on elk distribution and population counts are poorly understood. Also, we have inadequate understanding of whether these local declines are related to natural population variations, in which case the trend would be considered acceptable, or due to unwanted anthropogenic causes, which would signal deteriorating conditions. The composition of bulls and cows in the population is in good condition, apparently stable, and of no concern.

Table 4.1.3. Resource condition summary for Roosevelt elk in Olympic National Park.

Resource Measures	**Areas**	**Condition Status and Trend**	**Rationale**
Relative Abundance	Queets	🟡	Counts of elk in the Queets Valley declined approximately 22% from 1985-2010.
	Hoh	🟡	Counts of elk in the Hoh Valley declined approximately 18% from 1985-2010.
	South Fork Hoh	🟡	Counts of elk in the South Fork Hoh Valley declined approximately 60% from 1985-2010
Population Composition	Pooled	🟢	Bull:cow ratios are among highest ever reported.
Overall condition and trend for Roosevelt Elk		🟡⇩	The resource condition is of low to moderate concern because elk counts have declined on three trend-count areas since 1985. Confidence is low based on the limited survey coverage of the park's elk herds and uncertain baselines of the natural (i.e., pre-settlement) condition.

4.1.2. Riparian Vegetation

Andrea Woodward, Kurt Jenkins, Ben Gustafson, Josh Laterell, and Mark Harmon

Introduction

The western low-elevation forests of the Olympic Peninsula found in OLYM provide some of the finest examples of temperate, old-growth rainforest in the United States. Collectively, these forests of massive trees, covered with epiphytes and occupying the valley bottoms of major scenic rivers, comprise some of the park's most popular destinations and serve as key habitat for Roosevelt elk. Besides their aesthetic value, these forests serve many essential ecological functions. In particular, vegetation found in close proximity to the active channel at the interface of terrestrial and riverine ecosystems (hereafter, riparian vegetation) helps to stabilize river shorelines, while also contributing large wood (i.e., logs) to rivers. This wood, in turn, affects sediment deposition, channel evolution (Hupp and Osterkamp 1996), and in-stream habitat characteristics for fish and other aquatic organisms (Montgomery et al. 1996). Riparian vegetation also delivers leaves and other fine organic materials into the water, affecting nutrient cycling and aquatic productivity. Additionally, riparian vegetation provides shade that moderates soil and water temperature (Chen et al. 1999), while adding structural diversity along the river's bank to enhance streamside habitats for many terrestrial wildlife species. In low gradient west-side Olympic Peninsula river valleys, riparian functions are provided by an array of vegetation patches of different ages that are contacted, eroded, and created by the river as it scrolls across the valley bottom.

Features of Valley Floor Environments in Olympic National Park

The geomorphic structure and substrate for plant community development of the primary river valleys on the west side of OLYM is founded on terraces of till deposited by retreating Pleistocene glaciers or subsequent Neoglacial outwash (Crandall 1965, O'Connor et al. 2003). Over time since glaciation, the river has continually eroded older surfaces and deposited new ones, which coupled with channel incision has created new terraces. While the morphology of the valley bottom may change with the gradual loss of older glacial terraces, the parent material is sufficiently similar on newer surfaces that the same general pattern of vegetation succession ensues (Fonda 1974, Van Pelt et al. 2006).

Riparian and floodplain vegetation in the Queets Valley, Olympic National Park. Photograph courtesy of NPS/Phil Kennedy.

Vegetation patch types representing stages of succession on floodplains and terraces of the Olympic river systems have been described variably, but they all depict a progression of plant community development. For our purposes, we adopted the series of fluvial (produced by action of the river) patch types described by Latterell et al. (2006; Figure 4.1.7) and elaborated by Van Pelt et al. (2006) in the Queets Valley. These five patch types reflect gradients in flood frequency, substrate age, and soil development including areas affected by frequent flooding (pioneer bars, developing and established floodplains) to higher areas away from the channel where competition and gap formation drive community dynamics (transitional and mature fluvial terraces; Latterell et al. 2006). Pioneer bars represent the youngest surfaces still scoured frequently by high waters, and are dominated by cobbles, gravels and silts, as well as newly established red alder (*Alnus rubra*) and willow (*Salix sitchensis* and *S. scouleriana*). Developing floodplains, which may be flooded during most years, are dominated by 5-20-year old willow and red alder stands. Established floodplains, where floods are less frequent, are dominated by an overstory of mature red alder and regenerating Sitka spruce. They may also contain dense understories of salmonberry (*Rubus spectabilis*) and vine maple (*Acer circinatum*), and occasionally regenerating bigleaf maple and black cottonwood. Transitional fluvial terraces are the youngest surfaces generally free from flooding, and are dominated by Sitka spruce, with bigleaf maple, black cottonwood, and vine maple lower in the canopy (Fonda 1974, Balian and Naiman 2005, Latterell et al. 2006). Mature fluvial terraces develop into multilayered and multispecies old-growth stands dominated largely by Sitka spruce and western hemlock in the overstory and huckleberry (*Vaccinium alaskense* and *V. parvifolium*) and other shrub species in the understory. Together, these diverse vegetation patch types all contribute to the riparian function of these systems.

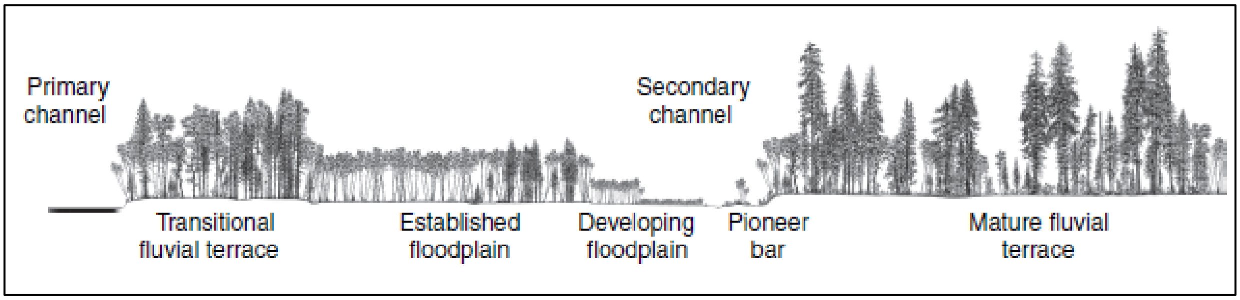

Figure 4.1.7. Profile drawing of patch types in an actual (half) cross-section of the Queets River valley floor. Large wood is not depicted, but may be present above and below ground where it can play a fundamental role in shaping landform evolution. Pioneer bars and developing floodplains commonly occur near margins of the primary channel, as well. Original artwork by R. Van Pelt. From Latterell et al. (2006).

At the spatial scale of individual vegetation patches, plant succession may proceed directly through all stages or it may be interrupted randomly at any stage by a disturbance. Consequently, at small spatial scales individual patches rarely reach an equilibrium composition (Naiman et al. 2010). However at the reach scale, the proportion of patch types is often relatively constant for over hundreds of years (Beechie et al. 2006). Key to the maintenance of this “shifting-mosaic steady state” (Bormann and Likens 1979, Latterell et al. 2006) of forest patch types is the initiation of bars and islands through the provision of logs to channels (Fetherston et al. 1995, Abbe and Montgomery 1996).

The provision of logs is arguably the most important role of riparian vegetation in shaping channel morphology in the Pacific Northwest. Older Pacific coastal floodplain forests are among the most productive in the world (McKee et al. 1982, Means et al. 1996, Balian and Naiman 2005, Naiman et al. 2009) and support some of the fastest growing and largest-known trees in the world (Van Pelt 2001). When these large trees are undercut by floods and fall into the river, they have a significant effect on dynamics of river flow by influencing channel shape and pool frequency (Montgomery et al. 1996, O'Connor et al. 2003, Beechie et al. 2006), while also protecting and stabilizing sites for colonizing vegetation (Abbe and Montgomery 1996, Naiman et al. 1998). Log jams are initiated by a “key member”, the key structural element of log jams, which has been physically defined by some to be a log that is at least 50 cm in diameter and 5 m long (Bisson et al. 1987, Sedell et al. 1988, Robison and Beschta 1990). In Olympic rivers, key members greater than 100 cm diameter are typical (Latterell et al. 2006, Van Pelt et al. 2006).

Logjam on the Elwha River. Photograph courtesy of the USGS/Kurt Jenkins.

The presence of a large root mass (known as a root wad) rather than a strict size threshold may be the most important factor in stabilizing key members (Abbe and Montgomery 1996). Key members in the Queets River were observed to be primarily Sitka spruce or other conifers (Hyatt and Naiman 2001, Latterell 2005). Cottonwood trees grow quickly and can achieve key-member diameters in 50-100 years and can potentially quickly restore large wood to rivers that have been logged (Collins and Montgomery 2002). However, deciduous species are depleted faster from channels than conifer species (Harmon et al. 1986, Hyatt and Naiman 2001, Latterell et al. 2006) and provide less durable ecological effects than conifers. In addition to key members, log jams consist of racked members and loose debris, which collect on the key members and may include deciduous species and smaller conifers (Collins and Montgomery 2002). Log jams cause complex flow patterns known as “island-braiding” where the river channel flows around islands formed behind log jams rather than in simple meanders typical of many low gradient river sections (Naiman et al. 2010). The successional sequence that results from vegetation and landform evolution, erosion-resistance conferred by persistent log jams, and a highly productive ecosystem with long-lived conifers sustains the development of potential sources of logs to maintain floodplain structure.

Acting on the mosaic of vegetation and landforms, elk have long been recognized as a primary architect of forest vegetation structure and composition in the Olympic rainforest valleys through the effects of their browsing and trampling. Early biologists commented frequently on the “over-browsed” appearances of many of the key browse species of Roosevelt elk (Schwartz and Mitchell

1945, Newman 1953, 1954). Many biologists have suggested that such browsing may affect the recruitment (i.e., the establishment of young saplings) and size of the more palatable species of trees and shrubs, ultimately maintaining a more open forest understory and favoring the development of a grass-dominated herbaceous layer (Happe 1993, Schreiner et al. 1996). Moreover, selective browsing of the more palatable tree species (e.g., western hemlock, bigleaf maple, and cottonwood over Sitka spruce and red alder; Jenkins 1981, Harmon and Franklin 1983), may affect the species composition of floodplain forests.

Assessment Goals and Objectives

We evaluated riparian vegetation by assessing the condition of valley floor vegetation communities in OLYM's west-side rivers, recognizing that fluvial terraces and floodplain communities all contribute to riparian ecological functions. In particular, we focus on assessing whether grazing, browsing, and trampling by native elk, and to a lesser extent deer, has affected the abundance of browse species and the provision of large logs to rivers. We reviewed historical written records, aerial photographs, and previously published and unpublished field data to determine historical conditions of fluvial terrace and floodplain plant communities in the park. We also investigated long-term trends in vegetation structure, composition, and landscapes that may affect long-term riverine processes. We acknowledge there are other important components of valley bottom vegetation condition, such as the prevalence of exotic plants, which is evaluated in Chapter 4.4.3.

Approach and Methods

To assess direct effects of elk on browse species, we examined temporal changes in (1) percent vegetation cover of key elk browse species, and (2) recruitment of selected species of overstory trees in established floodplains and fluvial terraces. To assess potential changes in provision of logs to rivers, and ultimately the patch dynamics and composition of fluvial terraces and floodplains, we also examined (3) the percent of conifer vegetation present along the active river channel that may become key member in debris jams, and (4) valley floor patch composition along the major west-side river systems. Because information for each of these measures was limited, each measure was evaluated opportunistically in different river valleys and reaches commensurate with data quality and quantity.

Vegetative Cover of Key Elk Browse Species

Effects of differential feeding among plant species by elk and deer on forest structure and composition have been investigated using exclosure studies. In these studies, researchers examine vegetation changes in plots where ungulates have been fenced out (Figure 4.1.8). Twenty-three ungulate exclosures were built on river terraces of west-side rivers in what is now OLYM between 1935 and 1958 (Woodward et al. 1994). Most exclosures were built on the Hoh, South Fork Hoh, Queets, and Quinault Rivers, although the Elwha and Bogachiel Rivers each contained one exclosure. These early exclosures were typically small (most less than 0.2 ha). Although these studies had many pitfalls by modern standards (including lack of randomization, lack of consistent sampling and monitoring methods, and potential fence line effects), these exclosures were revisited and sampled in the 1990s (Woodward et al. 1994), and effects of elk exclusion assessed retrospectively.

We summarized trends in early measurements of herbs, and described the size of shrub species and age distribution of vine maple inside and outside of these exclosures.

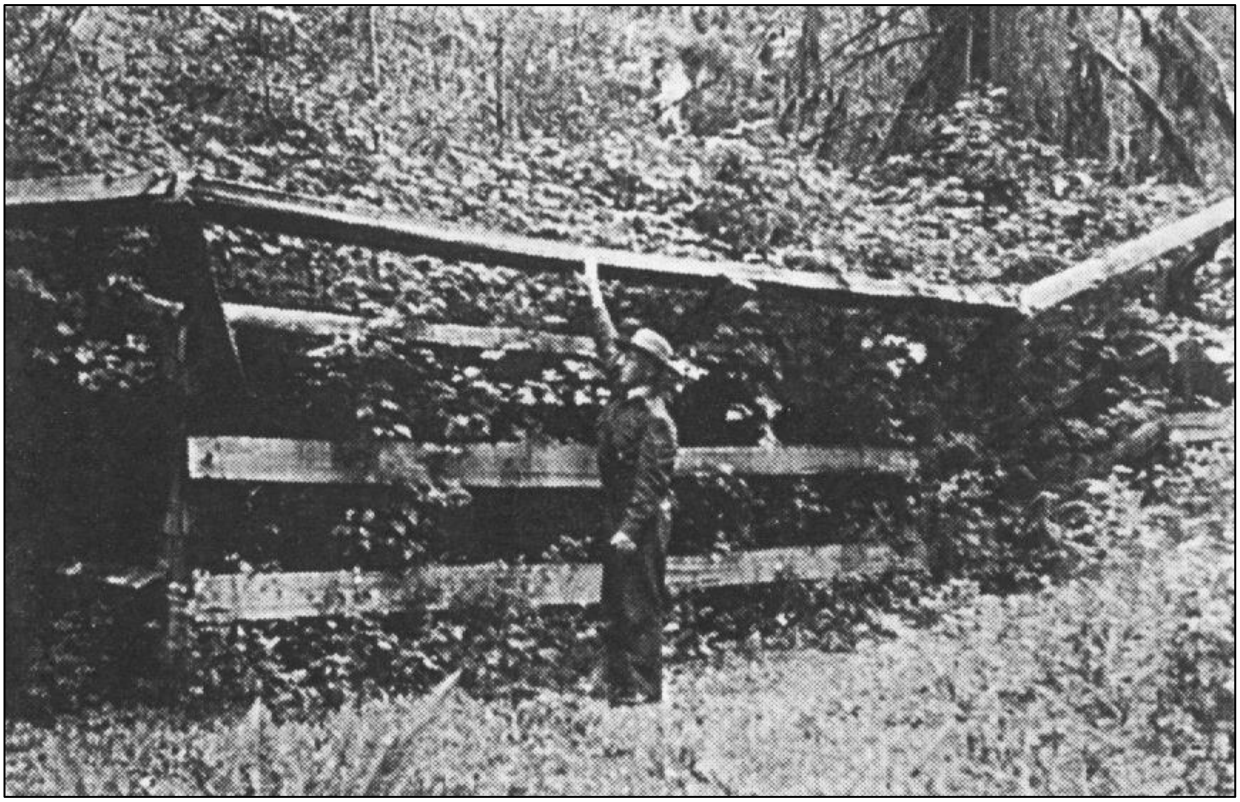

Figure 4.1.8. Elk exclosure built in 1935 in the Quinault Valley and photographed in 1943. Source: NPS.

Two larger (0.5 ha) ungulate exclosures were built in 1980 to represent an older and younger fluvial terrace of the South Fork Hoh River (Schreiner et al. 1996). These exclosures remained largely intact until at least 2005 and were sampled with a consistent protocol using four permanently marked 50-m transects established inside and outside of each exclosure. We and others measured herbaceous vegetation cover using 25 20x50-cm plots placed at alternate meter marks along each transect. We assessed shrub cover using the line intercept method (Grieg-Smith 1983); that is, we measured the percentage of each 50-m transect that intercepted the vertical projection of shrub foliar cover. Vegetation characteristics were measured annually from 1980 to 1984, then again in 1986, 1988, 1992, 1996, 2007 and 2016. Here, we report changes over three decades inside and outside these two exclosures.

Recruitment of Overstory Trees

We assessed the effect of elk foraging on tree seedling establishment using several data sets. First, we reviewed conclusions from Woodward et al. (1994) based on their previous examination of tree establishment inside versus outside of the early set of exclosures (i.e., those constructed in the 1930s and 1950s). Further, we summarized counts of tree seedlings by height class and species made periodically from 1980-2016 within 1-m belt transects established both inside and outside of the

South Fork Hoh exclosures (i.e., 0.5-ha exclosures built in 1980). We also examined previously published analyses of tree cores comparing timing of western hemlock establishment on mature fluvial terraces (Harmon and Franklin 1983), and of cottonwood and bigleaf maple (Beschta and Ripple 2008) establishment in established floodplains. Both of these last two data sets were collected to examine correlations between tree establishment dates and historical abundance of elk. Finally, we considered Houston's (1994) observations of cottonwood size and spatial distribution made in response to the concern that cottonwood was not establishing in developing floodplain forests during recent decades.

Conifer Vegetation along Active Channel

We interpreted historical aerial photographs and recent (2011-2013) Light Detection and Ranging (LiDAR) imagery (Puget Sound LiDAR Consortium 2011) to determine if there were changes over time in the percent of active channel length abutting conifers large enough to serve as key members of log jams. We used channel boundaries derived from 1939-2013 aerial photographs (see Chapter 4.1.3) to determine the extent of active channel contact with large conifers over time. Conifers are easily distinguished from deciduous trees in photos, but we could not identify species. Because Sitka spruce is the most common riparian conifer species, we assumed all individuals were Sitka spruce for simplicity. We used recent LiDAR images from 2011-2013 to determine forest canopy height during those years. Then we applied allometric equations that related tree height to diameter or age (Hegyi et al. 1979, Hanus et al. 1999) to identify trees that would have been large enough (>50 cm in diameter) to be key members at the time when previous images were taken. If trees in early photos were missing in 2011 due to changes in channel location, we estimated the height of the nearest similarly appearing trees. We considered trees whose crown radius (and therefore roots) were within 20 m of the active channel to be vulnerable to being undercut, which compares with a mean estimate of 13 m/yr mean annual channel migration rate for the Queets River determined over >60-yr intervals (O'Connor et al. 2003, Latterell et al. 2006). Van Pelt et al. (2006) considered Sitka spruce with a 25-m crown diameter to be "large" on the Olympic Peninsula. We considered trees less than 26 m apart to form a continuous line of potential key member delivery along the river bank at the time of each photo.

Valley Floor Patch Composition

Latterell et al. (2006) assessed changes in valley floor patch composition from 1939-2002, based on aerial photo interpretation of four study reaches in the Queets Valley. We extended this analysis to 2009 by obtaining aerial photographs from 2009 (U.S. Department of Agriculture 2009) and LiDAR imagery from 2011 (Puget Sound LiDAR Consortium 2011) and examining the same study reaches and sample points as Latterell et al. (2006). Sample points were originally established in a 100-m grid across 4 study reaches, for a total of 886 points. Each point was originally classified in each geo-referenced image available since 1939 by the following patch types: primary channel, secondary channel, pioneer bar, developing floodplain, established floodplain, transitional fluvial terrace and mature fluvial terrace as described earlier. We combined patch categories that are difficult to discern in photos (i.e., developing and established floodplains, transitional and mature terraces) for this analysis. Having LiDAR data dramatically improved our ability to classify patches compared with the original study.

Reference Conditions

To the extent possible, we inferred prehistoric forest composition and historical vegetation conditions at the time of early EuroAmerican settlement to establish benchmarks for comparison to the present vegetation. We obtained limited inferences on vegetation composition over millennia from studies of pollen grains from wind-pollinated plants preserved as accumulations in lake and bog sediments. Second, consideration of the entire record of observations made by early explorers and settlers (Lien 2001) provided a glimpse of how vegetation appeared at the time of settlement and how it changed in the early twentieth century. To evaluate changes in the availability of large trees along the active channel and changes in vegetation patch composition (Approach and Methods, measures 3-4), we used the historical range of variability determined from the early part of the 1939-2009 photo-sequence to qualitatively assess current conditions.

Pollen Studies

Hansen (1941) and Heusser (1964, 1974, 1978) sampled fossil pollen deposits in eight low elevation west-side river valley sites located in the Hoh, Bogachiel, and Humptulips valleys on the Olympic Peninsula in the 1940s, 1960s and 1970s to describe historic pollen abundance. These data put current vegetation in a long-term context and may indicate whether a change happened at the time of settlement. These studies indicated that western hemlock and alder have dominated pollen deposition since at least the mid-1800s. Sitka spruce was also consistently apparent but in smaller amounts than western hemlock, reflecting its relative abundance (Heusser 1969). Bigleaf maple appeared since the mid-1800s at 2 sites but constituted <2% of any pollen sample; black cottonwood appeared at only one site, also at <2% of any pollen sample.

Interpreting relative species abundance from pollen samples is complicated by the differences among species in pollen production, distance of dispersal, and rate of decay. When recent pollen deposits are compared with extant vegetation, the pollen spectrum of Sitka spruce has been shown to be in accord with actual distribution (Heusser 1969). Problematic genera include cottonwood (Populus genus) pollen, which has been under-represented relative to its abundance in vegetation while alder (Alnus genus) is over-represented (Leopold et al. 1964, Brubaker et al. 2005), possibly because cottonwood pollen does not preserve well or travel far from the source (Edwards and Dunwiddie 1985). In a study of modern pollen rain relative to vegetation composition, Heusser (1969) only found black cottonwood pollen at a Hoh River site where large black cottonwood trees (>30 m tall) were locally abundant. Bigleaf maple also does not seem to leave a strong pollen signature in river valleys. Florer (1972) observed that maple (Acer genus) pollen did not appear in pollen rain at one site where it was abundant in the living vegetation. Nevertheless, looking over millennia at core samples from west-side river valleys, black cottonwood and bigleaf maple do show up in small quantities at intervals (Heusser 1964, 1974, 1978), indicating that they have been a consistent component of the flora. Pollen studies provide relatively coarse descriptions of vegetation whose sensitivity to a disturbance such as early human settlement might be questioned. However, Heusser (1974) interpreted an increase in alder pollen around the turn of the twentieth century to be caused by clearing of forests by settlers and subsequent colonization by alder.

Written Accounts

The written records of early explorers and settlers (Lien 2001) provided a glimpse of how vegetation appeared at the time of settlement and how it changed in the early twentieth century. We also looked at the field notes from General Land Office surveys that occurred in the late 1800s and early 1900s. We found these vegetation descriptions to be spatially extensive and primarily descriptive of mature forests, possibly for the purpose of describing timber resources. While others have gleaned valuable vegetation composition information from these records (Collins and Montgomery 2002, O'Connor et al. 2003), we considered the search to be more time consuming than was feasible for this project, especially with a potentially small reward.

Aboriginal people may have occupied the Olympic Peninsula as early as 11,000 years before present (Schalk 1988). While a wet climate and acidic soils impede preservation of archeological artifacts (Wessen 1990), evidence from around 3,000 years before present indicates a semi-sedentary human population exploited marine and other natural resources (Croes and Hackenberger 1988, Schalk 1988, Wessen 1990). The decimation of these people during the late 19th century altered whatever effects they might have had on the peninsula, including effects on elk numbers and distribution as well as burning and selective harvest of forest vegetation. The extent of their influence is unknown.

The first EuroAmerican explorers and pioneers of the Olympic Peninsula and early government biologists left numerous qualitative accounts of riparian and floodplain vegetation conditions over time. Because these accounts provide the best available glimpse into the range of historical vegetation conditions, they are particularly important to our interpretation of major deviations between current and historical vegetation conditions as related to wolf extirpation and elk populations. We provide complete transcripts of some of the most descriptive accounts of elk effects on vegetation conditions in Appendix E. Below we summarize key points derived from these records relevant to range conditions and trends.

Early Exploration and EuroAmerican Settlement

The early explorers only occasionally described species presence, but their accounts often mentioned the presence of alder, cottonwood, willow, and bigleaf maple (Appendix E). Accounts of Banta (1889), Gilman (1890), and Morganroth (1991; referencing observations from 1889-1903) are particularly interesting because they described gravel bars and early transitional floodplains comprised of small cottonwoods—a notable difference from most gravel bars in the park today. It is not clear how cottonwood distribution may have varied longitudinally up valleys because two of the early descriptions appear to reference mainly the lower reaches of the rivers (Banta 1889, Gilman 1890). Morgenroth (1991) also described young alders, willows and cottonwood trees growing on gravel bars in the rain forest valleys, causing great piles of logs to be windrowed, referencing in general terms fluvial processes that are understood today as key factors influencing both channel and floodplain dynamics. The early historical record of understory vegetation conditions generally reflected the practical perspectives of the pioneer or explorer as related to ease of travel or land clearing, but they provide clues relevant to interpreting the effects of elk. Accounts range from mention of the distinctive openness of forest understories maintained by elk, to other descriptions of a sea of undergrowth. Several early reports describe an open forest understory maintained by browsing elk, where elk trails and open park-like conditions made travel easy. For example, from

1890-93, Morganroth (1991; referencing that period) wrote: (1) of the Bogachiel Valley, "*we continued…through several broad, parklike bottoms…apparently a favored stomping ground [of the elk] as the benchland had been well browsed and there was open grassland near the river"; (2)* of the Hoh Valley, "*elk trails were going in and coming out everywhere and the moss-covered ground made walking easy, especially through areas where elk had browsed*"*;* and (3) of the South Fork Hoh, "*I found the South Fork Valley covered with …low undergrowth, much of which had been killed out by overgrazing of elk. This made our travel comparatively easy*." Similar descriptions left behind in the journals of the Press Expedition, the first recorded trans-Olympic traverse, described an area in the Elwha Valley as a "*veritable elk pasture*" (Barnes 1890). The same year, during the summer, a member of the U.S. Army's O'Neill expedition described an area in the Queets Valley, apparently near Hee Hee Creek where "*Numberless elk had kept the under growth down*" (Fisher 1890). Clearly, areas where elk maintained an open park-like understory were common on the primary winter ranges of the Olympic Valleys, although their extent is difficult to discern.

These descriptions contrast markedly with those of E. S. Meany (1890) of the Press Expedition near the journey's end as the party descended the Quinault Valley. In one day, Meany wrote: (1) "*We…travelled the length of the island through a dense jungle of underbrush*"; (2) "*We continued all day down the river through a dense underbrush consisting of a tangled thicket of salmon berry bushes, vine maple, and all other usual small growth that can be imagined*"; and (3) *"…the bank of the river becoming now so dense with underbrush as to be almost impenetrable.*" It is perhaps telling that Meany also reported that this was "*one of the hard days of the trip*", that members of the party were "*sick with dysentery*" and running critically low on essential provisions (flour and meat). Because the expedition party appeared to be at a physical and emotional low point, this description of unbearable vegetation, recorded on a single hard day of travel (i.e., a localized area), may not be broadly representative of elk winter range characteristics of the time.

The temporal juxtaposition of these writings suggests that forest understories were highly variable. Indeed, just the day before and a few miles from where Meany (1890) described the party's battle with impenetrable vegetation, he described "*a large open bottom, enabling us to see up and down the valley*." Similarly, just a few miles upriver from the "*veritable elk pasture*" reported on the Elwha River, Barnes (1890) reported traveling along the bottomlands through an old burn that had grown into a "*tangled thicket*", forcing the party to literally hack their way through. Nearly 50 years later, James (1934) described "*impenetrable jungles*", areas of extensive wind throw of trees, and areas where the "entire browse has been cleaned out" by elk, all in proximity. Similarly, Sumner (1939) wrote that elk maintained open conditions along the human trails where travel was easy and thickets formed where elk "*found it slightly more difficult to travel*." From the earliest reports through to the mid-1900s, both before and after wolves were eradicated, understory density and human travel conditions appeared to vary over fine spatial scales. Early writings suggest that understory density varied with respect to conditions that impeded travel by elk, including gullies, burns, blowdowns, and vine maple thickets (Barnes 1890, Bailey 1918, James 1934, Sumner 1939). To anyone who has travelled any distance off trails in the rainforest valleys of OLYM, this variation rings true today.

Early Biologists: The Range Conditions

The historical record is unequivocal that elk began exerting an ever more pronounced effect on floodplain and terrace vegetation as their populations increased in response to protective measures, including wolf eradication, at the turn of the 20th century. From 1918 to the 1950s, several biologists described the remarkable influence of elk browsing on vegetation, which was frequently reported as very heavy or overbrowsed (Bailey 1918, Riley 1918, Skinner 1933a, Murie 1935a, Murie 1935b). Accounts described how elk had “cleaned up”, “cleaned out”, “destroyed”, “inhibited”, “eliminated”, “highlined” or simply “overbrowsed” the most palatable shrub species, notably salmonberry, huckleberry, vine maple, willows, cottonwoods, or bigleaf maple (Appendix E). Reports that elk had inhibited reproduction or recruitment of cottonwoods and bigleaf maples were commonplace (Cliff 1934, Murie 1935a, Murie 1935b, Newman 1954). The hedged condition of individual shrubs was often described as “highlined”, “stubby”, and “broomlike” in appearance.

Early biologists reported variable browsing pressure, both spatially and over time. Most reported that overbrowsing was greatest in the Hoh, Queets, and Quinault Valleys and less in the Bogachiel and Elwha Valleys (Murie 1935a, Murie 1935b, Sumner 1938, Schwartz 1939). Schwartz (1939) interpreted this variation among winter ranges as likely related to greater extent of preferred fluvial terrace and floodplain habitats in the Hoh, Queets, and Quinault Valleys. Heavy browsing was frequently reported in a 15-mile reach of the Hoh Valley from near the current park boundary up to Olympus Guard Station, in the Queets from Harlow Bottom up to the confined head of the floodplain, and in parts of the Quinault Valley. Olaus Murie (1935b) reported that browsing pressure was most intensive at the upper reaches of each of the primary elk valleys. Temporally, early homesteaders reported that during the period of lowered elk densities, understory shrubs, particularly salmonberry, had been denser further up valley in many of the watersheds. Following elk population recovery the elk ranges were open and park-like due to intensive elk browsing (Murie 1935b).

The contributing role of domestic livestock to the range condition of the 1920s-50s has been underappreciated in the recent past and bears closer review. Early pioneers settling in bottomland forests in and near the current park generally kept livestock, including cattle, sheep, goats, and pack animals (horses and mules). The potential effects of domestic livestock on vegetation conditions in the western rainforest valleys was first mentioned in the 1930s and trespass grazing by unauthorized stock persisted until the 1950s, well after the establishment of the park (Skinner 1933b, Murie 1935a, Newman 1953, 1958). Joseph Dixon (1943), an early NPS wildlife biologist, believed “*this unauthorized trespass by domestic stock within the park as the greatest present menace to native wildlife in Olympic National Park.*” He estimated about 100 cattle ranged in the Queets Valley, throughout the primary elk ranges, and recognized similar problems near the current Hoh Ranger Station, and in the North and East Forks of the Quinault. As late as 1958, Newman reported unauthorized livestock in four areas of the park, most critically the Hoh and Quinault Valleys, but also the Queets and Elwha. These early observers believed that much of the over-browsed condition was related to grazing by domestic livestock. Clearly not all of the overuse was attributed to domestic livestock as indicated by Newman’s (1953) description in the Quinault Valley: “*Up the river, above the range of the good-blooded Angus cattle, the elk forage plants are in only fair condition.*”

Biologists reported a gradual improvement in elk range conditions over time, first in response to the harvesting of nearly 1200 elk during 1935-38 (including 507 from just the Hoh Valley), prior to the park's establishment (Schwartz 1939). Schwartz reported that the percent of browsed twigs of vine maple declined from 90% to 40% and that of huckleberry decreased from 80% to 50% in the Hoh Valley from 1936-1940. By the 1950s, Sumner (1952) reported that both elk and range were in good condition and improved from conditions witnessed in the 1930s. A few years later, Newman (1958) concluded that "*elk herds are numerous in some areas and are causing an over-browsed look to some plants, but the range is not in such poor condition that a reduction in elk is necessary*." He also concluded that erosion was not a significant problem due to the abundance of big trees, logs, mosses, and other debris along the rivers, even if the browse plants were reduced. Jenkins (1981) reported that twig utilization measured in the Queets, Quinault, and Bogachiel winter ranges were generally less than those reported 40 years previously by Schwartz (1939). In addition to the effects of early hunting, the removal of domestic livestock and the gradual recovery of cougar populations may have assisted with the gradual improvement of range conditions.

Reference Conditions: Browse Species

The accounts of early explorers reveal that elk left an indelible imprint on forests and landscapes before EuroAmericans appreciably altered landscapes, wolves, or elk populations. The anecdotal descriptions of vegetation correspond well with the longer-term condition discerned from pollen, where west-side river valleys have been dominated for millennia by western hemlock and red alder with Sitka spruce also consistently present. Black cottonwood has been a part of the flora but has been patchy in time and space such that its easily decomposed pollen has not been well preserved in the pollen record. Bigleaf maple has also been patchy in space but at some sites has been more consistent through time than cottonwood (Heusser 1964). The vegetation pattern must have been patchy because reports describe areas where heavy browsing and grazing pressure maintained open understories, while others describe very dense understories. We conclude that cottonwood may have been a component of developing floodplains at the turn of the last century, at least in the lower river reaches where most descriptions originated. The condition of many other browse species appear to have improved after elk populations declined and livestock were removed from park lands.

Reference Conditions: Provision of Logs to Rivers

Reference conditions regarding provision of logs to rivers can be best approximated by the early period of the time-series of available data described below. Trends in these data over time indicate change from the reference condition and will be discussed along with the data.

Results and Assessment

Vegetative Cover of Key Elk Browse Species

Woodward et al. (1993, 1994) summarized the results of vegetation changes within ungulate exclosures erected from the 1930s to 1950s. Shrub cover increased in most exclosures (11 of 14), while grass cover decreased inside the majority (10 of 14) of exclosures over the period when data was collected, which ranged from 3 to 21 years. Ferns tended to remain constant or increase inside the exclosures (increase in 3 of 14, no change in 4, not reported in 7), whereas the response of forbs was inconsistent. Huckleberry clumps increased in size and stem density inside of two exclosures where that trait was explicitly measured. Stem density of vine maple was relatively consistent inside and outside of exclosures, but stems were longer inside exclosures and a greater proportion of stems were established from seed inside the exclosure than outside. Most stems outside the exclosures originated from adventitious sprouting of established shrubs.

A more complete time series of cover measurements associated with two of the exclosures established in the 1950s allowed a more thorough examination of the timing and duration of vegetation responses to ungulate exclusion (Figure 4.1.9). Vegetation responses were more pronounced in the exclosure near the Elwha River, representing the red alder (established floodplain) community (Figure 4.1.9). In the Elwha stand, shrub cover began increasing rapidly approximately 4 years following exclosure establishment, and grass cover began to decrease appreciably relative to controls after about 8 years. Forb and fern responses were inconclusive. These trends remained evident for the duration of sampling.

Vegetation response to ungulate exclusion was less dramatic in a Sitka Spruce-dominated community on an established river terrace in the Bogachiel River valley (Figure 4.1.9) compared with the Elwha alder stand. Responses of grasses, shrubs, and forbs inside the exclosure were minimal, whereas cover of ferns increased beginning approximately 3 years following ungulate exclusion.

Results describing changes in abundance of selected plant species in the more recently established ungulate exclosures in the South Fork Hoh (Schreiner et al. 1996) confirm the general trends in vegetation responses to protection from herbivores seen in earlier exclosure studies. A comparison of vegetation cover inside and outside of two exclosures eight years after construction confirmed that grass cover was lower, and fern and shrub cover was greater inside than outside the exclosure in vegetation patches that were initially grass-dominated. In moss-dominated patches, differences in grass, shrub, and fern cover were indistinct, but forb cover was greater inside than outside the exclosures. Outside the exclosures, Schreiner et al. (1996) demonstrated that fruit-bearing shrubs that are highly palatable to elk and deer – notably salmonberry, elderberry (*Sambucus racemosa*), and thimbleberry (*Rubus parviflora*) – were restricted largely to refugia. Refugia were defined as areas naturally protected from large herbivores atop or between large fallen logs or on elevated root masses of wind-thrown trees.

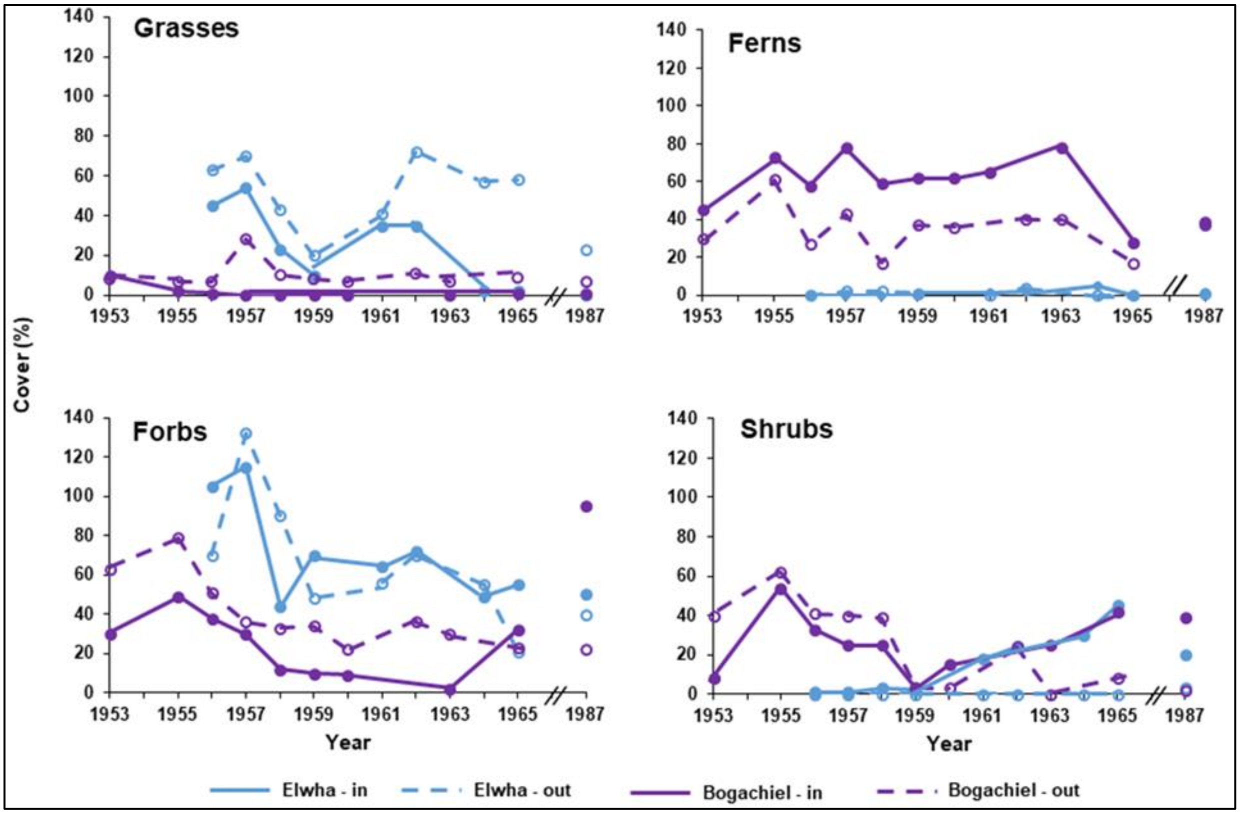

Figure 4.1.9. Percent cover of plant species groups inside ("in") and outside ("out") of elk exclosures established in two river valleys in the 1950s. Note that percent cover was derived by summing the cover estimates for individual species and may exceed 100%. Point symbols indicate when sampling occurred. Data source: Woodward 1993, 1994.

Our time-series analyses of vegetation cover in these two exclosures from 1980-2016 confirm many of the same temporal trends evident from the 1950s exclosure studies, at least for the first decade or more following exclusion. Palatable shrub and fern species such as salmonberry, lady fern (*Athyrium felix-femina*), and deer fern (*Blechnum spicant*) increased in one or both exclosures, whereas grass cover declined following exclusion (Figure 4.1.10).

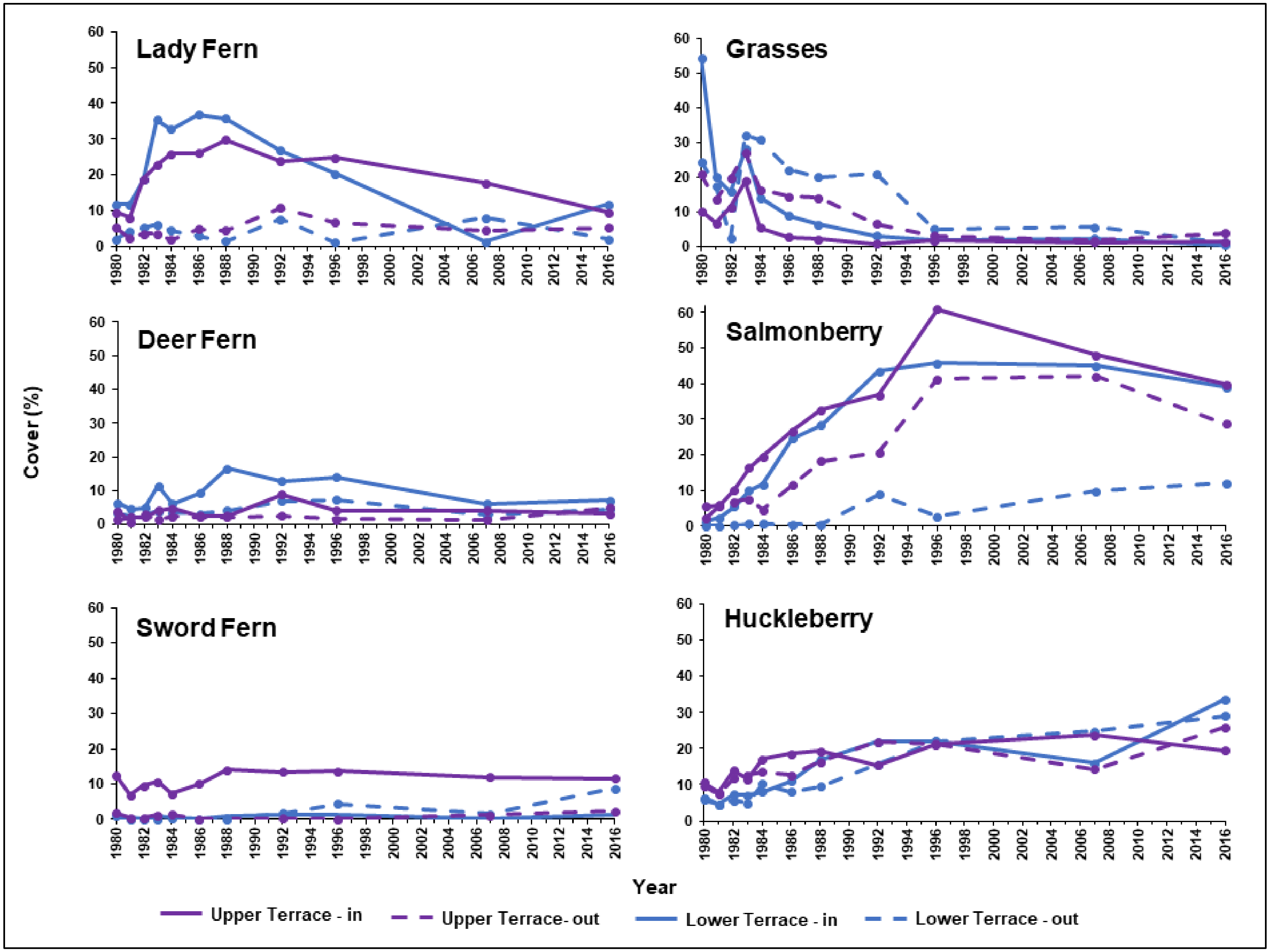

Figure 4.1.10. Percent cover of selected fern, grass, and shrub species sampled inside ("in") and outside ("out") of elk exclosures established on two terraces of the South Fork Hoh River floodplain in 1980. Point symbols indicate when sampling occurred. Source: HJ Andrews Long-Term Ecological Research.

After approximately the first decade, however, the effect of elk exclusion on plant community structure appeared to diminish as other factors became more prominent. Over the years, many large trees came down in and around the exclosures, some breaching the fences and requiring repairs to prevent entry by elk or deer. The needed repairs were accomplished until 2007. During a visit in in 2016, we noticed that limited entry was possible. However, the only changes in established trends in vegetation cover between 2007 and 2016 were an increase in lady fern and huckleberry inside the lower terrace exclosure (Figure 4.1.10) – not results expected from a potential increase in ungulate herbivory. The downed trees created large new canopy gaps, in some cases dramatically changing light availability, while also creating new refugia (natural protection from elk and deer herbivory), and providing new seed beds for tree establishment. The result was a very different patchwork of light availability, grazing pressure, and shrub and tree development both inside and outside the exclosures from what was evident during prior decades. We speculate that many differences in vegetation structure attributed to elk herbivory in previous decades may now be obscured by overwhelming changes in growing conditions both inside and outside the exclosures. Lady fern and deer fern are good examples of increasing cover due to elk removal followed by decline in subsequent decades, likely due to competition with salmonberry for light.

Previously, we reported that counts of elk in the South Fork Hoh have declined by about 60% since 1985 (Chapter 4.1.1). The decline in elk has been accompanied by an observable increase in shrubs and ferns outside of the exclosures (Figure 4.1.11). This change in elk density may partially explain the recent uptick in salmonberry and sword fern (*Polystichum munitum*) cover outside of the lower terrace exclosure and slight increase in sword fern outside of the upper terrace exclosure (Figure 4.1.10). Because lady fern and deer fern are highly preferred forages, they may not increase unless elk are completely absent. Surprisingly, huckleberry has increased inside and outside of both exclosures since early in the record. This was our best opportunity to compare among observable changes, elk density and the sensitivity of plot data. While changes in elk density resulted in an obvious change in vegetation to the human eye, the changes in measured plant abundance were comparatively small. We surmise that natural variation in many factors influencing plant growth, including herbivory, light availability, and growing conditions have combined to obscure the specific effects of elk.

Figure 4.1.11. Increase in understory vegetation cover outside of an ungulate exclosure in the South Fork of Hoh River, 1986-2017. The photographs were taken in 1986 (left, source: NPS) and 2017 (right, source: USGS). Arrows mark the location of the same fenceline post for contrast. The exclosure is located on a lower alluvial terrace approximately 4 km from the Olympic National Park Boundary and about 200m from the South Fork Hoh River.

Recruitment of Overstory Trees

Results from elk exclosures built in the 1930s and 1950s indicated that elk impeded the establishment of western hemlock in favor of Sitka spruce, but the effect was observed primarily in situations where rotting wood was available as a substrate for germination of tree seedlings (Woodward et al. 1994). Summaries of tree establishment from the more recent South Fork Hoh exclosures also showed an initial increase in western hemlock seedlings when elk were absent (Figure 4.1.12). Subsequent declines in western hemlock seedling abundance were likely due to natural thinning of some seedlings and others becoming trees. By contrast, Sitka spruce seedlings and saplings were more abundant outside of exclosures, indicating that factors other than elk were controlling establishment (Figure 4.1.12). While Stolnack and Naiman (2010) also concluded that factors other than elk controlled Sitka spruce establishment, they detected no clear relationships between establishment and indicators of soil moisture conditions, disturbance frequency, and competition. Inhibition of western hemlock establishment by elk is expected from its greater palatability to elk compared with Sitka spruce (Jenkins and Starkey 1991). However, tree ages sampled over 4 ha of the South Fork Hoh valley did not show higher western hemlock establishment at the turn of the 20th century when elk numbers were low (Harmon and Franklin 1983), even though the sample included a size cohort that seemed likely to have been the appropriate age. These contrasting observations may indicate that elk densities were not greatly reduced or not reduced for long enough in the South Fork Hoh River valley at the turn of the 20th century to inhibit establishment of western hemlock. Harmon and Franklin (1983) hypothesized that western hemlock establishment may be more sensitive to changes in availability of rotten wood and light than to elk abundance.

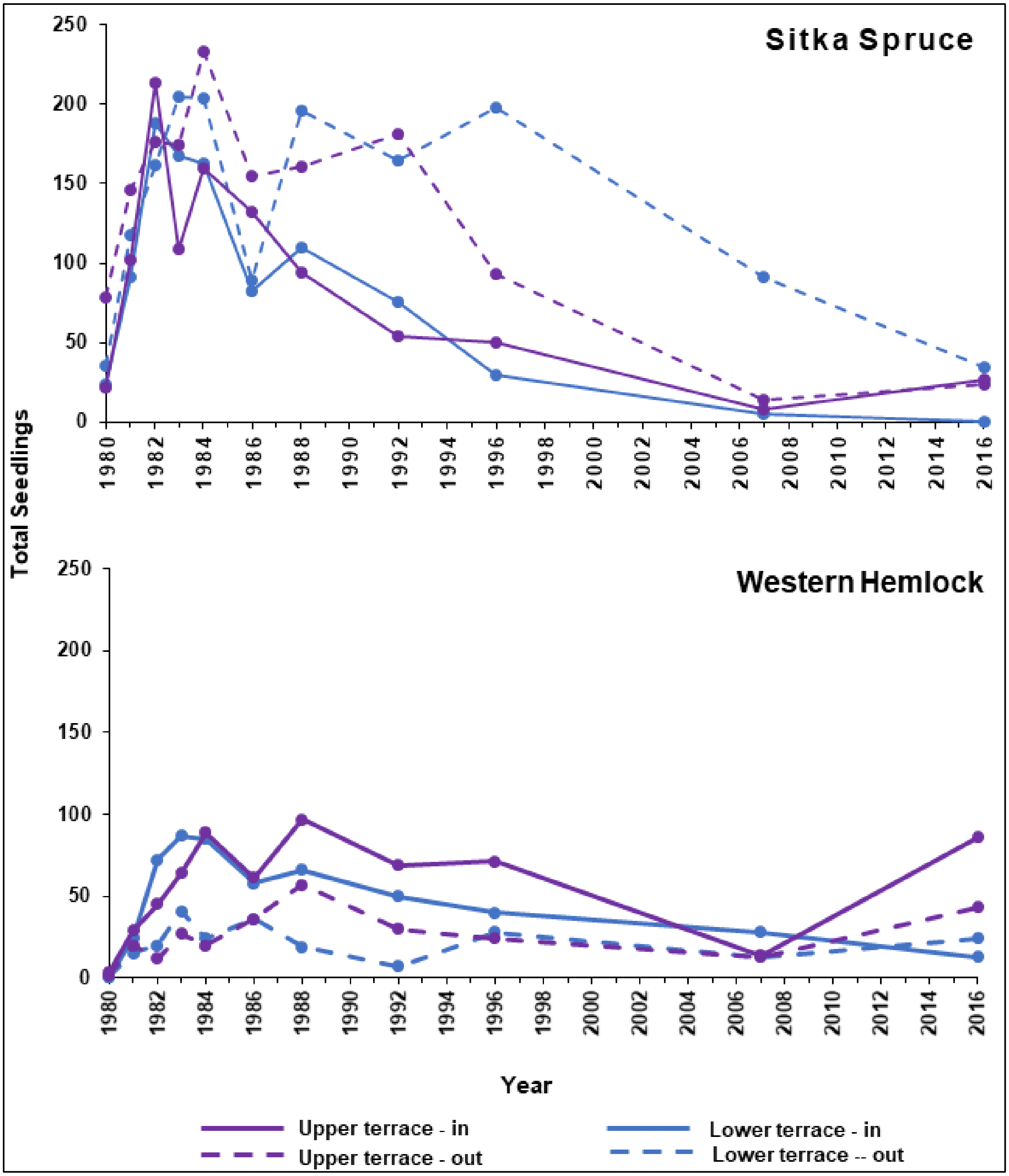

Figure 4.1.12. Total number of Sitka spruce and western hemlock seedlings enumerated in four 50-m belt transects sampled inside and outside of elk exclosures established on terraces in the South Fork Hoh River floodplain in 1980. Point symbols indicate when sampling occurred. Source: HJ Andrews Long-Term Ecological Research.

Concern that browsing by elk has negatively affected cottonwood and bigleaf maple abundance has been expressed since elk populations peaked in the 1930s. Olaus Murie (1935b) mentioned a lack of bigleaf maple and Newman (1953, 1954) was concerned that poor cottonwood and bigleaf maple recruitment was due to elk browsing. Jenkins (1981) found that the abundance of cottonwood seedlings and saplings was greater in the Bogachiel Valley where overall browsing pressure was lower than in the Quinault and Queets Valleys. While studying the Hoh River floodplain, Fonda (1974) found no cottonwood trees <11 cm in diameter at breast height, whereas Van Pelt et al. (2006) also found very few cottonwoods in young stands in the Queets. Based on aerial photos and ground plots, Houston (1994) determined that while cottonwood recruitment was occurring near river margins, proximity of mature cottonwoods and situations that created refugia from ungulate browsing were required for cottonwoods to attain larger size classes. Shafroth et al. (2016) suggest that cottonwood establishment may have been enhanced in areas avoided by elk as long as gravel bar habitat was available. Beschta and Ripple (2008) showed that recruitment of cottonwood and bigleaf maple has been extremely poor since wolves were extirpated and elk populations increased early in the last century (Figure 4.1.13). Based on these studies, it seems evident that ungulate browsing deters cottonwood growth beyond the seedling stage in areas of high elk density where refugia are rare or absent.

While elk herbivory appear to play and important role limiting cottonwood and bigleaf maple establishment, other factors may also be relevant. Black cottonwood occurs along the Pacific coast to the Rockies from Kodiak Island, Alaska to San Francisco, California and in scattered populations east of the Rockies (DeBell 1990). Much of the physiological and autecological research on this species has occurred in drier environments where cottonwood is a keystone riparian species rather than the coastal Pacific Northwest (see Braatne et al. 1996). Nevertheless, general conclusions likely hold throughout the range.

Cottonwood seedlings can potentially establish on newly available surfaces, however a particular sequence of hydrologic and biologic events is required for establishment. These required conditions may occur at irregular intervals depending on flow conditions (Braatne et al. 1996). Ideally cottonwood flowering and pollination occurs during peak spring flows so that seed dispersal coincides with declining flows (Stettler 2009). The window of opportunity is small because seed viability is only 2 weeks, and only 2-3 days if seeds are wet (DeBell 1990). Then soil must stay moist for at least 1-2 weeks and water table decline cannot exceed root growth. These conditions exist close to the river’s edge where the possibility of scouring is also greatest, thus flow must remain benign. Cottonwood tends to establish in a narrow band parallel to the river at an elevation that is not too low as to be scoured and not too high as to quickly dry (Rood et al. 2003, Stettler 2009). Black cottonwood establishes best where water tables remain close to the surface and where flooding is not normally accompanied by massive sedimentation. These often are areas where a natural levee occurs between black cottonwood and the river (Hawk and Zobel 1974). In the Pacific Northwest, cottonwood can also establish in places not adjacent to rivers as long as soil moisture is high enough (Niemiec et al. 1995).

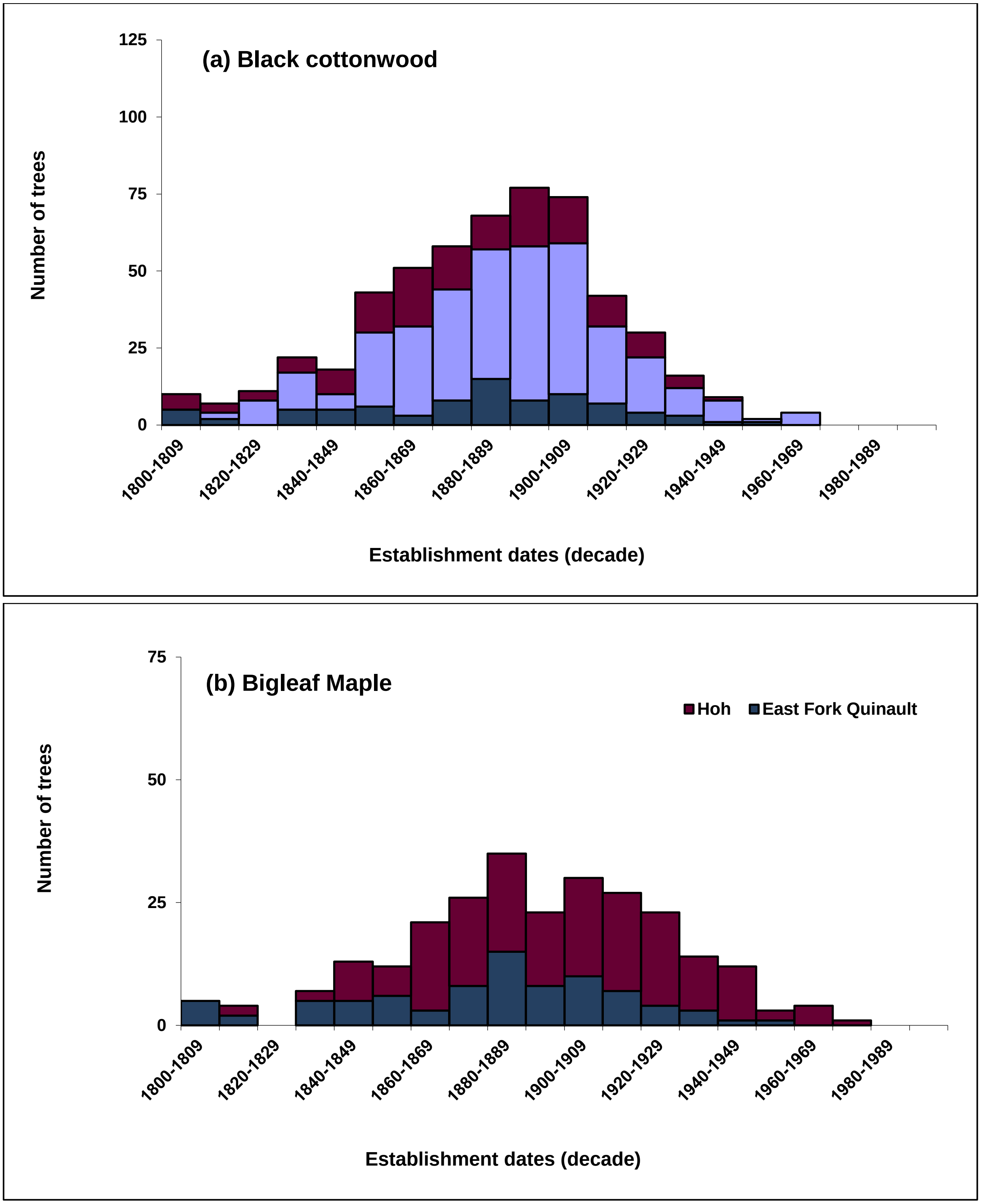

Figure 4.1.13. Establishment of black cottonwood and bigleaf maple in west-side river valleys of the Olympic Peninsula (Hoh, Queets, East Fork Quinault). Modified from Beschta and Ripple (2008).

Evidence suggests that bigleaf maple is not a generalist colonizer of main stem gravel bars. Rather, bigleaf maple seeds establish best on mineral soil and organic substrates, preferably those that stay moist during the growing season. These conditions pertain on colluvial and alluvial soils where there is both abundant moisture and deep gravelly soils. Fonda (1974) found that bigleaf maple in the Hoh valley was confined to colluvial fans, which occur at the confluence of tributaries and the main stem river. Bigleaf maple is not as tolerant of flooding as red alder and black cottonwood, but it can survive short periods of inundation (Minore and Zasada 1990). Consequently, bigleaf maple tends to successfully establish at greater distances from the water's edge than other riparian tree species (Minore and Zasada 1990).

Conifer Vegetation along Active Channel

Our analysis of the proportion of active channel margin in contact with large conifers capable of becoming key members of debris jams showed consistent results among the Hoh, Queets and Quinault Rivers (Figure 4.1.14) Approximately 30% of the active channel margin was adjacent to large conifers from 1939 until approximately 1980. After 1980, there was a steady increase in contact of the active river channel with large conifers, perhaps due to the erosion of older vegetated landforms, which corresponds with the shift in regime to higher river flows beginning in 1976 (see Chapter 4.1.3). Higher flows potentially coupled with higher sediment load due to melting glaciers likely caused increased bank erosion (East et al. 2017), which may have exposed more old-growth forest to the river margin.

Valley Floor Patch Composition

Over a 60-year interval (1939-2009) there has been little variation in the abundance of four valley floor vegetation patch types at 4 sites in the alluvial valley of the Queets River (Figure 4.1.15). Patch types include river channel (flowing water), pioneer bars (young surfaces still frequently flooded), floodplain (alder and willow dominated surfaces infrequently flooded), and alluvial terraces (free from flooding). While there have been differing responses among sites and some variation among years, patch abundance remains well within the range of variation seen over the last seventy years, even during recent decades when flood forcing was greatest. Patch composition has also remained relatively constant despite changes in ungulate (including livestock) herbivory effects which appeared to be greatest during the 1930s and to have diminished subsequently. The most notable trends have been toward less channel ($p = 0.023$) and more floodplain ($p = 0.210$) in recent decades, but the change has been minor.

These results showing less channel area contrast with the increase in channel width seen by East et al. (2017). Besides considerations of sampling error and differing spatial scales, the difference can perhaps best be explained by the different measurements used for two separate objectives. East et al. (2017) included islands within the channel in measuring channel width while Latterell et al. (2006) counted vegetated islands as pioneer bars or floodplain. The results showing little change in patch abundance also contrast with our observation of an increase in the length of bank that is in contact with large trees (Figure 4.1.14). This contrast may be explained by the large area of terraces (Figure 4.1.15), most of which is not adjacent to the river. A minor loss in terrace area due to increased bank erosion could nevertheless appear as a substantial change in length of mature fluvial terrace adjacent to the active channel.

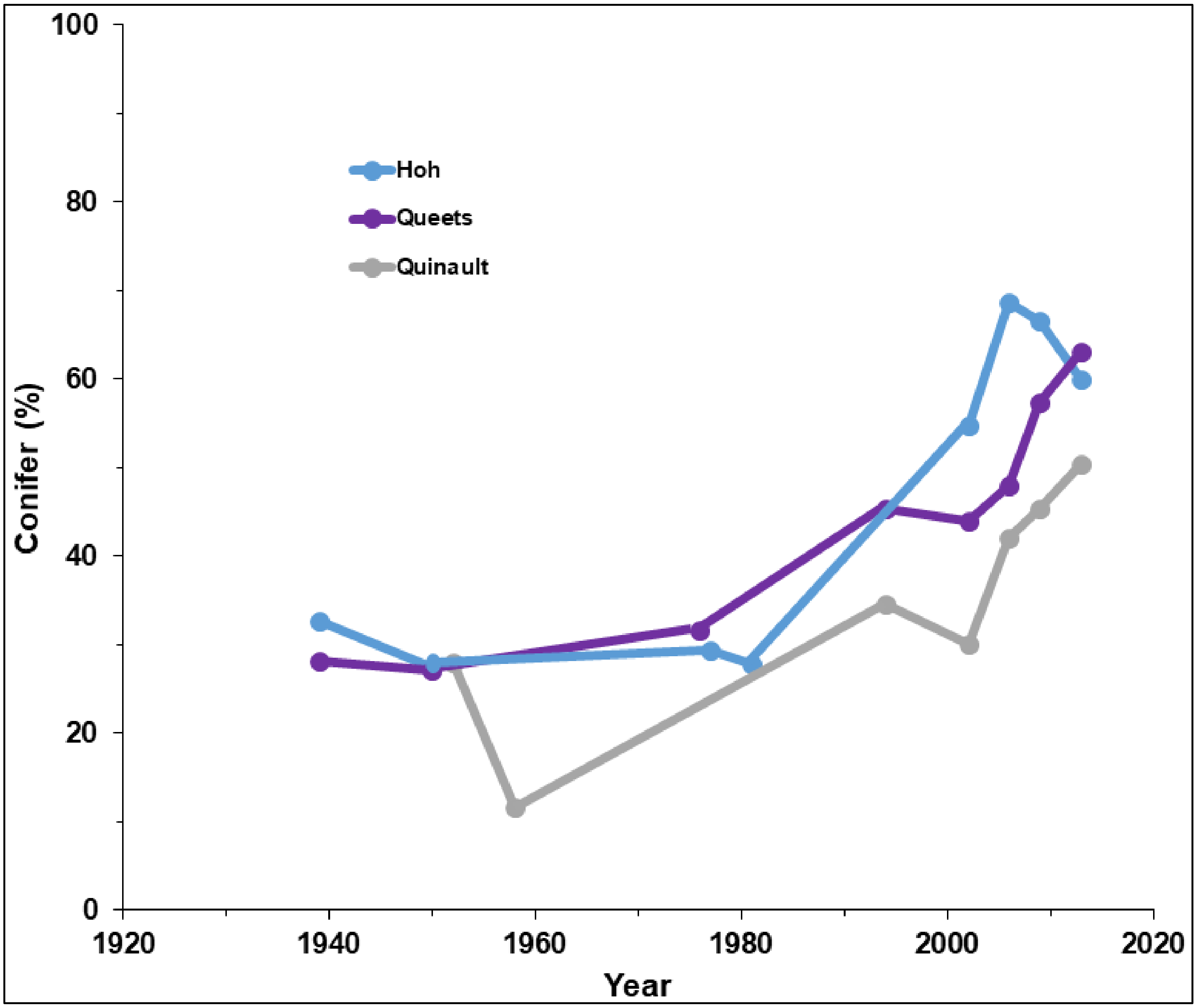

Figure 4.1.14. Portion of active channel margin occupied by coniferous trees with sufficient height to be >50 cm in diameter. Source: USGS https://www.sciencebase.gov/catalog/item/5a14b60fe4b09fc93dcfd810.

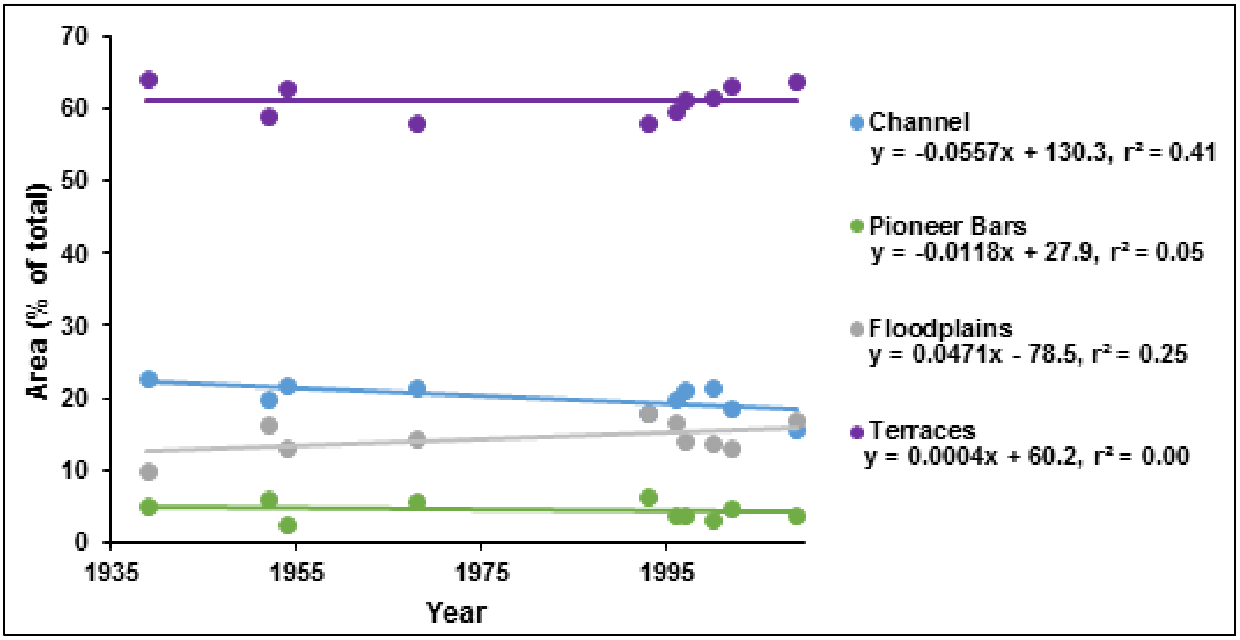

Figure 4.1.15. Trends in abundance of four landform types in floodplains of the Queets River, 1939-2009 (data points are averages of four plots [Latterell et al. 2006] supplemented with new assessment from 2009).

Although river channels are extremely dynamic, with mean annual migration rate of 13 m/yr in the Queets (Latterell et al. 2006), the net abundance of vegetated patch types has remained relatively constant. This result contrasts with that of Beschta and Ripple (2008) who concluded that erosion of older river terraces could result in their elimination over time. While we recognize that some terrace erosion may represent loss of Pleistocene landforms that cannot be replaced in an ecological time frame, terraces created from Holocene and ongoing contemporary processes are continually eroding and developing. Van Pelt et al. (2006) estimated that 100+-year old terraces support mature spruce forests, which likely provide similar ecological functions as far older terraces. In their assessment, Beschta and Ripple (2008) focused exclusively on the patch-specific erosion rates measured by Latterell et al. (2006) and failed to consider patch regeneration. Our results indicate the composition of patch types reflects a shifting-mosaic steady state over the past several decades (Figure 4.1.15). The slight trend toward more floodplain patches and no change in terrace abundance suggests that supplies of logs will be sufficient to maintain historic influences of riparian vegetation on fluvial dynamics into the future.

Elk Effects on Valley Floor Vegetation

Elk have clearly affected valley floor vegetation through preferential grazing that favors grasses and hinders palatable shrubs, especially salmonberry (Happe 1993, Schreiner et al. 1996), while also favoring the establishment of Sitka spruce and alder over western hemlock, cottonwood, and bigleaf maple (Schwartz and Mitchell 1945, Woodward et al. 1993, Woodward et al. 1994). These potentially profound effects of elk on valley floor vegetation appear to be limited by the patchy nature of both elk use and vegetation structure. At the finest scale, grazing affects forest understory growth in grassy patches more than patches with a mossy matrix (Happe 1993, Schreiner et al. 1996).

Grass patches are created in moist areas when fallen trees cause light-filled gaps; these grassy understories are subsequently maintained by elk grazing (Happe 1993). Some areas are not used by elk because they are protected atop windthrown logs or when fallen logs block or re-direct elk movement (Harmon and Franklin 1983, Houston 1994, Schreiner et al. 1996). Although intensive herbivory pressure restricted the distribution of many palatable shrubs to such refugia, windfall is very common (often covering up to 10% of the ground; Harmon and Franklin 1983). Based on the rapid response of shrubs to herbivore exclusion and the ready source of seed provided from an abundance of refugia, we conclude that herbivory effects on forest understory vegetation cover is ephemeral and very dynamic both spatially and temporally. Based on the recent indications that elk populations have declined in local areas (Chapter 4.1.1) and vegetation cover of some preferred shrub and fern species has increased, we conclude that the lower elk browsing pressure has increased understory vegetation in some areas.

It is likely that high elk numbers are an important factor limiting the establishment of cottonwood and perhaps bigleaf maple and that the effect seems to have persisted at least until the 1990s (Houston 1994) or early 2000s (Beschta and Ripple 2008). However, there are other factors that influence the abundance of these species in riparian areas. Despite incomplete knowledge of factors affecting cottonwood and bigleaf maple establishment, we conclude that the apparently poor tree establishment relative to historical descriptions warrants concern over the current and future conditions of these tree species in riparian forests. Changing climate, hydrology and elk herbivory may have combined to reduce cottonwood recruitment in OLYM below reference conditions. The prevalence of regenerating hemlock and Sitka spruce show no reason for concern for these species. Although western hemlock is selectively browsed by elk (Leslie et al. 1984), it germinates prolifically on logs that are out of reach of elk.

The length of experimental reaches where active channel abuts large conifers has increased since the flow regime change beginning in the mid-1970s (East et al. 2017). Results indicate that there is abundant contact between active channel and large conifers, suggesting there continues to be a high potential for delivery of logs into the active channel. Results from the analysis of abundance of riparian vegetation patch types indicate that the amount of mature fluvial terrace, the source of logs to rivers, has not changed appreciably since 1939. These indicators suggest there has not been a detected loss in either of the availability of large wood for potential delivery into riverine systems or changes in floodplain landscape composition despite changes in active channel width (Chapter 4.1.3).

Uses and Limitations

Our analysis of vegetation trends using both pollen and historical records and limited studies of elk browsing legacies has offered new perspectives on the historical and current condition of the temperate coniferous rainforest communities along the park’s major river systems. The work corroborates previous findings that elk likely play a key role in influencing the tree composition in floodplain and terrace forest communities, notably through their effect on cottonwood and maple establishment, and that they profoundly affect vegetation composition and structure of the understory more broadly throughout floodplain and alluvial terrace forests. The historical context provided here clearly suggested that abundant elk populations on the peninsula have always left a marked imprint on the vegetation, and that appearance of heavy browsing pressure is not solely or even primarily the result of wolf eradication. Review of the literature revealed that effects of domestic livestock may have been greater during the early decades following park formation than was previously appreciated, and that vegetation conditions have improved since vegetation was maximally affected by joint livestock and peak elk populations of the 1930s. The long-term examination of two 0.5-ha exclosures helped to also illuminate the complex interplay between elk browsing, windfall, and light on the responses of vegetation. Further, our work suggests that due to the prevalence of large Sitka spruce along the river margins and rapid colonization of alluvial landforms by alder and willow, reductions in the availability of large cottonwoods have likely not caused fundamental changes in long-term delivery and role of logs in river channels.

Nevertheless, large uncertainties remain in our understanding of elk-forest dynamics. Our conclusions are limited by lack of detailed vegetation descriptions preceding EuroAmerican settlement, and lack of replicated long-term vegetation monitoring plots. Our interpretations on the conditions and trends of key browse species and tree recruitment are based on localized, relatively short-term studies and anecdotal reports of observations by early explorers and settlers of the peninsula. Valley floor vegetation in the Pacific Northwest is extraordinarily diverse (Latterell et al. 2006, Van Pelt et al. 2006), and includes many long-lived species and successional patterns that are manifested over centuries rather than years or decades. Given the dynamic nature of vegetation both spatially and temporally in response to frequent disturbances, and the limited-scale of data collected to date, we are limited in our ability to offer firm conclusions on the long-term health and condition of vegetation composition in the face of changing predator, elk, and livestock populations. It is also difficult to project observed changes, such as the abundance of large trees adjacent to the active channel, for the centuries it may take for effects to have ecological consequences, especially given uncertainty regarding changing climate.

Although the series of exclosure studies have been informative in revealing the rapid responses of understory vegetation to total loss of elk from an ecosystem, they are much less helpful in discerning the effects of more subtle changes in elk density. Our exclosure studies revealed the complex interplay among multiple factors affecting vegetation cover and relative abundance, leading us to believe that even the relatively large 0.5- ha exclosures we studied may not be large enough to capture the full complexity of elk-vegetation dynamics over the long term.

Emerging Issues

Two drivers of valley floor vegetation structure and composition are currently changing in ways that may alter forest structure in the future. First, changing climate may directly affect vegetation by altering growing conditions. Climate change may indirectly influence vegetation due to the projected change in the amount and timing of river flow following shifts in precipitation regime and the relative influence of glaciers. Even though the flow regime shift since the mid-1970s (Chapter 4.1.3) has not been apparent in the abundance of valley floor vegetation patch types to date, it does appear to be affecting the amount of mature forest adjacent to the active channel. As elaborated in Chapter 4.1.3, the predicted increased incidence or magnitude of winter flooding may portend increased changes in valley floor patch composition in the future. Second, valley floor vegetation composition and structure may be affected if the apparent recent decline in the west-side elk population continues. This is already evident in the South Fork Hoh valley where the number of elk has declined and floodplain vegetation structure is changing.

Information Needs/Gaps

Relationships of Cottonwood to the Environment

Evidence suggests that cottonwood recruitment has declined over the last century in west-side river valleys. Elk herbivory is likely an important factor in the decline, but additional information on relationships of cottonwood to other environmental factors, specifically in the Pacific coastal environment, would be useful to better understand long-term effects of elk and other variables on cottonwood establishment. Suggested studies might include comparisons of recruitment and age structure of cottonwood trees among key watersheds of the park, along the longitudinal gradient from upper to lower river valleys, and in relation to substrate, soil, and flow regime and effects of particular high flow events on recruitment. The removal of the Elwha dams has uncovered suitable substrate for cottonwood establishment, which may create an opportunity to conduct research to better understand cottonwood recruitment dynamics. Anecdotal observations of cottonwood establishment and effects of browsing in the former reservoirs on the Elwha River emphasize the importance of substrate stability and the capability of some members of a cottonwood cohort to escape herbivory as important factors determining successful establishment (J. Chenoweth personal communication, August 2017).

Long-term Monitoring of Understory Browse and Tree Recruitment in Valley Floor Systems

Most previous studies of browse conditions and tree recruitment have been relatively short-term studies associated with ungulate exclosures built away from the active channel. A well-designed network of long-term, replicated reference plots on these valley bottom forests, including younger surfaces and the active channel, would be very helpful in determining the relative importance of disturbance regimes such as wind-fall, flood events, and ungulate herbivory on plant community composition and structure. Such a network would better describe the vegetation mosaic created by disturbance events and elk movements than establishing new exclosures. Also, replicating the relatively broad-scale surveys of understory patch types and shrub refugia in the South Fork Hoh (Schreiner et al. 1996) would help to improve understanding of changes in forest understory conditions since the late 1980s in this area.

Key Functions of Cottonwood and Bigleaf Maple

Given that cottonwood and perhaps bigleaf maple, may be in decline, it would be useful to better understand the potentially unique ecological values of these species and the effects of their lower abundance. Riparian trees provide shade, nutrients and habitat to riparian ecosystems. For example, cottonwood is a preferred browse species for beavers on the Olympic Peninsula (Knapp 2009), and abscising leaves of cottonwood may provide an abundant food source for elk and deer during autumn (Leslie et al. 1984). Bigleaf maple trees have the potential to host a tremendous amount and variety of epiphytic plants (Naiman et al. 2010) and produce leaf litter that increases potassium, calcium, and magnesium in soil compared with conifer litter (Turk et al. 2008). Perhaps these species also provide unique conditions for certain bird or insect species as well as other ecological benefits. Cottonwood also has the potential to quickly grow to key-member size so it may temporarily restore this function of situations where large coniferous trees have been depleted (Collins and Montgomery 2002).

Functional Redundancy of Deciduous Species

The deciduous species willow, alder, and cottonwood are important pioneers of new fluvial landforms and influence the subsequent stabilization and development of these areas. Given that cottonwood is apparently playing a lesser role than historically, it would be valuable to better understand the functional redundancies and any unique facilities among these species, specifically regarding geomorphology.

Effects of Climate Change on Log Delivery and Retention

Understanding the future of the reciprocal relationship between riparian vegetation and fluvial dynamics depends on projecting changes in fluvial dynamics due to changing climate. In particular, understanding the effects of changing flow regime on the delivery and retention of large wood in riverine ecosystems, and consequently, on sediment dynamics is needed to forecast the potential long-term consequences of changing flow regime on riverine ecosystems of the park. Repeated collection of LiDAR data in coming years would provide a more detailed view of changes in geomorphology and vegetation structure than we can discern from aerial photos alone. Future work may profit from study designs that provide sufficient detail on areas near the active channel to obtain better understanding of ecological dynamics in areas that supply large wood to river channels than we currently have from studies covering the entire valley bottom.

Resource Summary

We rated condition of riparian vegetation, which depends on the stability of both composition and relative abundance of valley floor patch types, as currently stable yet warranting moderate concern (Table 4.1.4). The very limited historical record, lack of long-term monitoring plots, and highly variable conditions reported in the historical literature all contribute to a rating of low confidence. The concern arises mostly due to the mounting evidence that changes in elk distribution or abundance has likely contributed to an unnaturally low rate of establishment of certain riparian tree species (i.e., cottonwood and bigleaf maple) over the last 90 years. We rate the overall condition as stable based on ecological considerations that the prevalence of other riparian tree species provides system resiliency to loss of palatable deciduous shrubs.

Table 4.1.4. Resource condition summary for riparian vegetation in Olympic National Park.

Resource	Species	Status and Trend	Rationale and Reference Conditions
Landscape Dynamics	Historic soil disturbance (relative severity and potential duration of impacts)	🟢 ⇔	While some areas were heavily impacted, historic impacts occurred prior to park establishment and many areas are assumed to have recovered. Trend is assumed given the protected status of these areas now.
Cover of Key Browse Species	General	🟢	Anecdotal accounts and limited field measurements suggest current cover conditions of key browse species are within the range of natural variation. Despite low cover of certain species, they seem stable within refugia where they are protected from browsing.
Seedling Establishment	Cottonwood	🟡	Lack of recruitment in the understory signals long-term loss of mature overstory trees.
Seedling Establishment	Bigleaf Maple	🟡	Lack of recruitment in the understory signals long-term loss of mature overstory trees in some plant communities.
Seedling Establishment	Western Hemlock	🟢	Although elk browse western hemlock selectively over Sitka spruce, there are abundant down and dead logs where western hemlock becomes established out of reach of browsing elk.
Seedling Establishment	Sitka Spruce	🟢	Elk do not selectively browse Sitka spruce. Records from the South Fork Hoh suggest an increase of Sitka spruce but spatial extent of this trend is not known.
Conifer Vegetation along Active Channel	General	🟢	The supply of trees adjacent to the active channel and having sufficient size and species composition to become key members of log jams has increased since the mid-1970s. The most likely cause is a change in flow regime at that time.
Floodplain Patch Composition	General	🟢	Despite river channel widening (Ch 4.3), the relative composition of floodplains and river terrace plant communities is relatively stable and remains within the historical range of variability since 1939. Unknown changes in future flow regimes as related to climate change contributes uncertainty.
Overall condition and trend of Riparian Vegetation		🟡	The resource condition is of moderate concern primarily due to lack of recruitment of palatable tree species. Although cottonwood and maple recruitment has diminished over the last century, conditions seem stable with respect to key ecological functions of riparian ecosystems.

4.1.3. River Geomorphology

Amy East, Kurt Jenkins, Tim Beechie, Jennifer Bountry, Mark Mastin, and Tim Randle

Introduction

Rivers are key features of the physical and ecological structure and function of OLYM. River channels and related environments (floodplains and side channels) are places of critical importance to water, nutrients, and carbon cycles, and provide essential habitat for wildlife (see review by Gurnell 2014). Many of OLYM's rivers drain the heart of the Olympic Mountains. These watersheds connect the high, glaciated uplands to lower-elevation valleys and ultimately, the Pacific Ocean, the Strait of Juan de Fuca, and Hood Canal. OLYM river corridors span over 2400 m of elevation and contain numerous physical environments, habitat types, plants, and animals. As such, rivers are not only a key natural feature of OLYM but also an important part of the social and economic value of this region. Many park visitors value the experience of hiking along or fishing in a river corridor, and the relatively pristine environment of OLYM rivers provides excellent opportunities for such experiences.

Elwha River. Photograph courtesy of USGS/Kurt Jenkins.

River geomorphology (landform shape, topography, and appearance) commonly changes over time, driven by both physical and ecological factors. Disentangling the various factors that influence river evolution often involves substantial uncertainty (Fetherston et al. 1995, Abbe and Montgomery 1996, Gurnell 2014, Corenblit et al. 2015, Gran et al. 2015). Physical factors that can alter river course or morphology (landform shape and appearance) include the amount of flow and sediment the river carries, the riverbed gradient, and the amount of large wood in the river system (logs >1 m in diameter; Wohl 2013). Effects of climate change may also contribute to changes in rivers such as those found in the alpine watersheds draining the central portion of the glaciated Olympic Mountains (Brocklehurst and Whipple 2002, Micheletti et al. 2015). Ecological factors affecting river morphology include the amount and type of riparian vegetation, especially large trees with roots large and deep enough to stabilize banks. Intensive herbivory and trampling by ungulates may reduce the cover and density of important riparian plant species, with potential consequences for river evolution (Beschta and Ripple 2006, 2008). Beschta and Ripple (2008) asserted that decades of overabundant elk populations in OLYM may have promoted a fundamental shift in river morphology from single-thread to a braided condition (i.e., a river with multiple channels split by gravel bars or islands).

Understanding the ways in which rivers have changed throughout the historical record, and identifying factors that have caused that change, is important because the physical, geomorphic condition of rivers in turn influences the ecological setting, including habitat availability for aquatic and terrestrial organisms and fluxes of sediment and nutrients through river-corridor landscapes. Understanding changes in river systems also has direct application to park managers because roads and other park infrastructure are present on the relatively flat floodplains and alluvial terraces of major river systems.

We evaluated river-channel change through time in the four largest rivers in OLYM—the Hoh, Queets, Quinault, and Elwha Rivers. Our purpose was to assess the current condition of riverine systems and processes in OLYM, while also developing a more comprehensive conceptual understanding of physical and ecological forces influencing river-system dynamics. We examined the historical aerial photographic record and interpreted those data in the context of hydrologic history, inferred sediment supply, elk populations, and empirically derived understanding of geomorphic processes.

Approach and Methods

We examined annual to multidecadal changes in two metrics to assess changes in the morphology and condition of OLYM rivers: (1) width of the active flow zone and (2) channel-braiding index. These are both key characteristics that define river planform (i.e., the shape and appearance of a river channel when viewed from above). We quantified these characteristics within selected reaches of the Hoh, Queets, Quinault, and Elwha Rivers (Figure 4.1.16; Table 4.1.5). To better understand causal mechanisms of change, we related changes in these measures over time to variations in mean annual stream flows and flood peaks. We examined statistical relationships between channel morphology, flood history, and mean-annual-flow history using generalized linear models. To identify whether a sudden shift in river planform may have occurred before the aerial photographic record began (prior to 1939), we examined historical stream-gage records for evidence of changes in riverbed elevation. We also consulted maps and surveys from the late 1800s and early 1900s, written historical accounts and photographs, and empirically derived slope–discharge relationships to channel form. Thus, we attempted to characterize river evolution as thoroughly as possible over decades that have seen variations in flow regime and sediment supply due to climatic fluctuations and landslides, fluctuations in herbivore and riparian-forest communities, as well as localized direct human modifications, especially the legacy of large-wood removal on the Quinault River (Bountry et al. 2005). Additional detail on our methods and results are available in a companion publication (East et al. 2017).

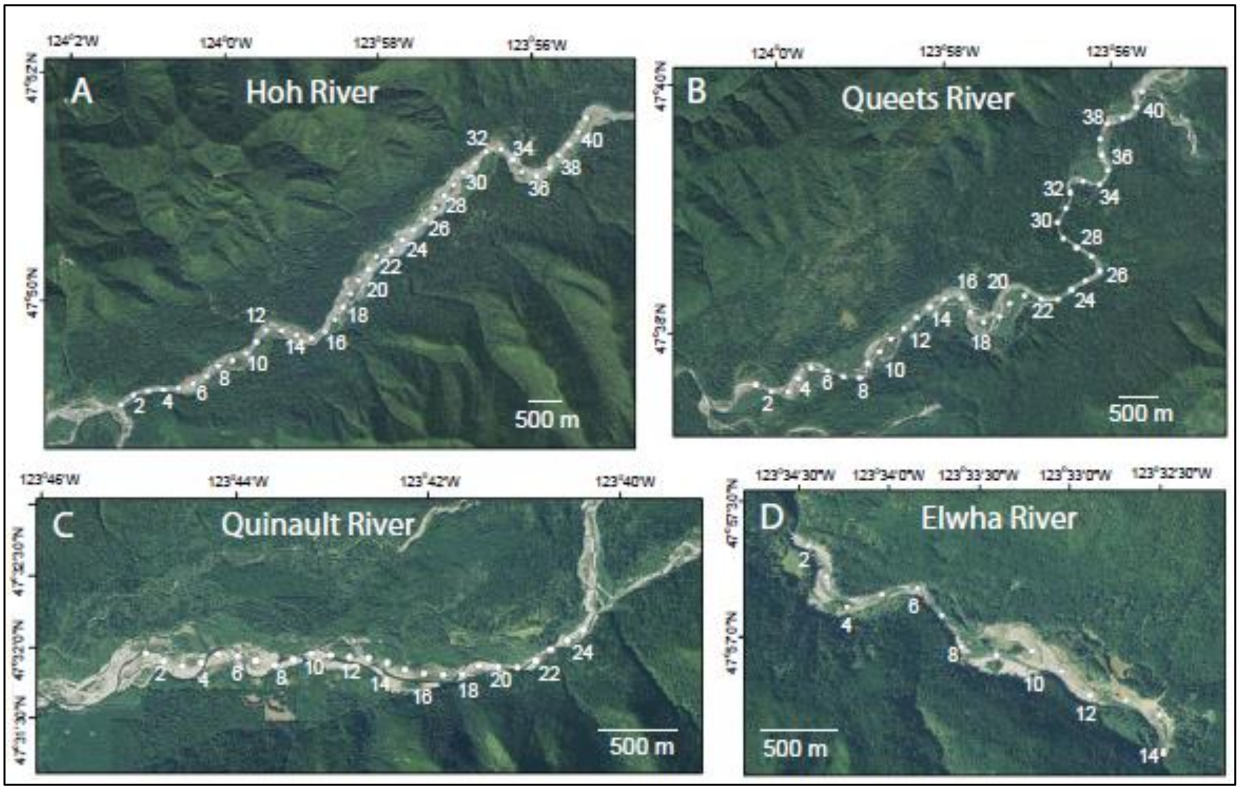

Figure 4.1.16. Aerial orthoimages of study reaches on the (A) Hoh River; (B) Queets River; (C) Quinault River (upstream of Lake Quinault); and (D) Elwha River. Images are from summer 2013 (USDA 2013). White circles show points 250 m apart used to assign transect locations on these and all other sets of aerial photographs for each reach. For clarity, only even-numbered points are labeled.

Table 4.1.5. Details of study reaches where river morphology was interpreted from aerial photographs of the Hoh, Queets, Quinault, and Elwha Rivers. Data sources: U.S. Geological Survey, 2017a, 2017b, 2017c, 2017d, 2017e.

River	**Reach Length (km)**	**Reach Gradient (m/m)**	**Basin Area Draining to Reach**	**Distance Above River Mouth (river-km)**	**Details of Study Reach**	**Mean Annual Flow (m^3/s)**	**2-year Flood Peak (m^3/s, Using Data Through 2016)**
Hoh	10	0.0056	324 km^2	46 from mouth to downstream end of study reach	Immediately upstream of South Fork Hoh confluence	72.2 (data from 1961–2016)	929 (using annual peak data from 1961–2016)
Queets	10	0.0042	452 km^2	39 from mouth to downstream end of study reach	Includes 2.5-km-long region where active flow zone is confined (one-third the width in the unconfined section)	124 (data from 1931–2016, but missing 1950–1974)	1910 (using annual peak data from 1931–2016, but missing 1967–1973)
Quinault	6.5	0.0045	478 km^2	9.5 from lower end of reach to Lake Quinault, 66 river-km to river mouth	Historical wood removal affected this reach. Reach is just downstream of North Fork Quinault confluence	81.4 (data from 1912–2016)	612 (using annual peak data from 1912–2016)
Elwha	3.25	0.0166	194 km^2	29 from mouth to downstream end of study reach	Geyser Valley; upstream of area influenced by sediment deposition when Glines Canyon Dam formerly impounded Lake Mills	42.7 (data from 1898–2016)	408 (using annual peak data from 1898–2016)

Measuring Channel Width and Braiding from Aerial Photographs

For each river, we selected an alluvial reach (where the river flows predominantly through its own sediment deposits rather than directly through and over bedrock) along a part of each river corridor. We chose reaches that had minimal recent land-use change adjacent to the river, and that were not influenced by substantial spatial changes in river discharge or sediment supply (Figures 4.1.16, 4.1.17). We note two exceptions. First, forest patches have been clearcut on the south bank of the Quinault River outside the park boundary (Figure 4.1.16C). Second, a landslide just above the Elwha River reach affected the Elwha River in the 1960s, as discussed below.

Figure 4.1.17. Aerial and oblique photographs showing representative morphology of study reaches on the Hoh (A-B), Queets (C-D), Quinault (E-F), and Elwha (G-H) Rivers. Note large logjam filling side channel of the Queets River (Panel C), anthropogenic clearings south of the Quinault River outside the park (asterisks in Panel E). By contrast, large clearing in Panel G is a natural feature. Aerial images from National Aerial Imaging Program (USDA 2013), field photographs by A. East.

We measured channel width and braiding on aerial imagery for each of the four rivers. We obtained aerial orthoimages as well as aerial photographs that had been georeferenced using ground control points (Piety et al. 2004, Bountry et al. 2005, Latterell et al. 2006, USDA 2006, Kloehn et al. 2008, USDA 2009, 2013). On each set of photographs, we evaluated width by digitizing the margins of the recently active flow zone, assuming that the unvegetated portion of the floodplain had been occupied recently by flow (Sear et al. 1995, Kondolf et al. 2002). We also digitized the midline of all active channels within the study reach, i.e., all wetted channels with surface-water connection to the mainstem river.

Along each study reach we established fixed points 250 m apart using ArcGIS™, in the center of the active channel in the most recent (2013) aerial images (Figure 4.1.16). At each point, we established transects perpendicular to the flow. We measured the number of braids that each transect crossed, as well as the active width (distance between left and right margins of the recently active flow zone, i.e., channel and adjacent unvegetated floodplain region). We calculated two indices of channel braiding: (1) the sum of the lengths of all channels, divided by the length of the mainstem river (Friend and Sinha 1993), and (2) the mean number of channels crossed by transects (Howard et al. 1970).

Stream-Flow Measurements and Peak Flood Events

We compared aerial photographic records of channel planform to river flow records obtained from USGS stream gages (http://waterdata.usgs.gov/nwis/, accessed 30 October 2017; Figure 4.1.18). Discharge recorded at each stream gage was somewhat higher than in the respective study reaches because the river gages were always downstream of the reference reaches and ungaged tributaries entered the mainstem rivers between the study reaches and the gage sites. However, the hydrographs were useful as a means to resolve seasonal and interannual flow variability and to identify approximate recurrence intervals of high flows. To assess decadal-scale changes in flood regime, and in particular to evaluate changes in flood activity over time, we calculated two-year-flood-magnitude (Q2) values from the record of annual flood peaks.

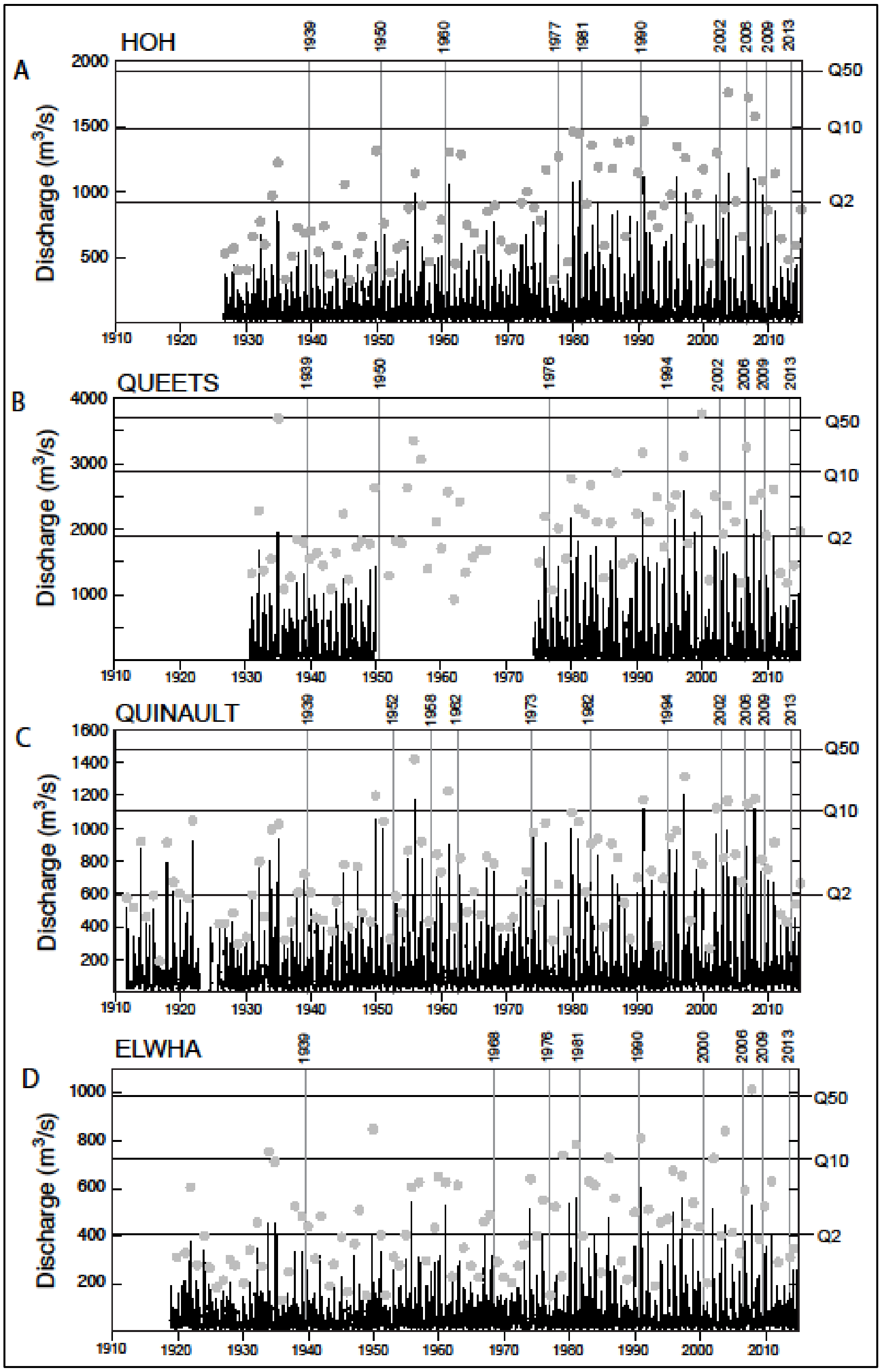

Figure 4.1.18. Hydrographs for the period of record for the (A) Hoh River, at USGS gaging stations 12041000 and 12041200 (see East et al. (2017), for explanation of combined gage records); (B) Queets River, at USGS gaging station 12040500; (C) Quinault River, at USGS gaging station 12039500; and (D) Elwha River, at USGS gaging station 12045500. Black lines show daily average discharge; gray circles show instantaneous annual peak flow values. Horizontal lines Q2, Q10, and Q50 indicate peak-flow magnitudes with 2, 10, and 50-year recurrence intervals, respectively, calculated for the period of record for each gage, up through and including water-year 2014 (September 30, 2014). Vertical gray lines show years corresponding with the photographic record.

We also analyzed stage-discharge history from 1950 and earlier, to determine whether bed-elevation changes occurred that might represent a large sediment pulse associated with a shift to wider, more braided channels. For that analysis, we selected discharge with a 1–1.5-year return interval, for which rating curves are reasonably well defined, and obtained the corresponding stage (water-surface elevation) from each stage-discharge rating curve.

Reference Conditions

River-channel characteristics as they existed prior to European settlement and the subsequent disruption of predator and prey populations would provide the best reference conditions for interpreting the condition of riverine processes today. Unfortunately, the aerial photographic record for river channels begins in 1939, more than 35 years after EuroAmerican settlers were firmly established in these environments. Although we lack a reference for detailed geomorphic interpretations from before 1939, we inferred historical conditions based on interpretations of cadastral maps from the late 1800s and early 1900s. We relied primarily on General Land Office (GLO) records, available from the Bureau of Land Management (BLM) (2016). For each study reach, we compared historic and current river width from our aerial photographs to measurements from the GLO records. We also examined (1) written descriptions and photographs from early explorer accounts of OLYM rivers relating to channel morphology (Wood 1967, Lien 2001, Jefferson County Historical Society 2010, Washington State Library 2015); (2) historical stream-gage records of river stage and discharge relationships; and (3) empirically derived slope-discharge relationships with river morphology (Leopold and Wolman 1957, Ferguson 1987, Knighton and Nanson 1993).

Results and Assessment

Overview

Our analysis of changes over time in flood magnitudes confirmed that flood peaks had been relatively low in the late 1960s and early 1970s, but increased thereafter (see also Piety et al. 2004, Bountry et al. 2005, Beschta and Ripple 2008, Czuba et al. 2012, Tohver et al. 2014, Castino et al. 2016). The magnitude of the 2-year flood over the most recent four decades has been 10–35% greater than over the entire period of record; details are given below for each river individually. The general increase in flood activity after the mid-1970s coincided with the onset of a wet phase of the Pacific Decadal Oscillation (PDO, an index of monthly sea-surface temperature anomalies over the North Pacific; Mantua et al. 1997). This mid-1970s climatic transition has been identified as a major atmospheric and hydrologic shift that affected a large region of the Pacific in both the northern and southern hemispheres (Castino et al. 2016).

The aerial photographs reveal clearly that OLYM rivers are dynamic systems that showed substantial spatial and temporal variation between 1939 and 2013 (Figures 4.1.19–4.1.25), including repeated geomorphic disturbance and recovery over time scales of less than a decade. These findings generally agree with those of previous studies (O'Connor et al. 2003, Piety et al. 2004, Bountry et al. 2005, Latterell et al. 2006). At any given time, the width of the recently active flow zone varied substantially along the length of each reach; width in some parts of a reach can be 6–9 times as great as width measured within other parts of the same reach in the same set of images. River channels varied from having (rarely) single-thread morphology to, more commonly, 2–4 wetted braids. Nearly every part of each reach changed over time due to meander migration of individual channels and avulsion (splitting off) of new channels. We found that when the recent flow history included a larger flood, river channels were wider and more braided. These statistically significant relationships (determined from generalized linear models) between recent flood activity and channel width and braiding were found for peak-flow (flood) history for all rivers ($p \leq 0.05$), but did not hold for mean annual flow ($p > 0.05$), indicating that high flows during winter (when the largest flows typically occur in any given year) are more important to determining channel morphology than are spring snowmelt flows (consistent with findings by Draut et al. 2011).

The aerial photographic record showed many examples of widening and braiding that were associated with recent occurrence of large floods. Photographs also showed recovery after disturbance, when the active river corridor became narrower (due to vegetation growth) and less braided during intervals with lesser peak flows. This was most pronounced during the late 1960s and early 1970s, when annual peak flows were relatively low. Flood peaks were generally greater between the late 1970s and 2010s than they were in the late 1960s and early 1970s (East et al. 2017; see details for each river, below). Over much of each reach, the active river corridor shifted back and forth laterally by hundreds of meters. These rates and styles of channel change are consistent with other studies of Pacific Northwest riparian landscapes in showing major spatial and temporal heterogeneity (Fetherston et al. 1995, Abbe and Montgomery 2003, Beechie et al. 2006, Latterell et al. 2006).

All four rivers widened significantly ($p \leq 0.05$) after having been relatively narrow in the 1960s–1970s (Figure 4.1.25; see also East et al. 2017). Widening thus corresponded temporally with the greater flood activity of the past several decades, although the Elwha River also widened in response to a landslide-induced sediment pulse, discussed below. The Hoh River showed a significant increase in channel braiding over the aerial photographic record (Figure 4.1.25A); the other rivers showed no trend in the degree of braiding.

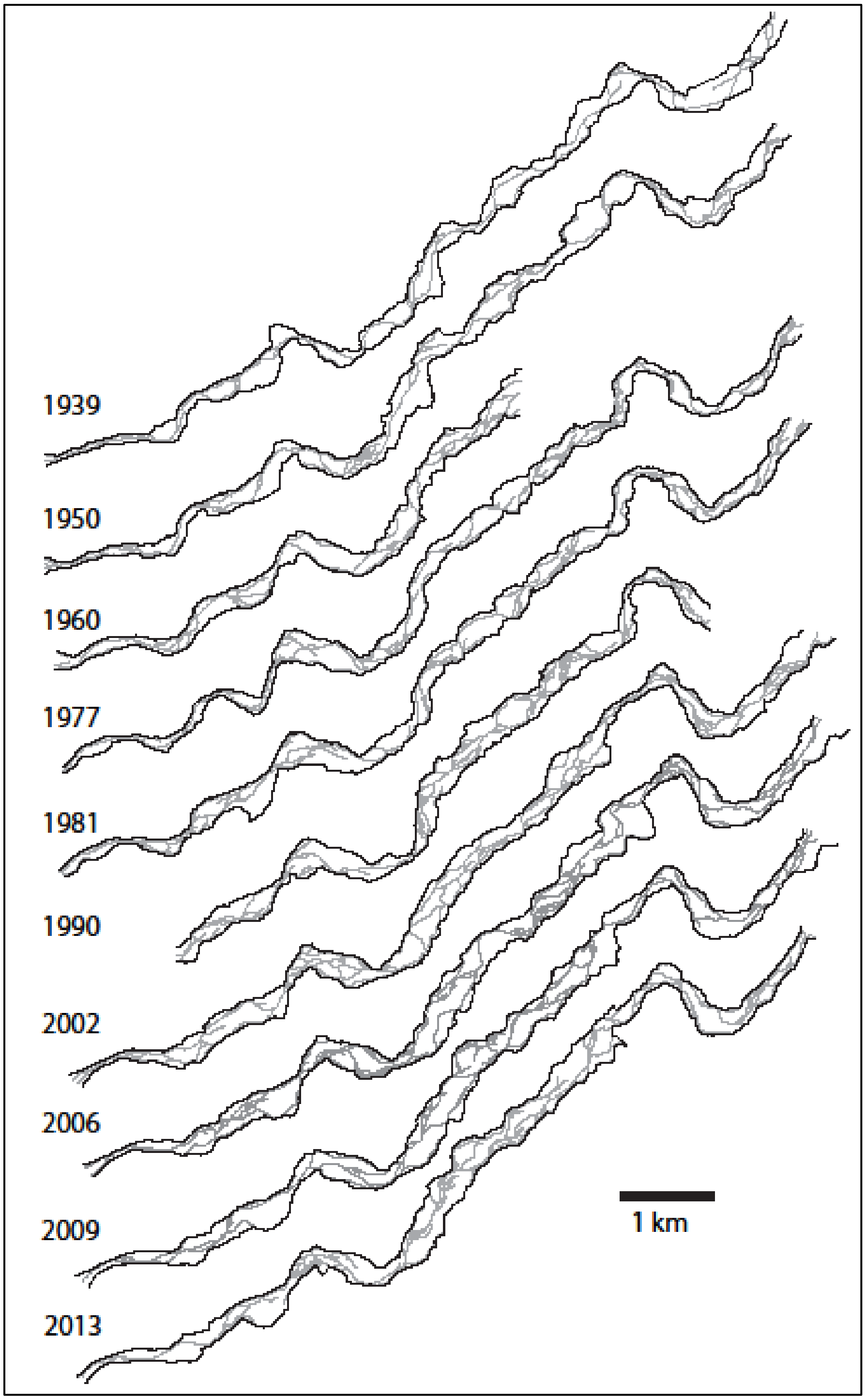

Figure 4.1.19. Digitized margins and braids of the Hoh River, 1939-2013 (gray, outlines of channel midlines; black, margins of recently active flow zone). Source: East et al. 2017.

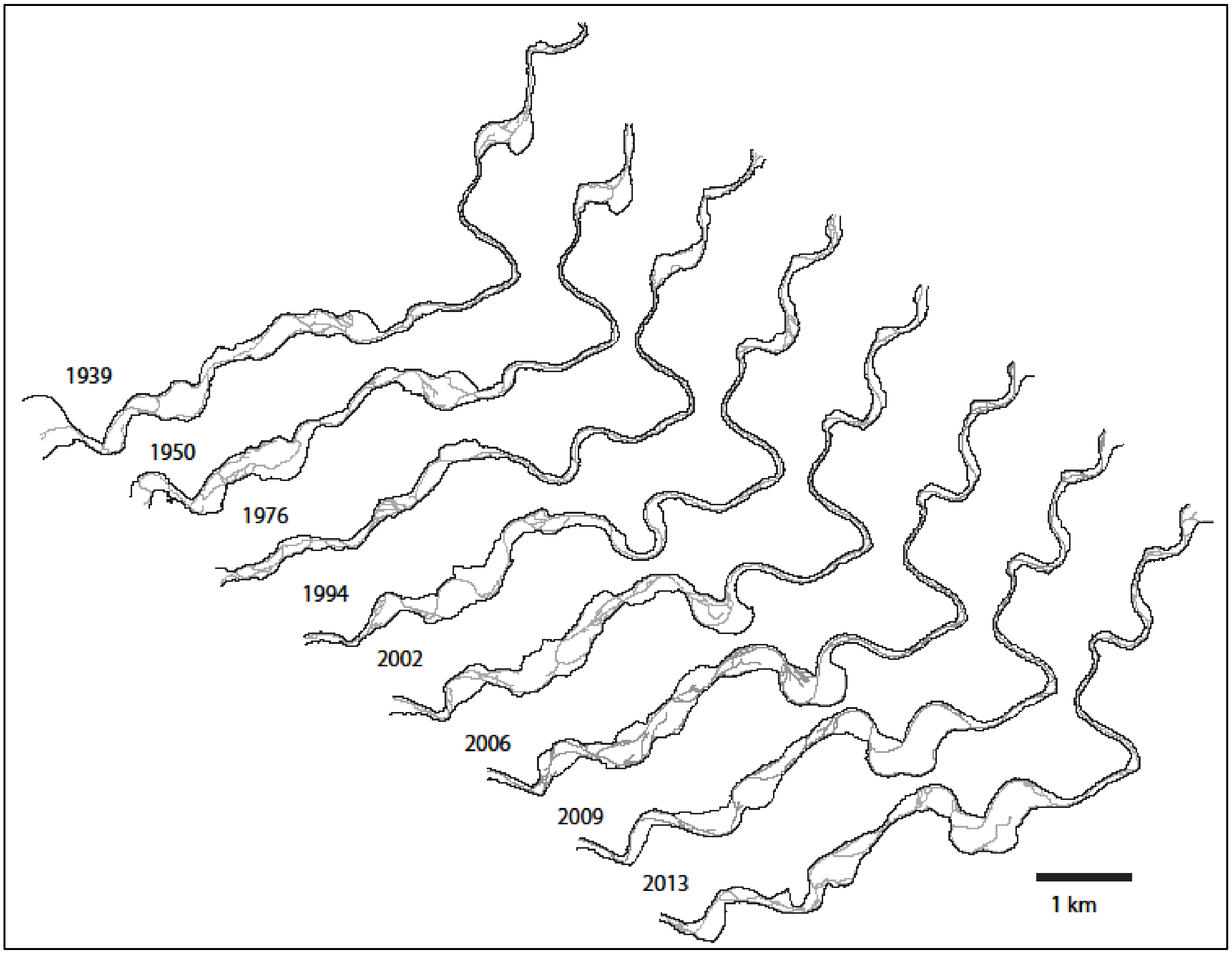

Figure 4.1.20. Digitized margins and braids of the Queets River, 1939-2013 (gray, outlines of channel midlines; black, margins of recently active flow zone). Source: East et al. 2017.

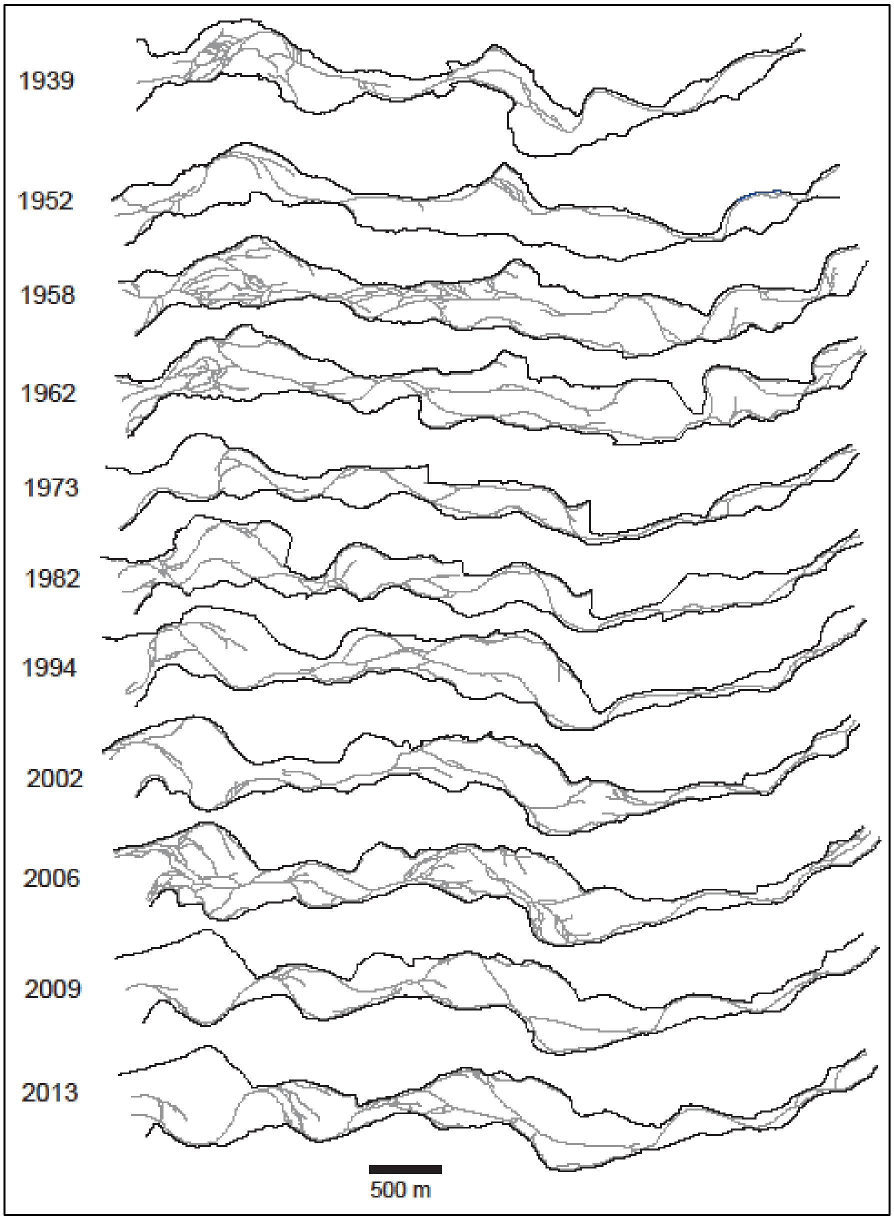

Figure 4.1.21. Digitized margins and braids of the Quinault River, 1939-2013 (gray, outlines of channel midlines; black, margins of recently active flow zone). Source: East et al. 2017.

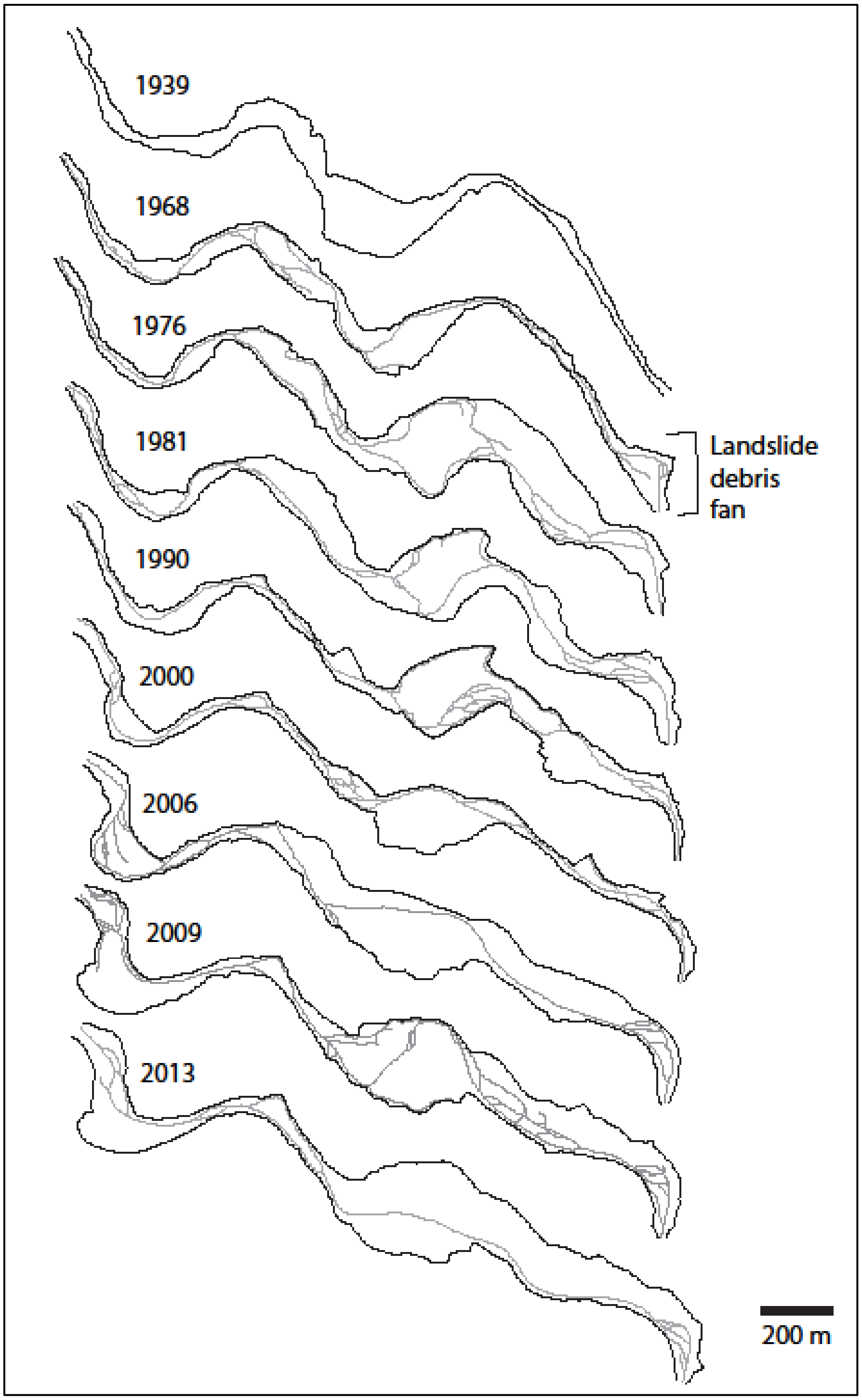

Figure 4.1.22. Digitized margins and braids of the Elwha River, 1939-2013 (gray, outlines of channel midlines; black, margins of recently active flow zone). The 1939 images did not have sufficiently high resolution to show braids. Landslide-debris fan is indicated at upstream end of reach in 1968, composed of reworked material from a 1967 landslide. Source: East et al. 2017.

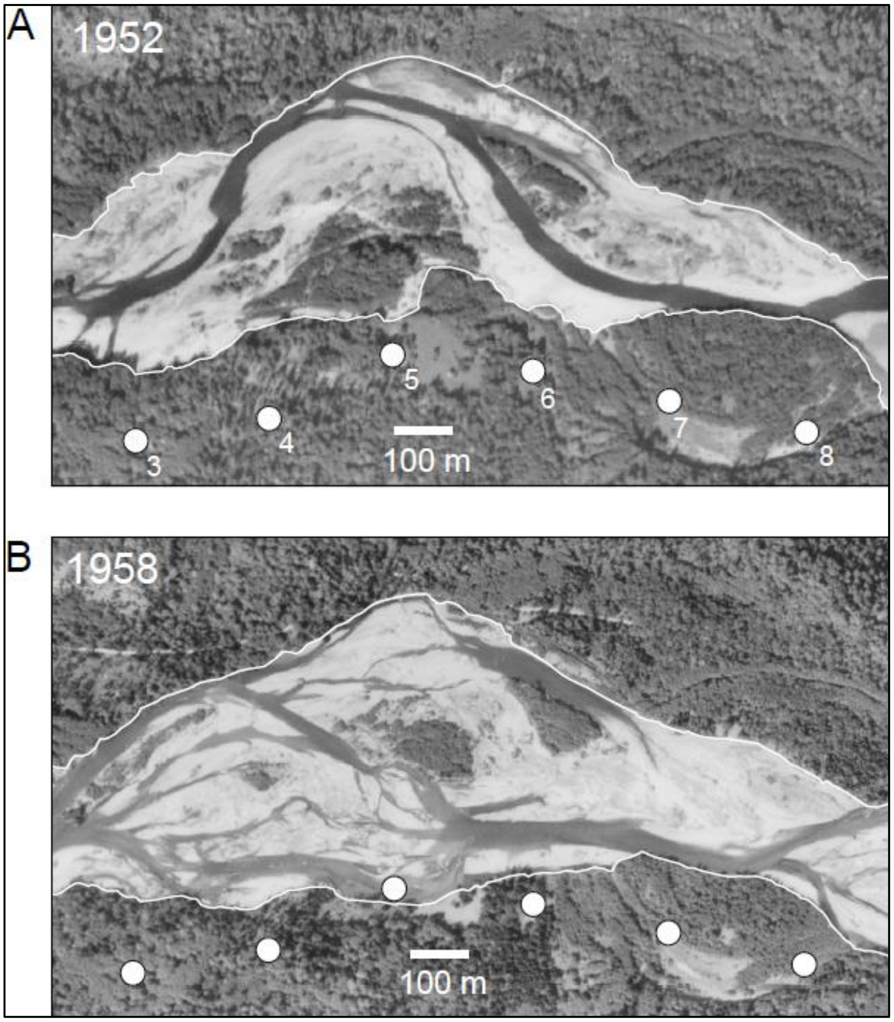

Figure 4.1.23. Photographs from the Quinault River study reach showing an example of disturbance, with active channel widening and becoming more braided between (A) 1952 and (B) 1958, an interval that encompassed the second-highest flood peak on record for that river (an event slightly below the Q50 value). White circles show points corresponding to transects 3–8 (Figure 4.1.16C); circles indicate approximate center of active river corridor as of 2013. White lines show the margins of the inferred recently active flow zone. Source: East et al. 2017.

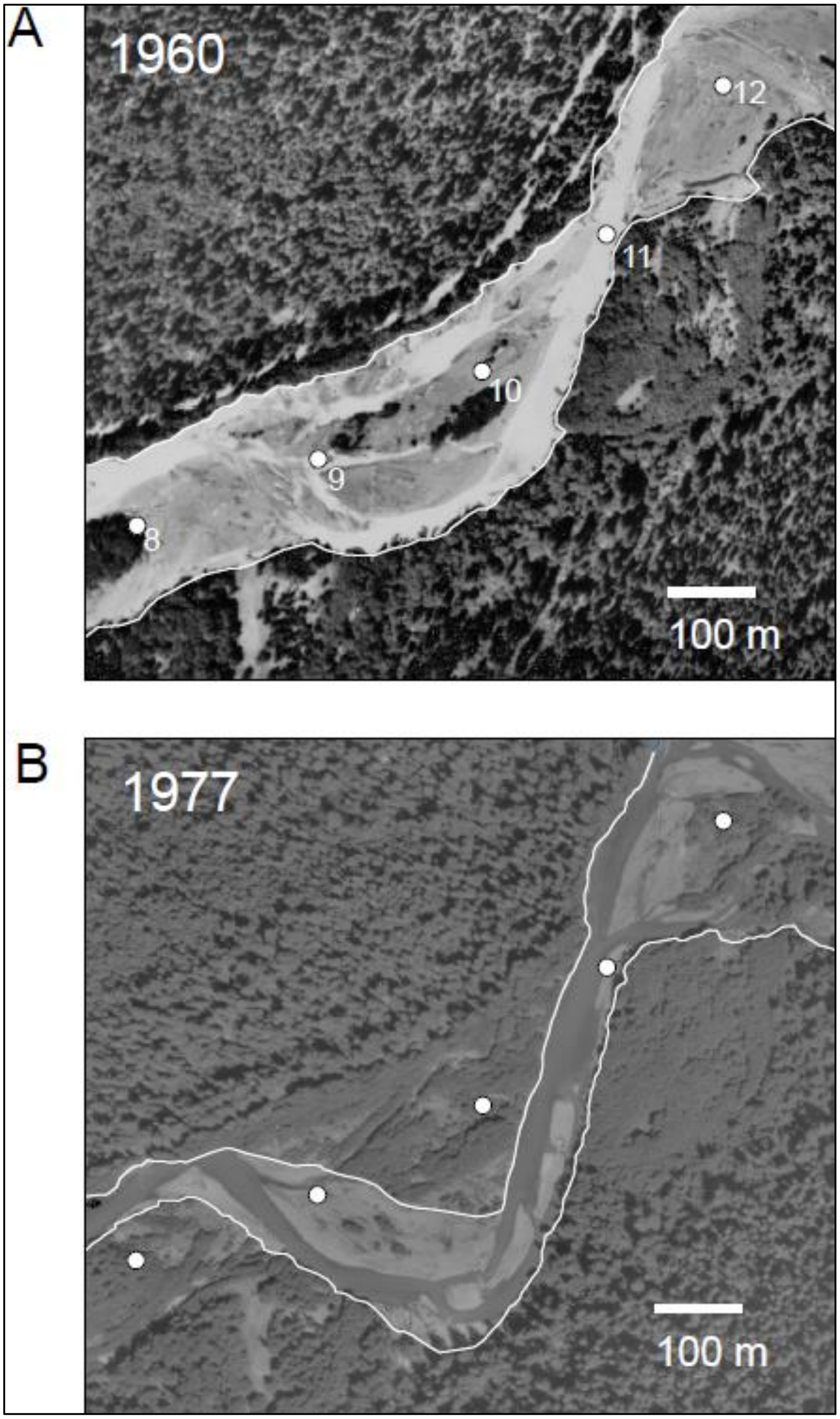

Figure 4.1.24. Photographs of the Hoh River study reach showing narrowing and simplification (decreased braiding) between (A) 1960 and (B) 1977. White circles show points corresponding to transects 8–12 (Figure 4.1.16A); circles indicate approximate center of active river corridor as of 2013. Source: East et al. 2017.

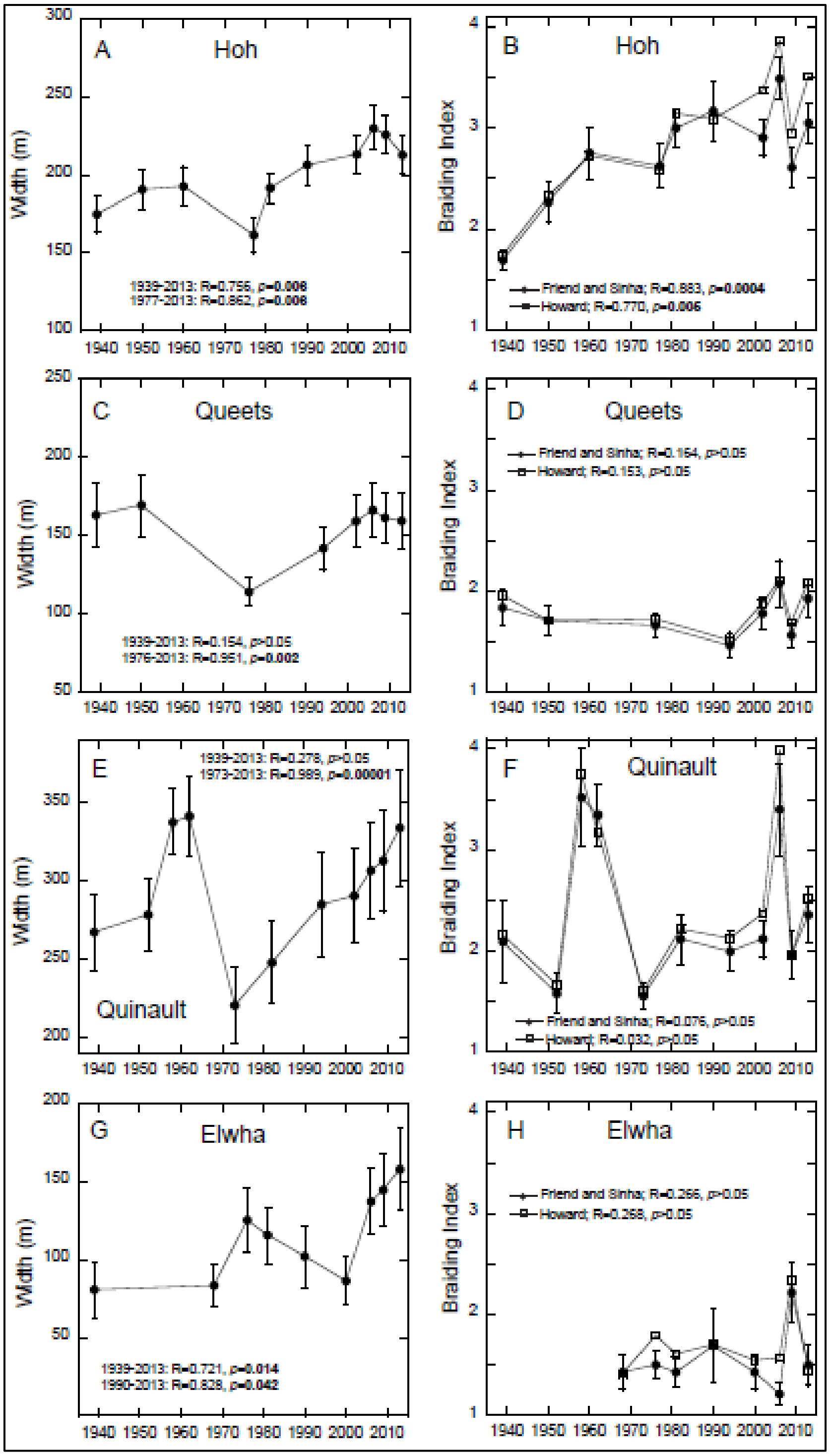

Figure 4.1.25. Mean width of the recently active flow zone, and braiding indices of Friend and Sinha (1993) and Howard et al. (1970), for study reaches on the Hoh River (A, B), Queets River (C, D), Quinault River (E, F), and Elwha River (G, H). The legend for braiding indices in (B) also applies to (D), (F), and (H). Error bars indicate standard error of the mean. R and *p* values on each plot were determined by Student's *t* tests on linear regressions for the temporal spans indicated. Source: East et al. 2017.

Hoh River

The Hoh River record included a decrease in channel width and braiding in 1977 relative to 1960 (Figures 4.1.19, 4.1.24, 4.1.25A, B), which interrupted the significant multidecadal trends toward increased width and braiding of this reach. That interval corresponded with low peak-flow conditions culminating a relatively quiescent flood regime of the late 1960s and early 1970s (the Hoh River flow during winter 1976–77 contained the lowest annual flow peak in the Hoh discharge record; Figure 4.1.18A). Thus, although that year was otherwise hydrologically normal, with near-average spring snowmelt flows, the lack of channel disturbance by winter flood events was unusual, and abundant vegetation growth narrowed the active river corridor. Width and braiding increased significantly during subsequent decades with higher peak flows (Figure 4.1.25A, B).

Notes from a GLO cadastral survey of the Hoh River in 1919 include width measured at one location within our study reach where the survey crossed orthogonal to the river corridor (BLM 2016). That measured width of 77 m was narrower than measured at the same location on the 2013 aerial photographs (133 m), although the 1919 measurement was within the range of spatial and temporal variability for this reach that we had measured in the aerial photographic record.

The three highest flood peaks recorded on the Hoh River have occurred since 2002 and include three events exceeding the calculated 10-year flood peak (Figure 4.1.18A). We measured the greatest width and braiding on the 2006 images (Figure 4.1.25A, B), which followed the flood of record by less than three years. Beginning in the fall of 1977, when the onset of a wet PDO phase signaled a widespread climatic shift over much of the Pacific region (Castino et al. 2016), the 2-year flood peak calculated for the Hoh River for water years 1978–2013 was 1024 m^3/s, whereas the 2-year flood for the entire period of record at gaging station 12041200 was 924 m^3/s.

Queets River

Mean channel width in the Queets River study reach decreased by 33% between 1950 and 1976 (Figure 4.1.20, 4.1.25C). Because discharge data are missing over much of that interval (Figure 4.1.19B), we cannot establish clear temporal correspondence between flow and channel changes. However, if the Queets River flow history behaved similarly to that of the other rivers in having had a relatively quiescent flood regime in the late 1960s and early 1970s, then results would be consistent with narrowing of the active flow zone over that time step.

A 1906 GLO survey of the Queets River measured channel width at three transects orthogonal to the river corridor within our study reach (BLM 2016). Those measurements of 74–138 m are similar to or slightly narrower than at the same locations in 2013 (74–172 m). However, they are within the range of variability that we measured in this reach throughout the aerial photographic record.

The flood of record for the Queets River occurred in December 1999 (a peak slightly above the 50-year flood value; Figure 4.1.18B). We measured increased width and braiding, and major channel reorganization, over the 1994–2002 photographic interval spanning that flood (Figure 4.1.25C, D). Additional braiding fluctuations occurred in the late 2000s, although width was essentially constant after 2002. The 2-year-flood peak value for 1978–2013 at station 12040500 was 2131 m^3/s, whereas the two-year peak calculated for the whole period of record is 1902 m^3/s.

Quinault River

Between 1952 and 1958 the mean active-channel width of the Quinault River study reach increased by 18%, concurrent with substantially increased braiding (Figures 4.1.21, 4.1.23, 4.1.25E, F). That interval included a November 1955 flood that was the second-highest peak in the Quinault discharge record, slightly below the 50-year flood (Figure 4.1.18C). Width and braiding decreased substantially thereafter, dropping by 35% between 1962 and 1973. Width and braiding reached the lowest recorded values for this river during this time due to vegetation encroachment and abandonment of several channels in the upper 3 km of the reach (Figure 4.1.21). That channel narrowing and simplification accompanied relatively quiescent flows—no flows approached the 10-year flood between 1962 and 1973, in contrast to larger floods before and after that interval (Figure 4.1.18C). The Quinault channel became significantly wider after 1973, accompanied by generally larger floods; the two-year flood value for 1978–2013 was 808 m^3/s, whereas the two-year flood for the entire Quinault River record is much lower at 595 m^3/s. We identified a short-lived braiding increase between 2002 and 2006 accompanied by a modest width increase (Figure 4.1.25E, F); this coincided with a 10-year flood in October 2003.

A 1908 GLO Quinault River survey included four width measurements within our reach that were orthogonal to the river corridor (BLM 2016), ranging from 163 to 354 m. These generally agreed with widths at those locations on the 2013 aerial photographs (231–334 m), and were within the range that we measured in the Quinault reach throughout the aerial photographic record.

The Quinault reach is wider than the Hoh and Queets reaches (which have comparable gradient), despite the discharge being less than that of the Hoh or Queets Rivers. Our median reach-averaged width value for the Quinault over the photographic record (299 m) was ~30% greater than on the Queets (227 m) and 50% greater than on the Hoh (200 m). We did not detect significant differences in braiding between the Quinault reach and those on the Hoh or Queets Rivers.

A legacy of anthropogenic physical alterations appeared to affect the Quinault River (Bountry et al. 2005, Herrera Environmental Consultants Inc. 2005). The wider active flow zone on the Quinault compared to that of the Hoh and Queets Rivers is, we suspect, most likely due to logjam and tree removal and other riverbank disturbances decades ago contributing to a more mobile channel than might have persisted without the wood removal. The other three study reaches were affected by such activities much less or not at all (Bountry et al. 2005, Latterell and Naiman 2007). A cursory GIS analysis of the 2013 aerial images conducted as part of this study shows that the Quinault reach still has less large wood than the other rivers—1.9% by area, compared to 2.7% on the Hoh, 2.5% on the Queets, and 5.4% on the Elwha reach (the Elwha, being smaller and steeper, is expected to have proportionally more in-stream wood than the other rivers).

Elwha River

Some of the measured changes in the Elwha River study reach corresponded to known landslide activity. All four watersheds have active landslides (QIN 1999, Lyon 2003, Godaire and Kimbrel 2015), but the Elwha study reach is the only one affected by landslide debris entering the mainstem channel immediately upstream from the study reach (350 m upstream, in a bedrock canyon; another landslide occurs 2.8 km upstream of the reach). As described by Tabor (1987) and Acker et al.

(2008), the landslide 350 m upstream from Geyser Valley dammed the river temporarily in 1967. Reworked material from the failed landslide dam had formed a 25-ha debris fan in the upper part of our study reach as of 1968. This substantially widened the upstream-most part of the reach between 1939 and 1968, although the new sediment had not moved far enough downstream by 1968 to increase mean width over the entire reach (Figures 4.1.22, 4.1.25G, H). Over subsequent decades the river reworked the landslide debris into several surfaces of different elevation; the debris fan directed the river more toward the right bank, further increasing the sediment and wood load added to the river (Acker et al. 2008). The reach-scale width increase as of 1976 relative to 1968 is attributable to progression of the new landslide sediment downstream. The upper half of the reach accounted for most of the increased width and braiding over those 12 years (Figure 4.1.22). Any changes in the Elwha channel associated with quiescent flow periods in the early 1970s (as observed in the other river corridors) would have been obscured by the landslide sediment pulse dispersing through the reach.

The second major channel change evident on the Elwha reach occurred after 2000, when reach-averaged width nearly doubled (from 86 m to 157 m between 2000 and 2013), affecting all parts of the reach (Figure 4.1.25G). Within that transition to a wider reach, we measured a distinct short-lived braiding increase between 2006 and 2009 that coincided with a 50-year flood in December 2007 (Figures 4.1.18D, 4.1.25H). Field topographic surveys in lowermost Geyser Valley corroborated the changes visible in aerial images and indicated that the new channels formed between September 2007 and April 2008 (Draut et al. 2011). Such close temporal correspondence between the channel changes observed in 2008 with the 50-year 2007 flood suggests a direct causal relationship. In the 2–3 years following that flood, lower Geyser Valley gradually returned to having one to two active threads (Figure 4.1.22; Draut et al. 2011, East et al. 2015). In general the widening of the Elwha reach in recent decades is consistent with greater flood activity compared to earlier in the record; the 2-year-flood value for 1978–2013 on the Elwha River was 477 m^3/s, whereas the two-year-flood value for the entire period of record is 421 m^3/s.

A 1911 GLO survey recorded width orthogonal to the river corridor at one location within our Elwha River study reach, measuring 64 m where we measured a width of 72 m on the 2013 aerial photographs. The 64-m width is within the range of spatial and temporal variability throughout the aerial photographic record.

What are the Primary Drivers of Channel Change?

Rivers are fundamentally shaped by both the water and sediment that they carry. The combination of high flows and available sediment is especially important in determining channel morphology: sediment loads typically increase exponentially with flood magnitude (Leopold et al. 1964). In the Pacific Northwest, annual sediment loads during years with large floods are commonly an order of magnitude greater than annual loads during years with modest peak flows (e.g., Major 2004, Czuba et al. 2012). In this analysis, we demonstrated relationships between channel evolution and flood history on the four study reaches in OLYM, and suggest mechanisms where sediment supply may have influenced observed dynamics.

Strong statistical correspondence between recent flood history and channel width and braiding implied that study reaches on the Hoh, Queets, Quinault, and Elwha Rivers respond primarily to flow forcing, including both floods and relatively quiescent periods. Intervals with large floods generally were accompanied by widening of the active flow zone and increased channel braiding, whereas intervals with smaller recent peak flows corresponded to narrowing and decreased channel braiding. The significant widening of all four rivers since the mid-1970s is attributable to the intensification of regional flood activity since that time (Mantua et al. 1997, Piety et al. 2004, Bountry et al. 2005, Czuba et al. 2012).

River width and braidedness are also sensitive to sediment supply, with increased supply commonly promoting bed aggradation (increased bed elevation), which leads to greater channel width and braiding (Smith and Smith 1984, Hoffman and Gabet 2007, Podolak and Wilcock 2013). Although river sediment load has not been measured on these rivers over much of the historical period, we surmise that Olympic river morphology is sensitive to sediment supply based on several distinct opportunities to observe channel response to increased sediment supply on the Elwha River. The increased width and braiding through the Geyser Valley reach of the Elwha River in the late 1960s and 1970s accompanied dispersion of a landslide-induced sediment pulse (Acker et al. 2008). A similar, but larger, response occurred on a downstream section of the Elwha River beginning in 2012 during a sediment pulse from two large dam removals, which was accompanied by a temporary increase in channel braiding of ~50% (East et al. 2015).

We suggest that the significant multidecadal trend toward greater braiding of the Hoh River may be attributable to increased sediment supply caused by glacial retreat in the upper watershed. OLYM glaciers have retreated rapidly over recent decades, concurrent with regional warming (Rasmussen and Conway 2001, Malcomb and Wiles 2013). Since 1980, OLYM has lost 34% of its glacial ice area and 82 glaciers have disappeared entirely (Riedel et al. 2015), exposing unconsolidated, unvegetated sediment high in the watersheds. The Hoh watershed contains by far the greatest proportion of Olympic Mountains glacial ice volume (65%), and the basins with the next-most have much less: the Elwha has 11%, the Queets has 9%, and the Quinault and several other basins each have <5% (Riedel et al. 2015). Given its high proportion of high alpine terrain, the Hoh River could be particularly sensitive to potentially increased sediment supply associated with high-altitude warming, including new sediment resulting from glacial retreat, shrinking perennial snow fields, melting of permafrost, and mass wasting of recently deglaciated valley walls (see synthesis assessment by Lane et al. 2017, of changes in watershed sediment flux following alpine glacier recession). Sediment-flux measurements from the Hoh River would be required to confirm or refute this explanation with certainty.

Previous studies provide evidence that enhanced bar and braid formation in rivers may reflect sediment-supply increase driven by glacier recession. Increased sediment load during rapid glacier recession has been documented from lake cores in British Columbia (Leonard 1997, Menounos and Clague 2008, Schiefer et al. 2010), and bar development in gravel-bed channels is known to vary with bedload flux (e.g., O'Connor et al. 2014). Several studies have demonstrated that proglacial sediment deposits can be an important (though temporally variable) sediment source to downstream rivers, contributing 60–80% of the fluvial suspended sediment during some ablation (melting) seasons in British Columbia (Orwin and Smart 2004, Leggat et al. 2015). Warburton (1990) found that 23% of fluvial sediment export from a glaciated catchment in Switzerland originated from proglacial sediment sources. Less is known, however, about how sediment export from recently deglaciated areas affects downstream river morphology. One study has shown increased river width and braiding downstream from retreating glaciers in British Columbia (Miles and Allegretto 2011). Finally, Czuba et al. (2012) investigated rivers that drain retreating glaciers on Mt. Rainier, Washington, and found that channel width correlated with the proportion of sedimentary debris mantling the glacier; their study did not measure braiding. We cannot explain the increased braiding on the Hoh River by pointing to excessive sediment supply from another source such as channel widening and bank retreat, as our GIS analysis shows that the Hoh has actually widened (i.e., eroded its banks) proportionally less than the Quinault in recent decades and to a similar degree as the Queets, neither of which shows a significant braiding increase.

The aerial photographic record clearly indicated that the study reaches on the Hoh, Queets, Quinault, and Elwha Rivers were braided at the time aerial photographs were first taken in 1939. Beschta and Ripple (2008) raised the possibility, however, that river planform may have been affected by high elk populations that preceded the aerial photographic record. Is the modern sensitivity of channel planform to flood activity unnatural and recent, having developed only since the early 20th century? Evaluating these questions is complicated by acceleration of glacial retreat in western North America in the 1920s–1940s (Spicer 1989, Leonard 1997, Menounos 2006, Koch et al. 2007, Malcomb and Wiles 2013), which might have increased sediment supply and thus the degree of channel braiding at that time. However, even in light of these complications, our examination of the historical record prior to 1939 does not support the idea that a fundamental shift in river morphology occurred prior to the photographic record.

Accounts from early explorers and settlers are equivocal as to whether OLYM river morphology was fundamentally different in the late 1800s and early 1900s than today. Because early explorers and photographers did not intend to document fluvial geomorphology explicitly, it is seldom clear whether narratives refer to single- or multi-thread channels, or whether oblique historical photographs show the only extant channel or one of several. However, the Press Expedition of 1889–1890 referred to islands and side channels on the Elwha and Quinault Rivers (Wood 1967), the 1878 Watkinson Expedition mentioned the upper Quinault River having wide gravel bars (Lien 2001), and 1891–1895 surveys of the Hoh River below our study reach mapped bars and braids (GLO 1896). Journal entries from the 1890s (published a century later by Morganroth 1991) described dynamic western Olympic Peninsula rivers, with broad gravel bars and channels that “meander from year to year from one side of [the] banks to the other.” Other early records mention narrow, vegetation-choked channels where, today and in the recent past, a river corridor has substantial unvegetated width (Bountry et al. 2005, Beschta and Ripple 2008). In rivers as spatially and temporally variable as these, though, a large difference in unvegetated width observed at two long-separated points in time may be real but not indicative of permanent change or long-term trends. Some early photographs also may represent localized observations where a channel was especially difficult to navigate by boat, rather than spatially averaged geomorphic conditions—expedition documents and photographs sometimes focused deliberately on the greatest challenges of the journey. If channels were indeed generally narrow and vegetated around the turn of the twentieth century, the cause may have been a quiescent flood regime, analogous to the early 1970s, but without sufficiently long discharge records it is impossible to be certain. The GLO surveys of 1906–1919 indicated that widths in the study reaches we examined were within the range represented by the aerial photographic record.

If OLYM rivers had undergone a major widening and transition to greater braiding over two decades between the start of elk population growth (1910s) and the first aerial photographs (1939), as Beschta and Ripple (2008) suggested, such a geomorphic adjustment presumably would have involved substantial erosion, sediment export, and corresponding aggradation in the lower portions of the river corridors as a sediment pulse dispersed (Jacobson and Gran 1999). The stage–discharge history for the stream-gage sites on the four rivers we studied did not show a coherent, substantial stage increase before 1939 to suggest major aggradation (East et al. 2017). On the Hoh River, stage at a discharge of 226 m^3/s fluctuated negligibly (within a range of 0.14 m) between 1926 and 1950. The Queets gaging station showed minor bed aggradation in 1935 (a stage increase of 0.33 m at 991 m^3/s) immediately following a 50-year flood, but stage then decreased by 0.2 m over the next decade. The Quinault record showed stage at 283 m^3/s varying by no more than 0.1 m between 1911 and 1950 (below Lake Quinault). The Elwha record showed minor scour and fill (within a 0.32-m range) between 1918 and 1927 with no directional trend. Even though sediment-trapping lakes limit applications of these stage histories on the Quinault River and (after Glines Canyon Dam was built in 1927) on the Elwha River, there does not appear to be evidence for a system-wide, major fluvial sediment pulse to support an interpretation of a fundamental geomorphic shift shortly before the start of aerial photography. In contrast, the Elwha gaging station showed aggradation of 1.3 m when a major sediment pulse passed through in 2012–2013 after dam removal (East et al. 2015).

General theory of river geomorphic development strongly suggests that rivers such as the Hoh, Queets, Quinault, and Elwha Rivers are likely to be braided rather than single-thread. Comparing gradients and discharge records for the four study reaches to threshold criteria for braided river systems indicates that Olympic rivers are well within the ranges of values typical of braided-channel development (Figure 4.1.26; Leopold and Wolman 1957, Ferguson 1987, Knighton and Nanson 1993, Van den Berg 1995, Eaton et al. 2010, Mueller and Pitlick 2014). Based on these empirically derived common characteristics of braided rivers, it is unlikely that these river reaches were ever single-thread, as that would require a reduction of river gradient (slope) by an order of magnitude, or a discharge reduction by 1–2 orders of magnitude.

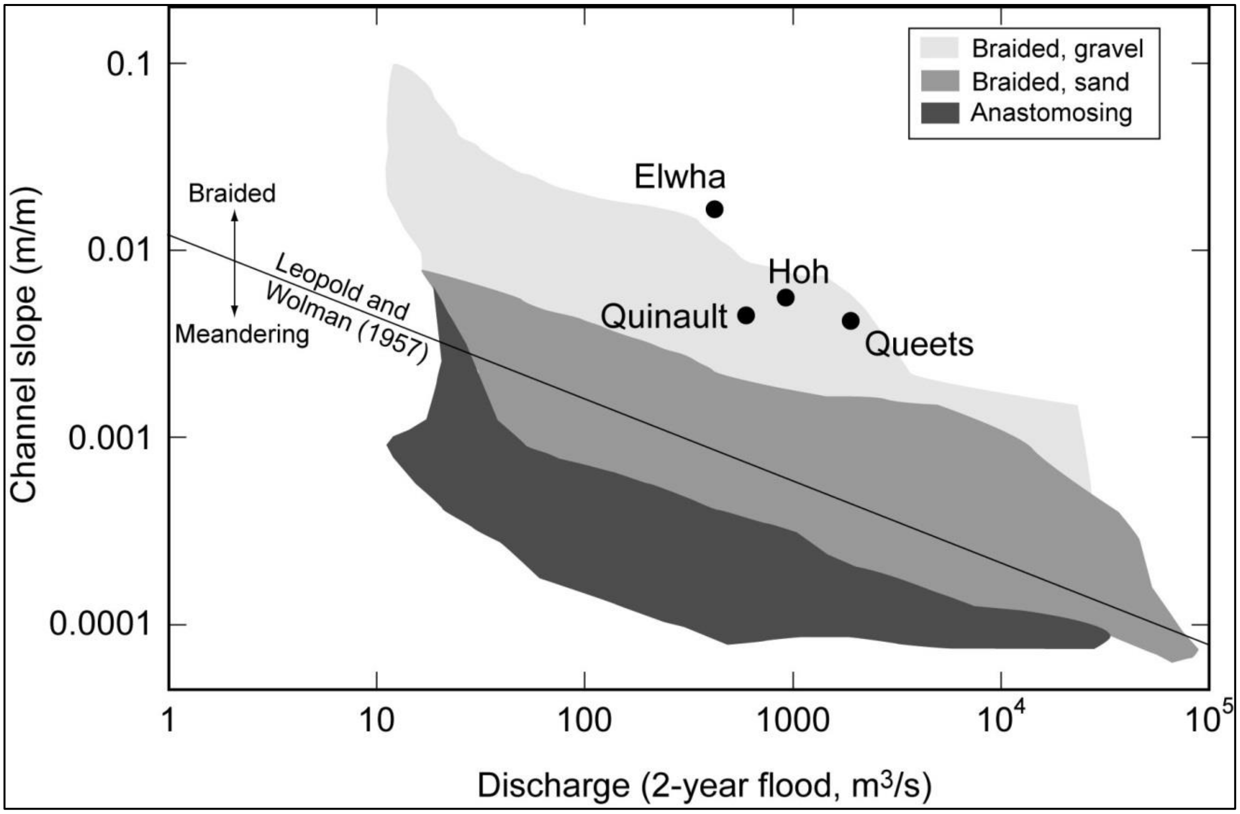

Figure 4.1.26. Slope–discharge plot showing the four Olympic National Park study reaches compared to channel patterns defined by previous empirical studies. Gray-shaded regions indicate where most data fall for braided gravel-bed rivers, braided sand-bed rivers, and anastomosing rivers (Ferguson, 1987; Knighton and Nanson, 1993). The diagonal line between braided and meandering channels shows the boundary defined by Leopold and Wolman (1957).

Uses and Limitations

The interactions among physical and ecological processes that determine river morphology and evolution are complex. Inherent randomness and feedback interactions among flow, sediment, and ecosystems mean that there are no easy answers to the question of what factors drive channel change, and whether patterns of change have departed from the natural range of variability.

Our analyses of the 74-year photographic record provided clear evidence that river planform change responds rapidly to changes in river flow and suggests that changes in braidedness may reflect sedimentary changes associated with glacier recession. Hence, the assessment of river width and braiding, alongside the examination of flood history, provides a baseline condition of rivers today that will be useful for examining future trends in river planform as well as factors driving future changes.

Our inferences have been constrained by limitations in the existing data. For example, we could not evaluate the potential influence of glacier recession on channel change via changes in sediment supply without direct sediment-load measurements and recent repeated topographic measurements from the upper portions of the Hoh watershed. Our inference of no fundamental shift in river morphology in the first few decades of the 20th century (before aerial photographs were taken in 1939) is based on inference from indirect sources of information: stream-gage records, written historical accounts, a limited number of early oblique ground-based photographs, and maps that depend in part upon the surveyors' interpretations of channel morphology or their intended resolution. Our inferences from those sources of information, although as robust in this report as we consider reasonable given the data quality, are limited by the lack of aerial imagery pre-dating the loss of wolves and population increase of elk.

Emerging Issues

The evolution of the OLYM river systems is likely to include additional widening and increased braiding over multidecadal time scales if flooding intensifies in the future, as predicted by climate-change and hydrologic models. Predictions of 21st-century climatic change for this region include fewer years with large snowpack, more precipitation falling as rain rather than snow in response to cool-season warming, and short-term intense rainfall causing more-frequent winter floods (Jakob and Lambert 2009, Mantua et al. 2010, Tohver et al. 2014, Lute et al. 2015, Vano et al. 2015). Models show that mid- to high-elevation regions of the Olympic Mountains will be especially prone to future increased flooding (Tohver et al. 2014), though presumably these effects would be modulated by annual to interdecadal cyclicity (PDO and El Niño-Southern Oscillation, ENSO). Additional hydrologic alterations are anticipated from glacial retreat, especially to late-summer streamflow (Riedel et al. 2015).

These climatic and hydrologic changes could increase watershed sediment yield in the Olympic Mountains (that is, the amount of sediment produced per unit watershed area). Sediment yield is likely to increase not only due to the loss of glaciers, perennial snow fields, and permafrost in high-altitude headwaters as a result of long-term warming (e.g., Micheletti and Lane 2016, Lane et al. 2017), but also due to increased landslide activity as a result of greater rainfall (Jakob and Lambert 2009). Studies of Olympic Mountains landslide frequency had not yet detected an increase in landsliding as of the late 1990s and early 2000s (QIN 1999, Lyon 2003), although we anticipate seeing additional landsliding activity in the future. The widening of OLYM river channels, which we attribute largely to their sensitivity to more intensive flooding and possibly increased sediment supply under a warmer 21st century climate, could exacerbate the erosion of ancient, high-elevation river terraces, a phenomenon noted by Beschta and Ripple (2008). The rate of such terrace loss will be highly variable in time and by location, determined by the dynamics of individual flood flows and local interactions of flow and large wood. Additional widening of river channels could potentially destabilize park roads and other infrastructure situated on river floodplains and alluvial terraces.

Information Needs/Gaps

Influence of glaciers and glacier retreat on riverine processes

Evaluation of the rates and locations of glacier recession in OLYM upper watersheds would allow better assessment of the role of high-elevation warming and glacier loss in shaping riverine processes. Temporal change analyses such as that of Riedel et al. (2015) are of great value for identifying and quantifying the rates, styles, and locations of glacier recession under the current rapidly warming climatic conditions. The loss of glaciers, perennial snow fields, and permafrost as climate warms has important implications not only for environmental changes occurring at high mountain elevations, but also for streamflow hydrology and sediment supply to these river corridors (e.g., Micheletti and Lane 2016), especially to the Hoh watershed. The glacial hydrologic contribution to OLYM rivers is particularly important in late summer (Riedel et al. 2015). Although sediment export from recently deglaciated areas is demonstrably important in some high alpine terrain (Leggat et al. 2015, Lane et al. 2017), distinguishing recent, contemporary glacial and proglacial sediment input would be difficult without spatially intensive monitoring, because rivers also access sediment from previous glacial occurrences as well as landslide deposits and other sediment sources (Orwin and Smart 2004).

Suspended-sediment concentrations

At present, suspended-sediment concentrations are not measured directly in the upper reaches of OLYM rivers. Initiating a sampling program to quantify suspended- or bed-sediment load over time in the upper Hoh River would provide valuable data for evaluating future effects of climate change, particularly flooding, landslides, glacier erosion, and channel planform.

Mass-wasting/landslide events

Continued monitoring of mass-wasting events (i.e., the downhill movement of soil and rock through landslides and smaller mass movements), such as through the current NPS landscape dynamics monitoring program (Antonova et al. 2010) or expanded similar efforts would also be useful. Additional quantitative information on mass-wasting occurrence would enhance future assessments of ongoing and landscape reorganization driven by climate change in the Olympic Mountains (Singh et al. 2015, Schildgen et al. 2016). Digital surface modeling methods (e.g., airborne LiDAR, see below) would be particularly useful for assessing mass-wasting influences on OLYM river dynamics.

Continued recording of streamflow

Recording streamflow at USGS gaging stations is of high value for identifying temporal fluctuations in river discharge and flow forcing—the dominant driver of changes in river channels.

Continued collection of digital, aerial orthoimages, and airborne LiDAR topography

These data types collected at regular intervals (<5 years for aerial photography, at least every decade for LiDAR-based digital elevation models) would provide valuable information that can be used in the future to assess geomorphic evolution as well as changes in forest and riparian vegetation. Airborne LiDAR topography would allow for detailed quantitative measurements of topographic changes in proglacial regions, to determine whether effects of glacial retreat and warming are indeed causing geomorphic changes that release additional sediment downstream. LiDAR-based terrain models could be used to detect, e.g., erosion of proglacial sediment, valley-wall mass wasting, or permafrost degradation in high alpine areas that ultimately affect river morphology and riverine ecosystems and resources downstream.

Resource Summary

We identify the physical condition and attributes of OLYM river channels as changing significantly—due to natural climatic cycles superimposed on long-term, human-driven climatic changes, including warming of high-alpine terrain (Table 4.1.6). However, although we judge these changes to be important, we do not ascribe those changes or trends to represent deteriorating conditions or loss of "integrity" *per se*. Rather than representing an improvement or deterioration in conditions, they may simply represent variations in natural processes that reflect variations in flood magnitude and frequency. These findings of recent, annual- to multidecadal-scale change are judged to be of moderate to high confidence: if the precipitation and temperature regimes of the Pacific Northwest evolve as models now predict over the long term, which would in turn likely trigger increased landslide activity and glacier recession, we have high confidence in our projections of future trends in riverine processes.

Table 4.1.6. Resource condition summary for rivers in Olympic National Park.

Resource	Indicator	Status and Trend	Rationale and Reference Conditions
Hoh	Width	🟡	Fluctuates considerably; has widened significantly since the 1970s
	Braidedness	🟡	Increased braidedness, likely caused by increased bedload associated with glacial retreat
Queets	Width	🟡	Fluctuates considerably; has widened significantly since the 1970s
	Braidedness	🟢	No change in braidedness
Quinault	Width	🟡	Fluctuates considerably; has widened significantly since the 1970s
	Braidedness	🟢	No change in braidedness
Elwha	Width	🟡	Fluctuates considerably; has widened significantly since the 1970s
	Braidedness	🟢	No change in braidedness
Overall condition and trend of Rivers		🟡⇔	All rivers have widened over time, and the Hoh river showed increased braidedness, but we do not ascribe these trends as representing improved or deteriorating conditions.

4.2. What are the effects of resource harvesting for human use on ecosystem function and sustainability of the resource?

Olympic National Park exists as an island of protected land on the Olympic Peninsula, surrounded by multiple jurisdictions of managed lands and ultimately by the Pacific Ocean, Strait of Juan de Fuca, and Puget Sound. Within this geographical context, OLYM is tasked with conserving and managing aquatic resources that are harvested both inside and outside of the park's boundaries, and that spend much of their life cycle in the ocean, where they are subject to a host of other natural and anthropogenic pressures. In this section, we address two harvested resources of concern to the park. First, we consider the condition of Pacific salmonids in the park (Chapter 4.2.1). The park contains key freshwater habitat for Pacific salmonids where they are protected from harvest, have productive conditions for spawning and rearing, and provide critical ecological functions. These salmonids are also part of sport, commercial, ceremonial and subsistence fisheries outside the park. Here, we carefully examine 17 salmonid stocks on five park rivers to address the influence of a century of harvest and hatchery practices. Second, we assess the condition of a population of razor clams found on OLYM's Kalaloch Beach (Chapter 4.2.2). Razor clams are harvested within the park's boundaries, leading to unique challenges balancing tribal and public harvest opportunities with a sensitive and declining population whose dynamics are not well understood.

Summer coho below salmon cascades on the Sol Duc River. Photograph courtesy of NPS.

4.2.1. Pacific Salmonids

Jeffrey J. Duda, Samuel J. Brenkman, and Patrick Crain

Introduction

Olympic National Park forms a core stronghold for Pacific salmonids (a family-level designation that includes all five species of Pacific salmon, steelhead, and coastal cutthroat trout) on the West Coast, with 12 major watersheds; approximately 6,000 km of rivers and streams; and two large lakes, Lake Crescent and Lake Ozette (Figure 4.2.1). The park is a World Heritage Site, a designated Biosphere Reserve, and the only NPS unit outside of Alaska with substantial populations of native anadromous salmonids (i.e., fish that migrate up rivers from the sea to spawn). Five species of Pacific salmon (*Oncorhynchus* spp.) and steelhead , comprising about 70 unique populations, are part of a diverse fish assemblage, with 31 native species inhabiting large contiguous areas of protected, relatively pristine habitat. The high quality habitat in OLYM serves as an important sanctuary for anadromous fish that helps protect the underlying productivity and resiliency of coastal rivers.

Salmonid populations that inhabit OLYM rivers are of high ecological, recreational, and cultural importance to the public and eight Treaty Tribes, significantly contributing to sport, commercial, ceremonial, and subsistence fisheries. Salmon also play a key role in both aquatic and terrestrial ecosystems, serving as a principal node in intricate aquatic and terrestrial food webs (Willson and Halupka 1995, Cederholm et al. 1999, Helfield and Naiman 2006). To set the context for our assessment of salmonid stocks in OLYM, we first provide an overview of the history of fisheries management in the NPS as a whole and OLYM in particular. We then provide some background on salmonid management in Washington State and on the history of artificial propagation practices (i.e., hatcheries) as related to this assessment. Definitions of fisheries terms are provided in Table 4.2.1.

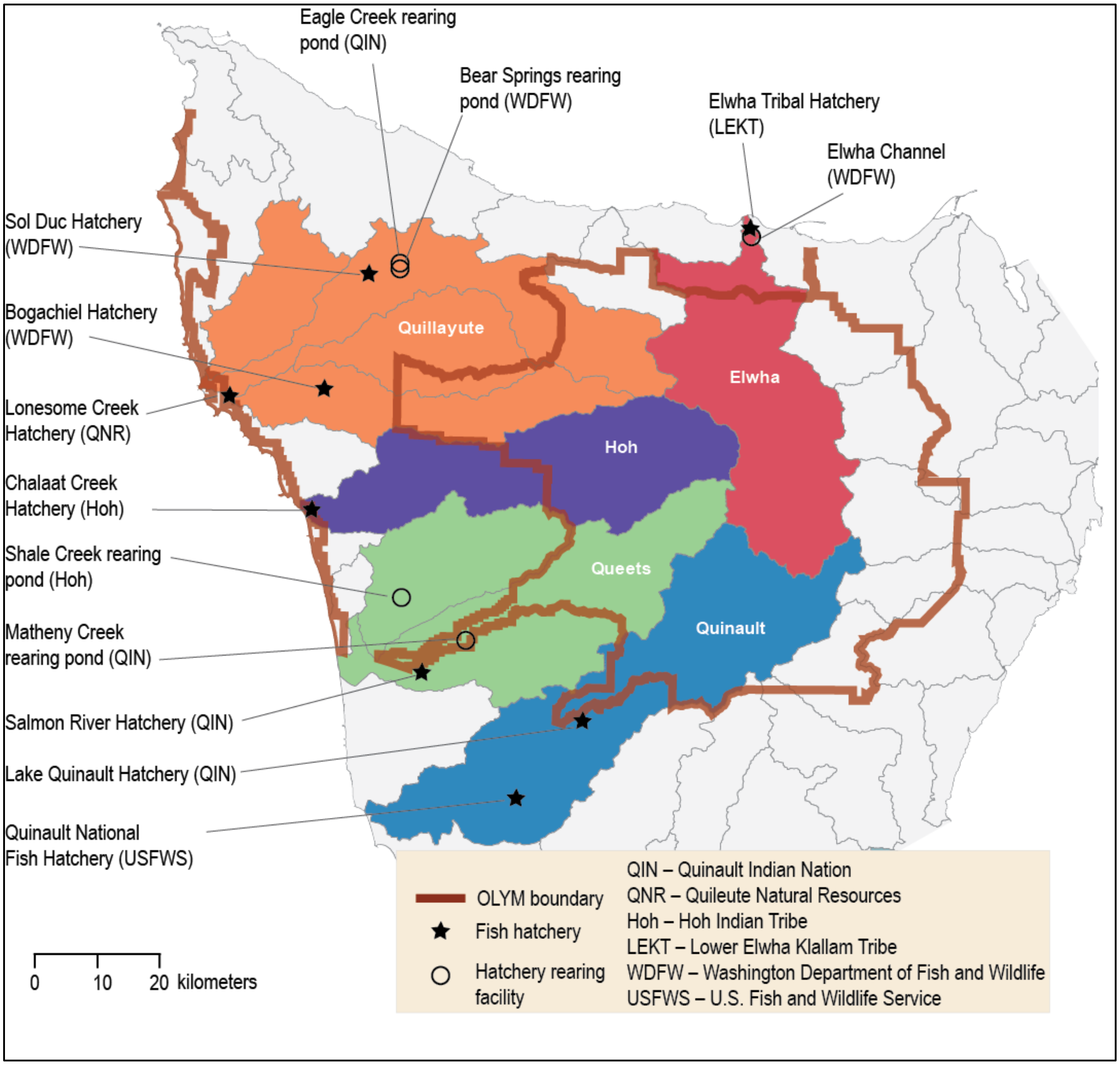

Figure 4.2.1. Locations of state, federal, and tribal salmonid hatcheries and rearing facilities in selected Olympic Peninsula watersheds. Watersheds shown in color are those focused upon in this report. Source: NPS.

Table 4.2.1. Definitions of fisheries terms used in this chapter.

Fisheries Term	Definition
Stock	Subpopulations of a particular species of fish occurring in a particular location or with particular attributes typically of interest to fisheries managers.
Evolutionary Significant Unit	A distinct grouping of populations designated for conservation or management purposes.
Escapement	The number of anadromous fish that escape the fishery and spawn.
Run timing	The time of year that adult salmon enter a river to spawn, which is generally specific to a given stock and defined by the beginning, peak, and end of the run.
Escapement goal	The number of adult salmon in a particular stock intended to escape the fishery to spawn, often to achieve the maximum sustained yield.
Maximum sustained yield	The largest average catch that can be taken continuously from a stock under existing environmental conditions.
Wild origin	Refers to locally adapted fish with little to no introgression from conspecific hatchery-origin fish
Natural origin	Refers to offspring resulting from mating in the wild between one or more parents of hatchery or unknown origin
Hatchery origin	Refers to artificially propagated fish
Terminal area	An area in which fish stocks begin to separate from one another; usually near a river mouth
Terminal run size	The total number of adult fish returning to a river, including those that escape to spawn and those that are harvested in the river.
Mean annual harvest rate	Over the period of record, the average number of fish harvested divided by the terminal run size.

Fisheries Management in the National Park Service

Fishing has been a traditional recreation activity in most of the U.S. National Park System since congressional authorization of Yellowstone National Park in 1872 (Panek 1994). The dichotomy of the park's mission to provide for use and enjoyment while conserving resources in an unimpaired manner has led to a long history of apparently contradictory management activities (Sellars 1992, Brenkman et al. 2014). Fishing is one of the few consumptive resource activities in national parks, and perhaps more extensively than any other wildlife, the Park Service historically manipulated fish populations (Sellars 1992, Panek 1994). Early management efforts in western parks enthusiastically sanctioned the taking of fish and the stocking and introduction of nonnative species to enhance recreational fishing and gain public support for a growing park system. This has led to significant biological and ecological changes to native fish populations from stocking programs intended for sport fisheries (e.g., Knapp et al. 2001, Schindler et al. 2001, Downen 2004) as well as those intended for conservation (e.g., Metcalf et al. 2012, Love Stowell et al. 2015). Today, most NPS units no longer stock fish in natural areas and instead rely on natural reproduction and angling regulations that promote both conservation and fishing opportunities.

Fisheries management in the NPS is governed by the principles and guidelines of the 1916 National Park Service Organic Act (16 U.S.C. et seq.), the enabling legislation of the NPS. Additional authorities and directions related to fisheries management are contained in the Redwood National Park Expansion Act of 1978 (16 U.S.C. 79a-79q); certain park units' enabling legislation; and other federal statutes including the Endangered Species Act, the National Environmental Policy Act, the Wilderness Act, the Clean Water Act, the Wild and Scenic Rivers Act, the Fish and Wildlife Coordination Act, the Magnuson Fishery Conservation and Management Act, the Migratory Game Fish Study Act, the Salmon and Steelhead Conservation and Enhancement Act, and the Lacey Act.

Generally, NPS fisheries management objectives seek to balance preservation of aquatic species and resources with providing visitors use and enjoyment of their national parks through recreational fishing opportunities. A management focus on maintaining naturally functioning aquatic ecosystems also provides opportunity to preserve, restore and protect viable fish populations and the genetic, behavioral, and ecological conditions that allow them to persist. With these broad goals in mind, decisions on harvesting fish within most NPS waters are made based on the species and its biological characteristics. Commercial fishing in National Park units is allowed only where specifically authorized by federal law or treaty rights. When commercial fishing is authorized, the NPS works with the tribes and the fishing industry to balance harvest with the health of the aquatic ecosystem. In general, the goal is to ensure that harvest does not unacceptably impact park resources or natural processes, including natural distributions, densities, and age-class distributions of fish.

Many NPS units protect watersheds and aquatic habitats that serve as refuges for native fish species. Habitat protection and restoration of depleted native fish stocks and the recovery of threatened and endangered fishes are among the highest priorities for fisheries management in the National Park System. Restoration of native fish populations and communities may involve the control or eradication of nonnative species that have contributed to depletion through predation, hybridization, or competition for limiting resources. Activities to achieve restoration may also include reintroduction to unoccupied habitats, barrier removal or modification, and implementation of special fishing regulations.

Fisheries Management in Olympic National Park

Fishery resources in rivers draining from the park are managed by OLYM, Washington Department of Fish and Wildlife (WDFW), and Treaty Tribes. Olympic National Park has exclusive federal jurisdiction to manage recreational fisheries within the park boundaries. Washington Department of Fish and Wildlife has jurisdiction of recreational and non-tribal commercial fisheries outside the park boundaries. The Treaty Tribes regulate commercial and subsistence gill-net and on-reservation tribal-guided sport fisheries. Park managers work annually with State and tribal managers to develop fishing regulations on a watershed-by-watershed basis. Sport fishing regulations in OLYM generally promote the catch-and-release of wild fish and the retention of nonnative and hatchery fish in coordination with the WDFW and the area tribes.

The coastal river systems draining from OLYM support sport fishing and commercial, ceremonial, and subsistence gill-net fisheries, with fish populations subjected to intensive fishing pressure and harvest during most months of the year. The highly popular sport fisheries that include guided and non-guided sport fishing for Pacific salmon and steelhead are economically important to local communities. Commercial catches of Pacific salmonids in rivers draining from the park are sold into local, regional, and national markets. Subsistence catch is for personal consumption and ceremonial catch is taken for cultural events by Treaty Tribes. Additionally, ocean harvest of some Pacific salmonids destined for OLYM rivers occurs over a broad geographic area ranging from Southwest Alaska to northern California. Ultimately, adult salmonids that "escape" harvest in recreational and commercial fisheries contribute to the abundance of spawning salmon in OLYM waters.

Management Framework for Pacific Salmonids in Washington

To more fully understand the complexity of fisheries management in OLYM, it is useful to lay out the legislation guiding salmonid management within the State of Washington. Much of the information in this subsection comes from National Oceanic and Atmospheric Administration (NOAA) Fisheries (http://www.westcoast.fisheries.noaa.gov/fisheries/salmon_steelhead/united_states_v_washington.html, accessed 8 October 2017). Pacific salmon and steelhead fisheries are managed under a complex framework based on a set of interrelated legal cases, U.S. law, and international treaties. These include the 1974 federal court case (U.S. v Washington) decided by U.S. District Court Judge George Boldt (commonly referred to as the Boldt Decision), a related case known as Hoh v. Baldridge, provisions of the Pacific Salmon Treaty between the U.S. and Canada, and the guidance found in the Magnuson-Stevens Act. The Boldt Decision re-affirmed the Treaty Tribe's right to harvest salmon and steelhead. Five treaties between the United States and various Washington tribes (1854 through 1856) described the reserved tribal fishing rights in common with citizens of the territory and included the treaties of Medicine Creek, Quinault, Neah Bay, Point Elliott, and Point-No-Point. Findings from the Boldt Decision clarified the treaties with regard to the allocation of salmon harvests between tribal and non-tribal fishers, holding that the tribes are entitled to a 50 percent share of the harvestable run of fish. Another important component from the Boldt Decision is the doctrine of "foregone opportunity" to address situations where, *"…one party, either Indian or non-Indian, chooses not to harvest their share of the salmon. In that case the other party may notify the non-harvesting party that they intend to harvest the balance of the non-harvesting party's salmon…Foregone opportunity applies only within one season and between parties, not between gear groups*" (Madson and Koss 1988:10).

The Hoh v. Baldridge case (522 F. Supp. 683), dictated that fishery management plans must take into account returns to individual streams if the fisheries might affect an individual tribe. This established another key management principle that requires river-by-river or run-by-run fisheries management (http://www.westcoast.fisheries.noaa.gov/fisheries/salmon_steelhead/united_states_v_washington.html, accessed 11 October 2017).

The Pacific Salmon Treaty, signed in 1985 by the U.S. and Canada, governs salmon fisheries in the two countries by establishing a forum for equitable sharing of harvest and conservation. The treaty is implemented through 4 panels, with representatives of both the US and Canada sitting on each panel. Independent annexes to the treaty provide guidelines for management of each species of salmon. These annexes are periodically renegotiated, which can result in substantial changes in management objectives.

Finally, the Pacific Fisheries Management Council exercises federal jurisdiction of harvest in coastal ocean waters 3–200 miles off of the coasts of Washington, Oregon, and California. For inland waters including the Puget Sound and Olympic Peninsula rivers, State and tribal representatives negotiate and develop annual salmon management plans for commercial and recreational fisheries in the North of Falcon process (http://www.westcoast.fisheries.noaa.gov/fisheries/salmon_steelhead/north_of_falcon.html, accessed 6 October 2017). The Puget Sound Salmon Management Plan, published in 1985, is the implementation framework for the allocation, conservation, and equitable sharing principles of United States v. Washington that governs management of salmon resources in Puget Sound between the Puget Sound Treaty Tribes and State of Washington. It defines the basis for deriving management objectives and allocation accounting, prescribes procedures for information exchange and dispute resolution, and includes provisions for annual review and modification. The State and tribes have managed the fisheries subject to the Puget Sound Plan through annual or multi-year agreements (e.g., the Comprehensive Chinook Plan or the Comprehensive Coho Plan). While management of the coastal river systems is not covered by the Puget Sound Management Plan, many of the principles are carried over, along with guidance from the Hoh v. Baldridge case.

For decades, harvest and escapement levels have been largely governed by the principle of Maximum Sustained Yield (MSY), in large part because it was established by the landmark 1974 Federal court case *U.S. vs Washington*. The theoretical underpinning of MSY is that there exists a maximum level of harvest for any given population which can be sustained in perpetuity (Ricker 1975). In theory, if one understands the underlying productivity of a population, this harvest level can be calculated and used to establish management objectives that will ensure a stock's persistence over time. Generally, these management objectives are either expressed in terms of "escapement goals" (number of adult fish which survive to spawn) or "harvest rates" (proportion of the total population which may be harvested).

Artificial Propagation of Salmonids on the Olympic Peninsula

Another tool of salmon management in Washington involves artificial propagation of salmon in hatcheries. The first hatchery in Washington State opened in 1895—for over a century, fish hatcheries have been viewed as a substitute for addressing root causes of declines in abundance, primarily loss of habitat, blockage of migratory routes, and overharvest (Waples 1999). In most cases, hatchery fish are used to supplement catches in recreational and commercial fisheries. Hatcheries spawn adult fish, culture embryos until hatching, rear juveniles within the artificial conditions of the hatchery environment and release individuals at various sizes into the river during a time when they can begin migration from freshwater to marine waters to complete their lifecycle.

On the coastal rivers of the Olympic Peninsula, there is an extensive array of federal, State, and tribal hatchery facilities that have released millions of chum, coho, pink, Chinook, sockeye, and steelhead into rivers, both inside and outside of the national park boundaries, to supplement commercial and recreational catches (Figures 4.2.1 and 4.2.2). More recent goals include using hatcheries for the conservation of imperiled populations of salmonids (Flagg and Nash 1999, Peters et al. 2014). The primary hatchery managers in the watersheds that are the focus of this chapter include WDFW, USFWS, the Lower Elwha Klallam Tribe (LEKT), the Hoh Tribe, the Makah Tribe, the Quileute Tribe, and the Quinault Indian Nation (QIN). The State and many tribes use various marking techniques (e.g., clipping the adipose fin, thermal marking of otoliths) to enable fishers and scientists to distinguish between hatchery and wild salmonids. There are some instances when hatchery fish are not externally marked (e.g., stocks listed under the Endangered Species Act, such as Elwha River Chinook salmon).

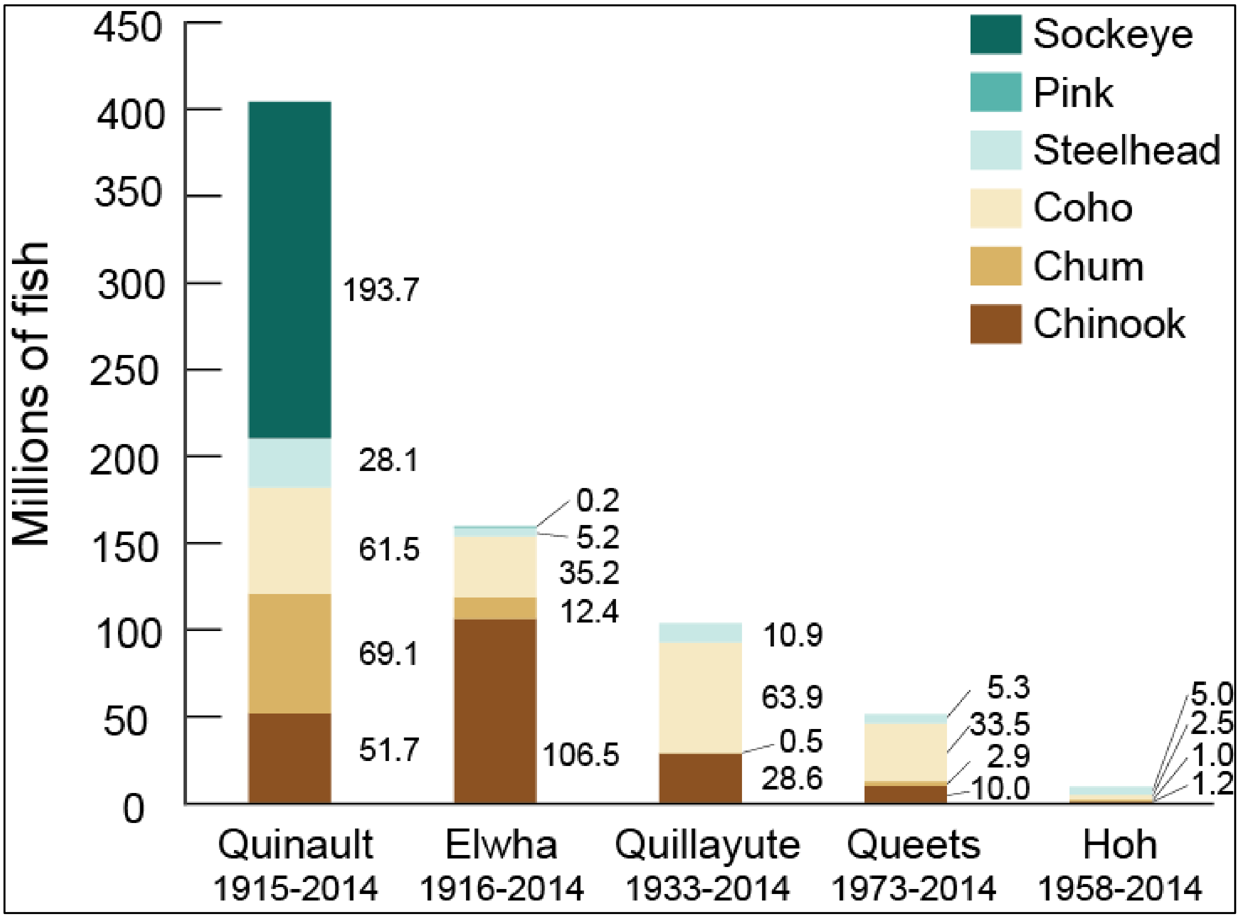

Figure 4.2.2. Total numbers of hatchery-origin Pacific salmonids by species released into the Quinault, Elwha, Quillayute, Queets, and Hoh River systems. The available period of record is shown below watershed name. Data from various hatchery records are summarized in Appendix F.

Hatchery programs operate under agreements and policies most recently guided by the Hatchery Reform Project, a project funded by congress in 1999 (Blankenship and Kern 2008). The State of Washington has further developed guidelines under their Hatchery and Fishery Reform Policy (Mobrand et al. 2005). The Salmonid Disease Control Policy requires testing to reduce the spread of infectious diseases among hatcheries and receiving waters (Fisheries Co-Managers of Washington State 2006). Additionally, Hatchery Genetic Management Plans are technical documents that describe the composition and operation of individual hatchery programs, with a goal of providing biologically-based artificial propagation management strategies that ensure the conservation and recovery of salmon and steelhead populations listed under the Endangered Species Act. Finally, A Joint Wild Salmonid Policy (1997) is designed to protect, restore, and enhance the productivity, production, and diversity of wild salmonids and their ecosystems to sustain ceremonial, subsistence, commercial, and recreational fisheries, non-consumptive fish benefits, and other related cultural and ecological values.

Assessment Goals and Objectives

Despite the high level of protection afforded within the boundaries of OLYM, including fishing regulations intended to balance recreational fishing activities with conservation and protection of salmon and other fishes (see discussion in Brenkman et al. 2014), management challenges exist for most rivers that contain Pacific salmon and steelhead. These include the well-known "four H's" of salmon management: hatcheries, harvest, hydroelectric dams, and habitat (Nehlsen et al. 1991, Ruckelshaus et al. 2002, McClure et al. 2003). The range-wide decline of salmon stocks drives management challenges and has led to listings under the Endangered Species Act, with 28 Evolutionary Significant Units of salmon and steelhead listed range wide (NOAA 2016), 17 in Washington State, and 5 in the waters of OLYM (Table 4.2.2). Recognizing the importance of Olympic Peninsula waters for anadromous fish, other authors have summarized the status of selected salmon and steelhead populations. Our reporting builds upon these previous stock status assessments conducted over two decades ago by Houston and Contor (1984), SASSI (1994), Huntington et al. (1994), and McHenry et al. (1996).

Table 4.2.2. Federally listed fish species in rivers draining from Olympic National Park. Data source: Federal Register 1999, NOAA 2016.

Species	Evolutionary Significant Unit	Listing Status (date listed)	Watersheds
Bull trout *Salvelinus confluentus*	Coastal-Puget Sound	Threatened (June 1998)	**Elwha[1], Hoh[1], Queets[1], Quinault[1]**, Skokomish, Dungeness, Gray Wolf
Chinook salmon *Oncorhynchus tshawytscha*	Puget Sound	Threatened (March 1999)	**Elwha[1]**, Dungeness, Dosewallips, Duckabush, Skokomish
Sockeye salmon *Oncorhynchus nerka*	Lake Ozette	Threatened (March 1999)	Lake Ozette
Steelhead trout *Oncorhynchus mykiss*	Puget Sound	Threatened (May 2007)	**Elwha[1]**, Dungeness, Graywolf, Skokomish
Chum salmon *Oncorhynchus keta*	Hood Canal summer-run	Threatened (March 1999)	Dungeness, Dosewallips, Duckabush, Hamma Hamma

[1] Watersheds identified in bold font are those analyzed in detail in this report.

In this assessment, we focused on two of the "4 H's" factors — the influence of past and present harvest and hatchery practices in key OLYM river systems. The other two factors, habitat and hydroelectric dams, while important, are not as influential within park borders and are not addressed in this report. For example, the protected salmonid habitat within OLYM is arguably as pristine as anywhere in the conterminous United States (Houston and Contor 1984) and removal of the two dams on the Elwha River is one of the largest salmon restoration projects in North America (Duda et al. 2008). Only one other river system draining from the park, the North Fork Skokomish, is influenced by hydroelectric dams. However, we recognize that habitat outside the park boundary has degraded from pristine conditions due to a variety of factors which likely contributes to the number of fish returning to each watershed. For example, the survival of juvenile fish that originated within the park but migrate outside of the park to rear could be effected. The State of Washington and local tribes have developed detailed habitat limiting factors analyses for each watershed.

Our goal was to establish a current baseline to assess the relative status and trends of key Pacific salmonid populations in rivers that drain from the park. We summarized annual escapement, catch, terminal run size, and hatchery release records for Chinook, coho, and steelhead, including 15 stocks from four Olympic Peninsula rivers (Hoh, Queets, Quinault, Quillayute) with sufficient data availability. We also summarized hatchery release records for chum, pink, and sockeye salmon for rivers where hatchery plantings exist, even though these stocks are not specifically addressed in the resource summary. The Elwha River, being the location of a large-scale restoration project, was treated differently than the other four rivers considered above, and included hatchery and harvest summaries for selected stocks with available data. We summarized approximately 5334 and 4517 records related to hatchery and harvest data, respectively, from the early 1900s to 2014. In addition to assessing the relative status of 17 key salmonid populations, we have provided a relational database of legacy and current harvest and hatchery information for future use by park managers.

Approach and Methods

We assessed the condition of 15 key salmon stocks from four major river systems (Hoh, Queets, Quillayute and Quinault) draining from OLYM (Figure 4.2.1) using the following measures: (1) trends in escapement and terminal run size of wild populations; (2) harvest rates and the annual number of hatchery and wild salmonids harvested; and (3) the annual number of hatchery releases. For each of the 15 salmon stocks, we derive one estimate of condition that incorporates these three measures (see 4.2.1 Resource Summary).

This report focuses primarily on coastal river systems that drain the western portion of OLYM (Quillayute, Hoh, Queets, and Quinault river systems) and the Elwha River, the site of a significant salmon restoration project that drains northward into the Strait of Juan de Fuca (Brenkman et al. 2008b, Duda et al. 2008, Pess et al. 2008). Adding the Elwha increases the number of examined stocks in this report to 17. We placed emphasis on these river systems because of the presence of important wild salmonid populations, intensive recreational and commercial fisheries on those populations, and data sets for salmonids stocks that date back, in some cases, to the early 1900s. The salmon stocks of emphasis include spring/summer Chinook, fall Chinook, summer and fall coho, and winter steelhead. The referenced season before each population refers to general entry timing from the ocean to the river. We analyzed escapements, terminal run sizes, and harvest rates for wild stocks only. We also report total catch (commercial, ceremonial and subsistence, and sport) for both hatchery and wild stocks within each river. A list of fisheries terms used in this chapter and our definitions is provided in Table 4.2.1.

Fisheries data related to harvest and hatcheries were taken from multiple sources (Appendix F). The contemporary sources, related to the co-management of fisheries by tribes and WDFW, were available in various online databases and published and non-published forms. Additionally, we reviewed historical data from various sources, including long-term records maintained at OLYM and data tables and appendices from the decision documents related to the Boldt Decision. Data from these sources were compiled and placed into a relational database for analysis of status and trends.

Escapement

Escapement levels, or the numbers of fish returning to spawn, were determined on a stock-by-stock basis by fisheries co-management conducted by WDFW and Washington Treaty Tribes. For most stocks in Washington coastal rivers, an escapement objective (goal) has been established to guide fishery managers in annual planning and conducting fisheries to harvest the number of "surplus" fish. The "actual spawners" are derived from a pool of "recruits" which are those adult salmon that have survived the ocean phase of their life history, and have the potential to spawn. The number of recruits minus the number of adults needed to meet the escapement objective is considered harvestable surplus, and is the main target of recreational and commercial fisheries.

Annual escapement estimates for each stock and river were obtained from WDFW and Treaty Tribes including annual reports maintained by the Technical Committee of the Pacific Salmon Commission (http://www.psc.org, accessed 9 October 2017) and other unpublished records associated with each particular river and stock. In general, escapement was estimated from expanded redd (a depression in the river's gravel bed dug by spawning females) counts that occur during a specific period for each stock (e.g., October 15 to December 1 for fall Chinook salmon on the Queets River). Surveys were conducted in each river (e.g., foot, boat, and helicopter surveys), with surveyors recording new redds each week. Cumulative redd counts for each index reach were used to estimate spawning curves, and expansions were then used to estimate spawning escapement levels. In most cases, this involved assumptions that relate the total number of redds to the total number of fish (male + female) and that index reaches were representative of the spawning distribution. We provide a brief description for the methods used for each stock below.

Harvest (1900s to 2014)

We obtained historical canning records from various documents and presented summaries of them below in the reference condition section. Commercial and sport harvest records were obtained from over 50 different sources of published and unpublished records and data files at OLYM. These sources include annual reports of tribal fisheries programs, status review documents from NOAA fisheries (e.g., Gustafson et al. 1997), Elwha River dam removal technical documents, and unpublished data records.

Hatchery Releases (1915 to 2014)

A key to understanding hatchery production is to summarize the magnitude of releases among species throughout coastal rivers of the Olympic Peninsula. State and federal agencies and tribes track hatchery releases by hatchery or stock but no comprehensive summary has been completed for Olympic Peninsula watersheds. We summarized temporal and spatial hatchery releases from federal, State, and tribal hatcheries from 1915 to 2014. The summary included stock origin, numbers of juveniles released, and release locations of 34 hatchery-produced salmonid stocks.

Hatchery releases into Olympic Peninsula rivers include salmonids derived from non-indigenous and indigenous stocks or transferred from other areas. Genetically segregated brood stocks are generally derived from hatchery-origin adults returning to the hatchery each year. Segregated programs are structured to create a genetically distinct, hatchery-adapted population (Flagg 2015). Genetically integrated brood stocks include natural-origin fish in the broodstock each year to maintain genetic integration with the natural population. For most integrated programs, the goal is to minimize the genetic and reproductive fitness differences between the hatchery broodstock and the natural spawning population from which they were derived (Flagg 2015). Fish size at release varies greatly among species and ecotypes with rearing time up to 18 months.

The accuracy and interpretation of hatchery release data is challenging. Data on release locations, species, run type, annual number of releases, and hatchery stock origin were obtained from multiple published and unpublished sources. The primary source was data from the Regional Mark Information System (RMIS), an online database operated by the Pacific States Marine Fisheries Commission, Regional Mark Processing Center (http://www.rmpc.org; Nandor et al. 2010). This online web application allows users to build queries and search long-term information about hatchery release programs throughout the West Coast, including those on target rivers of the Olympic Peninsula discussed in this chapter. The RMIS database has consistent annual records of hatchery releases into Olympic Peninsula rivers since about the 1970s.

We summarized 5,334 records from RMIS and estimated the annual and total numbers of hatchery fish released by run type (e.g. spring, summer, fall), by species, and for each river over the period of record. For cases where "run type" was unspecified in RMIS, we reported the numbers by species for a given river and designated those records as "unspecified" run type in the Appendices. Therefore, the total number of hatchery releases of a given species was the sum of records for run type plus those designated as "unspecified." Consequently, hatchery release numbers by run type for a given stock represent an absolute minimum number of fish released for most stocks. We determined origin of hatchery stocks for each species from the RMIS field entitled "hatchery stock location."

Additional hatchery release information was obtained from unpublished data and historical records contained at OLYM (Appendix F). We included hatchery release numbers from unpublished and historical sources for any years that were not included in RMIS.

Reference Conditions

Information on historical abundances of Pacific salmonids on the Olympic Peninsula is very limited. The authenticity, accuracy and interpretation of historical data is challenging, yet the insights from including these data are often very important for conservation (McClenachan et al. 2012). However, anecdotal reports, gray literature, and canning records provide some perspective on the duration of fisheries and the relative past magnitude of fish population sizes. Historical data on spawning escapements and total run sizes prior to the 1950s are not available and in this chapter we use data from the mid-1970s to assess trends in terminal run size (Figure 4.2.3). We found records of intensive harvest from canning records in coastal rivers that dated back to 1911 (Cobb 1930). Harvest of Olympic Peninsula salmonids occurs in marine fisheries from Washington to Alaska (Weitkamp 2010) and in terminal freshwater fisheries in the lower sections of rivers.

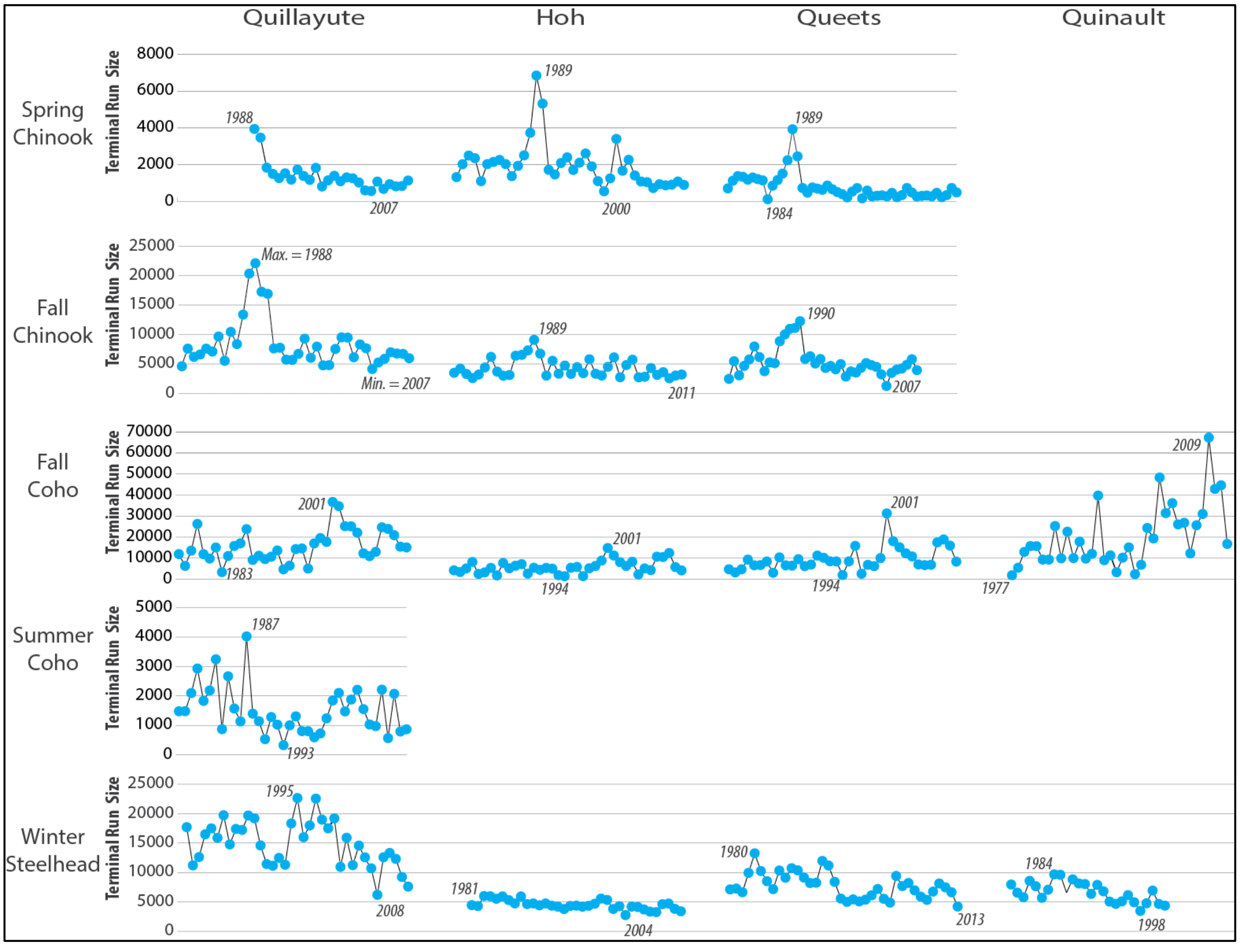

Figure 4.2.3. Terminal run size estimates by salmonid stock and river for four focal Olympic Peninsula watersheds. Terminal run size (i.e., total number of wild adult fish returning to each river, inclusive of harvested fish and escapement estimate) given over time from 1976 until 2013, with years of maximum and minimum values indicated.

As of the late 1800s, commercial salmon fisheries and canneries on the Olympic Peninsula were an important part of the regional economy and salmon have been harvested since time immemorial by Coast Salish people. Johnson (2013) reported that: "*J.W. Hume was one of the earliest to start commercial fishing in the area. In 1892 he purchased a cannery built in Port Angeles a year earlier […] and Hume packed 1.46 million pounds or 22,100 cases of salmon at Port Angeles harbor. The fish were taken 'mainly in the vicinity of that place,' as well as from King, Skagit, and Mason counties.[…] he used 12 seines in waters near the Port Angeles cannery. In 1899, the company used 2.04 million pounds of salmon [...].*"

Johnson (2013) also reported that: "*By 1912 Neah Bay had become a salmon canning outpost [….]The sight is said to be most impressive at night when most of the vessels are in harbor [and] their thousands of lights make the little port look like a large city. The bay had become a 'floating city' of 6,000 persons devoted to catching and processing salmon: two canneries, three curing plants, 115 purse seine boats, 350 gasoline trollers, and 600 sail and rowboats [….]*."

Hoh River

In the Hoh River, there are no available catch or abundance data prior to the 1930s for salmonids although a fish cannery operated from 1917 to at least 1921 (McMillan 2006). An assessment of historical canning records revealed 2,593 cases of canned salmon packed on the Hoh River from 1917 to 1921 with 48 one-pound cans per case (Cobb 1930). Wendler and Deschamps (1955) reported annual catches of fall coho up to 7455 fish in 1947 (mean annual catch = 4,122 from 1935 to 1954) and Chinook up to 2973 in 1953 (mean 1,016; 1,935–1,954). From 1933–1977, the peak annual catch of winter steelhead in the tribal fishery was 4,320 winter steelhead in 1976–1977 (WDG 1979). McMillan (2006) estimated historical winter steelhead abundances averaged from 7,938 to 13,230 fish from 1948–1961, with current abundances at 34-56% of the 1948-1961 average.

Queets River

From 1905 to at least 1927, a fish cannery was in operation on the Queets River (Cobb 1930). An assessment of historical canning records revealed that 37,403 cases of canned salmon and steelhead were packed on Queets River from 1912 to 1927 (Cobb 1930). From 1933–1976, the peak annual catch of winter steelhead in the tribal fishery was 13,182 winter steelhead which occurred in 1953–1954 (WDG 1979). Wendler and Deschamps (1955) reported annual catches of fall coho up to 17,689 fish in 1947 (mean 10,712; 1935-1954) and Chinook up to 9,249 in 1951 (mean 2,967; 1935–1954). McMillan (2006) reported estimated historical winter steelhead abundances ranging from 48,980 to 81,633 fish, with current abundances at 7.6–12.6% of this level.

Quinault River

An assessment of historical canning records revealed that 187,333 cases of canned salmon were packed on the Quinault River from 1911 to 1928 (Cobb 1930). Although small, the packs on these streams were consistent and profitable for the Native groups that supplied much of the salmon. In 1915, Natives at Quinault caught 355,007 sockeye valued at $80,348.60 (Johnson 2013), equivalent to over $1.8 million dollars in 2016. Sockeye returns to Lake Quinault between 1908 and 1912 averaged 380,000 fish per year with a peak estimated total return of up to 681,000 sockeye in 1908. An examination of harvest data for Quinault sockeye revealed a marked decline in stock productivity during the 1940s and 1950s (QIN 1981). Wendler and Deschamps (1955) reported annual catches of fall coho up to 44,966 fish in 1935 (mean 21,039; 1935–1954) and Chinook up to 7,372 in 1935 (mean 2858; 1935-1954). From 1940–1976, the peak annual catch of winter steelhead in the tribal fishery was 8,712 winter steelhead (1951/1952; WDG 1979). McMillan (2006) reported estimated historical winter steelhead abundance of 19,000 fish in 1952 with current abundances at 25.7% of the 1952 level.

Quillayute River

An assessment of historical canning records revealed 7,156 cases of canned salmon packed on the Sol Duc River from 1912 to 1915 (Cobb 1930). In 1955, commercial fishing in the Quillayute River was abolished upstream of the reservation (Wendler and Deschamps 1955). Wendler and Deschamps (1955) reported annual catches of fall coho up to 17,322 in 1952 (mean 8,588; 1935–1954) and Chinook up to 4,478 in 1941 (mean 1,817; 1935-1954). From 1945–80, the annual peak catch of winter steelhead in the tribal fishery was 10,504 winter steelhead (1974/1975; WDG 1979). McMillan (2006) reported estimated historical winter steelhead abundances averaged 17,614 from 1948–1961 with current abundances at 82.7% of the historical average.

Total run size for major winter steelhead stocks on the Olympic Peninsula during the early 1980s was ~70,000 fish (Light 1987). Harvest of steelhead in Washington in 1991 was the lowest recorded since 1962 (Cooper and Johnson 1992). The popularity of sport fishing for steelhead in Washington steadily increased after the end of World War II where the number of fishers increased from 41,000 anglers in 1948-49 to 77,300 anglers in 1952–53 (Larson and Ward 1955).

Elwha River

Several estimates of historical production of Pacific salmonids in the Elwha River were made as part of the dam removal planning process (Winter and Crain 2008). Estimates of historical salmon numbers used a variety of methodologies and were based upon available habitat in the river, production estimates from other watersheds, and early hatchery records. Because of the methods used and perhaps the available information at the time of the estimates, historical production values reported by Winter and Crain (2008) were highly variable. They reported the following: Chinook (average = 7,966; range = 1,284–17,493), coho (average =10,744; range = 3,520–19,143), steelhead (average = 4,143; range = 483–5,757), pink (average = 72,624; range = 3,147–137,600), chum (17,128; range = 9,042 – 25,600), and sockeye (3,042; range = 85 – 6,000). Observations by tribal elders and early settlers provided qualitative information such as, "*the fish were so plentiful that there was no need to select 'good' areas*" (Lane and Associates 1990). An early homesteader wrote on November 9, 1897, "*The salmon lay there with their backs out of water. All I had to do was to reach over them, hook the hook in their back and pull them out*" (Johnson 2013).

Results and Assessment

Hoh River

Watershed Overview

The Hoh River basin is the third largest drainage on the Olympic Peninsula, draining 894 km^2. The river flows 91 km from glaciers and ice fields on the slopes of Mount Olympus and descends 1,216 m in elevation to its confluence with the Pacific Ocean. Forty-four named tributaries flow into the Hoh River system. The South Fork Hoh, the major tributary to the Hoh River, drains 130 km^2 and flows westward until it joins the main stem Hoh at river kilometer (hereafter "rkm") 49. Fifty-eight percent of the Hoh River watershed (a 45-rkm reach) occurs in OLYM. The lower portion of the river flows through State, tribal, and private lands. The Hoh Basin has a maritime climate and receives an annual mean precipitation of 358 cm, most of which occurs from November to April. The annual median daily flow of the Hoh River is 51 m^3/s and mean daily flow is 71 m^3/s (England 2003).

Fish Assemblage Overview

The Hoh River supports populations of fall coho salmon, spring/summer and fall Chinook salmon, chum salmon, summer and winter steelhead, coastal cutthroat trout, bull trout, mountain whitefish (*Prosopium williamsoni*), sculpin, longnose dace (*Rhinichthys cataractae*), and Pacific lamprey (*Entosphenus tridentatus*) (Brenkman and Corbett 2005). Anadromous salmonids have access to at least 84 rkm up to the headwaters of the Hoh River and up to rkm 22 in South Fork Hoh.

Hoh spring/summer Chinook salmon are of wild origin and enter the river from April to August and spawn from late August to mid-October (Jorgensen et al. 1984, SASSI 1994). Hoh fall Chinook are primarily of wild origin and enter the river from September to December, and spawning occurs from mid-October through December in the main stem river primarily between Nolan and Owl Creeks (Jorgensen et al. 1984, SASSI 1994). Hoh River coho have been managed as a wild stock and enter the river from September to mid-February and spawn from October to mid-January (Jorgensen et al. 1984, SASSI 1994). Winter steelhead enter the river from November through May and spawn from March to June (Jorgensen et al. 1984). Chum enter the river from late September to early December (Mattson et al. 1980).

Stock status and trends

Spring/Summer Chinook

Hoh River spring/summer Chinook were classified as healthy in Huntington et al. (1994) and Salmon and Steelhead Stock Inventory (SASSI 1994). The established escapement goal for Hoh wild Chinook is 900 fish and wild escapement averaged 1,399 fish from 1976 to 2013 (Figure 4.2.4). Estimates of escapement ranged from a high of 4,697 fish in 1989 to a low of 492 fish in 2000. The escapement goal was met in 29 of 38 years (76%) since 1976 and in 4 of the last 10 years in the record (inset of Figure 4.2.4).

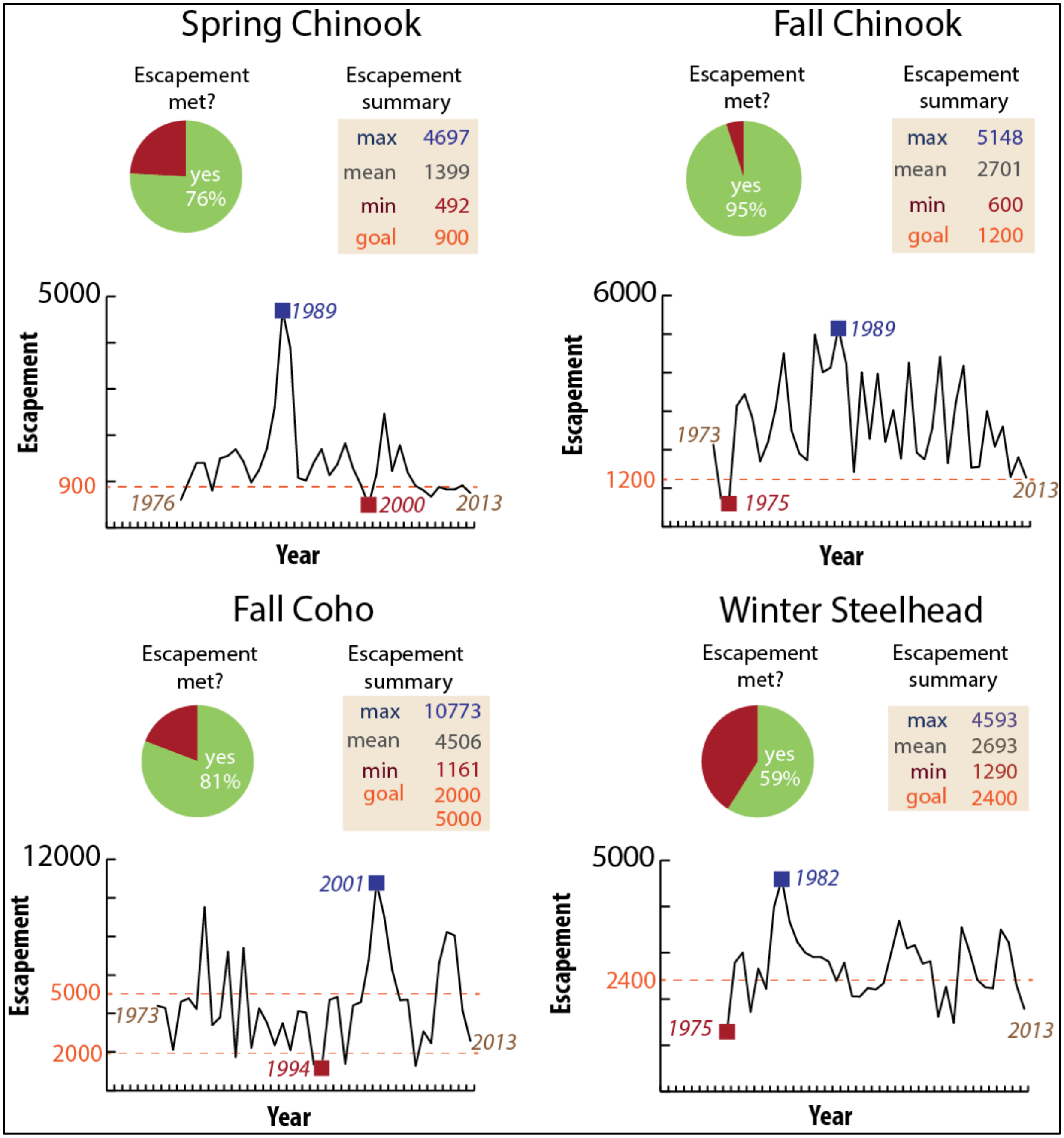

Figure 4.2.4. Trends in annual escapement levels of adult fall Chinook salmon, spring Chinook salmon, fall coho salmon, and winter steelhead in the Hoh River system. Each line graph depicts trends in annual escapement of wild fish during the period of record with the dotted line denoting the escapement goal set by the fisheries co-managers. Pie charts show the percent of years the escapement goal was met (green) or missed (red) for each stock. Data were derived from escapement records summarized in Appendix H.

From 1976 to 2013, the mean terminal run size was 1,941 wild fish (range to 529 to 6,820; SD 1,227; Figure 4.2.3), with an average annual percent change of -2.4 % over the period of record (Table 4.2.3). Since 1976, the total number of Chinook (hatchery and wild) harvested in commercial and sport fisheries in the Hoh River were 17,840 and 6,065, respectively (cumulative data since 1935 shown in Figure 4.2.5a). The mean number of Chinook annually harvested in commercial and recreational fisheries, respectively, was 469 fish (SD 377) and 164 (SD 143) fish. The mean annual harvest rate of total run size (hatchery and wild caught in the river, but not in ocean fisheries) was 28% (range 3–55%; SD 13%).

Table 4.2.3. Trend in terminal run size for wild stocks of selected Olympic National Park rivers with sufficient time series data. The average annual percent change in terminal run size was calculated as the slope of the least-squares linear regression line of the logarithmic annual terminal run size (following Busby et al. 1996).

River	Stock	Average Annual % Change in Terminal Run Size (95% CI)	R^2	*p*
Hoh	Spring Chinook	-2.4 (-3.8 to -1.0)	0.25	0.001
Hoh	Fall Chinook	-0.07 (-1.8 to 0.4)	0.04	0.22
Hoh	Fall Coho	+2.0 (0.2 to 3.7)	0.12	0.03
Hoh	Winter Steelhead	-1.2 (-1.7 to -0.6)	0.38	0.001
Queets	Spring Chinook	-3.8 (-5.6 to -2.0)	0.35	0.001
Queets	Fall Chinook	-1.2 (-2.5 to 0.00)	0.10	0.05
Queets	Fall Coho	+2.7 (0.9 to 4.5)	0.21	0.004
Queets	Winter Steelhead	-1.3 (-2.0 to -0.5)	0.25	0.001
Quillayute	Spring Chinook	-4.4 (-6.1 to -2.8)	0.56	0.001
Quillayute	Fall Chinook	-1.0 (-2.1 to 0.2)	0.07	0.10
Quillayute	Fall Coho	+1.9 (0.4 to 3.4)	0.15	0.020
Quillayute	Summer Coho	-1.4 (-3.0 to 0.2)	0.08	0.08
Quillayute	Winter Steelhead	-1.2 (-2.0 to -0.4)	0.20	0.005
Quinault	Fall Coho	+4.9 (2.1 to 7.8)	0.27	0.001
Quinault	Winter Steelhead	-2.2 (-3.3 to -1.1)	0.40	0.001

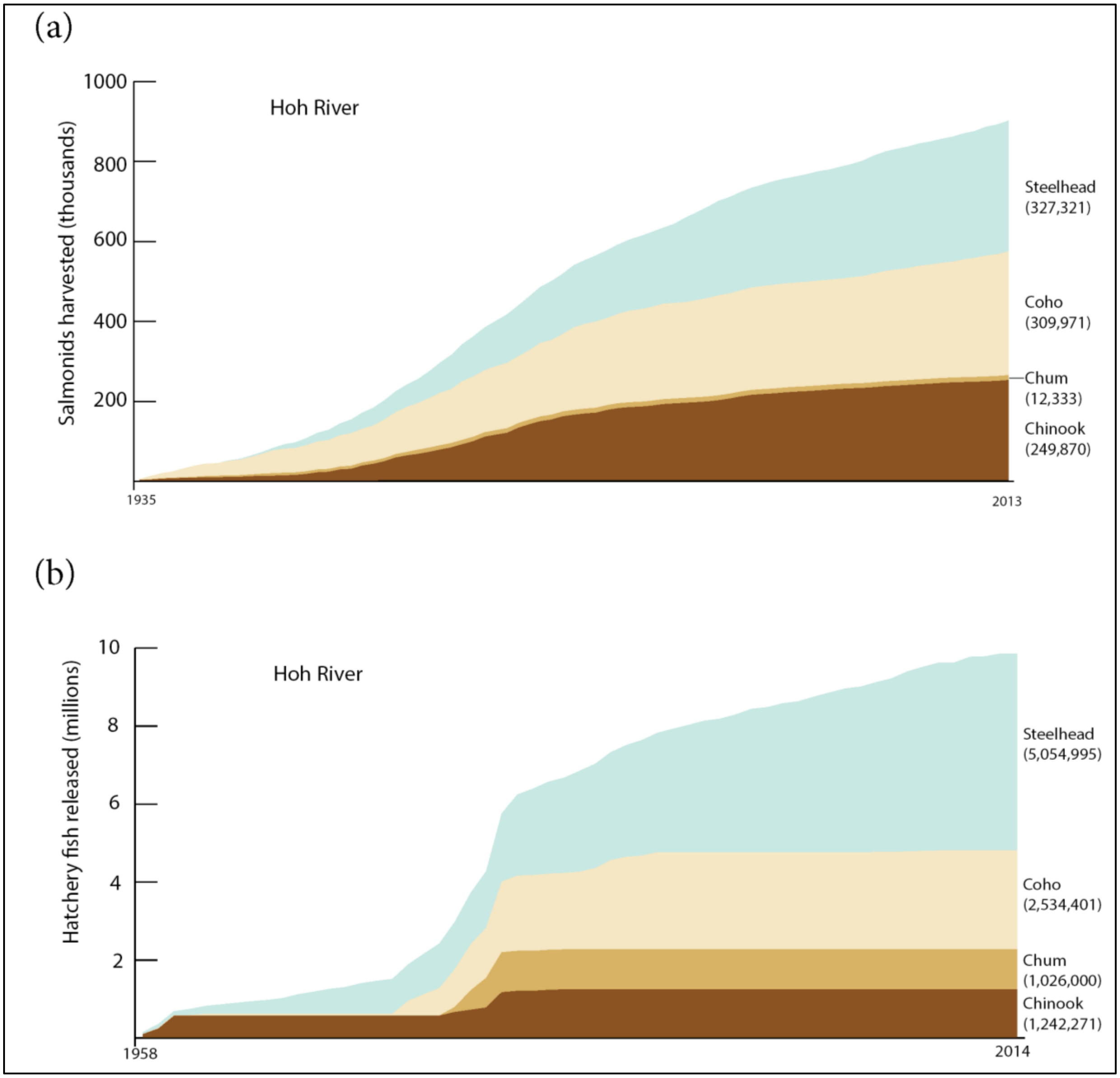

Figure 4.2.5. (a) Cumulative numbers of wild and hatchery Pacific salmonids harvested in commercial, ceremonial/subsistence, and sport fisheries in the Hoh River system from 1933 to 2013 and (b) cumulative numbers of hatchery raised Pacific salmon released into the Hoh River system from 1973 to 2014. Data were derived from hatchery and harvest records summarized in Appendix F.

Escapement for this stock was estimated from redd counts in index (weekly surveys) and supplemental areas (surveyed at peak of spawning) in the main stem and South Fork Hoh rivers and in tributaries. Surveys were conducted by foot and boat and estimates were derived using index timing curves. The total natural spawning escapement was calculated assuming 2.5 fish per redd. Spawning escapement for this stock was based on redd counts. The escapement floor policy of 900 for the Hoh spring/summer Chinook salmon was developed by Cooney (1984) and Quinault Department of Natural Resources (QDNR; 1982) based on spawner–recruit analyses. Stock production analysis of spawning escapement for brood years 1969 to 1976 was utilized to determine the initial escapement floor.

Fall Chinook
Hoh River fall Chinook were classified as healthy in Huntington et al. (1994) and SASSI (1994). The established escapement goal for Hoh wild fall Chinook was 1,200 fish (or 60% of total run) and the wild escapement averaged 2,701 fish (SD 1201) from 1973 to 2013 (Figure 4.2.4). Estimates of escapement ranged from a high of 5,148 fish in 1989 to a low of 600 fish in 1975. The escapement goal was met in 39 of 41 years since 1976 (95% of years) and in 10 of the last 10 years in the record (Figure 4.2.4).

From 1976 to 2013, the mean terminal run size was 3,881 wild fish (range 2,163 – 8,692; SD 1,609; Figure 4.2.3), and there was no significant trend in the time series over the period of record (Table 4.2.3). Since 1976, the total number of fall Chinook (hatchery and wild) harvested in commercial and sport fisheries in the Hoh River were 36,349 and 7,549, respectively (cumulative harvest data since 1935 shown in Figure 4.2.5a). The mean number of fall Chinook annually harvested in commercial and recreational fisheries were 957 fish (range 342 – 2,740; SD 570) and 204 fish (range 0 – 600; SD 140) fish, respectively. The mean annual harvest rate of total run size (hatchery and wild) was 30% (range 18 – 52%; SD 10%).

Escapement for this stock was estimated from redd counts in index (weekly surveys) and supplemental areas (surveyed at peak of spawning) in the main stem and South Fork Hoh rivers and in tributaries. Surveys were conducted by foot and boat and estimates were derived using index timing curves. The total natural spawning escapement was calculated assuming 2.5 fish per redd. Spawning escapement for this stock was based on redd counts. The escapement floor policy of 1,200 fall Chinook was developed by Cooney (1984) and QDNR (1982) based on spawner–recruit analyses. Stock production analyses of spawning escapements from 1968 to 1982 were utilized to determine the initial escapement floor.

Fall Coho
Hoh River coho were classified as healthy in SASSI (1994). The established escapement goal for Hoh wild coho is 2,000 to 5,000 fish and wild escapement averaged 4,130 from 1976 to 2012 (Figure 4.2.4). Estimates of wild escapement ranged from a high of 10,773 fish in 2001 to a low of 1,161 fish in 1994. The minimum escapement goal of 2,000 fish was met in 30 of 37 years since 1976 (81% of years) and in 9 of the last 10 years (Figure 4.2.4).

From 1976 to 2012, the mean terminal run sizes were 5,930 wild coho (range 1,404 – 14,801; SD 3157) and 525 hatchery coho (range 0 – 3,022; SD 606) (Figure 4.2.3). The trend in terminal run size of wild fish showed an average annual percent change of 2.0 % over the period of record (Table 4.2.3). Since 1976, the total number of fall coho harvested (hatchery and wild) in commercial and sport fisheries in the Hoh River were 70,557 and 8,314 fish, respectively (cumulative data since 1935 shown in Figure 4.2.5a). The mean number of fall coho annually harvested (hatchery and wild) in commercial and recreational fisheries was 1,907 fish (range 85 – 4,294; SD 1,195) and 225 fish (range 4 – 1,210; SD 249) fish, respectively. The mean annual harvest rate of total run size (hatchery and wild) was 33% (ranged 5 – 61%; SD 15%).

Winter Steelhead

Hoh River winter steelhead were classified as healthy in SASSI (1994). The established escapement goal for wild winter steelhead was 2,400 fish and wild escapement averaged 2,693 fish from 1975 to 2013 (Figure 4.2.4). Estimates of escapement ranged from a high of 4,593 fish in 1982 to a low of 1,290 fish in 1975. The escapement goal was met in 23 of 39 years since 1975 (59% of years) and in 5 of the last 10 years up to 2013 (Figure 4.2.4).

From 1979 to 2013, the mean terminal run size was 4,294 wild fish (range 2,541 – 5,783; SD 818; Figure 4.2.3), with the average annual percent change of -1.2 % over the period of record (Table 4.2.3). Since 1979, 36,298 wild winter steelhead were harvested in commercial fisheries (mean 1,037, range 258 – 1,927, SD 435) and 17,133 (mean 490, ranged 10 – 1,303; SD 322) in recreational fisheries, respectively (cumulative harvest data since 1935 shown in Figure 4.2.5a). The mean annual harvest rate for wild steelhead was 35% (range 7 – 54%; SD 12%). The ratio of hatchery:wild winter steelhead in the commercial fishery was 2:1 and in the sport fishery was 3.4:1.

Summary of Hatchery Releases

Annual hatchery releases of salmonids have occurred in the Hoh system since 1958. The Chalaat Creek hatchery facility is operated by the Hoh Tribe and located on the Hoh Indian Reservation near the river mouth. The facility opened in 1978 and released coho salmon (reared at Lake Quinault) and winter steelhead into the Hoh River. In addition to rearing tanks, the hatchery has a small incubation facility. The Quinault National Fish Hatchery previously released winter steelhead smolts into the Hoh River to support tribal commercial and non-Indian sport fisheries in the Hoh River.

Hatchery releases occurred throughout the main stem Hoh and tributaries. From 1958 to 2014, 9.8 million salmonids were released into the system including about: 4.1 million winter steelhead (54 years), 0.9 million summer steelhead (24 years), 0.9 million fall Chinook (12 years), 0.3 million spring Chinook (6 years), 0.04 million fall coho (4 years), and 0.3 million fall chum (2 years). Additionally, there were unspecific run-type origins that included: 217 steelhead (1 year), 94 Chinook (year), 2.5 million coho (20 years), and 0.8 million chum (3 years; Figure 4.2.5b; Appendix G). For hatchery releases into the Hoh River system, there were an estimated 16 non-indigenous stock origins among the salmonid species (Appendix G).

Queets River

Watershed Overview

The Queets River is the second largest drainage on the Olympic Peninsula. The river drains 1,157 km^2 and is located on the western slopes of the Olympic Mountains. The river flows 82.7 km from the headwaters and descends from 2,430 m in elevation to its confluence with the Pacific Ocean. There are 68 named tributaries in the Queets River system. Major tributaries include the Clearwater, Salmon, and Sams Rivers, and Matheny and Tshletshy Creeks. The upper 85% of the river flows through OLYM and enters the QIN reservation at rkm 10.9.

Fish Assemblage Overview

Fish species that inhabit the Queets River system include spring/summer and fall Chinook salmon, coho salmon, chum salmon, sockeye salmon, summer and winter steelhead trout, coastal cutthroat trout, bull trout, sculpin, mountain whitefish, dace, and Pacific lamprey. Bull trout are listed as federally threatened under the Endangered Species Act.

The salmonid community in the Queets River is comprised of wild, natural, and hatchery fish. Queets spring/summer Chinook are of wild origin, enter the river from February to August, and spawn in August and September. Queets fall Chinook enter the river from September to December. Spawning occurs from October through December in the main stem river, Matheny Creek, and Sams River. Queets coho are wild and hatchery origin, enter the river in August to December, and spawn from October to January. Winter steelhead are wild and hatchery origin, enter the river from November through May, and spawn from January to May (QDNR 1982).

Stock status and trends

Spring/Summer Chinook

Queets spring/summer Chinook were classified as depressed based on a short-term severe decline in escapement in SASSI (1994). The established escapement goal for Queets wild spring/summer Chinook is 700 fish (or 70% of run) and wild escapement averaged 876 fish from 1969 to 2013 (Figure 4.2.6). Estimates of escapement ranged from a high of 2,568 fish in 1989 to a low of 189 fish in 2003 (SD 591). The escapement goal was met in 22 of 45 years since 1969 (49% of years) and in 1 of the last 10 years since 2013 (Figure 4.2.6).

From 1976 to 2013, the mean terminal run size was 899 wild fish (range 150 – 3,954; SD 744), with an average annual percent change of -3.8 % over the period of record (Table 4.2.3). Since 1976, 6,244 wild Chinook were harvested in commercial fisheries (cumulative harvest data since 1933 shown in Figure 4.2.7a). The mean number of wild Chinook annually harvested in commercial and recreational fisheries from 1976 until 2013 was 164 (range 0 – 1,181, SD 262), and 25 (range from 0 – 129; SD 31), respectively (Figure 4.2.7). The mean annual harvest rate of total run size for hatchery and wild Chinook was 13% (range 0 – 58%; SD 15%).

Escapement for spring/summer Chinook was estimated from redd counts from August 15 to October 15. Surveys were conducted by foot and boat and estimates were derived using index timing curves. The total natural spawning escapement was calculated assuming 2.5 fish per redd. Spawning escapement for this stock is based on redd counts. The escapement floor policy of 700 for the Queets spring/summer Chinook salmon was developed by Cooney (1984) and QDNR (1982) based on spawner–recruit analyses. Stock production analysis of spawning escapement for brood years 1969 to 1976 was utilized to determine the initial escapement floor (Pacific Salmon Commission 2014).

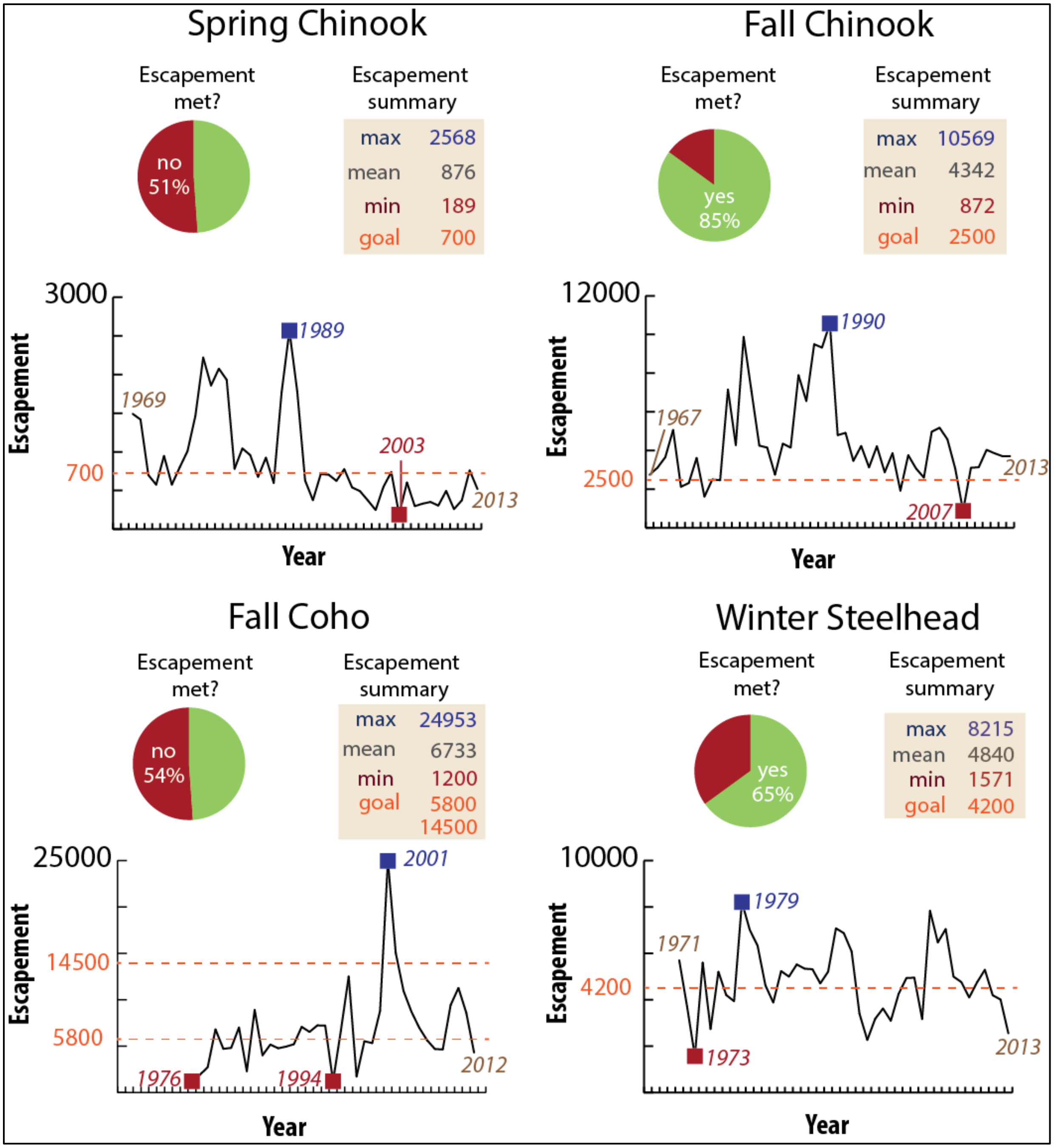

Figure 4.2.6. Trends in annual escapement levels of adult fall Chinook salmon, spring Chinook salmon, fall Coho salmon, and winter steelhead in the Queets River system. Each line graph depicts trends in annual escapement of wild fish during the period of record with the dotted line denoting the escapement goals set by each fisheries co-manager. Pie charts show the percent of years the escapement goal was met (green) or missed (red) for each stock. For fall coho salmon the lower of two escapement goals (5800) was used. Data were derived from escapement records summarized in Appendix H.

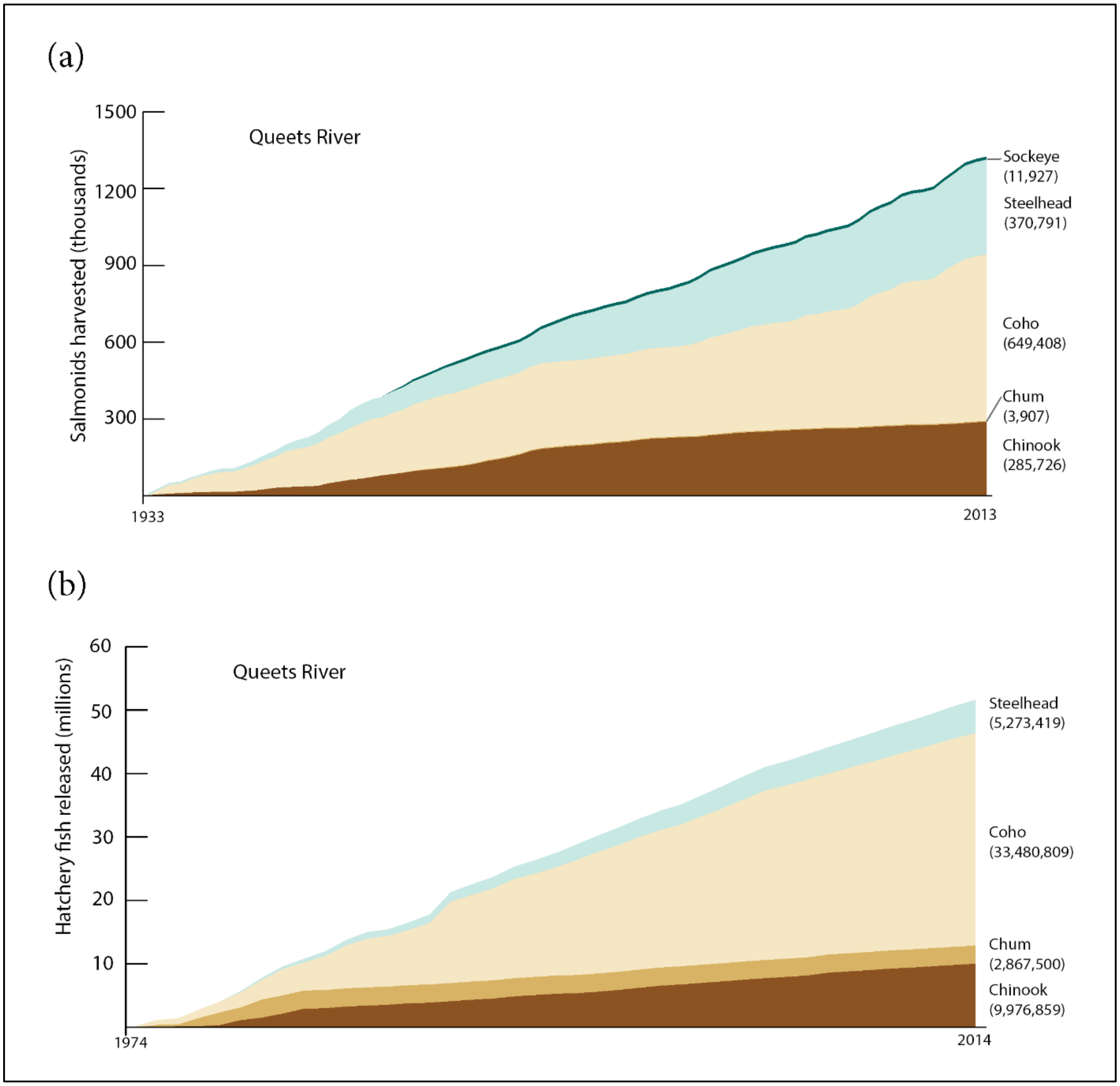

Figure 4.2.7. (a) Cumulative numbers of wild and hatchery Pacific salmonids harvested in commercial, ceremonial/subsistence, and sport fisheries in the Queets River system from 1933 to 2013 and (b) cumulative numbers of hatchery raised Pacific salmon released into the Queets River system from 1973 to 2014. Data were derived from hatchery and harvest records summarized in Appendix F.

Fall Chinook

Queets fall Chinook were classified as healthy (SASSI 1994) and in good reproductive status (Huntington et al. 1994). The established escapement goal for Queets wild fall Chinook was 2,500 fish (or 60 % of the run) and escapement averaged 4,342 fish from 1967 – 2013 (Figure 4.2.6). Estimates of escapement ranged from a high of 10,569 fish in 1990 to a low of 872 fish in 2007 (SD 2188). The escapement goal was met in 40 of 47 years since 1967 (85% of years) and in 9 of the last 10 years (Figure 4.2.6).

From 1976 to 2013, the mean terminal run size was 5,396 wild fish (range 1,288 – 12,297; SD 2,439), with an average annual percent change of -1.2 % over the period of record (Table 4.2.3). Since 1976, 66,117 wild Chinook were harvested in commercial fisheries (mean 1,740, range 262 – 3,851, SD 101) and 6,044 (mean 163, range 0 – 441; SD 122) in recreational fisheries, respectively (cumulative harvest data since 1933 shown in Figure 4.2.7a). The mean annual harvest rate of total run size for hatchery and wild Chinook was 32% (range 7 – 53%; SD 10%). Escapement for fall Chinook was estimated from redd counts from October 15 to December 1. Surveys were conducted by foot and boat and estimates were derived using index timing curves. The total natural spawning escapement was calculated assuming 2.5 fish per redd. The escapement floor policy of 2,500 for the Queets fall Chinook salmon was developed by Cooney (1984) and QDNR (1982) based on spawner–recruit analyses.

Fall Coho

Queets coho were classified as healthy in SASSI (1994). The established escapement goal for Queets wild coho is 5,800 to 14,500 fish (co-managers have set an escapement goal range for this stock) and escapement averaged 6,733 fish from 1976 to 2012 (Figure 4.2.6). Estimates of escapement ranged from a high of 24,953 fish in 2001 to a low of 1,200 fish in 1976 and 1994 (SD 4,451). The lower value escapement goal was met in 17 of 37 years since 1976 (46% of years) and in 6 of the last 10 years (Figure 4.2.6).

From 1976 to 2012, the mean terminal run size was 8,896 wild fish (range 1,370 – 30,661; SD 5,667), with an average annual percent change of 2.7 % over the period of record (Table 4.2.3). Since 1976, 289,747 wild and hatchery coho were harvested in commercial fisheries (mean 7,625, range 1,000 – 25,004, SD 6,713) and 14,132 hatchery and wild coho (mean 382, range 18 – 1,554; SD 414) in recreational fisheries, respectively (cumulative harvest data since 1933 shown in Figure 4.2.7a). The mean annual harvest rate of total run size of hatchery and wild fish was 42% (range 9 of total run size 68%; SD 14%).

Winter Steelhead

Queets winter steelhead were classified as healthy in SASSI (1994). Escapement goals differ between the State and the QIN. The WDFW escapement goal for wild winter steelhead in the Queets system is 4,200 fish (2748 for Queets only) and the QIN goal is 2,500 for the Queets system. Wild winter steelhead escapement averaged 4,840 fish from 1971 to 2013 (Figure 4.2.6). Estimates of escapement ranged from a high of 8,215 fish in 1989 to a low of 1,571 fish in 1975 (SD 1489). The WDFW escapement goal was met in 28 of 43 years since 1976 (65% of years) and in 6 of the last 10 years (Figure 4.2.6).

From 1976 to 2013, the mean terminal run size was 7,785 wild fish (ranged 4,240 – 13,280; SD 2,174), with an average annual percent change of -1.3 % over the period of record (Table 4.2.3). Since 1976, 90,897 wild steelhead (mean 2,392, range 425 – 5,498; SD 1,171) and 50,052 hatchery steelhead (mean 1,472, range 298 – 3,308; SD 766) were harvested in commercial fisheries. Since 1976, 13,488 wild winter steelhead (mean 355, ranged from 19 to 1,173; SD 279) and 21,453 hatchery steelhead (mean 631, range 71 – 2,014; SD 448) were harvested in recreational fisheries. The mean annual harvest rate of total run size of wild steelhead was 35% (range 9% – 54%; SD

11%). The ratio of hatchery:wild winter steelhead in the commercial fishery was 0.7:1 and in the sport fishery was 4.7:1.

Summary of Hatchery Releases

Annual hatchery releases of salmonids have occurred in the Queets system since 1974. The Salmon River Hatchery is the only facility located within the Queets system. The hatchery is owned and operated by QIN and is located at rkm 6.4 on the Salmon River, a tributary to the lower Queets River. The facility is operated for egg-take, spawning, incubation, and rearing of coho, Chinook, and winter steelhead. The facility annually releases 450,000 Chinook, 650,000 coho, and 150,000 winter steelhead into the Salmon River. Coho reared at the facility are of early-timed stock from the Quinault National Fish Hatchery.

Hatchery releases occurred throughout the main stem Queets and tributaries. From 1974 to 2014, 51.6 million salmonids were released into the system including about: 5.2 million winter steelhead (36 years), 9.5 million fall Chinook (40 years), 0.3 million spring Chinook (2 years), and 10.4 million fall coho (14 years) (Figure 4.2.7b; Appendix G). Additionally, there were unspecific run-type origins that included: 0.02 million steelhead (10 years), 0.2 million Chinook (1 year), 23 million coho (28 years), and 2.9 million chum (4 years; Figure 4.2.7b; Appendix G).

For hatchery releases into the Queets River system, there were an estimated 37 sources of non-indigenous stock origin among the salmonid species (Appendix G).

Quillayute River System

Watershed Overview

The Quillayute River drains 790 km^2 and flows in a westerly direction for 9 km to its confluence with the Pacific Ocean at La Push. Four large river systems drain into the Quillayute River including the Dickey, Sol Duc, Bogachiel, and Calawah Rivers. One hundred named tributaries flow into the Quillayute system.

Fish Assemblage Overview

Fish species that inhabit the Quillayute River system include spring/summer and fall Chinook salmon, summer and fall coho salmon, chum salmon, sockeye salmon, summer and winter steelhead trout, cutthroat trout, Dolly Varden (*Salvelinus malma*), sculpin, mountain whitefish, longnose dace, Pacific lamprey, and nonnative brook trout (*Salvelinus fontinalis*). There are no federally listed fish species in the Quillayute system.

Sol Duc spring Chinook are primarily of hatchery origin and enter the river from early April to late June with spawning from late August to October (SASSI 1994). Quillayute summer Chinook are primarily of wild origin and enter the river from April to mid-August with spawning from late August to mid-October (SASSI 1994). The majority of spawning is in the main stem Sol Duc, Bogachiel, and Calawah Rivers.

Quillayute fall Chinook are comprised of four stocks that include fish from the Bogachiel, Calawah, Sol Duc, and Dickey Rivers (SASSI 1994). Fall Chinook are of native origin and enter the river from September to late November, and spawning occurs from October to December (SASSI 1994).

Quillayute summer coho (Sol Duc) are of hatchery and wild origin. The stock is managed for hatchery production, which accounts for a majority (70-90%) of the terminal run to the river (SASSI 1994). Summer coho enter the river from July to mid-September and spawn from September to December upstream of river km 98 in OLYM. Quillayute fall coho are comprised of four stocks that include fish from the Bogachiel, Calawah, Sol Duc, and Dickey Rivers (SASSI 1994). Quillayute fall coho are of hatchery and wild origin and enter the river from mid-September to early December with spawning occurring from mid-November to mid-January (Quileute Tribal Fisheries Management 1979, SASSI 1994). Winter steelhead in the Quillayute system are of hatchery and wild origin. Wild winter steelhead are comprised of four stocks that include Bogachiel, Dickey, Sol Duc and Calawah River. Fish enter the river from December to June and spawn from mid-February to mid-June (SASSI 1994, McMillan et al. 2007).

Stock status and trends

Spring/Summer Chinook

Quillayute spring/summer Chinook were classified as healthy (Sol Duc) or unknown (Bogachiel, Calawah) in SASSI (1994). The established escapement goal for Quillayute spring/summer Chinook is 1,200 fish (adults and jacks combined) and escapement averaged 1,165 wild fish from 1976 to 2013 (Figure 4.2.8). Estimates of escapement ranged from a high of 3,800 fish in 1977 to a low of 502 fish in 2007 (SD 680). The escapement goal was met in 12 of 39 years since 1976 (31% of years) and in 0 of the last 10 years (Figure 4.2.8).

From 1988 to 2013, the mean terminal run size of wild Chinook was 1,370 fish (range 568 – 3,943; SD 771), with an average annual percent change of -4.4 % over the period of record (Table 4.2.3). Since 1976, 35,079 wild and hatchery Chinook were harvested in commercial fisheries (mean 948, range 100 – 3,400, SD 876) and 9,571 (mean 266, range 18 – 800; SD 175) in recreational fisheries, respectively (cumulative harvest data since 1933 shown in Figure 4.2.9). The mean annual harvest rate for wild and hatchery Chinook was 36% (range 12% – 59% of total run size; SD 12%).

Escapement for this stock is estimated from redd counts from index (weekly surveys) and supplemental areas (surveyed at peak of spawning) in the Bogachiel, main stem Calawah, North Fork Calawah and Sitkum Rivers. This has been used consistently in the Quillayute River System since the 1970s (Pacific Salmon Commission 2014). Surveys are conducted by foot, raft, boat, and helicopter. Using an appropriate redds per mile assignment, the information from index and supplemental surveys is then applied to other streams and segments that have historically had fish presence, but were not surveyed. These areas comprise the Quillayute River system stream mileage base that is consistently calculated to estimate escapement numbers. The number of redds is multiplied by 2.5 to estimate fish escapement.

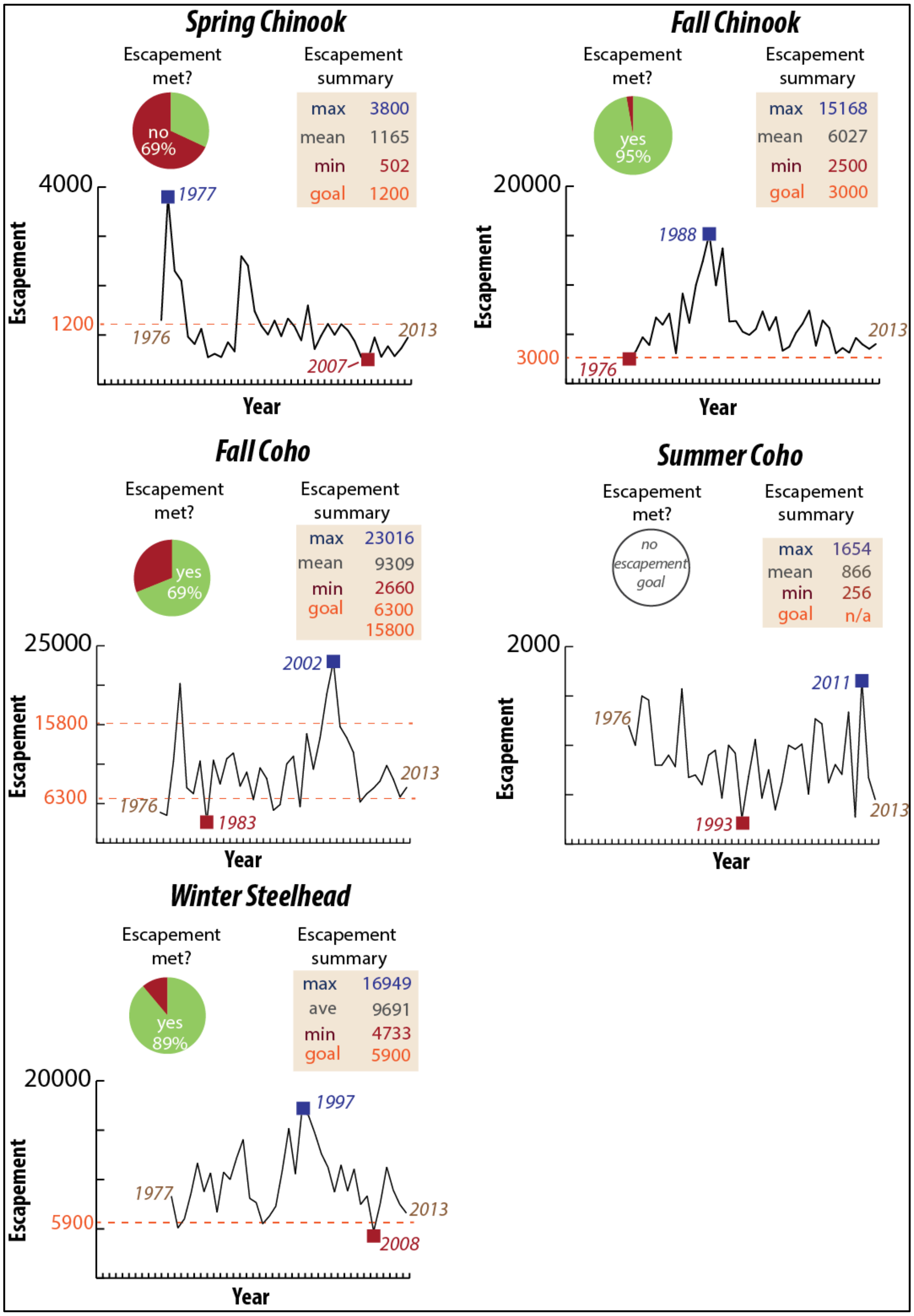

Figure 4.2.8. Trends in annual escapement levels of adult fall Chinook salmon, spring Chinook salmon, fall Coho salmon, summer Coho, and winter steelhead in the Quillayute River system. Each line graph depicts trends in annual escapement of wild fish during the period of record with the dotted line denoting the escapement goals set by fisheries co-managers. Pie charts show the percent of years the escapement goal was met (green) or missed (red) for each stock. Data were derived from escapement records summarized in Appendix H.

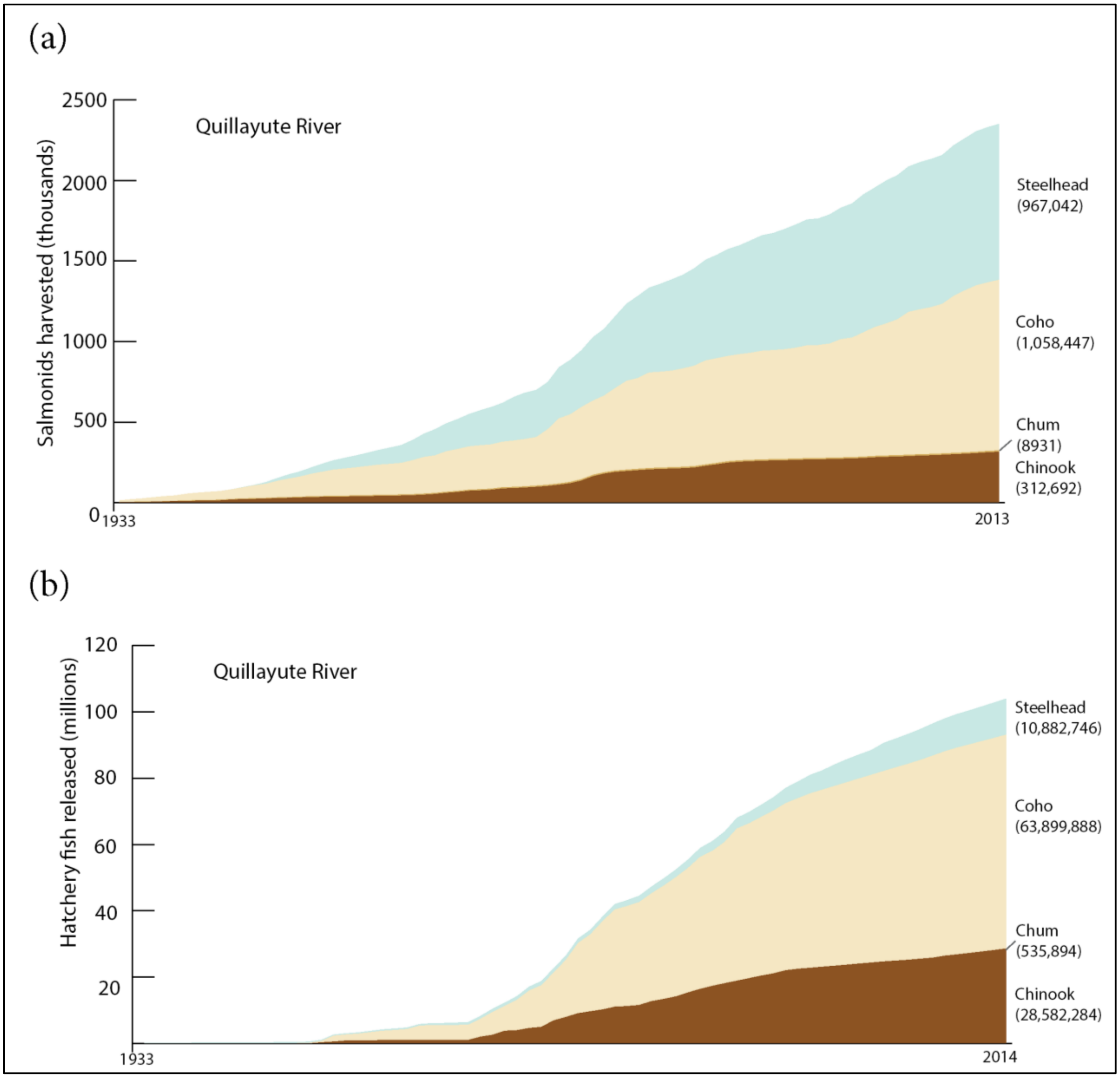

Figure 4.2.9. (a) Cumulative numbers of wild and hatchery Pacific salmonids harvested in commercial, ceremonial/subsistence, and sport fisheries in the Quillayute River system from 1933 to 2013 and (b) cumulative numbers of hatchery raised Pacific salmon released into the Quillayute River system from 1934 to 2014. The Quillayute watershed totals reflect data from major rivers including the Bogachiel, Sol Duc, Calawah, and Dickey rivers. Data were derived from hatchery and harvest records summarized in Appendix F.

Fall Chinook

Quillayute fall Chinook were classified as healthy in SASSI (1994). The established escapement goal is 3,000 fish (or 60% of the run) and escapement averaged 6,027 wild fish from 1976 to 2013 (Figure 4.2.8). Estimates of escapement ranged from a high of 15,168 fish in 1988 to a low of 2,500 fish in 1976 (SD 2962). The escapement goal was met in 37 of 39 years since 1976 (97% of years) and in 10 of the last 10 years (Figure 4.2.8).

From 1976 to 2013, the mean terminal run size was 8,535 wild fish (ranged from 4,137 to 22,115; SD 4202), with no significant trend over time (Table 4.2.3). Since 1976, 79,745 hatchery and wild Chinook were harvested in commercial fisheries (mean 2,155, range 282 – 7,996, SD 1,804) and 12,721 (mean 353, range 26 – 949; SD 226) in recreational fisheries, respectively. The mean annual harvest rate of total run size for hatchery and wild Chinook was 28% (range 8% – 61%; SD 13%). Terminal fisheries are managed for a harvest rate of 40%.

Escapement for this stock was estimated from redd counts in index (weekly surveys) and supplemental areas (surveyed at peak of spawning) in the Bogachiel, Sol Duc, Dickey, and Calawah Rivers and several smaller tributaries in the basin. This has been used consistently in the Quillayute River System since the 1970s (Pacific Salmon Commission 2014). Surveys are conducted by foot, raft, boat, and helicopter. Using an appropriate redds per mile assignment, the information from index and supplemental surveys is then applied to other streams and segments that have historically had fish presence, but were not surveyed. These areas comprise the Quillayute River system stream mileage base that is consistently calculated to estimate escapement numbers. The number of redds is multiplied by 2.5 to estimate fish escapement. The escapement goal for Quillayute fall Chinook salmon of 3,000 natural spawners is based on a spawner–recruit analysis developed by Cooney (1984) and QDNR (1982). Stock production analyses of spawning escapements from 1968 to 1982 were used to determine the initial escapement floor.

Summer Coho

Sol Duc summer coho were classified as healthy in SASSI (1994). There is no established escapement goal for Sol Doc wild summer coho. Escapement averaged 866 wild fish from 1976 to 2013 (Figure 4.2.8). Estimates of escapement ranged from a high of 1,654 fish in 2011 to a low of 256 fish in 1993 (SD 354).

From 1976 to 2013, the mean terminal run size was 1,505 wild fish (range 324 – 4,020; SD 805), with an average annual percent change of -1.4 % over the period of record (Table 4.2.3). Since 1976, 104,150 wild and hatchery coho were harvested in commercial fisheries (mean 2,815, range 70 – 11,985; SD 3,286) and 7,298 (mean 203, range 14 – 590; SD 163) in recreational fisheries, respectively. The mean annual harvest rate of the total run size for hatchery and wild coho was 32% (ranged 4% – 84%; SD 19%).

Fall Coho

Quillayute fall coho are comprised of fish that spawn in the Dickey, Sol Duc, Bogachiel, and Calawah Rivers. All stocks were classified as healthy in SASSI (1994). The established escapement goal is 6,300 to 15,800 wild fish (co-managers have set an escapement goal range for this stock). The hatchery escapement goal is 600 fish. Escapement of wild fall coho averaged 9,309 fish from 1976 to 2013 (Figure 4.2.8). Estimates of escapement ranged from a high of 23,016 fish in 2002 to a low of 2,660 fish in 1983 (SD 4549). The escapement goal of 6,300 fish was met in 27 of 39 years since 1976 (69% of years) and in 8 of the last 10 years (Figure 4.2.8).

From 1976 to 2013, the mean terminal run size was 15,864 wild fish (range 3,322 – 36,714; SD 7,761; Figure 4.2.3), with an average annual percent change of 4.9 % over the period of record

(Table 4.2.3). Since 1976, 383,874 wild and hatchery coho were harvested in commercial fisheries (mean 10,375, range 456 – 37,112; SD 9,151) and 39,102 (mean 1,086, range 18 – 4,260; SD 1,255) in recreational fisheries, respectively. The mean annual harvest rate of total run size for hatchery and wild coho was 35% (range 4% – 66%; SD 15%).

Winter Steelhead

Quillayute winter steelhead are comprised of fish that spawn in the Dickey, Sol Duc, Bogachiel, and Calawah Rivers. All stocks were classified as healthy in SASSI (1994). The established escapement goal for the Quillayute system is 5,900 wild fish. Specific escapement goals for each river are as follows: Dickey 123 fish, Sol Duc 2,910 fish, Bogachiel 1,127 fish, and Calawah 1,740 fish. Escapement of wild winter steelhead in the Quillayute system averaged 9,691 fish from 1977 to 2013 (Figure 4.2.8). Estimates of escapement ranged from a high of 16,949 fish in 1997 to a low of 4,733 fish in 2008 (SD 3109). The escapement goal of 5,900 fish was met in 33 of 37 years since 1977 (89% of years) and in 9 of the last 10 years (Figure 4.2.8).

From 1977 to 2013, the mean terminal run size was 14,877 wild fish (range 6,200 – 22,650; SD 3,981; Figure 4.2.3), with an average annual percent change of -1.2 % over the period of record (Table 4.2.3). Since 1977, 93,079 wild steelhead were harvested in commercial fisheries (mean 2,516, range 701 – 5,435; SD 1127) and 135,376 hatchery steelhead (mean 3,659, range 758 – 8,605; SD 1,930). 56,508 wild steelhead were harvested in recreational fisheries (mean 1,527, range 174 – 3,436; SD 925) and 117,157 hatchery steelhead (mean 3,166, range 677 – 6,518; SD 1,464). The mean annual harvest rate of total run size for wild steelhead was 28% (range 10% – 59%; SD 11%). The mean annual harvest rate of hatchery steelhead was 64% (range 32% – 86% of total run size; SD 12%). The proportion of hatchery: wild winter steelhead in the commercial fishery was 2.1:1 and in the sport fishery was 3.3:1.

Summary of Hatchery Releases

Multiple hatchery facilities operated by the Quileute Tribe and WDFW occur in the Quillayute system. The Quileute Tribe operates the Bear Springs Pond located on rkm 58 on the Sol Duc River, 8 km north of the Sol Duc Hatchery. The single pond rears 50,000 Chinook for later release into the Sol Duc River. Juvenile Chinook for the facility come from the Lonesome Creek Hatchery located in La Push. The WDFW Sol Duc Hatchery is located on the Sol Duc River north of Forks at rkm 48. The hatchery has facilities for egg-take, spawning, incubation, and rearing for Chinook, coho, and winter steelhead. The WDFW Bogachiel Hatchery began operation in 1976. The facility is used for egg-take, spawning, incubation, and rearing of summer and winter steelhead. Each year, the hatchery releases summer steelhead into the Calawah, Bogachiel, and Sol Duc Rivers. The Snider Creek Pond was operated by the Olympic Guides Association from 1986 to 2013 and raised winter steelhead at the confluence of Snider Creek and the Sol Duc River near rkm 71. The project harvested wild adults in the Sol Duc River and transported fish to the Sol Duc Hatchery for egg take and spawning.

Hatchery releases occurred throughout the Quillayute system. From 1933 to 2014, 103.9 million salmonids were released into the system including about: 9.2 million winter steelhead (57 years), 1.6 million summer steelhead (33 years), 6.2 million fall Chinook (22 years), 13.6 million spring Chinook (36 years), 15 million summer coho (38 years), and 0.7 million fall coho (4 years;

Figure 4.2.9b; Appendix G). Additionally, there were unspecific run-type origins that included: 0.1 million steelhead (3 years), 8.8 million Chinook (37 year), and 48.3 million coho (60 years; Figure 4.2.9b; Appendix G). For hatchery releases into the Quillayute system, there were an estimated 40 non-indigenous stock origins among the salmonid species (Appendix G).

Quinault River

Watershed Overview

The Quinault River originates from Anderson Glacier and drains 684 km^2 above the outlet of Lake Quinault (15 km^2). The river flows 111 km from the headwaters and descends from 2210 m in elevation to its confluence with the Pacific Ocean at Taholah Village. Sixty-four named tributaries flow into the Quinault River system. Major tributaries include the East Fork (234 km^2), North Fork (208 km^2), and Big and Graves Creeks. The entire East and North Forks of the Quinault are within the boundaries of the Olympic National Park. The river enters Lake Quinault and then flows for 53 km to the ocean.

Fish Assemblage Overview

Fish species that inhabit the Quinault River system include spring/summer and fall Chinook salmon, coho salmon, chum salmon, sockeye salmon, summer and winter steelhead trout, rainbow trout, cutthroat trout, bull trout, Dolly Varden, sculpin, mountain whitefish, longnose dace, largescale sucker (*Catostomus macrocheilus*), Pacific lamprey, threespine stickleback (*Gasterosteus aculeatus*), and carp (*Cyprinus carpio*). Bull trout are listed as federally threatened under the Endangered Species Act.

Quinault spring/summer Chinook are of wild origin and enter the river from mid-March to August with spawning from July to mid-October (QIN 1981, 2008). Quinault fall Chinook are of wild and hatchery origin and enter the river from August through November (QIN 2008), and spawning occurs from mid-October to early December (QIN 1981). Quinault coho are of hatchery and wild origin and enter the river from August through October and from early September to January, respectively. Most of the hatchery-origin coho return to the QNFH racks on Cook Creek and wild-origin coho spawn from October to January (QIN 2008). Winter steelhead enter the Quinault River from November through late May and spawn from January to May (QDNR 1982, QIN 2008). Quinault sockeye salmon historically entered the river from early December to June and spawn from November to March (QIN 2008).

Escapement, Total Run Size, and Harvest Status and Trends

Spring/Summer Chinook

We were unable to obtain escapement and terminal run size data for this stock. Quinault spring/summer Chinook were classified as depressed based on a short-term severe decline in escapement in SASSI (1994). There is no established escapement goal for Quinault wild spring/summer Chinook. Since 1976, 4,041 Chinook were harvested in commercial fisheries (mean 109, ranged from 0 to 530, SD 122; Figure 4.2.10).

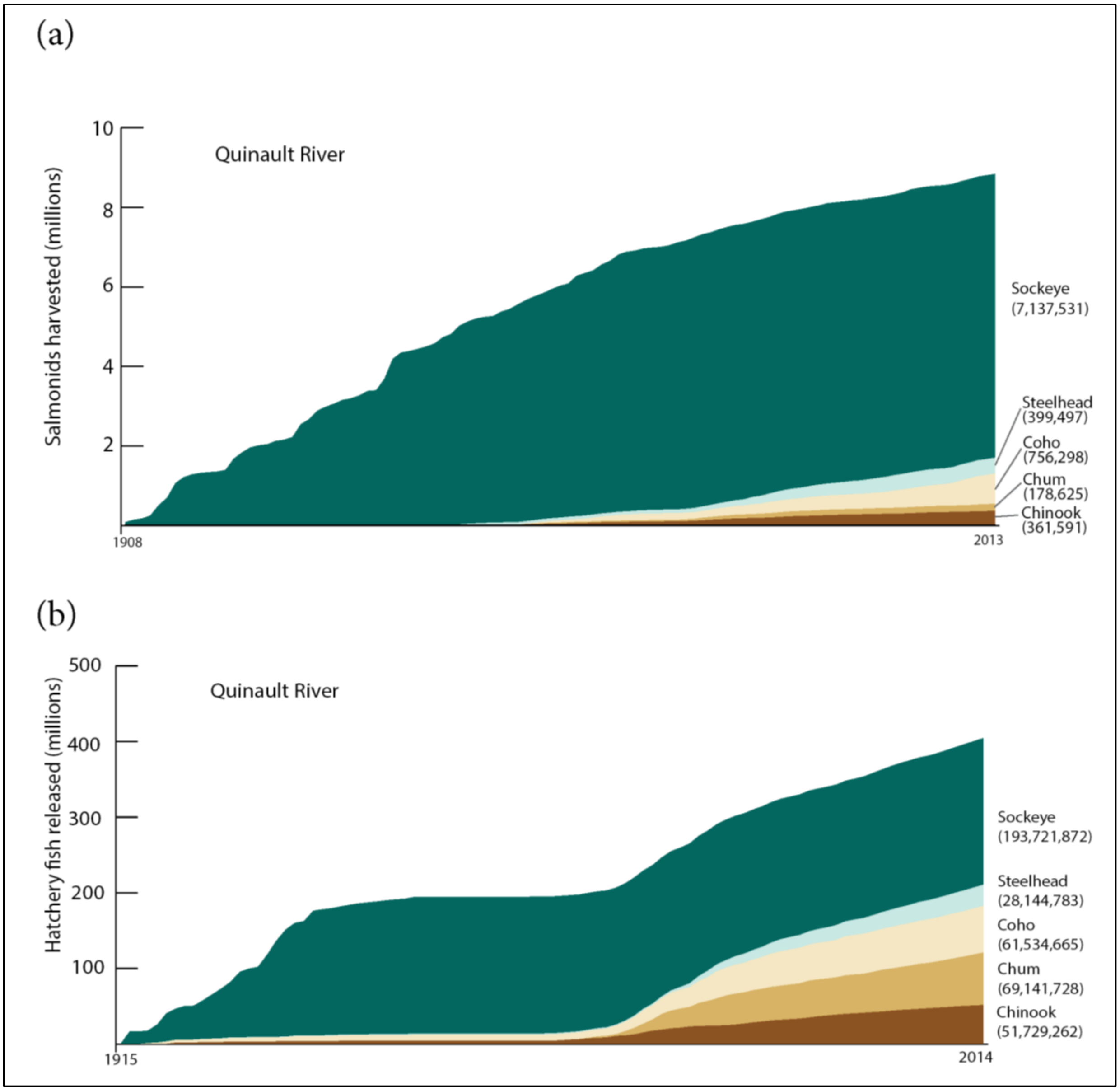

Figure 4.2.10. (a) Cumulative numbers of wild and hatchery Pacific salmonids harvested in commercial, ceremonial/subsistence, and sport fisheries in the Quinault River system from 1908 to 2013 and (b) cumulative numbers of hatchery raised Pacific salmon released into the Quillayute River system from 1915 to 2014. Data were derived from hatchery and harvest records summarized in Appendix F.

Fall Chinook

We were unable to obtain escapement and terminal run size data for this stock. Quinault fall Chinook were classified as healthy in SASSI (1994). There is no established escapement goal for Quinault wild Chinook. Since 1976, 217,013 hatchery and wild Chinook were harvested in commercial fisheries (mean 5,865, range 600 – 12,220, SD 2329).

Fall Coho

Quinault fall coho were classified as unknown in SASSI (1994). There is no established escapement goal for Quinault coho. Escapement averaged 7,454 wild fish from 1977 to 2012 (Figure 4.2.11). Estimates of escapement ranged from a high of 33,140 fish in 1989 to a low of 820 fish in 1994 (SD 7,640).

From 1977 to 2012, the mean terminal run size was 18,977 wild fish (range 600 – 66,094), with an average annual percent change of 4.9 % over the period of record (Table 4.2.3). Since 1976, 578,747 wild and hatchery coho were harvested in commercial fisheries (mean 15,642, range 365 – 54,882, SD 11,530) (Figure 4.2.11). From 2006 to 2012, 7,694 wild and hatchery coho were harvested in recreational fisheries (mean 1,099, range 325 – 2,047, SD 611). The mean annual harvest rate of total run size for hatchery and wild fish in the commercial fishery was 49% (range 8 – 76%; SD 14%).

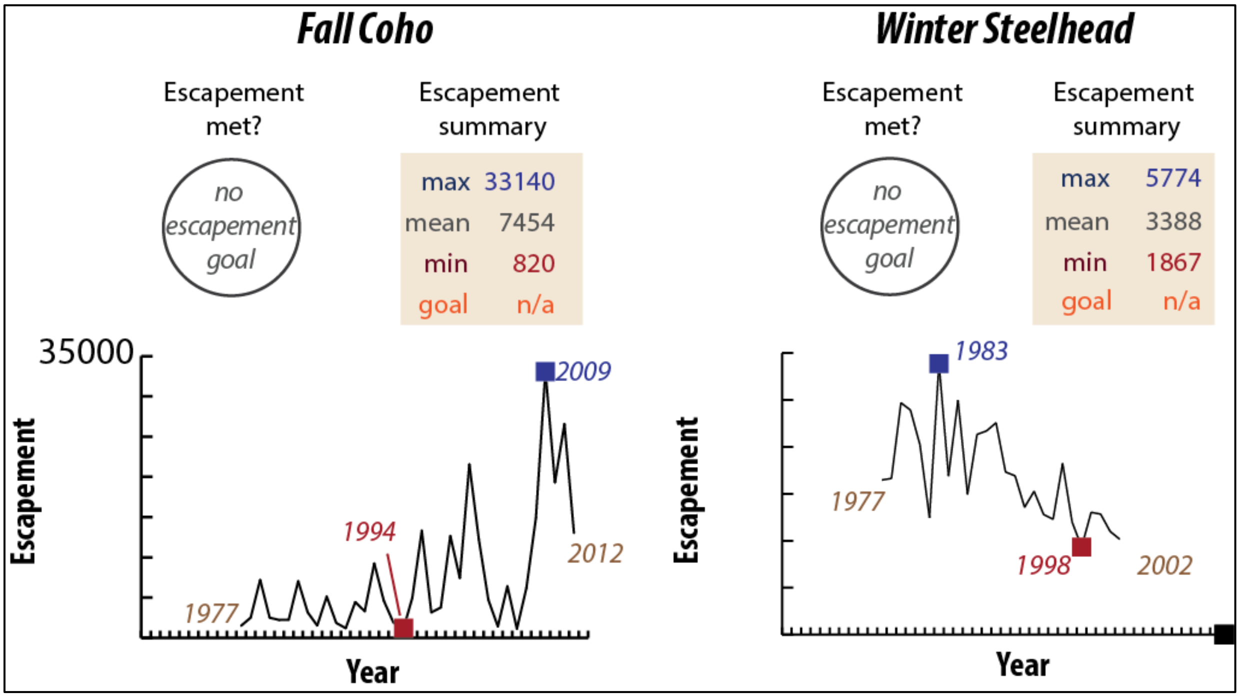

Figure 4.2.11. Trends in annual escapement levels of adult fall Coho salmon and winter steelhead in the Quinault River system. Each line graph depicts trends in annual escapement of wild fish during the period of record. There are no established escapement goals for fall Coho salmon and winter steelhead in the Quinault River. Data were derived from escapement records summarized in Appendix H.

Winter Steelhead

Quinault winter steelhead were classified healthy in SASSI (1994). There is no established escapement goal for Quinault winter steelhead. Escapement averaged 3,388 wild fish from 1977 to 2002 (Figure 4.2.11). Estimates of escapement ranged from a high of 5,774 fish in 1983 to a low of 1,867 fish in 1998.

From 1977 to 2002, the mean terminal run size was 6,640 wild fish (range 3,524 – 9,726; SD 1691), with an average annual percent change of -2.2 % over the period of record (Table 4.2.3). Since 1977, 75,279 wild steelhead (mean 2,788, range 670 – 6,068; SD 1192) and 191,408 hatchery steelhead (mean 7,089, range 1,068 – 15,979; SD 3,956) were harvested in commercial fisheries. 12,328 wild steelhead (mean 474, range 15 – 1,371; SD 301) and 18,007 hatchery steelhead (mean 692, range 61 – 1,656; SD 418) were harvested in recreational fisheries. The mean annual harvest rate of total run size for wild steelhead was 48% (range 15 – 65%; SD 11%). The mean annual harvest rate of hatchery steelhead was 67% (range 30 – 86%; SD 15%). The ratio of hatchery: wild winter steelhead in the commercial fishery was 2.9:1 and in the sport fishery was 3.0:1.

Summary of Hatchery Releases

Hatchery operations in the Quinault River system have a long history that can be partitioned into two time periods. The first period spanned from 1914 through 1947 with operation of the U.S. Bureau of Fisheries hatchery (Quinault Washington Station) at Falls Creek on Lake Quinault (Bountry et al. 2005). The second period spanned from 1968 to the present with operations at two hatcheries — the USFWS Quinault National Fish Hatchery at Cook Creek, a tributary of the lower Quinault River, and the Lake Quinault Tribal Hatchery on Lake Quinault.

The QIN owns and operates the Lake Quinault Tribal Hatchery located along the southwest shore of the lake. The facility began production in 1974 and is comprised of offshore floating net pens for raising Chinook, winter steelhead, and sockeye juveniles for release into Lake Quinault. The tribe also conducts egg takes and incubation for each species at the facility. In June, 2008, a Columbia River strain of infectious hematopoietic necrosis virus (IHNV, MD strain) was detected in juvenile Lake Quinault winter steelhead being reared in net pens in Lake Quinault. This was the first detection of this non-endemic strain, highly virulent to steelhead, in the Quinault River basin (USFWS 2009).

The USFWS Quinault National Fish Hatchery began hatchery production in 1968. The facility is for egg-take, spawning, incubation, and rearing of chum, coho, Chinook, and winter steelhead. The facility also supplies juvenile fish to the Lake Quinault Tribal Hatchery and to Chalaat Creek (Hoh) Hatchery. The Quinault National Fish Hatchery annually releases 1,500,000 chum, 600,000 Chinook, 803,000 coho, and 190,000 winter steelhead in Cook Creek. The hatchery also transferred pre-smolt steelhead annually to the Hoh River from the mid-1980s to around 2007.

Hatchery releases occurred at locations throughout the main stem Quinault, Lake Quinault, and tributaries. From 1915 to 2014, 404.3 million salmonids were released into the system including about: 193.7 million sockeye (63 years), 26.2 million winter steelhead (42 years), 51.0 million fall Chinook (69 years), 0.7 million spring Chinook (3 years), 9.7 million fall coho (29 years), and 69.1 million fall chum (45 years; Figure 4.2.10b; Appendix G). Additionally, there were unspecific run-type origins that included: 1.9 million steelhead (17 years) and 51.8 million coho (44 years; Figure 4.2.10b; Appendix G). For hatchery releases into the Quinault system, there were an estimated 59 sources that were of non-indigenous stock origin among the salmonid species (Appendix G).

Elwha River

Because of the large-scale dam removal and ecosystem restoration project occurring on the Elwha River, it was included in this section on salmon, although the available data are not comparable to the other four rivers present in this section. Thus, for some summary graphics (e.g., Figure 4.2.3) the

Elwha is not included. Nevertheless, we have included the Elwha River here as it will continue to be a focus of OLYM management as salmon populations respond to the project.

Watershed Overview

The Elwha River originates in OLYM and drains 833 km^2. The 6th-order river flows 72 km from glaciers and ice fields and descends from 1372 m at the headwaters to its confluence with the Strait of Juan de Fuca in the Pacific Ocean. Eighty-two percent of the watershed occurs in OLYM (19% of the area of the park) and is managed by the NPS as a wilderness area. The remaining portions of the river flow through State, private and tribal lands.

Two hydroelectric dams constructed in the early 1900s eliminated access for anadromous salmonids to 95% of the watershed. Removal of the Elwha Dam occurred in 2012 and the Glines Canyon Dam was removed in 2014. The geomorphology of the river is a series of alternating canyons and floodplains (Duda et al. 2008). The major canyons from mouth to headwaters include Elwha Canyon (1.7 rkm in length), Glines Canyon (0.8 rkm), Rica Canyon (1.9 rkm), Grand Canyon (5.5 rkm), an unnamed canyon (1.2 rkm) and Carlson Canyon (2.3 rkm). Forty-nine named tributaries flow into the Elwha River. Mean daily discharge of the Elwha River is 42 m^3/s, with annual minimum flows ranging from 8.5 to 14 m^3/s during summer (Curran et al. 2009).

Fish Assemblage Overview

Fish species that inhabit the Elwha River system include summer/fall Chinook salmon, coho salmon, pink salmon, chum salmon, sockeye and kokanee (landlocked version of sockeye) salmon, summer and winter steelhead trout, coastal cutthroat trout, bull trout, Dolly Varden, sculpin, threespine stickleback, Pacific lamprey, river lamprey (*Lampetra ayresii*), redside shiner (*Richardsonius balteatus*), eulachon (*Thaleichthys pacificus*), and non-native brook trout. Chinook salmon, steelhead, bull trout and eulachon are listed as federally threatened species under the Endangered Species Act (Brenkman et al. 2012).

The salmonid community in the Elwha River is comprised of wild, natural, and hatchery origin fish. Elwha Chinook are of hatchery and natural origin, with hatchery supplementation of the Chinook runs occurring continuously since the mid-1930s. Currently, over 90% of the returning adult Chinook are of hatchery origin (Duda et al. 2011, Anderson et al. 2013). Adult Chinook enter the river from May to mid-September and spawning occurs from August through October, with peak redd counts in mid-September. Since dam removal, Chinook salmon spawn in the main stem river and side channels downstream of the Glines Canyon Dam site and in Indian Creek and Little River.

Elwha coho are of hatchery and natural origin and enter the river from mid-September through December. Coho spawning begins in October and continues into January. Since dam removal, coho salmon spawn in the main stem river and side channels downstream of Rica Canyon and in Little River and Indian, Madison, Griff, Hughes, and Boulder Creeks. Winter steelhead in the Elwha River are of wild, natural, and hatchery origin, with the most recent hatchery component being wild captive broodstock. Native Elwha steelhead enter the river from January through June and typically spawn from March to early July (McMillan et al. 2015). Since the removal of the Elwha and Glines Canyon Dams, steelhead spawned in the main stem river upstream to Glines Canyon Dam site and a few adults were observed upstream of that site. Spawning also occurs in Little River and Indian, Hughes, and Griff Creeks. There have been no commercial or sport fisheries in the Elwha River from 2012 until the writing of this report (October 2016) as part of an agreement reached between the fisheries co-managers and the NPS to assist in restoration.

Escapement, Total Run Size, and Harvest Status and Trends

Chinook

Elwha Chinook were classified as depressed based on chronically low production (SASSI 1994). The established escapement goal for Elwha Chinook prior to dam removal was 500 naturally spawning fish and escapement averaged 1,253 fish from 1986 to 2014. Estimates of escapement ranged from a high of 5,228 fish in 1988 to a low of 163 fish in 1994 (SD 1,112). The escapement goal was met in 18 of 28 years since 1986 (64% of years) (Figure 4.2.12).

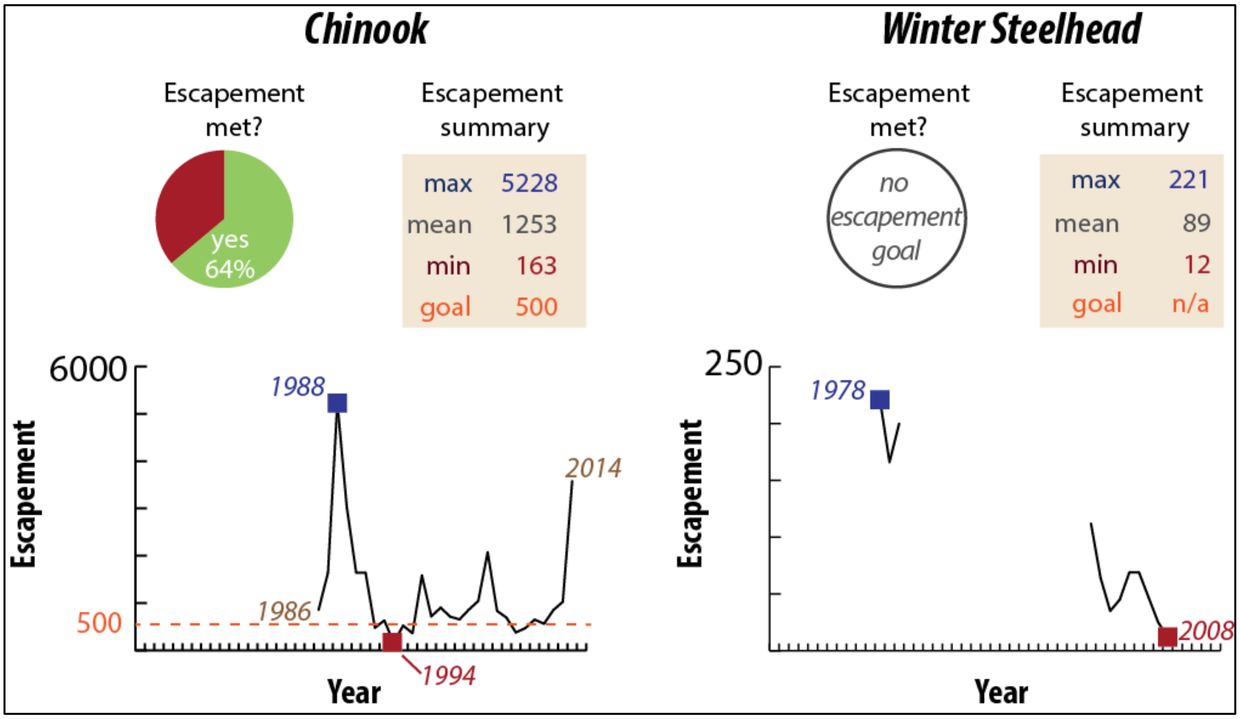

Figure 4.2.12. Trends in annual escapement levels of adult summer/fall Chinook salmon and winter steelhead in the Elwha River system. Each line graph depicts trends in annual escapement of wild fish during the period of record with the dotted line denoting the escapement goal set by the fisheries co-managers. Pie charts show the percent of years the escapement goal was met (green) or missed (red) for each stock. Data were derived from escapement records summarized in Appendix H.

From 1986 to 2013, the mean terminal run size was 2,865 Chinook salmon (both hatchery and wild combined; ranged from 1,141 to 8,667; SD 1,761). In fall 2005, 797 non-jack Chinook carcasses were sampled representing 20.6 % of the estimated escapement above the Elwha sonar site. Over 93% of the fish sampled were marked hatchery fish (Weinheimer et al. 2016). Since 1986, harvest in the Elwha River has been restricted and in very low numbers, with 2,411 hatchery and wild Chinook harvested during that period (mean 86, range 0 – 794, SD 180). The mean annual harvest rate of total run size for hatchery and wild Chinook was 2% (range 0% – 12%; SD 3.2%).

Fall Coho

Elwha fall coho were classified as healthy (SASSI 1994). There is no established escapement goal for Elwha wild coho. To date we have not found reliable terminal run size, escapement, and harvest data for Elwha coho salmon.

Winter Steelhead

Elwha winter steelhead were classified as depressed based on the chronically low production resulting from the loss of access to habitat from Elwha dams (SASSI 1994). There currently is no established escapement goal for Elwha wild steelhead. Estimates of escapement ranged from a high of 221 fish in 1978 to a low of 12 fish in 2008 (Figure 4.2.12).

From 1981 to 2011 (reported for 21 of 30 years) the mean total run size was 352 fish (range 45 – 835). We located harvest data of hatchery and wild fish from 1987 to 1999. During that time, 8,233 fish were harvested in sport fisheries (mean 633, range 84 – 2,299) and 10,513 fish were harvested in commercial fisheries (mean 809, 81 – 1,760).

Summary of Hatchery Releases

Elwha salmonid populations have been supplemented by hatcheries since 1915 (Brenkman et al. 2008a). There are two hatcheries on the Elwha River administered by WDFW (rkm 5.6) and the LEKT that have operated since 1976 and 1978, respectively. The tribal hatchery, located at rkm 1.6 operated from 1978 to 2014 and a new facility that began operation in 2014. Hatchery programs currently produce coho salmon (LEKT), Chinook salmon (WDFW), chum salmon (LEKT), pink salmon (LEKT and WDFW) and winter steelhead trout (LEKT). The Elwha Chinook hatchery program is integrated and incubation occurs at Hurd Creek (in the adjacent Dungeness River) and rearing occurs at Sol Duc River hatchery (http://wdfw.wa.gov/publications/01561/wdfw01561.pdf, accessed 30 October 2017).

Hatchery releases occurred throughout the main stem Elwha and tributaries. From 1916 to 2014, approximately 159.5 million salmonids were released into the system including about: 4.3 million winter steelhead (49 years), 0.5 million summer steelhead (27 years), 1.7 million spring Chinook (6 years), 104.8 million summer/fall Chinook (66 years), 5.9 million fall coho (13 years), 0.5 million chum (14 years), and 0.2 million pink (2 years; Figure 4.2.13; Appendix G). Additionally, there were unspecific run-type origins that included 0.4 million steelhead (2 years; Figure 4.2.13; Appendix G). For hatchery releases into the Elwha system, there were an estimated 21 sources of non-indigenous stock origin among the salmonid species (Appendix G).

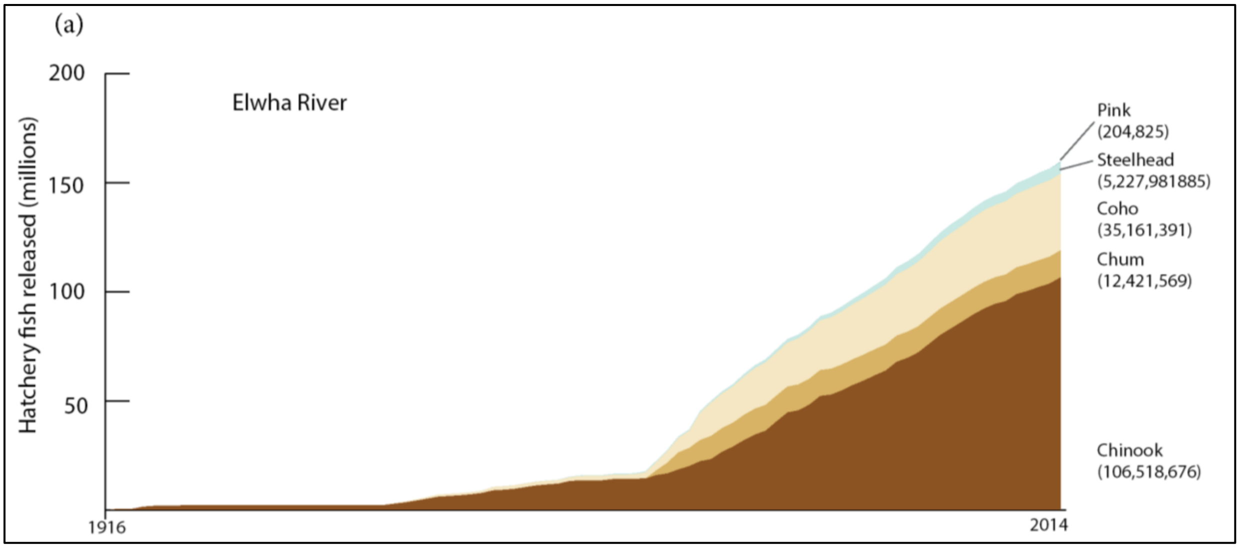

Figure 4.2.13. Cumulative number of hatchery origin Pacific Salmon released into the Elwha River system from 1916 to 2014. Data were derived from hatchery records summarize in Appendix F.

Uses and Limitations

We have compiled multiple types of long-term empirical information to evaluate the conditions of selected salmon stocks within OLYM. Because these data come from multiple sources, some easily available and others harder to locate, the exercise in data gathering in itself should prove useful to set a contemporary baseline of information spanning the recent decades since about the mid-1970s. For hatcheries and some historical harvest information, the record goes back even farther. It has also allowed us to identify important data gaps, especially in a context of useful information needs important to management. We also recognized that multiple lines of evidence were needed to accurately assess the status of stocks, and to identify additional data and management needs.

Discerning the status and trends of the fish stocks discussed in this chapter is difficult without a solid baseline condition for comparison. Most records on fish escapement, terminal run size, and number of fish harvested begin in the mid-20th century, well after stocks had been affected by human influences (i.e., the 4 H's). We used the best available data, which in many cases had a starting point in the mid-1970s. Having this be the "start" of the trendline (e.g., Figure 4.2.3) is problematic, because it suffers from what Pauly (1995) labeled a shifting baseline syndrome. This refers to a situation where fisheries biologists use the status of fish populations at the beginning of their career as the "baseline" to assess changes against, which is not necessarily reflective of a natural or historical state.

Placed into a context of escapement goals and the role that salmon nutrients have for the ecosystem, the potential implications of the shifting baseline concept come into focus. Recently, there has been a greater appreciation for the role that salmon play in providing nutrients into freshwater ecosystems—that are often nutrient limited—as well as riparian ecosystems (e.g., Cederholm et al. 1999). The pulse of these so-called "marine derived nutrients" (hereafter MDN) from spawning salmon can affect the growth rates and population densities of algae, aquatic invertebrates, and juvenile fish in the freshwater environment, as well as bears, eagles, and trees in the terrestrial environment (e.g., Kline et al. 1990, Schuldt and Hershey 1995, Bilby et al. 1996, Ben-David et al. 1998, Wipfli et al. 1998, Gende et al. 2002, Helfield and Naiman 2002, Wipfli et al. 2003). The transport of nutrients across ecosystem boundaries, from marine to freshwater and riparian, can represent a significant amount of the total nutrient loading of a riverine ecosystem (Stockner 2003 and references therein), affecting the dynamic cross boundary exchanges important within many riverine ecosystems (e.g., Nakano and Murakami 2001, Power 2001).

A key open question still facing fisheries scientists and managers is the degree to which salmon nutrient enrichment of freshwaters affects escapement levels and ecosystem productivity. Bilby et al. (2001) showed that streams with higher escapement levels increased the enrichment by salmon derived nitrogen in coho parr. Additionally, Knudsen et al. (2003) used a statistical simulation model to show that decreased escapement levels reduced the positive effects of marine derived nutrients to population growth. However, a significant and large-scale study has not yet been performed that demonstrates the direct role that MDN have in increasing escapement levels. As Larkin and Slaney (1997) state, "*Isotope signatures only reveal the movement of nutrients through the food chain and do not establish that nutrients from any source drive production*." If this link is ever explicitly demonstrated, it would make a stronger case for setting ecosystem based escapement goals rather than of maximum sustained yield (MSY).

Emerging Issues

Genetic Issues

Some level of genetic change relative to the natural population cannot be avoided in a cultured population (Waples 1999). A growing body of scientific literature has shown that hatchery-raised fish can have a number of possible impacts on native fish populations caused by competition, interbreeding, artificial selection, disruption of natural and sexual selection, and reduced genetic variation (e.g., Reisenbichler and McIntyre 1977, Chilcote et al. 1986, Reisenbichler and Rubin 1999, Araki et al. 2007, Christie et al. 2012). This has led to a reassessment of hatchery programs, and attempts to institute reforms of hatchery management practices to mitigate these negative effects (Natural Research Council 1996, Hatchery Scientific Review Group 2004, Mobrand et al. 2005, Kostow 2009). For example, the Elwha River has recently been designated as a wild steelhead gene bank, prohibiting the release of steelhead raised in state of Washington hatcheries.

Fish Pathology

Despite the obvious role fish diseases play in regulating salmon populations, the impacts of fish health typically do not appear in salmon recovery planning and restoration efforts (Hershberger et al. 2013). Little is known about how disease ecology interacts with factors such as population size, environmental stressors, habitat loss and alteration, and climate change; thus how these processes might impact future salmon populations. Because of the potential for catastrophic impacts, common fish pathogens are monitored at fish hatcheries on the Olympic Peninsula, but surveillance in wild, free-ranging populations is limited (Hershberger et al. 2013). Brenkman et al. (2008a) conducted an assessment of wild salmonid fish pathology in the Elwha River prior to dam removal. Such routine monitoring programs, while rare, could be instrumental in understanding current pathogen and disease profiles of important OLYM salmon populations in other rivers discussed in this report. A novel strain of IHNV previously located mostly in the Columbia River system has been detected in coastal steelhead stocks of the Olympic Peninsula (Breyta et al. 2013). How this and other disease issues emerge and impact salmon populations could be included in future management plans.

Climatic Impacts

Simulations predict increasing water temperatures and increasing thermal stress for salmon in western Washington that are slight for the 2020s and increasingly large later in the 21st Century (Mantua et al. 2010). By the 2080s, simulations predict a complete loss of snowmelt dominant basins in Washington. Rainfall-dominant systems will experience longer summer low-flow periods, increased streamflow in winter and early spring, declines in magnitude of summer low flows, and increases in winter flooding. The combined effects will likely reduce reproductive success and life history diversity for many salmon populations in Washington, including the Olympic Peninsula, through reduction in egg to fry survival, decreasing bioenergetic efficiencies, and changing food supplies (e.g., McCarthy et al. 2009, Benjamin et al. 2013). These climate change effects will likely vary according to both life history types and watershed types (Mantua et al. 2009).

MSY

The classical fisheries stock concept does not account for different productivity of the individual stocks or the variation in habitat quality from, for example, different tributaries or different environmental condition through time (e.g., Hilborn et al. 2003). A key element of fisheries management, often overlooked when operating under the prevailing salmon management paradigm, is curtailing of fishing effort when there is a recruitment failure, sub-optimal environmental conditions, or a decline in run size below forecasted levels used to set escapement and/or harvest goals. Additionally, theoretical deficiencies in spawner/recruit models and poor data quality on escapement levels can contribute to salmon population decline (Knudsen 2000). It is these and other problems that have led to questions of the utility of using MSY for fisheries management (e.g., Larkin 1977, Ludwig et al. 1993, Pauly et al. 2002). Exploring how escapement levels are currently performing, how to assess the level of risk for under-escaping and over-escaping fish runs with current escapement goals, and the role that modeling can play to better inform management are all issues worth considering in the future.

Information Needs/Gaps

Establishment of Escapement Goals

There is a need to establish escapement goals for selected wild salmon populations that inhabit rivers that drain from OLYM. From this report, we identified the following stocks that are managed for commercial and sport harvest without identified escapement goals: Sol Duc summer coho, Quinault chum, Quinault fall Chinook, Quinault spring/summer Chinook, and Quinault winter steelhead. Additionally, Elwha chum, pink, and sockeye salmon have no established escapement goals, although harvest has not occurred in several years due to the fishing moratorium associated with removal of the Elwha River dams.

Ohlberger et al. (2017) provided a current evaluation of productivity and abundance of selected Olympic Peninsula salmonid stocks using species specific life-history models to assess productivity and carrying capacity of the stocks. Estimates of escapement levels at MSY were provided and included a quantification of uncertainty for a range of different escapement goals. That report serves as a possible building block in the development of future escapement goals for stocks without established goals. The probability distribution for escapement at MSY can be used to select a desired level of confidence when defining or setting those goals.

Increased Monitoring and Harvest Management Considerations

To date, monitoring of wild salmonid escapement in OLYM has been limited by funding and logistical constraints. Knudsen (2000) discusses the shortcomings and consequences of escapement management based on limited information or poor data quality. In some cases on the Olympic Peninsula, escapement estimates of wild stocks are currently derived from surveys that occur in limited index reaches or are based on one or a few surveys during the peak of spawning. Escapement estimates can be biased when derived from surveys conducted at limited spatial and/or temporal scales (Ohlberger et al. 2017, Ohlberger et al. Submitted). Increased escapement monitoring for stocks that spawn in the park is dependent upon consistent and well-coordinated efforts among the State, tribes, and OLYM.

Future development or expansion of monitoring programs could include the latest technologies to assess wild salmonid escapements. Recently, side-scan sonar technology was used to estimate the escapement and run timing of winter steelhead and Chinook salmon in the Elwha River (e.g., Denton et al. 2014). Additionally, WDFW currently is using sonar to estimate the abundance of wild steelhead in the upper Hoh River from mid-January through May (M. Zimmerman, WDFW, pers. comm.). A better understanding of escapement, including the relative proportions of spawning of each stock that occurs inside and outside OLYM, would be useful in prioritizing management efforts for park managers.

Meaningful monitoring also requires an understanding throughout the continuum of freshwater, estuarine, and ocean habitats where salmonids complete their life histories. There is a need for additional monitoring of juvenile salmonids in coastal rivers to better understand in-river survival, productivity, and their abundances. Additionally, a better understanding of the interception of Olympic Peninsula wild populations in marine recreational fisheries is a key to effective in-river management. Genetic stock identification and analysis of coded-wire tag data of marine caught salmonids would provide information on the extent that OLYM stocks are caught in the ocean. One key example is that of Hoh spring/summer Chinook, where no marine interception estimates exist and consequently significant restrictions of in-river sport and commercial fisheries have occurred over the last decade.

Harvest management should acknowledge the role of salmon abundance to the aquatic ecosystem. Harvest influences salmon productivity by reducing numbers in the spawning population, reducing genetic diversity, and influencing species distributions. Finally, information is needed to fully understand the extent of bycatch and incidental mortality of Pacific salmonids in recreational and commercial fisheries in Olympic Peninsula rivers.

Hatcheries

The artificial propagation of Pacific salmonids in hatcheries has raised concerns regarding short and long-term genetic effects of hatchery-origin fish on fitness of natural populations for 40 years (Reisenbichler and McIntyre 1977). Since 1915, an estimated total of 729 million hatchery fish were released into five rivers draining from OLYM. Over the period of record, the estimated number of non-indigenous stock origins among species included the following for each river: Quinault (59), Quillayute (40), Queets (37), Elwha (21), and Hoh (16). There is a need to understand the extent of effects from hatchery releases on natural populations. The goals, release levels, stock origins, and effectiveness of hatchery programs require regular appraisal and future work could consider periodic genetic assessments to monitor any changes in the genetic composition of wild populations, particularly in rivers where extensive hatchery releases occurred and used non-indigenous broodstock sources.

Resource Summary

Salmon are a critical resource for the Olympic Peninsula and serve as a focal management issue for OLYM. Of the 17 stocks that we evaluated across 5 different river systems, we found cause for both concern and encouragement (Table 4.2.4). Of the 15 stocks with sufficient long-term data to assess status and trends, we found that six warranted significant concern and four warranted moderate concern, with the remainder in good condition that is relatively stable. This is a significant departure from earlier assessments on the status of Olympic Peninsula salmon stocks (Houston and Contor 1984, Huntington et al. 1994, SASSI 1994, McHenry et al. 1996). Observations include the following:

- The status of spring Chinook stocks across rivers is of concern, as this species consistently missed escapement goals, have declining terminal run sizes, and experienced commercial fishing closures.
- The status of fall Chinook stocks was one of the most stable fish runs, with good condition scores in the Hoh and Quillayute systems and a large-scale restoration project in the Elwha expected to improve population status.
- Quillayute summer-run coho salmon are an unusual stock, with unique migration and spawn timing. The spawning of wild summer-run coho salmon in the Sol Duc occurs primarily within Olympic National Park. Low escapement levels of wild summer-run fish, coupled with in-river harvest of hatchery coho cause concern for management of this unique stock.
- The Queets River system has the highest proportion of its watershed inside of OLYM, yet 3 of the 4 stocks were assessed as having significant concern. By comparison, the Quillayute had 3 of 5 stocks in good condition. Developing additional information about what is causing among river differences in status and trend is warranted.
- The magnitude of hatchery releases and the number of non-indigenous stock origins throughout rivers is cause for concern. An estimated 729 million hatchery salmonids were released into rivers reported herein. The Elwha and Quinault rivers had the longest record of hatchery releases dating back to 1915 for each. The highest total numbers of hatchery releases occurred in the Quinault, Elwha, and Quillayute rivers and the lowest number of releases occurred in the Hoh

river system. An estimated 59, 40, and 37 non-indigenous hatchery stock origins were used among salmonid species in the Quinault, Quillayute, and Queets rivers, respectively.

Table 4.2.4. Resource condition summary for Pacific salmonids in Olympic National Park, giving the condition and trend for each stock summarized in this chapter and a brief rationale statement. Further information for each criteria used in the assessment is provided in Appendix H.

River	Stock	Condition and Trend	Rationale
Hoh	Spring Chinook	🔴↔	Met escapement floor in 4 of last 10 years; commercial and recreational fishing closures; terminal run size slowly declining.
	Fall Chinook	🟢↔	Met escapement floor in 10 of last 10 years and 95% during period of record.
	Fall Coho	🟢↔	Made escapement goal in 9 of last 10 years and 81% during period of record.
	Winter Steelhead	🟡↓	Met escapement floor in 5 of last 10 years and 59% during period of record; High levels of hatchery releases (54 of 55 years) and significant fishing pressure.
Queets	Spring Chinook	🔴↔	Met escapement floor in 1 of last 10 years and 49% of period of record; No commercial harvest since early 2000's due to low returns.
	Fall Chinook	🟡↓	Met escapement floor in 9 of last 10 years and 85% during period of record; Terminal run size increasing recently from historical low; concern over hatchery broodstock program that uses wild fall Chinook.
	Fall Coho	🔴↔	Met escapement floor in 6 of last 10 years and 46% during period of record; High levels of hatchery releases (13.8 million fish from multiple stock origins) and intensive commercial harvest (42% average total harvest rate).
	Winter Steelhead	🔴↔	Met escapement goal in 6 of last 10 years and 65% during period of record; High levels of hatchery releases (37 of 37 years; 5.6 million fish released) and concern over lack of marking program for hatchery steelhead.
Quillayute	Spring Chinook	🔴↓	Made escapement goal in 0 of last 10 years and 31% during period of record; Terminal run size low with high levels of hatchery releases (36 of 37 years; 13.6 million fish released).
	Fall Chinook	🟢↔	Made escapement goal in 10 of last 10 years and 97% during period of record.

Table 4.2.4 (continued). Resource condition summary for Pacific salmonids in Olympic National Park, giving the condition and trend for each stock summarized in this chapter and a brief rationale statement. Further information for each criteria used in the assessment is provided in Appendix H.

River	Stock	Condition and Trend	Rationale
Quillayute (continued)	Fall Coho	🟢↔	Made escapement goal in 8 of last 10 years and 69% during period of record; High mean terminal run size with relatively low levels of hatchery releases (4 of 36 years in record).
	Summer Coho	🔴↓	No established escapement goal for unique wild stock that has 95% of spawning inside of OLYM (Sol Duc). Low wild escapement coupled with high hatchery releases (38 of 42 years; 15 million fish released) and concern with run timing overlap between wild and hatchery fish causing harvest impacts on wild stock.
	Winter Steelhead	🟢↔	Made escapement goal in 9 of last 10 years and 89% during period of record.High mean terminal run size.
Quinault	Fall Coho	🟡↔	Recent gains in terminal run trend over historical lows, but historical catch data suggest high production capacity of system; extensive hatchery releases from multiple stock origins. Concern over lack of escapement goal.
	Winter Steelhead	🟡↓	No established escapement goal with relatively high average harvest rate (48%) and extensive hatchery releases.
Elwha	Summer/Fall Chinook	◌	Limited historical data and emerging response to large-scale restoration.
	Coho	◌	Limited historical data and emerging response to large-scale restoration.
	Steelehead	◌	Limited historical data and emerging response to large-scale restoration

4.2.2. Razor Clams

Steven Fradkin

Introduction

Razor clams (*Siliqua patula*) are an abundant intertidal organism native to high energy sandy beaches of the exposed Washington coastline. Clam populations consist of identical males and females that broadcast spawn their gametes into the water column where fertilization takes place. The resulting planktonic larvae can drift 5-16 weeks and disperse tens of kilometers before settling out to beach sediments (Lassuy and Simons 1989). Razor clams become sexually mature in approximately 2 years at 76 mm in length, when they are considered adults for harvest purposes. In Washington, the razor clam lifespan is up to 9 years. Ecologically, razor clams are key players in nearshore nutrient cycling: they feed on abundant surf diatoms and subsequently export waste nitrogen that fuels nearshore productivity (Lewin et al. 1979). Razor clams are important food items for Dungeness crab (*Cancer magister*; Stevens et al. 1982) and marine fish (Hogue and Carey 1982, Kvitek and Oliver 1988), and are also consumed by sea otters (*Enhydra lutris*; Laidre and Jameson 2006).

Razor clam on Kalaloch Beach. Photograph courtesy of NPS/Steven Fradkin.

Culturally, razor clams are a highly valued fishery, important to Native American ceremonial and subsistence harvest as well as to local and regional non-tribal recreational harvest. The NPS has trust responsibilities to the Quinault Nation, Hoh, Quileute and Makah Tribes on the outer Washington coast that includes the responsible management of razor clam populations. Tribal harvest at Kalaloch beach within OLYM is enshrined in the 1855 Treaty of Olympia between the federal government and Quinault Nation and Hoh Tribe. These tribal treaty rights were further clarified by the 1994 Rafeedie decision in federal court that guaranteed the tribes 50% of the annual razor clam quota (total allowable catch) as determined by annual stock assessments. In addition to tribal considerations, generations of non-tribal Washingtonians have also relied on park beaches for clam harvest, and this constituency actively tracks park management of the resource.

In OLYM, there are 10 beaches where razor clams have been historically present (Figure 4.2.14; Olsen et al. 1974). These ten beaches were incorporated into the boundary of OLYM in 1986.

Regular stock assessment and park-sanctioned harvest activities are conducted only on Kalaloch beach, the longest and most productive beach. The other nine beaches are relatively small and inaccessible, and contain marginal populations likely derived from intermittent larval recruitment across years. Anecdotal accounts from local harvesters active on these beaches prior to their incorporation into OLYM suggested that these beaches have low clam densities and uneven size distributions.

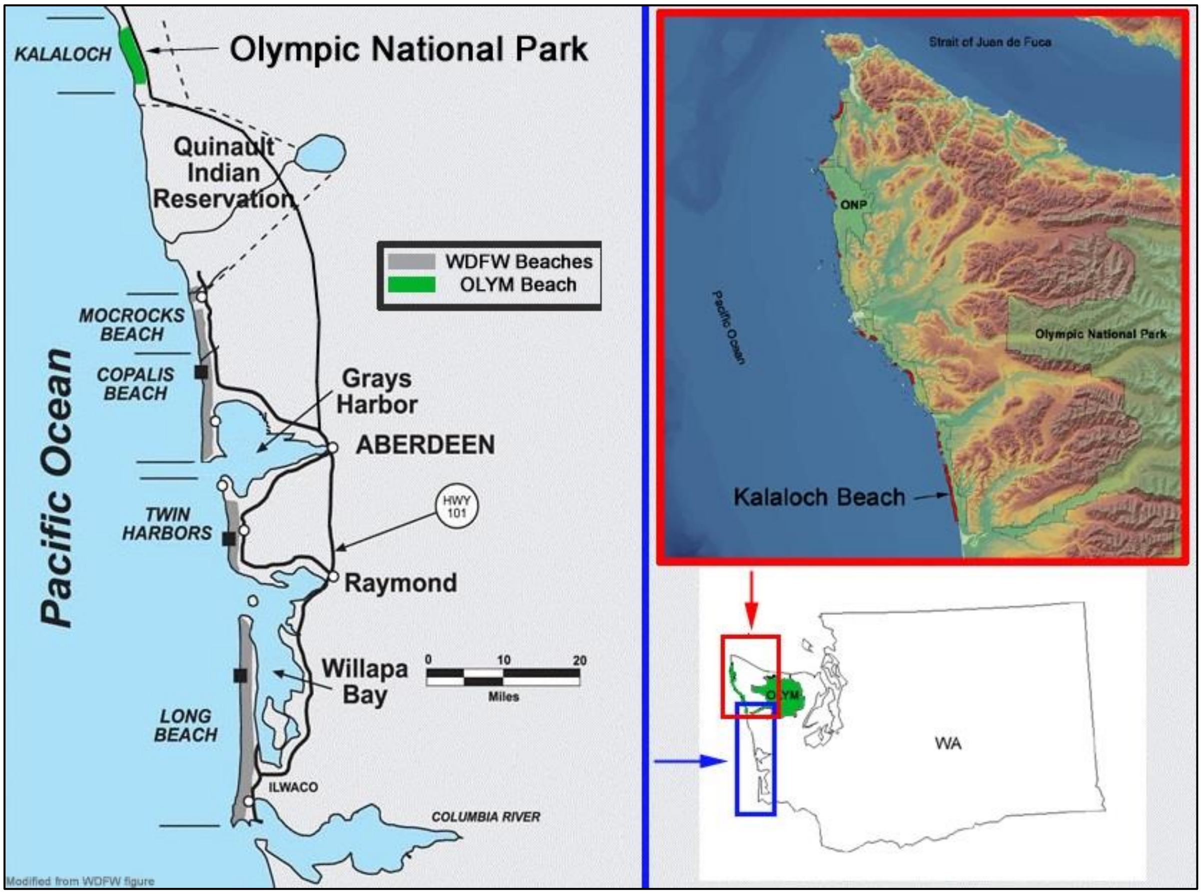

Figure 4.2.14. Location of razor clam beaches on the outer Washington coast. Ten razor clam beaches are found within Olympic National Park (indicated by red lines in the red inset box). NPS recreational harvest occurs only on Kalaloch Beach. Washington coast beaches that support recreational razor clam harvest (blue box inset) include state beaches managed by the WDFW (grey) and Kalaloch Beach managed by Olympic National Park (green). Source: WDFW and NPS.

Kalaloch Beach, on the southern end of the park shoreline, is approximately 6.8 km long (Figure 4.2.14). Kalaloch likely contributes a substantial number of propagules to the regional larval pool. However, it is unclear whether juvenile recruitment at Kalaloch comes from self-seeding, dispersal from extensive southern Washington clam beach larval sources (Figure 4.2.15), or a combination of both. Kalaloch is the only beach within OLYM boundaries where non-tribal recreational harvest is permitted. Formal stock and harvest assessments were begun by WDFW in 1995. OLYM began active management of the Kalaloch razor clam population in coordination with

WDFW in 2001. State razor clam beaches on the southern Washington shoreline are managed by WDFW.

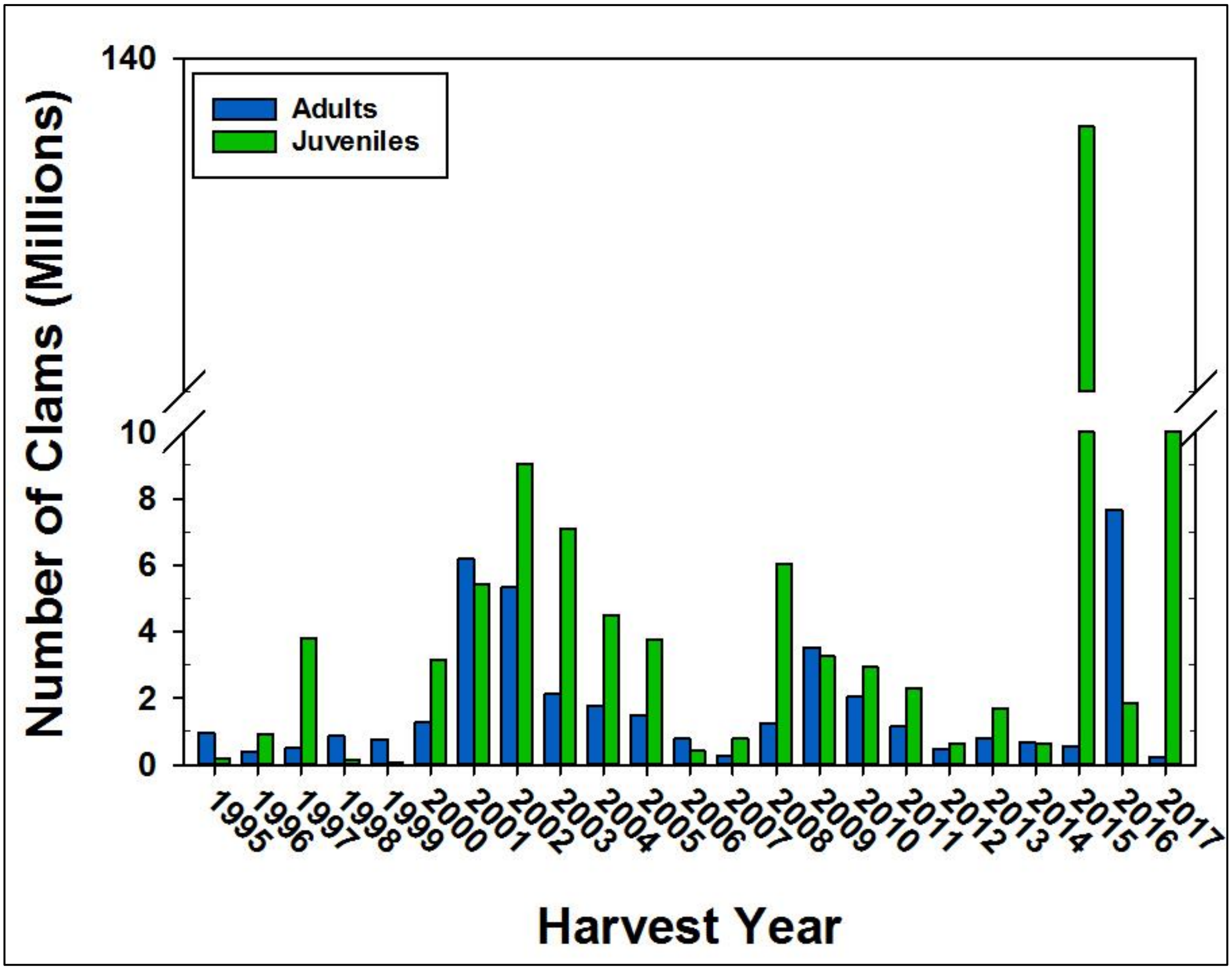

Figure 4.2.15. Abundance of adult and juvenile clams on Kalaloch Beach based on annual stock assessment data. Source: NPS, tribal, WDFW.

Since the inception of monitoring, the Kalaloch population has experienced marked fluctuations in adult abundance and recruitment (Figure 4.2.15), and the routine presence of biotoxins from harmful algal blooms (Figure 4.2.16). These phenomena have led to the frequent closure of the beach to non-tribal harvest (Figure 4.2.17) and a concern for the long-term condition of the Kalaloch razor clam population. Since 2006, the Kalaloch razor clam population has experienced chronic declines and low population levels, with non-harvest related annual adult mortality frequently exceeding 50% across multiple years (e.g. 2005-2007, 2009-2012), based on harvest estimates and changes in population size. During this period, the average adult size has been unusually small (<10 cm, Figure 4.2.18), suggesting that mortality has disproportionally affected larger size classes. As such, the park has restricted harvest over the last decade, with partial or full closures implemented in eight of the last 10 years (Figure 4.2.17).

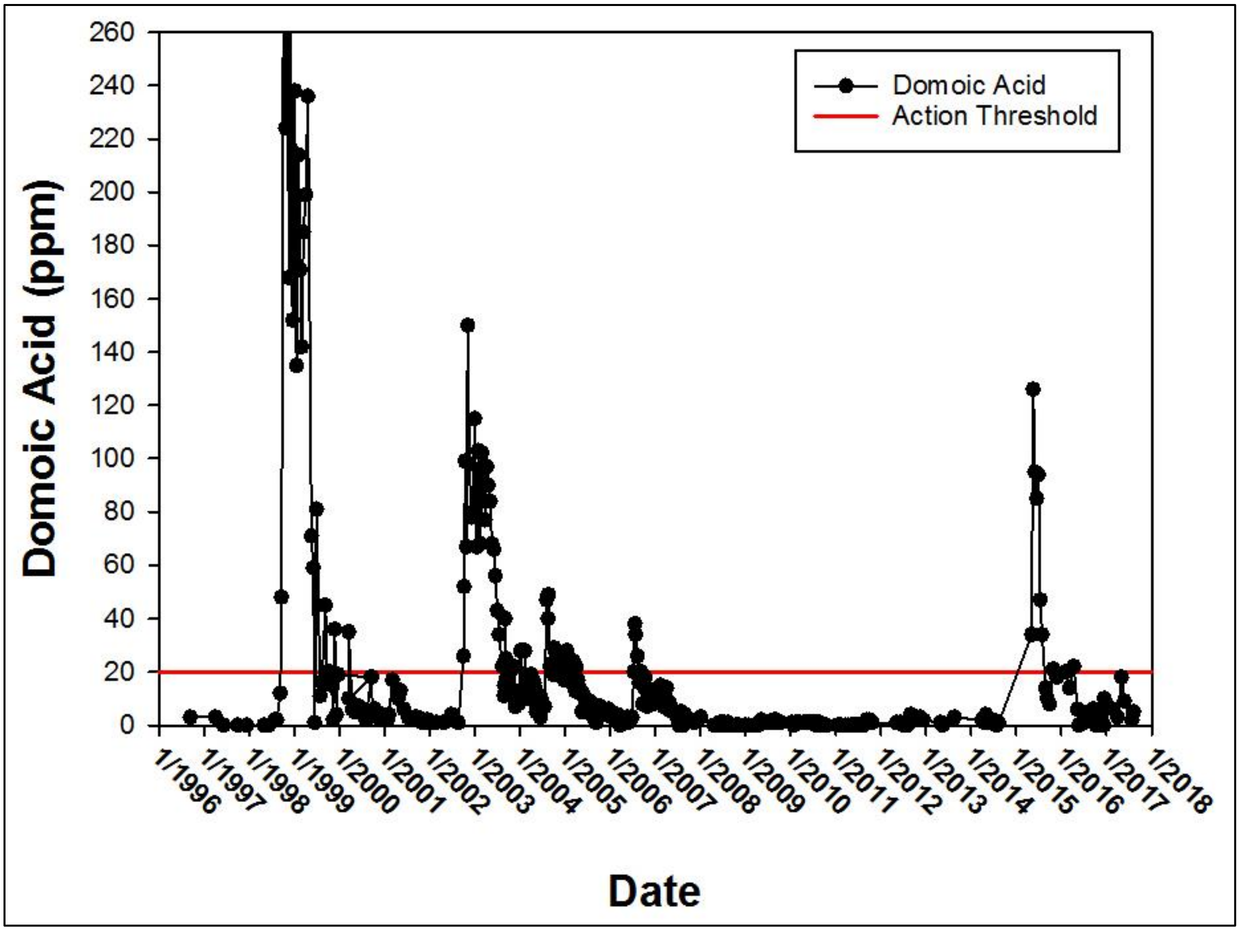

Figure 4.2.16. Domoic acid concentrations (from harmful algal blooms) in razor clam tissue from Kalaloch Beach. Red line denotes human health action threshold above which razor clam harvest is prohibited. Source: http://wdfw.wa.gov/fishing/shellfish/razorclams/domoic_levels.html, accessed 30 October 2017.

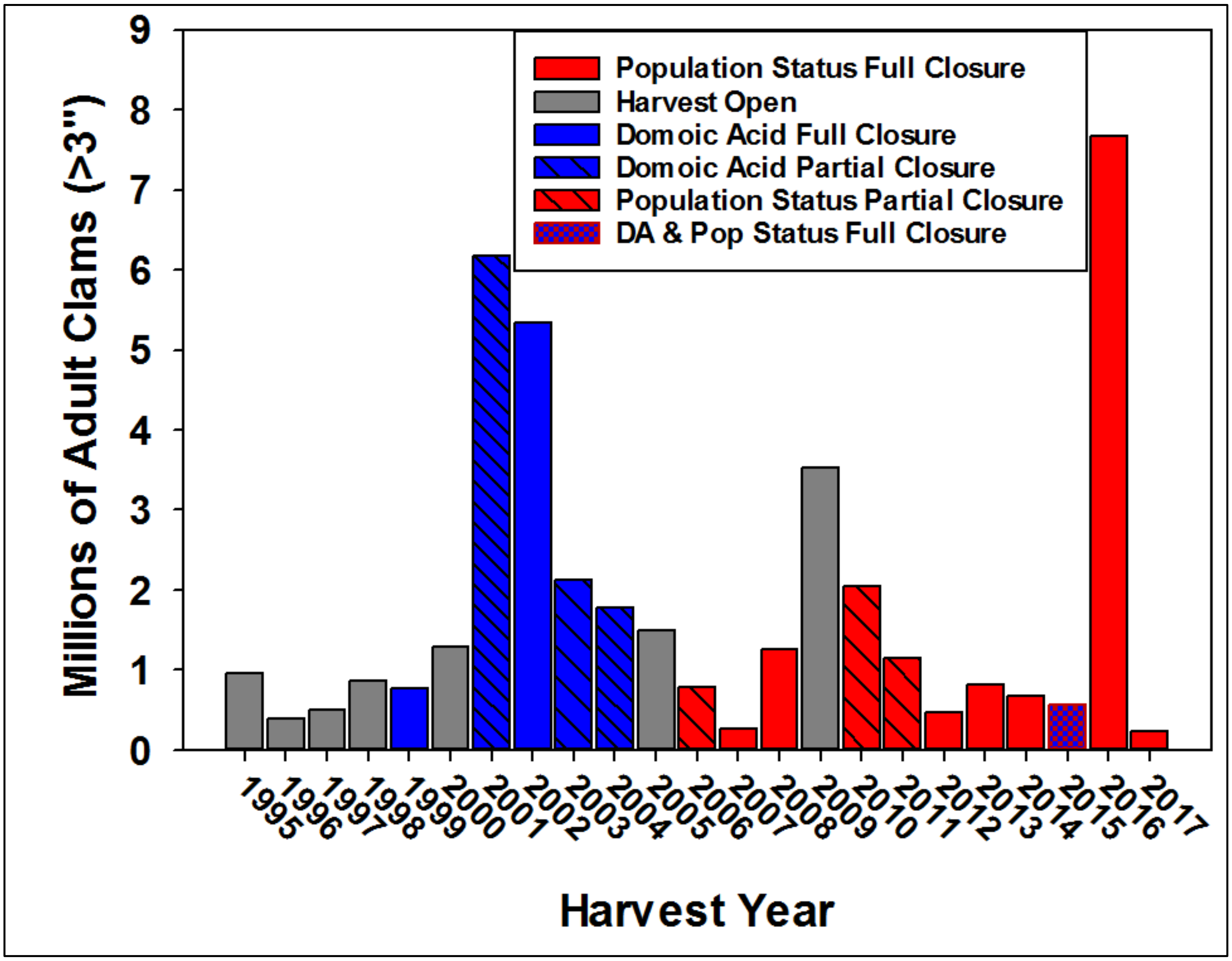

Figure 4.2.17. Annual razor clam harvest history for Kalaloch Beach overlaid on abundance of adult clams based on annual summer stock assessment data. Gray bars denote years with normal full harvest opportunities. Blue bars denote closure years due to high levels of domoic acid. Red bars denote closure years due to low adult abundance or small adult sizes. Open bars denote full closure to harvest, while hashed bars denote season where harvest closed prematurely due to domoic acid or population concerns. All closures denoted are for recreational harvest only. Tribal harvest occurred in all years except 1998. Source: NPS.

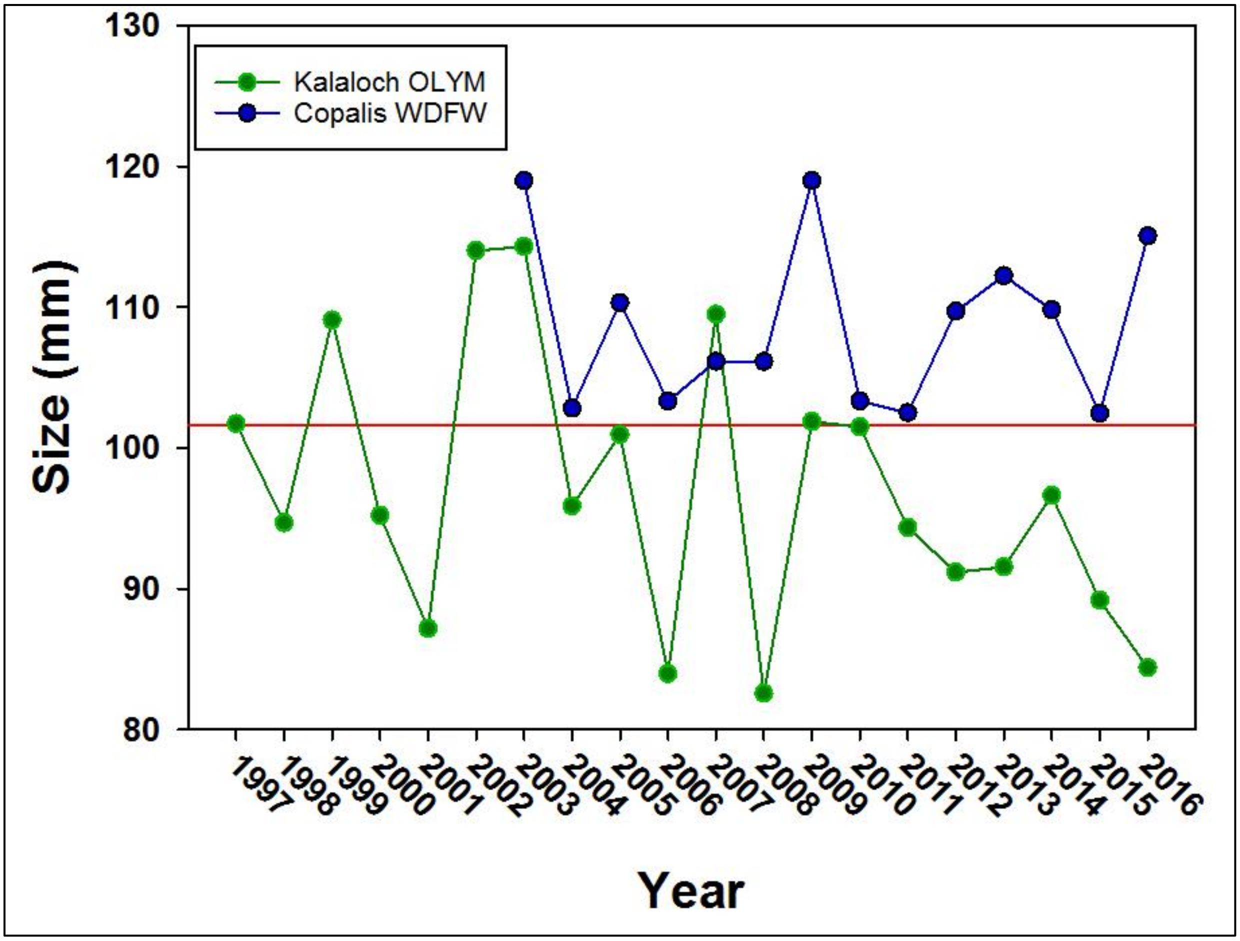

Figure 4.2.18. Comparison of annual average adult clam sizes at Kalaloch Beach (OLYM) and Copalis beach (WDFW). Red line denotes the 102-mm size threshold for what is considered a satisfactory clam for harvest purposes. Source: NPS and WDFW.

Approach and Methods

We used (1) razor clam population data derived from annual Kalaloch stock assessments and (2) data on potential stressors to assess the condition of the Kalaloch razor clam population. In particular, we look at the following measures of Kalaloch razor clam condition: (1) population abundance, (2) size-class distribution, and (3) average adult size. We then examine and discuss the potential impact of the following stressors to the population: harvest, pathogens, harmful algal blooms, ocean acidification, increased physical disturbance, and increased marine mammal predation.

Razor Clam Stock Assessment

Annual stock assessments are conducted at Kalaloch in late June or early July during negative low tides (<0.55 m) that expose the bulk of the intertidal clam beds. The 6.8-km beach is divided into four approximately 1.5-km-long sections with two random transects sampled in each section for a total of 8 transects. Of these transects, four are sampled by OLYM and four are sampled by the Quinault and Hoh treaty tribes with assistance from WDFW. Each shore-normal transect runs from the top of the clam bed towards the sea, with sampling levels every 15.25 m. At each sampling level, six plots are sampled using the pumped area method (Ayres and Simons 1999), where seawater is pumped into a 0.5-m^2 ring to a depth of 0.91 m, liquefying the sand. Razor clams are positively buoyant, thus all size classes (from 0.6 to 15.25 cm) float to the surface where they are counted, measured, and returned to the beach. Stock assessment data consist of abundance and size frequency information. For management purposes, stock assessments are used to estimate the total number of juvenile (<76 mm) and adult (>76 mm) clams. The harvestable quota, or total allowable catch, is determined as 25.4% of the total number of adult clams following WDFW protocols. Half of the total allowable catch is reserved as the tribal harvest quota, while the other half is reserved as the non-tribal harvest quota. Annual stock assessments began at Kalaloch in 1995, conducted by WDFW and the treaty tribes without active OLYM involvement. Olympic National Park began active involvement in management activities in 2001, and assumed full responsibility for the non-tribal stock assessment transects and harvest assessments in 2012.

Razor Clam Harvest Assessment

In years when harvest occurs, harvest assessments are conducted to determine the number of clams and proportion of the total allowable catch harvested. On each harvest date, the total number of harvesters is estimated by counting all people on the beach one hour before the low tide. Average digger success, or catch per unit effort, is estimated by interviewing diggers as they exit the beach to determine the number of clams harvested per person. The total clam harvest per date is estimated by multiplying the number of diggers by the average digger success, which is then subtracted from the total allowable catch. Tribes monitor their own harvest and report their data to WDFW and OLYM.

Pathogens

Washington razor clams have historically been afflicted by the bacterial gill pathogen NIX. In the 1980s, approximately 95% of the Washington coast razor clams were killed by bacterium NIX (Elston 1986). During this outbreak, northern razor clam populations in British Columbia, Canada, and Alaska were NIX-free, and archived Washington samples from before the outbreak were also NIX-free, suggesting that NIX was not in Washington razor clams prior to the 1980s. The origins and life cycle of NIX outside of razor clams are unknown. During the 1980s, the prevalence of NIX at Kalaloch was not assessed. NIX has been assessed at Kalaloch sporadically (Figure 4.2.19), starting in 2006. From 2008-2010, an NPS-funded project monitored NIX prevalence bimonthly. These assessments used classical histological methods of Elston and Peacock (1984), which were expensive and labor intensive. More recently, OLYM has assessed NIX prevalence at Kalaloch using a newly developed molecular probe (Jeremy Weisz/Linfield College and OLYM, unpublished data).

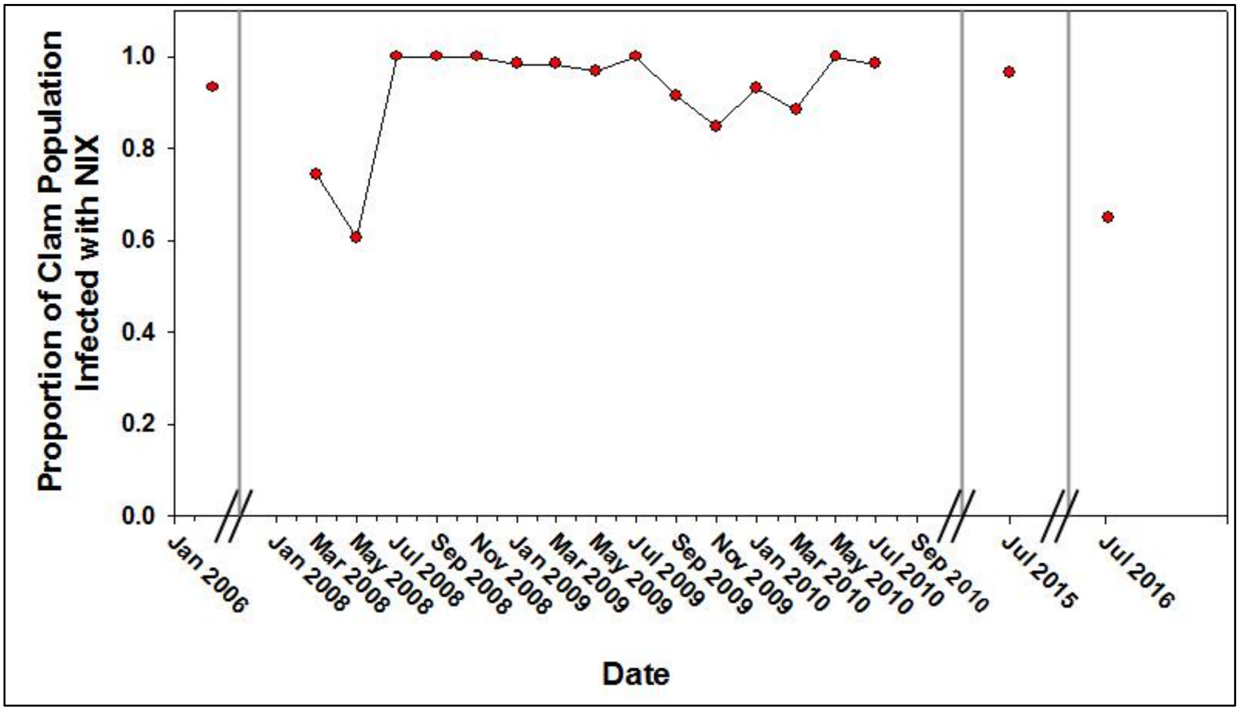

Figure 4.2.19. The prevalence of the bacterial gill pathogen NIX in Kalaloch razor clam population from 2006-2016. Source: NPS.

Harmful Algal Blooms

Domoic acid, produced by the bloom-forming surf diatom *Pseudo-nitzschia* spp., is the dominant biotoxin that razor clams accumulate through consumption of the algae (Horner et al. 1993). A potent neurotoxin, domoic acid is lethal to mammals in high doses, while causing short-term memory loss at sublethal doses. Domoic acid accumulates in the fatty tissues of razor clams and can take weeks to months to depurate from clam tissues. While likely an indicator of broader oceanographic changes that may impact razor clams (Moore et al. 2008), harmful algal blooms, such as toxic *Pseudo-nitzschia* blooms, do not appear to directly impact razor clam populations. The Washington Department of Health monitors the concentration of domoic acid in Kalaloch razor clam tissue monthly. The human health threshold concentration for domoic acid in razor clam tissue above which harvest is prohibited is 20 ppm.

Ocean Acidification

Ocean acidification (OA) is the decrease in seawater pH and carbonate ion concentration caused by the diffusion of anthropogenic atmospheric CO_2 into the world's oceans associated with fossil fuel combustion. The waters of the Pacific Northwest Coast are experiencing chronically corrosive waters (Feely et al. 2012). While adult razor clams buried in sand beaches may be less sensitive to OA than their directly exposed oyster and mussel bivalve relatives (Widdicombe et al. 2011), water-column borne larval razor clams may be particularly susceptible to acidification (Waldbusser et al. 2015). OLYM has been conducting continuous monitoring of nearshore pH levels at Brown's Point at the northern end of Kalaloch Beach since 2010 (OLYM, unpublished data). This monitoring consists of sensors that measure pH and other water quality parameters from a tidepool that is inundated by nearshore water at higher daily tides. Only data used from periods of tidepool immersion are used to characterize intertidal OA that is relevant to razor clams. In order to characterize historical OA conditions at Kalaloch on the southern Olympic coast, we used data from 2000-2010 from Tatoosh Island on the northern Olympic coast that had been collected using a similar methodology to our data from Browns Point (Wootton and Pfister 2012). The use of these data appears reasonable, as overlapping Tatoosh and Browns Point data from 2011-2014 differ by an average of 0.003 ± 0.187 SD pH units (Wootton and Pfister, unpublished data). Only OLYM summer data (April-September) were used in the present analysis to facilitate comparability with the Tatoosh data that was collected only during the summer.

Sea Otter Predation

Sea otters were extirpated from the Washington coast by the fur trade in the early 1900s. In 1969, sea otters were reintroduced to the Washington coast and their numbers on the south coast have increased markedly over the last two decades (Figure 4.2.20). Sea otters are known to forage on razor clams (Riedman and Estes 1990), and annual aerial surveys of otter abundance along the outer Washington coast have been conducted by WDFW and the USFWS since 1999.

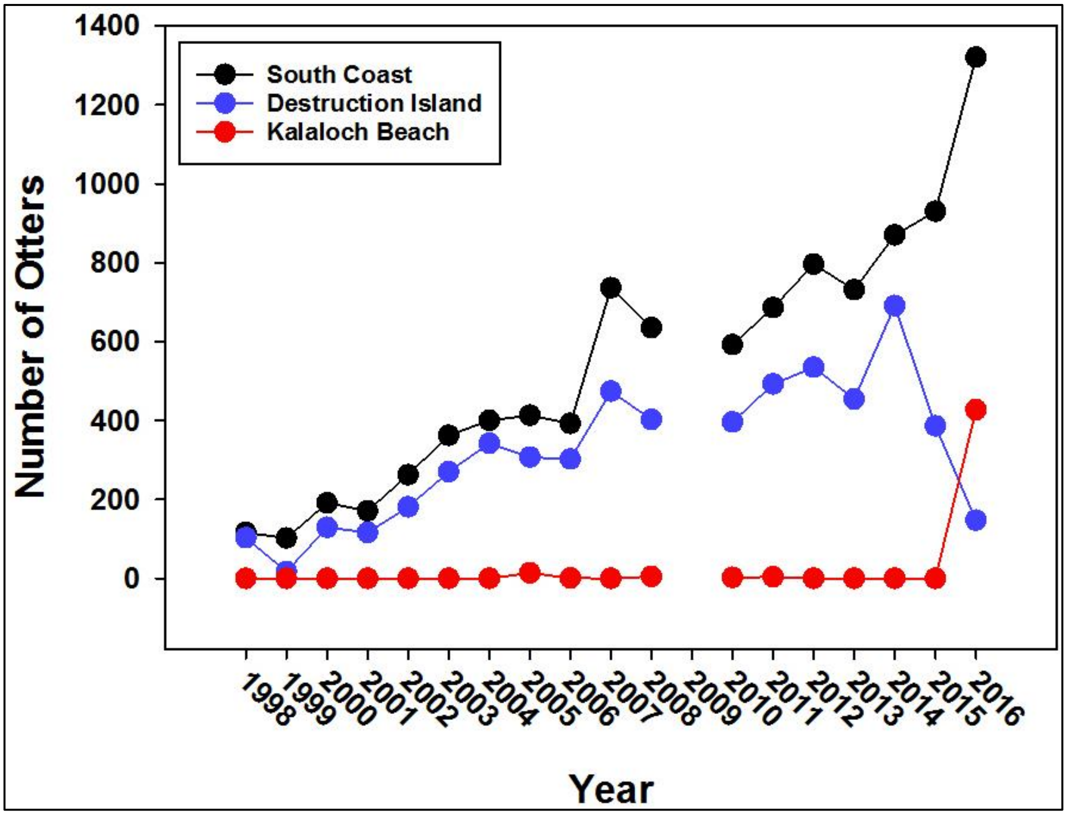

Figure 4.2.20. Population growth of the southern Washington sea otter population from 1998-2016 for (1) the total south-coast population from the Quinault reservation to north of Hoh head (black); (2) the proportion of the population observed around Destruction Island (blue); and (3) the proportion of the population observed along the coast near Kalaloch Beach (red). Data are derived from annual summer aerial surveys conducted by WDFW and USFWS (Jameson and Jeffries 1999-2014, Jeffries and Jameson 2015, Jeffries et al. 2016). Surveys were not conducted in 2009.

The effect of sea otter predation on the Kalaloch razor clam population was liberally estimated by calculating the annual proportion of the adult razor clam population consumed using Washington coast literature values of sea otter abundance (Jameson and Jeffries 1999, 2001, 2002, 2003, 2004, 2005, 2006, 2007, 2008, 2009, 2010, 2011, 2013, 2014, Jeffries and Jameson 2015, Jeffries et al. 2016), predation rates (Laidre and Jameson 2006), and razor clam size and abundances from annual stock assessment data (Appendix I).

Physical Disturbance

Razor clams have evolved in the wave-swept open coast environment. While they have limited ability to move laterally across the face of a beach, clams use their powerful feet to dig vertically in the sand at rates up to a third of a meter per minute to elude wave erosion and slow-digging harvesters (Lassuy and Simons 1989). As climate change proceeds, the frequency and magnitude of storm events with high wave action is expected to increase. Increased wave action may lead to increased excavation rates of razor clams. Once fully excavated from protective sands, the slightly buoyant clams are moved about by turbulent waters and have difficulty reestablishing into the beach, making them highly susceptible to predation. Excavated clams are usually assumed to be lost to the population. On the Oregon coast, an increase in the frequency and magnitude of large waves has been observed (Ruggiero et al. 2010). We examined wave data from the NOAA National Data Buoy Center (NDBC) Buoy #46041 off the southern Olympic coast offshore from Cape Elizabeth on the Quinault Reservation to assess whether physical disturbance of razor clams at Kalaloch beach may have increased. Specifically, we examined average daily, maximum daily, and annual maximum significant wave heights over the period of the razor clam population record.

Reference Conditions

Establishing reference conditions for the Kalaloch razor clam population presents a challenge. A healthy population should consist of numerous adult clams capable of sustaining harvest, and should contain a broad age/size class distribution with multiple cohorts, from newly settled juveniles to larger, older age classes (i.e., >127 mm; WDFW, unpublished data). Additionally, healthy razor clam populations typically do not experience greater than 50% non-harvest related mortality of adult clams across multiple years (Nickerson 1975). Historical Kalaloch stock assessment data suggest that abundances (Figure 4.2.15) and size structure (Figures 4.2.21 and 4.2.22) vary considerably across years. Frequently, adult abundances are low and the population consists of a single dominant cohort.

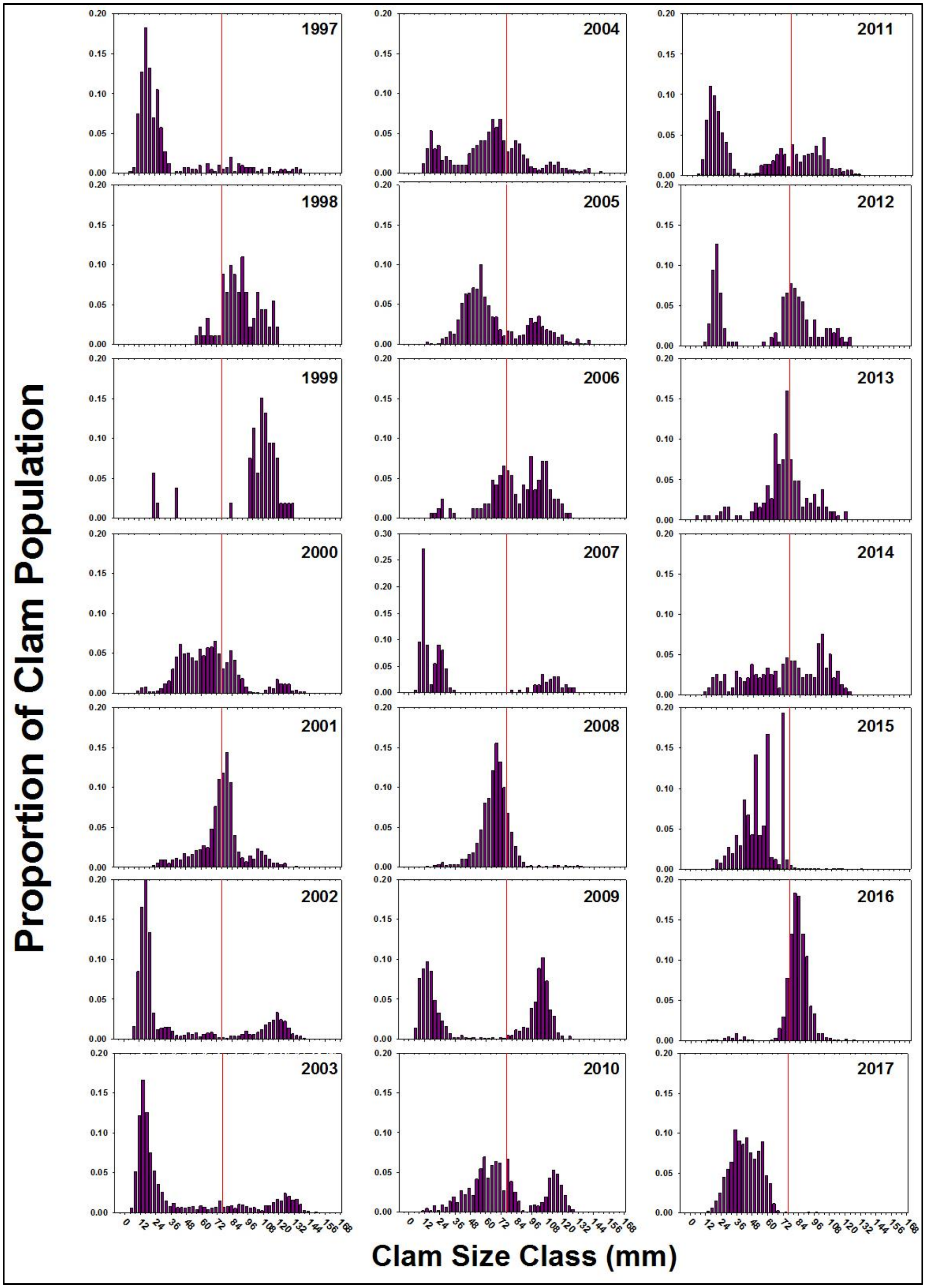

Figure 4.2.21. The size frequency distribution of the Kalaloch razor clam population from 1997-2017. The red vertical line in each sub-figure denotes the 76 mm size threshold between adult and juvenile clams. Source: NPS.

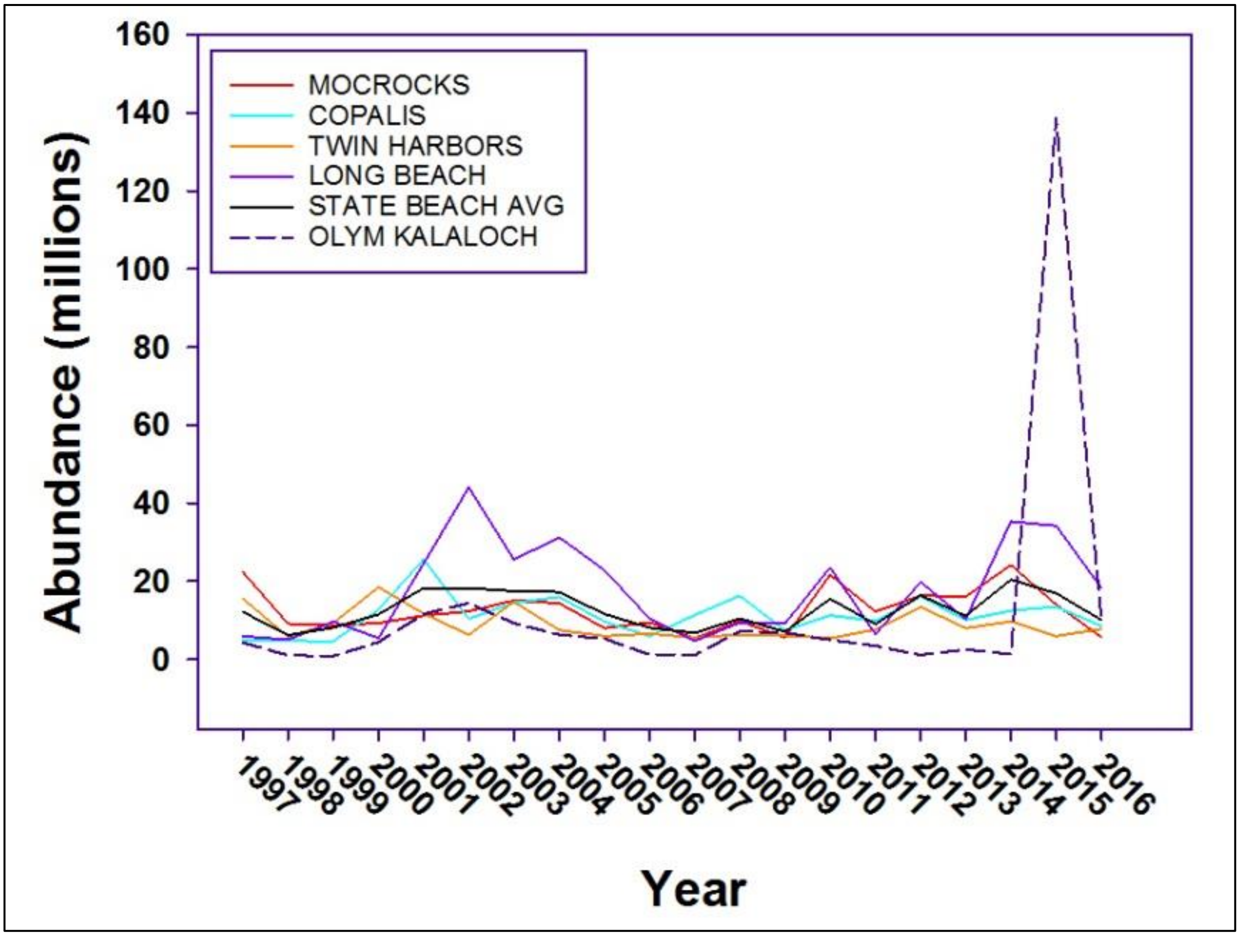

Figure 4.2.22. Trends in annual total clam abundance on outer Washington coast razor clam beaches from 1997-2017. The dashed line denotes Kalaloch Beach within Olympic National Park, and the solid black line denotes the average of the four WDFW managed beaches south of Kalaloch. Source: WDFW and NPS.

In comparison, the state beaches on the southern Washington coast (Figure 4.2.14) usually contain healthy populations. These beaches typically have higher clam abundances and broad size class distributions, representing regular recruitment and survival of larger adult clams. For the purposes of this condition assessment, we used Copalis Beach on the southern Washington coast as a reference condition beach. The population dynamics of Copalis Beach are typical of the southern state beaches (Figure 4.2.22). The size class distribution of Copalis is broad, with a relatively even distribution of multiple size classes, and with survival of larger size classes represented by a large average adult size (Figure 4.2.18).

Results and Assessment

In this assessment we evaluated razor clam population dynamics and size structure, along with a host of potential stressors including: harvest, pathogens, harmful algal blooms, ocean acidification, sea otter predation, and physical disturbance. Of these stressors, only the pathogen NIX appears to be a historical and present factor affecting the Kalaloch razor clam population. The other stressors considered warrant future consideration, as their effects along with the incidence of diseases are all expected to increase due to climate changes and further expansion of the reintroduced sea otter population.

Razor Clam Population Trends

Kalaloch razor clam survival is highly variable over time, but in most years new recruit abundance is near or greater than the adult population (Figure 4.2.15). While it is unclear what source beach recruits originate from, these data suggest that the population is not recruitment limited. Indeed, as recently as 2015 and 2017, Kalaloch experienced pulses of 138 and 97 million recruits respectively. These are the largest two recruitment classes ever observed on the Washington coast (Dan Ayres, WDFW, personal communication).

Adult survival appears to be an issue for the Kalaloch population. Over the period of record, the adult population abundance was usually less than 1.5 million clams (Figure 4.2.17), and there have been two periods in the last 11 years (2005-2007 and 2009-2012) where annual non-harvest mortality exceeded 50% of adults. More recently, from 2016 to 2017 there was 97% non-harvest mortality of adults.

The size/age structure of Kalaloch clams is often dominated by single size class cohorts (Figure 4.2.21). These cohorts are typically new-year recruits, or young adult cohorts. The largest size class reached at Kalaloch is about 120 mm. In contrast, the reference condition beach at Copalis is characterized by multiple cohorts across a broad range of size classes. The largest size class reached at Copalis is about 140 mm. While data are lacking, food availability may differ between Kalaloch and Copalis, leading to differential growth rates. However, as organisms with indeterminate growth, razor clam maximal size is determined by lifespan. Since 2010, the average size of an adult Kalaloch razor clam has declined markedly, from 102 mm to 83.8 mm (Figure 4.2.18). These declines were accompanied by frequent observations of large aggregations of empty clam shells at the high tide line. Over the same period, Copalis Beach has not shown a similar decline. For most of the period of record, Kalaloch has had average adult sizes at or below 102 mm, while the average Copalis adult has always been greater than 102 mm.

All of these data suggest that Kalaloch Beach is experiencing an agent(s) of mortality that disproportionately impacts adult clams, precluding survival to larger age classes that Washington razor clams typically achieve.

Razor Clam Harvest

Harvest has not been a constant feature of the annual razor clam population cycle. In 14 of the last 22 years, harvest has been completely or partially closed, due to domoic acid intoxication or to population status concerns (Figure 4.2.17). In years when harvest has been permitted, neither the recreational or tribal shares of the total allowable catch have come close to being utilized (Figure 4.2.23).

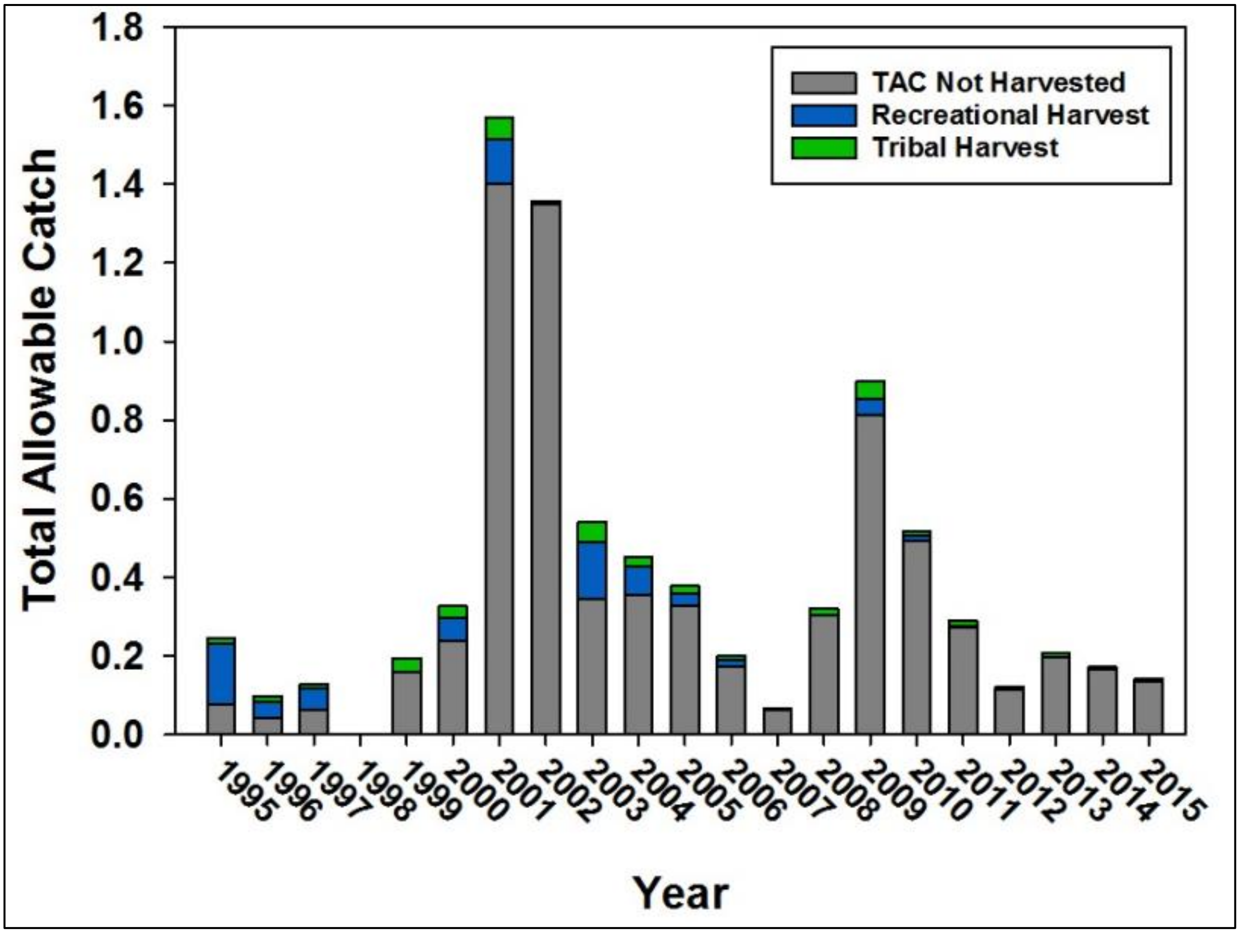

Figure 4.2.23. The total allowable catch (TAC) of Kalaloch razor clams by year with the amount taken annually by recreational and tribal harvesters. The TAC is calculated as 25.4% of the total adult population, with half allocated each to recreational and tribal harvesters. Source: NPS.

Pathogens

While the historical record of NIX prevalence at Kalaloch is incomplete (Figure 4.2.19), the sporadic data from 2006 to 2016 suggest that NIX is ubiquitous in the population. Throughout this period, at least 60% of the population was infected, with most sampling occasions showing infection rates greater than 80%. This period of high NIX infection coincided with two periods of marked population decline, (e.g. 2005-2007, 2009-2012, Figure 4.2.17), where harvest was inconsequential (Figure 4.2.23). It is unclear to what extent the southern State beaches are afflicted by NIX, as there have been no assessments conducted on State beaches since 2003. Prior to that time, sporadic assessments from the early 1990s and early 2000s show moderate to high levels of NIX prevalence (WDFW, unpublished data).

Harmful Algal Blooms

Toxic *Pseudo-nitzschia* blooms and high domoic acid concentrations in razor clam tissues were observed for the first time at Kalaloch in 1990, and occurred frequently through 2007 (Figure 4.2.16). These blooms led to harvest closure from 1999 to 2004 due to tissue concentrations above 20 ppm

(Figure 4.2.17). The presence of domoic acid does not appear to be related to the razor clam declines of 2005-2007 and 2009-2012.

Ocean Acidification

Intertidal seawater pH declined markedly on the Olympic coast from 2000-2010 (Figure 4.2.24). The rate of decline was over an order of magnitude greater than predicted by models (Wootton and Pfister 2012). From 2010 to 2016, the pH trend appears to remain relatively constant. The Kalaloch razor clam population appears to track the pH trend, with large adult population declines following the high years of 2001 and 2002 (Figure 4.2.17). However, subsequent years of high recruitment (2008 and 2015) with survivorship to high adult populations (2009 and 2016) suggest that lower pH levels are not the primary cause of population declines or recruitment failure (Figure 4.2.15). Larval bivalves have been shown to be more vulnerable to low pH and carbonate saturation states than adult bivalves, where larval bivalves under corrosive conditions have difficulty forming early shells and have decreased survival rates (Waldbusser et al. 2015). If current corrosive conditions were limiting the razor clam population, one would not expect to see substantial recruitment.

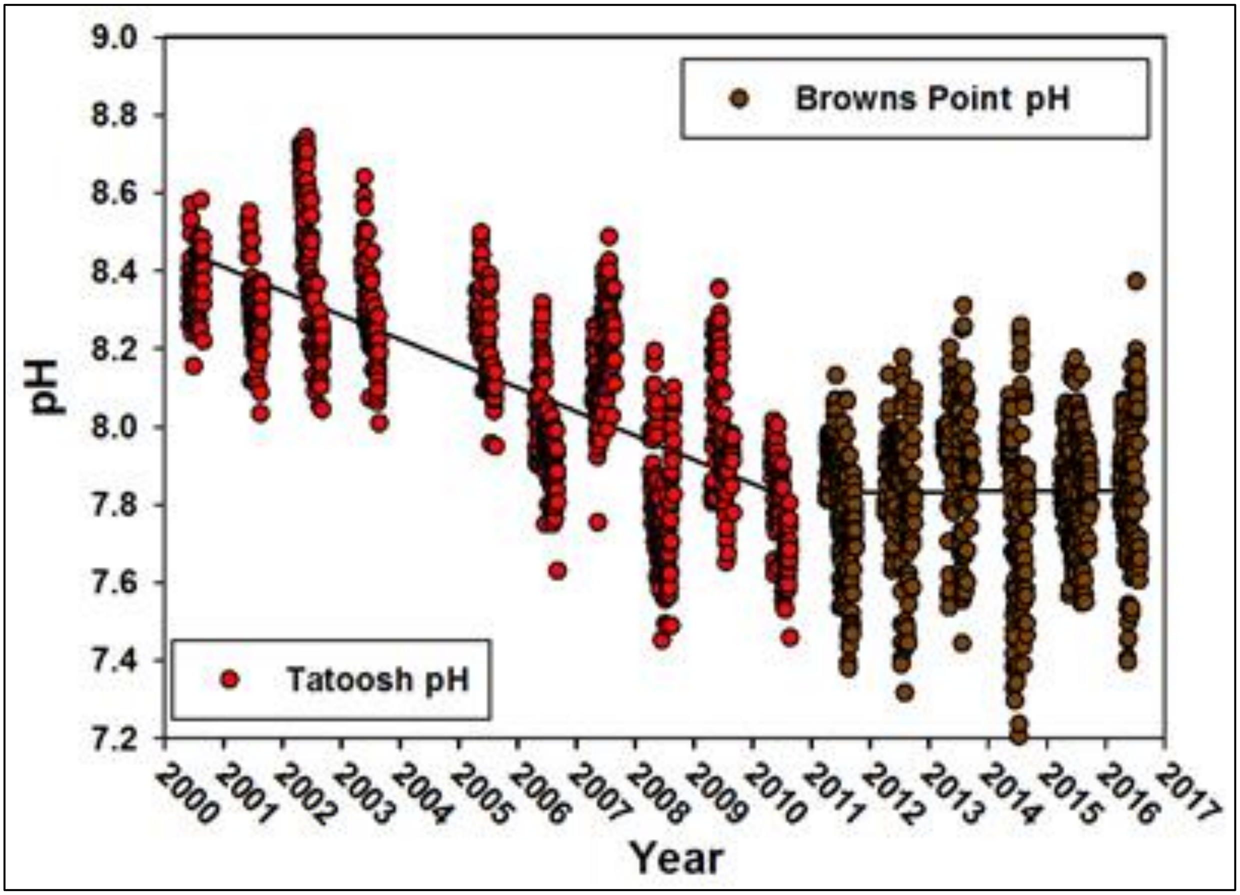

Figure 4.2.24. Trend in summertime (April through September) intertidal pH on the outer Olympic coast. Brown symbols denote data from Browns Point at the northern end of Kalaloch Beach (OLYM, unpublished data), while red symbols denote data from Tatoosh Island on the Northern Olympic coast. Source: Wootton and Pfister 2012.

Sea Otter Predation

Estimates of sea otter predation suggest that historical otter predation is not a significant factor in the razor clam declines we observed from 2004 to 2012 (Figure 4.2.25). From 1998 to 2016, otters are estimated to have consumed less than 8 percent of the annual clam population, with the percent consumed less than 3 percent in the majority of years. These liberal estimates likely overstate the impact of otters on Kalaloch razor clams. The number of sea otters on the south coast that may potentially feed on Kalaloch razor clams has risen from 116 to 1,302 animals from 1998 to 2016 (Figure 4.2.20). Throughout this period, the bulk of these south-coast otters have been observed around Destruction Island, approximately 4.8 km offshore of Kalaloch, with very few otters observed around the Kalaloch area. These aerial survey observations match anecdotal observations made by OLYM interpretive rangers and natural resource scientists. However, in recent years there appears to have been a shift in sea otter movement patterns. From 2015 to 2016, the number of otters observed at Destruction Island began declining while the south-coast otter population continued to climb. In 2016, substantial numbers of otters were observed on the Kalaloch coast for the first time since aerial otter surveys began being conducted in 1998. These observations are consistent with interpretive ranger anecdotal observations. These observations suggest that Destruction Island may have reached its carrying capacity causing large groups of otters to disperse in search of food sources. If this trend continues, otter predation on Kalaloch clams could increase and become a factor.

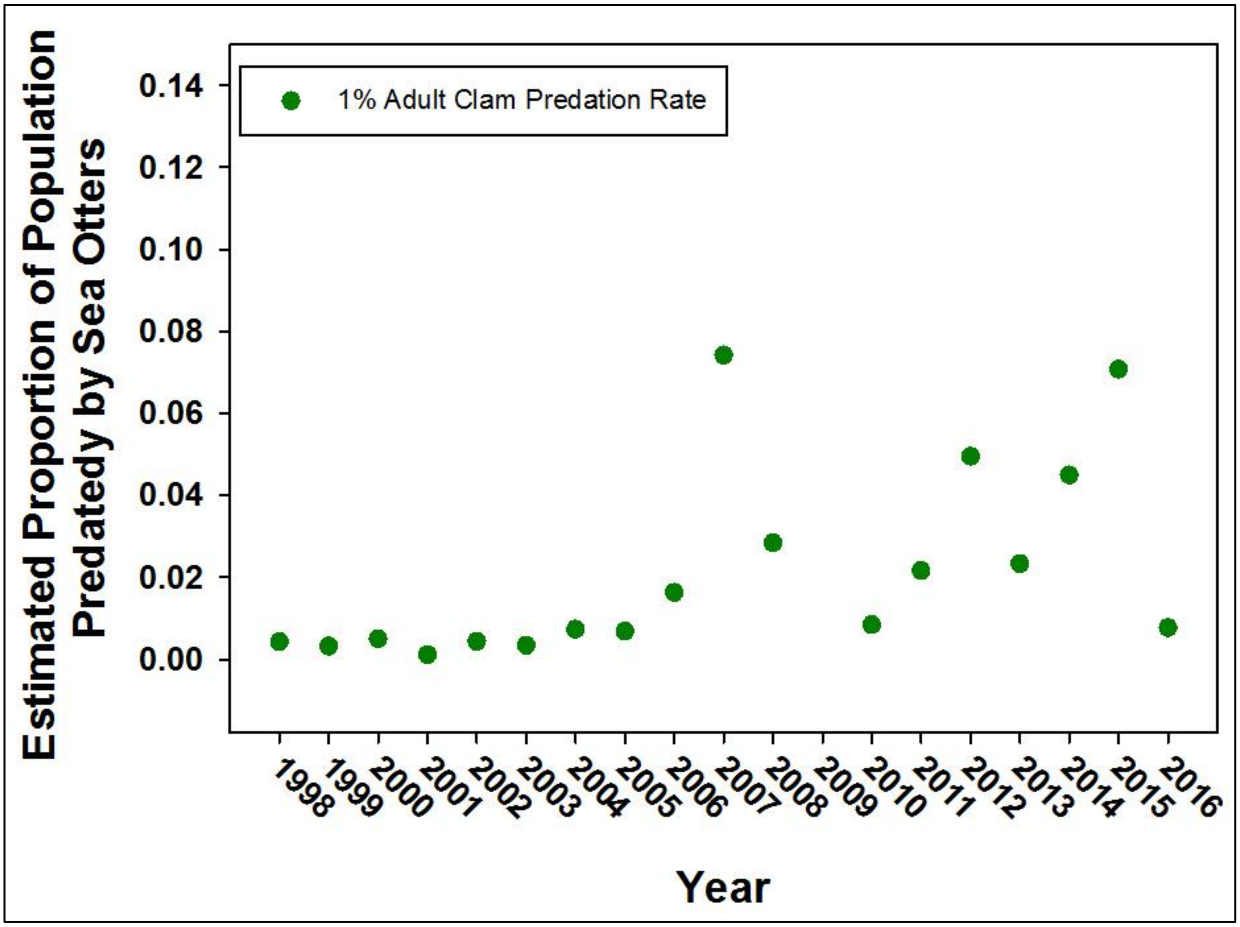

Figure 4.2.25. Estimated proportion of the annual adult Kalaloch razor clam population predated by sea otters. Estimated impact is calculated as: (6% of the area otter population)* (1% adult clam predation rate), where each otter consumes 30% of its weight each day and the average adult weight is based on the average weight from Riedman and Estes (1990).

Physical Disturbance

While maximal wave height appears to be increasing on the Oregon coast (Ruggiero et al. 2010), there was no obvious trend in increasing wave action at the NDCB Cape Elizabeth buoy from 1994 to 2016 (Figure 4.2.26). In particular, there is no discernable pattern of increased wave action coincident with razor clam population declines.

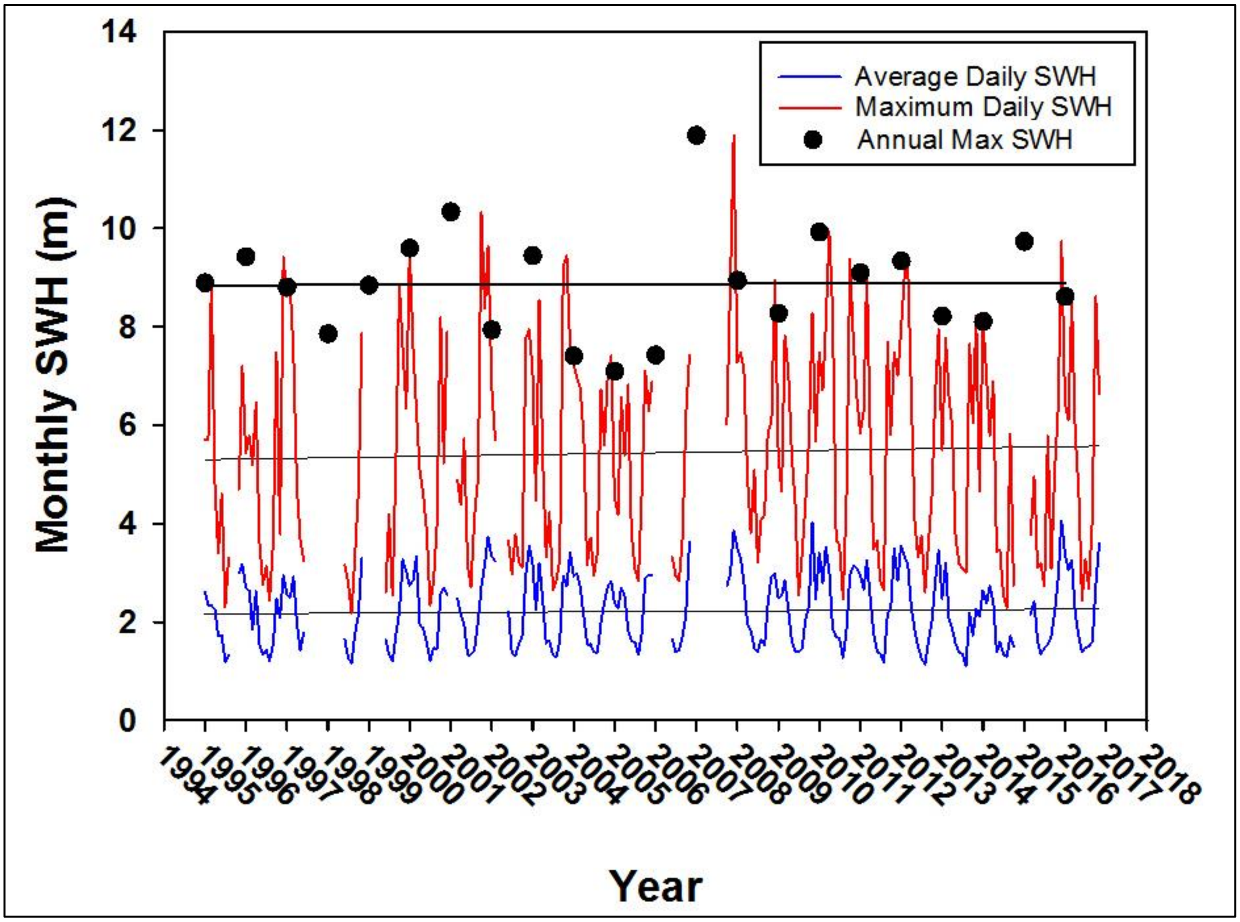

Figure 4.2.26. Average (blue) and maximum (red) daily significant wave height (SWH) and annual maximum SWH (black) for the Cape Elizabeth NDBC Buoy (46041) offshore near the Kalaloch area. Black lines are regressions for each dataset. Source: https://www.ndbc.noaa.gov/station_page.php?station=46041; accessed 4 October 2018.

Emerging Issues

Sea Level Rise

Rising sea levels associated with global climate change are expected to impact the outer Washington coast over the next 50-100 years, with the southern Olympic coast (Kalaloch) affected to a greater degree than the northern coast due to shorter-term uplift of land associated with the retreat of the Cordillarian ice sheet ~10,000 years ago. Current estimates for various scenarios predict a 3-45 cm sea level rise by 2050 and a 6-108 cm rise by 2100 (Mote et al. 2008). The impact of the high end of these estimates on razor clam populations is unclear. Rising sea levels will reduce the amount of intertidal beach habitat at lower elevations, making them subtidal habitats. Razor clams are known to be present in shallow subtidal habitats (Lassuy and Simons 1989), but the extent to which they persist over time and their role in overall population dynamics is unstudied. The creation of new intertidal beach at higher elevations via bluff erosion is likely, yet changes in sediment composition and the vertical extent of potential sand habitats has not been examined. The Olympic coastline is a historically eroding coastline. Razor clams have evolved in this environment, thus they will likely take advantage of new habitats created by sea level rise, as they have for millennia.

Physical Disturbance

The frequency and magnitude of storm events and associated physical disturbance of intertidal sediments is expected to increase with increasing climate change (Ruggiero et al. 2010). While not yet observed on the Olympic coast, increased future wave action has the potential to impact razor clams through excavation that leads to mortality.

Ocean acidification

Ocean acidification appears to have stabilized over the last 7 years (Figure 4.2.24). However, with atmospheric concentrations of CO_2 above 400 ppm, and increased CO_2 loading of the oceans likely to continue for the foreseeable future, continued acidification of Olympic coastal waters is predicted (IPCC 2014). Olympic coastal seawater is already corrosive to calcium carbonate structure for significant portions of the year. The presence of more frequent, more corrosive water will likely have impacts on razor clam recruitment and survival, similar to those already being observed on oysters within the Salish Sea (Feely et al. 2012).

Sea Otter Predation

The southern Olympic coast sea otter population has grown markedly over the last 18 years and appears to be reaching its carrying capacity. Beginning in 2015, large rafts of foraging otters have been observed making forays from Destruction Island. In 2016, for the first time, large numbers of otters were observed in the Kalaloch area, with concomitant decreases at Destruction Island 4.8 km offshore (Figure 4.2.20). This shift may be due to otters opportunistically taking advantage of unnaturally high numbers of juvenile razor clams at Kalaloch in 2015 (Figure 4.2.15), but it also suggests that the otter population is becoming resource limited and is foraging more broadly. As the otter population increases or stabilizes, it may rely on Kalaloch razor clams as a food source more than it has historically.

Pathogens

As climate change proceeds, frequency and magnitude of marine pathogen impacts are expected to increase (Burge et al. 2014). Since 1980, two pathogens new to the Olympic coast have profoundly impacted marine organisms. The bacterium NIX has caused mass mortality of razor clams, and is likely still a consequential agent of mortality. In 2013, sea star wasting disease first appeared on the Olympic coast (see Chapter 4.3.2), and has since been observed in more than 22 sea star species along the west coast of North America (Eisenlord et al. 2016). While the origins and environmental controls of these pathogens are poorly understood, as are most marine pathogens, new marine pathogens affecting intertidal organisms are likely to emerge with more frequency in the future.

Information Needs/Gaps

Razor clam population data at Kalaloch have largely been collected to support harvest management activities. Significant information needs exist to better understand razor clam population dynamics and the factors that influence them.

Population connectivity

Pelagic razor clam larvae are known to disperse tens of kilometers. Early genetic work suggests there is substantial gene flow between Washington coastal populations, with more population differentiation occurring at broader scales between Washington and Alaska (Le Clair and Pheps 1994). However, no data exists to elucidate from where beach recruitment originates, whether beaches are self-seeding or whether recruitment is linked to the production of other beaches. There may be opportunities to use modern genetic and isotopic techniques to determine the source/sink nature of Washington razor clam beaches. This information would be very useful in managing razor clam beaches as a network of interconnected beaches.

The Role of Subtidal Razor Clams

While the management focus of razor clam populations is in the low intertidal zone where they are harvested, clams are known to be present at least in the shallow subtidal, potentially up to about 12 m deep (Lassuy and Simons 1989). Very little information exists on the presence and persistence of subtidal clams, and there is no information on their role in sustaining intertidal populations. The presence of substantial subtidal populations has the potential to change our understanding of razor clam population dynamics and our management of them.

Pathogens

The pathogen NIX has a demonstrated potential to severely impact razor clam populations. NIX is still present at high incidence levels at Kalaloch, and is likely a factor in the current status of the population. To date, we have no understanding of the life history of the NIX bacterium, whether it has multiple hosts, whether it remains dormant in sediments, whether it has a pelagic life-stage, or what environmental factors influence its prevalence. The recent development of a NIX molecular probe provides promise to begin to be able to address some of these questions.

Sea Otter Predation

Current information on sea otter abundance in the Kalaloch area is based upon annual aerial surveys conducted by WDFW and the USFWS. Little current information on otter resource use and razor clam foraging exists. As the area otter population increases and foraging shifts occur, there is the potential for a greater impact on razor clams. Encouragingly, a University of Washington graduate student is currently conducting graduate work on coastal sea otter resource use, so new information may soon be available (Jessica Hale, personal communication).

Resource Summary

Using 23 years of annual stock assessment and harvest data for the Kalaloch razor clam population, we conclude that the condition of this population has been deteriorating since at least 2005 (Table 4.2.5). There have been multiple year runs of substantial non-harvest adult mortality, leading to low population sizes. The size class structure of the population is frequently dominated by single juvenile or young adult cohorts, suggesting poor survivorship to larger size class. In recent years, the average adult size has continually decreased, further suggesting that larger/older clams are not surviving. Conditions at Kalaloch appear to be continuing to decline. Low juvenile and adult survival from 2015 to 2016, poor harvest results during the fall of the 2016-2017 season, and low adult survival from 2016 to 2017, suggest that there has been continued high non-harvest mortality in the adult population.

Table 4.2.5. Resource condition summary for razor clams in Olympic National Park.

Resource	Measure	Condition Status and Trend	Rationale
Kalaloch razor clam population	Population size over time	🔴	Substantial non-harvest adult mortality
	Size-class distribution	🔴	Size class structure is uneven
	Average adult size	🔴	Mean adult size has decreased over time
Overall condition and satus of Razor Clams		🔴⬇	All measures of condition are of concern and have shown deterioration in condition over time

4.3. What are the population trends and ecological impacts of key declining and/or sensitive species or communities?

National parks play various roles in protecting habitat and populations of key species or taxa of conservation interest or concern. In some cases, national parks can contain most or all of an endemic species' range, and thus be critical to the long-term viability of unique species. National parks can also contain species that are declining for reasons outside of a park's control. In these cases, parks can serve as critical areas to monitor patterns of decline (and possibly recovery) in areas that may be less impacted by other natural and anthropogenic stressors than non-protected areas. Finally, national parks can serve as a bellwether for understanding natural fluctuations and long term trends in occupancy, range, and population size for sensitive species that may be declining elsewhere. Among others, OLYM serves all of these roles. In the following three sections, we address three resources of particular interest to the park that fall under these broad categories. First, we assess the condition of the Olympic marmot, an iconic yet declining endemic species of major conservation interest to the park (Chapter 4.3.1). Second, we assess the condition of the sea star community (particularly *Pisaster ochraceus*), which have declined in the park as well as regionally in association with an outbreak of disease (Chapter 4.3.2). Finally, we assess the condition of the amphibian community in the park, which represents taxa of regional and global concern due to severe and ongoing declines (Chapter 4.3.3).

We note here that several other resources in OLYM that could fall under this guiding question are covered in other sections of the report. The condition of the razor clam, a declining species found along OLYM's beaches, is covered under section 4.2.2 because the core monitored population is also harvested on selected years. The condition of the declining northern spotted owl is covered in section 4.4.1 because the primary threat to this species in the park is the invading barred owl, which is one of the invasive species we assess. As outlined in Chapter 3, other sensitive or declining species of concern to the park are being assessed as part of other projects, such as the recently reintroduced Pacific fisher.

4.3.1. Olympic Marmots

Rebecca McCaffery and Patricia Happe

<u>Introduction</u>

The Olympic marmot is a large, social, ground-dwelling squirrel (family Sciuridae) found in subalpine and alpine meadows and talus slopes of the Olympic Mountains (>1,400 m). This charismatic species is endemic to the Olympic Peninsula and >90% of the species' habitat is found within OLYM, making park resources critical to the survival of the species. The Olympic marmot was added to the State Candidate species list in 2008, and it was designated a State Endemic Mammal in 2009. Globally, it is listed as Least Concern by the International Union for the Conservation of Nature (IUCN). The demography and distribution of marmots is known better than many other species in OLYM as a result of historical (Meagher 1957, Barash 1968, Barash 1973, Wood 1973, Houston and Schreiner 1994) and recent (Griffin et al. 2007a, Griffin et al. 2007b, Griffin et al. 2008, Griffin et al. 2009, Witczuk et al. 2009, Griffin et al. 2010, Witczuk et al. 2013) research and monitoring activities, and the management implications of this research have been elaborated in detail.

Olympic marmot. Photograph courtesy of NPS/Pete Zaidel.

Olympic marmots are one of several marmot species found in western North America. Recent genetic work suggests that Olympic marmots diverged from the nearby and closely related Vancouver Island (*M. vancouverensis*) and hoary (*M. caligata*) marmots about 2.6 million years ago (Steppan et al. 2011, Kerhoulas et al. 2015). The next-closest- related species is the yellow-bellied marmot (M. flaviventris), which is widely distributed in the mountains of western United States and southwestern Canada. Like hoary and Vancouver Island marmots, Olympic marmots hibernate for 6 – 8 months a year, are not reproductively mature until age 3 or 4 (Barash 1973, Barash 1974), and typically reproduce biennially (Barash 1973), though they can reproduce in consecutive years (Griffin et al. 2007a). Juveniles usually do not disperse until after the second annual hibernation. The species lives in groups of 2 – 30 individuals, and digs extensive burrow systems in subalpine and alpine meadow habitat. These habitats range from <5 ha to >100 ha, and are surrounded by a mixture of rock, snow, and forested habitats (Griffin et al. 2008). Olympic marmots are primarily herbivores, feeding on grasses and roots in the spring and a variety of flowering plants and sedges found in their meadow habitat in the summer (Barash 1973). However, they can be carnivorous when vegetation is scarce, likely scavenging on carrion (Barash 1973).

Conservation concerns for the Olympic marmot originated in the 1990s. At that time, park staff and visitors first noted that marmots had disappeared from some parts of the park where they had been commonly viewed previously (Griffin et al. 2008). Subsequent research confirmed that marmot colony occupancy had declined over recent decades as evidenced by disappearance of long-occupied colonies, continuing declines in abundance at extant colonies, and large areas of recently abandoned habitat throughout the park (Griffin et al. 2008). Numerous threats were hypothesized to potentially affect Olympic marmots. These included conifer encroachment leading to subsequent habitat loss and fragmentation (Griffin et al. 2009), tourist activity (Griffin et al. 2007b), direct effects of climate change (Griffin et al. 2008), disease, inbreeding depression, and predation by coyotes (Witczuk et al. 2013). Sue Griffin, who conducted in-depth studies of marmot population dynamics in the park in relation to these threats, hypothesized that the synergistic effects of predation, habitat changes, and climate change may all be interacting with a slow life history and limited dispersal capabilities to impact the Olympic marmot range-wide (Griffin et al. 2008, 2009).

Continued concern over the status of Olympic marmots following research conducted in the 2000s led to the development of a park-wide monitoring program for the species (Witczuk et al. 2009). Over the past six years, OLYM has implemented this monitoring program by relying on citizen scientists to survey known and potential colony sites for the presence of Olympic marmots. The purpose of this assessment is to synthesize the results of all past research and monitoring with park-wide occupancy surveys that have been conducted since 2010 to provide the best information on status and trends in the species going forward.

Approach and Methods

We examined three measures to assess the condition of the Olympic marmot:

1. (1) Long-term occupancy of select colonies. Here, we summarized the presence of marmots at 28 well-studied historical and current colonies over a 58-year time period, beginning with literature values and culminating with recent citizen science monitoring data.
2. (2) Contemporary park-wide occupancy. Here, we examined park-wide occupancy in two ways:
    - (a) We evaluated park-wide occupancy from surveys conducted from 2002-2005 and 2010-2015 by comparing the presence of marmots at sites surveyed in both of these time periods.
    - (b) We used dynamic occupancy models to formally estimate occupancy using the 2010-2015 data, where we examined regional differences in occupancy as well as potential drivers of variation in occupancy over time.
3. (3) Demography. We synthesized demographic parameters measured for Olympic marmots as well as other marmot species from the literature.

Olympic Marmot Occupancy

The term “occupancy” refers to the proportion of sampling sites inhabited by a species of interest (MacKenzie et al. 2006). In this assessment, we refer to both “raw” (often referred to as “naïve”) occupancy and “robust” occupancy. Raw occupancy is a simple calculation of the proportion of sites where marmots or marmot sign was directly observed. We examine this metric in the long-term dataset (measure 1), as well as parkwide (measure 2a). This metric was reported when sampling replication was insufficient to estimate detection probability, which is needed for robust estimation. Because wildlife species are detected imperfectly (i.e., not seeing an individual on a site does not necessarily mean it is not present), presence or absence of a species at a site cannot be measured perfectly, and can lead to biased estimates of occupancy that are typically lower than true occupancy. A class of models called occupancy models uses information from repeated observations each year to estimate detectability and thus acquire an estimate of occupancy that accounts for imperfect detection (MacKenzie et al. 2003, MacKenzie et al. 2004, MacKenzie et al. 2005, MacKenzie et al. 2006). We estimated robust occupancy using the current (2010-2015) parkwide dataset, where many colonies were visited two or more times per year (measure 2b).

Long-Term Occupancy of Select Colonies

Surveys were conducted periodically by various researchers and park staff at several areas around Hurricane Ridge, Hurricane Hill, Obstruction Point, Royal Basin, and Deer Park from 1957 to 2015. Occupancy of select colonies was recorded by Mary Meagher (1957), David Barash (1966-69), William Wood (1972), and Janis Burger (1989), and was summarized in Houston and Schreiner (1994). Surveys took place during the active season for marmots (i.e., the summer months). Sue Griffin then resurveyed most of these colonies between 2002 and 2006 as part of her dissertation research to determine changes in raw occupancy status between the 1950s and the 2000s. Many of these colonies were monitored intensively during those years as part of Griffin's demographic research. Finally, park staff and citizen science volunteers have visited many of these historical colony sites from 2010 – 2015 as part of the citizen science monitoring program described below. We therefore have a 58-year record of raw occupancy patterns for select colonies in the northeastern part of the park (Table 4.3.1). We summarized raw occupancy of these colonies and described changes in the percent of colonies occupied over this time period.

Table 4.3.1. Occupancy monitoring datasets used to assess condition of Olympic marmots in Olympic National Park. Source: NPS.

Author	Year(s)	Geographic Extent of Colonies Monitored	How Data are used in the Assessment
Mary Meagher	1957	Select colonies around Hurricane Ridge, Hurricane Hill, Obstruction Point Road, Obstruction Point, and Deer Park	• Long-term raw occupancy of select colonies
David Barash	1966-1969	Select colonies around Hurricane Ridge, Hurricane Hill, Obstruction Point Road, and Deer Park	• Long-term raw occupancy of select colonies
William Wood	1972	Select colonies around Hurricane Ridge, Hurricane Hill, Obstruction Point Road, Obstruction Point, and Royal Basin	• Long-term raw occupancy of select colonies

Table 4.3.1 (continued). Occupancy monitoring datasets used to assess condition of Olympic marmots in Olympic National Park. Source: NPS.

Author	Year(s)	Geographic Extent of Colonies Monitored	How Data are used in the Assessment
Janis Burger	1989	Select colonies around Hurricane Ridge, Hurricane Hill, Obstruction Point Road, Obstruction Point, Royal Basin, and Deer Park	• Long-term raw occupancy of select colonies
Sue Griffin	2002-2006	Park-wide	• Long-term raw occupancy of select colonies • Contemporary park-wide assessment of raw occupancy
OLYM staff and citizen science volunteers	2010-2015	Park-wide plus select Olympic National Forest sites (since 2012)	• Long-term raw occupancy of select colonies • Contemporary park-wide assessment of raw occupancy; • Robust occupancy analysis

Contemporary Park-wide Occupancy

In addition to the long-term data on select accessible colonies described above, we also examined patterns and trends in park-wide raw occupancy. We relied on data collected as part of Griffin's dissertation research and the present citizen science monitoring program to examine patterns of raw occupancy in the mid-2000s and the past six years, respectively (Table 4.3.1). From 2002 - 2005, Griffin conducted systematic surveys of potential marmot habitat throughout the park to determine the current and recent distribution of the species. Researchers used GIS to identify potential habitat based on research by Barash (1973) and Wood (1973), and established 3,516 irregularly-shaped polygons of potential high-elevation habitat from this effort (Figure 4.3.1; Griffin et al. 2010). They then used a stratified random sampling approach to select polygons to survey from within the potentially accessible habitat, resulting in surveys of 809 polygons. Full details on site selection are provided in Griffin et al. (2008). Crews then visited each polygon on foot and systematically surveyed the area for presence of marmots and burrows. Sites were classified as occupied (marmot or active burrow detected), abandoned (sign of inactive marmot burrows), or no sign. Active burrows contained one or more of the following signs: fresh digging, clipped or compacted vegetation near the entrance, marmot odor or flies, and trails between entrances. Abandoned burrows were still easily visible, but had no sign of recent digging, clipped vegetation, tracks, or other sign of marmot activity. Griffin et al. (2008) estimated that abandoned burrows could be identified for up to 7-15 years, after which they would be too overgrown to discern.

Figure 4.3.1. Map of potential marmot habitat used for designing the monitoring study, based on Griffin et al. 2010, and later extended to Olympic National Forest by Olympic National Park staff. Source: NPS.

From this initial survey work, graduate student Julia Witczuk developed a park-wide monitoring program to determine changes in marmot colony occupancy over time. Full details on site selection and design of the monitoring program can be found in Witczuk et al. (2009). This monitoring program was implemented by the park using volunteer citizen scientists volunteers from 2010-2015. In this program, volunteers visited individual units (polygons of potential habitat) combined into a cluster of mapped habitat patches and grouped into trips that varied from a single-day trip to an eight-day backpacking trip (Figure 4.3.2). Some clusters were designated as core clusters to visit each year, while others were designated as clusters to visit occasionally (Witzcuk et al. 2009). Volunteers systematically searched each unit in the cluster looking for marmots or marmot sign and presence of both active and inactive (i.e., abandoned) burrows, following Witczuk and Griffin's protocol. If a marmot was seen, the patch was classified as occupied. If no marmot was seen, the whole patch was surveyed on foot (or using binoculars if deemed too steep or dangerous) to look for sign of recent marmot activity, such as active burrows. Volunteers were encouraged to take pictures for verification of burrow status and recorded track logs of their surveys using GPS. Sites without marmots were classified as "abandoned" or "no sign" as in Griffin's surveys. As part of the monitoring program, replicate surveys of certain sites were conducted in at least some years, with 25-30% of units visited >1 time each year. Volunteers conducted surveys in August and early September of each year. The goal of the program was to survey all core clusters each year (*sensu* Witzcuk et al. 2009), and 80% of the units in core clusters were typically successfully surveyed. Other clusters were designated for occasional surveys, and were typically surveyed every 2 – 3 years, often without replication within years. From 2010 – 2015, 2,345 surveys were conducted on 377 separate units of potential marmot habitat across the park and in nearby national forest units on the northeastern corner of the park (Table 4.3.2; Figure 4.3.2). Results of this survey effort were compared to Griffin's park-wide surveys as well as analyzed in the occupancy analysis described below (Table 4.3.2).

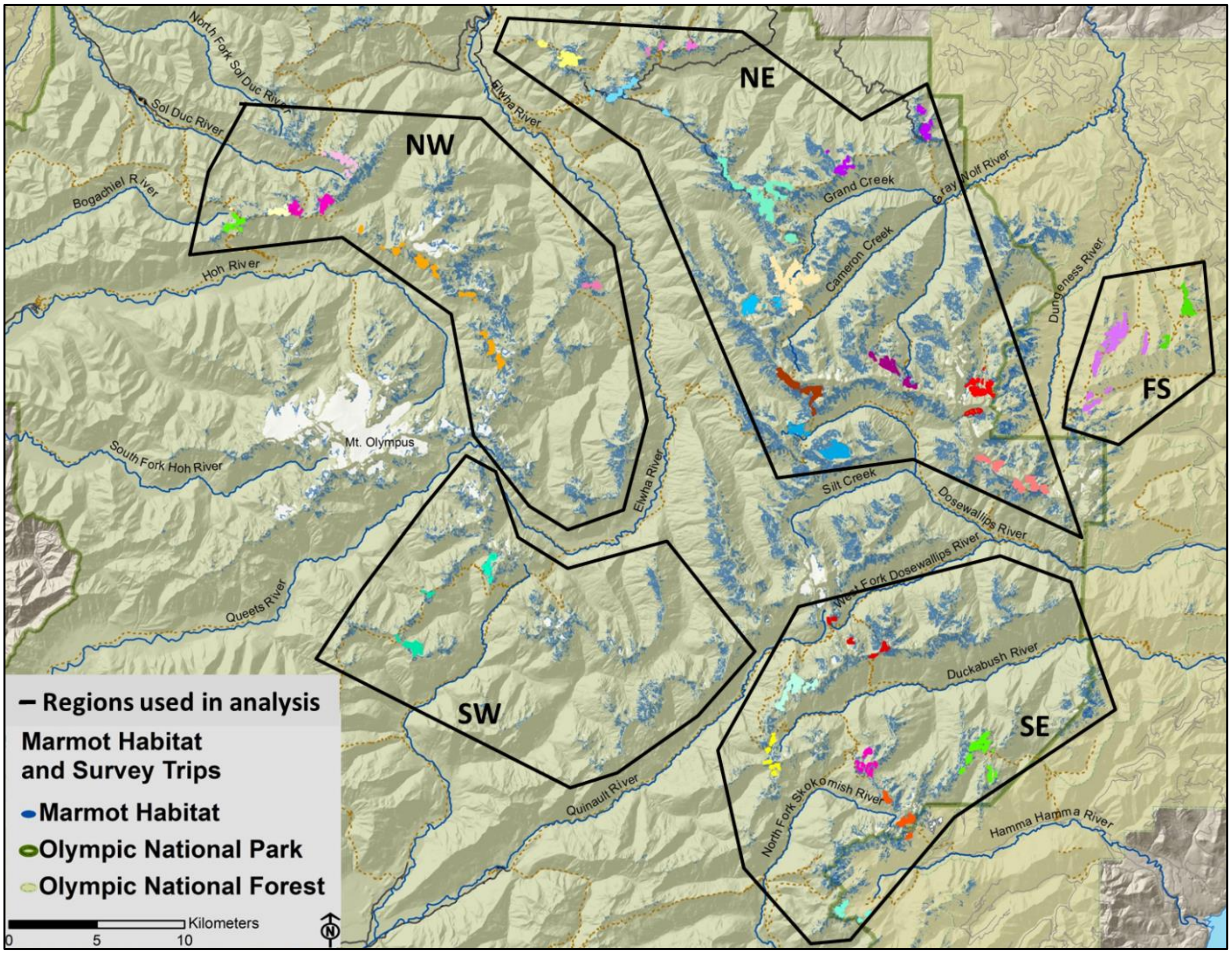

Figure 4.3.2. Marmot habitat and survey trips in Olympic National Park and Olympic National Forest. Colored polygons represent the different monitoring trips assigned to citizen science volunteers. Within each trip are several units of potential habitat to survey. For the analysis, the park was broken into different regions of interest: northwest (NW), northeast (NE), southeast (SE), southwest (SW), and Forest Service (FS). Source: NPS.

Table 4.3.2. Citizen science Olympic marmot monitoring survey effort by region (Forest Service units are included in the Northeast region), 2010 – 2015. Source: NPS.

Region	Number of Trips/Year	Number of Clusters	Number of Units	Total Number of Surveys, 2010-2015	Average Surveys per Unit per Year
Northwest	6	12	63	465 (19.8%)	1.2
Northeast	14	41	225	1423 (60.7%)	1.1
Southeast	6	15	76	373 (15.9%)	0.8
Southwest	1	3	13	75 (3.6%)	1.0
Total	27	71	377	2345	1.0

Raw Park-wide Occupancy

To describe range-wide changes in raw occupancy over two decades, we determined which units surveyed from 2010 – 2015 were also surveyed in the mid-2000s by intersecting Griffin's survey points with currently surveyed polygons, resulting in 171 overlapping polygons. We compared the qualitative results from the two survey periods. Specifically, we examined what percent of units found to be occupied in the mid-2000s were determined to be occupied at least once during 2010 – 2015.

Robust Occupancy Analysis, 2010-2015

To conduct a more statistically robust analysis of current occupancy (2010 – 2015), we used hierarchical Bayesian occupancy models implemented in R (R Core Development Team 2015) and JAGS (Plummer 2003) to determine occupancy of Olympic marmot colonies throughout the range of the species over time (McCaffery and Happe, *in prep*). This model allowed us to estimate occupancy, extinction (the probability of an occupied unit becoming unoccupied), and colonization (the probability of an unoccupied unit becoming occupied) rates while accounting for differences in detection probability. Further details on the model structure can be found in McCaffery and Happe (*in prep*).

Using these models, we first examined if park-wide occupancy had changed over time by estimating annual occupancy and the number of occupied sites without any covariates. We then examined if occupancy varied by region of the park (NW, NE, SW, and SE) and Forest Service (Figure 4.3.2). We did this analysis because of concerns over range contraction across the southern part of the range, where habitat is sparser and less connected than in the northern regions and where occupancy appears to be lower (Figure 4.3.1). We then examined if differences in occupancy over time could be explained by variation in snowpack during the previous winter (an index of winter severity) or growing degree days during the summer (an index of the length and warmth of the summer).

Olympic Marmot Demography

Demographic data for Olympic marmots are more limited than the presence/absence data described above. Early accounts of survival and reproduction in Olympic marmots were recorded by David Barash, who conducted his master's and dissertation research on the social behavior of Olympic marmots from 1966 – 1969 (Barash 1968, 1973). Subsequently, Sue Griffin conducted intensive demographic work as part of her dissertation research from 2002 to 2006 (Griffin et al. 2008). She studied marmots at three general locations in the park: Hurricane Ridge (including colonies near the visitor center as well as around Hurricane Hill), Obstruction Point (including colonies on the road to Obstruction Point), and Royal Basin. She used radio-telemetry and detailed field observations to estimate survival and reproductive rates for marmots in these three populations. She also synthesized Barash's field data to compare her estimates to his work. Details of the survey methods and analytical approaches are summarized in Griffin et al. (2008). In this assessment, we present the demographic rates from this paper, which include estimates derived from Barash's work as well as the detailed analyses from Griffin's work.

To provide more context for this demographic information, we conducted a literature review to summarize demographic rates of other North American marmot species. First, Griffin et al. (2008) summarized work on the nearby Vancouver Island marmot, which we include in this assessment. In addition, we assembled previously published demographic rates for both the hoary and yellow-bellied marmots. We were only able to find one study that estimated survival in the hoary marmot (Patil et al. 2013). Yellow-bellied marmot demographic rates come from a series of studies at a long-term field site near Gothic, Colorado. We summarized the most recent syntheses of spatial and temporal variation in survival from colonies at this site (Schwartz et al. 1998, Ozgul et al. 2006, Borrego et al. 2008).

Reference Conditions

Long-term Occupancy of Select Colonies

We used colony occupancy measured from the 1950s to 1989 as our reference conditions for the long-term occupancy of select colonies in the northeast part of the park. We use these data because these are the earliest quantitative data we have on marmot occupancy in the park, and they pre-date observations of declines by park staff and visitors in the 1990s.

Contemporary Park-wide Occupancy

We consider the current analysis of park-wide occupancy to provide a baseline by which the park can examine trends in Olympic marmot distribution into the future.

Olympic Marmot Demography

We use rates from the earlier literature on Olympic marmots (Barash 1973) and the nearby Vancouver Island marmot (Bryant 1996, Bryant and Page 2005) to compare to Griffin et al. (2008). We also looked at a range of demographic rates in more common species including the yellow-bellied (Schwartz et al. 1998, Ozgul et al. 2006, Borrego et al. 2008) and hoary (Patil et al. 2013) marmots. Collectively, these demographic rates provide a temporal reference (Barash 1973) and related species references (all other studies) to which we can compare recent demography of the species in OLYM.

Results and Assessment

Long-term Occupancy of Select Colonies

At areas monitored periodically from 1957 – 2015 (n = 28) in the northeastern portion of the park, we have seen a decline in occupied units from 28 to 17 (Table 4.3.3). This represents a loss of 40% of historical colonies that could be reliably relocated, and is driven by the loss of all colonies in the Deer Park/Blue Mountain area. Two units that were monitored historically are no longer monitored because the sites are unoccupied and habitat is no longer suitable due to conifer encroachment. These are located in the ski area at Hurricane Ridge. Three historical units were not included in this analysis because we could not determine their current location. Encouragingly, we have documented marmots in at least one year from 2010-2015 in 8 of 13 sites that were determined to have gone extinct in Griffin's surveys (Griffin et al. 2008; Table 4.3.3). These units were located in the Hurricane Ridge area and along Obstruction Point Road. Furthermore, no units that were occupied in the 2000s have become unoccupied in the 2010s. Therefore, the losses in these areas that appear to have occurred in the 1990s have not continued into the present, though the dynamics of units that have alternately been classified as occupied v. unoccupied in recent years is not well understood.

Table 4.3.3. Raw occupancy of select colonies of Olympic marmots in Olympic National Park from 1957 – 2015. ‘P’ indicates that marmots were present in that colony in a given year, ‘A’ indicates they were not seen, and dashes indicate that surveys were not conducted in that year. Sources for data in each year are indicated in the footnotes.

Area	**Colony**	**1957[1]**	**1966[2]**	**1967[3]**	**1968**	**1969**	**1972[4]**	**1989[5]**	**2002[6]**	**2003**	**2004**	**2005**	**2006**	**2010[7]**	**2011**	**2012**	**2013**	**2014**	**2015**
Hurricane Ridge area	1 – Lodge	–	P	P	P	P	–	P	A	A	A	A	A	A	A	A	A	A	P
	2 –Bartholomew	P	P	P	P	P	–	P	P	A	A	A	A	A	A	A	A	A	P
	3 – Widow	P	–	P	P	P	P	P	P	A	A	A	A	P	P	P	P	P	P
	4 – Henderson	P	P	P	P	P	–	P	A	A	A	A	A	–[8]	–	–	–	–	–
	5 – Henderson Annex	–	–	A	P	P	–	P	A	A	A	A	A	–[8]	–	–	–	–	–
	6 – Meander	–	–	P	P	P	–	P	A	A	A	A	A	A	A	P	A	A	A
	7 – Marigold	–	–	P	P	P	P	P	A	A	A	A	A	–	P	–	A	–	A
	8 – Picnic	P	P	P	P	P	P	P	P	P	P	P	P	A	P	P	P	P	P
Hurricane Hill area	9 – Ridgely	P	–	–	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P
	10–Allee/ Grasshopper	–	–	–	–	P	P	P	A	A	A	A	A	P	P	P	P	P	P
	11 – Cornus	P	–	–	P	P	P	P	P	P	P	P	P	P	P	P	P	P	P
	12 – Zenith	P	–	–	–	P	–	–	P	P	P	P	P	P	P	P	P	P	P
	13 – Aegean	–	–	–	P	P	–	A	P	P	P	P	P	–	P	A	A	A	P
Obstruction Point Road	14 – Steeple	P	–	P	P	A	–	P	A	A	A	A	A	A	A	A	A	A	A
	15 – Aureus (Eagle Point)	P	P	P	P	P	P	P	P	P	P	P	P	P	–	P	P	P	P
	16 – Aureus Annex	–	–	A	A	P	–	P	A	A	A	A	A	–	–	P	P	P	P

[1] Meagher (1957)

[2] Barash (1968)

[3] Barash (1973), 1967 – 1969

[4] Wood (1973)

[5] Houston and Schreiner (1994)

[6] Griffin et al. (2008), 2002 – 2006

[7] OLYM citizen science program, 2010 – 2015

[8] No longer surveyed because habitat is no longer suitable

Table 4.3.3 (continued). Raw occupancy of select colonies of Olympic marmots in Olympic National Park from 1957 – 2015. 'P' indicates that marmots were present in that colony in a given year, 'A' indicates they were not seen, and dashes indicate that surveys were not conducted in that year. Sources for data in each year are indicated in the footnotes.

Area	Colony	1957[1]	1966[2]	1967[3]	1968	1969	1972[4]	1989[5]	2002[6]	2003	2004	2005	2006	2010[7]	2011	2012	2013	2014	2015
Obstruction Point/ Grand Valley	17 – Badger Valley	P	–	–	–	–	P	P	P	P	P	P	P	P	P	P	P	P	A
	18 – High Trail	–	–	–	–	–	P	P	P	P	P	P	P	P	P	P	P	P	A
	19 – Grand Lake	P	–	–	–	–	P	–	P	P	P	–	P	–	–	–	P	P	P
Obstruction Point/ Grand Valley	20 – Moose Lake I	–	–	–	–	–	P	–	–	–	–	–	–	P	A	P	P	P	P
	21 – Moose Lake II	–	–	–	–	–	P	–	–	–	–	–	–	P	P	P	P	P	P
	22 – Grand Valley Tarn	P	–	–	–	–	–	P	–	–	–	–	–	P	P	P	A	P	A
Royal Basin	23 – East Side	–	–	–	–	–	P	P	P	P	P	P	P	P	P	P	P	P	P
	24 – West Side	–	–	–	–	–	P	P	P	P	P	P	P	P	P	P	P	P	P
Deer Park/Blue Mountain	25 – Basin East of Summit	P	P	P	P	–	–	P	A	A	A	A	A	–	A	A	A	A	A
	26	P	–	–	–	–	–	P	A	A	A	A	A	–	A	A	A	A	A
	27	P	–	–	–	–	–	P	A	A	A	A	A	–	A	A	A	A	A
	28	P	–	–	–	–	–	P	A	A	A	A	A	–	A	A	A	A	A

[1] Meagher (1957)

[2] Barash (1968)

[3] Barash (1973), 1967 – 1969

[4] Wood (1973)

[5] Houston and Schreiner (1994)

[6] Griffin et al. (2008), 2002 – 2006

[7] OLYM citizen science program, 2010 – 2015

[8] No longer surveyed because habitat is no longer suitable

Contemporary Park-wide Occupancy

Raw Park-wide Occupancy

During Griffin's park-wide surveys from 2002 – 2006, 219 of 809 units (27%) were classified as occupied, 111 (14%) were classified as abandoned, and 484 (60%) had no sign of marmots (Griffin et al. 2008). These data provided the first look at park-wide distribution of marmots. Because these sites were based on models of potential marmot habitat, the majority of units ("no sign") may never have contained marmots. There were large regional differences in occupancy status: the proportion of sites classified as occupied or abandoned (i.e., recent sign of marmots) was 11% in the southwest region, but >42% in the other three regions (northeast, northwest, and southeast). Furthermore, among polygons with sign of marmots (occupied or abandoned), a higher proportion of sites were recorded as abandoned in the southeast (78%) and southwest (58%) regions of the park compared to the northeast (26%) and northwest (24%) regions.

Park-wide, 163 units surveyed by Griffin from 2002-2006 were also surveyed at least once from 2010 to 2015 (Figure 4.3.3). Olympic marmots were present at 62% of sites from the mid-2000s that were also surveyed in the past 6 years. From 2010 – 2015, 66% of those sites were found to be occupied at least once during the survey period, representing a similar percent occupancy to the mid-2000s overall. Comparison of these two survey periods confirmed that no apparent recolonization has occurred in the southwest and southeast survey units that were determined to be unoccupied between 2002 and 2006, and suggested that some additional units in these regions have become unoccupied over the past ten years. Park-wide, we see some dynamics in apparent occupancy, with certain sites going from occupied to unoccupied, others going from unoccupied to occupied, and many remaining the same.

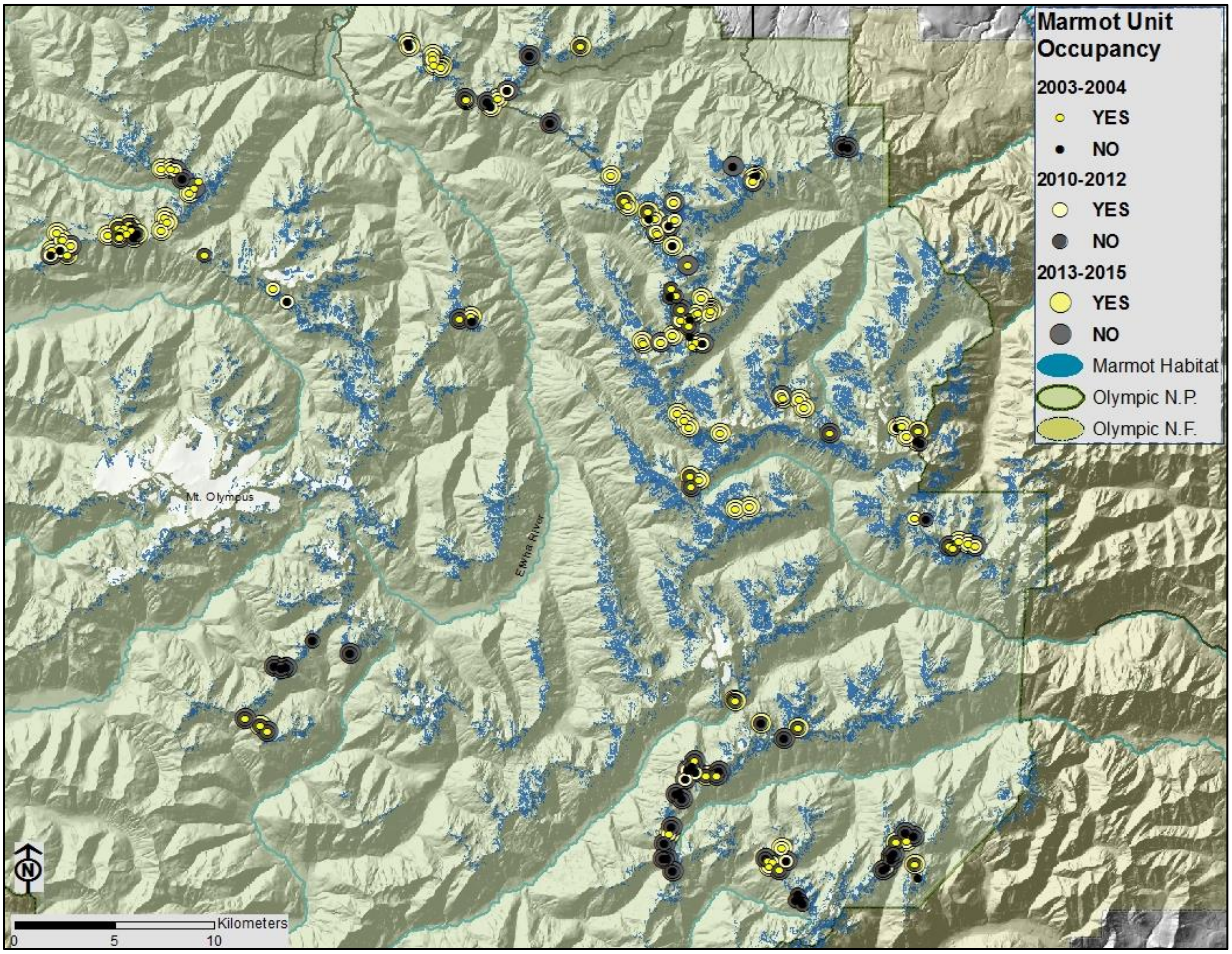

Figure 4.3.3. Comparison of naïve marmot occupancy in units surveyed by Griffin between 2002 and 2006 and by Olympic National Park volunteers from 2010 to 2015. "Yes" refers to an occupied site, and "no" refers to a colony classified as abandoned or where no sign was detected. Source: NPS.

Robust Occupancy Analysis, 2010-2015

Over the past six years, we found that Olympic marmot occupancy across its range has remained relatively stable (Figure 4.3.4). Estimated annual occupancy hovered between 50 and 60% of units. We note that raw values for occupancy rate are based on the units that were surveyed in those years only (not all units were surveyed in all years). By contrast, the occupancy models estimate occupancy for all survey units in each year, whether or not they were surveyed that year.

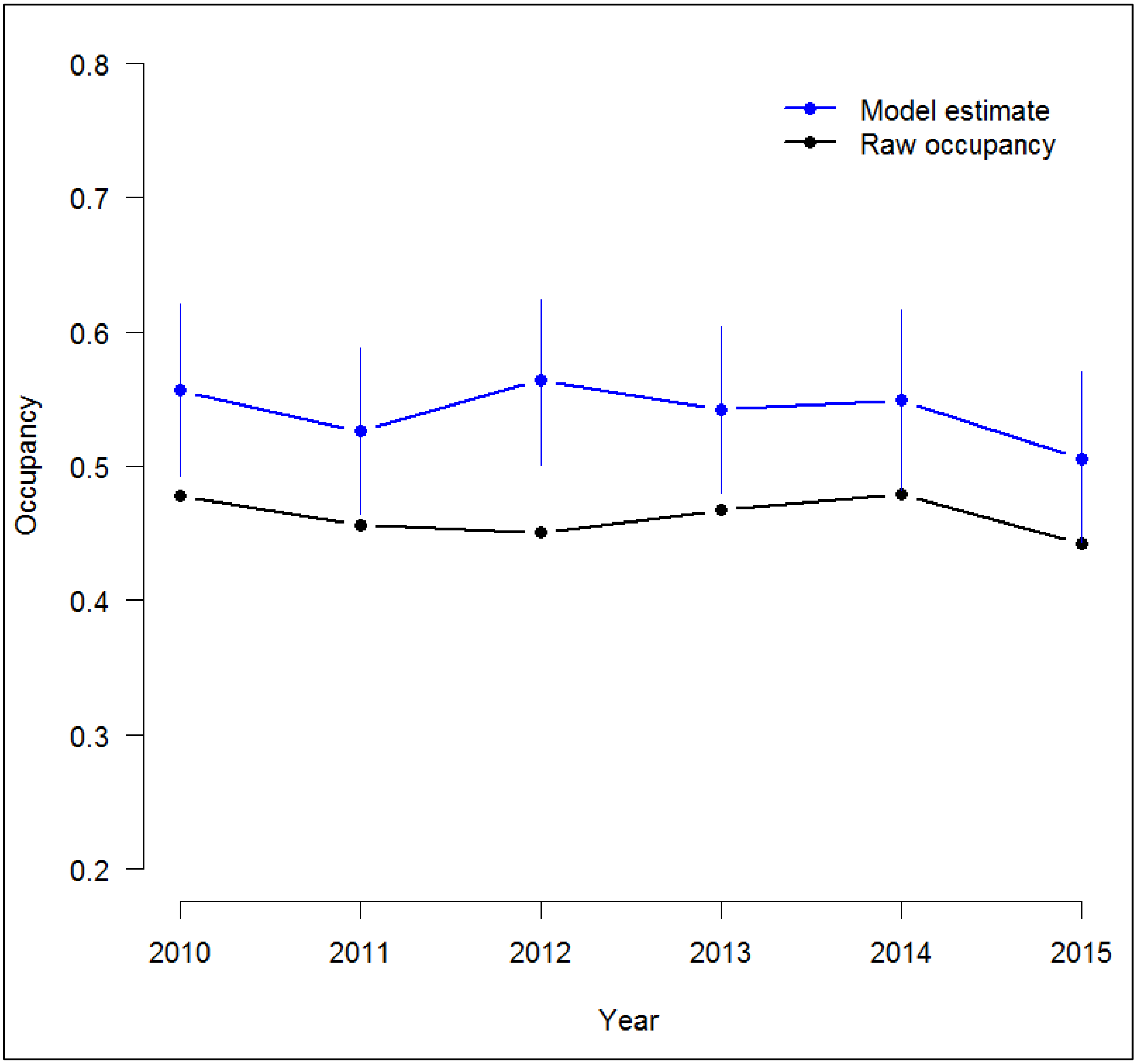

Figure 4.3.4. Park-wide estimated occupancy probability (blue line) with 95% credible intervals from 2010 to 2015, compared to raw percent occupied units uncorrected for detection probability. Source: analylsis of NPS data.

We found no relationship between occupancy probability and either snowpack in the previous year or growing degree days in the current year. These findings generally corroborate previous conclusions that changes in annual snowpack do not appear to directly relate to changes in marmot occupancy (Griffin et al. 2007a). However, we note that apart from 2015, snowpack levels were higher and less variable from 2010 – 2015 than they were during Griffin's study years (2002 – 2006).

Across OLYM, the highest occupancy probability was in the northeast region of the park (mean 0.69; Figure 4.3.5), where most of the potential marmot habitat is found (Figure 4.3.1). Occupancy was around 0.49 in the northwest region of the park, and was below 0.40 and more variable in all other regions (Figure 4.3.5). Mean colonization probability across years (the probability of an unoccupied unit becoming occupied) was highest in the northwest region (0.28 ± 0.05) and lowest in the northeast (0.17 ± 0.03) and southwest regions (0.17 ± 0.05), but confidence intervals overlapped across all regions. The mean probability of an occupied unit remaining occupied was highest in the northeast region (0.91 ± 0.02) and lowest in the southwest region (0.43 ± 0.12) and the Forest Service units (0.42 ± 0.12), signaling higher extinction rates in these latter two regions. Finally, mean turnover rates – an indicator of the stability of the occupancy patterns – were highest in the southwest region (0.63 ± 0.12) and lowest in the northeast region (0.08 ± 0.01). Together, these results indicate that occupancy is highest and most stable in the northeast region, and lowest and most erratic in the southwest region.

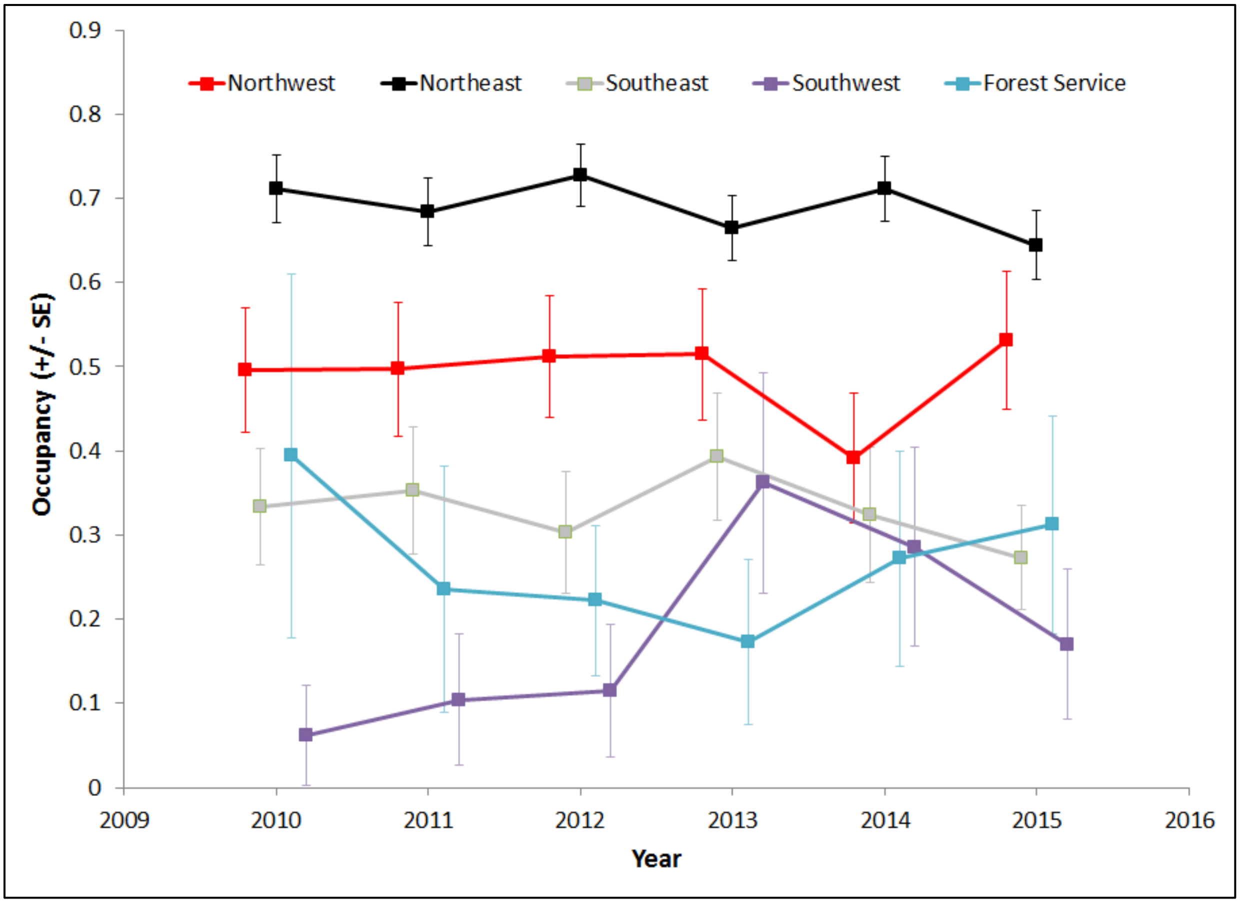

Figure 4.3.5. Modeled occupancy over time across five monitored regions. The four park regions correspond to the regions delineated in Griffin et al. (2008) and shown in Figure 4.3.2: northwest, northeast, southeast, and southwest. Forest Service units are found near the northeast corner of the park. Source: analysis of NPS data.

Olympic Marmot Demography

Olympic marmot demographic rates fall broadly into the range of values measured for other western North American marmot species (Table 4.3.4). However, estimates of demographic rates for Olympic marmots across three study areas in OLYM predicted a declining population (r = -0.07; Tables 4.3.2 and Griffin et al. 2008), where r values below 0 indicate a declining population. When vital rate estimates are broken down across Griffin's three study sites, Royal Basin appears to be more stable than the Hurricane Hill and Obstruction Point study populations (Table 4.3.5; Griffin et al. 2008). However, all three areas still showed declining growth rates.

Comparisons of Griffin's contemporary data to Barash's historical study of Olympic marmot demography suggest a decline in adult female survival (Table 4.3.4), which could be driving the declining population growth rate. However, we advise caution in comparing these two rates, since Barash used a crude calculation of survival of observed animals from a limited number of animals and study locations, whereas Griffin modeled survival in 40 females across three study populations using radio-telemetry data. A small change in the adult female survival rate in long-lived species can have a large influence on long-term population growth rate and viability of a population (Morris and Doak 2002).

Uses and Limitations

The historical occupancy data presented in this assessment provide a foundation for assessing changes in colony occupancy over a long (60-year) time frame, which is useful to documenting long-term changes in occupancy. However, these data are only available for a small number (~25) of colonies in a limited region of OLYM, and thus provided a limited understanding of how occupancy may have changed around the whole park over that time period. The range-wide comparison of raw occupancy data allowed us to better understand potential changes in occupancy over the past 10 – 15 years, but they do not provide a long historical perspective on park-wide occupancy. Finally, the contemporary analysis of occupancy in relation to various habitat and climate variables allowed us to assess potential drivers of change in occupancy more recently, and to assess occupancy in a statistically robust framework. These data will provide a useful baseline for future monitoring of park-wide marmot occupancy, and will further be used to refine the citizen science monitoring program. Our contemporary analysis is limited by relative lack of sampling in the southeast and southwest quadrants of the park compared to the more accessible and popular northeast and northwest quadrants of the park.

The demographic data provide a baseline for marmot survival and reproductive rates in certain populations, and include data from two periods. It is not known, however, how these data relate to the current demography of core populations in the park without long-term monitoring.

Table 4.3.4. Summary of historical and contemporary vital rate estimates for the Olympic marmot and other North American marmot species (mean values reported with 95% confidence interval [range] or standard error [single number] in parentheses). Adapted from Griffin et al. (2008) with the addition of data on yellow-bellied marmots (Schwartz et al. 1998, Ozgul et al. 2006, Borrego et al. 2008) and hoary marmots (Patil et al. 2013).

				Olympic Marmots	
Demographic rate	**Vancouver Island Marmots**[1]	**Yellow-bellied Marmots**[2]	**Hoary Marmots**[3]	**Historical**[4]	**Contemporary**[5]
Proportion of females weaning litters	0.41 (0.33 – 0.50)	–	–	0.41 (0.28 – 0.54)	0.36 (0.27 – 0.46)
Litter size	3.36 (0.83)	4.1 (0.1)	–	4.04 (1.20)	3.31 (1.14)
Female: male sex ratio	0.51 (0.41 – 0.61)	0.57 (All age classes) 0.7 (Adults only)	–	0.6	0.58 (0.44 – 0.70)
Juvenile survival	0.53 (0.42 – 0.65)	Females: 0.52 (0.42 – 0.63) to 0.78 (0.52 – 0.92) Males: 0.41 (0.06) to 0.60 (0.06)	0.40 to 0.93	0.55 (0.40 – 0.66)	0.60 (0.49 – 0.70)
Subadult survival	–	Females: 0.30 (0.19 – 0.45) to 0.78 (0.40 – 0.95) Males: 0.12 (0.03) to 0.26 (0.04)	0.25 to 0.98	–	0.83 (0.72 – 0.93)
Adult female survival	0.76 (0.68 – 0.83)	0.64 (0.57 – 0.71) to 0.76 (0.72 – 0.80)	0.48 to 0.76	0.89	0.69 (0.58 – 0.78)
Adult male survival	0.76 (0.68 – 0.83)	0.37 (0.07) to 0.55 (0.05)	0.48 to 0.76	–	0.79 (0.72 – 0.85)
Projected r	–	-0.04 (-0.10 – -0.01) to 0.08 (0.05 – 0.12)	–	–	-0.07

[1] Data were collected on remaining wild Vancouver Island marmots, located north of Cowichan Lake on the east side of the Island. Reproductive rates are from 36 litters and 134 females studied from 1987 to 1995 (Bryant 2005). Juvenile survival rates are from 75 juveniles (pers. comm. between A. Bryant and S. Griffin, reported in Griffin et al. 2008). Adult survival rates are from Bryant and Page (2005) using data collected from 1992 to 2004. No sample sizes were reported for subadult and adult survival rates.

[2] Studies were all conducted near Gothic, Colorado. Reproductive rates are from 1,534 marmots and 265 litters studied between 1962 and 1993 (Schwartz et al.1998). Male survival rates are from 1,541 individual males monitored between 1962 and 2006 (Borrego et al. 2008). Female survival rates and population growth rates are from 860 individual females monitored between 1976 and 2003 (Ozgul et al. 2006). Survival data come from four colony and 12 satellite sites, and ranges in survival and growth rates reflect spatial variation among these sites.

[3] Research on hoary marmots was conducted between 1999 and 2004 in the southwest Yukon Territory, Canada. Estimated rates reported are for 217 known-sex marmots (Patil et al. 2013). The authors did not find differences in male and female survival rates. Range represents annual variation.

[4] Historical data are from Barash (1973), who studied Olympic marmots from 1967 to 1969. Reproductive rates are reported for 59 females and 24 litters. Juvenile survival is reported from data on 64 individuals. Reproduced from Griffin et al. (2008).

[5] Contemporary data are from Griffin et al. (2008), who studied Olympic marmots from 2002 to 2006 at three locations. Reproductive rates are reported for 102 females and 29 litters. Survival rate samples sizes are: 89 juveniles, 35 subadults, 40 adult females and 82 adult males.

Table 4.3.5. Summary demography for Olympic marmots from 2002-2006 at three site groups, modified from Griffin et al. (2008). For litter size, the mean, with sample size, and standard deviation in parentheses, is shown. For all other rates, the sample size, and upper and lower 95% confidence limits are given. Some females were monitored as both subadults and adults.

Demographic Rate	**Hurricane Hill**	**Obstruction Point**	**Royal Basin**	**All Colonies**
Proportion of females weaning litters	0.38 (24; 0.19 – 0.59)	0.48 (44; 0.32 – 0.63)	0.21 (34; 0.09 – 0.38)	0.36 (102; 0.27 – 0.46)
Litter size	3.22 (9; 1.39)	3.33 (18; 1.08)	3.50 (2; 0.71)	3.31 (29; 1.14)
Female: male sex ratio	0.55 (11; 0.23 – 0.83)	0.59 (44; 0.43 – 0.74)	0.50 (4; 0.68 – 0.93)	0.58 (59; 0.44 – 0.70)
Juvenile survival (from emergence)	0.63 (24; 0.41 – 0.81)	0.57 (58; 0.43 – 0.70)	0.71 (7; 0.29 – 0.96)	0.60 (89; 0.49 – 0.70)
Juvenile survival (from tagging)	0.65 (20; 0.41 – 0.85)	0.64 (53; 0.50 – 0.77)	0.80 (15; 0.52 – 0.96)	0.67 (88; 0.56 – 0.77)
Adult female survival (annual)	0.62 (11; 0.47 – 0.75)	0.62 (18; 0.47 – 0.75)	0.80 (11; 0.61 – 0.91)	0.69 (40; 0.58 – 0.78)
Adult female survival (to July 15)	0.78 (11; 0.66 – 0.91)	0.78 (14; 0.66 – 0.91)	0.89 (10; 0.79 – 0.99)	0.83 (35; 0.72 – 0.93)
Adult male and subadult survival	0.78 (24; 0.69 – 0.85)	0.78 (40; 0.69 – 0.85)	0.89 (18; 0.75 – 0.96)	0.79 (82; 0.72 – 0.85)
Projected *r*	-0.12	-0.08	-0.01	-0.07

Emerging Issues

Climate Change

Many of the current hypothesized threats to the Olympic marmot can be linked to climate change. Both direct and indirect impacts of climate change present an ongoing threat to marmots globally (Armitage 2013). Warming temperatures have changed the timing of hibernation for many marmot species (Inouye et al. 2000) and have led to range shifts to higher elevations, where possible (Waltari and Guralnick 2009). Prolonged drought due to climate change could affect nutrient quality of the plants that marmots forage on during the active season and lead to decreased marmot fitness (survival and reproduction). Additionally, increase in the frequency of weather extremes – and interannual variability in weather conditions between heavy snow and drought – could lead to increased mortality and reduced reproduction (Armitage 2013). Changing temperature and precipitation regimes could also interact with other factors to affect marmot population viability. For example, by facilitating conifer encroachment to higher elevations, warmer temperatures could lead to loss of marmot habitat and dispersal restriction among habitat patches (Griffin et al. 2009).

Range Contraction and Colony Isolation

We are concerned about declines in occupancy in the southwestern region as well as some colonies in the southeast. These areas are isolated from the stable populations in the northeast part of the park: habitat is sparser, more linear, and separated from other habitat patches by valleys (Figure 4.3.1). We surmise that colony extinctions in this area have a particularly low probability of being recolonized due to low occupancy of adjacent areas and low connectivity with occupied habitat. More broadly, loss of habitat due to conifer encroachment upslope could lead to colony extinction and isolation. Range shifts due to increasing temperatures pushing marmots higher in elevation (where possible) could also lead to range contraction, increased isolation, and local extinction.

Coyote Predation

Coyotes are known predators of marmot species (Van Vuren 1991, Van Vuren 2001). Witzcuk et al. (2013) used an extensive scat analysis in marmot habitat to confirm that coyotes are the dominant carnivore in the OLYM high country, as well as the dominant predator of marmots (Witczuk et al. 2013). Coyotes were historically absent from the Olympic Peninsula, and were first documented around the time that wolves were extirpated (Scheffer 1995). Following their arrival, they increased in number and geographical distribution around the peninsula, though current abundance and rates of spread are unknown. Coyote predation may be linked to both climate change and vegetation characteristics. Lower snowpack may facilitate coyote movement to higher elevations earlier in the spring, and coyotes are more likely to travel in the open meadow habitats that marmots prefer than other mammalian predators of marmots such as bobcats (Witczuk et al. 2013).

Information Needs/Gaps

Changes in Temperature and Precipitation

Years of heavy snow into the spring can have a negative impact on marmot reproduction, but too little snow may remove important insulation for hibernating marmots (Inouye et al. 2000, Tafani et al. 2013) and exacerbate drought conditions in the summer. We saw no relationship between marmot occupancy and either peak snowpack or growing degree days (and index of summer length/intensity) over the past six year, but we anticipate that changing temperature and precipitation conditions could be important to marmot occupancy and population viability in the future. Long-term monitoring of marmot occupancy could include analyses of weather variation in relation to marmot occupancy or other measures of marmot population viability.

Habitat Quantity and Quality

Climate change could be impacting marmot habitat in two ways. First, warming temperatures may be facilitating the encroachment of conifers upslope into alpine meadows, leading to a reduction of marmot habitat that could impact total habitat available over time as well as dispersal potential between suitable habitat patches (Griffin et al. 2009). We conducted some preliminary analyses of occupancy in relation to vegetation composition, and results suggested that unit occupancy may be lower in sites with greater proportion of encroachment by trees and large shrubs, and higher in units with more meadow habitat. Future work could assess how vegetation characteristics are changing in modeled marmot habitat over time, and how that might be impacting changes in occupancy. Second, prolonged droughts (expected with climate change) could impact the quality of meadow habitat, which could have demographic consequences for marmots (Armitage 2013). Tracking drought conditions in relation to marmot occupancy and/or demography) could clarify the role of drought on marmot distribution and abundance.

Range Contraction

More frequent monitoring of areas of concern in the southwest and southeast would help the park understand the extent and timing of occupancy changes in these regions. Because of limited access, these areas are difficult areas to monitor compared to the northern parts of the range, but could be targeted when possible to have a better understanding of the occupancy dynamics in these regions. Along with increased monitoring of harder-to-access regions of concern, a formal connectivity analysis of potential marmot habitat and known colonies could help the park quantify how isolated different colonies or areas are from other areas and model the potential for recolonization using existing data on marmot dispersal rates and distances (Griffin et al. 2009).

Effects of Coyote Predation

There is considerable interest in (1) understanding how coyotes may directly be impacting marmot populations through predation, and (2) how coyote presence and predation activity in the higher elevations of OLYM may change if wolves return to the park, since wolves are known competitors of coyotes (Berger and Gese 2007). While we know that coyotes are the primary mammalian predator of marmots (Griffin 2007, Witczuk et al. 2013), we know little about their distribution, seasonal use of marmot habitat, and population dynamics in the park. Studies of coyotes in the subalpine and alpine ecosystems of OLYM would add to the park's understanding of their role in marmot population dynamics. For example, non-invasive sampling of coyotes via scat and hair collection could be used in a spatial capture-recapture framework to estimate coyote population size in regions of concern, or long-term cameras could be used to examine seasonal distribution of coyotes relative to snowpack.

Population Viability Assessment

Determining the long-term population viability and genetic health of the species would clarify the risks to species persistence over time, and could assess how changes in different population stressors or management strategies might affect population viability. While this kind of analysis would require additional data collection and expertise in population modeling, it would help determine trigger points for active intervention: when and how does the park engage in active translocation or rescue of individuals of the species?

Management Interventions

Tools such as fire (to reduce conifer encroachment) could be used to maintain habitat patches and enhance connectivity between patches. Assisted dispersal (where marmots are physically moved to abandoned but viable habitat) could be used to maintain or enhance connectivity among networks of marmot colonies, provided that chronic climatic and microclimatic stresses at abandoned sites are not harsher than at sites currently occupied by marmots. Modeling studies and controlled experiments could help address the need, feasibility, and utility of these and other tools to maintain current occupancy levels or restore population size and connectivity.

Effectiveness of Ongoing Citizen Science Monitoring Program

NPS managers are interested in determining the effectiveness of the citizen science program for assessing marmot occupancy. A goal of the analysis presented in this assessment was to gain a broader understanding of whether or not the current monitoring program is powerful enough to assess population trends in a meaningful manner. The program does generally appear to allow for robust estimates of occupancy in certain regions of the park, and a power analysis could help park managers understand the level of change that the program could detect, given different monitoring intensities. It is less clear how responsive measured changes in occupancy may be to changes in population dynamics or demography. Targeted studies that can assess how occupancy relates to dynamics of individuals colonies would help better guide how well occupancy tracks broader population dynamics in this system, though this would require more intensive study than limited monitoring resources currently allow. Finally, our analysis highlighted the potential challenge of citizen science in remote wilderness settings, since survey effort was lower in the more remote and difficult-to-access southern portions of the range. These regions often require longer trips to access sites and conduct surveys (i.e., up to eight days), and it is harder to find volunteers for this length of trip. Increased precision in understanding occupancy dynamics in the southwestern portion of the park in particular may require the use of paid natural resources technicians.

Resource Summary

Long-term monitoring of select colonies in the northeast part of the park has uncovered major declines between the 1950s and the early 2000s. Recent monitoring of these sites, as well as park-wide surveys for marmot occupancy, reveals populations that appear to be fairly stable (Table 4.3.6). Therefore, we characterize marmots as a resource of significant concern due to past declines and potential for future range contraction with an unchanging trend in the past six years. Climate change and coyote predation continue to be issues of concern that warrant further monitoring and study. Furthermore, while park-wide occupancy has been relatively stable, loss of colonies in the southern regions of the park warrants concern.

Table 4.3.6. Resource condition summary for the Olympic marmot in Olympic National Park.

Resource/Measure	**Condition Status and trend**	**Rationale**
Species occupancy – historical colonies	🔴	Major declines in colony occupancy measured between early studies and the mid-2000s at known, accessible colony sites
Raw parkwide occupancy	🟡	Similar occupancy measured in mid 2000s compared to 2010-2015 data at overlapping sites. Overall annual naïve occupancy higher in the present dataset compared to the mid 2000s, but depends on region of the park
Species occupancy – current monitoring program	🟡	Occupancy relatively stable in last 6 years of marmot monitoring program. No sign of major effects of potential stressors like snowpack variability.
Species demography	🔴	Studies are limited, but demographic rates appear similar to other species and past studies of Olympic marmots in most areas of the park. Concern over female survival and coyote predation effects leading to declining population growth rates.
Overall condition and trend of Olympic marmot	🔴 ↔	After declines in the 1990s, the current park-wide occupancy seems to have stabilized, but threats from coyote predation and climate change remain, and contraction of the range is of concern.

4.3.2. Sea Stars

Steven Fradkin

Introduction

Sea stars are vital and charismatic denizens of the rocky intertidal habitats that comprise over 50% of the OLYM shoreline. While park visitors gravitate to colorful sea stars during their trips to the margins of the sea, these echinoderms also play a substantial role in structuring intertidal communities. Of the 11 sea star species documented on the park shoreline, the ochre sea star, *Pisaster ochraceus* (hereafter "*Pisaster*"), is the most obvious, numerically abundant, and structurally important. For these reasons, *Pisaster* has received the most attention of all the park's sea stars. Found from the shallow subtidal to the mid rocky intertidal zone, this original keystone species (sensu Paine 1969) is a voracious predator, with an influence on its community disproportionate to its abundance. This influence ripples throughout the intertidal food web with its most obvious effect manifested as distinct horizontal bands of mussels, the lower bounds of which are often set by the upper levels of *Pisaster*'s predatory reach. While mussels are its preferred prey, *Pisaster* has a broad diet that also includes barnacles, snails, and other marine invertebrates (Sanford and Menge 2007).

Pisaster populations consist of outwardly identical males and females that release their gametes directly into the water column where fertilization occurs from May to July. Larvae feed in the water column for between 1-7 months before metamorphosis and settlement to rocky substrates as juvenile stars. *Pisaster* may live up to 20 years (Robles 2013). As an intertidal organism, *Pisaster* is exposed to two distinctly different thermal regimes. When the tide is in, *Pisaster* is covered with sea water that ranges between 7-14 ºC. When the tide is out, *Pisaster* is exposed to air and solar radiation that can result in body temperatures well above (summer) and well below (winter) submerged temperatures.

In June 2013, a large number of sick *Pisaster* individuals were observed while conducting annual long-term intertidal monitoring activities within OLYM (Figure 4.3.6). The symptomatic white surface lesions and graphic arm loss had not previously been observed on the Olympic coast. This affliction was subsequently identified as sea star wasting disease (hereafter "SSWD") and was observed across the west coast and Salish Sea from 2013 through 2017 (Figure 4.3.7; Stokstad 2014). While SSWD-like symptoms have been observed previously on the West Coast (Dungan et al. 1982, Eckert et al. 2000, Bates et al. 2009), the current SSWD event is different in several ways. Previous SSWD-like events have affected a limited number of sea star species in a restricted geographic locale for a relatively short duration.

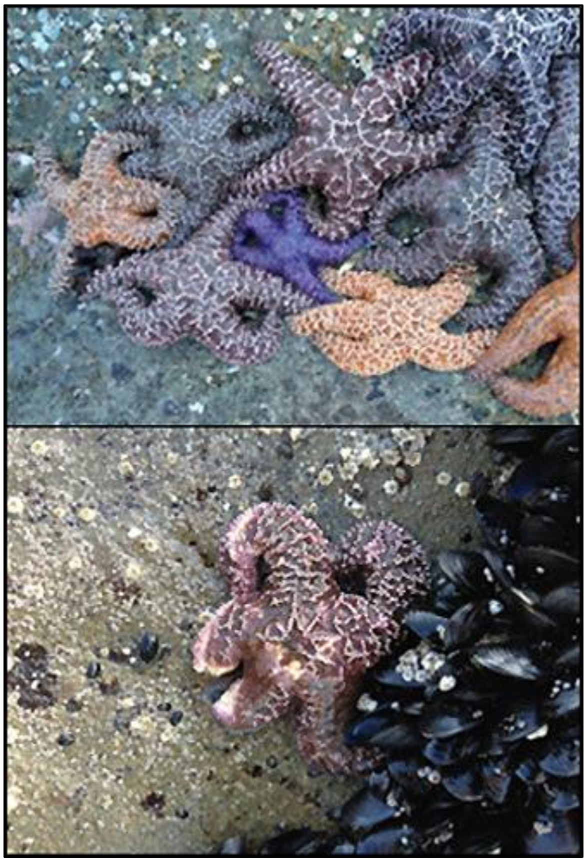

Figure 4.3.6. Top: Healthy *Pisaster* at Sokol Point. Bottom: *Pisaster* exhibiting symptoms of SSWD at Starfish Point. Photos courtesy of NPS/Steven Fradkin.

In this assessment, we examined trends in sea star relative abundance as related to the recent outbreak of SSWD in OLYM's coastal strip. We focus on the condition of *Pisaster*, which is the most ecologically dominant intertidal sea star and has been disproportionately affected by SSWD.

We also report trends in other species where data allow (Table 4.3.7), and identify all species where SSWD has been observed in OLYM.

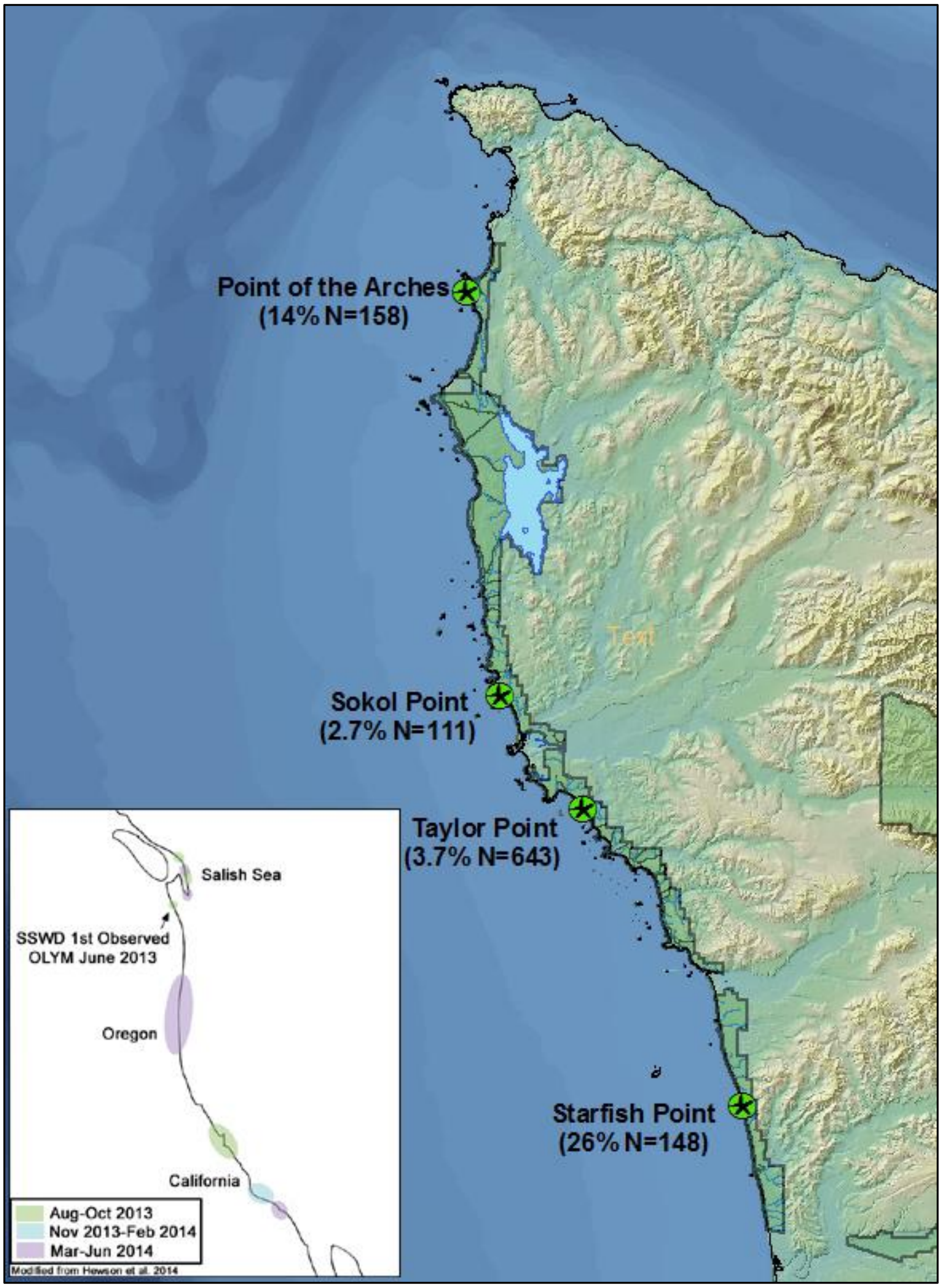

Figure 4.3.7. Map of the Olympic National Park coastal strip, with the four sea star long-term monitoring plots denoted by sea star symbols. The percent of *Pisaster* population displaying SSWD symptoms in June 2013 and the number of sea stars surveyed is shown for each plot. Inset map illustrates the time course of SSWD along the U.S. west coast through June 2014. Source: NPS; Inset map modified from Hewson et al. 2014.

Table 4.3.7. Sea star species documented in Olympic National Park rocky intertidal habitats, and how they are included in this assessment.

Scientific Name	Common Name	SSWD in[1] OLYM	SSWD in[2] Species Range	Assessment
Dermasterias imbricata	Leather star	Yes	Mid	Not included, not in monitoring plots, low tidal elevation
Evasterias troschelii	Mottled star	Yes	High	Not included, not commonly found in monitoring plots
Henricia spp.	Blood star complex	Yes	Mid	Included, limited relative abundance data
Leptasterias spp.	Six-rayed star	Yes	Mid	Included, limited relative abundance data
Mediaster aequalis	Vermilion star	No	Low	Not included, not in monitoring plots, low tidal elevation
Orthasterias koehleri	Rainbow star	No	Mid	Not included, not in monitoring plots, low tidal elevation
Patiria miniata	Bat star	No	Mid	Not included, not in monitoring plots, low tidal elevation
Pisaster ochraceus	Ochre sea star	Yes	High	Included, focal species: relative abundance, size class distribution, and prevalence of SSWD over time
Pycnopodia helianthoides	Sunflower sea star	Yes	High	Included, anecdotal observations, not in monitoring plots, low tidal elevation
Solaster dawsoni	Morning sun star	No	High	Not included, not in monitoring plots, low tidal elevation
Solaster stimpsoni	Striped sun star	No	Mid	Not included, not in monitoring plots, low tidal elevation

[1] Observation of SSWD in species in OLYM since 2013.

[2] Level of mortality associated with observed SSWD throughout the species range since 2013 as documented by the Multi-Agency Rocky Intertidal Network (MARINe; pacificrockyintertidal.org, http://www.eeb.ucsc.edu/pacificrockyintertidal/data-products/sea-star-wasting/species_affected_2016_0825.pdf, accessed 23 October 2017).

Approach and Methods

To determine the condition of *Pisaster* populations, we used NPS data from coast-wide sea star monitoring plots, in addition to SSWD surveys and continuous intertidal temperature data from the Starfish Point area of the southern park shoreline. We examined the following measures: (1) relative abundance of *Pisaster* at four monitoring sites; (2) size frequency of *Pisaster* over time at four monitoring sites; and (3) monthly prevalence of SSWD in relation to sea temperature anomalies at a focal monitoring site. Changes in size frequency, specifically a pulse in the relative abundance of small sea stars, provide a signal that a new generation has recruited into the population. To determine condition of the broader sea star community, we examined population trends of two other species when data were sufficient to extract time series of abundance from our intertidal monitoring plots. We also summarized observations of the SSWD presence in all OLYM sea star species, and the level of mortality observed in these species throughout their range outside of OLYM.

Sea star monitoring sites were established in 2008 at four sites on the Olympic coast as part of the park's long-term intertidal monitoring program (Figure 4.3.7). This methodology is consistent with similar surveys being conducted by the Multi-Agency Rocky Intertidal Network (MARINe) consortium at hundreds of sites, including NPS marine units, across the west coast of North America from Alaska to Mexico. Sites were monitored annually during a June index period to control for seasonal movements in sea star elevation (Robles 2013). At each site, we monitored three permanently marked, unequally-sized (~35-40 m^2) plots in the mid-rocky intertidal zone that represent suitable *Pisaster* habitat. All sea star species were counted in each plot, and the size of each *Pisaster* was measured. Abundance data from each plot were added together to obtain a total count for each site. *Pisaster* size was measured as the length to the nearest 5 mm from the middle of the central disc to the tip of the longest arm.

After discovery of SSWD in June 2013, we began regular surveys of SSWD incidence in *Pisaster*. We did not have easy access to the mid and north coast of OLYM, so we focused on the relatively accessible Starfish Point area on the south coast to monitor SSWD. Surveys were conducted monthly beginning in September 2013, except when the combination of inadequate low tides and/or high swell precluded access to the intertidal zone. During each survey, technicians walked haphazardly through the rocky intertidal for approximately 45 mins, counting and recording the disease class of at least the first 100 *Pisaster* encountered, but occasionally counting over 500 *Pisaster*. The disease class was based on a standardized classification scheme used by the MARINe research consortium (http://www.eeb.ucsc.edu/pacificrockyintertidal/data-products/sea-star-wasting/index.html, accessed 23 October 2017).

We included intertidal temperature data from the Starfish Point mid-intertidal zone temperature data logger, which provided a temperature reading every half hour from 2005-2016. We used these data to compute the monthly temperature anomaly for 2013-2016, defined as the number of degrees Celsius a given month deviates from the 2004-2014 average intertidal temperature at Starfish Point for that month.

Reference Conditions

NPS *Pisaster* abundance data for the Olympic coast were collected from 2008 to 2016. Data sets for other west coast regions go back to 1989. These longer-term data sets suggest that *Pisaster* populations in Alaska, the Salish Sea, and the outer coast of Washington have been gradually declining from 2008 to 2013. In contrast, populations in California and Oregon appear to be relatively stable from 2000-2013. However, given the relatively short time series of Olympic coast *Pisaster* data, it is unclear whether the gradual declines are part of a longer-term cycle, or indicative of a true decline. The onset of SSWD in 2013 has contributed to marked population declines. For this condition assessment, the population status and trend prior to 2013 was used as the reference condition.

Results and Assessment

Pisaster Abundance

From 2009-2016 there was a statistically significant decline (p=0.0176) in *Pisaster* abundance on the Olympic coast across the four monitoring sites (Figure 4.3.8). This decline appears to have begun prior to the onset of SSWD in 2013: the Point of the Arches, Sokol Point, and Starfish Point sites all showed declines between 2009 and 2012. After the onset of SSWD, declines were apparent at all 4 sites. These post-SSWD declines are consistent with observations of *Pisaster* declines across the entire west coast recorded by MARINe partners (Eisenlord et al. 2016, Menge et al. 2016; http://www.eeb.ucsc.edu/pacificrockyintertidal/index.html, accessed 23 October 2017), although the severity of the Olympic coast decline has been markedly lower than that observed in other regions of the west coast. In most areas, post-SSWD declines have been drastic, including local extirpation at locations within the Salish Sea and on the California coast.

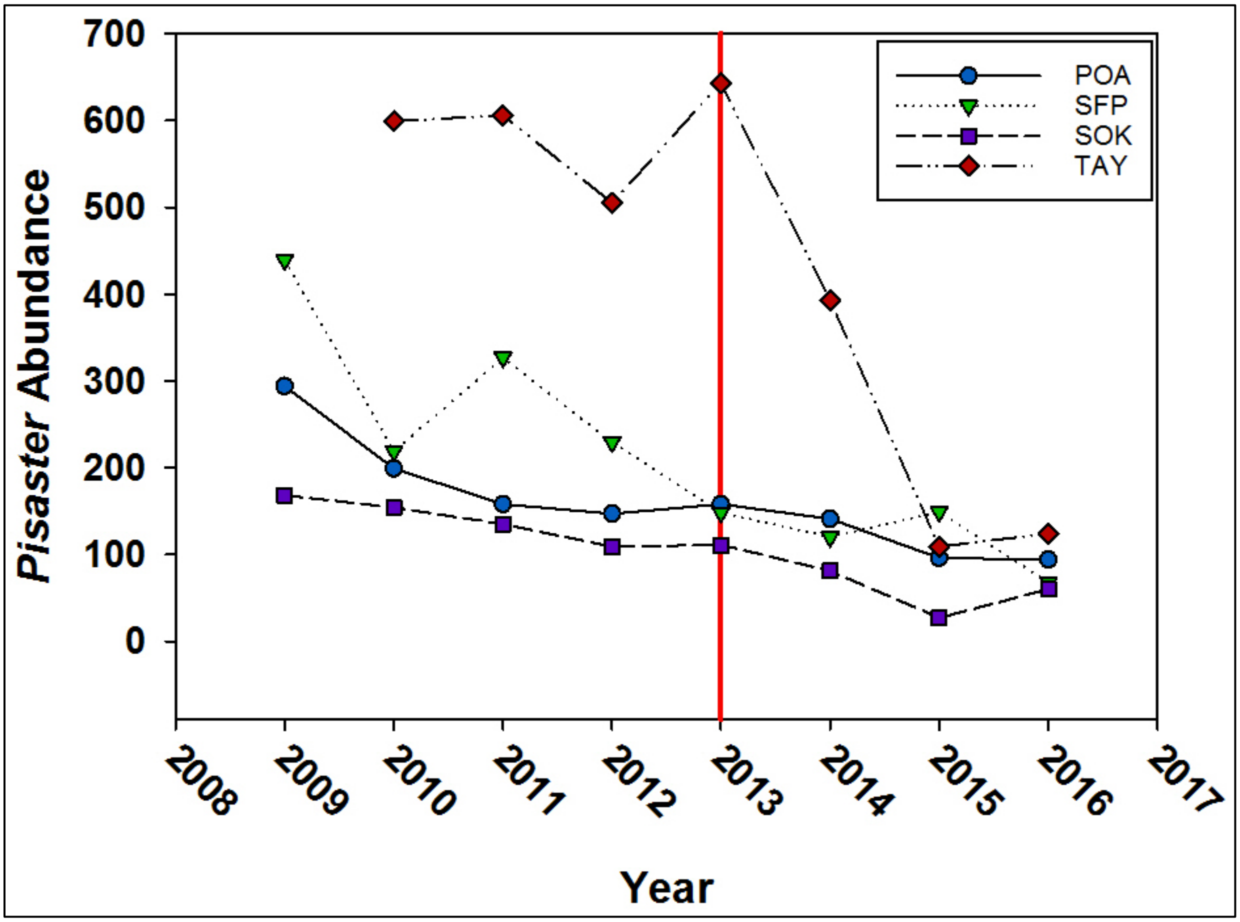

Figure 4.3.8. Trends in *Pisaster* abundance at long-term monitoring plots on the Olympic coast. From north to south: POA = Point of the Arches, SOK = Sokol Point, TAY = Taylor Point, SFP = Starfish Point. The red vertical line at 2013 represents the onset of SSWD on the Olympic coast. Source: NPS.

Pisaster Population Size Structure

Juvenile recruitment of Olympic coast *Pisaster* was very low from 2009-2013 (Figure 4.3.9), as indicated by the unimodal size frequency comprised of relatively large sea stars and few small sea stars through 2013. In 2014 to 2015 there was an uncharacteristically large addition of small young sea stars indicating a pulse of recruitment on the Olympic coast. Representative data for Starfish Point are presented in Figure 4.3.9. Data for Point of the Arches, Sokol Point and Taylor Point (not shown) are consistent with the Starfish Point data. This cohort of small sea stars has persisted into 2017 (data not shown). This recruitment pulse was also observed sporadically elsewhere throughout *Pisaster*'s range in California, Oregon, and in the Salish Sea (Menge et al. 2016).We have also recently observed a second large recruitment pulse on the Olympic coast in the winter of 2016-2017 (OLYM, unpublished data). Recruitment pulses are atypical for *Pisaster* populations in the Pacific Northwest (Menge et al. 2016). It is unclear what has caused them, though data from the Oregon coast suggest that SSWD did not trigger higher larval settlement (Menge et al. 2016). Rather, SSWD appeared to have indirectly facilitated higher recruit survival, possibly due to increased prey abundance caused by adult mortality prior to new larval settlement. This hypothesis is consistent with field observations of *Pisaster* mass spawning in early June on the Olympic coast from 2009-2013, suggesting adequate larval supply, but poor post-settlement survival (OLYM, unpublished data).

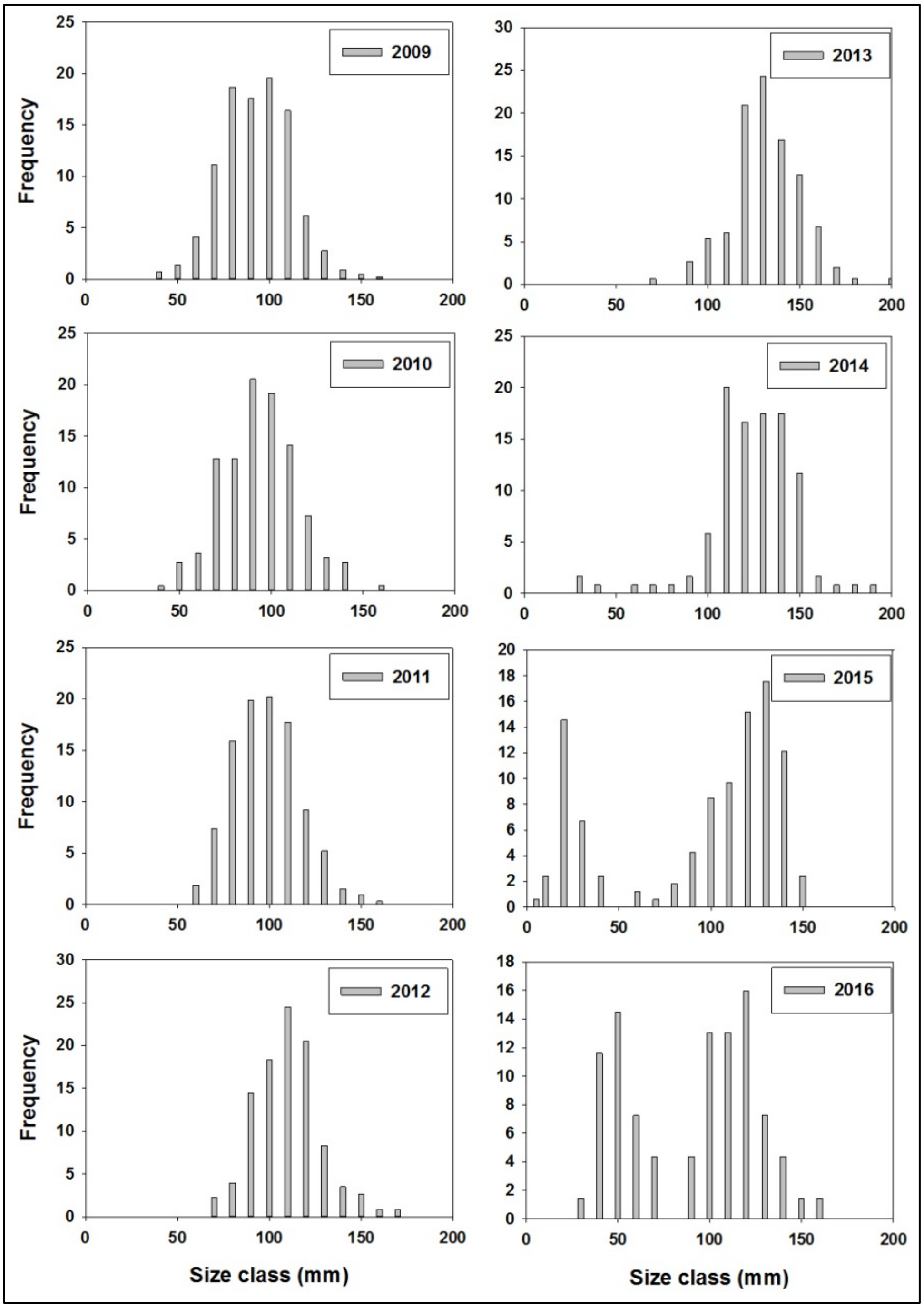

Figure 4.3.9. Annual size class frequency distributions of *Pisaster* across all plots at Starfish Point. Size classes are binned by 10 mm increments. Size frequency patterns at Starfish Point are similar to those observed over the same timescale at the Point of the Arches, Sokol Point, and Taylor Point monitoring sites. Source: NPS.

Pisaster SSWD Prevalence and Water Temperature

At the outset of SSWD in June 2013, the highest incidence of diseased *Pisaster* (26%) was observed on the south Olympic coast, with smaller percentages of the *Pisaster* populations exhibiting disease symptoms on the mid and north coast (Figure 4.3.7). Subsequent monthly surveys focusing on the Starfish Point area showed a decline in SSWD incidence to below 10% of the population through the winter of 2013-2014, and the absence of SSWD through the summer of 2014 (Figure 4.3.10), when SSWD was at a high level in the Salish Sea (Eisenlord et al. 2016). In the winter of 2014-2015 SSWD reappeared on the Olympic coast at the highest incidence yet recorded (~65%) that sustained through the winter and spring, when SSWD was very rare in the Salish Sea. Olympic SSWD was present at moderate levels through the summer of 2015 and the winter of 2015-2016. While SSWD has been observed in both adults and juveniles at all four OLYM sites, it appears to be more prevalent in adults, with less than 4% of juveniles ever showing SSWD symptoms. These results are consistent with the observations of Menge et al. (2016) from the Oregon coast, who suggest juveniles may be less susceptible to SSWD due to their smaller surface area exposure to water-borne pathogens or possibly other physiological differences. Alternately, diseased juveniles may die faster, making their detection less likely.

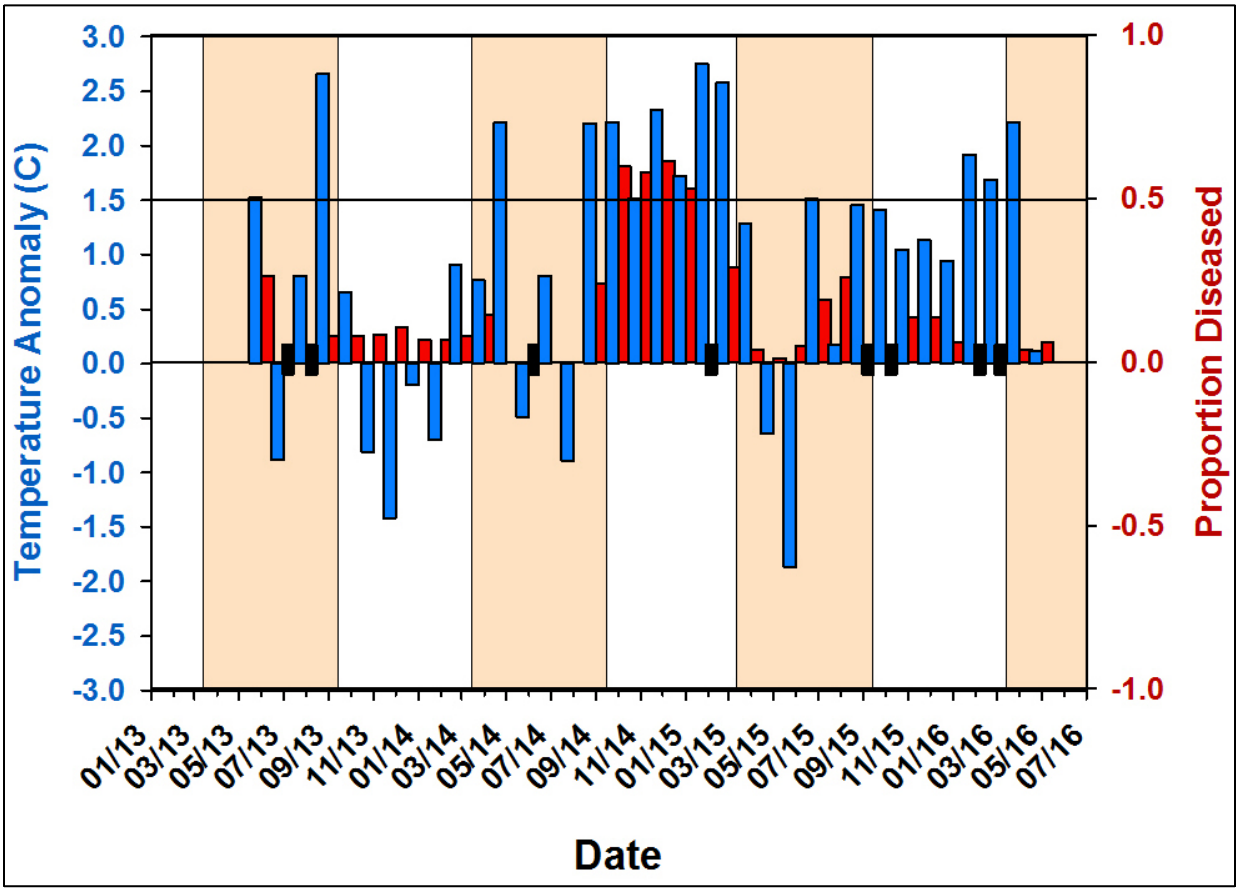

Figure 4.3.10. Temporal patterns of proportion of the Starfish Point *Pisaster* population displaying SSWD symptoms (red bars), and the monthly intertidal seawater temperature anomaly at Starfish Point (blue bars). Shaded areas represent the summer months. Black bars denote periods where SSWD surveys did not occur. Source: NPS.

Previous examples of SSWD-like symptoms have been associated with elevated water temperature (Eckert et al. 2000). We and others (Eisenlord et al. 2016) have hypothesized that the current SSWD event is also associated with elevated temperature. We observed a significant correlation (p=0.0126, Pearson correlation coefficient=0.465) between SSWD prevalence and the deviation of monthly average water temperature from the long-term average. Higher-than-average water temperatures and increased prevalence of SSWD coincided with the coastal arrival of the "blob" in September 2014 and the strong El Nino from summer 2015 to winter 2015/2016 (Figures 4.3.10). The blob was a persistent mass of anomalously warm water that developed in the offshore waters of the NE Pacific during the winter of 2013-2014 (Bond et al. 2015, Di Lorenzo and Mantua 2016). Similarly, the onset of SSWD in June 2013 coincided with elevated temperature.

The relationship between increased levels of SSWD and anomalously warm monthly water temperature is intriguing because there does not appear to be an absolute temperature threshold that instigates SSWD. Rather, SSWD appears to be facilitated by water that is warmer than normal for a particular time of year, suggesting that *Pisaster* is acclimated to seasonal temperatures.

We found a similar relationship between SSWD prevalence and the deviation of the monthly average temperature of rocky intertidal surfaces from the long-term average when exposed at low tide, a proxy for what *Pisaster* experiences under such circumstances. (p=0.001, Pearson correlation coefficient=0.585). However, water rather than aerially exposed temperature is likely the dominant influence on SSWD. The subtidal sunflower sea star (*Pycnopodia helianthoides*), a dominant predator not aerially exposed, has been devastated by SSWD to the point near functional extinction throughout most of its range (Montecino-Latorre et al. 2016). Similarly, *Pisaster* spends the majority of a tidal cycle submerged (Robles 2013). While summertime exposed temperatures can cause heat stress and mortality in *Pisaster* (Robles 2013), maximal exposed temperatures have not increased since the onset of SSWD (F=0.1147, p=0.7355).

Other Sea Star Species

Data on sea star species other than *Pisaster* are limited, as long-term monitoring plots were established primarily to characterize trends in *Pisaster* due to its outsized role in the OLYM rocky intertidal zone. However, two other species, the blood star (*Henricia* spp.) and the six-rayed star (*Leptasterias hexactis*) are regularly found in some of the OLYM monitoring plots. Overall, populations of these species have remained relatively stable, even after the onset of SSWD (Figure 4.3.11). SSWD has been observed occasionally in both species; however, the incidence rate has always been very low (<1%) and does not appear to have had a population-level effect. An exception is the *Leptasterias* population at Sokol Point, which has increased dramatically since the onset of SSWD in 2013. Menge (1972) observed that *Leptasterias* increases in abundance and size in the absence of *Pisaster*, which has a broadly overlapping diet with *Leptasterias*. It is possible that the decline in *Pisaster* at Sokol Point to the lowest level at our 4 monitoring sites (Figure 4.3.8) has released *Leptasterias* from competitive constraints, allowing the population to increase. Alternately, Sokol Point may have just experienced unusual settlement from a larvae-rich water mass.

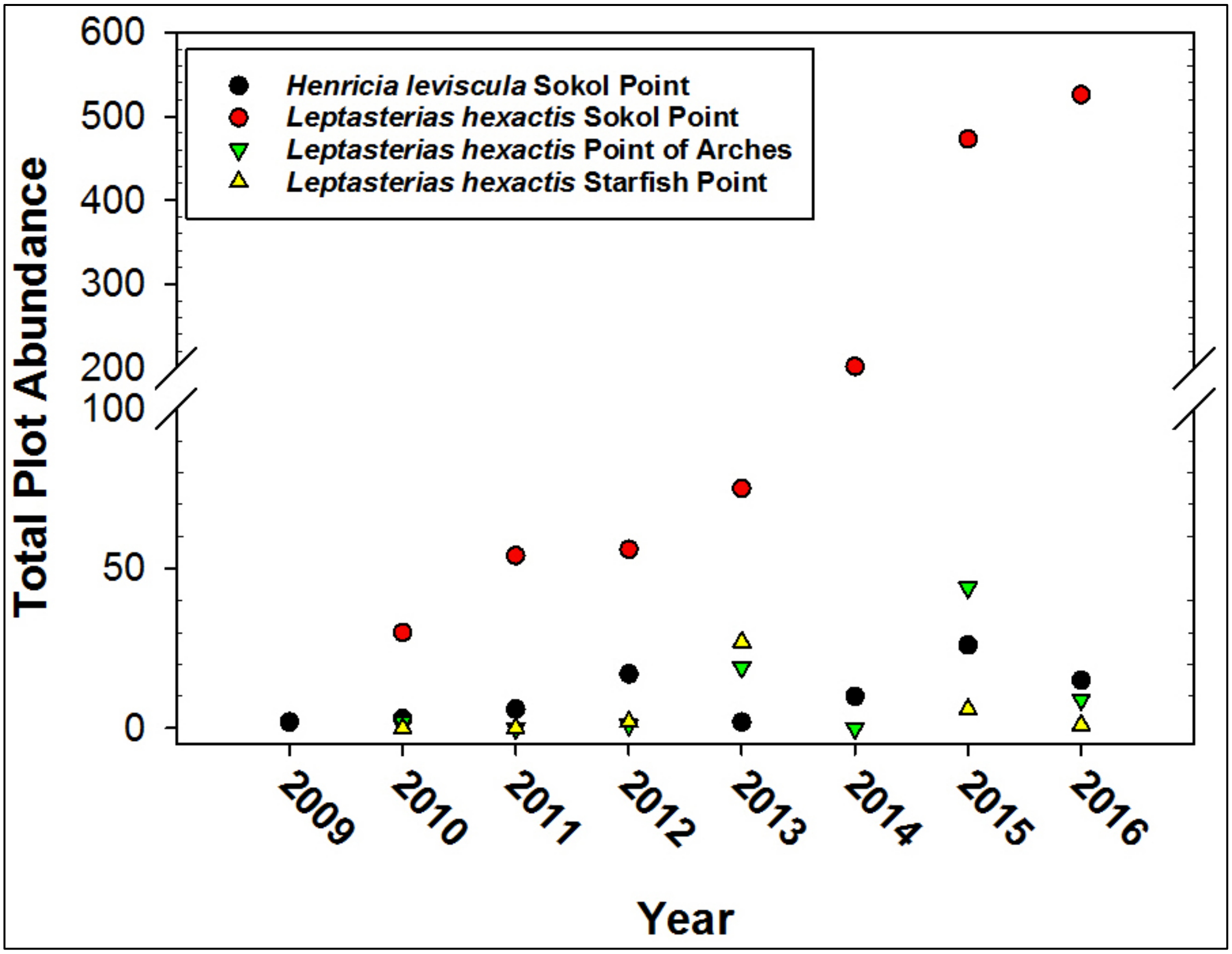

Figure 4.3.11. Abundance of blood sea stars (*Henricia* spp.) and six-rayed sea stars (*Leptasterias hexactis*) in permanent plots at Olympic National Park long-term monitoring sites. Only data from sites where species are found in most years are presented. Source: NPS.

The sunflower star, *Pycnopodia helianthoides*, is a keystone species of the lower intertidal and subtidal zone. This species does not occur in the higher elevation OLYM long-term monitoring plots. Prior to 2013, large adult *Pycnopodia* were commonly observed opportunistically throughout the lower intertidal zone of OLYM. Since 2013, no adult *Pycnopodia* have been observed. These observations are consistent with the collapse of *Pycnopodia* elsewhere in Washington (Montecino-Latorre et al. 2016). However, in the summers of 2015 and 2016, several newly recruited juvenile *Pycnopodia* were observed at Point of the Arches, suggesting that recruitment may potentially rebuild the *Pycnopodia* population.

All 11 sea star species documented in OLYM have displayed SSWD symptoms since 2013 within their respective ranges in California and the Salish Sea (Montecino-Latorre et al. 2016, http://www.eeb.ucsc.edu/pacificrockyintertidal/index.html). However, we have only observed SSWD in six species within OLYM (Table 4.3.7). This is likely due to a lack of sampling effort, as the five species undocumented for SSWD are relatively uncommon and all live in the lower intertidal zone outside of our monitoring plots. For all of the OLYM low intertidal species (Table 4.3.7), we do not have enough information to be able to assess the level of SSWD impacts on their populations. Of the 22 sea star species known to be afflicted by SSWD on the Pacific west coast, quantitative information on disease prevalence and population impacts is lacking for the vast majority of species (Montecino-Latorre et al. 2016).

Emerging Issues

SSWD Persistence

The dominant emergent issue for *Pisaster* on the Olympic coast is the persistence and level of impact of SSWD in *Pisaster* and other OLYM sea star populations. SSWD has been present from 2013 through early 2017, and it is unclear whether SSWD will remain a long-term stressor of sea star populations, or whether it will fade away with time. It is also unclear whether the gradual decline in *Pisaster* prior to 2013 is part of a longer-term population cycle, or symptomatic of environmental changes detrimental to *Pisaster*. Continued long-term monitoring should provide valuable insights into the potential environmental controls of SSWD, along with insights into the population dynamics and persistence of *Pisaster*.

Long-term Changes in Intertidal Community

Potential long-term changes in *Pisaster* populations could have vast consequences for the community structure of intertidal communities, as suggested by the increase in *Leptasterias* in one of our plots. The decrease or loss of this keystone species in the short- or long-term may affect vertical zonation patterns, species richness and diversity, as well as the relative abundance of competitors and prey species in the rocky intertidal. Analyses of community change in our long-term monitoring plots in conjunction with monitoring *Pisaster* dynamics would clarify the consequences of SSWD-induced changes in these communities. While we observed SSWD impacts in OLYM *Pisaster*, impacts in the Salish Sea and California have been more severe, including local extirpation. Our studies of community change, along with other on-going monitoring efforts on west coast that span a range of environmental conditions, will help to elucidate the environmental stressors influencing SSWD and its community impacts.

Marine Heatwaves

An emerging issue associated with SSWD and *Pisaster* population declines is the increased frequency and magnitude of marine heatwaves that potentially stress sea stars and lead to disease outbreaks. The blob, a persistent anomalously warm water mass that lasted from 2013 through 2014 before breaking apart as the 2015/2016 El Niño developed, re-formed in the winter of 2016. In both instances, the blob was formed by a lack of winter cooling of waters heated by normal summer conditions (Di Lorenzo and Mantua 2016). If ocean conditions that generate marine heatwaves become more common, outbreaks of diseases such as SSWD may become more common and more intense.

Information Needs/Gaps

Understanding Causes and Controls of SSWD

Managers need better information on the cause and controls of SSWD in *Pisaster*. While SSWD is thought to be caused by a densovirus that has been in the Pacific marine environment since at least the 1940s (Hewson et al. 2014), the clinical causes, environmental controls, and long-term ecological consequences of SSWD remain to be determined. Our understanding of the long-term dynamics of *Pisaster* populations is also inadequate. Prior to the onset of SSWD, declines in OLYM *Pisaster* populations were apparent, although the cause and potential cyclic nature of these changes in abundance are unknown. Continued monitoring of *Pisaster*, its thermal environment, and its prey community dynamics is needed to clarify potential long-term demographic patterns.

Understanding Pycnopodia Population Dynamics

There is also a dearth of information on *Pycnopodia* population dynamics in the lower intertidal and subtidal zone in OLYM. Data on population size and trends over time would provide valuable insights into the short- and long-term effects of SSWD on this species. Data on Olympic Peninsula *Pycnopodia* population dynamics could be compared with dynamics measured for this species within the Salish Sea. Fortunately, starting in 2015, the Olympic Coast National Marine Sanctuary and NOAA Northwest Fisheries Science Center began a set of subtidal dive transects to quantify the abundance of subtidal marine invertebrates and fish. This annual summertime sampling collects data on *Pycnopodia* abundance that should help to fill this information gap.

Resource Summary

Sea stars play a vital role in structuring intertidal communities on the Olympic coast. All eleven Olympic sea star species have displayed symptoms of SSWD, and *Pisaster* and *Pycnopodia* have been particularly impacted (Table 4.3.8). Fortunately, Olympic populations have not been affected as strongly as populations in the Salish Sea and in California. However, there has been a significant long-term decline in OLYM *Pisaster* populations, and no adult *Pycnopodia* have been observed on the OLYM coast since 2013. SSWD is still persistent in Olympic sea star populations. Successful recruitment of juvenile *Pisaster* is a hopeful sign of sea star resilience. Recruitment also appears to be occurring in *Pycnopodia*, holding promise for rebuilding of their populations.

Table 4.3.8. Resource condition summary for sea stars in Olympic National Park.

Species	Measure	Condition Status and Trend	Rationale
Pisaster population dynamics	Population trend	🔴	Significant population decline 2009-2016, including declines prior to SSWD outbreak in 2013
	Incidence of disease	🔴	Persistent SSWD in the system up to the present.
	Age class distribution	🟡	Two years successful recruitment post-SSWD, which may indicate some population recovery. Longer trends unknown.
Pycnopodia	Abundance	🔴	No observations of adults post SSWD, although signs of limited recruitment. Trends unknown.
Other sea star species	Population trends	🟡	Apparent increase or stable population in other affected species, but data are limited to two species and a few locations.
	Incidence of disease	🟡	Found in all 6 OLYM species and likely in others. Trends unknown.
Overall condition and trend of Sea stars		🔴⇩	Persistent SSWD since 2013 and ongoing declines in the adult *Pisaster* populations; Limited information on other species and successful recruitment in *Pisaster* leads to medium confidence in the assessment

4.3.3. Amphibians

Rebecca McCaffery, Michael J. Adams, Amanda M. Kissel, and Wendy J. Palen

Introduction

Amphibian declines are an unprecedented global conservation challenge that has been described as the beginning of a sixth mass extinction event in earth's history (Wake and Vredenburg 2008). One third of US amphibians are considered to be threatened at some level by the IUCN (Stuart et al. 2004). This is more than any other class of animal and recent information suggests this is a conservative estimate. One of the more alarming aspects of amphibian decline is the unexplained disappearance of amphibian populations from habitats that appear undisturbed. Indeed, documented declines in National Parks are one factor that galvanized concern for amphibians in the late 1980s (Corn et al. 1989, Corn 1994). Thus, the NPS plays an important role in this global issue both by protecting amphibians and by providing a 'natural laboratory' to help us understand different drivers of population and assemblage change in amphibians. Conversely, amphibians can be viewed as indicators of global environmental change that cannot be stopped at reserve boundaries. The issue of amphibian declines in protected areas has high public recognition.

Its size, wilderness status, location, elevation range, and diversity of amphibians all make OLYM an important reserve for protecting amphibians and for studying patterns of change in amphibians. The park has excellent baseline inventories of its unique amphibian fauna but has almost no information on trends. This fauna includes the most ancient living anuran in the world (coastal tailed frog, *Ascaphus truei*), regional endemics, and species that are known to be declining in portions of their range. Estimates of trends for nearly every species could be produced with relative ease using the inventories from a decade ago as a baseline. Moreover, this baseline and the characteristics of OLYM allow us to ask questions about major global threats to amphibians. In this assessment, we collate existing data on the distribution of amphibians in OLYM and summarize research results from a long-term study of a focal, high-elevation amphibian population that has implications for other populations and species.

Approach and Methods

We examined two key measures for amphibians in OLYM: (1) species diversity and distribution throughout the park, and (2) demography of a focal Cascades frog (*Rana cascadae*) population in the northern subalpine region of OLYM. The amphibian fauna of OLYM is discussed relative to known stressors affecting amphibian populations globally and within the context of protection within OLYM.

Species Diversity and Distribution

Extensive inventories of amphibians were conducted in OLYM from 1996 to 2002, which provided information on species occurrence (Bury and Adams 2000, Adams et al. 2001, Adams and Bury 2002, Adams et al. 2005, Galvan et al. 2005). Our assessment of species distribution relied predominantly on a previous report that compiled all amphibian records for the national parks in Washington State (Galvan et al. 2005). The document covered most of the inventory work that was completed 1984-1995.

Herein we report species richness and identify species present within the park. For each species we identify legal status, IUCN status, distribution, and conservation concerns. We also present a map of spatial distributions of all amphibian species known in the park, and describe the larger distribution of the species in the region.

Breeding male and female Cascades frogs. Photograph courtesy of Amanda Kissel.

The US Geological Survey's Amphibian Research and Monitoring Initiative (ARMI) and Simon Fraser University (SFU) support an intensive monitoring site for the Cascades frog in the northern subalpine region of OLYM called Seven Lakes Basin. These data have been collected from 2002 to the present. The data analyzed are from thirteen small ponds on the southern side of Clear Lake. Frogs were hand captured during thorough surveys of each pond during the breeding season. Two or three consecutive surveys (one per day) were conducted each year. Surveys occurred shortly after ice out, which is when frogs were breeding. Frogs were individually marked with passive integrated transponder (PIT) tags if their snout-to-vent length (a common measure of size in amphibians) was greater than 37mm. This size roughly coincides with the transition from juvenile to adult so the study focuses on adult frog population trends. We used a robust design analysis (Pollock 1982) linked to a suite of 14 climate covariates to estimate annual survival and population size (Kissel et al., *in prep*). We used AIC_c to rank models, and model averaged those that contained 95% of the weight (Burnham and Anderson 2002). The population estimate represents the number of individuals (males and females separately) in the population including individuals that are temporary emigrants (i.e., are not present at the pond in a given year but are part of the population that uses the pond).

Additionally, from 2013 to 2015, crews from SFU counted the number of egg masses in all survey ponds that had Cascades frog breeding (n=19) in the Seven Lakes Basin. Ponds were categorized into four hydroperiod classes (short, intermediate, perennial, and permanent), and monitored throughout the summer to track which ponds dried before tadpoles were able to develop, metamorphose, and disperse to other ponds. Short hydroperiods hold water for a few weeks to months, intermediate ponds hold water for a matter of months and may dry sooner in drought years, perennial ponds dry in only in extreme drought years, and permanent ponds do not dry.

Reference Conditions

Species Diversity and Distribution

Baseline conditions measured during the 1984 – 1995 inventory work (presented herein) can serve as reference conditions, but no effort is underway to determine the extent of changes since then.

Cascades Frog Demography

We do not have data on Cascades frog abundance and demography at our focal monitoring site prior to 2002. Therefore, the monitoring data we have collected and continue to collect represent a baseline understanding of how population size fluctuates over time in a dynamic, protected montane environment.

Results and Assessment

Species Diversity and Distribution

There are 13 amphibian species found in OLYM. This includes 5 frogs and 8 salamanders (Table 4.3.9). Two species, the western toad and Van Dyke's salamander, are listed as Candidate species for listing by Washington State. Four species are listed as State Monitor species in Washington State: coastal tailed frog, Cascades frog, Cope's giant salamander, and the Olympic torrent salamander. These are species for which the state wants to monitor status and distribution due to lack of information or restricted ranges. Twelve species have fairly wide distributions in the park, while one species, Van Dyke's salamander, has a more restricted distribution (Figures 4.3.12, 4.3.13, and 4.3.14). Within their respective ranges, nine species are considered stable, three species are thought to be decreasing, and the status of one species is unknown (Table 4.3.9).

Table 4.3.9. Management status, global trend, and distributional extent of amphibians present in Olympic National Park.

Species		**Management Status**				**Spatial Extent**	
Scientific Name	**Common Name**	**Washington**[1]	**Federal**[1]	**IUCN**[2]	**NatureServe**[3]	**Inside Park**	**Outside Park**
Anaxyrus boreas	Western toad	Candidate	NL	Least Concern (decreasing)	G4	Wide	Western US
Pseudacris regilla	Pacific treefrog	NL	NL	Least Concern (stable)	G5	Wide	PNW[4]
Ascaphus truei	coastal tailed frog	Monitor	NL	Least Concern (unknown)	G4	Wide	PNW
Rana aurora	red-legged frog	NL	NL	Least Concern (stable)	G4	Wide	PNW
Rana cascadae	Cascades frog	Monitor	NL	Near Threatened (decreasing)	G3G4	Wide	PNW
Ambystoma gracile	Northwestern salamander	NL	NL	Least Concern (stable)	G5	Wide	PNW
Ambystoma macrodactylum	long-toed salamander	NL	NL	Least Concern (stable)	G5	Wide	PNW
Dicamptodon copei	Cope's giant salamander	Monitor	NL	Least Concern (stable)	G3G4	Wide	Sub-PNW
Ensatina eschscholtzii	Ensatina	NL	NL	Least Concern (stable)	G5	Wide	Western US
Plethodon vandykei	Van Dyke's salamander	Candidate	NL	Least Concern (stable)	G3	Narrow	Sub-PNW
Plethodon vehiculum	western red-backed salamander	NL	NL	Least Concern (stable)	G5	Wide	PNW
Rhyacotriton olympicus	Olympic torrent salamander	Monitor	NL	Vulnerable (decreasing)	G3	Wide	Olympic Peninsula
Taricha granulosa	Rough-skinned newt	NL	NL	Least Concern (stable)	G5	Wide	PNW

[1] NL = not listed

[2] IUCN = International Union for Conservation of Nature

[3] G = Global; 3: Vulnerable; 4; Apparently Secure; 5: Secure

[4] PNW = Pacific Northwest

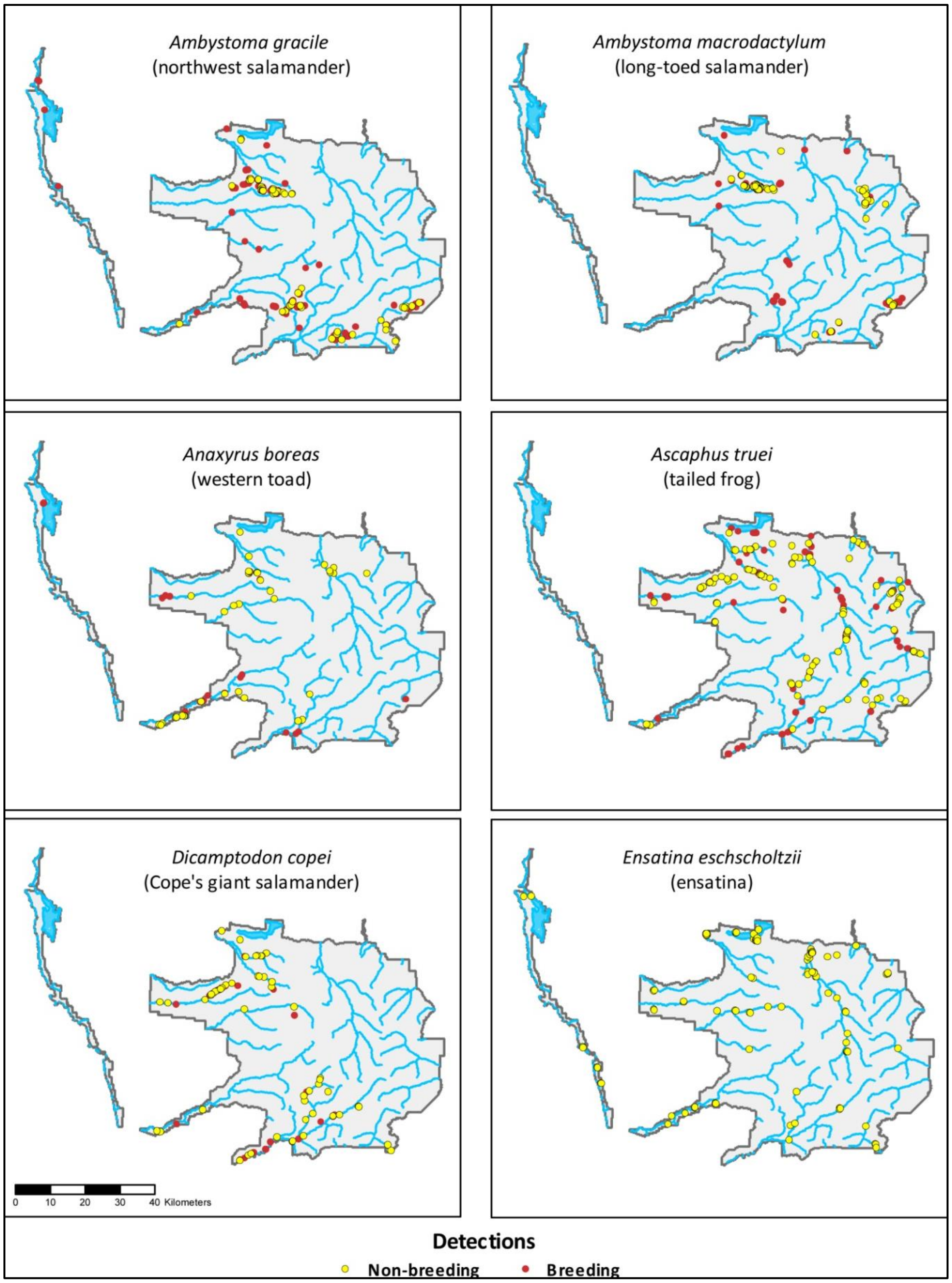

Figure 4.3.12. Distribution of amphibian species found in Olympic National Park during park-wide surveys; northwest salamander, long-toed salamander, western toad, tailed frog, Cope's giant salamander, and ensatina. Figure modified from maps prepared in Galvan et al. 2005.

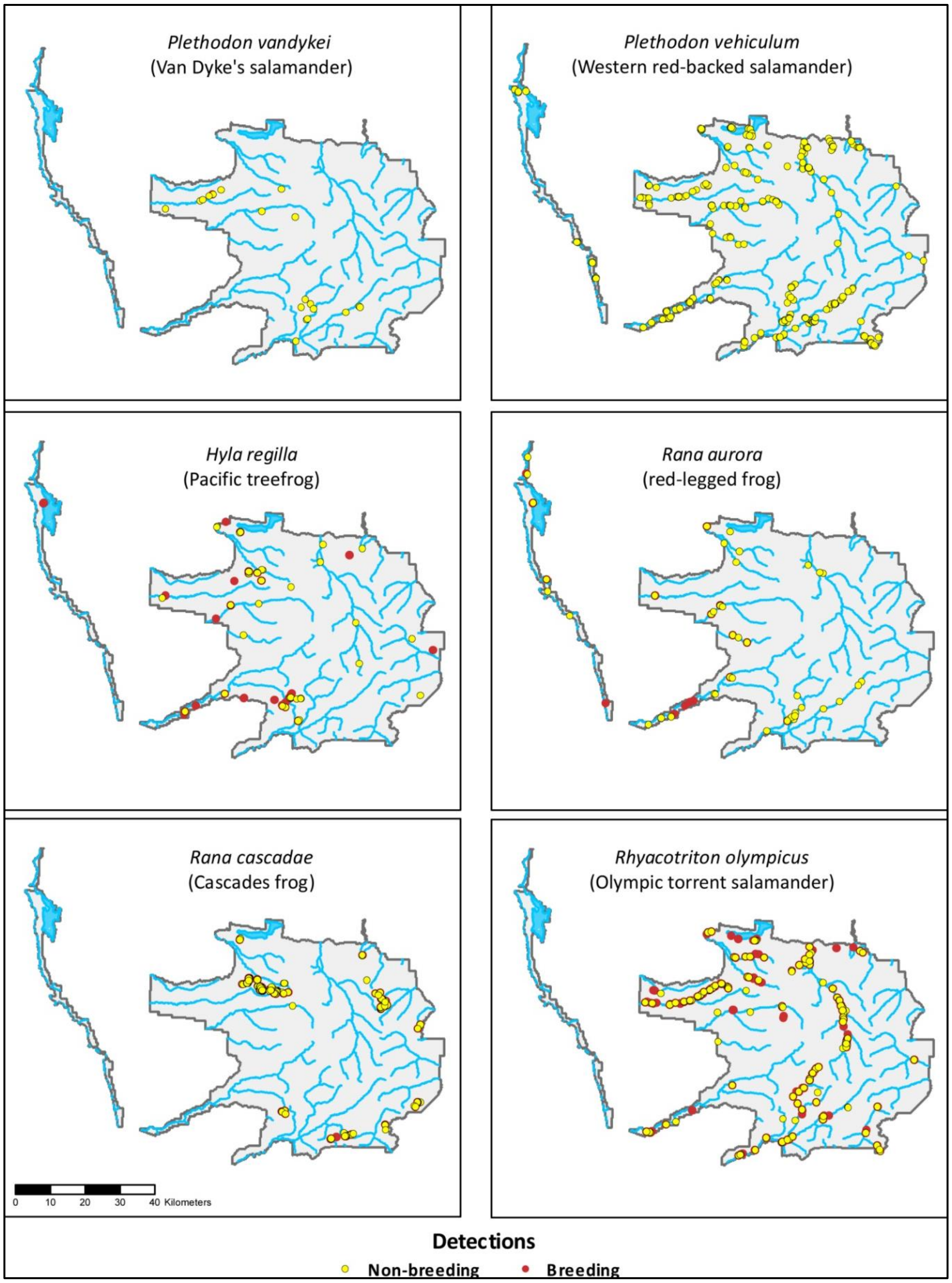

Figure 4.3.13. Distribution of amphibian species found in Olympic National Park during park-wide surveys;Van Dyke's salamander, western red-backed salamander, pacific treefrog, red-legged frog, Cascades frog, Olympic torrent salamander. Figure modified from maps prepared in Galvan et al. 2005.

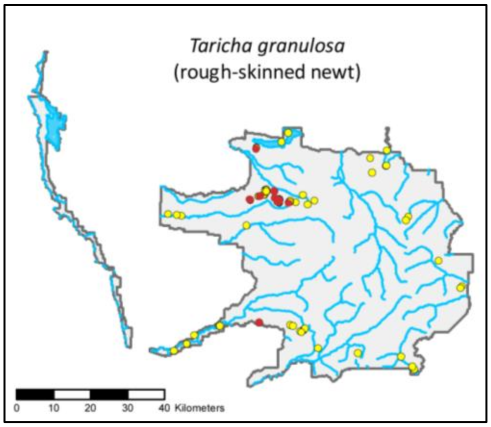

Figure 4.3.14. Distribution of amphibian species found in Olympic National Park during park-wide surveys; rough-skinned newt. Figure modified from maps prepared in Galvan et al. 2005.

Cascades Frog Demography

Annual estimates of abundance for the Cascades frog in the Seven Lakes Basin between 2002 and 2015 ranged from 3 (0.013 standard error [SE]) females and 35 (0.56 SE) males in 2004 to 58 (0.92 SE) females and 175 (2.14 SE) males in 2013 (Figure 4.3.15). Over this time period, geometric mean population growth rate was 1.02, indicating a stable population size on average. There was high interannual variation in population size, and overall estimates suggest a small population size (Figure 4.3.15), however estimates of female population size may be low in years when sampling took place late (i.e. after egg masses were laid) and females may have migrated from the sampled ponds, and egg mass counts from 2013 to 2015 were much higher (Figure 4.3.16). Mean annual female survival was 0.74 (0.05 standard deviation [SD]), and male survival was 0.81 (0.04 SD). The top two climate variables correlated with adult survival were winter length and precipitation in the driest quarter (i.e., summer precipitation, Kissel et al., *in prep*). Both variables were positively correlated, such that increases in winter length and summer precipitation were associated with increases in survival.

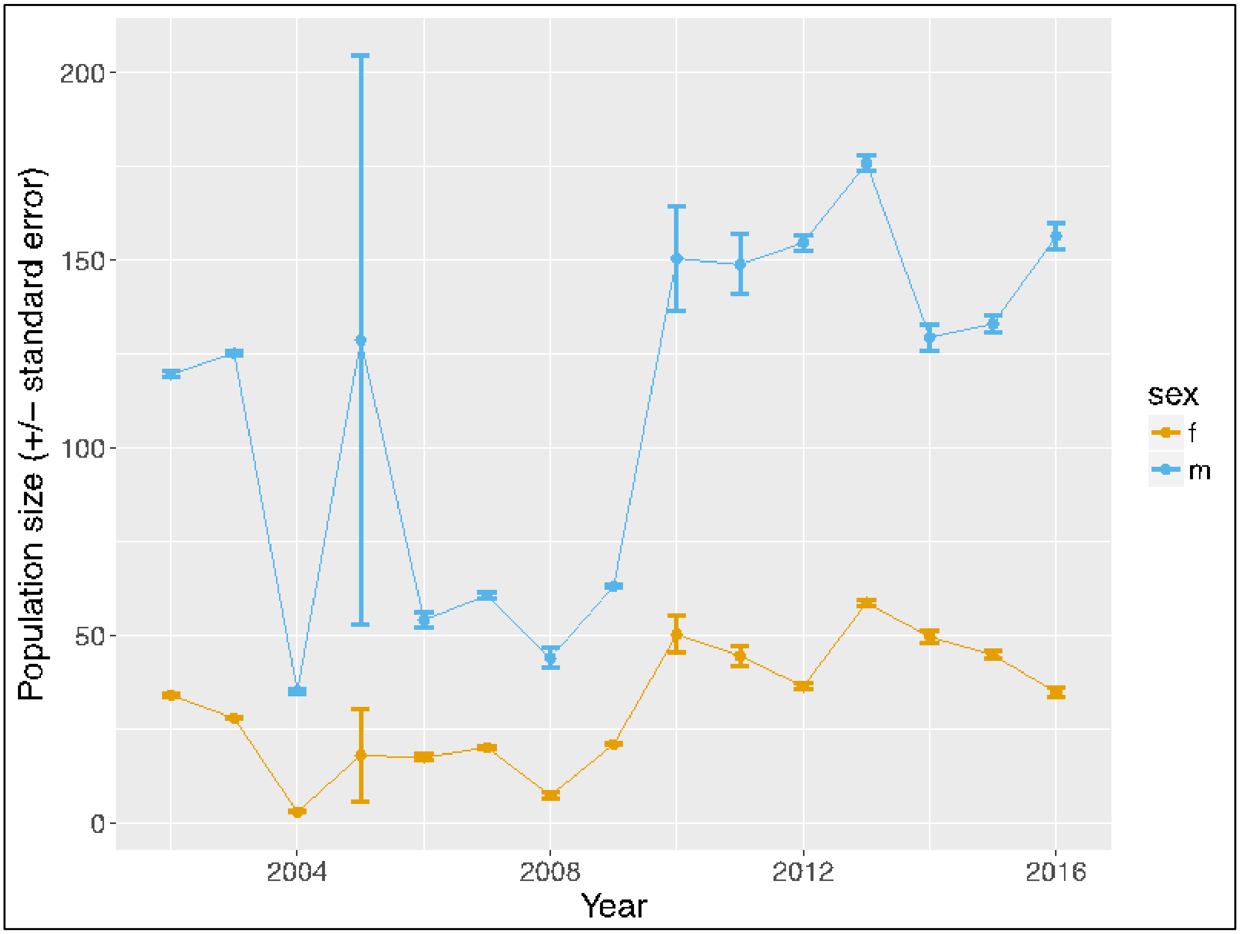

Figure 4.3.15. Population estimate for Cascades frog in the Seven Lakes Basin from 2002-2015. Female population size is in orange, and male population size is in blue.

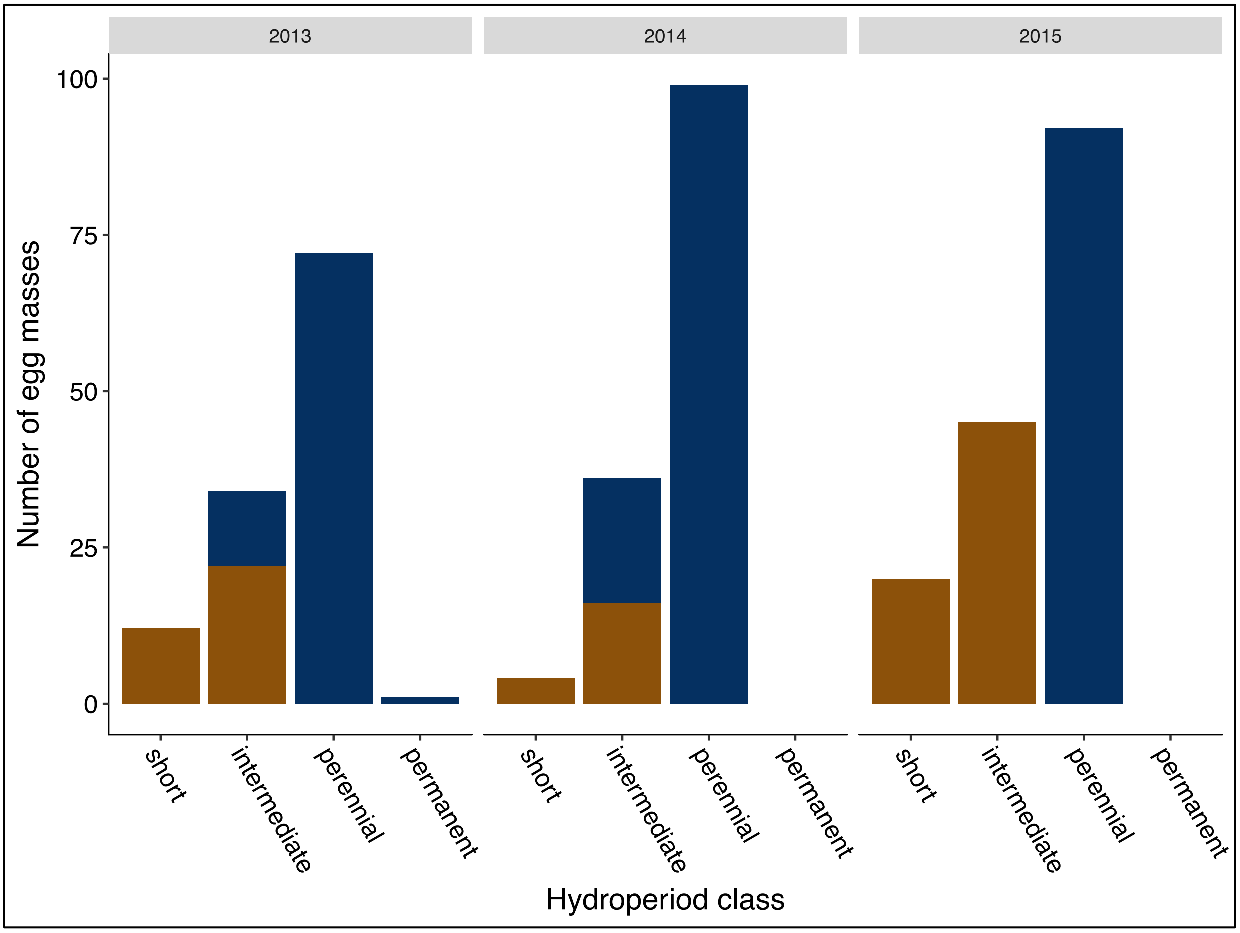

Figure 4.3.16. The portion of reproductive effort lost (expressed as number of egg masses) due to pond drying (orange bars) compared to the total number of egg masses laid in each hydroperiod class (blue bars) from 2013 to 2015.

The percentage of reproductive effort lost as a result of pond drying was 29% in 2013, 14% in 2014 and 41% in 2015 (an extreme drought year). In all three years, all eggs laid in short hydroperiod wetlands (i.e., wetlands that typically hold water for a matter of weeks and dry annually) desiccated before metamorphosis occurred, and in 2015, all eggs laid in intermediate hydroperiod ponds (i.e., seasonal wetlands that dried each year, but held water later than the "short hydroperiod" wetlands) were lost due to pond drying as well (Figure 4.3.16).

Uses and Limitations

Inventories of all species conducted in the 1980s and 1990s provided an important baseline for amphibian distribution throughout the park. However, they do not allow us to determine trends in distribution or occupancy for any of these species. Repeat inventories, particularly of key habitats or species, would be necessary to better understand how distribution patterns and occupancy may be changing over time.

Focal monitoring of the Cascades frog in Seven Lakes Basin showcases the interannual variability that can be seen in temperate, pond-breeding amphibian populations over time, and provides a context for understanding high-elevation pond breeding amphibian populations in the park. However, the study is limited to one species in one population, and we cannot be sure how these results may extend to other populations and species in OLYM. Although we found relationships between survival and climate variables in Seven Lakes Basin (Kissel et al., *in prep*), there may be other, unknown drivers of population variability affecting this and other populations.

Emerging Issues

Introduced Salmonids

Salmonids have been widely introduced to formerly fishless mountain lakes and are known to reduce or displace some species of amphibians (Adams et al. 2001, Pilliod and Peterson 2001, Knapp 2005). Species such as the northwestern salamander and long-toed salamander that occupy and rely on permanent lakes to complete their life history are particularly vulnerable (Tyler et al. 1998, Hoffman et al. 2004, Pilliod et al. 2010). Western toads and rough-skinned newts are exceptions that seem to coexist relatively well with fish (Welsh et al. 2006, Pilliod et al. 2010): these two species are noted for high levels of toxins in their skins. In NOCA mountain lakes, introduced trout have been documented as negatively affecting the abundances and distributions of native biota including amphibians (Hoffman et al. 1996, Liss et al. 1998, Tyler et al. 1998). These negative effects are most often related to trout population density and lake productivity (Tyler et al. 1998), and thus are not consistent across the landscape. Adams et al. (2001) found that Cascades frog and long-toed salamander were more likely to be found at ponds and lakes without fish (brook trout) but did not find a negative relationship between northwestern salamanders and fish. OLYM has relatively high numbers of fishless ponds and lakes compared to the rest of the region, which may provide a refuge for amphibians from introduced fish. However, research suggests that climate change might decrease the relative availability of small ponds that currently provide a refuge from fish (Lee et al. 2015).

Climate Change

The detailed data collected in the upper Sol Duc watershed of OLYM over the past ~15 years suggest that Cascades frog population sizes in the area are highly variable. Higher survival is correlated with longer winters and increases in summer precipitation, such that by the 2080s (A1B climate emissions scenario), the population is projected to no longer be stable as a result of shorter winters, drier summers, and increased frequency of pond drying (Kissel et al., *in prep*). The persistence of many wetlands in OLYM is projected to decline by large magnitudes, especially in short and intermediate hydroperiod wetlands (Lee et al. 2015) that hold 46% of Cascades frog reproductive effort (Kissel et al., *in prep*). The combination of the historic introduction of non-native trout into most large lakes in OLYM and the prediction that many smaller wetlands will experience much more frequent drying in the next century lead to the expectation that the status of many amphibian species within the park are at very high risk despite their current stability (Ryan et al. 2014).Overall, larval mortality due to pond drying is expected to increase 13% on average by the 2080s (A1B climate emissions scenario), which could tip the population from stable to declining (Kissel et al., *in prep*). While this research has been focused on a single population, the patterns seen in Seven Lakes Basin may portend challenges to pond-breeding amphibians throughout OLYM.

Disease

Most of the more enigmatic amphibian declines (i.e., mysterious declines in protected areas) worldwide seem to be explained by a disease called chytridiomycosis and its interactions with other stressors. Chytridiomycosis is caused by the fungal pathogens *Batrachochytrium dendrobatidis* (Bd) and *B. salamandrivorans* (Bsal). Bd and Bsal are the only chytrids known to specialize on amphibians. Susceptibility to both pathogens is highly variable and not well understood.

Most current information on resistance comes from studies of Bd. Bd was discovered in 1999 (Longcore et al. 1999). There are seven known strains of Bd and virulence varies. It kills amphibians by dehydration and is said to be the greatest single threat to biodiversity of any wildlife pathogen due to the number of species it affects and its ability to drive populations and species to extinction. Peptides produced by immune response and by bacteria that live on the skin of amphibians play a role in resistance (Woodhams et al. 2007, Lam et al. 2010). Environmental factors like temperature also play a role (Woodhams et al. 2003, Spitzen-van der Sluijs et al. 2014). Bd is present in many pond-breeding populations in the Pacific Northwest without clear effect. The pathogen may be having low-level effects that are not as obvious as the waves of decline and extinction seen in other parts of the world. Alternatively, declines in the Pacific Northwest may have already occurred and we now have relatively resistant populations. It is also possible that severe declines occur during particular environmental conditions that happen intermittently; however, the pathogen may not be a problem for some species or areas.

Bsal was discovered when it was found to be responsible for salamander die-offs in Europe (Martel et al. 2013), and it has not yet been found in North America. The only species in OLYM that is known to be affected by Bsal is the rough-skinned newt, for which it is highly lethal, but only a handful of North American species have been tested. The Pacific Coast is considered high risk for Bsal invasion and effects (Richgels et al. 2016). Recently, researchers have discovered Bsal on a frog species used in animal trade (Nyugen et al, *in press*), suggesting that the Bsal threat should not be limited to salamanders, and reinforcing concerns over the role of amphibian trade in spreading this disease.

Information Needs/Gaps

New Surveys of Species Occurrence

Comprehensive inventories for all amphibians present in OLYM have been completed. Most amphibians in OLYM are in the IUCN Least Concern category meaning there is not information to suggest they be Red-Listed at some level. There is not typically quantitative information on trends in species that can be used in the IUCN assessment, and broadly distributed species are less likely to be Red-Listed. A recent study of U.S. amphibians showed that IUCN Least Concern amphibians were disappearing from 2.7% of the ponds and other places where they occur each year and the trend was worse on NPS than on other types of land (Adams et al. 2013). While this is a substantial rate of loss, it would be difficult to notice in broadly distributed species without periodically assessing the status and trends of these species. Given the fact that more than a decade has passed since a thorough inventory of amphibians in OLYM was conducted, a survey designed to detect changes in occurrence patterns is an important need.

Effects of Climate Change on Amphibians in OLYM

Ongoing monitoring and research in Seven Lakes Basin suggests that high-elevation pond-breeding amphibian populations will be threatened by the shorter winters, warmer summers, and increased rates of pond drying predicted with climate change in OLYM (Kissel et al., *in prep*). It is less clear how well the predictions for this Cascades frog population may be generalizable to other populations and species throughout OLYM. Climate projections suggest that rates of pond drying and loss will increase (Lee et al. 2015), and efforts to better understand how pond loss will affect amphibian occupancy across OLYM are underway (SFU, unpublished data). It is unknown how climate change might impact stream-breeding amphibians in OLYM.

Effects of Non-native Fish on Amphibians in OLYM

While OLYM has less non-native fish presence in high mountain lakes than other parts of the region, climate change could make fish-impacted lakes (i.e., permanent waterbodies) more critical to amphibian persistence in these ecosystems (Ryan et al. 2014). Comprehensive surveys of fish presence and amphibian occupancy in relation to climate change would help clarify which amphibian populations in the park are most vulnerable to climate change. These types of studies could help prioritize potential areas for non-native fish removal.

Resource Summary

We find that the amphibian community in OLYM is a resource of moderate concern and currently unknown trend (Table 4.3.10). Most species found in OLYM are considered regionally stable and species inventories have been completed, showing a wide distribution of most species. However, we have no information on changes in occupancy or abundance except one population. Focal studies of the Cascades frog in the Seven Lakes Basin show high interannual population size, but no systematic increase or decrease in population size over time. High variation in population size warrants concern, especially with climate change. Ongoing threats to amphibians in OLYM include climate change, non-native fishes, and disease. These all warrant continued attention, monitoring, and research.

Table 4.3.10. Resource condition summary for amphibians in Olympic National Park.

Resource/Measure	Condition Status and trend	Rationale
Species occurrence	🟢	Species inventories have been completed and most species are considered regionally stable, though status in the park is unknown
Trends in focal Cascades frog population	🟡	Population generally stable, but highly variable and could be subject to change in the future, especially in relation to changing climate. Demography limited to a single population.
Overall condition and trend of Amphibian community	◌	Past amphibian inventories point to wide distributions of park species, but status of all species is unknown and occupancy of all species is due to be revisited. Focal study of a Cascades frog population shows a stable but variable population size, warranting moderate concern. Ongoing threats include climate change, non-native fishes, and disease.

4.4. What species or ecosystem functions are threatened by non-native or invasive species?

Non-native invasive species present one of the most persistent threats to global biodiversity (Vitousek et al. 1997, Wilcove et al. 1998). These species can have deleterious effects on specific species of conservation concern (Wilcove et al. 1998, Clavero and García-Berthou 2005), as well as broader communities and ecosystems (Gurevitch and Padilla 2004), leading to biotic homogenization over space and time, species extinction, and the altering of evolutionary trajectories of native species (Mooney and Cleland 2001). Moreover, system stressors like climate change can influence or change how invasive species spread or interact with their environments and the response of native species (Hellmann et al. 2008). In this assessment, we examined the impact of three invasive species or groups of species: (1) barred owls (Chapter 4.4.1); (2) balsam woolly adelgid (Chapter 4.4.2); and (3) non-native invasive plants (Chapter 4.4.3). The barred owl directly competes with the federally threatened northern spotted owl; the balsam woolly adelgid affects the structure and potentially the composition of the subalpine fir community; and non-native invasive plants are a pervasive threat to many communities and ecosystems within the park.

Mount Olympus. Photograph courtesy of NPS.

4.4.1. Barred Owl (Northern Spotted Owl)

J. David Wiens, Scott Gremel, and Patricia Happe

Introduction

Conservation of the northern spotted owl (*Strix occidentalis caurina*), a federally threatened sub-species, has played a central role in the management of older forests in the Pacific Northwest (Thomas et al. 1990, USFWS 2011b, Dugger et al. 2016). Spotted owls are nocturnal, apex predators that use large home ranges and nest, roost, and forage in predominantly older forests. Initial conservation efforts for northern spotted owls began as early as 1973 in Oregon, but the sub-species was not federally listed as threatened until 1990 (USFWS 1990). The original listing was based on the owl's strong association with old conifer forest and declining trends in both old-growth forest habitat and owl populations (USFWS 1990). Management of spotted owls has been a complicated and controversial effort that has led to much federal land being reserved as habitat for the owl and associated wildlife (Noon and Franklin 2002). Despite these efforts, populations of spotted owls have continued to decline since population monitoring began in 1985 on demographic study areas located throughout the owl's geographic range. A recent meta-analysis of demographic rates of spotted owls indicated that competition with encroaching populations of barred owls (*S. varia*) was a primary cause of these population declines, but that forest conditions, local weather, and regional climate were also relevant sources of demographic variation to spotted owls (Forsman 2011, Dugger et al. 2016).

Competitive Interactions between Spotted Owls and Encroaching Barred Owls

Competition with rapidly increasing populations of barred owls has emerged as a prominent and complex threat to the long-term persistence of northern spotted owls. Once limited to forests of eastern North America, the barred owl is a medium-sized, ecologically similar *Strix* owl species whose newly extended geographic range now completely overlaps that of the northern spotted owl (Gutiérrez et al. 2007, Livezey 2009). The specific cause of this range expansion is largely unknown, but landscape changes facilitated by European settlement or historical changes in climate are factors that may have enabled barred owls to expand from eastern to western North America (Monahan and Hijmans 2007, Livezey 2009). Newly colonizing barred owls in western North America have been classified as native invaders—species that, under the influence of events such as climate change or human modifications to the landscape, have become invasive by expanding their populations into new areas (Valéry et al. 2009, Carey et al. 2012, Wiens et al. 2014). Initial colonization of different forest types in the Pacific Northwest was variable, but the barred owl now appears to outnumber spotted owls throughout the entire range of the threatened subspecies (Wiens et al. 2011, Dugger et al. 2016, Holm et al. 2016). Compared to spotted owls, barred owls are slightly larger, use smaller home ranges (Hamer et al. 2001, Wiens et al. 2014), have more diverse diets (Hamer et al. 2001, Wiens et al. 2014), and use a broader range of forest conditions for nesting (Pearson and Livezey 2003) and foraging (Hamer et al. 2007, Singleton 2015, Singleton et al. 2010, Wiens et al. 2014, Weisel 2015). More recently, pilot studies in California have demonstrated that experimental removal of barred owls from forests historically occupied by spotted owls can improve population trends of spotted owls (Diller et al. 2016, Dugger et al. 2016). Collectively, these studies provide strong evidence that competition with an increasing number of barred owls is causing accelerated population declines of spotted owls.

Northern Spotted Owls and Barred Owls in Olympic National Park

The Olympic Peninsula comprises one of eight long-term study areas where demographic rates of northern spotted owls have been monitored to assess the effectiveness of the Northwest Forest Plan in maintaining populations (Lint et al. 1999, Dugger et al. 2016). The Olympic demographic study area included 54 historical breeding territories of spotted owls monitored in OLYM, and 45 territories monitored by the USFS on the surrounding Olympic National Forest (ONF; Gremel 2014). Each territory included multiple activity centers (i.e., nests) that were used by individually color-marked spotted owls in one or more years of the study (Gremel 2014, Dugger et al. 2016). Populations of spotted owls have been monitored in OLYM consistently since 1992 and in ONF since 1987 (Anthony et al. 2006, Forsman 2011, Dugger et al. 2016).

The Olympic demographic study area is generally representative of habitat conditions on federal lands on the Olympic Peninsula, although the proportion of suitable forested habitat for spotted owls is greater in the National Park than in surrounding landscapes (Holthausen et al. 1995, Anthony et al. 2006). Olympic National Park, which includes the largest tract of late-seral and old-growth coniferous forest in the northern portion of the spotted owl's geographic range, historically comprised a large and regionally significant population of spotted owls (Gremel 2014). Historical estimates of population size indicate that approximately 230 territorial pairs of spotted owls occupied OLYM during the early 1990s (Seaman et al. 1996), prior to when barred owls became abundant in the region. Since that time, status and distribution of spotted owls in OLYM has changed dramatically (Gremel 2005, 2014). The most recent estimates of annual rates of population change from the Olympic demographic study area indicated that the local population was declining at rate of 3.9% per year, and that the rate of decline was increasing (Dugger et al. 2016).

The barred owl's range expansion into the Pacific Northwest has proceeded from north to south (Livezey 2009), so OLYM was likely populated by barred owls earlier than many other forests in the northern spotted owl's geographic range. Barred owls were first documented on the west side of the Olympic Peninsula in 1985 (Sharp 1989), and their invasion into OLYM is well documented (Gremel 2005, 2014). The effects of barred owls on territory occupancy, survival, reproduction, and population trend of spotted owls on the Olympic study area have been evaluated in previous, range-wide analyses of spotted owls (Forsman 2011, Dugger et al. 2016). These studies combined data on spotted owls and barred owls from both OLYM and ONF, however, so results are not specific to OLYM, and potential differences in owl populations between these land ownerships (and associated management strategies) have not been investigated.

Barred owl. Photograph courtesy of the NPS.

Assessment Goals and Objectives

The goal of our assessment was to better understand the colonization patterns of barred owls into OLYM and their effects on the occurrence and distribution of the federally threatened northern spotted owl. We also wanted to evaluate whether trends in occurrence of spotted owls and barred owls on the broader Olympic demographic study area were representative of conditions specific to OLYM. Thus, our specific objectives were to: (1) determine long-term (1992–2013) trends in occupancy dynamics of both spotted owls and barred owls, including colonization and local extinction rates, and (2) characterize the effect of barred owl presence on territory occupancy dynamics of spotted owls on OLYM versus ONF.

Approach and Methods

We used three key measures to assess the status of spotted owls and barred owls at OLYM: (1) long-term trend in the annual proportion of historical breeding territories of spotted owls that were used by spotted owls versus barred owls; (2) change in local extinction and colonization rates of spotted owls and barred owls over time; and (3) the consequence of invading barred owls on territory occupancy dynamics of spotted owls. We evaluated potential differences between OLYM and ONF for each of these measures to better understand park-specific changes in the occurrence of each owl species.

Trend in Territory Occupancy of Spotted Owls and Barred Owls

Herein, we refer to territory occupancy of spotted owls as the detection/non-detection of spotted owls at historical nesting territories monitored by field crews during the breeding season (1 March–31 August). Historical territories were defined as a landscape patch that represented the cumulative areas of use by a spotted owl, or pair of spotted owls, during the study period (Figure 4.4.1; Dugger et al. 2016:62 - 63). For spotted owls, data for our assessment included detection histories obtained from annual surveys conducted at 54 historical territories in OLYM, and 45 historical territories in ONF (Gremel 2005, 2014). For barred owls, which were not specifically surveyed, data for our assessment were based on ancillary detections of individuals recorded during surveys of spotted owls (Gremel 2005, Dugger et al. 2016). Barred owls have smaller spatial requirements and can achieve greater densities relative to spotted owls (Singleton et al. 2010, Wiens et al. 2014), so historical territories of spotted owls could include multiple individuals or pairs of territorial barred owls. As a consequence, our inferences to barred owls were more appropriately interpreted as *use* of historical territories of spotted owls, rather than *territory occupancy* by barred owls (see MacKenzie et al. 2003).

Recent analyses of spotted owls and barred owls in the Olympic demographic study area used a conditional, two-species occupancy model to determine co-occurrence dynamics of the two species (Dugger et al. 2016). Our intention was not to repeat these analyses and quantify competitive interactions between the species, but rather to use single-species analyses of site occupancy (MacKenzie et al. 2003) to evaluate trends in distribution and occurrence of each owl species separately (MacKenzie et al. 2003, Olson et al. 2005, Dugger et al. 2011, Sovern et al. 2014). In short, this method uses detections and non-detections of spotted owls or barred owls over repeated visits to sites (territories) to estimate the annual proportion of sites occupied (or used) by spotted (or barred) owls, as well as colonization and local extinction probabilities, while accounting for imperfect detection of owls present during field surveys. We used these analyses to assess trends in territory occupancy and use of each owl species in relation to land ownership (OLYM vs. ONF) and terrain conditions, in addition to examining changes in colonization and local extinction rates relative to historical reference conditions (see below). A previous analysis of barred owls in OLYM suggest that areas colonized first by barred owls included low-elevation patches of older forest with relatively flat terrain, often with a component of deciduous trees and riparian areas (Gremel 2005). We investigated the effects of terrain conditions, including mean elevation and terrain ruggedness, on annual colonization rates of barred owls by including these factors as site-specific covariates in our analyses. Terrain features were calculated from a 30-m resolution digital elevation model of the study area. We provide details of our analyses of territory occupancy dynamics for each owl species in Appendix J.

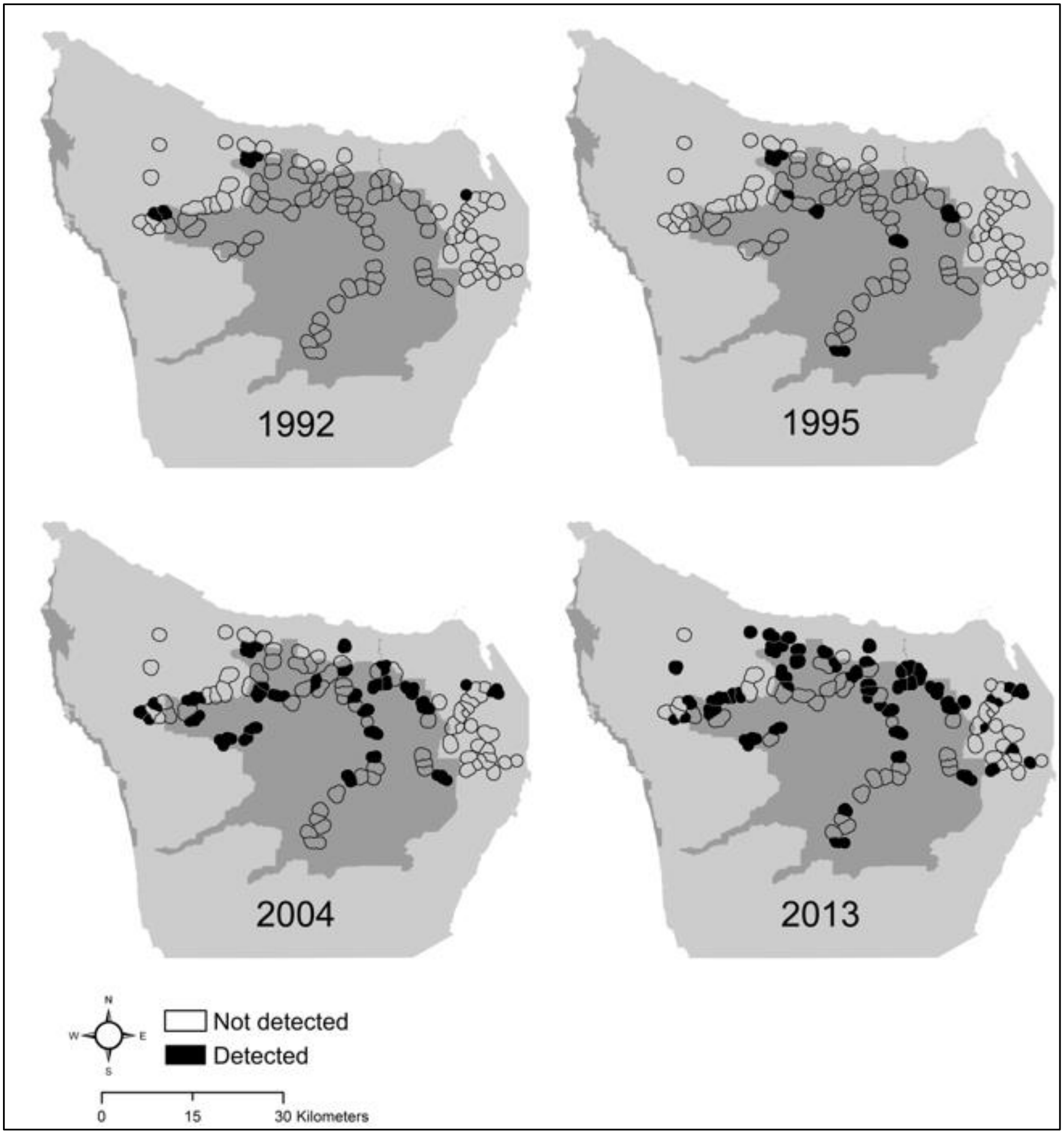

Figure 4.4.1. Territories of spotted owls (polygons) with detections of barred owls for selected years on the Olympic demographic study area, Washington. Olympic National Park is shown in darker gray. Source: NPS.

Change in Local Extinction and Colonization Rates of Spotted Owls and Barred Owls

Processes that determine annual changes in territory occupancy of spotted owls or barred owls are the colonization of a previously unoccupied site, and the local extinction of a previously occupied site. Here, colonization is defined as the annual probability of an unoccupied site becoming occupied by the focal owl species in the following year, whereas local extinction is the annual probability of an occupied site becoming unoccupied in the following year. We used results from our single-species occupancy analyses to estimate recent changes in annual colonization and local extinction of each owl species relative to historical reference conditions during 1992 – 1995 (see below).

Effects of Barred Owls on Spotted Owls

Previous studies have shown that barred owls can have a strong influence on territory occupancy, colonization, local extinction, and detectability of spotted owls (Olson et al. 2005, Dugger et al. 2011, Sovern et al. 2014). Similar to these studies, we used detection/non-detection information on barred owls during surveys of spotted owls to assess how detections of newly colonizing barred owls affected territory occupancy dynamics of spotted owls over time (Appendix J).

Reference Conditions

Reference conditions for the key measures we used for spotted owls and barred owls were based on historical conditions that existed when the first complete inventory and population estimate of spotted owls was completed in 1992–1995 (Seaman et al. 1996). We defined this time period as a state in the past corresponding with low occurrence of barred owls, prior to when their populations became more established (i.e., barred owls detected at <10% of historical territories of spotted owls; Gremel 2005, 2014, Dugger et al. 2016). The focal period of our assessment (1992–2013) was one of rapid range expansion and local population increases for barred owls in the Olympic study area (Figure 4.4.1), so we identified three time-periods in which to examine departures from historical reference conditions:

- 1992–1995 (barred owls detected at <10% of spotted owl territories)
- 1996–2004 (barred owls detected at <30% of spotted owl territories)
- 2005–2013 (barred owls detected at ≥30% of spotted owl territories)

Results and Assessment

Trend in Territory Occupancy Dynamics of Spotted Owls and Barred Owls

Spotted Owls

We calculated year- and species-specific estimates of territory occupancy, local extinction, and colonization to contrast trends in use of historical territories by spotted owls and barred owls (Figure 4.4.2). Consistent with previous studies (Dugger et al. 2016), we found a strong and negative annual time trend in occupancy of historical territories by spotted owls (Figure 4.4.2A). In addition, we found minor differences in estimates of territory occupancy between OLYM and ONF (greater on OLYM), particularly during the later years of the study. We found no evidence of a difference in local extinction rates between OLYM and ONF (Appendix J).The probability that a previously occupied site would become unoccupied in the following year (i.e., local extinction) showed a time trend in which local extinction increased steadily during the early years of the study, and then increased more slowly in later years (Figure 4.4.2B). The probability that a previously unoccupied territory would become re-occupied (i.e., colonization) declined sharply during the study, but annual colonization rates were 5–19% greater on OLYM relative to ONF (Figure 4.4.2C). Detectability of spotted owls during surveys varied within and among years, but was not strongly related to land ownership (Appendix J).

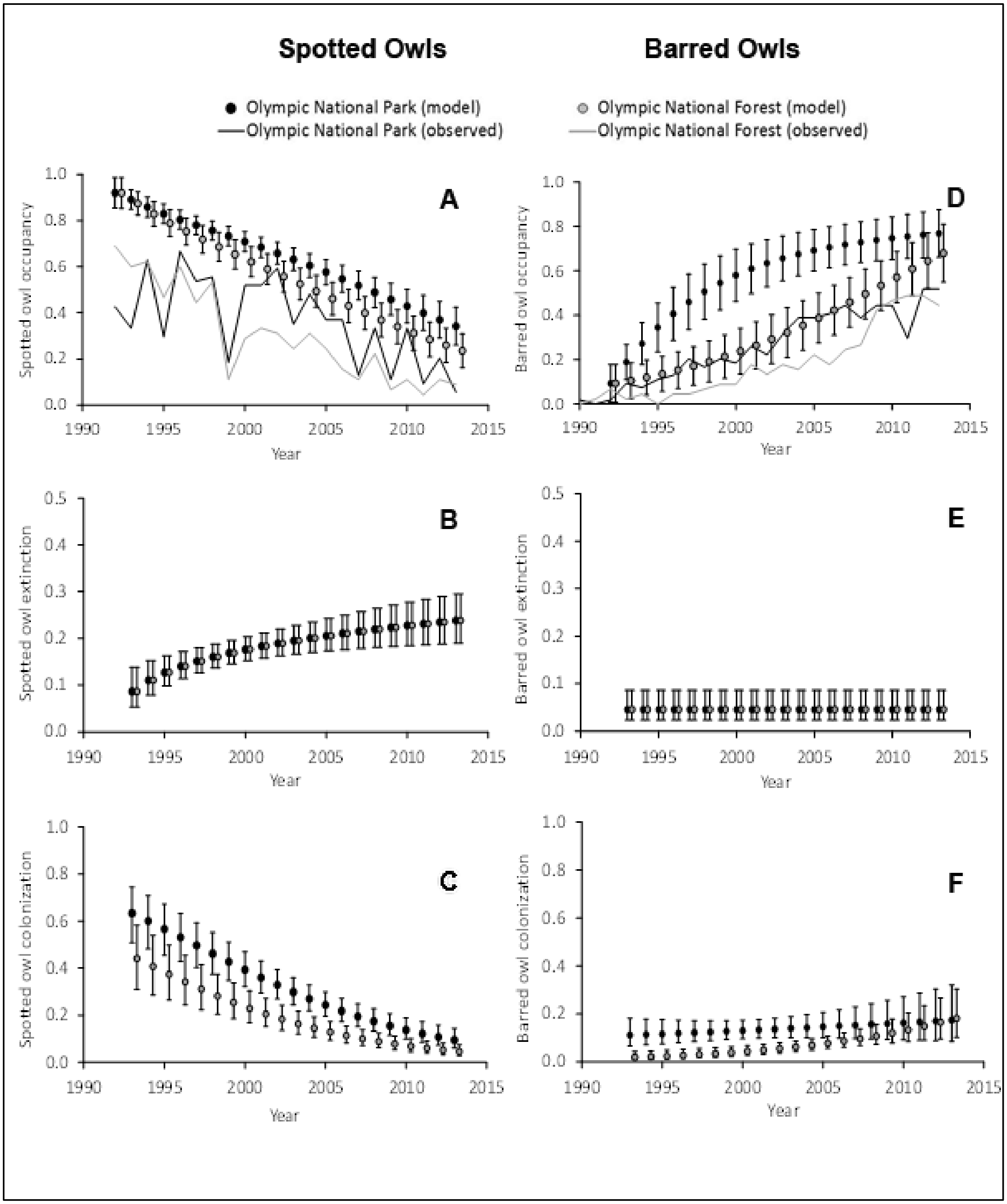

Figure 4.4.2. Estimated annual probabilities (± 95% confidence intervals) of territory occupancy (A, D), local extinction (B, E), and colonization (C, F) of northern spotted owls and barred owls on Olympic National Park and Olympic National Forest, Washington, 1992 – 2013. Estimates are from the best-supported single-species occupancy models developed for each owl species. Source: analysis of NPS data.

Barred Owls

Consistent with field observations, the estimated probability of use of historical spotted owl territories by barred owls in 1992 was relatively low (0.09, SE=0.04) and similar between OLYM and ONF (Figure 4.4.2D). In contrast to spotted owls, local extinction of barred owls remained relatively low and constant during the study period (Figures 4.4.2E). Colonization of historical spotted owl territories by barred owls gradually increased over time, but the annual rate of colonization by barred owls was greater on OLYM relative to ONF, especially during early- to mid-years of the study (Figure 4.4.2F). Consistent with our predictions, topographic roughness of spotted owl territories (i.e., standard deviation of elevation within the site) explained some of the spatial variability in local colonization rates of barred owls, with colonization being greater in relatively flat areas with low values of topographic roughness (Appendix J). Our analysis and results also showed that the probability of detecting barred owls that were present during surveys of spotted owls increased over time and varied between land ownerships (greater on ONF; Appendix J).

Effects of Barred Owls on Spotted Owls

Consistent with previous studies (Sovern et al. 2014, Dugger et al. 2016), the detection of barred owls during surveys of spotted owls was strongly associated with an increase in the likelihood that a site previously occupied by spotted owls would become unoccupied (Figure 4.4.3). We also found strong support for a negative effect of barred owls on re-colonization of historical territories by spotted owls (Appendix J), which paralleled findings reported by Dugger et al. (2016). We found no evidence that the effects of barred owls on occupancy dynamics of spotted owls varied between OLYM and ONF, as shown by a lack of support in our analysis for an interaction between the effects of land ownership and presence of barred owls on occupancy parameters (Appendix J). On each survey occasion, the detection of ≥1 barred owl decreased the probability of detecting spotted owls that were present by an average of 27% across all years.

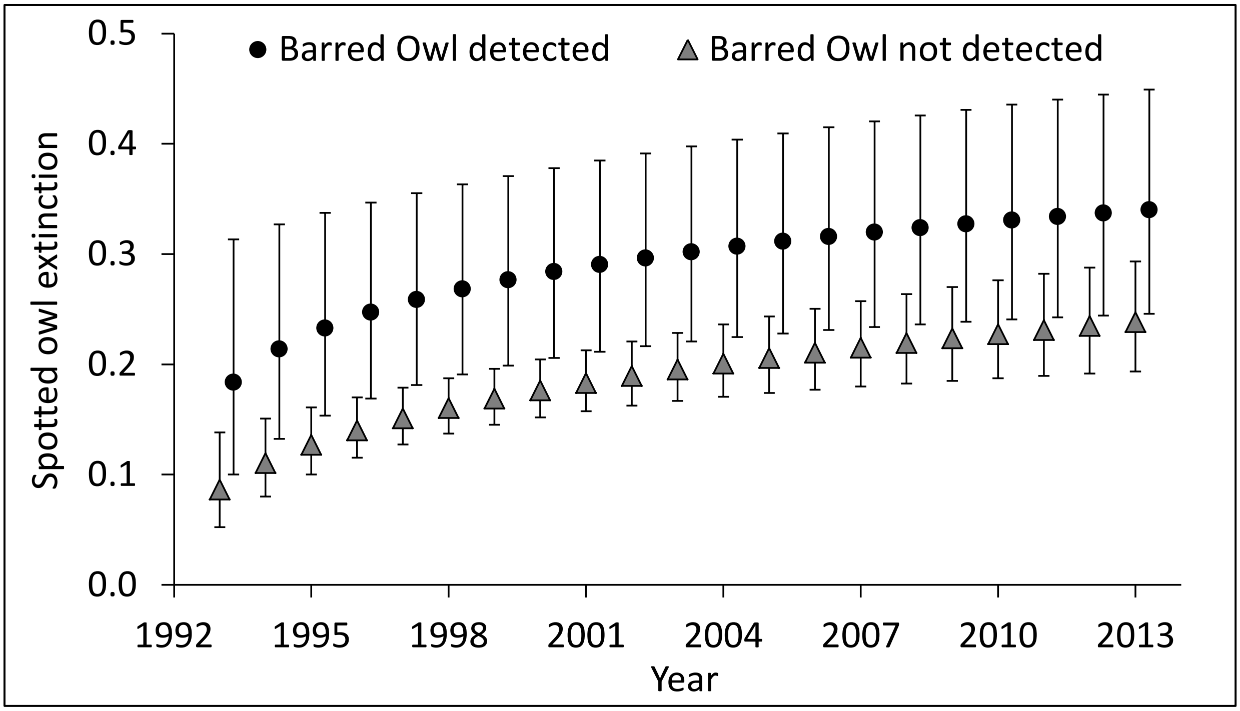

Figure 4.4.3. Year-specific estimates (±95% confidence intervals) that a territory occupied by spotted owls on Olympic National Park, Washington, would become unoccupied in the following year (extinction) when barred owls were detected or not detected during annual surveys of spotted owls, 1992–2013. Source: analysis of NPS data.

Change from Reference Conditions

Our analyses and results confirmed that annual estimates of territory occupancy for spotted owls were declining rapidly on both OLYM and ONF (Figure 4.4.2A). By 2004, (i.e., 10 years after historical reference conditions), the overall probability of territory occupancy by spotted owls on OLYM had declined by 34% (Table 4.4.1). During this same time period, we estimated an 86% increase in the use of spotted owl territories by barred owls (Figure 4.4.2D, Table 4.4.1). By the end of the study period (2013), territory occupancy rates of spotted owls on OLYM had declined by an average of 63% relative to reference conditions. This result was generally consistent with findings of Dugger et al. (2016:60), who reported that territory occupancy of spotted owls on the Olympic study area had declined by 75% between 1994 and 2013. However, by including the effect of land ownership in our assessment, we further showed that declines in occurrence of spotted owls were more extreme at ONF (76% decline) relative to OLYM (63% decline; Table 4.4.1). Conversely, use of spotted owl territories by barred owls increased dramatically from 0.09 in 1992 to 0.77 in 2013 on OLYM (an 88% increase), and from 0.09 to 0.68 on ONF (an 86% increase).

Table 4.4.1. Estimated probability of territory occupancy of northern spotted owls and use of historical spotted owl territories by barred owls in Olympic National Park and Olympic National Forest, Washington, during reference years of low occurrence of barred owls (1992, 1995) relative to years when barred owls became more established in the region (2004, 2013). We report the standard error of estimates in parentheses.

Owl Species	**Land Ownership**	**Estimated Probability of Territory Occupancy (Spotted Owl) or Use (Barred Owl)**			
		1992	**1995**	**2004**	**2013**
Spotted Owl	OLYM	0.920 (0.034)	0.829 (0.022)	0.605 (0.027)	0.342 (0.042)
	ONF	0.920 (0.034)	0.788 (0.029)	0.481 (0.035)	0.221 (0.035)
Barred Owl	OLYM	0.093 (0.043)	0.344 (0.056)	0.675 (0.050)	0.769 (0.055)
	ONF	0.093 (0.043)	0.135 (0.040)	0.354 (0.057)	0.680 (0.066)

Uses and Limitations

For long-lived territorial species like spotted owls and barred owls, trends in territory occupancy are a good indicator of long-term population trends (Tempel and Gutiérrez 2013). Our assessment clearly showed that territory occupancy of spotted owls was in sharp decline on OLYM, and that this was caused by annual increases in desertion of sites with concurrent reductions in re-colonization of sites that had become unoccupied. The rapid and dramatic increase in detections of barred owls was strongly related to annual increases in local extinction rates of spotted owls, and barred owls also had a negative effect on an observer's ability to detect spotted owls during standardized field surveys.

Information from our assessment can be used to help determine whether demographic analyses for the Olympic demographic study area (Dugger et al. 2016) are representative of owl populations specific to OLYM. We found that territory occupancy of spotted owls on OLYM was slightly greater than on the ONF, but that time trends in territory occupancy, local extinction, and colonization on both OLYM and ONF were generally consistent with time trends and parameter estimates reported for both these areas by Dugger et al. (2016). Information on the differential patterns of colonization by barred owls we identified within OLYM versus ONF can help guide future management decisions concerning spotted owls and barred owls, and identify priority research needs. For example, our assessment showed that barred owls initially colonized OLYM more rapidly than ONF, and that the subsequent rate of decline in territory occupancy of spotted owls was reduced on OLYM relative to ONF (Figure 4.4.2). Given the distinct differences between these two land ownerships in historical management regimes and amount of suitable owl habitat (Holthausen et al. 1995), this finding suggests that as barred owls invaded the region, habitat conditions in OLYM may have allowed spotted owls to persist longer on their territories than in the surrounding, more heavily managed landscapes. Regardless, our assessment suggested that with increasing numbers of barred owls, any advantage that may have existed in OLYM was eventually lost, as spotted owl occupancy rates reached similarly low levels on both ownerships by the end of the study period.

Monitoring protocols for spotted owls were originally designed to determine occupancy and reproduction rates at a sample of territories historically occupied by the species. As a consequence, barred owls were not systematically surveyed and the information available on their occurrence patterns is restricted to incidental observations during surveys of spotted owls, which likely underestimates their population status (Wiens et al. 2011, Dugger et al. 2016). We were able to partially account for this potential source of bias by accounting for imperfect detection probabilities of barred owls in the analytical framework we used. Nonetheless, barred owls have much smaller spatial requirements than spotted owls (Wiens et al. 2014), which implies that historical territories of spotted owls are likely to harbor large numbers of barred owls. Detection/non-detection data do not provide information on numbers of barred owls detected per spotted owl territory, and this was an important limitation of our study.

Similar to previous studies, results of our occupancy analyses indicated a greater occurrence of barred owls than uncorrected estimates based on incidental field observations during surveys of spotted owls (Hamer et al. 2007, Singleton et al. 2010, Wiens et al. 2011, 2014, Yackulic et al. 2014). Thus, a major advantage of our assessment was that it accounted for imperfect detection of barred owls during surveys of spotted owls in determining occupancy dynamics. A potential limitation of our single-species assessment of the effects of barred owls on spotted owls, however, was that it did not specifically account for imperfect detection of barred owls when considering the effects of dynamic territorial interactions between these species (e.g., Dugger et al. 2016). Nonetheless, our primary results and overall conclusions regarding the negative consequences of barred owls on spotted owls matched those of Dugger et al. (2016) regardless of the type of analytical approach used.

Emerging Issues

Potential Effects of Barred Owls on Other Sensitive Wildlife

In addition to impacts on spotted owls, changes in the abundance and distribution of a generalist apex predator like the barred owl can have potentially strong cascading effects on prey populations and food-web dynamics in the Pacific Northwest (see review by Holm et al. 2016). Prey species that could potentially be disrupted by barred owls at OLYM are indicated by recent dietary studies (Graham 2012). Diet samples collected from barred owls in OLYM included mostly forest mammals (71% of total prey numbers), with birds, amphibians, reptiles, insects, snails, and small fish comprising the remaining proportion of the diet (Table 4.4.2). Northern flying squirrels (*Glaucomys sabrinus*), shrews (*Sorex* spp.), and moles (family *Talpidae*) were the most common mammals in diets of barred owls, but flying squirrels had the greatest dietary contribution in terms of biomass (Graham 2012). Flying squirrels are considered a keystone species in older forests, and are an important prey species for northern spotted owls and many other native predators (Carey 1995, Smith and Green 2005). Given that northern flying squirrels are a primary contributor to dietary biomass of barred owls (Graham 2012), sharp increases in predation of flying squirrels are likely as barred owl populations continue to expand.

Table 4.4.2. Diet composition of barred owls on Olympic National Park, Washington. Data from Graham, 2012.

Taxonomic Category	Prey Species	% of Prey Numbers[1]	% of Prey Biomass[1]
Mammals	Shrews (*Sorex* spp.)	18.7	2.8
	Moles (Family Talpidae)	11.7	13.2
	Rabbits or hares (Family Leporidae)	0.5	3.1
	Northern flying squirrel (*Glaucomys sabrinus)*	15.5	34.0
	Douglas' squirrel (*Tamiasciurus douglasii)*	2.1	7.7
	Western spotted skunk (*Spilogale gracilis)*	0.5	5.3
	Pacific jumping mouse (*Zapus trinotatus)*	1.6	0.7
	Microtis spp.	7.9	4.6
	Unidentified vole or mouse *(Muridae* spp.)	1.6	0.8
	Ermine (*Mustela erminea)*	0.5	0.5
	Total	**71.7**	**78.1**
Birds	–	6.4	17.4
Amphibians	–	7.0	2.6
Frogs	*Rana* spp.	4.8	2.4
Salamanders	–	2.1	0.2
Reptiles	–	0.5	0.9
Insects	–	10.7	<0.1
Snails	–	3.2	0.5
Fish (small salmonids)	–	0.5	0.4

[1] Total number of prey items was 187 and total prey biomass was 11,445-g.

As generalist and opportunistic predators, barred owls capture a greater proportion of diurnal, terrestrial, and aquatic prey than northern spotted owls (Hamer et al. 2001, Forsman et al. 2004, Wiens et al. 2014). Barred owls have also reached densities in the Pacific Northwest that are far greater than historical populations of northern spotted owls (Wiens et al. 2011, 2014, 2016). These life history traits indicate that barred owls are not direct functional replacements of northern spotted owls in forested ecosystems of the Pacific Northwest, and that a wide range of prey species may be affected if they replace northern spotted owls (Holm et al. 2016). We hypothesize that expanding barred owl populations will result in increased predation pressure on traditional and naïve prey species within OLYM, and suggest that this is a highly relevant emergent issue that warrants further consideration.

Climate Change

Climate change can impact populations of spotted owls through changes in weather, forest cover, prey availability, and other ecological interactions. In a recent study, population growth of spotted owls was positively associated with wetter-than-normal conditions during the nesting season, which likely increased the size and availability of prey populations (Glenn et al. 2010). The rate of population growth and reproduction of spotted owls was also negatively associated with cold, wet winters and the number of hot summer days during the nesting season. This information suggests that projected future climate conditions of warmer winters and drier summers have the potential to negatively affect reproduction, recruitment, and consequently population growth rates for spotted owls (Glenn et al. 2010). How changes in climate may influence barred owls is currently unknown.

Small Population Size and Loss of Genetic Diversity

Loss of genetic diversity within a small population can decrease adaptive potential and cause inbreeding depression because of a higher probability that closely related individuals will mate. Genetic evidence indicates that loss of genetic variation and increased potential for inbreeding depression is greatest in the northern portions of the spotted owl's geographic range (Funk et al. 2010). This suggests decreased evolutionary potential of spotted owls and increased vulnerability to extinction as populations decline to small numbers in OLYM (Funk et al. 2010).

Disease and Pathogens

Disease exposure could be a secondary consequence of climate change, blood parasites, or interactions with barred owls. Lewicki et al. (2015) found that spotted owls had a greater diversity of parasitic *Haemoproteus* spp. than barred owls, in addition to a higher rate of infection. In addition, avian malaria (*Plasmodium* spp.) was found to be common in barred owls, but was only recently documented in spotted owls. Spotted owls are susceptible to West Nile virus and experience high rates of mortality when exposed (Courtney et al. 2004), however it is unknown what, if any, population-level impacts the disease has caused.

Information Needs/Gaps

Management Options for Barred Owls

Barred owls now occur in large enough numbers that the long-term persistence of spotted owls at OLYM may be in question without additional management intervention. Under at least some conditions, populations of spotted owls have responded positively to lethal removal of barred owls during pilot removal experiments (Diller et al. 2016, Dugger et al. 2016). These studies and others (Wiens et al. 2014, Yackulic et al. 2014) support the hypothesis that removal of barred owls may be able to slow or reverse spotted owl population declines on at least a localized scale. Nonetheless, we emphasize that the effectiveness and feasibility of barred owl removal for conservation of spotted owls in northern portion of the spotted owl's geographic range, where populations have declined the sharpest, is currently unknown. A large-scale research program on experimental removal of barred owls to benefit northern spotted owls was initiated by federal agencies in Oregon and Washington in 2015, with initial results expected in 2019 (Wiens et al. 2016).

Improved Monitoring Techniques for Spotted Owls and Barred Owls

Early on in the development of a monitoring program for spotted owls in OLYM, mark-recapture and random census-plot methods were both considered as options for population monitoring. The decision was made to use mark-recapture, which provides estimates of demographic rates (e.g., recruitment and survival). Precise estimates from mark-recapture studies require large samples of marked spotted owls, which was the case during the reference conditions of our assessment in the early 1990s (Lint et al. 1999). In recent years, however, the low number of individual spotted owls detected in OLYM suggests a need to transition to alternative monitoring techniques that may be better suited for sparse or rare species. Survey designs using automated bioacoustic recorders may be appropriate and effective, as audio recordings have already been used as a replacement for identifying individuals (Trifa et al. 2008) or for conducting occupancy surveys and analyses for sparsely distributed species (Zwart et al. 2014). Annual detection rates of barred owls during surveys of spotted owls at OLYM were very low during our study (7–25%; Appendix J), and the use of automated recorders for monitoring spotted owls could also increase the detection of barred owls, or other species of interest. Another advantage of automated recorders is that they are especially well-suited for remote or difficult to access areas like OLYM, as visits need only be made when deploying and picking up the recorders, or replacing batteries. Acoustic monitoring methods are also passive, which could reduce potential negative interactions between spotted owls and barred owls responding to broadcast calls used in current surveys. Use of automated detection methods in combination with the analytical framework of occupancy modeling may provide efficiencies for monitoring spotted owls as they become rarer and more difficult to detect, but inferences based on these studies are specific to sample sites and do not provide information of individual population dynamics that mark-recapture methods do. Efforts are currently underway to test the cost-effectiveness and efficacy of using automated field recorders to track territory occupancy dynamics of spotted owls and barred owls at OLYM (Gremel 2014).

Based on the following indicators, we conclude that the detrimental affect of barred owls on northern spotted owls warrants significant concern in OLYM and surrounding lands (Table 4.4.3). Further, we conclude with high confidence that the overall condition and trend of the northern spotted owl population is declining in OLYM and the adjoining ONF.

Resource Summary

- Long-term monitoring of northern spotted owls in OLYM showed that territory occupancy rates declined from 0.92 in 1992 to 0.34 in 2013 (a 63% decline from reference conditions). This result was consistent with findings from recent meta-analyses of spotted owls by Dugger et al. (2016), who found that territory occupancy of spotted owls had declined by an average of 75% between 1994 and 2013 on both OLYM and ONF combined.
- Declines in territory occupancy rates of spotted owls over a 22-yr study period (1992–2013) were more pronounced in ONF (76% decline) relative to OLYM (63% decline).
- Probability of barred owl use at breeding territories of spotted owls increased from 0.09 in 1992 to 0.77 in 2013 in OLYM (a 88% increase from reference conditions).
- Analyses specific to both OLYM and ONF showed a strong, negative relationship between detections of barred owls and territory occupancy of spotted owls.
- We found that barred owls initially colonized OLYM more rapidly than ONF, but that the rate of decline in territory occupancy of spotted owls was greater in ONF relative to OLYM.
- We found no evidence that the effect of barred owls on spotted owls varied between land ownerships (OLYM, ONF).

Table 4.4.3. Resource condition summary for northern spotted owls and barred owls in Olympic National Park.

Resource/Measure	Condition Status and Trend	Rationale
Trend in territory occupancy of spotted owls, 1992 – 2013.	🔴	Territory occupancy of spotted owls declined ~63% from historical reference conditions
Trend in use of historical spotted owl territories by colonizing barred owls, 1992 – 2013.	🔴	Use of spotted owl territories by barred owls increased ~88% from historical reference conditions
Change in colonization, and local extinction rates of spotted owls	🔴	Positive trend over time in local extinction rates of spotted owls, with negative trend in colonization of historical breeding territories
Effect of barred owls on territory occupancy dynamics of spotted owls	🔴	Strong association between detections of barred owls and local extinction of spotted owls; negative effect of barred owls on detection rates of spotted owls
Overall condition of Barred Owl	🔴	Use of historic spotted owl territories by colonizing barred owls has increased over time.
Overall condition and trend of Northern Spotted Owl	🔴 ⬇	Northern spotted owl territory occupancy has decreased. Local extinction rates by northern spotted owls have increased over time.

4.4.2. Balsam Woolly Adelgid (Subalpine Fir)

Karen Hutten and Andrea Woodward

Introduction

Balsam Woolly Adelgid (BWA) is an exotic insect that affects some species of true firs (*Abies* spp.), including subalpine fir and Pacific silver fir in OLYM. Balsam woolly adelgid was introduced from Europe in the early 1900s and was first documented in the southern Olympic Peninsula in 1955. It then spread to Hurricane Hill by 1970 and became widely dispersed by the 1990s (Hutten 2015; Figure 4.4.4). Infested fir trees are weakened and gradually killed over a number of years as feeding BWA alter tree cell growth, inhibit transport of water and nutrients, and create symptomatic swellings, typically at branch nodes. Control agents or methods have not been successful or feasible at the forest level. Balsam woolly adelgid has caused 40 to 70% mortality of subalpine fir over a 35 to 45 year period at sites in Washington and Oregon (Mitchell and Buffam 2001).

Figure 4.4.4. Tree mortality observed along a high-elevation ridge of Olympic National Park from fixed-wing aircraft, August 6, 2012. Photograph courtesy of Karen Hutten.

Subalpine fir on the Olympic Peninsula is limited to high elevations, generally in the northeast, where it pioneers harsh environments and creates conditions suitable for establishment of other species (Biek 2000). It is difficult to predict what the loss of subalpine fir would mean to subalpine forests in the northeastern region of OLYM, but the consequences to the entire ecosystem are expected to be substantial (Liebhold et al. 1995). Subalpine fir is considered to be a foundation species, which “defines 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). In addition to securing soil and creating conditions conducive to establishment of other tree species, subalpine fir provides habitat and food for a variety of large and small mammals and birds (Uchytil 1991).

Approach and Methods

We examined the following specific measures to assess the spread of BWA in OLYM and its effects on the health of subalpine fir ecosystems: (1) distribution of BWA-infested subalpine forest through time using aerial detection survey data, (2) extent of subalpine fir forest decline over time using satellite imagery, and (3) basal area of dead and unhealthy (dying) trees associated with change in spectral magnitude in satellite imagery.

Distribution of BWA

We first described the apparent broad-scale distribution and spread of BWA in Oregon and Washington through time as detected based on visible symptoms by ADS data. We reported the apparent spread of BWA across this region, placing the BWA infestation into a larger regional context and demonstrating the dispersal capability of the insect. We then summarized ADS data from OLYM over time to show the spread of BWA symptoms within the park from 1970 – 2012.

Aerial detection survey data are the longest-term comprehensive description of the disturbance regime, and are collected jointly by the USFS and the Washington DNR. These data have been collected annually since 1949 and describe the location of symptoms of forest insects, disease, weather-related damage, and other forest health stressors (Johnson and Wittwer 2008). The surveys are conducted from fixed-wing aircraft typically flying at 185 km/hr (115 mi/hr) and 500 m (1640 ft) above ground level. Observers evaluate a swath 2.4 km (1.5 mi) wide when surveying mountainous areas and sketch the location of disturbances on topographic maps. Assessment of disturbance agents is based on the occurrence of pest-specific damage ‘signatures’ consisting of foliage color, canopy texture, tree species identity, and season. In addition, observers estimate the severity of crown damage in three classes (high, moderate, and low) and in the case of mortality, the number of trees affected or trees per acre affected (McConnell et al. 2000). A subset of accessible polygons has been ground-truthed to verify BWA presence in association with the visual signature of BWA (Overhulser 2004, Hutten 2015).

Extent of Subalpine Fir Decline

To assess the annual extent of tree health decline associated with BWA in OLYM we used data from Landsat satellite images (30 m pixels) trained by ADS (Hutten 2015). Use of satellite data compensates for the tendency of ADS to overestimate area affected and to vary annually due to changes in observers. Satellite imagery was analyzed using the LandTrendr process (Kennedy et al. 2010), which involves tracking changes in spectral reflectance for individual pixels through a time series of Landsat images available since 1984. The method is able to characterize landscape change as a continual process operating at slow (e.g., regrowth, disease spread) or fast (e.g., fire) rates rather than simply a contrast between conditions at two points in time. Using the LandTrendr methodology (Kennedy et al. 2010) and overlays of ADS, we determined the spectral signature indicative of BWA activity (Hutten 2015). Areas described as having BWA infestation by ADS were associated with a gradual (long-duration, low-magnitude) change in spectral reflectance within a particular range of values of the spectral index Normalized Burn Ratio (NBR). The NBR is an index calculated from the difference in strength of near infrared versus shortwave infrared spectral bands in Landsat images and has been shown to effectively detect changes in forest health (Cohen et al. 2010). We quantified the trend and extent of BWA-associated forest decline using the appropriate range of the NBR derived from the LandTrendr analysis.

Basal Area Affected

The basal area of trees damaged by BWA was estimated by comparing the magnitude change in NBR for 50 18-m radius sample plots with a field assessment of trees in the plots. Health of individual trees and the presence of small BWA-symptomatic swellings can be seen in the field but not in satellite imagery. We related the basal area of dead and unhealthy trees occurring in a field plot to the change in spectral reflectance in Landsat imagery for the same location. A regression equation describes the relationship and can be used to remotely estimate the basal area of dead and unhealthy trees for an area based on change in spectral reflectance.

Reference Conditions

Because BWA is an exotic disturbance agent, the reference condition is found in subalpine fir stands that lack symptoms of BWA infestation. After introduction to the Olympic Peninsula in 1955, BWA spread in gradual to punctuated episodes, leaving as much as 90% of the subalpine fir forest infested when field work was completed in 2011 (Hutten 2015).

Results and Assessment

Distribution of BWA

Balsam woolly adelgid damage was detected by ADS in Oregon in 1953, spread to the Olympic Peninsula in 1955, and to OLYM by 1970 (Figure 4.4.5). Since then it has spread more widely at high elevations in the eastern Olympic Mountains (Figure 2.3.4) in areas modeled to have subalpine fir forests (Figure 4.4.6).

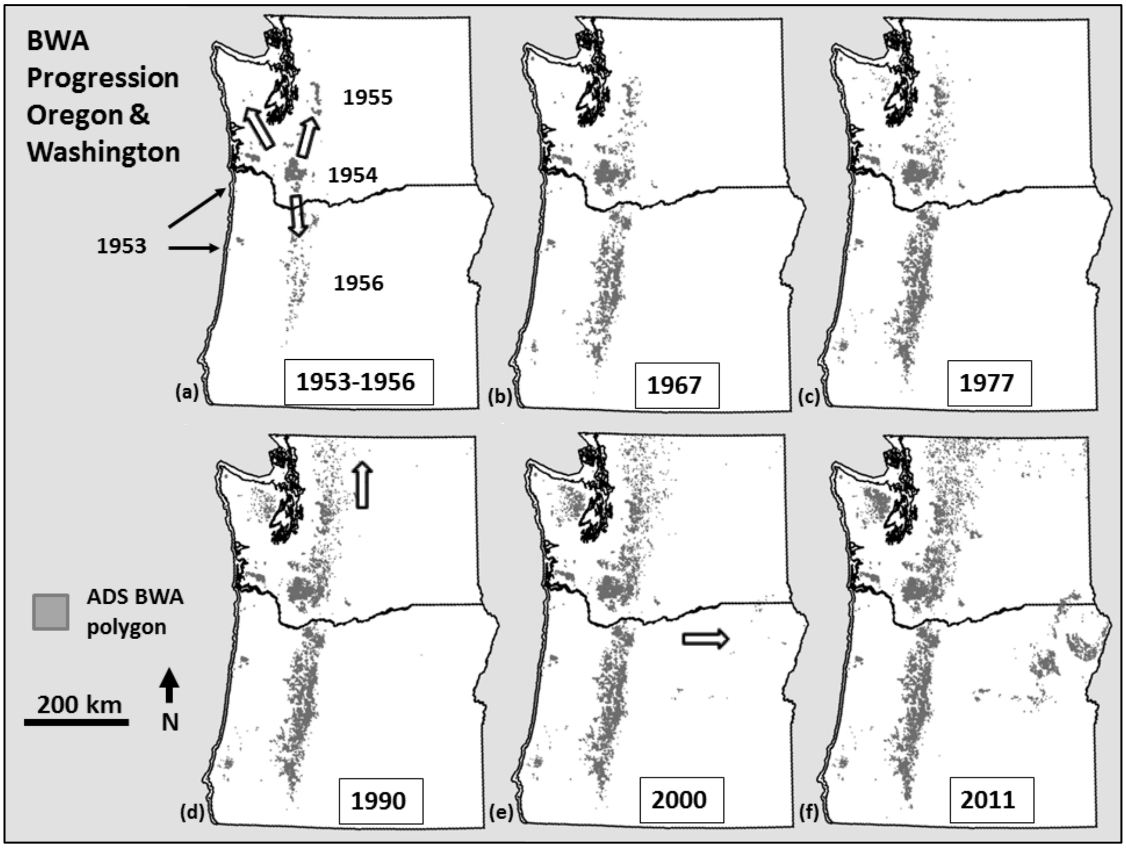

Figure 4.4.5. Spatio-temporal progression of BWA activity across Oregon and Washington documented by cooperative DNR and USFS aerial detection surveys (ADS), 1953 to 2011. Black arrows show where BWA was first sighted in Oregon (1953) along the coast; white block arrows indicate dominant new direction of movement, and BWA continues spread within infested areas as well. From Hutten (2015). Source: USDA Forest Service Pacific Northwest Region Aerial Detection Surveys.

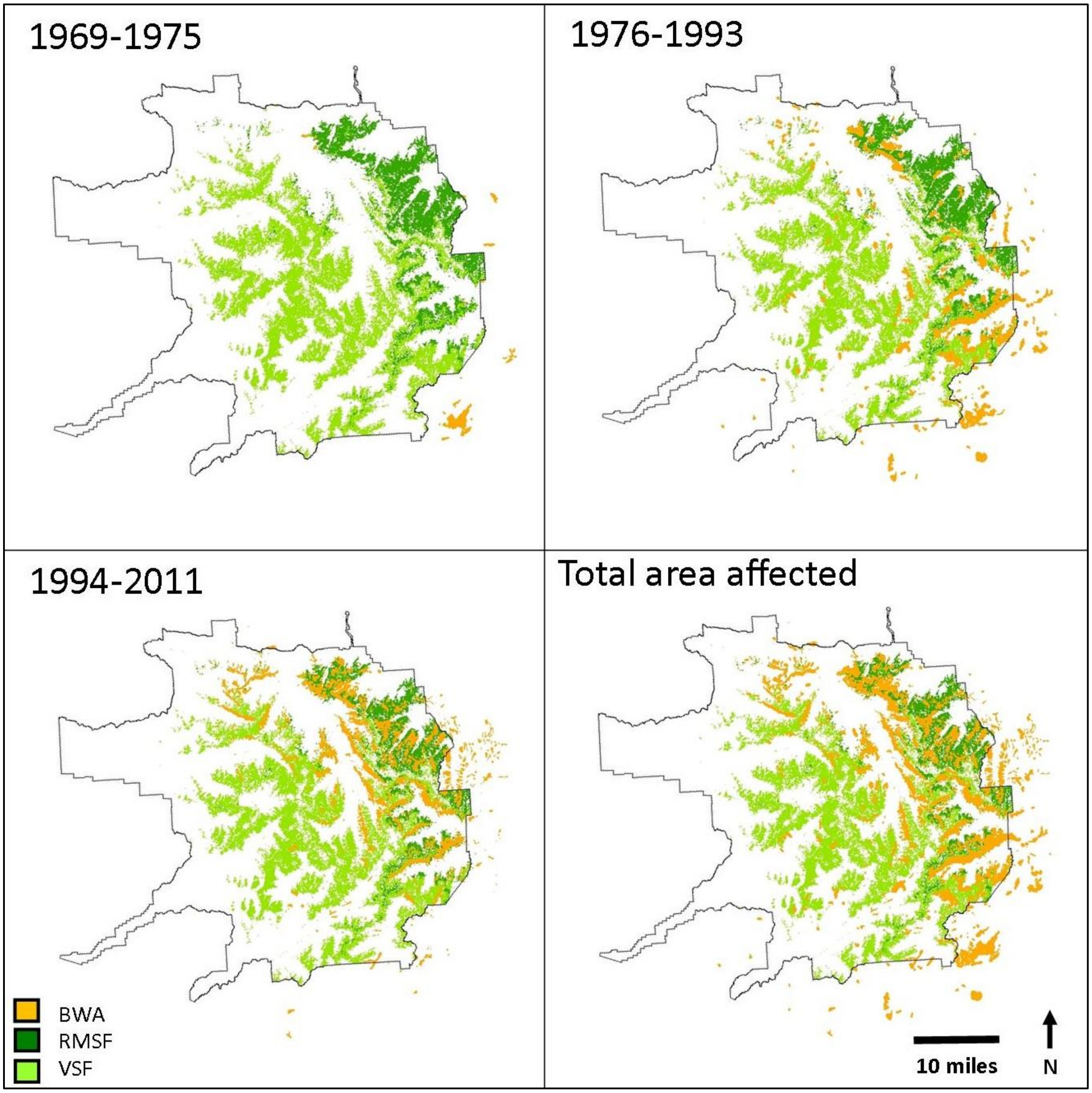

Figure 4.4.6. Spread of BWA (orange) across the Olympic Peninsula over three time periods and total area affected from ADS. BWA occurs within two forest types that contain subalpine fir (National Vegetation Classification Standard macro groups): RMSF = Rocky Mountain Subalpine & High Montane Conifer Forest (dark green), VSF = Vancouverian Subalpine Forest (light green). Source: USDA Forest Service Pacific Northwest Region Aerial Detection Surveys, and the National Vegetation Classification Standard, Version 2.

Linking Spectral Signature with Incidence of BWA

In the Olympics, BWA infestations were associated with a low-intensity change in spectral NBR values. This was determined by observing the coincidence of NBR trends with BWA-infested areas as detected by ADS. When we graphed low-magnitude spectral decline relative to elevation for all of OLYM, we found that the distribution matched that of the host tree (subalpine fir) as well as ADS-detected BWA (Figure 4.4.7). Ground plots established in these areas confirmed the presence of BWA (Hutten 2015).

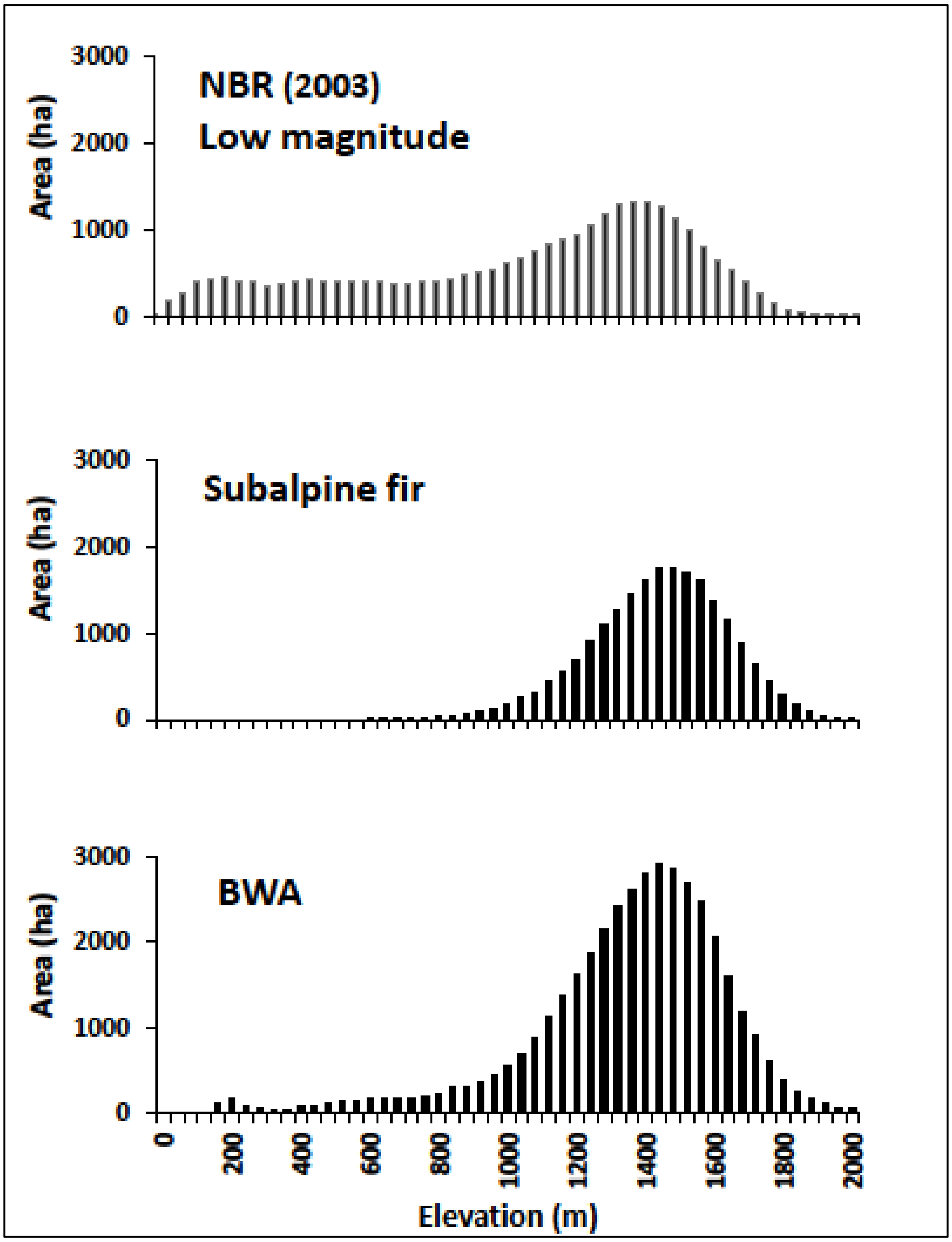

Figure 4.4.7. Elevational coincidence of low magnitude change in forest health, subalpine fir, and ADS-detected BWA. Source: Hutten 2015.

Extent of Subalpine Fir Decline

LandTrendr analysis of OLYM shows that low-magnitude decline in NBR, characteristic of BWA infestation, began prior to 1985 and increased steadily in extent from 1993 to 2007 at an average rate of 1,605 ha per year (Figure 4.4.8). The rate of change was greatest during warm phases of the PDO when the extent increased at an average rate of 1,945 ha per year. The affected area declined in total extent from 2007 to 2012 at a rate of 1,542 ha per year; this may represent a decrease in BWA effects due to a reduction in drought stress, a decline in BWA activity, or a decrease in live host trees and ingrowth of non-host tree species. These changes are consistent with BWA being a sap-sucking insect: symptoms may appear worse during hot or dry periods due to a combination of BWA-induced stress and water stress. More time is needed to determine whether this is a long-term trend or a short-term pause. Low-magnitude decline in forest health has affected at least 24,857 ha of subalpine forest area since 1985 (~10,000 ha were already in decline prior to 1985). This is nearly 7% of OLYM including both forest and non-forest areas.

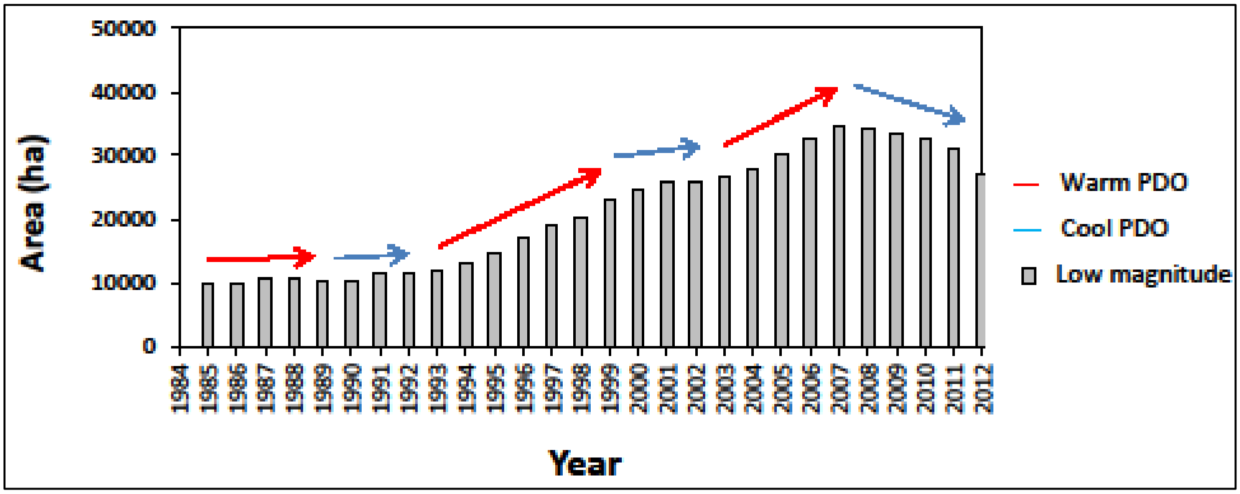

Figure 4.4.8. Total land area affected by BWA in Olympic National Park, 1984-2012, as indicated by LandTrendr analysis (low magnitude decline in NBR), and in relation to PDO phase. Source: modified from Hutten 2015.

Basal Area Affected

There is a positive linear relationship between the magnitude of change in NBR and the basal area of dead and unhealthy trees (Figure 4.4.9). The regression equation for this relationship may be used to remotely estimate the impacts of BWA on subalpine fir trees across OLYM with the use of LandTrendr processed satellite imagery. We estimated that the total basal area of dead and dying trees in subalpine fir forest affected by BWA was 858,259 m^2 as of 2012. This is equal to 13.5 million subalpine fir trees of average size (28.5 dbh), although confidence in this number is compromised by the relatively low value of r^2 (0.36). Damage estimates were restricted to areas within forest types containing subalpine fir that also demonstrated magnitude change in the range typical for BWA infestations. Magnitude change was further limited by the bounds of the model and is thus a conservative estimate of tree basal area affected by BWA.

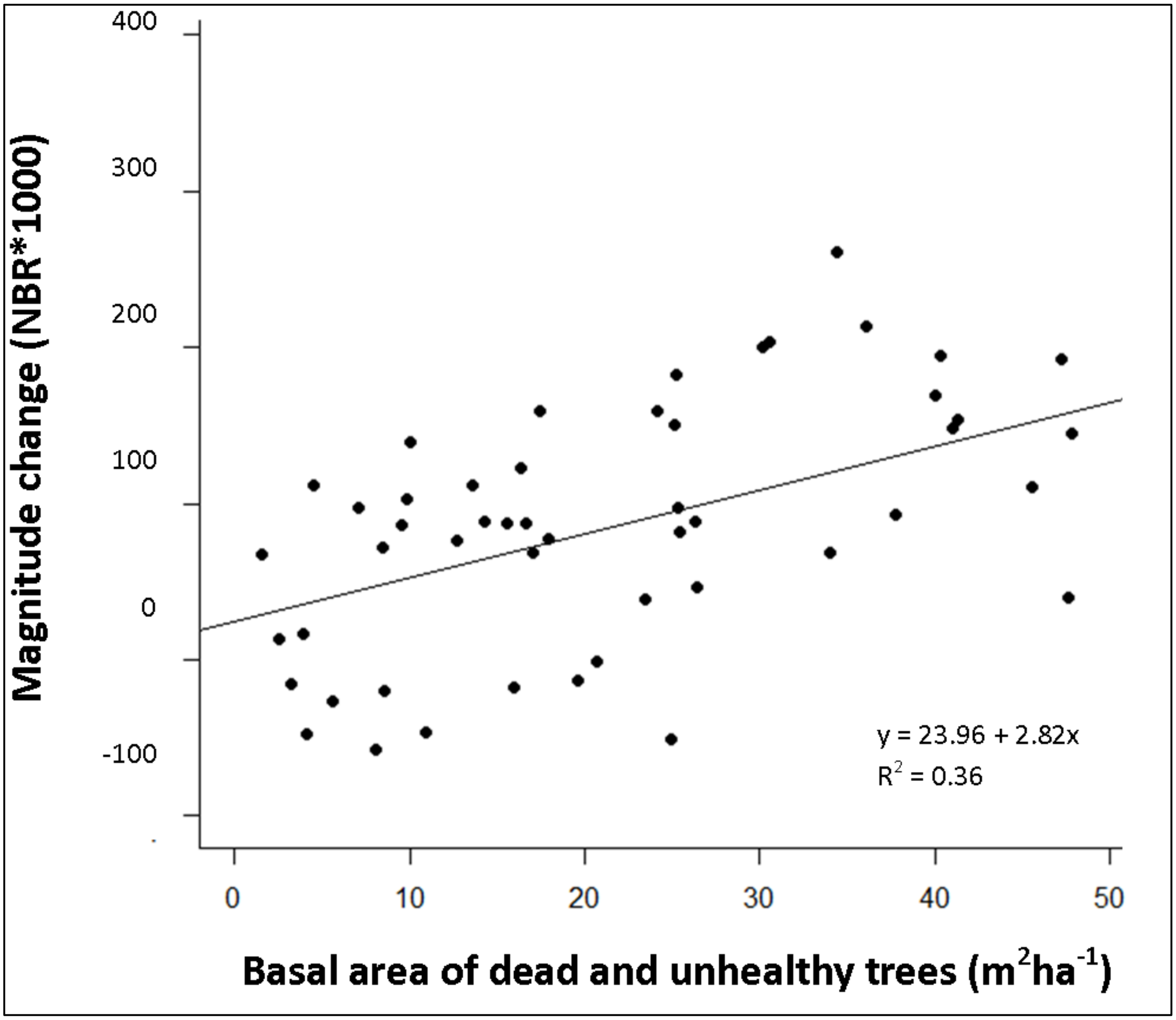

Figure 4.4.9. Relationship between NBR and basal area of dead and unhealthy trees. Source: Hutten 2015.

Uses and Limitations

Our approach to investigating forest health, using two remote sensing methods and field sampling, improved our ability to see broad-scale and emergent patterns and to verify active processes. Land managers are challenged by the narrow perspective in space and time that is generally available to them. It is not feasible to visit every forest location in OLYM, and it is not possible to evaluate trends from a single observation. Aerial detection surveys have extended our ability to observe over large areas at the resolution of the human eye, and to document past conditions. Similarly, Landsat imagery offers a broad-scale view, is unbiased relative to ADS observations, and past image data can be revisited repeatedly. Meanwhile, field observations inform us about current fine-scale processes that are necessary for interpreting broad-scale patterns. We used these methods together to capitalize on strengths and compensate for limitations.

Satellite imagery used in conjunction with ADS can reduce observation bias and the potential for over- or under-estimating infestation area or severity by human observers. Creating disturbance maps using the ADS sketchmapping method is highly subjective and therefore variable among observers (Klein et al. 1983). The ADS data are best used for detecting new infestations and identifying possible disturbance agents based on species and patterns of affected trees, but may not precisely determine the area affected (Johnson and Wittwer 2008). Mapping accuracy has been improved with the advent of digital technology (digital aerial sketchmapping; Schrader-Patton 2002), including touch screens and integrated GPS. Nevertheless, remotely determining the cause of a disturbance, either with imagery or by human observation, will remain subjective for the foreseeable future.

Describing change in vegetation based on reflectance values captured in satellite images can be subject to error due to differences in atmospheric moisture and plant phenology between images. LandTrendr methods compensate for this error by using algorithms to select and delineate the dominant change trend. Secondarily, this can result in an over simplification or misrepresentation of trend for individual pixels when change is complex or subtle. However, at broad scales and for an agent such as BWA that causes widespread and continuous long-term decline, the dominant trend is evident and becomes more certain over time.

The range of NBR values used to describe low-magnitude decline used by Hutten (2015) as an indicator of BWA damage was determined by its ability to distinguish areas identified by ADS; hence, NBR is not a direct measure of BWA activity. Other sources of damage with a similar spectral signature could be counted as BWA damage, and the two agents may be difficult to separate if they occur in the same forest type. Conversely, on the occasion that BWA causes high-magnitude decline perhaps by interacting with other factors such as opportunistic bark beetles, weather events (wind), or climate cycles (e.g. warm weather and drought conditions), BWA damage may not be detected at these locations. Such synergistic effects can increase the overall rate and magnitude of decline, and the relative effects of multiple agents (a complex) attacking one host species, are not easily differentiated. Nevertheless, it is sometimes possible to separate relative impacts from multiple biotic agents using elevation, habitat, host species, or additional spectral indices (e.g., Meigs et al. 2015, Senf et al. 2015).

While the methodology used to identify the spectral signature of BWA can be used elsewhere, the results reported here are specific to OLYM during a particular time period (1985-2012). Our methods may be used to monitor future activity at OLYM and may be applied elsewhere with the understanding that forest composition and disturbance agents and their magnitude can change over space and time.

Emerging Issues

Balsam woolly adelgid is not the first and is unlikely to be the last non-native biotic agent to affect forests in OLYM. White pine blister rust pre-dates BWA and has killed numerous 5-needle pines including western white pine and white bark pine (USFWS 2011b). With circumglobal transport of people, plants, and wood products, the future introduction of non-native organisms is a real and constant threat (Pejchar and Mooney 2009). Specialists remain vigilant against insects and disease that are known threats to local tree species (e.g. Asian long-horned beetle and Asian gypsy moth;

Haack et al. 2010, USDA 2014), but new sources continue to surprise managers (e.g. golden-spotted oak borer in southern California and sudden oak death in Oregon, Goheen et al. 2002, Coleman et al. 2015).

Human-assisted movement of BWA and other insects and disease into unaffected ecosystems is an ongoing issue. Insects may be transported short or long distances when for example BWA-infested Christmas trees are trucked from the farm to homes or businesses, or dying BWA-infested trees are cut down and transported to campgrounds as firewood. For some invasive insects and disease, quarantines are established around known infestations, and movement of wood products from these locations is restricted by law (http://www.dontmovefirewood.org, accessed 01 June 2016); however, BWA is not on this list and the laws do not restrict transport from areas harboring new or lessor known invasive organisms. Preventative measures must therefore be accomplished through education and voluntary action led by organizations such as Don't Move Firewood (http://www.dontmovefirewood.org, accessed 01 June 2016).

Finally, climate change is likely to interact with disturbance agents and further impact sensitive subalpine fir communities. Vegetation interacts with and responds to changes that exceed the adaptive range of individual species in surprising and unpredictable ways. This was the case with yellow cedar (*Cupressus nootkatensis*) in southeast Alaska when early snow melt left shallow roots exposed to spring freeze events and ultimately caused the death of yellow cedar at low elevations across >500,000 ac (Hennon et al. 2016). A reduction in snow accumulation also affects water storage and distribution, and the effect of summer drought which can alter tree distribution (Lutz et al. 2010). Subalpine fir is adapted to high-elevation and high-latitude environments where heavy snowpack conditions allow it to successfully out-compete other tree species along the alpine treeline (Uchytil 1991). It is also more drought tolerant than some conifers, thus climate change could allow subalpine fir to move up in elevation, or increase in abundance in moist forests that become more dry (Rochefort and Peterson 1996, Zolbrod and Peterson 1999). However, interaction with insects and disease increases the complexity because warmer temperatures favor insect population growth, and drought conditions stress trees and reduce their ability to defend against insect attacks (Bentz et al. 2010). Increased endemic insect and disease activity are expected to accompany warming temperatures (Littell et al. 2010). Add to that an introduced invasive species like BWA and the effects become more pronounced.

Information Needs/Gaps

Invasive Species

Managers of protected areas would benefit from an improved understanding of invasive insects and pathogens; how they are introduced and spread. An assessment of the type of activities that result in the introduction and spread of invasive species could be used to improve guidelines and education programs that inform visitors, increase awareness, and change behavior. Protected lands would also benefit from an assessment and restructuring of broad-scale policy and laws to reduce the risk of invasive species introductions.

Collaboration

Cooperation and sharing of information and resources across management jurisdictions is critical for addressing invasive species that move readily across land ownership boundaries. Olympic National Park would benefit by regularly reviewing ADS reports and online disturbance maps. Information on new and emerging non-native threats, such as gypsy moth (*Lymantra dispar dispar*) trap results and movement of the emerald ash borer (*Agrilus planipennis*), is also available from the Washington Department of Agriculture pest program and the Washington Invasive Species Council. Park staff could contribute to USFS/DNR efforts by confirming ADS observations with field investigations and communicating their findings on datasheets that are provided online. Sharing of field data would inform ADS and may improve surveys and the monitoring system overall.

Disturbance Agent Detection

There is a need to further develop our remote-sensing capabilities for detecting and differentiating disturbance agents, especially insects and pathogens. A combination of spectral indices and GIS data layers can be used to identify attributes unique to specific agents or complexes. The Laboratory for Applications of Remote Sensing in Ecology lab in Corvallis, Oregon continues to work with willing land managers to fine tune these skills.

Restoration

Some subalpine fir trees show resistance to BWA. Individual trees may have a more effective defense system or tolerate BWA better when environmental conditions support tree health. Identifying and monitoring resistant trees may help with future restoration activities. Seeds may be collected and tested in breeding programs such as at Dorena Genetic Resource Center in Cottage Grove, Oregon.

Predators for BWA

Members of the family Adelgidae have no known parasites and are susceptible to few fungi (Schooley et al. 1984). This may be because BWA feeding causes lethal damage to trees before BWA populations are numerous enough to attract and sustain high numbers of predators (Mitchell and Buffam 2001). More than 30 predators have been imported and released since 1933 in an effort to control BWA (Montgomery and Lyon 1995), but none have been effective (Schooley et al. 1984), although some may persist in the environment for at least 20 years (Humble 1994). Most work on biological control agents is several decades old and could perhaps be revisited with greater success.

Adaptive Management

A better understanding of disturbance agents and regimes, interactions with climate change, and vulnerability of tree species in OLYM would help inform potential management options. Unexpected consequences may result from the compounded effects of multiple disturbances (Paine et al. 1998), and it is difficult to predict precisely how and which resources will be affected (e.g. yellow cedar decline). However, understanding the kinds of events that have occurred in the past and that may occur in the future can aid managers in building scenarios and developing flexible management plans that are able to adapt to change with new information. Activities of skilled personnel would ideally be prioritized to carry out proactive management, develop adaptive management plans, and focus on collaboration to improve landscape change monitoring.

Resource Summary

We determined that the condition of subalpine fir communities resulting from presence of BWA is of moderate concern, with a deteriorating condition over time (Table 4.4.4). BWA effects have been observed in subalpine forests of OLYM since 1970 with increasing severity from 1993-2007. By 2007, BWA-type decline was detected on 24,857 ha, covering nearly 7% of the total area of OLYM. Many subalpine fir trees have died, which changes tree species composition and has unknown consequences for ecosystem function or animals that rely on subalpine fir.

Table 4.4.4. Resource condition summary for balsam woolly adelgid and subalpine fir in Olympic National Park.

Resource/Measure	Condition Status and Trend	Rationale
Distribution of BWA	🟡	ADS indicates that BWA distribution has increased in subalpine forest of OLYM since detection in the 1970s.
Extent of subalpine fir decline	🟡	Detection of health decline in subalpine fir stands has increased at a rate of 1,605 ha per year since 1985 and totals 24,857 ha.
Basal area of Subalpine fir affected	🟡	Basal area of dead and dying trees is linearly related to magnitude of spectral change; unhealthy trees have increased over time and can be calculated with a regression equation.
Overall condition of Balsam woolly adelgid	🟡	BWA has been present since 1970 with increasing severity until 2007.
Overall condition and trend of Subalpine fir	🟡⇩	Many trees have died, with unknown future consequences for these communities and ecosystem function.

4.4.3. Invasive Non-native Plants (Native Plants)

Andrea Woodward, Joshua Chenoweth, and Janet Coles

Introduction

Non-native plant species have been part of the Olympic flora since the earliest written records: in 1890, botanist L.F. Henderson noted non-native species he observed at Lilliwaup on Hood Canal while accompanying the O’Neil expedition (Henderson 1891). The first attempted comprehensive list of non-native species was published by Jones (1936). Thus non-native species have been documented in Olympic Peninsula ecosystems since the arrival of EuroAmericans and long before the park’s establishment.

Many terms have been used to describe species transported to places outside of their native range by human activities, and these terms have varied by agency and changed over time (Table 4.4.5). Of these species, some are considered invasive because they can spread widely without the aid of human cultivation in a new environment and negatively affect native plant communities. Invasive species that are recognized by federal, state, or local governments to threaten agricultural crops, local ecosystems, or fish and wildlife habitat are given the legal designation "noxious weed" (Washington State Noxious Weed Control Board, http://www.nwcb.wa.gov/, accessed 30 October 2017) and are subject to regulations concerning control measures (Table 4.4.5).

Table 4.4.5. Glossary of terms used to describe non-native plant species.

Term	Definition
Non-native	A species that occurs in a given location as a result of direct, indirect, deliberate or accidental actions by humans (NPS definition). Can also be called "exotic", "alien", or "introduced."
Invasive	A non-native species whose introduction does, or is likely to cause economic or environmental harm or harm to human, animal, or plant health (Executive Order 13112, 1999)
Noxious weed	A legal term describing an invasive species that is recognized by federal, state, or local governments to threaten agricultural crops, local ecosystems, or fish and wildlife habitat

In general, most non-native plant species have minor effects on natural ecosystems. For example, of the 1,352 vascular plant species in OLYM (NPSpecies 2017), 248 are not native to OLYM, but only approximately 38 of these species and varieties are of concern to park managers because they are considered threatening to park resources (Chenoweth et al. 2011); of those that are of concern, 14 have "high priority" for control (Table 4.4.6). Though relatively few, some invasive non-native species can be extremely disruptive by interfering with natural processes, altering disturbance regimes and biogeochemical cycles, or threatening the survival of naturally evolved plant assemblages and abundances of native species (D'Antonio and Vitousek 1992, Hiebert and Stubbendieck 1993, Vitousek et al. 1996, 1997, Mack et al. 2000, Asner and Vitousek 2005, Strayer et al. 2006). In fact, invasive non-native species have been identified as one of the biggest threats to biodiversity, ecosystem function, and community interactions (Boersma et al. 2006). An example in the Pacific Northwest is the invasive knotweeds (*Polygonum x bohemicum*, *P. cuspidatum*, and *P. sachalinense*). These large knotweeds invade riparian zones and replace important woody species such as red alder, which reduces the nutrient subsidy from riparian litterfall to aquatic systems (Urgenson et al. 2009). Moreover, exotic species can disrupt the accurate presentation of a historic scene and damage historic or archeological resources (Hiebert and Stubbendieck 1993).

The list of "high-priority"' and "primary" invasive non-native plant species for OLYM (Figure 4.4.10, Table 4.4.6) was developed from a combination of sources. In general, the list consists of Washington State Noxious Weeds (http://www.nwcb.wa.gov/, accessed 17 October 2017) specifically found in Clallam County (http://www.clallam.net/weed/, accessed 17 October 2017). However, some of these species are not included on the OLYM list because they are rare, apparently not spreading or have low impact in OLYM; others were added (cheatgrass [*Bromus tectorum*], English holly [*Ilex aquifolium*], laurel cherry [*Prunus laurocerasus*], perennial pea [*Lathyrus latifolius*], purple foxglove [*Digitalis purpurea*], and small everlasting peavine [*L. sylvestris*]) because they have high impact and may be more common outside of OLYM than inside. High-priority species are those identified by Clallam County as required or encouraged to be controlled and had a “serious” or “significant” threat rating in Olson et al. (1991). Several other species with lower state and county ratings were also designated as high priority in OLYM because of their high potential impact, ease of spread and/or high potential for control (cultivated knotweed [*Polygonum polystachyum*], English holly, and evergreen blackberry [*Rubus laciniatus*]). These priorities are being revisited using more formal methods during revision of the exotic plant management plan.

Figure 4.4.10. Sample of invasive non-native plant species in Olympic National Park: a) Scot’s broom (*Cytosus scoparius*) in the Elwha Valley; b) Himalayan blackberry (*Rubus armeniacus*) around Andrew’s Homestead in the Queets Valley; c) Bohemian knotweed (*Polygonum x bohemicum*) near Lake Ozette; and d) Canada thistle (*Cirsium arvense*) along the Queets River. All photographs courtesy of NPS.

Table 4.4.6. Invasive non-native plant species thought to threaten park resources in Olympic National Park. Some species are high priority for control (High Priority) while others are of concern (Primary). Nomenclature corresponds to Washington State Noxious Weed Control Board. N/A indicates species that are not yet ranked by NatureServe or included on the Washington State noxious weed list.

Level of priority	Species	Common Name	Life Form	NatureServe Invasiveness Rank	WA State Noxious Weed List
High-priority invasive non-native plant species	*Cirsium arvense*	Canada thistle	Forb	HIGH	Class C
	Cytisus scoparius	Scot's broom	Shrub	HIGH	Class B
	Geranium robertianum	herb Robert	Forb	N/A	Class B
	Hedera helix	English ivy	Vine	HIGH	Class C
	Hypericum perforatum	common St. John's wort	Forb	HIGH	Class C
	Ilex aquifolium	English holly	Small tree	HIGH	N/A
	Leucanthemum vulgare	oxeye daisy	Forb	MEDIUM	Class B
	Phalaris arundinacea	reed canarygrass	Graminoid	HIGH	Class C
	Polygonum cuspidatum	Japanese knotweed	Shrub	HIGH	Class B
	Polygonum polystachyum	cultivated knotweed	Forb	HIGH	Class B
	Polygonum sachalinense	giant knotweed	Shrub	HIGH	Class B
	Polygonum x bohemicum	Bohemian knotweed	Shrub	HIGH	Class B
	Senecio jacobaea	tansy ragwort	Forb	LOW	Class B
	Rubus laciniatus	evergreen blackberry	Shrub	N/A	N/A
Primary invasive non-native plant species	*Bromus tectorum*	cheatgrass	Graminoid	HIGH	N/A
	Buddleja davidii	butterfly bush	Shrub	HIGH	Class B
	Centaurea debeauxii ssp. *Thuillieri*[1]	meadow knapweed	Forb	N/A	Class B
	Centaurea diffusa	diffuse knapweed	Forb	HIGH	Class B
	Centaurea jacea	brown knapweed	Forb	N/A	Class B
	Centaurea stoebe[1]	spotted knapweed	Forb	HIGH	Class B
	Cirsium vulgare	bull thistle	Forb	MEDIUM	Class C
	Clematis vitalba	evergreen clematis	Vine	MEDIUM	Class C
	Conium maculatum	poison hemlock	Forb	MEDIUM	Class B
	Daphne laureola	spurge laurel	Shrub	MEDIUM	Class B

[1] These species occur near the park boundary and will be high priority if found inside Olympic National Park

Table 4.4.6 (continued). Invasive non-native plant species thought to threaten park resources in Olympic National Park. Some species are high priority for control (High Priority) while others are of concern (Primary). Nomenclature corresponds to Washington State Noxious Weed Control Board. N/A indicates species that are not yet ranked by NatureServe or included on the Washington State noxious weed list.

Level of priority	Species	Common Name	Life Form	NatureServe Invasiveness Rank	WA State Noxious Weed List
Primary invasive non-native plant species (continued)	*Digitalis purpurea*	purple foxglove	Forb	MEDIUM	N/A
	Echium vulgare	common viper's-bugloss	Forb	N/A	Class B
	Heracleum mantegazzianum	giant hogweed	Forb	MEDIUM	Class A
	Hieracium aurantiacum	orange hawkweed	Forb	MEDIUM	Class B
	Hypochaeris radicata	Hairy cat's ear	Forb	N/A	Class C
	Iris pseudacorus	yellow iris	Forb	HIGH	Class C
	Lathyrus latifolius	perennial pea	Forb	N/A	N/A
	Lathyrus sylvestris	small everlasting peavine	Forb	N/A	N/A
	Linaria dalmatica ssp. *dalmatica*	dalmatian toadflax	Forb	N/A	Class B
	Linaria vulgaris	butter and eggs	Forb	HIGH	Class C
	Lythrum salicaria	purple loosestrife	Forb	HIGH	Class B
	Potentilla recta	sulfur cinquefoil	Forb	HIGH	Class B
	Prunus laurocerasus	laurel cherry	Shrub	N/A	N/A
	Rubus armeniacus	Himalayan blackberry	Shrub	MEDIUM	N/A

Records of non-native plant species sightings in OLYM were first systematically summarized and prioritized in 1991 (Olson et al. 1991), following the OLYM draft Resource Management Plan call to initiate an exotic plant management program in 1990. The NPS as a whole is mandated to combat invasive non-native species (Executive Order 13112), they are a management priority (NPS 2006), and the need to address them has resulted in funding for teams of non-native plant management technicians (Exotic Plant Management Teams, EPMTs) to work throughout the national parks (Beard and Gibson 2011). In this assessment, we summarize what is known about the status and spread of non-native plant species in OLYM with the goal of prioritizing future monitoring and management of these species.

Approach and Methods

Our goal was to describe the status of non-native plant distribution and abundance in OLYM using the following measures: (1) spatial distribution of invasive non-native plant species based on accumulated georeferenced records of invasive non-native species occurrence, (2) rate of change of non-native or invasive non-native species abundance in areas having repeated surveys, and (3) status of invasive non-native species establishment in the de-watered reservoirs following dam removal on the Elwha River. The de-watered reservoirs may provide ideal habitat for establishment of invasive species.

Spatial Distribution of Invasive Non-native Plant Species

The spatial distribution on invasive plant species can be best understood from maps of geo-referenced vegetation survey data. The most thorough compilation of geo-referenced data for OLYM was developed by Jones and Halpern (2007) as part of an effort to model potential habitat locations for select invasive non-native species. Most data were collected in the early 2000s by NPS and Olympic National Forest. Some data were collected to specifically target non-native species while others were collected during sampling of vegetation for other reasons. Jones and Halpern (2007) created a geodatabase for 9 plant species: Canada thistle (*Cirsium arvense*), herb Robert (*Geranium robertianum*), English ivy (*Hedera helix*), English holly, Himalayan blackberry (*Rubus armeniacus*), evergreen blackberry, reed canary grass (*Phalaris arundinacea*), giant knotweed (*Polygonum sachalinense*), and Japanese knotweed (*Polygonum cuspidatum*). Georeferenced treatment locations and some survey data have also been collected by the EPMT, however only data from 2015 and 2016 are currently available. These data have been combined into one geodatabase for the 9 species originally compiled by Jones and Halpern (2007) as well as bohemian knotweed (*Polygonum x bohemicum*). The EPMT data were reported as polygons and were converted to points by estimating the center point of each polygon. Georeferenced data from a survey of the road adjacent to Lake Crescent are also available but were not incorporated into the Jones and Halpern (2007) database.

Rate of Change

Describing rate of spread of non-native species in the park depended on having repeat records, but these were relatively rare. Plant surveys in OLYM are typically time-consuming, expensive, and difficult to conduct due to inaccessibility. Hence, it was unusual for areas to be revisited when so much of the park had never been surveyed. Nevertheless, there were several sources of information that indicated changes in the distribution and abundance of non-native or invasive species over time:

- *Peninsula-wide surveys:* Non-native species have been mentioned since the earliest botanists visited the peninsula (Henderson 1891). Comprehensive floras of the Olympic Peninsula have been produced over time (Jones 1936, Buckingham and Tisch 1979, Buckingham et al. 1995), and a new flora is currently being compiled by botanist Fred Weinmann. Non-native species documented by 1900 were cited by Jones (1936). We compare the number of species included in these records over time without regard to whether they are considered invasive with the aim of describing the rate of introduction of new species to the area. These numbers are a crude estimate of the number of non-native species because they do not take into consideration comprehensiveness of survey or changes in taxonomy.
- *Park-wide surveys:* The first comprehensive compilation of known locations of non-native species in OLYM (Olson et al. 1991) was qualitatively compared with a survey of areas thought to be susceptible to invasion (Rochefort et al. 2016, data collected in 2002). The earlier dataset

(1991) was based on incidental sightings and targeted surveys; locations were expressed as regions of the park with certain specific locations (e.g., campgrounds) sometimes listed. The later data (2002) were collected using a random survey design of susceptible areas (i.e., developed zones, roads, trails, riparian/reservoir areas). We also have an estimate of nonnative species from NPSpecies (2017), the validated list of plant and animals found in national parks.

- *Lake Crescent road surveys:* Repeat surveys of Highway 101 along the edge of Lake Crescent were conducted in 2002 (Rochefort et al. 2016) and in 2014 (Chenoweth 2014). The 2002 survey had a randomized statistical design and results indicated all points that were sampled whether they had non-native species or not. Plots were 100 m^2 and cover of non-native species were recorded in cover classes. These results can be compared with the later survey (Chenoweth 2014), which was a continuous survey along both sides of the 20-km road section for 22 invasive non-native (of which only 14 were found) and 4 rare plant species that could potentially occur. Populations less than 30 m in length parallel to the road were mapped as points; populations greater than 30 m were mapped as lines extending between the beginning and end of the population parallel to the road.
- *Park homestead surveys:* The river valleys of the Olympic Peninsula were settled beginning in the 1890s (Evans 1983). A retrospective study of distribution of invasive non-native species (Williams 1975), followed by later surveys (Del Moral 1985, Riege 2000, Riege and Del Moral 2004, Acker et al. 2014), focused primarily on 5-6 homesteads of the original 64 homesteads in the Queets Valley. Results indicated establishment success, spread, and effectiveness of control efforts for a selection of species, albeit in atypical park environments. Acker et al. (2014) also surveyed for invasive non-native species along the trail and went farther into the park beyond the homesteads.

Status of Elwha Reservoirs

Removal of the Glines Canyon (in 2014) and Elwha (in 2012) dams on the Elwha River drained reservoirs Lake Mills and Lake Aldwell, respectively, and exposed over 300 ha of bare mineral soil. We assembled the data on non-native plant colonization that took place on these exposed reservoir beds following dam removal. We focused on the former Lake Mills reservoir for this assessment because it is entirely within OLYM boundaries. Extensive efforts were made to map and remove sources of propagules prior to dam removal, and to subsequently remove invasive non-native species and to establish native species. Because it was infeasible to address the over 100 non-native species known to be present prior to dam removal, this project focused on twenty Elwha-specific “primary species of concern” that were considered invasive and likely to occur in the area of interest (Chenoweth et al. 2011).

Two methods were used to monitor species in the dewatered reservoirs during and after dam removal: permanent plot surveys and field mapping of invasive non-native plants. Seventy-one 50-m^2 permanent plots were randomly established in the reservoirs to determine presence and frequency of native and non-native species. Presence/absence of the primary species of concern was assessed over the entire plot; abundance was indicated by frequency in 4 1-m^2 subplots in annual surveys since dam removal. In addition, survey crews traversed the entire dewatered reservoir to provide a comprehensive map of invasive species populations so as to more thoroughly determine the extent of invasive species distribution than could be obtained from plots. Georeferenced locations of invasive non-native species were recorded as a point feature while areas surveyed without invasive species were recorded as line features. Data were also collected to describe the density of invasive plants in each located population.

Reference Conditions

The appropriate reference condition for non-native plants is an absence of species transported to the park through human activities. While restoring park lands to the reference condition is likely impossible, it is nevertheless a baseline for evaluating trend. Because historic data on invasive non-native species are very limited, this assessment serves as a baseline or reference for current park conditions, to which future management of invasive non-native plant species can be compared.

Results and Assessment

Spatial Distribution of Invasive Non-native Plant Species

The invasive non-native species locations compiled from Jones and Halpern (2007) and the EPMT include 9 of the 14 high-priority species of concern to park management (Rochefort et al. 2016, Table 4.4.6) plus Himalayan blackberry; Scot's broom (*Cytosus scoparius*), one knotweed species, oxeye daisy (*Leucanthemum vulgare*), common St. John's wort (*Hypericum perforatum*), and tansy ragwort (*Senecio jacobea*) were not included (Figure 4.4.11). We consider the compiled map to be a useful assessment of focal areas and abundance of invasive species because it includes most high-priority species, even though it does not cover all invasive species considered threatening (Table 4.4.6). Some mapped points are inaccurate because those populations have been reduced or eradicated by the EPMT. However, historic locations are indicators of imperiled areas requiring vigilance, because they have a higher probability of recolonization due to potential seed bank and incomplete treatment.

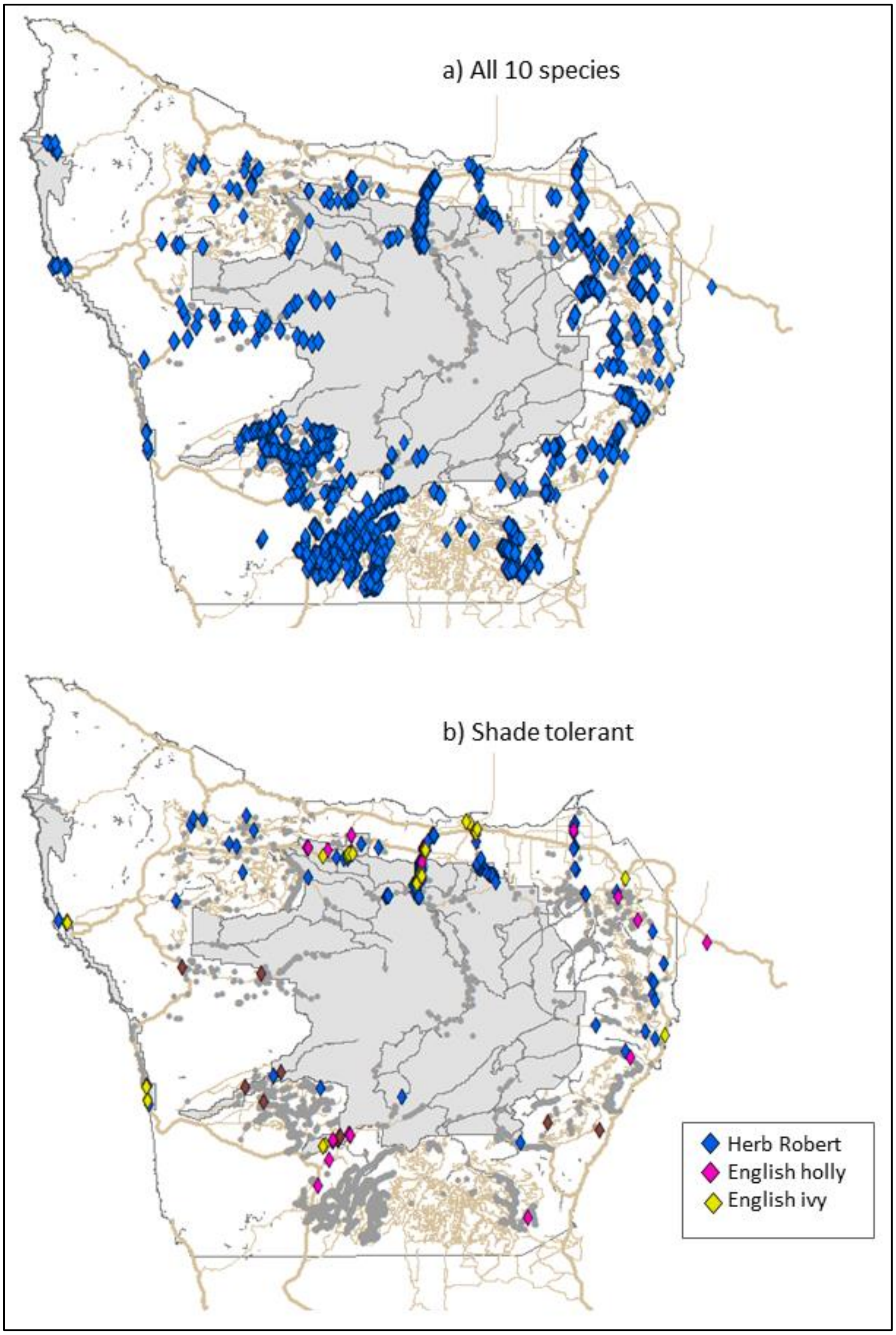

Figure 4.4.11. Distribution of georeferenced locations of 10 non-native invasive plant species in Olympic National Park Showing a) all georeferenced species, and b) shade tolerant species Source: Jones and Halpern (2007) and NPS.

Included in the data points compiled by Jones and Halpern (2007) are results from a random survey design of susceptible areas (developed areas, road, trails and riparian zones; Rochefort et al. 2016, data collected 2002). The random survey attempted to provide a comprehensive view of the distribution of all non-native plant species in OLYM. Rochefort et al. (2016) found non-native species in 72% of randomly sampled plots while the species mapped by Jones and Halpern appeared in less than 10% of the same sample, suggesting a substantial gap between distribution of high-priority invasive species and all non-native species. The five most common species found by Rochefort et al. (2016) – smooth and hairy cat's ear (*Hypochaeris glabra* and *H. radicata*), creeping buttercup (*Ranunculus repens*), wall lettuce (*Mycelis muralis*), and white clover (*Trifolium repens*) – are not considered management priorities. Only hairy cat's ear and wall lettuce are included on the Washington State noxious weeds list.

Maps of data from both Jones and Halpern (2007) and the non-native survey (Rochefort et al. 2016) show that developed areas and roads are the areas most affected by non-native species (Figure 4.4.11, Rochefort et al. 2016). In particular, the Hoh, Lake Ozette, Quinault, and Elwha roads, developed areas around Lake Crescent and Lake Quinault, and the Queets trail in the vicinity of old homesteads have significant infestations of invasive non-native species. The eastern side of the park was not well surveyed in the Rochefort et al. (2016) sample and one might expect more species in the vicinity of the Dosewallips road. While other points from the Jones and Halpern (2007) dataset do not show high-priority species of concern in this area, there are known populations of these species. For example, herb Robert has been present on the Dosewallips road since the early 2000s and is also found around vacation homes near Staircase.

Several OLYM high-priority invasive non-native species – namely herb Robert, English ivy, and English holly (Figure 4.4.11) – can establish under forest canopies and perhaps invade the interior of the park. Presently herb Robert is the most abundant high-priority species, having spread from the site of first observation near Heart of the Hills in the mid-1970s (Jones and Halpern 2007) across the northern part of the Olympic Peninsula with a few sightings on the coast in the southern peninsula. Knotweed can also establish under forest canopy but is typically considered a greater threat to riparian areas than forests (Washington Department of Agriculture 2005).

English holly and English ivy, which arrived sometime between 1936 and 1979 (Olson et al. 1991), are not common but they are both bird dispersed and may be establishing in areas that are not frequently traveled, especially near developed areas where they are currently known (Jones and Halpern 2007).

Reed canary grass and knotweed species (Figure 4.4.12) are significant threats to riparian areas because they are able to suppress native biodiversity with consequences for river flow dynamics and habitat for aquatic organisms (Schooler et al. 2006, Urgenson et al. 2009). Specific consequences include the invasion of reed canary grass in Ozette River and Big River riparian areas with potential to deter the recovery of Lake Ozette sockeye salmon (Haggerty et al. 2009). Also, the dramatic spread of knotweed from one population just outside of the park boundary along 30 miles of the Hoh River in 2-3 years (Silver and Hutten 2005) has required at least a decade of work to control (Silver and Fawcett 2007). Reed canary grass is also abundant at Irely Lake.

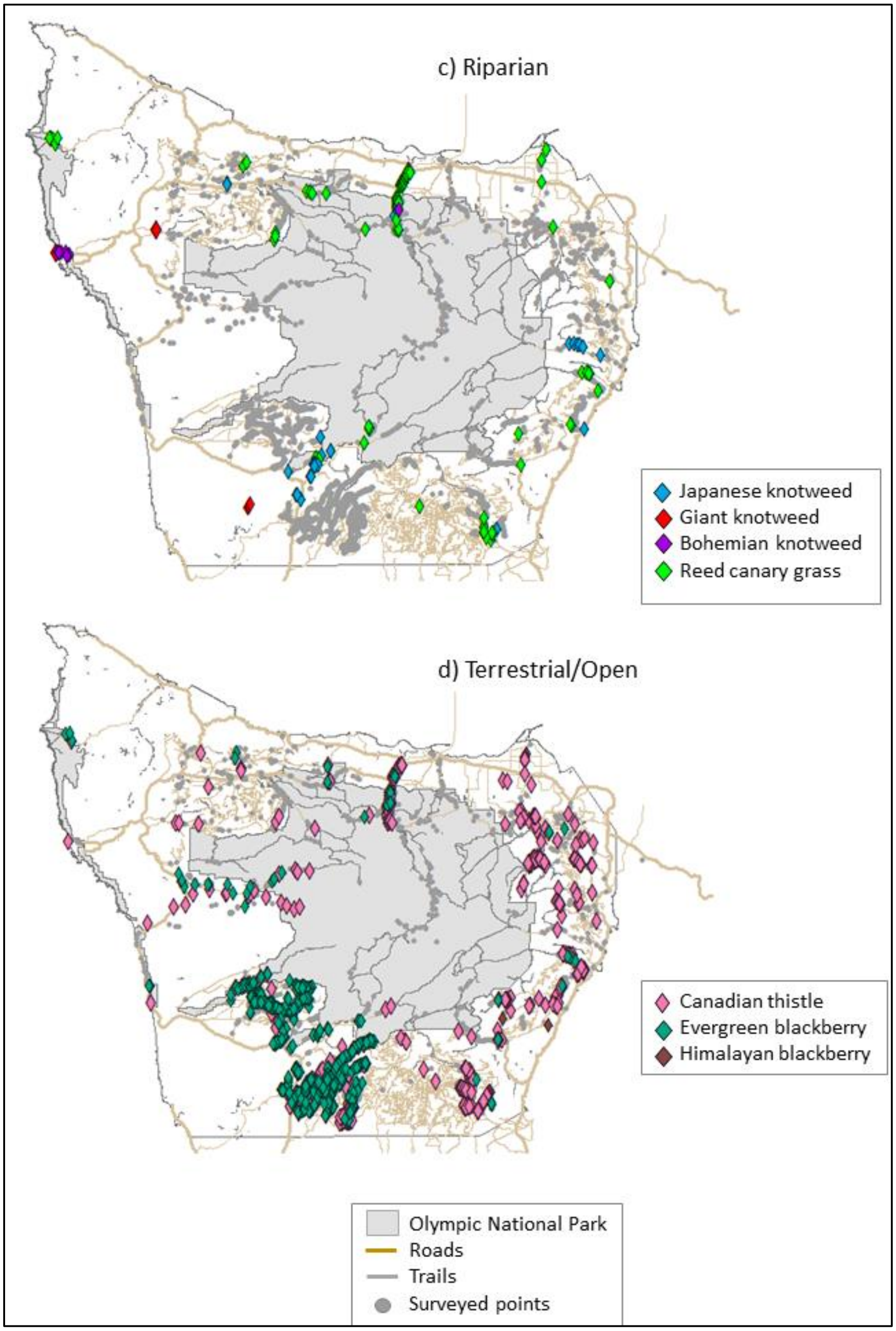

Figure 4.4.12. Distribution of georeferenced locations of 10 non-native invasive plant species in Olympic National Park Showing c) riparian species, and d) species found in terrestrial/open habitats. Source: Jones and Halpern (2007) and NPS.

High-priority species that are limited to terrestrial open areas are Canada thistle and blackberry species (Figure 4.4.12), although Canada thistle often occurs near rivers. The fact that all three plant species have been present on the Olympic Peninsula since before 1900 (Jones and Halpern 2007) and are not widespread in the park may indicate that their potential for establishment and persistence is limited. However, habitat models of Canada thistle, herb Robert, evergreen blackberry, English holly and English ivy suggest that all species are in the early stages of invasion (Jones and Halpern 2007, Jones et al. 2010). The potential for further spread is supported by results of a survey of the Queets drainage showing Canada thistle far up the valley (Acker et al. 2014).

Highway 101 is a major transportation corridor for not only park visitors, but also Olympic Peninsula residents and commercial traffic. A continuous survey of the 19 km of road adjacent to Lake Crescent (Chenoweth 2014) showed that it is heavily infested with invasive species. A total estimated affected area of 1.0 ha was dominated by everlasting peavine (0.57 ha, extending 14.2 km) and herb Robert (0.24 ha, extending 5.1 km). Scot’s broom and reed canary grass were also abundant, extending 2.6 km and 3.1 km, respectively. Although invasive populations were discontinuous, most stretches of the road were affected.

Modeled high risk areas for herb Robert, English ivy and English holly are mostly outside of the park and <10% of the park is at moderate to high risk of being invaded. These areas are primarily in large river valleys at low elevation. Areas at high risk for Canada thistle are also in large river valleys while evergreen blackberry has potential habitat in large river valley and coastal areas on the west side of the Olympic Peninsula (Jones and Halpern 2007, Jones et al. 2010, Figure 4.4.13).

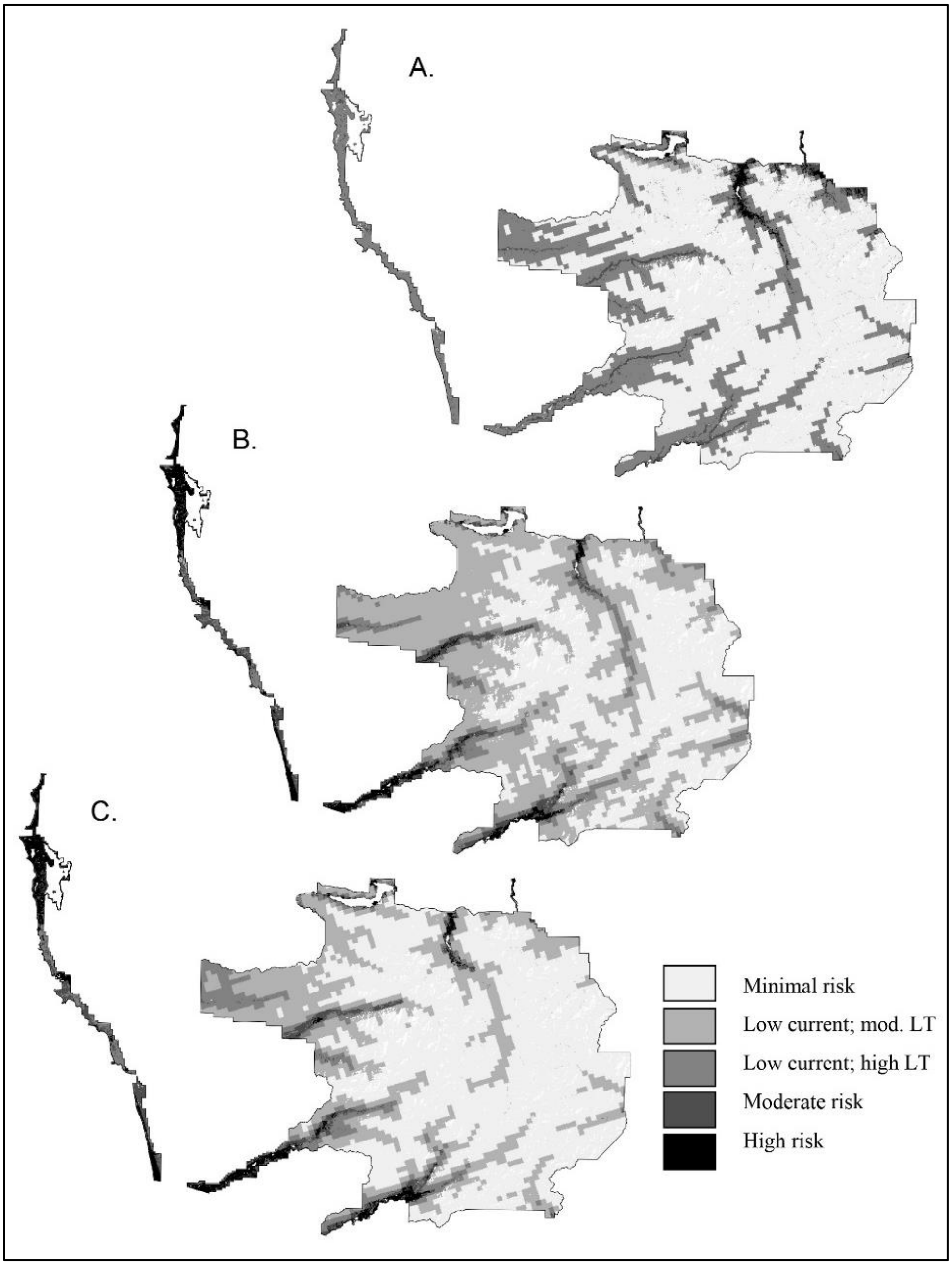

Figure 4.4.13. Areas modeled to have high risk for invasion by (A) herb Robert, (B) English ivy, and (C) English holly (from Jones and Halpern 2007). Categories of risk are: minimal risk; low near-term (current) risk but moderate long-term (LT) risk; low near-term (current) risk but high long-term risk; moderate near-term and high long-term risk; and high risk in the near and long term.

Rate of Change

Peninsula-wide Surveys

Non-native species were present on the Olympic Peninsula at the arrival of EuroAmerican explorers (Henderson 1891) and have increased at a steady rate of approximately three species per year since then (Figure 4.4.14), based on peninsula-wide accounts and floras. The rate of increase and total numbers appear to be lower in the park, but non-native species are numerous. Park data points are from Olson et al. (1991) and NPSpecies (2017).

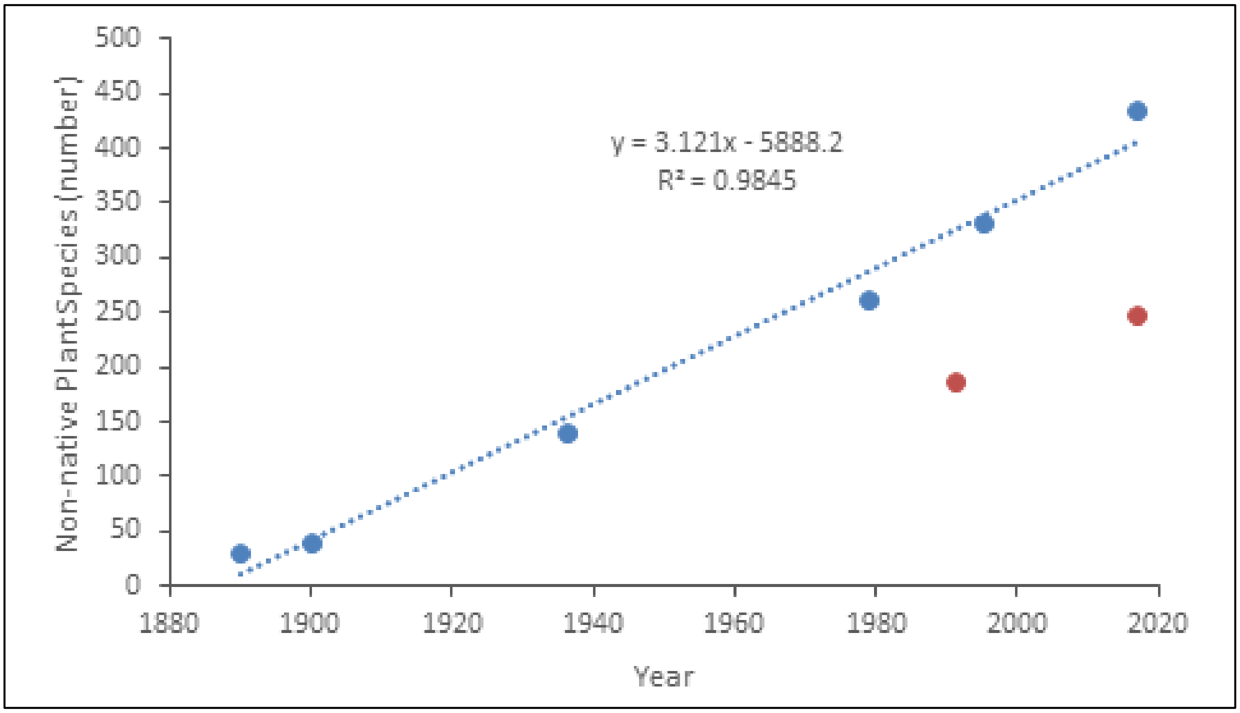

Figure 4.4.14. Trend in number of non-native species in the Olympic Peninsula (blue) and Olympic National Park (red) floras over time. Source: Henderson (1891), Jones (1936), Buckingham and Tisch (1979) Buckingham et al. (1995), and NPS.

Park-wide Surveys

The most comprehensive assessments of invasive non-native species distributions were conducted in 1991 and 2002 (Rochefort et al. 2016). Because methods were very different, the two data sets can only be qualitatively compared (Table 4.4.7). Nevertheless, it does appear that herb Robert and evergreen blackberry became more wide-spread in OLYM during the 11-year interval, while English ivy and English holly may have decreased due to control efforts. Canada thistle, orchard grass (*Dactylis glomerata*), oxeye daisy, common dandelion (*Taraxacum officinale*) and tansy ragwort were determined to be widespread in both surveys.

Table 4.4.7. Comparison between compilation of invasive species records, surveys, and level of concern in 1991 (Olson et al. 1991) versus systematic survey conducted in 2002 and level of concern (Rochefort et al. 2016) for species considered high priority in either study. Darker colors indicate greater apparent abundance. Abundance was measured as number of populations (1991) or percentage of survey plots having each species (2002).

		Level of Concern		Abundance		Location	
Species	**Common name**	**1991**	**2016**	**1991**	**2002**	**1991**	**2002**
Agropyron repens	Quackgrass	Significant	–	>50	?	Sweet's field, Elwha	–
Cerastium fontanum	Common chickweed	Significant	–	16-50	?	Hoh, Elwha, Staircase, South Fork Hoh	Heart of the Hills
Cirsium arvense	Canada thistle	Serious	High	>50	5	widespread	Hoh, Kalaloch, Lake Crescent, Olympic Hot Springs ,Lake Ozette, Queets, Quinault, Rialto beach
Cytisus scoparius	Scot's broom	Significant	High	>50	6	widespread	Lake Crescent, Kalaloch, Elwha dam
Dactylis glomerata	Orchard grass	Significant	–	>50	11	widespread	widespread
Geranium robertianum	Herb Robert	Low	High	<5	9	absent	Elwha, Lake Crescent, HQ, Heart of the Hills, Kalaloch, Quinault
Hedera helix	English ivy	Significant	–	16-50	2	Quinault, Beach 1, Rialto, Kalaloch, Mora, Ozette Ranger Station, Lake Crescent	Lake Crescent, Quinault, OLYM headquarters, Mora
Ilex aquifolium	English holly	Serious	–	>50	3	Quinault, Kalaloch, Beach 4 parking, Lake Ozette, Lake Crescent, Elwha Ranger Station, Altaire	Quinault, OLYM headquarters, Lake Crescent, Kalaloch
Lathyrus sylvestris	Everlasting peavine	Significant	–	6-15	4	Lake Crescent, Sol Duc road	Lake Crescent, Elwha dam

Table 4.4.7 (continued). Comparison between compilation of invasive species records, surveys, and level of concern in 1991 (Olson et al. 1991) versus systematic survey conducted in 2002 and level of concern (Rochefort et al. 2016) for species considered high priority in either study. Darker colors indicate greater apparent abundance. Abundance was measured as number of populations (1991) or percentage of survey plots having each species (2002).

		Level of Concern		**Abundance**		**Location**	
Species	**Common name**	**1991**	**2016**	**1991**	**2002**	**1991**	**2002**
Leucanthemum vulgare	Ox-eye daisy	Significant	–	>50	11	widespread	Elwha, Hoh, Kalaloch, Elwha, Lake Crescent, Heart of the Hills, Olympic Hot Springs, Lake Ozette, Quinault, Sol Duc
Phalaris arundinacea	Reed canary grass	Serious	High	>50	3	Lake Ozette, Lake Crescent, Hurricane Ridge road (low)	Lake Crescent, Lake Ozette, Quinault, Olympic Hot Springs
Poa pratensis	Kentucky blue grass	Serious	–	>50	?	widespread	Queets, Obstruction Point
Polygonum cuspidatum	Japanese knotweed	Significant	High	6-15	?	Quinault, Sol Duc resort	Quinault, Rialto beach
Polygonum sachalinese	Giant knotweed	Significant	High	6-15	?	Quinault, Elwha, Hurricane Ridge	Rialto beach
Polygonum x bohemicum	Bohemian knotweed	Significant	High	?	0.4	absent	absent
Rubus armeniacus	Himalayan blackberry	Significant	High	Insuff. data	?	no locations identified	Queets, Quinault, Kalaloch
Rubus lacinatus	Evergreen blackberry	Significant	High	Insuff. data	9	no locations identified	Queets, Hoh, Olympic Hotsprings, Quinault, Kalaloch, Lake Ozette
Senecio jacobea	Tansy ragwort	Significant	–	16-50	7	Quinault, Queets road, Lake Crescent, Hurricane Ridge road	Heart of the Hills, Lake Crescent, Quinault, Lake Ozette, Rialto beach, Bogachiel, Kalaloch

Table 4.4.7 (continued). Comparison between compilation of invasive species records, surveys, and level of concern in 1991 (Olson et al. 1991) versus systematic survey conducted in 2002 and level of concern (Rochefort et al. 2016) for species considered high priority in either study. Darker colors indicate greater apparent abundance. Abundance was measured as number of populations (1991) or percentage of survey plots having each species (2002).

		Level of Concern		Abundance		Location	
Species	**Common name**	**1991**	**2016**	**1991**	**2002**	**1991**	**2002**
Taraxacum officinale	Common dandelion	Serious	–	>50	10	widespread	Deer Park, Quinault, Kalaloch, Olympic Hot Springs, OLYM headquarters, Lake Crescent, Staircase, Elwha campground, Sol Duc Resort, Heart of the Hills, Hoh, Lake Ozette

Evergreen blackberry has been present on the Olympic Peninsula since before 1900 (Jones 1936) and was common along the Washington coast in 1906 (Piper 1906). At present, it is much more abundant outside of the park than inside (Figure 4.4.12) but given the wide distribution in 2002, it is surprising that it was not documented inside the park in 1991 (Olson et al. 1991). In contrast, herb Robert was first seen in the 1970s at Heart of the Hills, then spread up the Hurricane ridge road, to the Elwha valley by 1992 (Tisch 1992), and across the northern edge of the Olympic Peninsula by 2002 (Rochefort et al. 2016). As evidence that it can spread under forest canopies, it is often found along deer trails far from roads and other human impact areas including a 20-ha forested area well off-trail in the Elwha Valley above the former Lake Mills reservoir (EPMT data).

Lake Crescent Road Surveys

By comparing observations taken in 2002 from randomly distributed plots with observations made within 100 m of the center of each plot in 2012, we can begin to assess change over time of invasive species abundance. These results reflect the net effects of control efforts and spread of populations. Using a distance criterion as high as 100 m perhaps creates a bias toward exaggerating the number of species in 2014. Fourteen plots from the 2002 survey were included in the analysis, and the analysis was restricted to the 22 species surveyed in 2014.

Results show substantial declines in Scot’s broom and tansy ragwort (Figure 4.4.15), two targets of intensive control efforts beginning in 2001. However, we have no estimate of error to evaluate the statistical significance of the difference. Other changes are comparatively minor. Nevertheless, with the exception of herb Robert, there were fewer observations of the 22 target species at the locations sampled in both 2014 and 2002. Avoiding an increase in the number of populations of these aggressive species is an achievement. We were not able to compare the size of the populations from these data.

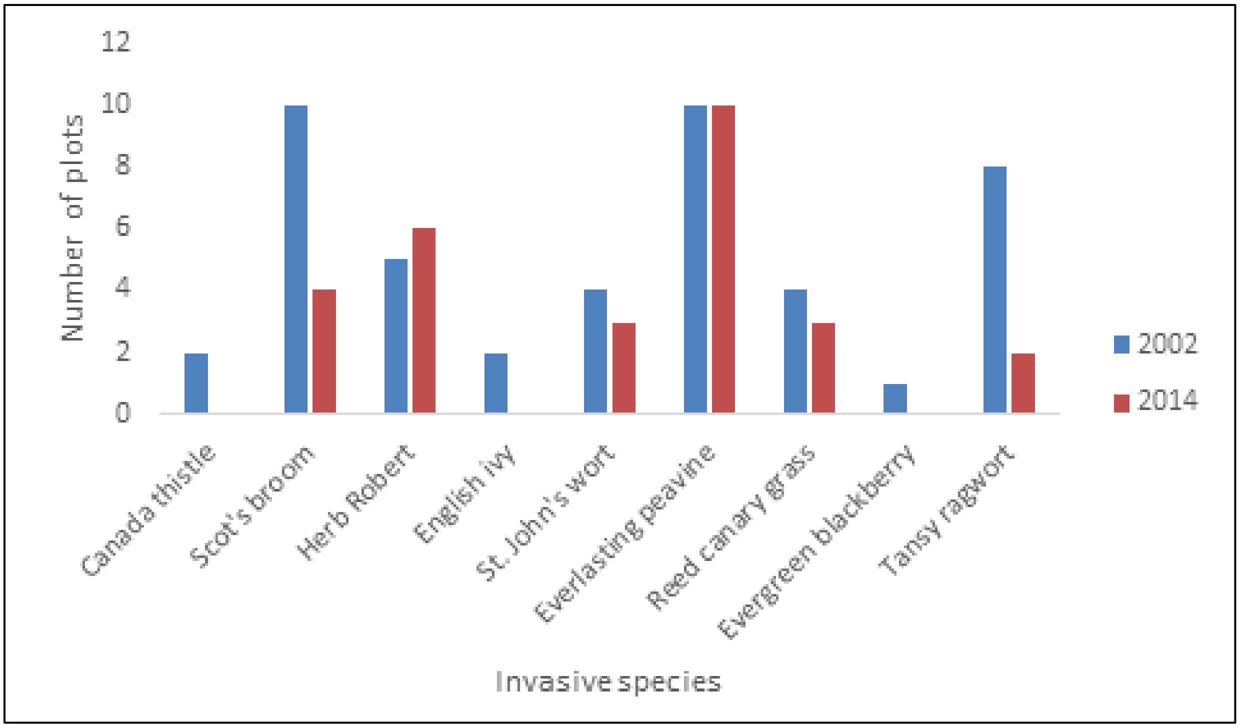

Figure 4.4.15. Incidence of invasive species at plots near Highway 101 adjacent to Lake Crescent in 2002 and 2014. Source: NPS.

Park Homestead Surveys

Perhaps the longest-term data set to track spatial distribution of non-native species comes from abandoned homesteads in the Queets River valley. These sites were caused by historical human activity and are not typical of most of the park but they are instructive regarding the effectiveness of control methods. The clearings were created beginning in 1890 (Evans 1983), were acquired by OLYM in 1940, and were vacated by 1953. In the mid-1970s, Williams (1975) surveyed the size of remaining clearings and examined species abundances in six fields (ranging in size from 4.5 to 15 ha), although not all data were collected at all plots. Later, Del Moral (1985) resurveyed five of the six fields originally surveyed by Williams. Acker et al. (2014) re-surveyed the original six between 2009 and 2013 for giant bentgrass (*Agrostis gigantea*), Canada thistle, English holly, reed canary grass, knotweeds, Himalayan blackberry, evergreen blackberry and tansy ragwort. Meanwhile, control efforts began in 2006 for Canada thistle, bull thistle (*Cirsium vulgare*), purple foxglove, English holly, evergreen blackberry and tansy ragwort.

Six species were assessed in all three surveys (Figure 4.4.16); two plant species (Canada thistle, reed canary grass) were assessed in four fields, and four plant species (English holly, Himalayan and evergreen blackberry, and tansy ragwort) were assessed in five fields. Their abundance can be coarsely described by comparing the percent of fields in which each species occurred during each sample period. Control efforts seem to have been effective for the less abundant English holly, which was effectively removed in the field where efforts were made to control it. Reed canary grass has not been treated, but it has not spread from the field where it was initially introduced. Given the great potential of reed canary grass to disrupt native ecosystems, it seems that control may be possible and especially desirable. Canada thistle and evergreen blackberry seem to be harder to control, perhaps because they are both aggressive and abundant within fields (Acker et al. 2014). Tansy ragwort was not present in the original survey but has increased dramatically despite control efforts. However, it should be controllable as it only spreads short distances (Wardle 1987).

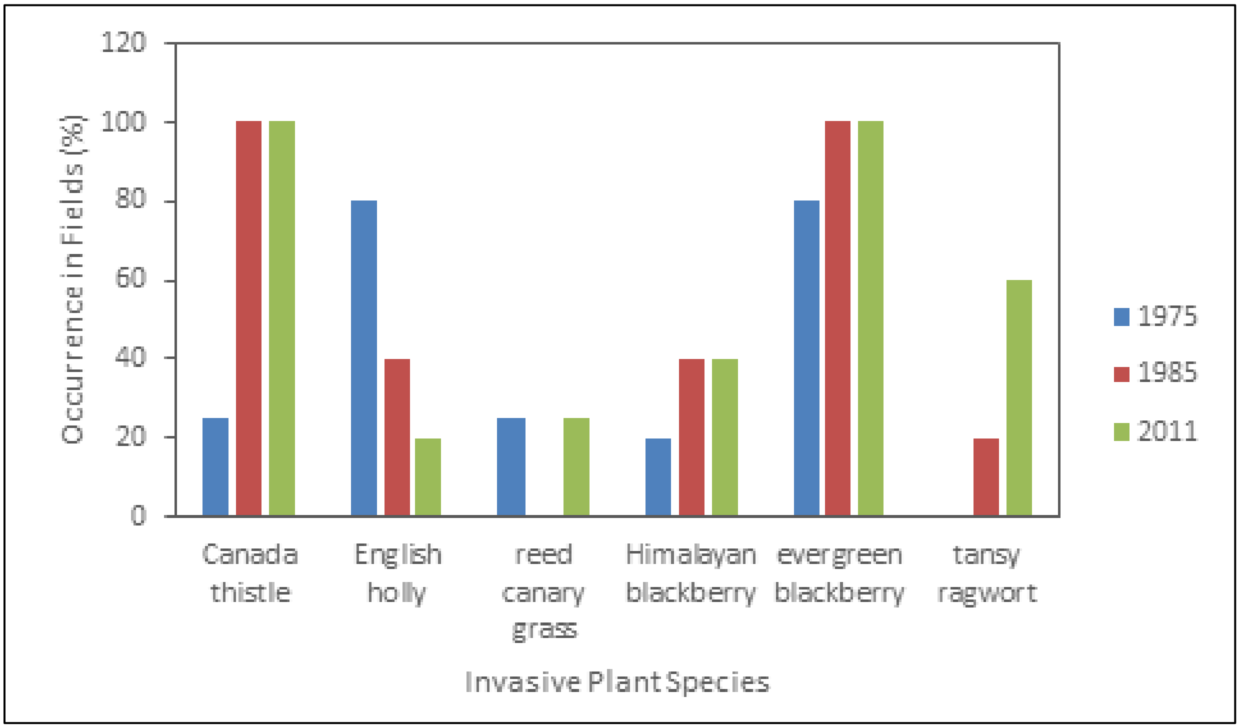

Figure 4.4.16. Occurrences of selected species in six old fields found in the Queets Valley. Data in 1985 were collected at only five of the six fields, so are the minimum number of plots with occurrence. Data Source: Del Moral (1985) and NPS.

Acker et al. (2014) surveyed near the trail upstream from the homestead fields to see how much deeper into the park the target species had spread beyond these sources. Evergreen blackberry had moved the farthest upstream and was seen more than 8 km above the last homestead. Canada thistle and Himalayan blackberry had spread approximately 5.6 km and less than 1.6 km beyond the last field, respectively. No species were found for 5 km beyond the last observation of evergreen blackberry, setting a benchmark against which to measure future spread.

Status of Elwha Reservoirs

In 2016, 254 points representing seven different invasive non-native species were recorded in the former Lake Mills reservoir (Table 4.4.8). The vast majority of the locations represented two species, Canada thistle (169 locations) and herb Robert (56 locations; Figure 4.4.17). The Canada thistle locations were well distributed in the former reservoir while herb Robert observations were mostly concentrated in the northern half of the reservoir. Most of the herb Robert locations were clustered along streams that cross roads, which serve as a source of herb Robert, before draining into the reservoir (Whiskey Bend Road to the east and the Olympic Hot Springs Road to the west): they are heavily infested with herb Robert despite past and ongoing eradication efforts. Scot's broom and common St. John's wort were mostly clustered around the former dam site. Prior to dam removal, the berms around the dam were heavily infested with these and other invasive species, notably cheatgrass. During dam removal, the contractors bulldozed the berm into the former reservoir to create access roads. These now-abandoned road beds are the source of thousands of Scot's broom seedlings each year and could be a problem for as long as the seed bank remains viable, estimated to be 80 years (Zouhar 2005). Cheatgrass seed does not have long viability (Hulbert 1955), so the absence of new plants suggests that seeds did not survive disturbance of the berm.

Table 4.4.8. Invasive species established in former Lake Mills reservoir.

Species	Common Name	GPS Points	Total Estimated Cover (m^2)
Buddleja davidii	Butterfly bush	1	Single plant
Cirsium arvense	Canada thistle	169	972
Cytisus scoparius	Scot's broom	16	25
Geranium robertianum	Herb Robert	56	417
Hypericum perforatum	Common St. John's wort	8	>8
Rubus armeniacus	Himalayan blackberry	4	6

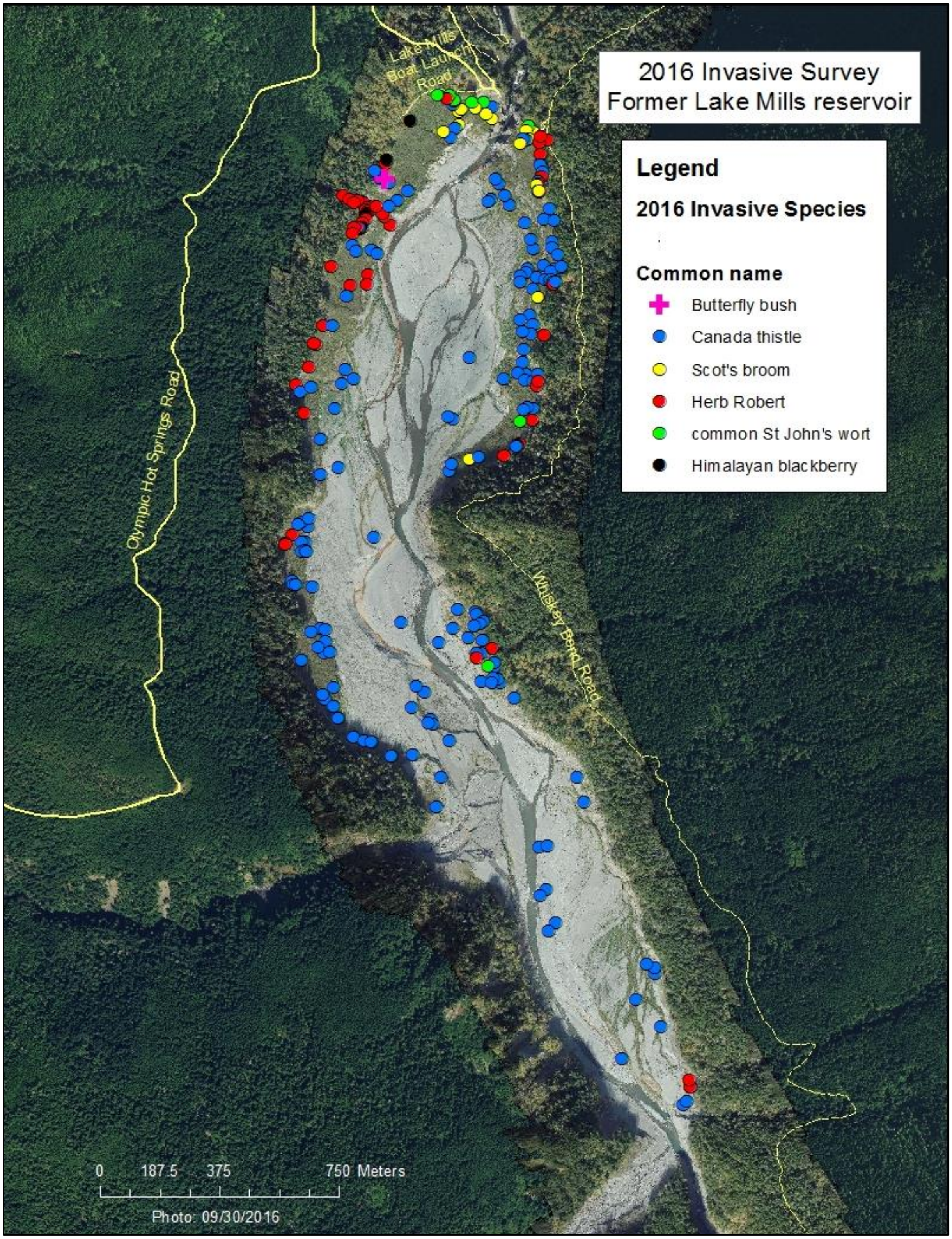

Figure 4.4.17. Distribution of invasive plant species on the former Lake Mills reservoir in 2016. Source: NPS.

A single plant of butterfly bush (*Buddleja davidii*) was located northwest of the upper reaches of the newly exposed Lake Mills reservoir in 2016 (Figure 4.4.17) and offers a unique opportunity to observe the invasion process. Butterfly bush was not known to occur in the park prior to this sighting. Although this location is far from the riparian zone typically invaded by butterfly bush in the Pacific Northwest, this species is known to invade a wide range of habitats and thrives in many disturbed environments (Tallent-Halsell and Watt 2009). It is unknown how it could have reached this location because it was far from seed sources or water courses. Moreover, wind dispersal distances are estimated to be in the tens of meters and there is no known population within that distance. The single specimen was removed in 2016, but with potential annual seed production of 100,000 to 3 million seeds (Tallent-Halsell and Watt 2009) and the possibility that the plant was present since dam removal but not detected until 2016, the plant is likely to have seeded the area and warrants continued monitoring.

Results from permanent plots in 2016 indicate that non-native species represented an average of 18% of the species richness in the former Lake Mills reservoir. Only two high-priority species were present in the permanent plots, Canada thistle (three plots, 0.01% frequency) and herb Robert (one plot, <0.01% frequency). Velvet grass (*Holcus lanatus*) became abundant in 2016 with 25% frequency but was considered to be a species of secondary concern prior to dam removal (Chenoweth et al. 2011). The abundance of velvet grass is likely influencing the relative frequency of non-native species. Since 2012, non-native species frequency has steadily increased on fine-sediment sites, where most vegetation occurs within the former Lake Mills reservoir, while native species have declined slightly (Figure 4.4.18). It is not unusual to see a decline in species richness during natural primary succession as annuals and short-lived perennials give way to a few longer-lived species such as red alder and other woody species. Presently, native woody species dominate the developing vegetation, with over 11,400 stems per acre and average tree heights over 6 m. Although the relative frequency of non-natives is gradually increasing, native forest species are still dominant and are not likely to be impacted by the frequency of species such as velvet grass and other weedy herbaceous species. Continued monitoring will indicate whether non-native species persist and/or have a legacy effect on the development of a fully functional alder or cottonwood community.

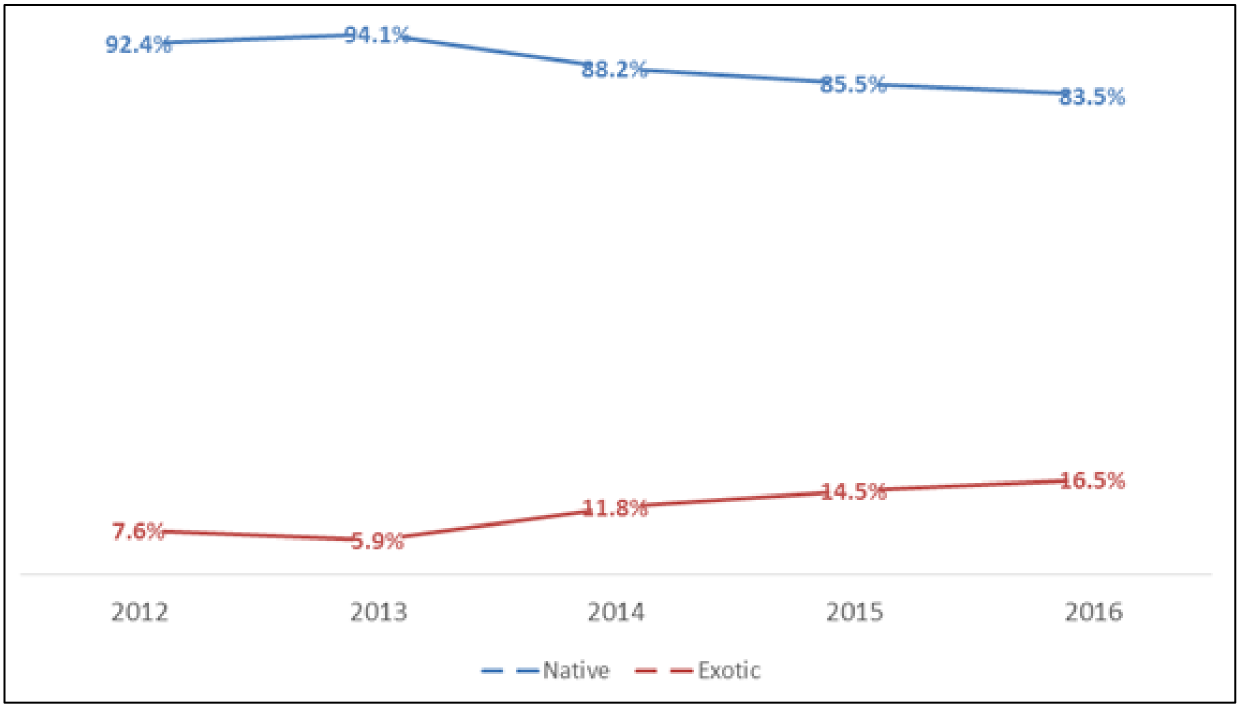

Figure 4.4.18. Plant species frequency at permanent monitoring plots in the former Lake Mills reservoir over time. Source: NPS.

Threat of Non-native Species to OLYM Resources

The primary mission of NPS is to conserve park ecosystems as unimpaired by human activities as possible. As early as 1933, NPS policy recognized the harmful effects of non-native plants and animals (Albright 1933), with perhaps the most fundamental being loss of biodiversity (Wilcove et al. 1998). There are over 200 non-native plant species in OLYM, of which 38 are thought to threaten park resources because of their aggressive displacement of native species and potential effects on ecological function (Mack et al. 2000, Chenoweth et al. 2011).

The future status of non-native plant distribution depends on changes in the threat and effectiveness of control efforts in the park and outside of park boundaries. Historic records indicate that the number of non-native plant species in the Olympic Peninsula flora has increased linearly at approximately three species per year since records began and shows no sign of abating. Limited records from inside the park indicate that there are fewer non-native species than outside the park (Figure 4.4.14). Contributing factors for this difference may include the lower abundance of disturbed habitat (including developed areas, roads, clear cuts), less hospitable habitats for non-native species at high elevations, and less comprehensive surveys due to challenging access within the park. In any case, the potential for non-native species to establish in OLYM will continue to be a management challenge.

Control efforts appear to have been successful in eliminating populations of some species and checking the spread of others in local areas (Figures 4.4.13 and 4.4.15). Elimination seems possible for small populations, even for aggressive species with the potential to spread vegetatively (e.g., Canada thistle, evergreen and Himalayan blackberry). In contrast, large, aggressive, vegetatively propagated species like the knotweeds can take extensive efforts and potentially more than a decade of work to eliminate, as was true on the Hoh River just outside of the park (Silver and Fawcett 2007). This situation occurred when one small population was not addressed for several years.

The combination of pre-dam removal treatment of invasive species, revegetation efforts using native species, and control of invasive non-native species following dam removal have resulted in the successful establishment of native vegetation on the bed of the former Lake Mills reservoir. Problem species include Scot's broom and common St. John's wort, which are primarily limited to the former dam site and could be eradicated with time and attention. Canada thistle and herb Robert are widely distributed, still have source populations, and may be difficult to control. Velvet grass was not anticipated to be as aggressive as it has proven to be.

The concerted effort to prevent or control spread of invasive species in the Lake Mills reservoir following dam removal has been successful. Most invasive species have a limited distribution and are unlikely to impair continued establishment of native vegetation. However, some species will need continued attention to be kept under control (e.g., Scot's broom). It is important to note that the park service continues to aggressively treat invasive species in the former reservoirs. Each year, a crew of at least four people combats all the major species from May through October. However, funding for a dedicated Elwha crew ended after the summer of 2017, reducing treatment efforts to 1-2 weeks per year by the regional EPMT. Extension of control efforts on the former Lake Mills reservoir would help ensure that gains made in controlling the spread of invasive species are not diminished in the future.

Uses and Limitations

This synthesis of available data on non-native plant distribution successfully quantified the threat to park resources and showed that it is not diminishing. We were also able to summarize localized repeated surveys to show the net effect of spread versus control efforts. It is useful to demonstrate effectiveness given the resources currently invested in addressing invasive plants.

Limitations to drawing conclusions from these data are primarily due to the paucity of data repeated in particular places using consistent methods or a sample design. Consequently, conclusions are imprecise and lack an assessment of confidence. Moreover, existing surveys cover a relatively small portion of the park.

Some modeling of potential habitat characteristics and spatial distribution has been conducted (Jones and Halpern 2007, Jones et al. 2010). These efforts are limited by potentially inadequate descriptions of habitat because species may not yet occupy all suitable niches. Moreover, these efforts do not consider the effects of climate change on future habitat distribution.

Emerging Issues

Prioritizing Management Efforts to Address Non-native Plants

Setting priorities for targeting control efforts is a fundamental aspect of managing non-native plant species owing to the large number and extent of the threats. Traditionally, management has focused on identifying the most dangerous species based on their characteristics (e.g., Randall et al. 2008), which has led to some species being addressed while others are accepted as part of the system. However, it is becoming more widely recognized that site characteristics and pathways are other important dimensions for prioritizing management action (Giljohann et al. 2011, McGeoch et al. 2016). For example, sites can be considered multi-dimensionally along with species characteristics and pathways. Alternatively, some sites can simply be prioritized and managed to defend against all invaders (e.g., areas above a certain elevation).

Monitoring and Managing Spread of Invasive Non-native Plants

There is potential for existing populations of invasive plants to expand in distribution from frequently disturbed areas (roads, trails, administrative areas) to eventually encompass more pristine areas of the park, including alpine meadows and sensitive riparian areas. The primary vectors of dispersal are machinery (e.g., heavy equipment and passenger vehicles), wind, water, birds, and recreational use of trails by humans and pack stock, roughly in that order of importance. At the same time, globalization and consequent international trade are having the unintended consequence of globalizing the world's flora, meaning that the influx of non-native species is not expected to slow (https://www.currentresults.com/Invasive-Species/global-708131.php, accessed 18 October 2017).

New tools are being developed to improve response to increasing establishment and spread of non-native species. A national effort was begun in 2004 by the Federal Interagency committee for the Management of Noxious and Exotic Weeds to develop and implement National Early Detection and Rapid Response System for Invasive Plants in the United States, including techniques of early detection-rapid response (Westbrooks 2004). The goal was to develop a coordinated framework of public and private partners to address new invasive plants by efficiently communicating early reports – including vouchering, verification, and assessment of new records – so that rapid response efforts can be more effective. These efforts are being enhanced by the development of cell phone "apps" and other tools to support citizen science, which enlists the general public to report sightings of invasive species (e.g., Washington Invasive Species Council, http://www.invasivespecies.wa.gov/report.shtml, accessed 18 October 2017; Early Detection and Distribution Mapping System, http://www.eddmaps.org/west, accessed 18 October 2017; What's Invasive Community Data Collection, http://www.whatsinvasive.org, accessed 18 October 2017). Information networks may be needed at regional to international spatial scales (Simpson et al. 2009). New prioritization methods are also being developed to prioritize species based not only on their probability of occurrence, but also by their effect on biodiversity, ease of detection, and ease of control to determine a surveillance strategy that minimizes expected management costs (Hauser and McCarthy 2009). Finally, new ways to treat invasive species are being explored. In one example, an innovative study found success using mulch made from English ivy to suppress herb Robert (Lintz et al. 2011).

Effects of Park Management

There is a growing recognition that routine activities by the park's facilities maintenance program may be contributing to the spread of noxious weeds within the park. Areas where materials (e.g., gravel, surplus lumber) and heavy equipment are stored are often infested with weeds, which then become established when the materials or machinery are used in an uninfested part of the park.

Enhanced vigilance of machine and storage sites and associated focused control efforts would help ensure that invasive non-native plant species are not present in these sites and inadvertently transported. Another routine park maintenance activity – mowing roadsides – can also facilitate the spread of noxious weeds if mowing occurs as existing roadside weed populations are setting seed. On the other hand, if mowing occurs as weeds are beginning to flower it can greatly reduce the volume of seed that is set.

Revegetation of Areas Where Weeds Have Been Removed

Typically, weeds are removed or sprayed and then the area they occupied is left unmanaged. As a result, these areas often become quickly re-occupied by weeds. A program to revegetate weed-treated areas with native seeds and plants, especially those that were densely weed infested, would decrease the likelihood that treated areas need repeated weed removal treatments.

Changing Ecological Relationships of Invasive Plants

While there is currently some understanding of the potential severity of ecological effects regarding individual non-native species, the ecological relationships may change with time. Due to evolutionary changes in both native and non-native species, shifts in species composition of communities (e.g., predators and competitors of invasive species may become more abundant), accumulation of materials (e.g., leaf litter or allelopathic chemicals may change community composition), and interactions with abiotic variables (e.g., invading species may affect fire regime), the perceived threat from particular non-native species may increase or decrease over time (Strayer et al. 2006, Walther et al. 2009). Consequently, time since invasion is an important dimension for assessing effects of non-native species, suggesting that prioritization of species for management action will change with time.

Effects of Climate Change

Climate change is expected to have a profound effect on invasion dynamics and on how ecosystems are affected by non-native species (Hellmann et al. 2008, Pauchard et al. 2009, Pejchar and Mooney 2009, Walther et al. 2009). Changing abiotic conditions will directly affect the distribution of non-native and native plants, thereby influencing where invasions can happen and the resistance or resilience of the native community to invasion. Climate change will also have indirect effects on plant habitats through changes in disturbance regime, dispersal dynamics, and the distribution of human activities. High-elevation environments may be especially susceptible to invasion due to changing climate (Pauchard et al. 2009). As habitats change and species migrate, the difference between migration and invasion may become blurred (Walther et al. 2009). Non-native species may eventually be needed to maintain ecosystem function as native species become maladapted. These potential effects will pose serious challenges to developing park management policy that is adapted to a changing environment.

Vulnerability of Forest Ecosystems

Invasion ecologists have long assumed that disturbance enables non-native plant invasions and that undisturbed plant communities are more resistant to invasion (Levine et al. 2004). Additionally, invasive plants are commonly fast-growing, shade-intolerant species. Together, these characteristics suggest that undisturbed forests are much less likely to be invaded than open areas. However, there is global evidence that the assumption that forests are relatively immune to invasion is less justified than previously thought (Martin et al. 2009). Several non-native species on the Olympic Peninsula are known to thrive under closed forest canopy (e.g., herb Robert, English ivy, English holly) and more are likely to arrive with time.

Interactions Between Fire and Invasive Species

A warmer climate and drier summers may result in an increase in fire frequency. Consequent disturbance of native vegetation may provide habitat for non-native plant species. Initial evidence from the Heatwave Complex of fires, which occurred in 2009 in the eastern part of the park, indicated that early and repeated treatment of invasive species is effective in preventing their establishment within burned areas (Acker 2012).

Information Needs/Gaps

Integrating, Validating, and Georeferencing

The park has many records of invasive species locations that have not been digitized. Having these data combined into a single spatial database would provide context to help determine an effective strategy for integrated management of invasive plant species. In particular, Scot's broom is a species of concern whose distribution is currently missing from compiled georeferenced records.

Many invasive species are not expected to survive in colder conditions (Millennium Ecosystem Assessment 2003). However, written reports by backcountry rangers describe six or more locations of evergreen blackberry, Canada thistle and herb Robert in the mountainous park interior. Moreover, EPMT records show invasive species (e.g., ox-eye daisy) at high elevations on the Hurricane Ridge Road. Furthermore, anecdotal reports of invasive species are accumulating, particularly for herb Robert. These reports have not been validated with surveys, georeferenced, and integrated into the non-native species database. Coordination among random reports, EPMT data, and validation surveys by park staff will maximize understanding of the state and dynamics of non-native species spread.

Monitoring Non-native Species

Frequent and comprehensive inventory and monitoring of at least front-country areas would help the park understand the extent of non-native species distribution and the effectiveness of control and prevention efforts. Furthermore, regular monitoring would provide the basis for understanding long-term consequences of non-native species on native ecosystems, as well as how climate change may interact with non-native plants to impact native communities. This effort would require resources to summarize and integrate data from monitoring and exotic plant management as well as to conduct analyses.

Resource Summary

We determined that the condition of invasive non-native plants is of moderate concern, but that they are causing a deteriorating condition of native plant communities over time (Table 4.4.9). The number of non-native plant species has been increasing in the park and across the peninsula, with opportunity for more species to invade the park. In addition, resources for the monitoring, mapping, and control of invasive species (including funding for EPMT) within OLYM are currently limited.

Table 4.4.9. Resource condition summary for invasive non-native plants and native plants in Olympic National Park.

Resource/Measure	Condition Status and Trend	Rationale
Spatial distribution of invasive non-native species	🟡	Invasive non-native speices are widely distributed on the Olympic Peninsula but are mainly found outside of the park; uncertainty is due to lack of a comprehensive survey in the park.
Rate of change of invasive species abundance in areas having repeated surveys	🟡	Control methods seem to be effective in some circumstances and seem to keep populations stable in others; the threat from outside of the park is increasing; time series of data to assess rate of change are rare and local.
Status of invasive species establishment in the de-watered reservoirs following dam removal on the Elwha River	🟡	There is not yet certainty about the effectiveness of control efforts; there is still ample opportunity for invasion, especially in areas where natives are slow to establish (e.g, coarse sediments where Scot's broom might establish)
Overall condition of Invasive non-native plants	🔴	Moderate concerns expressed for individual measures indicate that effective management actions are still possible if undertaken urgently. Invasive species pose a significant threat to park resources if left unchecked.
Overal condition and trend of Native plant	🟡⇩	The threat is increasing; condition of large areas of the park is unknown; local areas may be vulnerable due to management practices regarding roads and parking lots as well as visitor effects.

4.5. What are the ecosystem effects of airborne contaminants?

4.5.1. Air Quality and Contaminants

Tonnie Cummings, Steven Perakis, and Collin Eagles-Smith

Introduction

Air quality is an essential resource of all units of the National Park System. It affects human health and visitor enjoyment, and good air quality helps ensure the integrity of park resources and values. In addition to safeguards provided by the NPS Organic Act, the 1977 Clean Air Act amendments identified 48 NPS areas - including OLYM - as Class I air quality areas and afforded them special air quality protection. In fact, the 1977 amendments give federal land managers an "*affirmative responsibility*" to protect the air quality related values in Class I areas. Air quality related values are resources sensitive to air pollution and include visibility, lakes, streams, vegetation, soils, and wildlife.

There are many sources of air pollution; some are natural and some are anthropogenic, i.e., human-caused. The NPS focuses on reducing the impact of anthropogenic pollution on park resources. Some human sources and activities that could affect OLYM include manufacturing and industrial processes, vehicle emissions, agricultural and forestry practices, land disturbance, and prescribed fire (Cummings 2013; Figure 4.5.1). Trans-Pacific transport is also a significant source of air pollution to the west coast of North America (Yu et al. 2012).

Air pollutants of concern include sulfur and nitrogen compounds, fine particulates, ground-level ozone, and persistent bioaccumulative toxins, such as mercury. Potential effects include visibility impairment; ozone-induced human health problems and damage to vegetation; aquatic and terrestrial acidification and eutrophication; and neurological, respiratory, and other health issues associated with exposure to toxins. To better understand and protect air quality, the NPS and collaborators have monitored air quality and air quality related values at OLYM since 1980 (Figure 4.5.2). This work has been described in detail in Eilers et al. (1994), Air Resource Specialists (2003), and Cummings (2013). Below we introduce the key air quality and contaminant measures covered in this assessment.

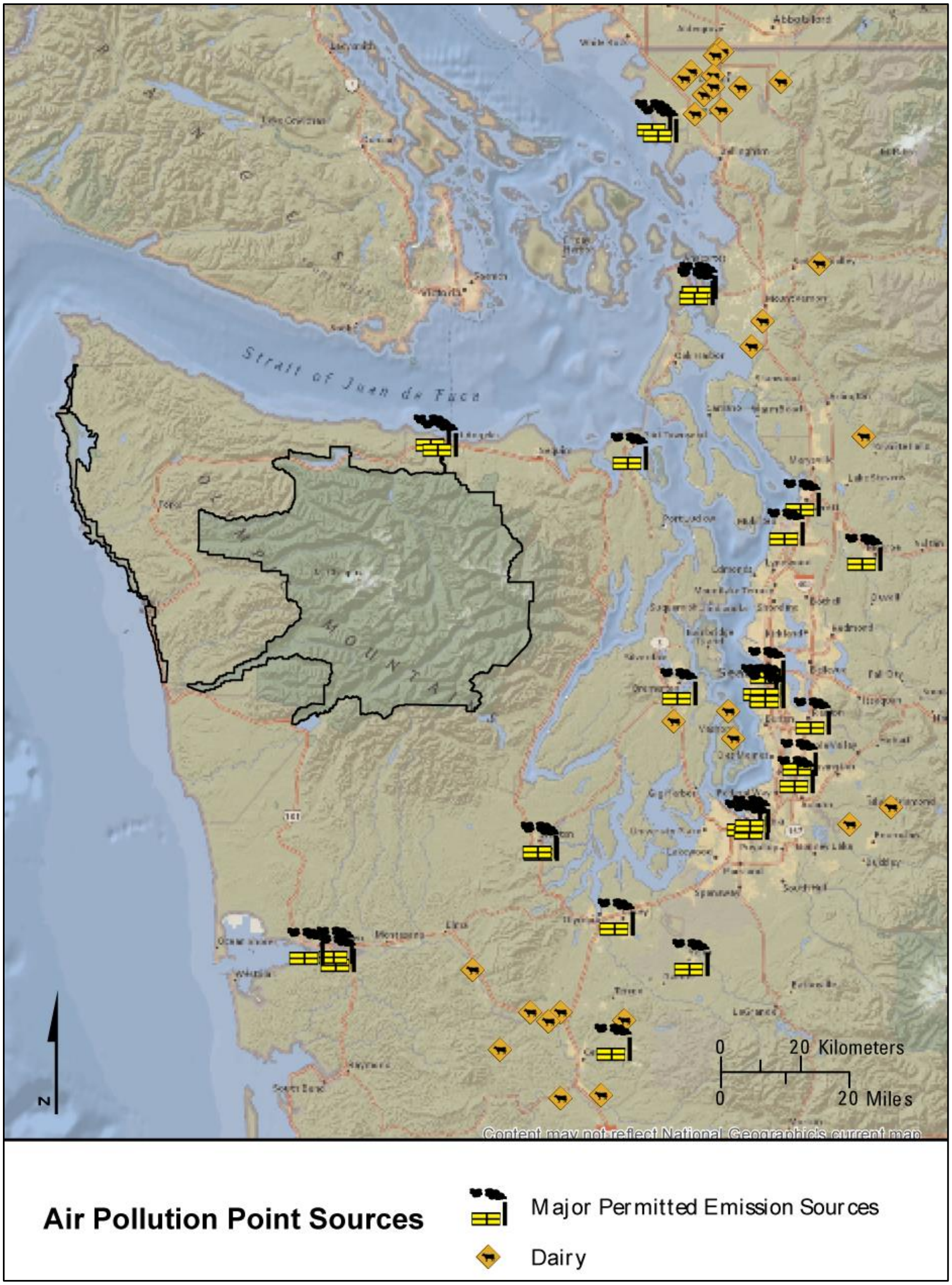

Figure 4.5.1. Locations of some air pollution sources near Olympic National Park. Major permitted emission sources emit greater than 100 tons per year of sulfur dioxide, nitrogen oxides, particulate matter, or volatile organic compounds. Source: NPS.

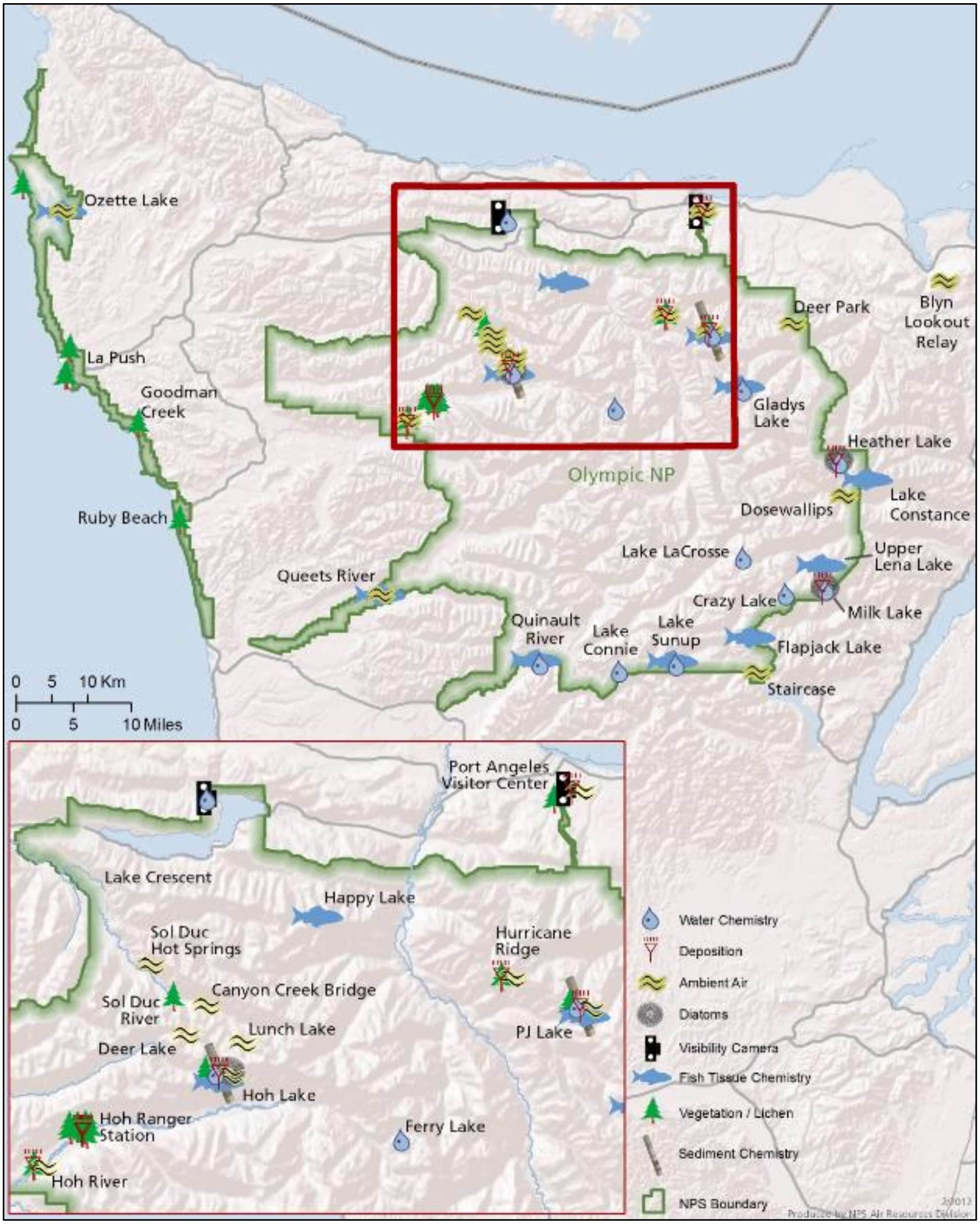

Figure 4.5.2. Locations of some of the air quality-related monitoring and research conducted at Olympic National Park from 1980-2012. Source: Cummings 2013.

Visibility

Among the experiences that visitors to national parks treasure is the breathtaking scenery – majestic mountains contrasted against a pure blue sky or a spectacular array of stars at night. In a survey conducted at OLYM in 2000, visitors ranked scenic views as one of the most important park resources to protect (Kulesza et al. 2013). Visibility-impairing particles include anthropogenic pollutants as well as natural compounds like soil and sea salt aerosols. Fine particles in the atmosphere absorb or scatter light, causing haze, reducing visibility, and degrading scenic views (Hand et al. 2011). Fine particles are also a significant concern for human health because they lodge deep in the lungs and can cause respiratory problems (Dockery 2009).

The 1977 Clean Air Act amendments set a specific goal for visibility protection in Class I areas: "*the prevention of any future, and the remedying of any existing, impairment of visibility in mandatory Class I federal areas which impairment results from manmade air pollution*" (42 U.S. Code § 7491). To further the visibility goal, in 1999, the U.S. Environmental Protection Agency (USEPA) promulgated the Regional Haze Rule, which requires each state to develop a plan to improve visibility in Class I areas, with the goal of returning visibility to natural conditions, i.e., no human-caused impairment, by 2064. The Regional Haze Rule requires improved visibility on the haziest days, with no degradation on the clearest days. If a state does not make reasonable progress in achieving visibility improvement, it may be required to implement additional pollution reduction strategies.

Visibility has been monitored at OLYM since 1980 (Cummings 2013). Photographs (taken since 1980) provide qualitative documentation of visibility conditions (Figure 4.5.3). Atmospheric particle monitoring conducted as part of the national Interagency Monitoring of Protected Visual Environments Program began in 2001. The particle monitoring allows for identification of the chemical species and sources of visibility impairment in the park, and it is used to document long-term visibility trends including progress in meeting Regional Haze Rule goals.

Figure 4.5.3. Representative photographs of clear and hazy days at Lake Crescent in Olympic National Park. Standard visual range in the picture on the top is about 212 km while standard visual range in the picture on the bottom is about 85 km. Source: from the Air Quality Web Cameras website archive for OLYM, NPS 2016a.

Ozone

Ozone is a respiratory irritant that can trigger a variety of human health problems including chest pain, coughing, throat irritation, and congestion. Ozone also affects vegetation, causing significant harm to sensitive plant species (USEPA 2014). Ozone enters plants through leaf openings called stomata and oxidizes plant tissue, causing visible injury (e.g., stipple and chlorosis) and growth effects (e.g., premature leaf loss; reduced photosynthesis; and reduced leaf, root, and total size).

Over the years, ozone data have been collected at many locations in OLYM (Cummings 2013). Ozone concentrations were measured with passive samplers at six sites in the park during the ozone season (approximately May-September) from 1995-2003. Year-round monitoring was conducted at three locations from1981-2005 and portable monitors were used during the ozone season at two sites (2004-2011). Ozone monitoring was discontinued at OLYM in 2011 due to consistently low values. The closest currently operating monitor is at Cheeka Peak, approximately five km north of OLYM, and it is considered a good surrogate for the park.

Nitrogen and Sulfur Deposition

Airborne pollutants are eventually deposited through either wet deposition (i.e., rain, snow, clouds, and fog) or dry deposition (i.e., particles and gases) onto vegetation, soils, streams, and lakes. Sulfur and nitrogen deposition can have a significant effect on natural systems, and nitrogen is of particular concern in the western U.S. where many ecosystems are naturally nitrogen-limited. Over time, excess nitrogen deposition alters biodiversity and plant and soil chemistry, with cascading effects through ecosystems (Cummings et al. 2014). Excess nitrogen deposition also leads to increased nitrate leaching to water bodies, where it can cause eutrophication, acidification, or dead zones.

The NPS, other land managers, and the USEPA use critical loads to determine the threshold for ecosystem sensitivity to atmospheric deposition. A critical load is technically defined as "*the quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment are not expected to occur according to present knowledge*" (Nilsson and Grennfelt 1988). Critical loads are typically expressed in terms of kilograms per hectare per year (kg ha^{-1} yr^{-1}) of wet or total (wet plus dry) deposition. Critical loads can be developed for a variety of ecosystem responses, including shifts in aquatic plankton or terrestrial lichen and plant species, changes in soil chemistry, and lake and stream acidification. In general, as deposition increases, additional resources are affected and ecological effects become more pronounced (Cummings et al. 2014; Figure 4.5.4). The goal of the NPS is to limit nitrogen and sulfur deposition to levels that do not exceed the minimum critical load for a park's most sensitive resources. In the Pacific Northwest, critical loads have been developed for nitrogen deposition, but not for sulfur deposition.

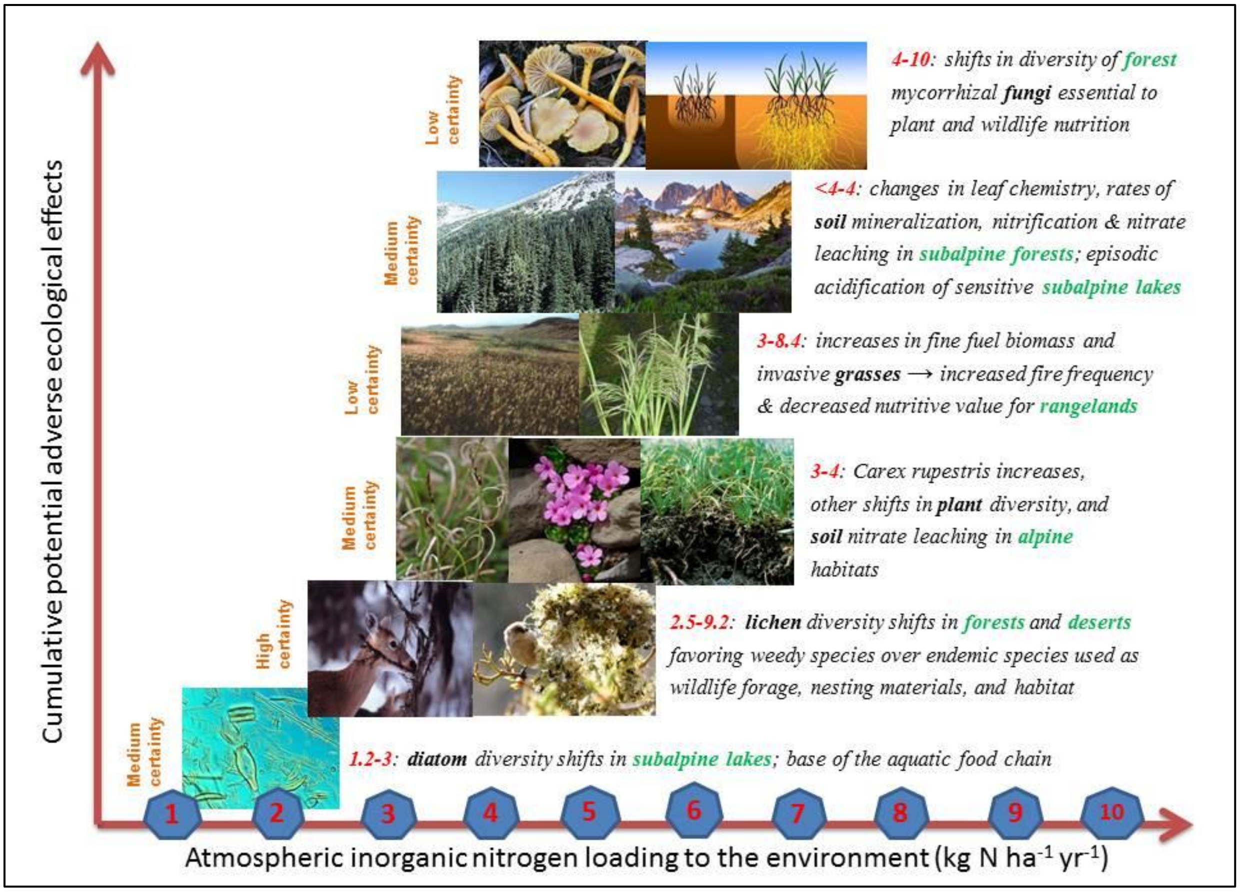

Figure 4.5.4. Cumulative potential adverse ecological effects associated with atmospheric nitrogen deposition in the Pacific Northwest. The reliability assessments are as follows: High Certainty when a number of published papers of various studies show comparable results, Medium Certainty when the results of some studies are comparable, and Low Certainty when very few or no data are available in the Pacific Northwest so the applicability is based on expert judgment. Source: Cummings et al. 2014.

Wet deposition of nitrogen (only inorganic species) and sulfur has been monitored at OLYM since 1980 through the National Atmospheric Deposition Program (NADP). The NADP measures concentrations and calculates annual wet deposition of pollutants. Researchers have, and are, investigating the effects of nitrogen deposition on lichens, alpine lake diatoms, and alpine vegetation at OLYM to determine critical loads.

Mercury

Mercury is a globally pervasive, highly toxic pollutant that is broadly distributed to even remote environments through wet and dry atmospheric deposition (Krabbenhoft and Sunderland 2013). Mercury is also a naturally occurring component of soils in geologically enriched areas that are associated with mercury-containing ore bodies. In aquatic ecosystems, inorganic mercury can be microbially converted to its most bioaccumulative and toxic form, methylmercury (Morel et al. 1998). Fossil fuel combustion is the primary source of "new" mercury releases to the environment (Agnan et al. 2016). Only approximately 20% of anthropogenic mercury emissions occur in the western half of North America (Eagles-Smith et al. 2016b), yet the Pacific Coast Range and Olympic

Peninsula receive some of the highest amounts of atmospheric mercury deposition in the Continent (Figure 4.5.5). This is largely a function of the high precipitation rates, but also because of trans-Pacific transport of atmospheric mercury released in other parts of the world (Weiss-Penzias et al. 2016). Atmospheric mercury is also taken up and sequestered in plant tissue, and it is ultimately stored in organic material of soils. The high productivity rate of Northwestern forests makes them some of the largest pools of inorganic mercury in the western United States (Obrist et al. 2016).

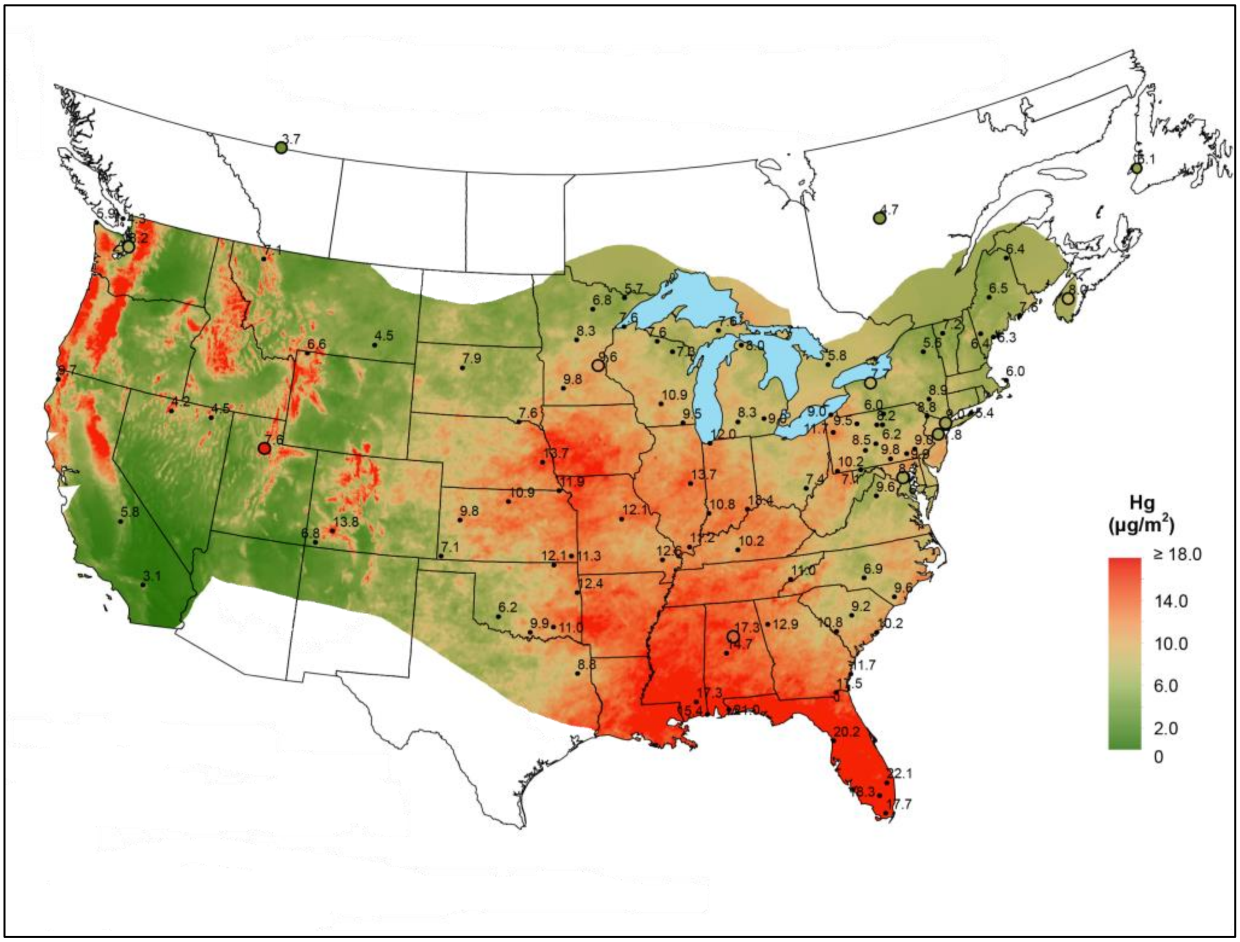

Figure 4.5.5. Mercury wet deposition estimates for the continental U.S. during 2014. Estimates are from NADP Mercury Deposition Network data (NADP 2016).

As mercury bioaccumulates through food webs it can reach concentrations in higher-order consumers that are associated with endocrine disruption (Tan et al. 2009), behavioral effects (Burke et al. 2010), and impaired reproduction (Scheuhammer and Sandheinrich 2008). Ecological risk is generally highest in top trophic level organisms, such as fish-eating birds (Wiener et al. 2003), but recent evidence suggests that invertivorous wildlife such as songbirds and bats may also be at risk (Yates et al. 2014, Jackson et al. 2015). Human exposure to mercury is primarily through fish consumption, thus sites with higher fish mercury concentrations pose a greater risk to human health. Methylmercury production and entry into the food web is largely governed by ecological processes and site-specific biogeochemical conditions of aquatic ecosystems. OLYM contains a diversity of aquatic habitats, climatological gradients, as well as variation in mercury deposition across the park. These factors have created a mosaic of mercury sensitive habitats that likely result in substantial variation in mercury exposure throughout the park.

Concern associated with the potential risks of trans-Pacific atmospheric mercury deposition, coupled with the measurement of fish with elevated mercury concentrations in some OLYM water bodies, has led to focused research in the past 5 years on mercury distribution throughout the park. Researchers have examined the variability and drivers of fish mercury concentrations in lakes (Eagles-Smith et al. 2014), the influence of food web structure on mercury bioaccumulation in salamanders (Pfleeger 2015), and songbird exposure to mercury near water bodies throughout the park. Additionally, OLYM has served as one of two intensive parks for the Dragonfly Mercury Project, a citizen science initiative to examine mercury contamination in national parks throughout the country using aquatic dragonfly larvae as bioindicators (Eagles-Smith et al. 2016a).

Other Persistent Bioaccumulative Toxins

In addition to mercury, persistent bioaccumulative toxins include other heavy metals, current and historic use pesticides, industrial chemicals, and by-products of fuel combustion. Concerns mainly pertain to impacts on humans and wildlife. Effects vary with the type of pollutant, but include declines in reproductive success, growth, and neurological function, and increased disease susceptibility (Landers et al. 2008).

While there have been a few studies of non-mercury persistent bioaccumulative toxins at OLYM that involved limited sampling (Cummings 2013), a significant effort was undertaken at the park as part of the intensive Western Airborne Contaminants Assessment Project (WACAP). The objective of WACAP was to determine the risk from airborne contaminants to ecosystems and food webs in 20 national parks in the western U.S. (Landers et al. 2008).

Approach and Methods

Visibility, Ozone, and Atmospheric Deposition

This air quality assessment used the methods developed by the NPS Air Resources Division (ARD) for a consistent Servicewide approach to evaluating conditions and trends in visibility, ozone, and atmospheric deposition at NPS units throughout the continental U.S. (NPS 2017). In brief, data collected by federal, state, and local monitoring networks are evaluated with an Inverse Distance Weighted (IDW) interpolation method to estimate air quality conditions for parks. For the IDW method, values are derived from several monitors in the area, with data from the closest stations to a park given greater weight than data from stations farther away. The estimates are based on 5-year averages, and the values in each park are compared to ARD benchmarks for specific measures of ozone, visibility, and atmospheric deposition (Table 4.5.1). Benchmarks are established based on regulatory standards, natural visibility goals, and ecological thresholds, and are described in the Reference Conditions section below.

Table 4.5.1. Indicators and specific measures for air quality condition assessments. Source: based on NPS 2017.

Indicator	Specific Measure
Visibility	Visibility on mid-range days minus natural visibility condition on mid-range days (in deciviews)
Ozone	Human health: 4th-highest daily maximum 8-hour concentration (in parts per billion); Vegetation health: 3-month maximum 12-hour W126[1] (in parts per million-hours)
Atmospheric Deposition	Sulfur wet deposition (in kilograms per hectare per year) Nitrogen wet deposition (in kilograms per hectare per year)

[1] The W126 is based on a cumulative sum of hourly ozone concentrations during a rolling 3-month period, where the hourly values are weighted according to their magnitude.

The ARD's Air Quality Condition and Trends website (NPS 2016b) provides information on visibility, ozone, and nitrogen and sulfur deposition for OLYM. The website is updated every few years based on recent monitoring data. The ARD calculates short-term trends from data collected over a 10-year period at on-site or nearby representative monitors. Short-term trends are calculated for sites that have at least six years of annual data and an annual value for the end year of the reporting period. A non-parametric regression technique called the Kendall-Theil method is used to determine statistically significant trends ($p \leq 0.10$). This condition assessment used 2009-2013 interpolated data from on-site and off-site monitors. The short-term trends relied on 2004-2013 data. Visibility, nitrogen deposition, and sulfur deposition trends were based on data collected in the park. Ozone trends were based on data from the nearby Cheeka Peak site.

The evaluation of nitrogen critical loads for OLYM used the results from ARD's Critical Loads and Estimated Exceedances website (NPS 2016c). The methods followed the approach described in Pardo et al. (2011), which recommended a range of critical load values for each of the Level 1 ecoregions identified in the ecosystem classification system developed through the Commission for Environmental Cooperation for North America (CEC 1997). OLYM encompasses both the Northwestern Forested Mountains and the Marine West Coast Forests ecoregions, and nitrogen critical loads have been identified for one or more of the following components in each of those ecoregions: forests (i.e., trees and soils), herbaceous plants and shrubs, lichen and bryophytes, mycorrhizal fungi, and nitrate leaching. As noted previously, critical loads for sulfur have not yet been developed for OLYM because of more pressing concern over nitrogen deposition. Critical loads were compared to 2010-2012 3-year average total (i.e., NADP monitored wet plus modeled dry) nitrogen deposition to identify possible exceedances. An exceedance suggests increased potential of ecological harm.

Mercury

Qualitative risk classifications for fish and birds were assigned using mercury concentrations in both potential diet items (dragonfly larvae), and fish (muscle) and bird (blood) tissues from various locations throughout OLYM. Dragonfly larvae were collected as part of a national-level citizen science study evaluating dragonfly larvae as bioindicators of mercury contamination in national parks (Eagles-Smith et al. 2016a). Potential risk to humans associated with fish consumption was estimated using mercury concentrations in sport fish from various lakes in the park. The status of mercury contamination in biological resources of OLYM was assessed using existing data on mercury concentrations in dragonfly larvae (N = 653 samples from 12 locations), fish (N = 319 samples from 14 locations), and songbirds (N = 146 samples from 11 locations).

Other Persistent Bioaccumulative Toxins

As part of WACAP, researchers used passive air sampling devices, along with snow, conifer needle, lichen, surface water, sediment, and fish tissue samples, to assess persistent bioaccumulative toxins in the Hoh and PJ Lake watersheds at OLYM. Concentrations of persistent bioaccumulative toxins in biota were compared to literature-based human and wildlife health thresholds.

Reference Conditions

Visibility

Visibility conditions and trends are expressed in terms of a haze index which correlates incremental changes in haziness to corresponding changes in perceived visibility. The haze index is reported in deciviews (dv). The dv scale is near zero for a pristine atmosphere and increases as visibility degrades.

The ARD’s condition assessments are based on estimated average visibility on mid-range days (40th to 60th percentile) minus the estimated natural visibility on mid-range days (NPS 2017). The estimated value is compared to ARD benchmarks (Table 4.5.2). The difference between estimated current conditions and estimated natural visibility represents the human contribution to visibility impairment. The ARD computes visibility trends from the haze index values on the 20% haziest days and the 20% clearest days. The Clean Air Act visibility goal requires visibility improvement on the 20% haziest days, with no degradation on the 20% clearest days.

Table 4.5.2. Benchmarks for visibility condition. Source: NPS 2017.

Category	Visibility (dv)
Warrants significant concern	>8
Warrants moderate concern	2-8
Resource is in good condition	<2

Ozone

The ARD’s condition assessments for human health risk from ozone are directly related to the EPA’s primary National Ambient Air Quality Standard of a 4th-highest daily maximum 8-hour ozone concentration of 70 parts per billion (ppb; NPS 2017). The maximum estimated ozone concentration at a park is compared against ARD benchmarks (Table 4.5.3).

Table 4.5.3. Benchmarks for human health condition for ozone. Source: NPS 2017.

Category	Ozone concentration[1] (ppb)
Warrants significant concern	≥71
Warrants moderate concern	55-70
Resource is in good condition	≤54

[1] Estimated or measured 5-year average of annual 4th-highest daily maximum 8-hour concentration.

Although the primary National Ambient Air Quality Standard is not a good predictor of vegetation response to ozone, the USEPA has not set a secondary standard that focuses on vegetation. However, in its recent policy assessment of the ozone standards, the USEPA discussed use of the W126 to assess plant response (USEPA 2014). The W126 preferentially weights the higher ozone concentrations most likely to affect plants and sums all of the weighted concentrations during daylight hours. The highest 3-month period that occurs during the growing season is reported in parts per million-hours (ppm-hrs). Based on the information from the USEPA, research indicates for a W126 value of:

- ≤7 ppm-hrs, tree seedling biomass loss is ≤2 % per year in sensitive species; and
- ≥13 ppm-hrs, tree seedling biomass loss is 4–10 % per year in sensitive species.

The ARD compares maximum calculated W126 values at a park to benchmarks tied to the research results to assess vegetation condition related to ozone (NPS 2017, Table 4.5.4).

Table 4.5.4. Benchmarks for vegetation condition for ozone. Source: NPS 2017.

Category	Ozone Concentration[1]* (ppm-hrs)
Warrants significant concern	>13
Warrants moderate concern	7-13
Resource is in good condition	<7

[1] Estimated or measured 5-year average of the maximum 3-month 12-hour W126 concentration.

Nitrogen and Sulfur Deposition

The ARD's condition assessments for nitrogen and sulfur deposition are based on wet deposition only, rather than total deposition, because the evaluation relies on data collected through the 250-plus NADP monitoring sites. Wet deposition is calculated by multiplying nitrogen or sulfur concentrations in precipitation by normalized precipitation amounts (NPS 2017). A park's maximum calculated deposition is then compared to benchmarks based on the results of studies that related the amount of atmospheric deposition to aquatic ecosystem health (Table 4.5.5). If a park is considered very highly sensitive to acidification or nitrogen nutrient enrichment relative to other Inventory and Monitoring parks, the condition is adjusted to the next worse condition category.

Table 4.5.5. Benchmarks for nitrogen and sulfur deposition condition. Data source: NPS 2017.

Category	Deposition (kilograms hectare^{-1} year^{-1})
Warrants significant concern	>3
Warrants moderate concern	1-3
Resource is in good condition	<1

The ARD bases wet deposition trends on concentrations of nitrogen or sulfur pollutants in precipitation, rather than on deposition amounts of pollutants, so that yearly variations in precipitation amount do not influence trend analyses (NPS 2017). Sulfur wet deposition trends rely on precipitation sulfate concentrations. Nitrogen wet deposition trends use the sum of the molecular weight ratios of nitrate and ammonium concentrations in precipitation.

According to Pardo et al. (2011), the minimum nitrogen critical loads for Northwestern Forested Mountains ecological components range from 2.5-5.0 kg ha^{-1} yr^{-1} and the range for the Marine West Coast Forests is 2.7-5.0 kg ha^{-1} yr^{-1} (Table 4.5.6). Deposition above these levels indicates potential harm to nitrogen-sensitive park resources.

Table 4.5.6. Minimum nitrogen critical loads (in kg ha^{-1} yr^{-1}) for five ecosystem components in the Northwestern Forested Mountains and Marine West Coast Forests ecoregions. Data source: based on Pardo et al. 2011.

Ecoregion	Forests[1]	Herbaceous Plants and Shrubs	Lichens and Bryophytes	Mycorrhizal Fungi	Nitrate Leaching
Northwestern Forested Mountains	4.0	4.0	2.5	5.0	4.0
Marine West Coast Forests	5.0	N/A	2.7	5.0	N/A

[1] Trees and soils.

Mercury

Risk potential associated with mercury exposure was estimated using literature-based benchmarks for possible effects to fish and wildlife. Estimated impairment severity indices and the associated mercury concentration benchmarks are provided in Table 4.5.7. Fish tissue (muscle) benchmarks are derived from Lepak et al. (2016) and are associated with biochemical/gene expression (low severity), behavior/reproduction/histology (moderate severity), and growth (high severity) effects. Benchmarks associated with mercury concentrations in fish diet were derived by back-calculating the dietary mercury concentrations associated with fish muscle benchmarks using a literature-derived biomagnification factor estimate (Lavoie et al. 2013). Bird tissue and dietary benchmarks are derived from risk estimates summarized in Ackerman et al. (2016), and are associated with oxidative stress and altered gene expression (low severity), behavioral impairment and reduced reproductive success (moderate severity), increased probability of impaired reproduction (high severity), and reproductive failure, decreased survival (extreme severity). Severity benchmarks are derived from the Great Lakes Fish Advisory Workgroup (2007), and are based upon workgroup's consumption advice. The 'below likely impairment' benchmark is associated with unlimited consumption; low severity equates to a recommendation of up to 2 fish meals per week; moderate severity equates to a recommendation of up to 1 fish meal per week; high severity equates to a recommendation of up to 1 fish meals per month; extreme severity equates to a recommendation of no fish consumption. It is important to note that these human consumption guidelines are not associated with official state or federal agency guidelines and differ from benchmarks applied by the Washington Department of Health.

Table 4.5.7. Benchmarks (nanograms/gram wet weight) for estimated impairment severity in fish, wildlife (birds), and humans.

Estimated Impairment Severity	Fish		Wildlife (birds)		Human
	Diet	**Tissue (muscle)**	**Diet**	**Tissue (blood)**	**Sport fish fillet**
Below likely impairment	<25	<200	<100	<200	<50
Low	25–40	200–300	100–180	200–1,000	50-110
Moderate	40–120	300–1,000	180–300	1,000–3,000	110–220
High	>120	>1,000	300–400	3,000–4,000	220–950
Extreme	N/A	N/A	>400	>4,000	>950

Other Persistent Bioaccumulative Toxins

For WACAP, Landers et al. (2008) evaluated the risk to humans and piscivorous wildlife from eating fish contaminated with persistent bioaccumulative toxins. Unfortunately, information about harmful levels of those contaminants is limited to a handful of chemicals and species. For humans, health thresholds were calculated for subsistence and recreational fishers for 13 non-mercury persistent bioaccumulative toxins. Fish tissue concentrations of four non-mercury contaminants were compared to health thresholds for three wildlife species (Table 4.5.8).

Table 4.5.8. Fish contaminant health thresholds for piscivorous wildlife (in nanograms/gram wet weight. Data source: Landers et al. 2008.

Pollutant	Belted Kingfisher (*Ceryle alcyon*)	Mink (*Mustela vison*)	River Otter (*Lontra canadensis*)
Total PCBs[1]	440	130	180
Total DDTs[2]	20	360	490
Total Chlordanes	4.5	830	1,140
Dieldrin	360	20	30

[1] polychlorinated biphenyls

[2] dichlorodiphenyltrichloroethane

Results and Assessment

Visibility

Estimated average visibility on mid-range days at OLYM was 10.5 dv. Subtracting the park's estimated natural visibility of 5.1 dv on mid-range days, the assumed contribution from human-caused haze was 5.4 dv. Compared to ARD's benchmarks, visibility at OLYM warranted moderate concern. The 10-year trends showed a statistically significant improvement in visibility on both the 20% haziest and the 20% clearest days (Figure 4.5.6), indicating that for the 2004-2013 timeframe, the park was meeting Regional Haze Rule requirements for improvement on the worst days and no degradation on the best days.

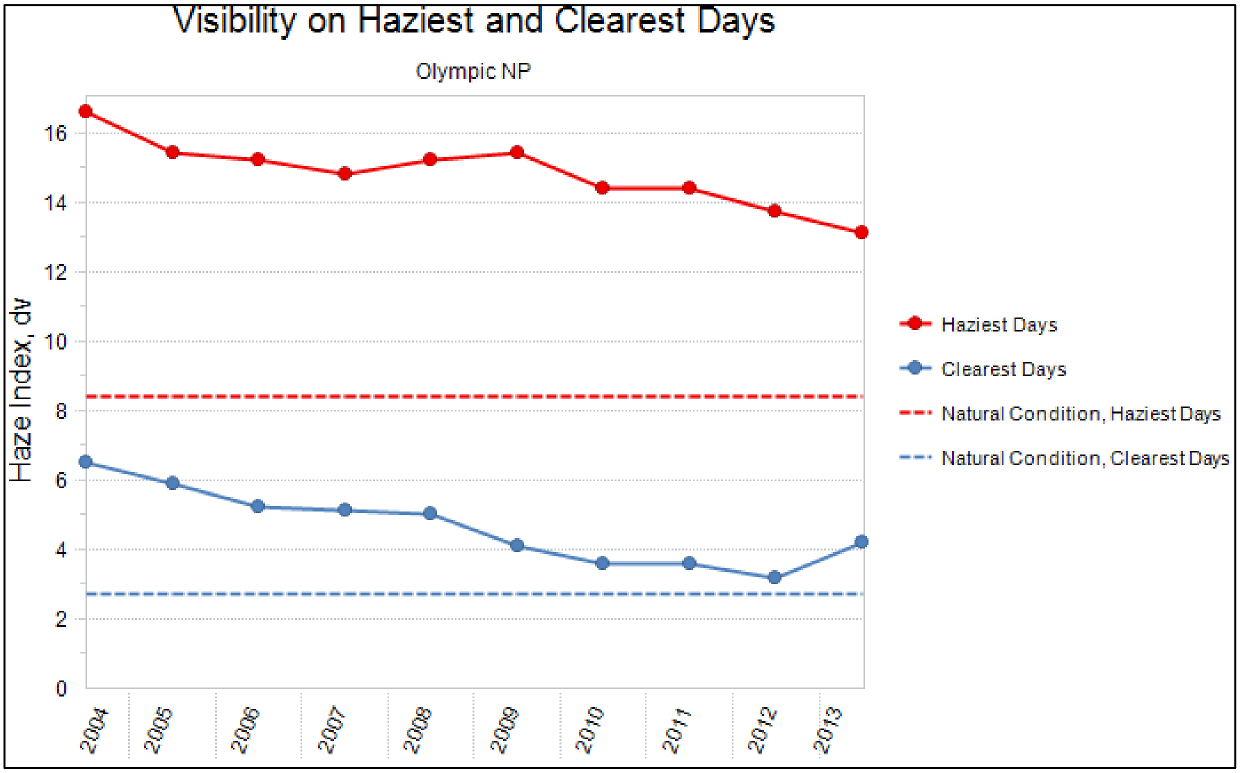

Figure 4.5.6. 2004-2013 visibility trends at Olympic National Park. Source: NPS 2016b.

Ozone

The 4th-highest daily maximum 8-hour ozone concentration for OLYM was 54.2 ppb, which is well below the primary National Ambient Air Quality Standard of 70 ppb. The maximum 3-month 12-hour W126 was 1.6 ppm-hrs, which is much lower than levels known to harm vegetation, i.e., 7-13 ppm-hrs. Compared to ARD benchmarks for ozone, human health and vegetation were in good condition. For 2004-2013, there were no statistically significant short-term trends in ozone at the park (Figures 4.5.7 and 4.5.8).

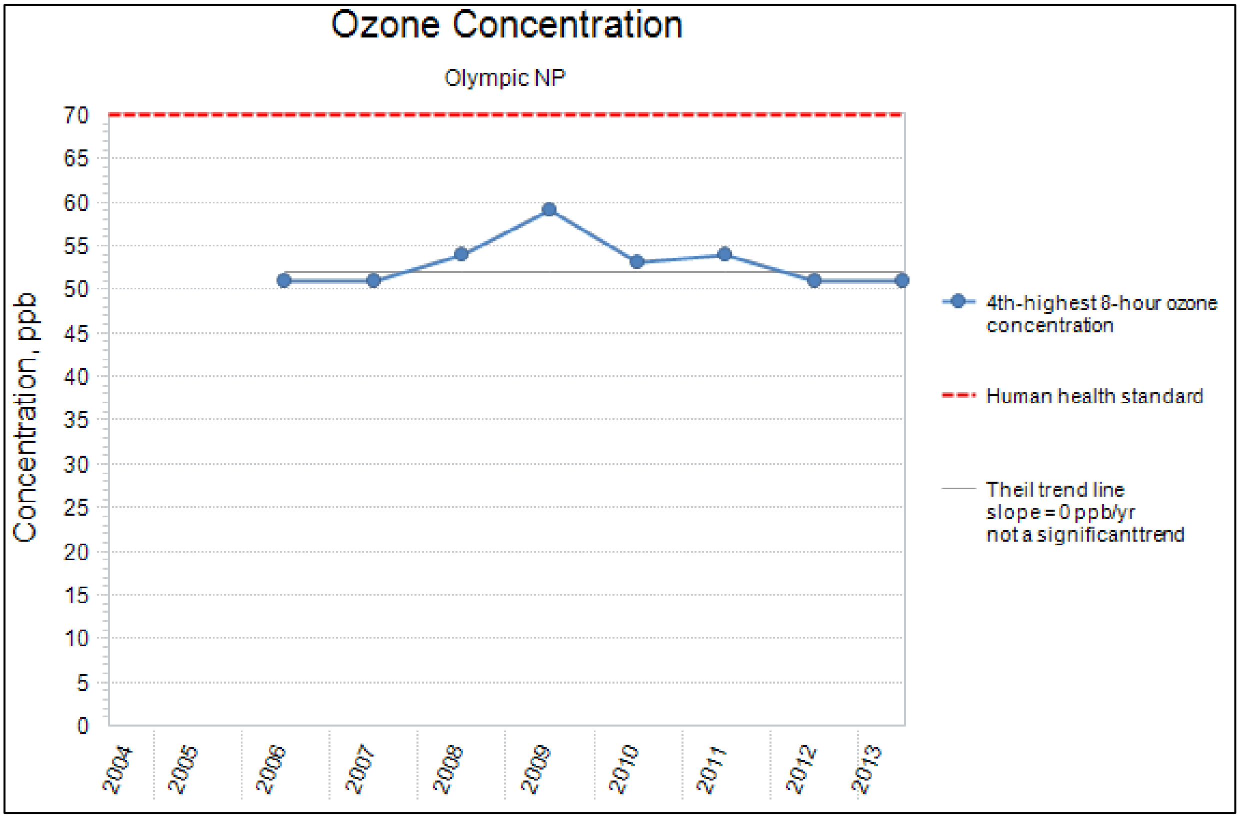

Figure 4.5.7. 2004-2013 4th-highest daily maximum 8-hour ozone trends at Olympic National Park. Source: NPS 2016b. Site was established in 2006.

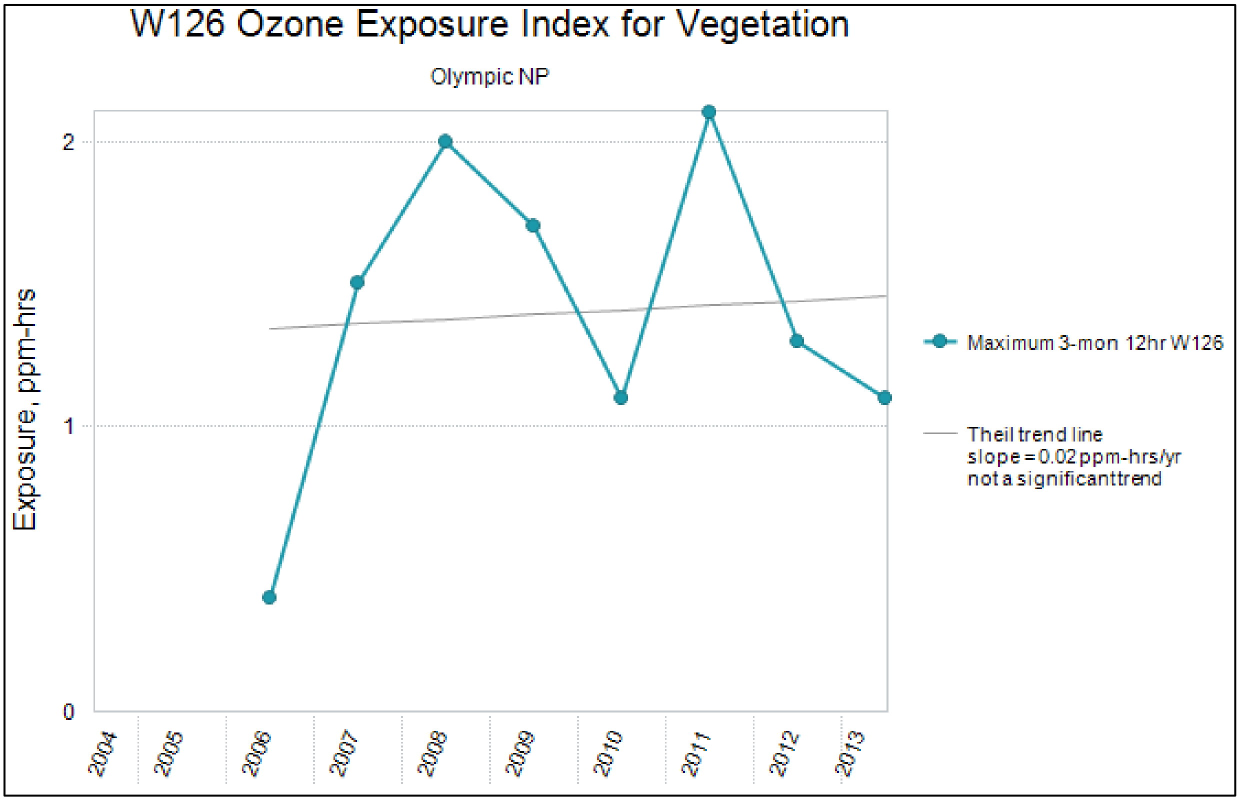

Figure 4.5.8. 2004-2013 maximum 3-month 12-hour W126 ozone trends at Olympic National Park. Source: NPS 2016b. Site was established in 2006.

Nitrogen and Sulfur Deposition

Estimated wet nitrogen deposition at OLYM was 2.8 kg ha^{-1} yr^{-1}. Compared to ARD deposition benchmarks, this level normally warrants moderate concern. However, because OLYM's ecosystems may be very highly sensitive to nitrogen nutrient enrichment relative to other Inventory and Monitoring parks (Sullivan et al. 2011a, b), the condition was elevated to significant concern. Estimated wet sulfur deposition at the park was 5.5 kg ha^{-1} yr^{-1}, a level that warrants significant concern when compared to ARD deposition benchmarks. In addition, OLYM was ranked as being very highly sensitive to acidification relative to other Inventory and Monitoring parks (Sullivan et al. 2011c, d).

While there was no significant short-term trend in wet nitrogen concentration in precipitation at OLYM (Figure 4.5.9), there was a statistically significant increase in sulfur concentration from 2004-2013 (Figure 4.5.10). Precipitation sodium and chloride concentrations are closely correlated with sulfate concentrations, suggesting marine sources influence sulfur trends at OLYM (NADP 2016).

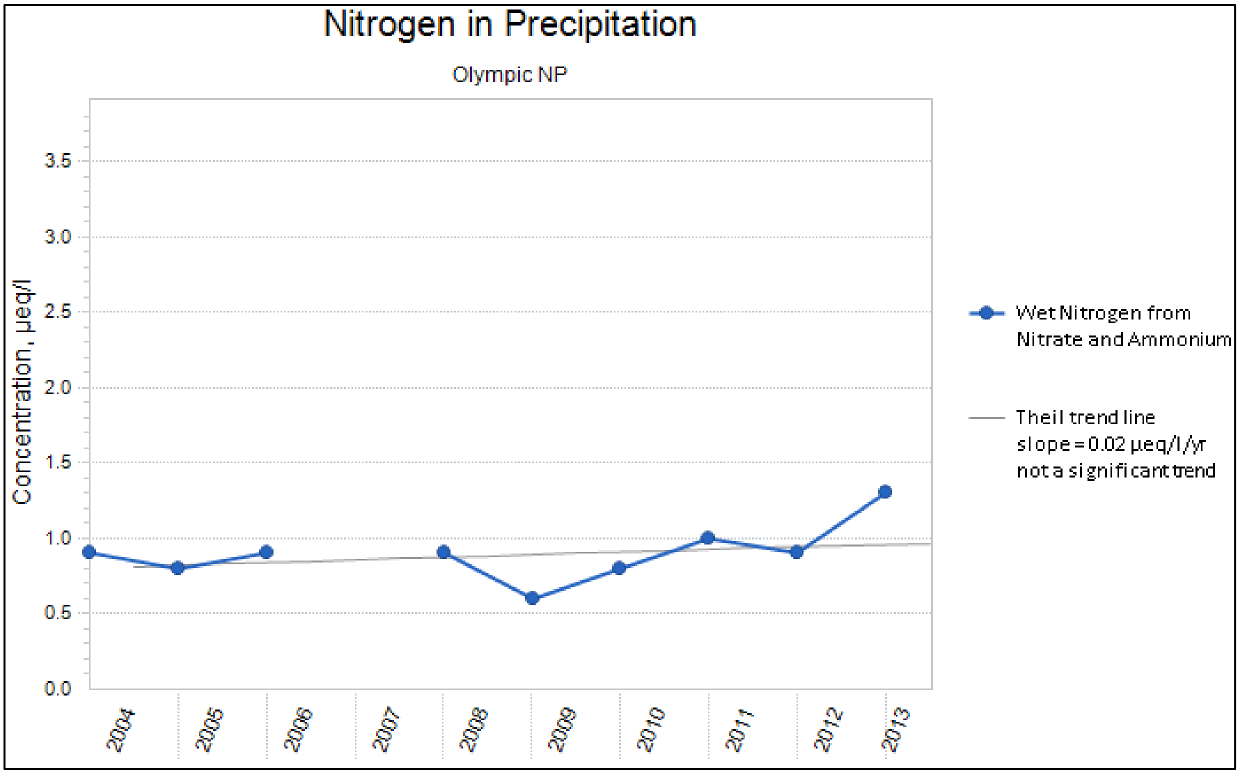

Figure 4.5.9. 2004-2013 trend in nitrogen (nitrate plus ammonium) concentration in precipitation, in microequivalents per liter (µeq/l), at Olympic National Park (from NPS 2016b). 2006 and 2007 data did not meet ARD completeness criteria so are not included in the trend analysis.

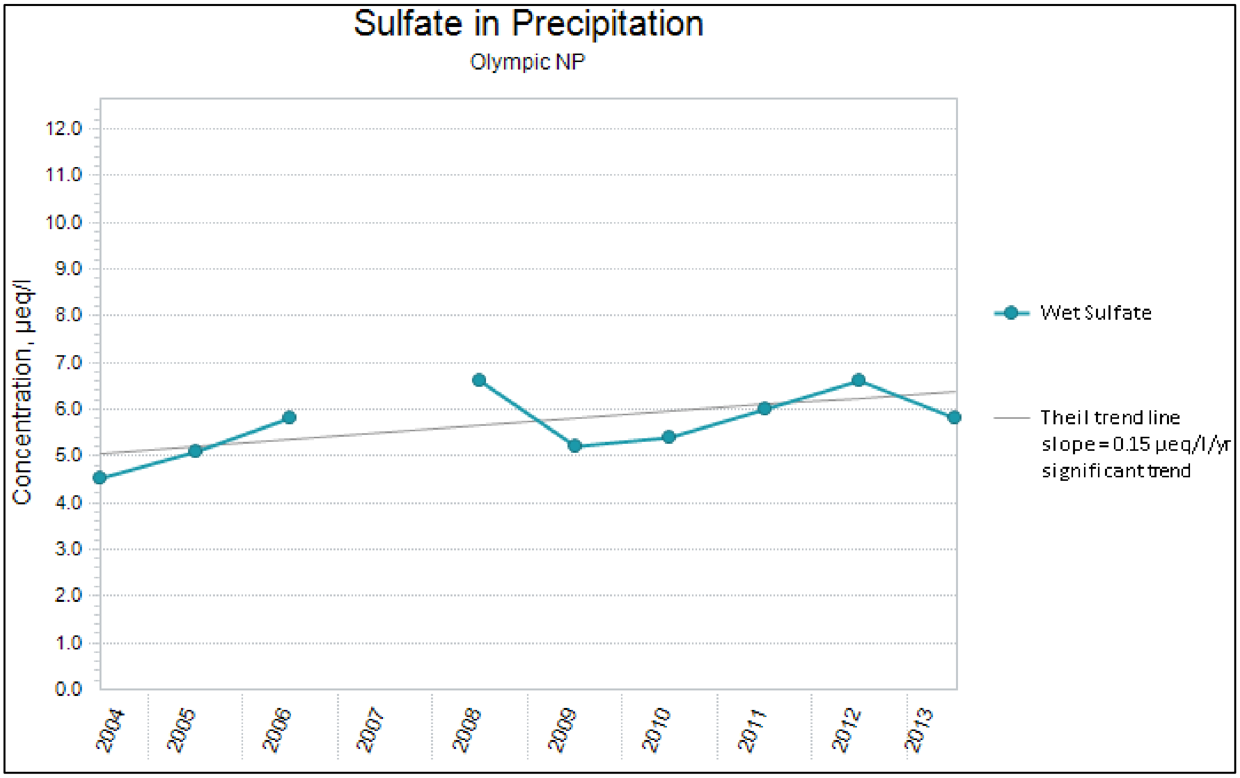

Figure 4.5.10. 2004-2013 trend in sulfur (as sulfate) concentration in precipitation, in microequivalents per liter (µeq/l), at Olympic National Park (from NPS 2016b). 2006 and 2007 data did not meet ARD completeness criteria so are not included in the trend analysis.

Based on estimated total (i.e., NADP monitored wet plus modeled dry) nitrogen deposition, minimum nitrogen critical loads were potentially exceeded at OLYM for all five ecosystem components described by Pardo et al. (2011; Table 4.5.9). There have also been two studies investigating critical loads for park lakes. Based on historic changes in diatom species composition of sediment cores, Sheibley et al. (2014) determined the critical load at Hoh Lake had been exceeded and estimated a critical load value of 1.2 kg ha^{-1} yr^{-1} of wet nitrogen. A subsequent *in situ* nutrient enrichment experiment by Williams et al. (2016) concluded that while phytoplankton in OLYM's mountain lakes are highly sensitive to future increases in nitrogen deposition, none of the three lakes studied at OLYM currently exceeded critical loads.

Table 4.5.9. Estimated 2010-2012 three-year average total (i.e., NADP monitored wet plus modeled dry) nitrogen deposition and minimum critical loads for five ecosystem components at Olympic National Park. Source: NPS 2016c.

Ecoregion	Total Nitrogen Deposition	Forests[1]	Herbaceous Plants and Shrubs	Lichens and Bryophytes	Mycorrhizal Fungi	Nitrate Leaching
Northwestern Forested Mountains	4.8	4.0[2]	4.0[2]	2.5[2]	5.0[2]	4.0[2]
Marine West Coast Forests	5.2	5.0[2]	N/A	2.7[2]	5.0[2]	N/A

[1] Trees and soils

[2] Values indicate a potential critical load exceedance (also shown in red text).

Mercury

Mercury concentrations in dragonfly larvae are highly variable, ranging from less than 5 nanograms/gram wet weight (ng/g ww) to nearly 950 ng/g ww (Figure 4.5.11). Moreover, there was substantial variability in average mercury concentrations among sites, suggesting that habitat- or site-specific processes may be influencing entry of mercury into the food web (Figure 4.5.11). In fact, there was a 48-fold difference in dragonfly larvae total mercury concentrations between the sites with the highest and lowest average concentrations. Across all 12 sampled park locations, 69%, 48%, and 12% of dragonflies had mercury concentrations that were above low-, moderate-, and high-severity benchmarks, respectively, for potential effects to fish. Mercury concentrations in 15%, 10%, and 7% of dragonflies exceeded low-, moderate-, and high-severity benchmarks for potential effects to birds.

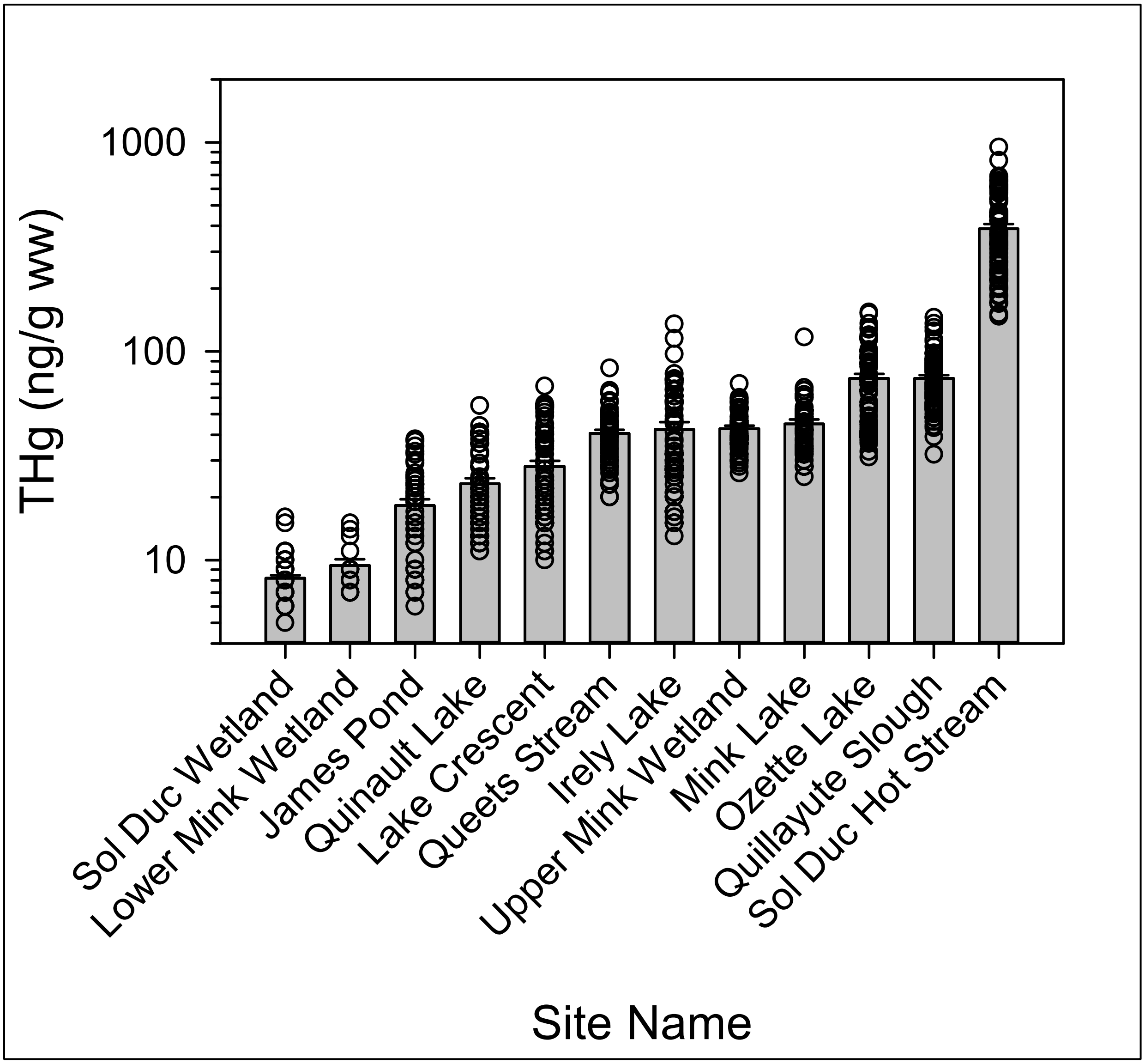

Figure 4.5.11. Total mercury (THg) concentrations in dragonfly larvae collected from select aquatic locations throughout Olympic National Park. Concentrations are reported as nanograms of mercury per gram of tissue on a wet weight basis. Source: https://www.sciencebase.gov/catalog/item/5b92cffce4b0702d0e80a2d5 (accessed 4 October 2018).

There was less variation in fish mercury concentrations across the park than dragonflies, though concentrations still spanned from 25 to more than 1200 ng/g ww (Figure 4.5.12). On a site-specific basis, mercury concentrations had a >4-fold range between the sites with the highest and lowest mercury concentrations in fish (Figure 4.5.12). Most of the fish sampled in OLYM contained mercury concentrations that were below benchmarks associated with deleterious effects, but 6% of fish contained mercury concentrations placing them at risk for low-severity impairment, and 1% and 0.6% of fish contained enough mercury to be classified as moderate- or high-severity impairment, respectively. In contrast, 35%, 8%, 1%, and 1% of fish contained sufficient mercury concentrations to potentially cause low-, moderate-, high-, and extreme-severity impairment, respectively, in fish-eating birds.

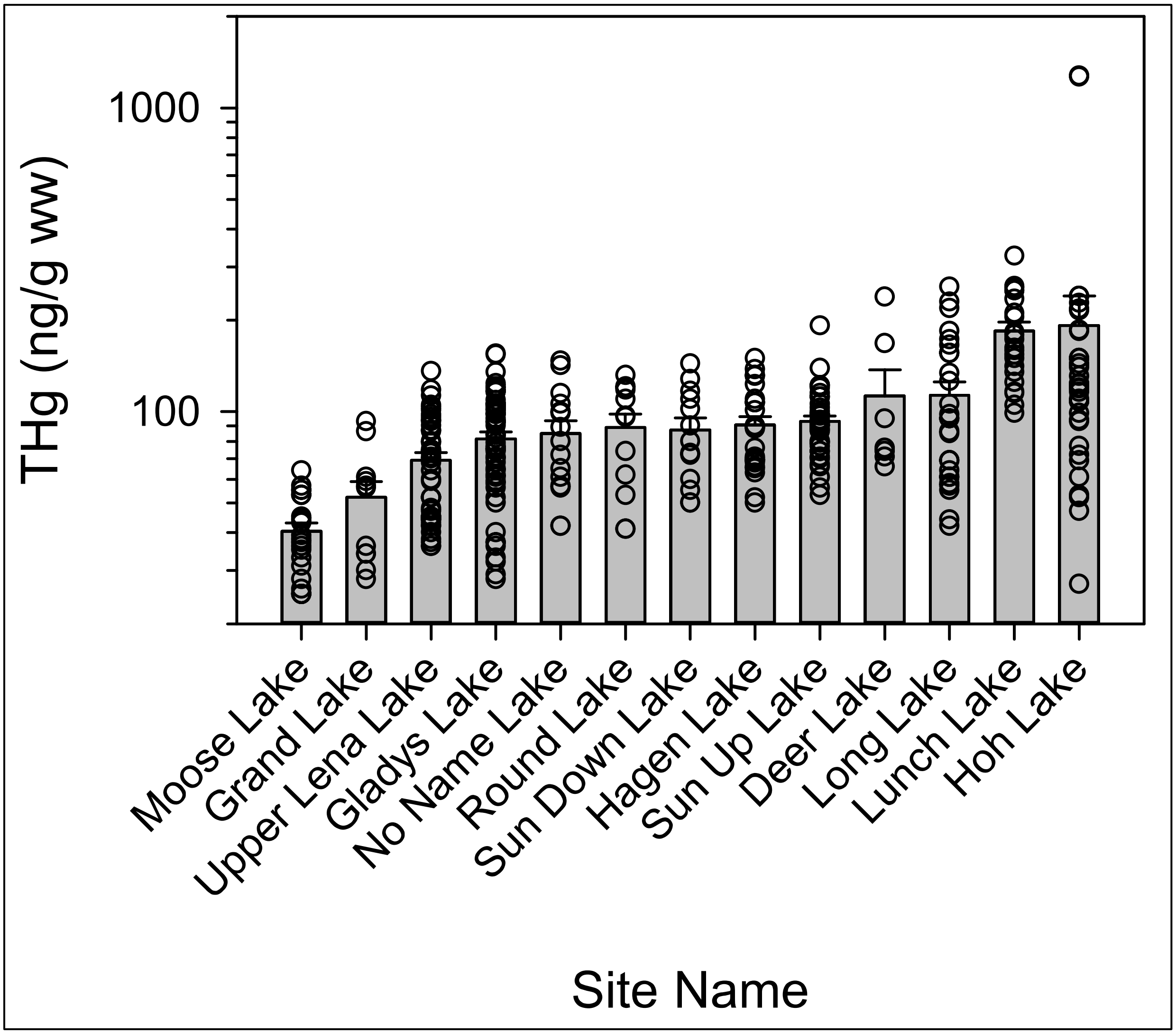

Figure 4.5.12. Total mercury (THg) concentrations in muscle tissue of fish (brook trout, cutthroat trout, and rainbow trout) collected from select aquatic locations throughout Olympic National Park. Concentrations are reported as nanograms of mercury per gram of tissue on a wet weight basis. Source: Pfleeger 2015.

With regard to human health concerns, 16% of fish had mercury concentrations that were below any consumption guideline benchmarks, whereas 84% of fish contained enough mercury to fall into categories recommending limited fish consumption to 2 meals per week or less. Of those, 28%, 5%, and 1% of fish contained mercury concentrations associated with consumption guidelines of up to: 1 meal per week, 1 meal per month, or no consumption, respectively.

Mercury concentrations in riparian songbirds can indicate the availability and risk of mercury to terrestrial wildlife that rely upon energetic subsidies from aquatic habitats. Across 17 species of songbirds sampled at 11 sites across OLYM, mercury concentrations ranged 116-fold from the lowest to the highest individual. Across species, mercury concentrations were lowest in Swainson's Thrush (*Catharus ustulatus*), Dark-eyed Junco (*Junco hyemalis*), and Song Sparrow (*Melospiza melodia*), and highest in Common Yellowthroat (*Geothlypis trichas*), Pacific-slope Flycatcher (*Empidonax difficilis*), and Yellow-rumped Warbler (*Dendroica coronata*) (Figure 4.5.13). Across all sites and species, mercury concentrations in 46% of songbirds sampled were above the benchmark associated with low-severity impairment, and approximately 1% of birds exceeded the moderate-severity impairment benchmarks (Figure 4.5.13).

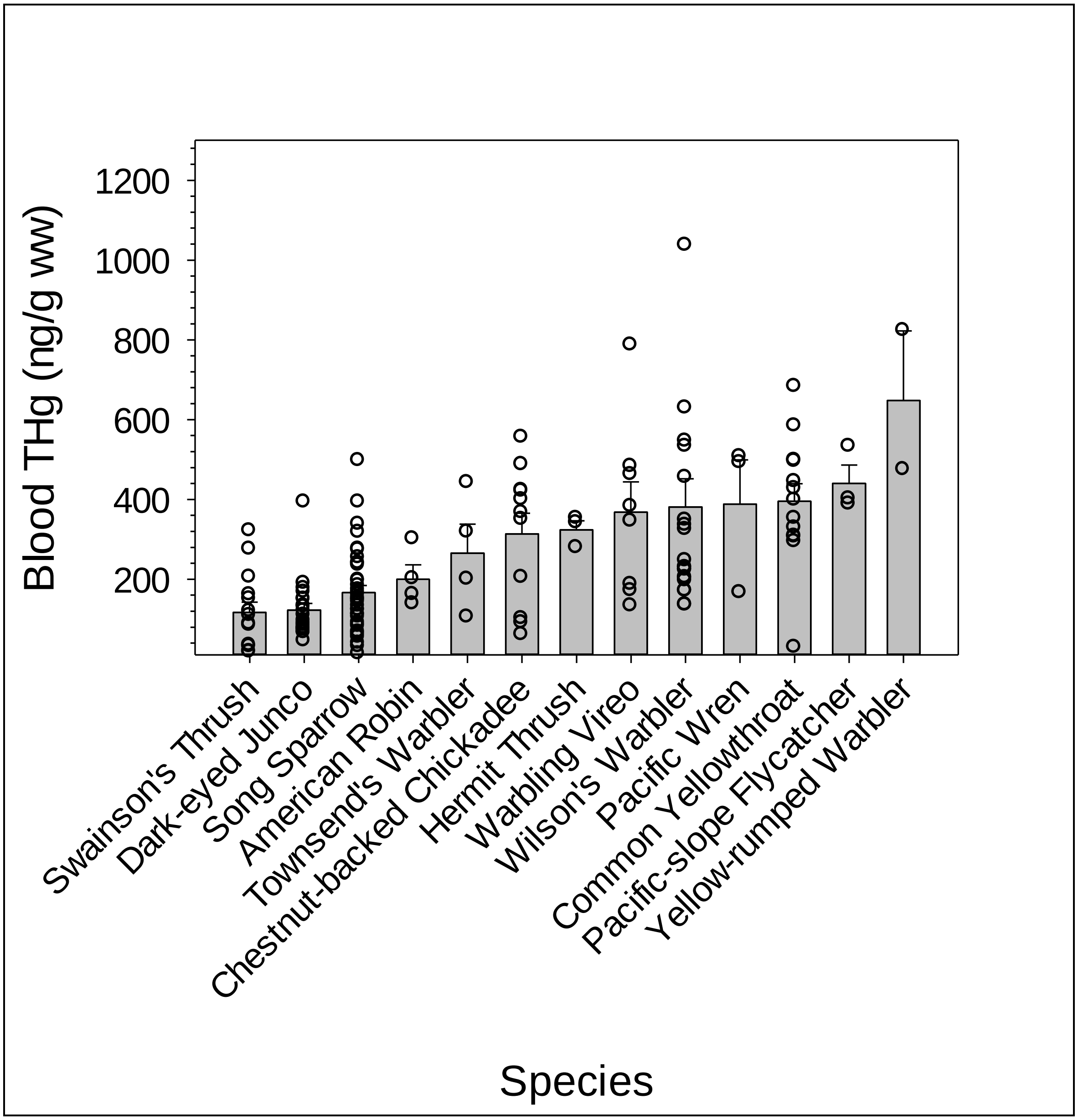

Figure 4.5.13. Total mercury (THg) concentrations in blood of various songbird species sampled from select locations throughout Olympic National Park. Concentrations are reported as nanograms of mercury per gram of blood on a wet weight basis. Source: USGS unpublished data.

Other Persistent Bioaccumulative Toxins

Results from the WACAP study (Landers et al. 2008) showed concentrations of mercury, polycyclic aromatic hydrocarbons, endosulfans, α-hexachlorocyclohexane, hexachlorobenzene, and dacthal in vegetation from OLYM were at mid- to upper ranges compared to other parks. Concentrations of chlorpyrifos, trifluralin, and polychlorinated biphenyls were low. The vegetation samples were collected along an elevational transect. While some toxins, such as polychlorinated biphenyls and pesticides, increased in vegetation with elevation, concentrations of polycyclic aromatic hydrocarbons decreased. The authors hypothesized that due to high forest productivity at OLYM, pesticide accumulation in vegetation may contribute significant contaminant loads to the ecosystem via canopy throughfall and needle litter-fall. Concentrations of pesticides in fish from OLYM were low to average compared to other parks, and concentrations of non-mercury contaminants did not exceed human or wildlife health thresholds (Figure 4.5.14).

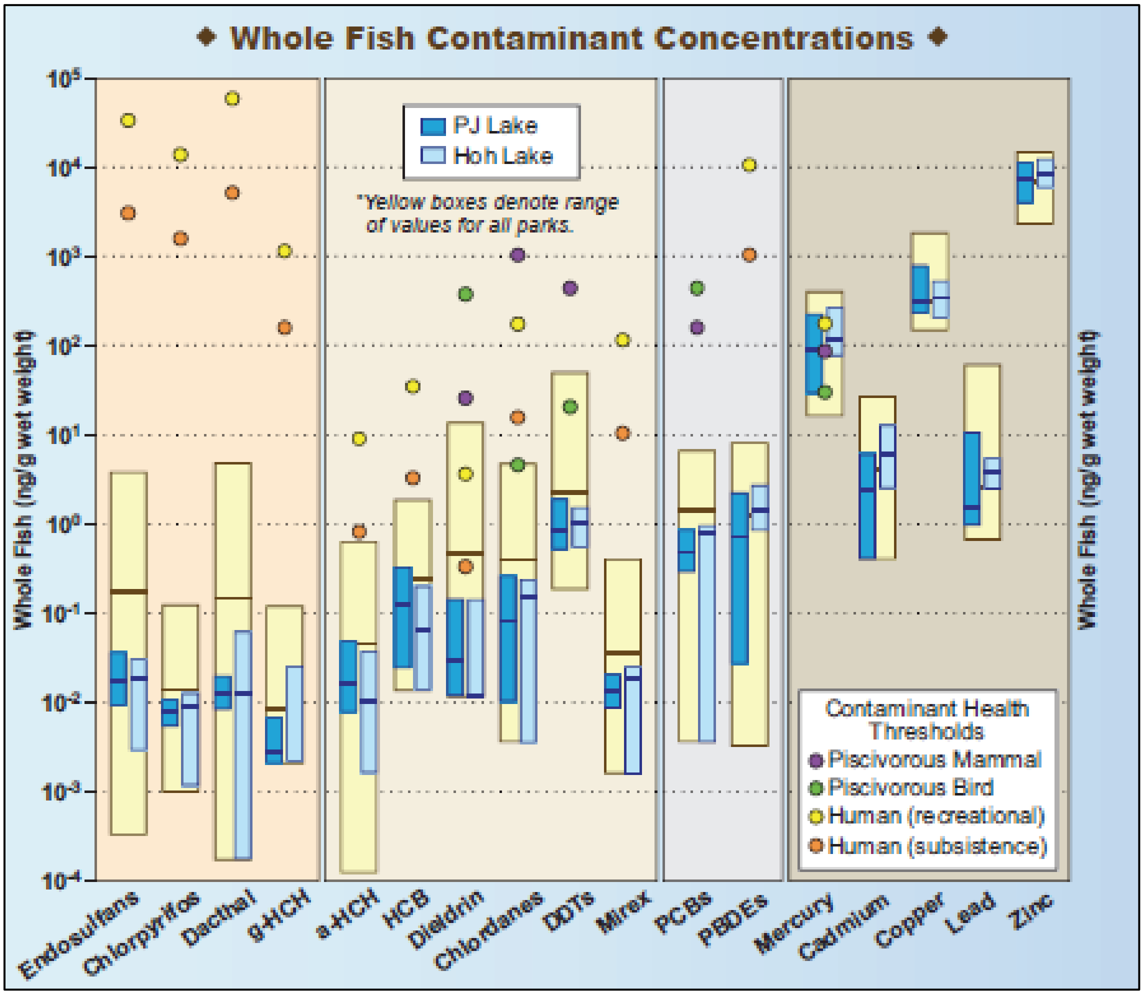

Figure 4.5.14. Concentrations of persistent bioaccumulative toxins detected in fish collected from PJ and Hoh Lakes in Olympic National Park as part of the WACAP study (from Landers et al. 2008). Horizontal lines in boxes indicate median values. From left to right, pollutant categories represented by large shaded boxes are Current-Use Pesticides, Historic-Use Pesticides, Combustion By-products, and Metals. Concentrations are reported as nanograms of mercury per gram of tissue on a wet weight basis.

Uses and Limitations

A significant amount of air quality monitoring and research has been conducted at OLYM since 1980. This assessment followed ARD's approach for assessing condition and trends of visibility, ozone, and sulfur and nitrogen deposition (NPS 2017). Air Resources Division has no Servicewide method for assessing park condition related to persistent bioaccumulative toxins. Thanks to OLYM's concerns about mercury, this assessment was able to rely heavily on extensive mercury effects data collected in the park.

Condition assessments for visibility, ozone, and nitrogen and sulfur deposition at OLYM relied on the handful of monitors located in the Pacific Northwest. However, based on ARD's criteria (NPS 2017), there is high confidence in the assessments for the park. The visibility metric used in ARD's assessments was specifically designed to evaluate regional visibility rather than estimate impacts from local sources; in fact, visibility calculations for the entire lower 48 states are based on interpolations from a total of approximately 150 monitors. Ozone, too, is a regional pollutant, and while monitoring at several locations in the park has shown some elevational differences, the ozone interpolation likely represents the condition at OLYM adequately. The sulfur and nitrogen deposition assessments incorporate a precipitation model that increases the amount of deposition as precipitation increases. The highest deposition modeled at OLYM was compared to ARD's deposition benchmarks. Nevertheless, because there are indications nitrogen critical loads may be exceeded at OLYM, it would be useful to corroborate the interpolated deposition values with additional on-site data, particularly with high-elevation data.

There is limited information about the sensitivity of OLYM's resources to nitrogen and sulfur deposition. Aside from lichens, none of the nitrogen critical loads developed by Pardo et al. (2011) incorporated data collected in the park. The lack of ecological response studies, combined with the limitations of the nitrogen deposition estimates, increases uncertainty about the applicability to the park of critical loads developed elsewhere.

The recent, extensive sampling for mercury concentrations in organisms throughout the park suggests that mercury exposure is highly variable among locations, and that in some locations mercury may be bioaccumulating to concentrations that threaten the health of fish and wildlife. The cycling of mercury is complex and biological exposure can be decoupled from inorganic mercury sources. Instead, the habitat and landscape factors that influence mercury methylation may be more important and additional research and modeling to better understand these relationships could help better predict risk across the park. In particular, this could be valuable for identifying lakes that may contain fish with mercury concentrations that exceed human health criteria. Additionally, although mercury concentrations in some individuals exceed suggested impairment benchmarks, there is tremendous variability in sensitivity to mercury among species. Thus, it is unclear if organisms inhabiting the park are currently suffering deleterious effects due to mercury exposure. Assessments of reproductive health and endocrine disruption tied to mercury exposure would be informative to better elucidate potential deleterious effects. While there are significant amounts of data about mercury concentrations in fish, birds, and invertebrates collected throughout the park, there is limited information about the extent of non-mercury persistent bioaccumulative toxins exposure or contaminant effect thresholds at OLYM.

Emerging Issues

Climate Change

Climate change may exacerbate air pollutant concentrations and effects on resources. For example, increased summertime temperatures may lead to higher ozone levels (USEPA 2009). Increased temperature and changes in precipitation patterns could enhance nitrogen deposition-associated effects on plant biodiversity, forest health, and nutrient cycling in ecosystems (Cummings et al. 2014).

Trans-Pacific Air Pollution

Data indicate that trans-Pacific air pollution is increasing (Lin et al. 2014). While there are encouraging reports recently that China is taking steps to reduce emissions, it is unclear the degree to which these changes will resolve concerns across all air pollutants, and whether other nations upwind of OLYM will also strengthen emission controls.

Information Needs/Gaps

Data to Fill Gaps in Understanding of Ecological Response to Nitrogen Deposition in the Region

Ecoregional critical loads identified in Pardo et al. (2011), which represented state-of-the-art estimates of available information, may not be entirely appropriate for OLYM because they are based on data from locations as far away as Colorado and Alaska. The challenges in better defining nitrogen critical loads for the park include: (1) orographic variations in deposition are not adequately captured by regional deposition models, (2) regional critical loads data for forests (other than for lichens) are lacking, and (3) potential interactions of climate change and nitrogen deposition are complicated (Cummings et al. 2014). Understanding climate change interactions with nitrogen requires fundamental information on how climate impacts ecosystem sensitivity to nitrogen deposition (Greaver et al. 2016).

Data to Better Understand and Model the Spatial Extent of Mercury and Other Persistent Bioaccumulative Toxins Exposure, Concentrations, and Effects at OLYM

This would include toxicological studies, as well as process-based studies to better understand the factors influencing mercury cycling and bioaccumulation in the park.

Improved Estimates of Nitrogen Deposition in the Pacific Northwest

Current monitoring focuses on inorganic nitrogen and wet deposition; contributions from organic nitrogen and dry deposition are mostly unknown, but are expected to rival wet inorganic nitrogen inputs. Moreover, because nitrogen deposition in the region is relatively low and levels are often close to the lower end of the critical load range identified for sensitive species, more precise site-specific deposition estimates would allow park managers to more accurately assess impacts on OLYM’s ecosystems.

Predictions of Future Trends in and Sources of Air Pollution

Emissions will be influenced by U.S. and global economies. Regulatory activities, such as USEPA's periodic review of the National Ambient Air Quality Standards or implementation of standards to reduce hazardous air pollutants such as mercury, could reduce emissions.

Data Regarding the Synergistic Effects of Air Pollutants on Resources

For example, nitrogen and ozone are the two most common air pollutants in North America (Taylor et al. 1994) and can interactively shape growth, mortality, and other ecological processes in a variety of forest types (Grulke and Balduman 1999, Ollinger et al. 2002), though information from Pacific Northwest forests is lacking.

Studies of the Effect of Vehicle Emissions on Roadside Air Quality

Air pollutants associated with vehicles include particulate matter, nitrogen oxides, benzene and other toxins. Research from Scotland, for example, shows impacts on vegetation due to increased nitrogen deposition near roads (Truscott et al. 2005).

Data on How Quickly Degraded Ecosystems Can Recover if Air Pollution is Reduced

Field observations and experiments in Europe and eastern North America have shown that ecological recovery following lowering of nitrogen and sulfur deposition can vary greatly depending on the response variable considered (Boxman et al. 1998, Lawrence et al. 2015).

Resource Summary

In summary, the condition of air quality indicators at OLYM varies (Table 4.5.10). While visibility is in moderate condition, the park is currently meeting Regional Haze Rule goals for improving visibility on the worst days and no degradation on the best days. Ozone concentrations are not a threat to either human health or vegetation. Both nitrogen and sulfur deposition warrant significant concern. Nitrogen deposition may be exceeding critical loads for OLYM's sensitive resources, but additional information is needed to confirm the applicability of critical loads developed outside the park. Mercury exposure exceeds low- and/or moderate-severity impairment benchmarks in a substantial proportion of invertebrates, fish, and birds throughout the park, but the prevalence of effects is not yet known. Additionally, the proportion of individuals exposed to elevated mercury varies among habitats and sites in the park, suggesting that some areas pose greater or lesser risk than others. Non-mercury persistent bioaccumulative toxins have been detected in the park, but their extent and effects are unknown.

Table 4.5.10. Resource condition summary for air quality and contaminants in Olympic National Park.

Measure	Condition and Trend	Rationale
Visibility	🟡 ⇧	Meets goals for improving visibility on the worst days; no degradation on best days
Ozone effects on human health	🟢 ⇔	No threat to human health; no trend over time
Ozone effects on vegetation health	🟢 ⇔	No threat to vegetation; no trend over time
Nitrogen deposition	🔴 ⇔	Exceeding critical loads for key resources; no trend over time
Sulfur deposition	🔴 ⇩	Exceeds ARD benchmarks; increasing trend over time
Mercury	🟡	Exposure exceeds low- or moderate-severity impairment benchmarks for diverse taxa
Other persistent bioaccumulative toxins	◌	Detected, but trends and effects unknown

Chapter 5. Discussion

5.1. Introduction

Rebecca McCaffery and Kurt Jenkins

The ecology of the Olympic Peninsula is ever-changing. During the Pleistocene, the ebb and flow of continental and alpine glaciers shaped its mountains and valleys and created its unique fauna and flora. Aboriginal peoples followed the retreating ice and affected the landscape through burning and harvest in poorly understood, yet likely significant ways. Much more recently, human activity has altered species composition, species abundance, and ecosystem processes. Direct activities such as logging, land-clearing, harvest, hunting, and non-native species introductions have shaped landscapes, changed food webs, and affected ecosystem function. Impacts of climate change and airborne contaminants pervade the peninsula to reach its most pristine landscapes. Amidst these changes, OLYM was established, bringing new protection to much of the peninsula. More recently, restoration projects such as dam removal on the Elwha River and reintroduction of fishers to OLYM have been undertaken to bring back some of the park's lost ecological integrity. These restoration activities themselves change ecosystems, as they conserve and restore key species and ecological function. Ultimately, this Natural Resource Condition Assessment is influenced by these historical and current natural and human-induced changes shaping OLYM.

Rialto Beach. Photograph courtesy of NPS/Danielle Archuleta.

In the NPS, new guidelines for resource stewardship embrace this notion of continuous change, couching the duty of resource stewardship and the preservation and restoration of ecological integrity within this context (Colwell et al. 2012). Our assessment of resource condition was designed to explicitly evaluate some of the ecological changes in OLYM that are indirectly or directly driven by current human activities or legacies of prior activity. The five guiding questions were developed to address this goal. Fundamentally, these are some of the biggest questions in conservation biology and natural resource management: impacts of species extirpations, impacts of human harvest, loss or changes in biodiversity, impacts of invasive species, and impacts of contaminants.

While our national parks represent some of the most protected ecosystems in the American landscape, they are constantly affected and changed by forces both within and outside of our control, as outlined above. Present conditions in OLYM must be interpreted in light of the fact that the park is nestled within a working landscape managed primarily for resource production or other human uses. Certain far-reaching stressors have caused irreversible changes to park resources. In conducting this assessment of OLYM's natural resources, we have addressed many issues that park decisions can directly affect, and others that are more elusive. Consequently, the questions grappled with here do not always lead to a management directive; in some cases, a decision may be to continue current management approaches, especially given limited resources. Where direct management activities are not feasible or possible, we highlight the importance of understanding the effects of changing resource condition on ecosystem function.

Within the uncertainty of past conditions and future trajectories, our goal was to assess to the best of our ability the condition of key resources in the present day. To do this, we had to rely on our understanding of historical conditions for various resources. We often had to interpret resources against idealized reference conditions that we will never return to, such as a landscape with no invasive species. Even early written accounts of the peninsula and park did not necessarily reference the landscape we would aim to return to, for example where predator and prey populations had already been heavily persecuted. Another challenge we encountered was "shifting baseline syndrome" (Pauly 1995), where documentation of historical conditions is lacking and current trends are judged against an insufficient time series of data, such as comparing current conditions to an already degraded "reference" state. Interpretations of current condition could be misleading if they are compared to reference conditions that do not represent accurate historical conditions. This affected how we could quantify or qualify current conditions in relation to historical conditions. Within these challenges, we acknowledged uncertainties, highlighted irreversible changes to historical ecosystems, and stressed the importance of assessing ecosystem function in light of these changes. In some cases, we have judged resource condition based on our assessment of whether basic ecosystem processes and functions are intact, even when the resource state may be altered.

In this chapter, we collate the work presented in this assessment to provide a comprehensive synthesis of the status of OLYM's natural resources, as viewed through the five guiding questions and twelve resources of focal interest (Table 5.1.1). The first four questions ask about ecosystem processes in relation to species loss, harvest, decline, and introduction. The final question addresses effects of airborne contaminants and air quality on ecosystem processes. Our synthesis moves beyond the individual resources assessed in Chapter 4 to provide a broader picture of natural resources conditions in OLYM, with discussion of both the challenges and opportunities for moving forward in managing and understanding key park resources and ecological processes.

Table 5.1.1. Resource condition summary for each focal resource. For guiding question 4, condition and trends are reported for the native species or community that would be impacted by changes in the non-native or invasive species or community.

Guiding questions	Resource	Condition and Trend	Rationale
Guiding question 1: What are the effects of the loss of top predators on ecosystem health?	Elk	🟡⇩	The resource condition is of low to moderate concern because elk counts have declined on three index count areas since 1985. Confidence is low based on the limited survey coverage of the park's elk herds and uncertain baselines of the natural (i.e., primitive) condition.
	Riparian Vegetation	🟡⇔	The resource condition is of moderate concern primarily due to lack of recruitment of palatable tree species. Although cottonwood and maple recruitment has diminished over the last century, conditions seem stable with respect to key ecological functions of riparian ecosystems.
	River Geomorphology	🟡⇔	All rivers have widened since the 1970s, and the Hoh river showed increased braidedness, but we do not ascribe these trends as representing improved or deteriorating conditions.
Guiding question 2: What are the effects of resource harvest for human use on ecosystem function and sustainability of the resource?	Pacific Salmonids	*Variable*	Assessments of Pacific salmonids vary by stock and river, where some stocks are deemed stable, some are in decline, and others have unknown status. See 4.2.1 for detailed assessment.
	Razor Clams	🔴⇩	All measures of condition (population size, size-class distribution, and average adult size) warrant significant concern and have shown deterioration in condition over time.
Guiding question 3: What are the population trends and ecological impacts of key declining and/or sensitive species or communities?	Olympic Marmots	🔴⇔	After declines in the 1990s, the current park-wide occupancy seems to have stabilized, but threats from coyote predation and climate change remain, and contraction of the range is of concern
	Sea Stars	🔴⇩	*Pisaster* has declined in all monitoring plots and continues to be affected by SSWD. New recruitment in this species may show prospect for recovery. Other species have shown symptoms of SSWD, but population-level effects are unknown. Climate-induced disease outbreaks may recur in the future.
	Amphibians	◌	Past amphibian inventories point to wide distributions of park species, but status of most species is unknown and occupancy of all species is due to be revisited. Focal study of the Cascades frog shows a stable but variable population size, warranting moderate concern. Ongoing threats include climate change, non-native fishes, and disease.

Table 5.1.1 (continued). Resource condition summary for each focal resource. For guiding question 4, condition and trends are reported for the native species or community that would be impacted by changes in the non-native or invasive species or community.

Guiding questions	Resource	Condition and Trend	Rationale
Guiding question 4: What species or ecosystem functions are threatened by non-native or invasive species?	Barred Owl (Northern Spotted Owl)	🔴⇩	Northern spotted owl territory occupancy has decreased while occupancy of those territories by invasive barred owls has increased. Local extinction rates by northern spotted owls have increased over time.
	Balsam Woolly Adelgid (Subalpine Fir)	🟡⇩	BWA has been present since 1970 with increasing severity till 2007. Many trees have died, with unknown future consequences for these communities and ecosystem function.
	Invasive Non-native Plants (Native Plants)	🟡⇩	The threat is increasing; control efforts are only effective against small populations and in local areas; resources for control efforts are limited; populations seems to be restricted to developed areas, road, and trails for now.
Guiding question 5: What are the ecosystem effects of airborne contaminants?	Air Quality and Contaminants	*Variable*	Resource condition varied among the air quality indicators, with some measures in good condition, some warranting moderate or severe concern, and some with unknown condition or trend. See 4.5.1 for detailed assessment.

5.2. What are the effects of the loss of top predators on ecosystem health?

Kurt Jenkins, Rebecca McCaffery, Amy East, and Andrea Woodward

We examined the effects of wolf extirpation on the ecological health of several major river systems in OLYM nearly 100 years after the fact. We summarized and reviewed the evidence of potential legacy effects of wolf eradication via direct effects on Roosevelt elk populations (Chapter 4.1.1), and indirect effects on riparian vegetation (Chapter 4.1.2) and river-channel morphology (Chapter 4.1.3). Here we expand the discussion to explore in greater depth the web of factors that may have interacted with wolf extirpation to lead to current conditions in the west-side rainforest valleys—the primary winter ranges of OLYM elk herds. We examine whether the loss of wolves appears to have affected natural ecosystem functions and processes to the extent that ecosystem health is threatened. For this task, we define a healthy ecosystem as one that is sustainable by virtue of maintaining its functional organization, productivity, and resilience to change (Rapport et al. 1998, Costanza and Mageau 1999).We build the argument that structural complexity, system productivity, and redundancy of ecological functions within OLYM's west-side river ecosystems have helped sustain key ecosystem processes and functions despite the loss of wolves and other changes to the ecosystem. Finally, we acknowledge limitations of our analysis by identifying several other less understood components of park ecosystems that may also be affected by trophic cascades resulting from loss of wolves.

5.2.1. Wolf-Elk Interactions

Elk populations clearly increased in the early 1900s alongside the gradual eradication of wolves, the campaign to reduce or eradicate cougars (Murie 1935b, Sumner 1938, Schwartz 1939), and a 28-year cessation of legal elk hunting (1905–1933; Schwartz 1939). Because the effect of wolf demise is forever intertwined with these other factors, the elk population increase at the turn of the 20th century must be viewed as a collective response to the suite of elk protection measures enacted at that time.

The evidence suggests that the rebounding elk population likely exceeded the ecological carrying capacity of its habitat (as reports of overbrowsing and die-offs suggest; Appendix E), and then declined to an intermediate density in between the historical low and high population levels (Beschta and Ripple 2008; Figure 4.1.3). The argument built by Beschta and Ripple (2008) implies that the population stabilized at unnaturally high densities after release from wolf predation, thus maintaining an unnaturally high level of herbivory. Early historical accounts of abundant elk wearing deep trails in the soil and suppressing understory vegetation, however, serve as a reminder that densities of elk today may not be exceptional compared to the early historical era (reviewed in Chapters 4.1 and 4.2 and summarized in Appendix E).

We speculate that increased cougar populations by the mid to late 1900s may have moderated the long-term effects of wolf eradication on elk abundance. Cougar populations may have increased as a result of diminished competition from wolves as well as from enhanced protection within the newly created and expanded national park. In instances where wolves have recolonized areas inhabited by established cougar populations, wolves and cougars compete for similar prey (Akenson et al. 2005, Kortello et al. 2007, Elbroch et al. 2015). Wolves interfere directly with cougars by killing them, usurping their prey, or displacing them to steeper or more rugged habitats (Ruth et al. 2005, Atwood et al. 2007, Bartnick et al. 2013). Therefore, cougars may be more abundant and more widely distributed today than prior to the elimination of wolves, particularly in OLYM’s interior where cougars have not been hunted for several decades. Individual male and female cougars reportedly kill about 24 and 3 elk per year outside the park on the Olympic Peninsula, respectively, with males selecting elk over deer, and females selecting deer over elk (Murphie and McCoy 2015). Both male and female cougars select elk calves over older elk (Murphie and McCoy 2015). Recent research based on 12 studies of elk calf survival and mortality throughout the Rocky Mountains indicated that the relative influences of cougar and wolf predation on elk populations are partially compensatory (i.e., as the effect of wolf predation increases, that of cougar predation decreases; (Griffin et al. 2011). The addition of wolves to a predator guild composed of cougars, black bears, and coyotes (which occasionally prey on young elk) increased neonatal mortality rates of elk calves by only about 4% (Griffin et al. 2011). The low rates of calf recruitment in OLYM’s population, particularly for year-round low-elevation resident elk (Schwartz 1939, Newman 1958, Jenkins 1981), are consistent with Newman’s (1958) conclusion that cougar predation and nutrient limitations are important contributing factors limiting elk populations in OLYM. Predation of elk calves by black bears on the peninsula, although poorly understood, may also play a role limiting elk populations (Raedeke et al. 2002).

In addition to directly affecting population size, wolves also influence the distribution and foraging patterns of elk, thus also their browsing signature on the landscape. Research following wolf restoration in the greater Yellowstone area showed that elk tended to move less predictably as wolf encounters increased. This change in movement included reduced home range fidelity, greater frequency of temporary long range movements, and even occasional switches in migratory patterns (Gude et al. 2006, Gower et al. 2009). Elk also altered habitat use patterns in the presence of predators, making greater use of higher-security forested habitats and less use of riskier open habitats in the face of predation risk (Creel et al. 2005, Fortin et al. 2005). Heightened awareness caused by the threat of predation may also cause individual elk to remain more vigilant while feeding—and consequently to forage less efficiently—when wolves are near (Liley and Creel 2007, Laundré et al. 2010). Similarly, we speculate that wolves may have prompted more frequent movements and less predictable distribution patterns in OLYM's elk. That is, elk may not have concentrated activity in the riparian zone and nearby floodplains as extensively in the past as they do currently (Jenkins and Starkey 1984, Schroer et al. 1993). We speculate that changes in elk distribution related to the loss of wolves may have affected and still affect the long-term recruitment of palatable deciduous tree species (such as black cottonwood and bigleaf maple) on developing floodplains of several major river systems in the park.

5.2.2. Wolf-Elk-Vegetation Interactions

After wolves were eliminated on the peninsula and elk populations increased, early biologists consistently and emphatically reported that palatable browse species were "overbrowsed", "degraded" or "impacted" on the principal low-elevation elk ranges in OLYM, particularly the Hoh, Queets, and Quinault Valleys (reviewed in Chapter 4.1.2). An increasing elk population with, perhaps, a more sedentary distribution pattern likely contributed to the degraded range conditions, but the influx of settlers' cattle at the same time challenges the notion of a single cause (Skinner 1933b, Murie 1935b, Dixon 1943, reviewed in Appendix E). Dixon (1943) considered "*the unauthorized trespass by domestic stock with the park as the greatest present menace to native wildlife*" in the park. By the 1950s, however, Newman (1958) considered that elk and vegetation had reached an acceptable equilibrium in the park coincident with the gradual removal of livestock and perhaps a gradual buildup of cougar numbers following park protection. Although many reports of historical range conditions were highly subjective, utilization of key browse species measured in the Queets and Quinault Valleys decreased between 1939 and 1980 (Jenkins 1981), supporting Newman's perception that equilibrium between vegetation and elk numbers had developed during the intervening time, a notion generally accepted decades later (Leslie and Starkey 1982, Happe 1993, Woodward et al. 1994).

Herbivory continues to be a driving force affecting the biomass and density of palatable shrub species in floodplain and fluvial terrace forests on the key elk ranges (Happe 1993, Schreiner et al. 1996). Newman (1958) first recognized that grazing and browsing by elk helped to maintain open meadow understories within the open overstory patches in fluvial terrace forests; i.e., that elk maintained habitats most suitable to their nutritional needs. Exclosure studies confirmed that browsing and grazing herbivores indeed maintained a mosaic of grass-dominated foraging patches within the coniferous forest matrix, while also optimizing nutrient qualities and spatial concentration of nutrients within the most productive and preferred foraging patches (Happe 1993). These observations together with the apparent decrease in browsing pressure helped transform the historic perspective—that intensive herbivory pressure was a severe problem—to a more ecological perspective that the observed browsing pressure likely represented the outcome of natural forage optimization processes (McNaughton 1979, Frank and McNaughton 1993).

Judging the "naturalness" of the past and current elk–vegetation relationship requires an understanding of pre-settlement reference conditions, including the range of natural variation in elk populations and herbivory influences. Modern exclosure studies demonstrate the remarkable short-term response of vegetation to the complete exclusion of deer and elk (Figures 4.1.9 to 4.1.12), but they provide no information regarding the range of natural variations in herbivory pressures that existed prior to European contact. Clearly, herbivores left a recognizable signature on the landscape prior to settlement, at least in localized areas (reviewed in Chapter 4.1.2., Appendix E), but the historical accounts are not sufficient to gauge the spatial extent of open park-like patches maintained by native herbivores.

Vegetation conditions have never been static, however. Wind acting in concert with insects and root pathogens is the predominant forest disturbance in moist lowland forests in the western Olympics (Agee 1994), affecting forests at small to vast spatial scales. Over the last century hurricane-force winds have hit the Olympic Coast on average about every 20 years, sometimes affecting broad swaths of forest (Henderson et al. 1989). Webster (1922), for example, described effects of the legendary windstorm of 1921 as follows: "*Formerly the elk traveled up and down this benchland [in the Hoh Valley just outside the current park boundary], now a mass of wreckage from the big storm, miles on miles of piled up trees, impenetrable by either deer or elk*." Wind also blows down individual to small groups of trees, thus creating frequently changing, small-scale variations in light penetration, understory vegetation composition and productivity (Happe 1993, Schreiner et al. 1996), as well matrices of downed wood on the forest floor (for an example see McKee et al. 1982).

The relationship between understory vegetation, large herbivores, and wind disturbance is a natural dynamic process of the Olympic rainforest communities (Schreiner et al. 1996). Although elk clearly shape the structure and composition of riparian vegetation communities, dead and downed wood (covering about 10% of the land in some areas, Harmon and Franklin 1983) clearly adds structural complexity that influences vegetation pattern at the patch and landscape scales. Large fallen logs provide natural refugia from browsing deer and elk. The rapid response of palatable shrubs to protection from herbivores within natural refugia or artificial exclosures provides testimony to the resilience of vegetation communities in OLYM's alluvial forests when relieved from browsing (Woodward et al. 1994, Schreiner et al. 1996). Collectively, the high productivity of the system, structural complexity of the understory, rapid response of vegetation to protection from herbivory, and the historical record indicated no obvious concerns over the condition of most of the key browse species measured at the landscape scale.

The increase of elk populations in the early 20th century appears to have had the most pronounced impact on the recruitment and establishment of palatable deciduous tree species, particularly black cottonwood and bigleaf maple in developing floodplain communities (Beschta and Ripple 2008).

Changes in herbivory patterns (potentially mediated by changes in the abundance, distribution, or behavior of large herbivores likely contributes to the scarcity of cottonwood and bigleaf maple regeneration on the most heavily used elk ranges (Balian and Naiman 2005, Van Pelt et al. 2006, Beschta and Ripple 2008). Establishment of cottonwood, however, depends on a narrow range of substrate and hydric conditions (Braatne et al. 1996), as well as specific disturbances creating those optimum conditions. Houston (1994), for example, observed small cottonwood trees in the Quinault Valley associated with bars created during specific flood events in the early 1950s and with log jams that may have accumulated sediments ideal for germination, while also providing protection from both herbivory and erosion. Acker et al. (2008) reported cottonwood trees establishing on landforms created by a natural landslide dam-break flood in the Elwha Valley. Further, our incidental field observations revealed bigleaf maple saplings growing in protected areas out of reach of elk, particularly on landslides or rock fans at the base of the valley walls in the Quinault and Hoh Valleys. Collectively, these observations indicate that while elk are clearly a major influence on the establishment of cottonwood and bigleaf maple within the west-side river valleys in OLYM, episodic disturbance histories may also play a role in their establishment over long time periods (Fonda 1974). Additional studies would help to better understand the multiple factors that affect cottonwood and bigleaf maple establishment and recruitment patterns on OLYM river floodplains.

5.2.3. Wolf-Elk-Vegetation-River Interactions

Multiple physical processes have dominated river-planform change in OLYM over the 74-year photographic record. In selected reaches of four gravel-bed rivers, channel width and braiding increased in relationship with (1) increased hydrologic forcing, especially the recent flood history; (2) increased sediment-supply that may be either episodic - such as from recent landslides - or, we surmise, long-term and ongoing, from effects of rapid glacial retreat, at least in the Hoh watershed; and (3) localized legacy effects of anthropogenic modification over the past century on the Quinault River.

Contrary to assertions of Beschta and Ripple (2008), we found no evidence that river systems in the west-side rainforest valleys have shifted from being predominantly single-threaded channels prior to wolf eradication to the mainly braided-channel systems seen today. Rather, multiple lines of evidence suggested that the braided river systems represent the natural condition for these rivers. In Chapter 4.1.3 and Appendix E we reviewed historical descriptions of braided and dynamic river systems reported by the first explorers and homesteaders in the western Olympics. Many of the rivers in OLYM have slope and discharge conditions typically expected to result in the development of braided channels (Figure 4.1.26). The tendency for river channels to become braided is further reinforced by inputs of large wood, a feature inherent to these systems (Naiman et al. 2010). We found no evidence in the stage–discharge history of these rivers to suggest that that there was a major sediment pulse prior to 1939 signaling a fundamental restructuring of river channels (which presumably would have been accompanied by substantial sediment export). The range of variability in channel width since 1939 encompasses the same range found in measurements from land-survey maps made between 1904 and 1919, further suggesting that no major geomorphic regime-shift occurred during the era of relatively high elk abundance in the early 20th century.

Although our data indicate that elk impacts on channel morphology are likely superseded by larger signals of physical controls, we do not dismiss the possibility that lesser or more dispersed herbivory by ungulates in the presence of wolves could enhance fluvial resilience to physical forcing through greater cottonwood and bigleaf-maple presence. We cannot determine whether channels would have evolved differently had more of those trees been present. We speculate, however, that overstory composition of riparian forests would not have changed sufficiently through gradual senescence of mature cottonwood and bigleaf maple trees to have fundamentally altered riverine processes prior to the photographic record. We also speculate that there are built-in functional redundancies in the vegetation community that likely mitigate the effects of diminished recruitment of young cottonwood and bigleaf maples on developing floodplains. Sitka spruce, red alder, and willows have high stem densities and basal area in developing floodplain communities; these species grow rapidly in this productive environment and are key players in vegetation successional pathways (Fonda 1974, McKe et al. 1982, Balian and Naiman 2005, Van Pelt et al. 2006). Even the presence of large trees, however, does not guarantee bank stability or resilience to flood erosion, given that channels with bankfull width >20 m also tend to be deep enough that they erode their banks beneath the rooting zone of even the largest trees in this setting (Beechie et al. 2006). Fallen, large-diameter conifers and root masses, primarily Sitka spruce in the western Olympic rivers (Harmon et al. 1986, Hyatt and Naiman 2001, Latterell 2005), provide key members to log jams, which create erosional resistance, stabilize new floodplain surfaces, and trap sediments (Fetherston et al. 1995, Abbe and Montgomery 1996). Although these are all functions also attributed to rooted streamside vegetation generally in riparian areas, log jams play a critical function enhancing and protecting developing floodplains during the extremely high flows typical of west-side OLYM rivers (Naiman et al. 2010).

We find little evidence supporting concerns that widening of the braided river channels over time threatens to disrupt the seasonal delivery of logs and large wood into the major river systems, which would disrupt key ecological functions and degrade fish habitat (Beschta and Ripple 2008). This concern arose from observations that many braided river channels were not in contact with a forested edge except during high flows (Beschta and Ripple 2008). We demonstrated that mean annual streamflow had no significant connection to channel widening, whereas the magnitude of annual peak flows shows strong statistical relationships with both channel width and the degree of channel braiding (East et al. 2017). This further suggests that the majority of wood delivery and transport occurs during high flows—a conclusion amply supported by field observations. The edges of the active channels today are in closer contact with large conifer trees than any time during the last 70 years (the extent of the photographic record) as a result of recent channel widening. Despite erosion of older surfaces, we see no evidence of change in the mosaic of landforms and plant communities over time that would threaten to diminish the source of large key members of log jams in the future.

Our interpretations, which highlight the primacy of physical over biological forcing of river morphology, differ from those of Beschta and Ripple (2008) in part because of different assessment methods and study design. In a previous publication we contrasted our analysis of river channel changes to what we believe was a less robust inference derived from spatial comparisons (and a space-for-time assumption) made among river reaches that differed not only in densities of elk, but also in basic underlying geology and geomorphic processes, particularly sediment supply. We infer that those differences in geologic setting readily explain the spatial geomorphic contrasts that Beschta and Ripple (2008) attributed to elk use. We also reviewed fundamental differences in the Olympic versus Rocky Mountain (Yellowstone) river systems and hydrology that likely contributed to differences in elk grazing influences on river channel morphology at the regional scale (although even in Yellowstone National Park, physical forcing by floods and sediment supply may provide the strongest controls on channel morphology; Meyer 2001).

5.2.4. Potential Effects of Wolves on Other Ecosystem Components

In assessing the impacts of wolf extirpation in OLYM, we focused on only a few measures of resource conditions in low-elevation river valleys. But the loss of wolves may also affect other components of the park's ecosystems. The extirpation of wolves and habitat changes on the Olympic Peninsula, for example, was followed by an expansion and increase in coyote populations during the early to mid-1900s (Scheffer 1995). Research indicates that coyote predation in the subalpine/alpine regions of the park has likely contributed to recent Olympic marmot populations declines (Witczuk et al. 2013, see Chapters 4.6 and 5.4). The ecological release of coyotes associated with the disappearance of wolves likely also affected other carnivores through competitive interactions, as well as populations of their prey (Prugh et al. 2009, Miller et al. 2012). Moreover, reduced herbivory pressures attributed to increased predation on ungulates have been correlated with increases in breeding bird density and diversity (Berger et al. 2001, Hebblewhite et al. 2005), beaver density (Hebblewhite et al. 2005), and berry production (Beschta and Ripple 2012a, Ripple et al. 2014). Beavers are widely recognized as ecosystem engineers (Wright and Jones 2006), suggesting that detriments to their populations may extend to other ecosystem functions and components (Wright et al. 2002, Wright et al. 2004). We have no data to document these potential effects in OLYM, but suggest that the influence of wolf loss in OLYM likely extends beyond the species and systems we assess in this report.

5.2.5. Conclusions

Elk have left an unmistakable imprint on OLYM ecosystems, but the legacy of wolf loss is only one of many forces that have shaped the lowland riverine systems. Floodplain vegetation, patch dynamics, and channel characteristics are under the continual powerful influence of natural disturbance regimes including infrequent large-scale wind and fire events, catastrophic floods, frequent small-scale wind throw, continual fluvial disturbances, and herbivory influences. Although herbivory effects have varied over the last century, owing in part to the loss of wolves, natural ecosystem processes seem largely intact, and resilient to natural disturbance regimes. As Van Pelt et al. (2006) observed, "*despite the apparent levels of chaotic destruction [in riverine floodplains associated with flooding], the riparian vegetative community is highly resilient as it quickly reorganizes into a diverse and productive ecological system*."

Based on our working definition of a healthy ecosystem—one retaining its parts, productivity, and resiliency—our assessment of the effects of loss of wolves on ecosystem health is mixed. The loss of wolves reduced diversity of the large mammalian predator guild. Although predation by cougars and nutritional limitations may have replaced the limiting influence of wolves on elk (to some unknown extent), the system has by definition lost functional redundancy and resiliency with the loss of the wolf. Similarly, the reduction in black cottonwood and bigleaf maple may not have had measurable effects on river channel dynamics due to the functional redundancy afforded by Sitka spruce and red alder, but what other ecological functions provided by cottonwood trees might be diminished by their decline? It is precisely to hedge bets against these uncertainties that the policies of the National Park Service strive to retain all component parts of ecosystems under its management (NPS 2006).

Although several key properties of the elk-vegetation-riverine systems appear resilient to the loss of wolves, we identified other resources likely affected by wolves, and additional resource concerns not related directly to wolves. For elk, we identify human uses on the park's boundary, the threat of disease, and consequences of a changing climate as the greatest concern to populations. For riparian vegetation, our greatest concern lies in the potential unmeasured ecological impacts of diminished recruitment of palatable riparian tree species such as cottonwood and bigleaf maple. Increased frequency and magnitude of winter floods in OLYM and changes in sediment transport may portend future changes in floodplain patch composition, river channels, and fish habitats, as well as enhanced threats to park infrastructure located on river floodplains. Continued monitoring and study of these other potential drivers of ecosystem change can help the park understand system dynamics and prioritize management into the future.

Recent studies indicate that herbivore impacts on vegetation and their cascading effects throughout ecosystems may be mediated along gradients of increasing primary productivity (Pringle et al. 2007, Hopcraft et al. 2010). Indeed, during a period of intensive browsing pressure in OLYM, Murie (1935b) observed that herbivores appeared to be less detrimental to vegetation or soils than in other areas, specifically referencing Yellowstone. A few decades later, Newman (1958) concluded that the abundance of big timber, downed logs, mosses, "*good turf*" and "*rapid and regular seasonal growth of forage plants*" afforded some protection from overbrowsing and erosion. Most recently, University of Washington ecologist Jerry Franklin elaborated this general concept during a visit to the South Fork of the Hoh River, suggesting that Sitka spruce – not the wolf – likely plays the predominant role shaping many key ecological processes in these valleys (J. Franklin, personal communication, April 2016). We echo the sentiment that the impacts of elk in the low-elevation rainforest valleys of OLYM may be different than ecosystems like those in Yellowstone as a result of profound differences in disturbance regimes, habitat structure, and primary productivity.

In the end, it is not surprising that we should leave with such uncertainty over the effects of an ecological perturbation that happened so long ago, especially in a complex, multi-causal system (Peterson et al. 2014). As a result, ecologists continue to vigorously debate the relative importance of wolf restoration on elk, vegetation, and river dynamics (Winnie 2012, Barber-Meyer 2015, East et al. 2017), even after watching the system adjust to the restoration of wolves in other regions. In Yellowstone all parties agree that wolves have changed the ecosystem appreciably since their return (Eisenberg 2010). Therefore, we conclude that even if key ecosystem processes in OLYM do not appear to be distinctly impaired as a result of the missing wolves, the system is likely different today than it was historically and that it would likely change again if wolves return—perhaps in ways we cannot easily visualize. The addition of wolves would likely affect the behavior and distribution patterns of elk, their pattern of herbivory on the landscape, and the establishment and recruitment of palatable tree species. The presence of wolves may have cascading effects on other carnivore populations and lower trophic levels. We are much less certain that these changes would add significant resilience to the system in mediating the effects of changing climate patterns, which include increased flooding, predictions of increased landslide activity, and probable downstream effects of glacier retreat, including river-channel widening and braiding. Ultimately, only time and perhaps the eventual return of wolves to OLYM will help to further unravel the story of how the loss of wolves once changed the ecological function of these systems—and how the system might change again.

5.3. What are the effects of resource harvesting for human use on ecosystem function and sustainability of the resource?

Rebecca McCaffery, Jeffrey J. Duda, Samuel J. Brenkman, and Steven Fradkin

Resource harvest has always been an integral part of the human ecology of the Olympic Peninsula. The riches of the peninsula's forests, fish, and wildlife have sustained generations of people, from the first people to set foot on these lands to the present day. Natural resource harvest is important commercially, recreationally, and culturally to many groups on the peninsula, and occurs on land, in rivers, and at sea. While OLYM's lands are largely protected from resource harvest, certain resources within the park legally are harvested within its borders (e.g., razor clams), or are impacted by harvest activities taking place outside the park (e.g., salmonids). In these cases, resource management does not stop at the park's boundary.

For this question, we examined the condition of Pacific salmonids and razor clams in OLYM. Both of these resources highlight the challenges of species and conservation management issues that reach far beyond the boundaries of the park. Our assessment of salmonids in particular highlighted the need for ongoing cooperation among various partners tasked with maintaining healthy salmon populations on the peninsula. Assessments of both salmon and razor clams reinforced the need for ongoing targeted monitoring that can address both harvest goals and ecosystem function of the resource.

5.3.1. Pacific Salmonids

Salmon are iconic symbols of the Pacific Northwest, and those that inhabit the waters of OLYM are important components of the park ecosystem and related cultural, ecological, economic, and recreational domains. Inside of OLYM, salmonids represent a key node within complex aquatic foodwebs, and their carcasses provide energy subsidies to terrestrial plant and animal species. Due to their anadromous life cycle, salmonids spend significant periods outside of park waters, where they are subjected to factors that ultimately affect their abundance, distribution, and genetic composition in park waters. Factors influencing populations include harvest, hatcheries, habitat degradation, disease, and ocean conditions. This dichotomy, in addition to the treaty, legal, and economic factors involved with salmon as a commodity, creates significant management complexities for fisheries management at OLYM (further discussed in Brenkman et al. 2014).

We completed a comprehensive synthesis of harvest, terminal run size, escapement, and hatchery release data for 15 key salmonid stocks from four major OLYM rivers. Gathering quantitative harvest and hatchery data has proven to be challenging due to lack of consistent record keeping, few records in electronic formats, and difficulty in obtaining data across the range of multiple sources. We provided quantitative measures as much as possible to assess recent status and trends of populations, but where appropriate relied upon both qualitative and quantitative descriptions to assess current health and status of OLYM salmonid stocks as compared to limited historical baselines.

Olympic National Park is managed as a natural area and as a Biosphere Reserve it is seen as a reservoir of both biodiversity and genetic diversity (Houston and Contor 1984, Lawrence et al. 2011). Our assessment was intended to gather appropriate information to address the question of ecosystem function and resource sustainability in light of the fact that salmonids are harvested. Three of the key datasets we compiled from various sources— in-river harvest estimates, escapement estimates, and hatchery plantings—spanned a period from the early 1900s to recent decades since the historic Boldt decision. Yet, placing these data in an appropriate historical context, where, for example, we could make quantitative comparisons with pre-settlement abundances for the species of interest was a challenge. Our attempt to examine historical canning and harvest records is barely adequate to assess orders of magnitude of abundance, and does not contain any estimates of variability or uncertainty. Despite these shortcomings, we can say with some level of certainty that, like other assessments throughout their range, the historical levels of Olympic Peninsula salmonid populations were larger than current levels and the widespread use of hatchery practices is a phenomenon that has expanded in a relatively short period of time compared with the evolutionary history of salmon. Thus the abundance of salmonids and associated marine-derived nutrients returning to park waters is also reduced from historical levels and the levels of genetic introgression from outside stocks due to hatchery planting is above historical baselines.

Our results showed a mixed assessment of the current status and recent trends of salmonid populations returning to park waters. Two-thirds of the 15 stocks assessed had some level of concern, with the other third being in good condition. Of the *Oncorhynchus* species reviewed, the sustainability of spring and summer-run Chinook salmon in coastal rivers is of utmost concern. Of the rivers reviewed, the sustainability of Queets River salmonid stocks is of most concern with three of four stocks assessed as having significant concern. This is of particular concern as 85% of the Queets River flows through OLYM. By comparison, the Quillayute system had 3 of 5 stocks considered in good condition. We also highlighted salmonid stocks where there are no established escapement goals yet harvest occurs annually (i.e., Sol Duc summer coho, Quinault stocks).Where some stocks consistently had low terminal run sizes and missed escapement goals (e.g., spring Chinook), others were more stable and more consistent in reaching their management goals (e.g., fall coho). Stocks such as summer coho have unique life histories and are very dependent upon OLYM waters for spawning and rearing. Because of management based on MSY principles and the available harvest data, we do know that harvest targets 30-70% of a given stock’s run size with peak annual harvest rates of up to 84% of the terminal run for some stocks (e.g. Sol Duc summer coho).

In addition to reducing abundance, commercial and recreational exploitation of fish can cause significant changes to life history and morphological traits of wild populations (Darimont et al. 2009). Although not evaluated in this report, fishing can be selective on life history traits and cause reductions in size and age and shifts in run timing (Todd and Larkin 1971, Hamley 1975, Trippel

1995, Quinn et al. 2006, Quinn et al. 2007, Kendall et al. 2009). Ricker (1981) reported that fishery selection contributed to declining trends in age and body size in many salmonid populations, though he recognized that these traits may be affected by several factors. Gillnets can be size-selective (Hamley 1975, Ricker 1981, Bromaghin 2005), and examination of age- and size-selective harvest of sockeye salmon in a commercial gillnet fishery over 57 years revealed higher susceptibility and exploitation of older and larger fish in most years (Kendall et al. 2009). There also are possible genetic effects of fishery selection on age and size at maturity (Law 2000, Olsen et al. 2004).

The scale and magnitude of hatchery practices has also become an important factor that affects wild salmonids in rivers draining from OLYM. The magnitude of hatchery practices, in terms of numbers released, duration of planting, and number of effected rivers, as well as the origin of donor stocks, poses risks to the sustainability of wild salmonids that originate in OLYM. The possible effects include genetic, fitness, behavioral, and ecological changes, as well as disease risks (Waples 1999). It also causes significant challenges to NPS management objectives. At the same time, hatcheries can play a role in recovery of federally listed stocks (e.g., Elwha River Chinook salmon; Peters et al. 2014) and supplement recreational and commercial fisheries. Recognizing the complexities and importance of the issue, more attention has been paid to the effects of hatcheries in the scientific literature and recent hatchery reform processes have focused on improving fish culture practices (e.g., Mobrand et al. 2005). Taken together, the effects of harvest and hatcheries, two of the main factors affecting salmon populations that are present to varying degrees across the watersheds we analyzed, pose risks to the sustainability of Pacific salmonid resources in OLYM.

The ability to maintain salmonids and their important role for natural aquatic ecosystem functioning within OLYM will depend upon continued cooperation among park, State of Washington, and Treaty Tribes, a fact long recognized among park managers (Houston and Contor 1984). This coordination and cooperation would benefit from expanded monitoring and assessment of salmonid resources to inform management. Escapement, run size, and harvest data are taken as true values, however, these types of data are likely associated with considerable uncertainty due to sampling or estimate error (Knudsen 2000, Ohlberger et al. 2017). Conservation and sustainable use of salmonids requires accurate stock assessments and escapement estimates, which could be enhanced through the use of new technologies and tools in fisheries management, such as side-scan sonar to estimate escapement, life cycle models, spawner surveys that occur over greater temporal and spatial scales, and risk assessments. Additional data and analyses could be used to reevaluate current management goals and criteria, as well as incorporating analyses of levels of uncertainty and risk into assessment (Ohlberger et al. *submitted*). This could inform important improvements to fisheries management of the rivers we assessed, such as establishing escapement goals for those stocks currently lacking them and determining if current escapement levels are effective for the long-term sustainability of salmon and their fisheries. This question would also be served by an assessment of alternative approaches to salmon management, like ecosystem based fisheries management (Pikitch et al. 2004), multispecies management, and precautionary principle approaches to both fishing stocks and fishing communities (e.g., Lauck et al. 1998, Hilborn et al. 2001). Any increased escapement of wild salmonid populations into OLYM will benefit the ecosystem and the role of the park as a stronghold of salmon conservation.

5.3.2. Razor Clams

Razor clams on Kalaloch Beach in OLYM represent a sensitive and declining population of this culturally important and highly sought-after clam species. Data have been collected over the past 23 years largely to support harvest management activities, yet these data have uncovered other stressors to the population that are generally poorly understood. For razor clams, human harvest does not appear to be the primary factor currently affecting the sustainability of this resource: harvests did not reach the allowable catch on most years when it was allowed, and full harvest opportunities only occurred in 8 of the last 22 years. Furthermore, substantial mortality has occurred on some years between the time when stocks were assessed and when the harvest season began. The clearest stressor to this population is the bacterial gill pathogen NIX, with other stressors present that could potentially play a role in future population dynamics.

Although harvest does not seem to be a major driver of sustainability of the resource, there are ways in which harvest activities could affect the population. First, while the dataset covers over 20 years of stock assessments, it is unknown if the current population may have legacy effects from historical overharvest. Furthermore, while harvest has been conservative, it could still represent an additive source of mortality that further impacts an already stressed population when it does occur. In addition to the clams that are harvested, people will toss back smaller, often damaged clams in hopes of landing bigger ones, which could compound effects of harvest. Selection for the largest clams in a population where mean clam size is declining could also contribute to a decline in mean clam size. Given the lack of regular harvest over the period of record, these factors are likely not major drivers of population dynamics, but should be considered as potential stressors to a struggling harvested population.

Although the effects of harvest appear minimal, this assessment revealed a lack of understanding of the factors affecting razor clam population dynamics. Results from this assessment can thus provide guidance for future monitoring activities. Our assessment showed that while recruitment may be adequate, adult survival (as measured by progression to larger age classes) and mean adult clam size seems to be lower in this population than in the reference population at Copalis Beach, Washington.

The factors regulating population size over time are generally poorly understood, and targeting monitoring and research to address these questions could help elucidate questions surrounding razor clam population dynamics. Studies that examine (1) what governs interannual fluctuations in razor clams populations of Kalaloch; (2) where sources of recruitment are located; (3) the role of subtidal clam populations in the dynamics of intertidal clam populations; and (4) the natural stressors to this population will help us understand recruitment dynamics and sources of mortality. Explicit examination of the biology of the NIX pathogen would elucidate how and when it impacts razor clam populations. Finally, ongoing monitoring of other potential stressors that could become more important with climate change, such as sea level rise, physical disturbance, and ocean acidification, should continue. A clearer understanding of population dynamics will go a long way to understanding if, when, and how this population can be sustainably harvested.

5.4. What are the population trends and ecological impacts of key declining and/or sensitive species?

Rebecca McCaffery, Michael J. Adams, Patricia Happe, and Steven Fradkin

Olympic National Park contains numerous species that are rare, declining, or otherwise sensitive to changes in the environment. For this question, we addressed the trends of three animal resources of concern to the park: the Olympic marmot, the sea star community, and the amphibian community. Although there are also many other species of concern in the park (reviewed in Chapter 2.3.2), the selected species span the elevational gradient from subalpine and alpine habitats of the Olympic mountains (marmots and amphibians), down to middle and low elevation forests, streams, and wetlands (amphibians), and out to coastal intertidal habitats (sea stars). Stressors include disease, habitat changes, predation, and non-native species impacts, but what unites the potential threats to these three resources is climate change. Factors that appear to be affecting these resources – or that may affect these resources adversely in the future – can nearly all be linked to the direct and indirect effects of long-term changes in climate across these diverse taxa and the park ecosystems they inhabit.

Each resource we evaluated has unique concerns with relation to climate change impacts. For the Olympic marmot, concerns over species viability consider both direct effects of changing snowpack and temperature, as well as the indirect and synergistic stressors of conifer encroachment/habitat loss due to changing climates, and easier access to marmots by predators such as the non-native coyote, which could occur with earlier snowmelt. For sea stars, the key concern to the community is the recent and ongoing outbreak of sea star wasting disease, but the underlying driver that may have facilitated the outbreak is suspected to have been warm seawater temperatures associated with climate change. For amphibians, warming temperatures and loss of snowpack may present future threats to the viability of certain populations, particularly those dependent on snow-fed or temporary wetlands in higher elevation systems. Stresses induced by climate can be exacerbated by other threats to amphibians such as non-native fish presence and disease outbreaks.

In this section, we synthesize the assessment of each of these three resources in turn, providing justification for and guidance on how OLYM may address future monitoring and management of these resources. As with the other resources, managers will need to grapple with stressors that are outside of their direct control, determine the best use of monitoring resources and research programs to track the ongoing status of these resources, and evaluate potential management decisions and tools that could be used to change trajectories for species of concern.

5.4.1. Olympic Marmots

The Olympic marmot is a charismatic, iconic, and endemic inhabitant of OLYM's high-elevation ecosystems. There is no doubt that the species has declined in recent decades, prompting detailed research (Griffin et al. 2008) and the establishment of a park-wide monitoring program (Witczuk et al. 2009). Our assessment indicated relatively stable occupancy of marmot colonies park-wide over the past six years. However, it highlighted ongoing declines in the southwest and southeast regions of the park, which could lead to a significant contraction of this species' range. Furthermore, stressors like coyote predation and climate change are of ongoing concern to the long-term viability of this species.

While there was no evidence of direct impacts of climate change to marmot demography (Griffin et al. 2007a) or occupancy (this assessment) to date, climate change may pose both direct and indirect challenges to marmots in the future. For species like the marmot that already inhabit environments at the upper elevational limits of their habitat, changes in snowpack and temperature, increased drought, conifer encroachment into meadow habitat, and increased predation pressure are accompanied by an inability to migrate to new habitats. Past radio-telemetry work has demonstrated low dispersal capability (Griffin et al. 2009), meaning that isolated habitat patches may be hard to naturally recolonize after they go extinct. Conifer encroachment on meadow habitat could degrade habitat and lead to increased colony isolation. Finally, historical colony loss occurred during a series of low-snowpack years, and detailed demographic work conducted by Griffin also occurred over a series of low-snow years. Without long-term studies that incorporate both high and low snow years, it is hard to determine what the demographic effects of snowpack could be, or to more clearly determine what mechanisms snowpack variability could affect.

Changes in snowpack and habitat caused by climate change could also pose threats to Olympic marmots by facilitating coyote predation. Coyotes have been shown to be an important predator of marmots (Van Vuren 1991, Van Vuren 2001), and OLYM is no exception (Witczuk et al. 2013). Coyotes were historically absent in OLYM (Scheffer 1995), and it has been posited that wolf extirpation led to mesopredator release across the American West (Ripple et al. 2013), which included the coyote. Coyotes have been documented to prey on marmots in OLYM since the 1960s (Barash 1973). In a recent study, coyotes were determined to be the dominant terrestrial marmot predator in OLYM (Witczuk et al. 2013). We hypothesize that increased conifer cover may facilitate coyote predation by allowing them to more frequently approach marmot colonies undetected. Further, lower snow pack could allow earlier access to the high country by these generalist predators. Indeed, declines in the 1990s coincided with several years of lower snowpack (NPS, unpublished data), and it is possible that conditions during that period facilitated increased predation pressure by coyotes on historical colonies.

The future dynamics of OLYM marmots is uncertain, but coyote predation and climate change, coupled with range contraction, are ongoing concerns. The successful citizen science program for monitoring marmots is a powerful way to keep collecting park-wide data on marmot occupancy. However, long-term studies of marmot survival would help inform underlying demographic patterns in the face of environmental variability. Furthermore, greater understanding of vegetation changes (i.e., conifer encroachment) using high resolution photography or LiDAR data would allow the park to examine the role of changing vegetation on marmot occupancy or demography. Finally, a greater understanding of the temporal and environmental factors affecting coyote predation would clarify the threat of coyote predation to marmot population viability. If wolves are reintroduced to or naturally recolonize OLYM, monitoring to assess potential changes in the behavior and abundance of coyotes in the high country would be warranted (e.g., Berger and Conner 2008). If continued declines and habitat changes are measured via occupancy analyses or other studies, further feasibility studies and modeling work may indicate adaptive management action is warranted, such as habitat amelioration or translocations.

5.4.2. Sea Stars

The sea star community – and *Pisaster ochraceus* in particular – is a colorful and critically important ecological component of OLYM's rocky intertidal ecosystem (Paine 1969). In 2013, an outbreak of a disease subsequently identified as SSWD caused mass mortalities of *Pisaster* and other species from Mexico to Alaska (Menge et al. 2016; Montecino-Latorre et al. 2016), including OLYM's coastline. This outbreak has been ongoing with varying intensity up to the present (Stokstad 2014, Eisenlord et al. 2016). Only time will tell how SSWD has impacted the intertidal community in OLYM now and into the future. Continued monitoring of this resource as part of the intertidal monitoring program is key to assessing recovery or continued decline and determining the potential repercussions that changes in abundance of the previously dominant *Pisaster* may have on the community. Pulses of *Pisaster* recruitment in the past three years, possibly due to competitive space released by adult mortality, suggest that the population may be recovering. Monitoring this new recruitment in the face of persistent SSWD will help explain disease dynamics in relation to sea star size (e.g., Eisenlord et al. 2016). Continuous monitoring of the broader intertidal community will help inform how loss and declines of this original keystone species (Paine 1969) affect the structure of the rest of the community. Monitoring may also shed light on how SSWD is affecting less dominant sea star populations.

In both OLYM (this assessment) and at sites in the San Juan Islands and South Puget Sound (Eisenlord et al. 2016), high prevalence of SSWD was associated with warm water temperature anomalies. In OLYM, an extreme anomaly ("the blob") occurred between September 2014 and April 2015, when the highest incidence of SSWD was detected. This marine heatwave occurred when waters normally heated in the summertime failed to cool in the winter (DiLorenzo and Mantua 2016). While SSWD persists in the environment, such an extreme marine heatwave has not re-occurred. These data suggest that climate change-associated marine heatwaves may facilitate outbreaks of SSWD like the one observed in OLYM and throughout the west coast. Monitoring of SSWD prevalence in conjunction with ocean temperature will help us understand how closely linked ocean temperatures may be to such disease outbreaks. If temperature anomalies like this one become more extreme or frequent, sea star populations may be more susceptible to future outbreaks.

Furthermore, monitoring SSWD, its potential drivers or facilitators, and the sea star community at OLYM contributes to the greater knowledge of how this disease is affecting sea stars based on datasets collected from California to Alaska, and represents some of the longer-term data we have on this phenomenon in these taxa. Dynamics of SSWD and *Pisaster* in OLYM are different than in other parts of the Pacific Northwest. Therefore, continued monitoring in OLYM is important to understanding the geographical variation in susceptibility to and recovery from SSWD now and in the future. In this sense, OLYM plays a key role in the larger conservation of these environments as we seek to understand (1) the impact of SSWD on *Pisaster* and other species; (2) the relative importance of environmental drivers of disease outbreaks; (3) the long-term impacts of declines or losses of *Pisaster* and other species on the rocky intertidal community; and (4) the potential for recruitment and recovery following the SSWD outbreak.

5.4.3. Amphibians

The amphibian fauna in OLYM is unique, diverse, and of regional and global importance. Amphibians are found in many habitats in the park, from high mountain lakes, to cascading mountain streams, and finally to low elevation lakes, rivers, and wetlands. Because amphibians are sensitive to temperature and other aquatic stressors, they serve as a barometer for freshwater ecosystem health in OLYM. Most amphibian species are widely distributed in the park, with the exception of the Van Dyke's salamander. While no species found in OLYM are listed at the federal level, certain species are of concern at the state or global level, because they are decreasing across their range (Western toad and Cascades frog), their status is unknown (coastal tailed frog), or they occupy a restricted range within the Pacific Northwest (Cope's giant salamander and Van Dyke's salamander) or the peninsula (Olympic torrent salamander). For all species, repeat surveys could help assess status in the park since the USGS conducted its species inventory 15 – 20 years ago.

The long-term monitored Cascades frog population in Seven Lakes Basin has shown no trend in growth over time, but has demonstrated remarkable interannual fluctuations. These types of fluctuations are not atypical for temperate, pond-breeding amphibians, but this high stochasticity warrants attention. High variability in population size over time is associated with increased risk of extinction in diverse species, and can be one of the greatest predictors of extinction risk for sensitive species (Fagan et al. 2001). Analyses of adult frog survival in this population show that survival increases with increased winter length and increased precipitation in the summer (Kissel et al., *in prep*). Therefore, a shift to longer, drier summers could lead to decreases in adult survival that quickly impact population size. Furthermore, recruitment success is tied to having a sufficiently long pond hydroperiod (i.e., length of time that a pond contains water each year) for developing tadpoles to metamorphose (McCaffery et al. 2014). With climate change, hydroperiods of breeding ponds are expected to shorten (Lee et al. 2015), and larval mortality in the Seven Lakes Basin is expected to increase 13% by the 2080s (A1B climate emissions scenario, Kissel et al., *in prep*). Overall, reductions in adult survival due to shorter winters and drier summers, coupled with an increase in larval mortality have the potential to push the population from stable to declining by as early as the 2040s (Kissel et al., *in prep*).

Changing climate conditions can also interact with the presence of non-native predatory fish in more stable, permanent wetlands to further challenge pond-breeding species (i.e., the climate vise, Ryan et al. 2014). This concern over climate impacts to high elevation pond-breeders points to a need for ongoing monitoring of focal populations like the Cascades frog in Seven Lakes Basin. While this research has been focused on a single population, trends and results may be applicable to many amphibian populations inhabiting the ponds and lakes of OLYM's subalpine region. Exploring management strategies such as targeted non-native fish removal could promote amphibian persistence in these vulnerable ecosystems. In a landscape with fewer semi-permanent wetlands, perennial waterbodies, many of which currently contain non-native fish, can provide key breeding and foraging habitat to sensitive amphibians.

In contrast to these in-depth studies at high elevation ponds, the status and trends of stream-breeding frogs and salamanders is largely unknown. Stream and river temperatures across the country are rising (Kaushal et al. 2010), and summertime stream temperatures are projected to increase in OLYM as well (Mantua et al. 2010). However the consequences of such potential changes to OLYM's amphibian stream breeders is unknown. Basic monitoring of key stream-breeding amphibian populations could help clarify their status and if or how they may be affected by climate change in the future. Finally, while disease has not been a significant issue in OLYM's amphibians to date, it is of concern for amphibians worldwide (Daszak et al. 2003, Muths et al. 2003, Lips et al. 2006, Martel et al. 2014). Precautions taken in lake, stream and wetlands monitoring programs to prevent the accidental spread of disease while continuing to regularly monitor for disease would help foster amphibian population health in the park.

5.5. What species or ecosystem functions are threatened by non-native or invasive species?

Rebecca McCaffery, Andrea Woodward, and David Wiens

Non-native species have been a part of the Olympic Peninsula since the first EuroAmerican explorers set foot in the area. Many arrived by accident as unnoticed travelers on boats, boots, and vehicles, while others, such as the mountain goat, were deliberately introduced. Yet others are slowly moving onto the peninsula as ranges expand and change (e.g., barred owl), perhaps facilitated by human-induced changes to the landscape or climate, or due to natural range expansion following the last ice age (e.g., porcupine). Some of these species persist at low densities and do not appear to alter ecosystem function or community composition, while others are deemed invasive due to their deleterious effect on native species or ecosystems (i.e., native invaders). With the exception of the mountain goat, which are susceptible to overharvest (and whose numbers and distribution are well known in the park), it is unlikely that many of these species can be fully eradicated: each is now part of OLYM's biota. This reality leads to the challenge of determining what management actions can – or should – be taken to (1) prevent further spread of these species, (2) limit damages they might incur, (3) protect native species, and (4) prevent introduction of new invasive species. As part of this process, the park may need to make difficult decisions on how much to prioritize investment in the control of an invasive species. In addition, new monitoring methods may be required to understand the distribution and abundance of invasive species of interest as well as effects of the presence and spread of these species on broader ecosystem processes, ultimately informing management strategy and prioritizing actions.

For this question, we examined three very different invasive species or species groups, each with unique challenges to monitoring, control, and management. First, we examined the effects of the invasive barred owl on the federally threatened northern spotted owl. The barred owl has aggressively displaced the northern spotted owl throughout its geographic range, leaving questions about the future of this iconic old-growth species in OLYM, as well as questions about the potential influence barred owls may have on other native species and OLYM's forest ecosystem. Next, we assessed the status of the invasive forest pest BWA. Like the barred owl, BWA is now part of OLYM's ecosystems, with implications for forest community structure as well as future detection and monitoring of pest species. Finally, we examined the status of invasive non-native plant species as a whole in the park. Contrary to the previous two examples, this assessment involved examining a large group of non-native plant species, where guidance for prioritization of future monitoring and management efforts among species is an additional need. In the following discussion of each of these invasive species/groups, we: (1) showcase the variety of ways that non-native and/or invasive species have infiltrated OLYM's ecosystems; (2) identify difficult challenges to management or partial control; and (3) highlight a need for active research and monitoring programs to better understand how the presence of these species will affect ecological function in diverse environments.

5.5.1. Barred Owl (Northern Spotted Owl)

In contrast to BWA and many invasive plant species, barred owls are native to eastern North America, but their range has expanded westward in recent decades. It is unclear to what extent the spread of barred owls has occurred naturally or been facilitated by historical human changes to the landscape (Gutiérrez et al. 2007), but they are now permanent residents of western forested ecosystems. As a generalist species, the barred owl has gained a strong foothold in western forests, outcompeting the federally threatened northern spotted owl throughout its range. Since the barred owl range expansion into OLYM, the species has gradually displaced northern spotted owls from their historical territories throughout the park, as demonstrated by 23 years of intensive demographic monitoring of northern spotted owls in the park.

In addition to competing with threatened northern spotted owls for habitat and food, the increasingly dominant presence of barred owls in OLYM introduces several monitoring challenges. First, because monitoring programs were designed specifically for the northern spotted owl, the occurrence and distribution of barred owls in the park is predominantly based on ancillary information collected during surveys of spotted owls. Such information is likely to underestimate the abundance (and ecological influence) of barred owls, and the slow replacement of spotted owls by barred owls further highlights the need to better understand how barred owls may be influencing broader ecological processes throughout the park. Improved monitoring of barred owls would lead to a better understanding of the extent of barred owl presence in OLYM. Automated field recorders are a promising new technology that could be used as a cost-effective way to track occupancy of spotted and barred owls, in addition to many other avian species. In addition, targeted demographic studies of barred owls would help clarify survival, recruitment, and population growth rate dynamics in OLYM. Second, little is known about the effects of barred owl population expansion on prey species and the broader community. As generalist and opportunistic predators (Hamer et al. 2001, Wiens et al. 2014) that reach higher natural densities than spotted owls (Wiens et al. 2011, 2014, Wiens et al. 2016), barred owls in OLYM may have consequential impacts on prey communities (Holm et al. 2016): their functional role may be different than the spotted owls they are replacing. Therefore, research and monitoring that examines the prey base in areas used by barred owls versus spotted owls would help understand the potential long-term consequences of barred owl expansion on the larger ecosystem.

In addition to new monitoring tools, barred owl expansion in OLYM raises the difficult question of management intervention. A current study is now underway to determine if experimental removal of barred owls can benefit northern spotted owls in Washington and Oregon, with initial results expected by 2019 (Wiens et al. 2016). This approach requires the successful removal of enough barred owls to allow spotted owls to recolonize usurped historical territories. An important consideration to any type of control effort for barred owls (either experimental or via management) is whether enough spotted owls remain on the landscape to reclaim historical territories once barred owls have been removed. At present, few northern spotted owls have been detected in the park relative to historical levels, but it is ultimately unknown if there are enough spotted owls to achieve recolonization in OLYM if barred owls were to be removed. A second challenge is the geographical scale of the barred owl's colonization capabilities. Because barred owls are ubiquitous in the west, there will always be new barred owls present to replace those that might be removed. Therefore, if the park were successful in controlling barred owl presence in OLYM via trapping or lethal control, the northern spotted owl could become the classic case of a conservation-reliant species (Scott et al. 2005, Scott et al. 2010, Goble et al. 2012); that is, one whose existence is wholly dependent on continuous human action. Experimental studies can address the short-term success of barred owl removal, but long-term persistence of viable, healthy northern spotted owl populations at larger geographic scales is less certain.

5.5.2. Balsam Woolly Adelgid (Subalpine Fir)

The assessment of BWA spread in OLYM demonstrated the challenges in detecting and assessing the impact of forest insect pest species. BWA, a European insect species that slowly kills fir trees over multiple years, is now widespread on the Olympic Peninsula. It has been present in North America since the early 1900s, reached the southern Olympic Peninsula in 1955, invaded OLYM by 1970, and reached peak severity in 2007. Despite long-term awareness of this forest pest in Pacific Northwest forests, the damage to subalpine fir forests of OLYM was not appreciated until it had become widespread and visible from the air. BWA is now pervasive in the subalpine fir forests of OLYM, and this delay in detection highlights a challenge for OLYM and other forested parks dealing with potential pest infestations in the future. Control at a large scale is not possible once invasive pests are established, suggesting that emphasis be placed on prevention of future invasions. Such prevention can be achieved by (1) increasing collaboration among management agencies to facilitate timely awareness of species spread across different land management jurisdictions, and (2) educating the public regarding inadvertent transport of infested wood products or on people or vehicles.

Now that BWA is part of OLYM's ecosystems, it is also important to determine how it will affect subalpine fir ecosystems, including the effect of dead trees on fire size and intensity, and how effects might be altered under different temperature and moisture regimes. As with the other resources examined in this assessment, one invasive species can nearly eliminate a host or competing native species and dramatically change an ecosystem. For future ecosystem restoration, managers may want to locate areas where subalpine fir trees appear to tolerate BWA either due to genetics or environmental conditions. Cones from tolerant or resistant trees could be collected for breeding and planting programs. The conditions at these locations may also indicate important refuge areas; evidence indicates that BWA impacts increase during warm periods making refuges even more critical with a warming climate. Because subalpine fir is faced with the double challenge of an invasive pest and climate change, managers may want to consult climate projections and use spatially explicit models to understand the future range of subalpine fir. It is possible that subalpine fir will regenerate in some areas while also experiencing periodic BWA outbreaks tied to climate cycles.

5.5.3. Invasive Non-native Plants (Native Plants)

Non-native plant invasions have been nationally recognized as a threat to the NPS mission to protect natural and cultural resources and make them available for future generations (Allen et al. 2009). On the Olympic Peninsula, the presence (Jones 1936) and potential impacts (Albright 1933) of non-native plant species have been recognized for over a century, yet we still know relatively little about the distribution and abundance of the vast majority of the invasive non-native plant species found in OLYM. We know even less about how these species may be impacting ecological function and native species' ecology in the park. The assessment documented a continuous increase in the number of non-native plants on the peninsula and in OLYM over time. Georeferenced records indicated that the greatest infestation of most high-priority park species of concern has occurred in developed areas and along roads. Finally, limited repeat surveys of select areas show increases in some invasive species and decreases in others, possibly due to successful control efforts in the latter case. These limited data highlight a need for better spatial and temporal monitoring data to quantify abundance and spread of key invasive non-native species. This could be accomplished by: (1) designing consistent inventory and monitoring protocols for focal areas in OLYM to understand the abundance and spread of invasive plant species; (2) using citizen science to collect opportunistic and targeted monitoring data on invasive species found in the park interior; and (3) integrating existing historical data to provide a more complete picture of what is currently known about the spatial extent of invasive species.

Concomitant with efforts to better understand the abundance and spatial extent of invasive non-native species over time, targeted studies could be designed to understand ecological impacts of invasive species on native communities. While there are examples of localized control and prevention of invasive plant establishment or spread, most of the invasive plant species documented and mapped in this assessment will always be part of OLYM's plant community. Therefore, recognizing and quantifying how they may shape or affect ecosystems can help prioritize management efforts and help manage for change. For example, how are invasive non-native species impacting ecosystem structure and function? Is there evidence of native species decline? Are invasive species affecting soil quality or habitat for species having higher trophic levels? Finally, invasive plant species spread and impact could be altered with climate change (Hellmann et al. 2008, Pauchard et al. 2009, Pejchar and Mooney 2009, Walther et al. 2009). Therefore, understanding the effect of invasive non-native species on ecosystem function in these changing environments will be important to making appropriate decisions about their management and their roles in various park ecosystems.

Finally, the assessment of the effort to map, monitor, and control invasive non-native plant establishment on the former Lake Mills reservoir on the Elwha River has highlighted the ability for the park to prevent spread in certain cases rather than waiting to control invasive species after they become established. This project demonstrated how concerted effort at the outset of a massive ground-disturbing project can help prevent establishment of invasive species of interest, and how monitoring can help track invasive species location data and allow for targeted control efforts. It remains to be seen whether the effort can be sustained to achieve long-term success. While this project has involved significant investment and results are not final, it showcases the park's ability to successfully prevent or control invasive species spread in key areas, at least for the short-term. This type of project shows how the park might be able to target early infestations or prevent invasive non-naitve species establishment in other areas that are being restored or that have the high potential for increase of invasive species.

In summary, invasive non-native plant species – like forest insect pests and animal invaders – represent a persistent threat to OLYM's native ecosystems. In certain cases, targeted efforts may be able to control or remove invaders from key locations or in association with certain projects. However, in most cases, there is little we can do to remove non-native species. Therefore, detecting new invaders, documenting and limiting their spread, understanding their ecological impacts, and continuing to manage for change will allow OLYM to best contend with the changes wrought by invasive species in the park.

5.6. What are the ecosystem effects of airborne contaminants?

Rebecca McCaffery and Tonnie Cummings

The park has a mandate to retain relatively pristine air conditions, yet the processes that might affect air quality and contaminant deposition are far-reaching. Given its location and the predominant air flow off the Pacific Ocean, there is a perception that OLYM has excellent air quality. However, extensive monitoring and research show that is clearly not the case. Regional and indeed global air circulation patterns, industry, and human use of landscapes can all have repercussions on the air quality and contaminants measured in OLYM. For example, Trans-Pacific air pollution is increasing (Lin et al. 2014), and the future trends in emissions from countries like China are unknown. Mercury deposition in OLYM is also impacted by trans-Pacific transport of atmospheric mercury (Weiss-Penzias et al. 2016). Furthermore, some air quality metrics that are currently stable may change with climate change. For example, changes in temperature and precipitation patterns could affect the impacts of nitrogen deposition on ecosystem function (Cummings et al. 2014).

In the assessment of seven air quality and airborne contaminant measures, conditions varied substantially, providing guidance on which areas of air quality are of most concern to the park. Arguably, the visibility provisions of the Clean Air Act provide the best opportunity to protect park air quality. To that end, continued operation of the Interagency Monitoring of Protected Visual Environments Program visibility site can help OLYM track progress in achieving Regional Haze Rule goals. While visibility is of concern, current trends show improved visibility on the worst days and no change over time in visibility measures on good days. Ozone is not currently an issue for the park, showing no threat to human or vegetation health, and no trends over time. However, ozone levels may increase with increased temperatures due to climate change (USEPA 2009), so continued ozone monitoring at nearby Cheeka Peak is recommended.

In addition to visibility, the effects of air pollution on OLYM's ecological resources via atmospheric deposition are an ongoing concern. Data indicate high-elevation vegetation and lakes, in particular, are nitrogen-sensitive, and that nitrogen deposition may already have exceeded critical levels.

Additional work is needed to quantify nitrogen deposition in the park and to refine nitrogen critical loads for OLYM. Sulfur deposition also exceeds ARD benchmarks and has shown an increasing trend over time. However, at this time, nitrogen deposition is considered a greater threat than sulfur deposition to Pacific Northwest ecosystems. If that changes, sulfur critical loads should be determined for OLYM. Elevated concentrations of mercury have been detected in fish, birds, and invertebrates collected throughout the park, but additional studies are needed to determine if biota are being adversely affected. Significant questions also remain regarding the extent and effects of non-mercury persistent bioaccumulative toxins at OLYM. These issues point to the value of continued monitoring of airborne pollutants via the NADP and more targeted studies.

Many of the trends in airborne contaminants are outside of the park’s control, but continued monitoring of these pollutants alongside focused studies can increase our understanding of how park ecosystems are affected by these stressors. Specifically, targeted monitoring and research could address: (1) how elevated levels of these contaminants are affecting individual species and ecosystem function; (2) how contaminant levels (such as mercury) vary spatially, and what the implications are for OLYM’s ecosystems; (3) how different contaminants may interact, leading to synergistic effects on ecosystems; and (4) how effectively ecosystems can respond if contaminant levels are reduced. While future trends and dominant sources of air pollution are unknown, OLYM can play a role in understanding how these contaminants impact the protected landscapes that we value.

5.7. Conclusions

In this “Age of the Anthropocene”, we use, manage, protect, and study systems that have almost inevitably been altered by human activity. Olympic National Park was established during a series of human-caused perturbations that indelibly shaped the lands of the peninsula. Climate change, airborne contaminants, and invasive species are now affecting ecosystems in new ways, even the places that we consider to be the most protected on the planet. And many of those protected, pristine places have legacies that pre-date their protected status. In this assessment, we have evaluated the condition of twelve resources as they relate to various ecosystem stressors, and synthesized that information in a way that can assist park managers in developing resource management plans.

Not surprisingly, our analyses have revealed complex systems with numerous interacting factors affecting the resources of interest. While enlightening in some places, our analyses have also revealed processes underway (e.g., due to ongoing climate change) that may be largely out of our control to address or manage in meaningful ways. In other places, the park may have difficult decisions to make. Our intention was not to determine these future steps but to provide analyses and syntheses that can help further the discussion of resource management priorities and viable stewardship strategies for the future.

Blue Glacier, Olympic National Park. Photograph courtesy of NPS.

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Appendix A. Species lists for Olympic National Park, including mammals, birds, fishes, amphibians, and reptiles.

Table A.1. Mammals of Olympic National Park.

Order	Family	Scientific Name	Common Name
Artiodactyla	Bovidae	*Oreamnos americanus*[1]	mountain goat
Artiodactyla	Cervidae	*Cervus elaphus roosevelti*	Roosevelt elk
Artiodactyla	Cervidae	*Odocoileus hemionus columbianus*	Columbian black-tailed deer
Carnivora	Canidae	*Canis latrans*	coyote
Carnivora	Canidae	*Vulpes vulpes*[1]	red fox
Carnivora	Felidae	*Lynx rufus rufus*	bobcat
Carnivora	Felidae	*Puma concolor oregonensis*	cougar
Carnivora	Mephitidae	*Mephitis mephitis*	striped skunk
Carnivora	Mephitidae	*Spilogale putorius*	spotted skunk
Carnivora	Mustelidae	*Enhydra lutris*	sea otter
Carnivora	Mustelidae	*Lontra canadensis*	river otter
Carnivora	Mustelidae	*Martes caurina*	marten
Carnivora	**Mustelidae**	***Mustela erminea olympica*[2]**	**Olympic ermine**
Carnivora	Mustelidae	*Mustela frenata*	long-tailed weasel
Carnivora	Mustelidae	*Mustela vison*	American mink
Carnivora	Mustelidae	*Pekania pennanti*	fisher
Carnivora	Otariidae	*Callorhinus ursinus*	northern fur seal
Carnivora	Otariidae	*Eumetopias jubatus*	Steller sea-lion
Carnivora	Otariidae	*Zalophus californianus*	California sea-lion
Carnivora	Phocidae	*Mirounga angustirostris*	northern elephant seal
Carnivora	Phocidae	*Phoca vitulina*	harbor seal
Carnivora	Procyonidae	*Procyon lotor*	raccoon
Carnivora	Ursidae	*Ursus americanus*	black bear
Chiroptera	Vespertilionidae	*Corynorhinus townsendii*	western big-eared bat
Chiroptera	Vespertilionidae	*Eptesicus fuscus*	big brown bat
Chiroptera	Vespertilionidae	*Lasionycteris noctivagans*	silver-haired bat
Chiroptera	Vespertilionidae	*Lasiurus cinereus cinereus*	hoary bat
Chiroptera	Vespertilionidae	*Myotis californicus*	California myotis
Chiroptera	Vespertilionidae	*Myotis evotis*	long-eared myotis
Chiroptera	Vespertilionidae	*Myotis lucifugus*	little brown bat
Chiroptera	Vespertilionidae	*Myotis volans*	long-legged myotis
Chiroptera	Vespertilionidae	*Myotis yumanensis*	Yuma myotis

[1] Non-native species.

[2] Endemic species or subspecies (also shown in bold text).

Table A.1 (continued). Mammals of Olympic National Park.

Order	Family	Scientific Name	Common Name
Lagomorpha	Leporidae	*Lepus americanus*	snowshoe rabbit
Rodentia	Aplodontiidae	*Aplodontia rufa*	mountain beaver
Rodentia	Castoridae	*Castor canadensis*	American beaver
Rodentia	Dipodidae	*Zapus trinotatus trinotatus*	Pacific jumping mouse
Rodentia	Erethizontidae	*Erethizon dorsatum*	porcupine
Rodentia	**Geomyidae**	***Thomomys mazama melanops***[2]	**Olympic mazama pocket gopher**
Rodentia	Muridae	*Myodes gapperi*	gapper red-backed mouse
Rodentia	Muridae	*Microtus longicaudus*	long-tailed vole
Rodentia	Muridae	*Microtus oregoni*	creeping vole
Rodentia	Muridae	*Microtus townsendii*	Townsend's vole
Rodentia	Muridae	*Mus musculus*[1]	house mouse
Rodentia	Muridae	*Neotoma cinerea*	bushy-tailed woodrat
Rodentia	Muridae	*Ondatra zibethicus*	muskrat
Rodentia	Muridae	*Peromyscus keeni*	Keen's mouse
Rodentia	Muridae	*Peromyscus maniculatus*	white-footed mouse
Rodentia	Muridae	*Phenacomys intermedius*	heather vole
Rodentia	Muridae	*Rattus norvegicus*[1]	Norway rat
Rodentia	Muridae	*Rattus rattus*[1]	black rat
Rodentia	Sciuridae	*Glaucomys sabrinus*	flying squirrel
Rodentia	**Sciuridae**	***Marmota olympus***[2]	**Olympic marmot**
Rodentia	**Sciuridae**	***Tamias amoenus caurinus***[2]	**Olympic yellow-pine chipmunk**
Rodentia	Sciuridae	*Tamias townsendii*	Townsend chipmunk
Rodentia	Sciuridae	*Tamiasciurus douglasii*	Douglas' squirrel
Soricomorpha	Soricidae	*Sorex bendirii*	Pacific water shrew
Soricomorpha	Soricidae	*Sorex cinereus*	masked shrew
Soricomorpha	Soricidae	*Sorex monticolus*	dusky shrew
Soricomorpha	Soricidae	*Sorex palustris*	water shrew
Soricomorpha	Soricidae	*Sorex trowbridgii*	Trowbridge's shrew
Soricomorpha	Soricidae	*Sorex vagrans*	wandering shrew
Soricomorpha	Talpidae	*Neurotrichus gibbsii*	shrew-mole
Soricomorpha	**Talpidae**	***Scapanus orarius***[2] **(Olympic clade)**	**coast mole**
Soricomorpha	Talpidae	*Scapanus townsendii*	Townsend's Mole

[1] Non-native species.

[2] Endemic species or subspecies (also shown in bold text).

Table A.2. Birds of Olympic National Park. This table includes all species ever seen in the park, including vagrant species with single records.

Order	Family	Scientific Name	Common Name
Accipitriformes	Accipitridae	*Accipiter cooperii*	Cooper's Hawk
Accipitriformes	Accipitridae	*Accipiter gentilis*	Northern Goshawk
Accipitriformes	Accipitridae	*Accipiter striatus*	Sharp-shinned Hawk
Accipitriformes	Accipitridae	*Aquila chrysaetos*	Golden Eagle
Accipitriformes	Accipitridae	*Buteo jamaicensis*	Red-tailed Hawk
Accipitriformes	Accipitridae	*Buteo lagopus*	Rough-legged Hawk
Accipitriformes	Accipitridae	*Buteo platypterus*	Broad-winged Hawk
Accipitriformes	Accipitridae	*Circus cyaneus*	Northern Harrier
Accipitriformes	Accipitridae	*Haliaeetus leucocephalus*	Bald Eagle
Accipitriformes	Cathartidae	*Cathartes aura*	Turkey Vulture
Accipitriformes	Pandionidae	*Pandion haliaetus*	Osprey
Anseriformes	Anatidae	*Aix sponsa*	Wood Duck
Anseriformes	Anatidae	*Anas acuta*	Northern Pintail
Anseriformes	Anatidae	*Anas americana*	American Wigeon
Anseriformes	Anatidae	*Anas clypeata*	Northern Shoveler
Anseriformes	Anatidae	*Anas crecca*	Green-winged Teal
Anseriformes	Anatidae	*Anas cyanoptera*	Cinnamon Teal
Anseriformes	Anatidae	*Anas discors*	Blue-winged Teal
Anseriformes	Anatidae	*Anas penelope*	Eurasian Wigeon
Anseriformes	Anatidae	*Anas platyrhynchos*	Mallard
Anseriformes	Anatidae	*Anas strepera*	Gadwall
Anseriformes	Anatidae	*Anser albifrons*	Greater White-fronted Goose
Anseriformes	Anatidae	*Aythya affinis*	Lesser Scaup
Anseriformes	Anatidae	*Aythya americana*	Redhead
Anseriformes	Anatidae	*Aythya collaris*	Ring-necked Duck
Anseriformes	Anatidae	*Aythya marila*	Greater Scaup
Anseriformes	Anatidae	*Aythya valisineria*	Canvasback
Anseriformes	Anatidae	*Branta bernicla*	Brant
Anseriformes	Anatidae	*Branta canadensis*	Canada Goose
Anseriformes	Anatidae	*Branta hutchinsii*	Cackling Goose
Anseriformes	Anatidae	*Bucephala albeola*	Bufflehead
Anseriformes	Anatidae	*Bucephala clangula*	Common Goldeneye
Anseriformes	Anatidae	*Bucephala islandica*	Barrow's Goldeneye
Anseriformes	Anatidae	*Clangula hyemalis*	Long-tailed Duck
Anseriformes	Anatidae	*Cygnus buccinator*	Trumpeter Swan

[1] Non-native species.

[2] Endemic species

Table A.2 (continued). Birds of Olympic National Park. This table includes all species ever seen in the park, including vagrant species with single records.

Order	Family	Scientific Name	Common Name
Anseriformes	Anatidae	*Cygnus columbianus*	Tundra Swan
Anseriformes	Anatidae	*Histrionicus histrionicus*	Harlequin Duck
Anseriformes	Anatidae	*Lophodytes cucullatus*	Hooded Merganser
Anseriformes	Anatidae	*Melanitta fusca*	White-winged Scoter
Anseriformes	Anatidae	*Melanitta nigra*	Black Scoter
Anseriformes	Anatidae	*Melanitta perspicillata*	Surf Scoter
Anseriformes	Anatidae	*Mergus merganser*	Common Merganser
Anseriformes	Anatidae	*Mergus serrator*	Red-breasted Merganser
Anseriformes	Anatidae	*Oxyura jamaicensis*	Ruddy Duck
Apodiformes	Apodidae	*Chaetura vauxi*	Vaux's Swift
Apodiformes	Apodidae	*Cypseloides niger*	Black Swift
Apodiformes	Trochilidae	*Calypte anna*	Anna's Hummingbird
Apodiformes	Trochilidae	*Selasphorus rufus*	Rufous Hummingbird
Caprimulgiformes	Caprimulgidae	*Chordeiles minor*	Common Nighthawk
Charadriiformes	Alcidae	*Brachyramphus marmoratus*	Marbled Murrelet
Charadriiformes	Alcidae	*Cepphus columba*	Pigeon Guillemot
Charadriiformes	Alcidae	*Cerorhinca monocerata*	Rhinoceros Auklet
Charadriiformes	Alcidae	*Fratercula cirrhata*	Tufted Puffin
Charadriiformes	Alcidae	*Ptychoramphus aleuticus*	Cassin's Auklet
Charadriiformes	Alcidae	*Synthliboramphus antiquus*	Ancient Murrelet
Charadriiformes	Alcidae	*Uria aalge*	Common Murre
Charadriiformes	Charadriidae	*Charadrius semipalmatus*	Semipalmated Plover
Charadriiformes	Charadriidae	*Charadrius vociferus*	Killdeer
Charadriiformes	Charadriidae	*Pluvialis fulva*	Pacific Golden-Plover
Charadriiformes	Charadriidae	*Pluvialis squatarola*	Black-bellied Plover
Charadriiformes	Haematopodidae	*Haematopus bachmani*	Black Oystercatcher
Charadriiformes	Laridae	*Larus argentatus*	Herring Gull
Charadriiformes	Laridae	*Larus californicus*	California Gull
Charadriiformes	Laridae	*Larus canus*	Mew Gull
Charadriiformes	Laridae	*Larus delawarensis*	Ring-billed Gull
Charadriiformes	Laridae	*Larus glaucescens*	Glaucous-winged Gull
Charadriiformes	Laridae	*Larus heermanni*	Heermann's Gull
Charadriiformes	Laridae	*Larus hyperboreus*	Glaucous Gull
Charadriiformes	Laridae	*Larus occidentalis*	Western Gull

[1] Non-native species.

[2] Endemic species

Table A.2 (continued). Birds of Olympic National Park. This table includes all species ever seen in the park, including vagrant species with single records.

Order	Family	Scientific Name	Common Name
Charadriiformes	Laridae	*Larus occidentalis X glaucescens*	Olympic Gull
Charadriiformes	Laridae	*Larus philadelphia*	Bonaparte's Gull
Charadriiformes	Laridae	*Larus pipixcan*	Franklin's Gull
Charadriiformes	Laridae	*Larus schistisagus*	Slaty-backed Gull
Charadriiformes	Laridae	*Larus thayeri*	Island (Thayer's) Gull
Charadriiformes	Laridae	*Rissa tridactyla*	Black-legged Kittiwake
Charadriiformes	Laridae	*Sterna caspia*	Caspian Tern
Charadriiformes	Laridae	*Sterna hirundo*	Common Tern
Charadriiformes	Laridae	*Sterna paradisaea*	Arctic Tern
Charadriiformes	Laridae	*Sterna forsteri*	Forster’s Tern
Charadriiformes	Laridae	*Thalasseus elegans*	Elegant Tern
Charadriiformes	Scolopacidae	*Actitis macularia*	Spotted Sandpiper
Charadriiformes	Scolopacidae	*Aphriza virgata*	Surfbird
Charadriiformes	Scolopacidae	*Arenaria interpres*	Ruddy Turnstone
Charadriiformes	Scolopacidae	*Arenaria melanocephala*	Black Turnstone
Charadriiformes	Scolopacidae	*Calidris alba*	Sanderling
Charadriiformes	Scolopacidae	*Calidris alpina*	Dunlin
Charadriiformes	Scolopacidae	*Calidris bairdii*	Baird's Sandpiper
Charadriiformes	Scolopacidae	*Calidris canutus*	Red Knot
Charadriiformes	Scolopacidae	*Calidris mauri*	Western Sandpiper
Charadriiformes	Scolopacidae	*Calidris melanotos*	Pectoral Sandpiper
Charadriiformes	Scolopacidae	*Calidris minutilla*	Least Sandpiper
Charadriiformes	Scolopacidae	*Calidris ptilocnemis*	Rock Sandpiper
Charadriiformes	Scolopacidae	*Calidris pusilla*	Semipalmated Sandpiper
Charadriiformes	Scolopacidae	*Gallinago gallinago*	Common Snipe
Charadriiformes	Scolopacidae	*Heteroscelus incanus*	Wandering Tattler
Charadriiformes	Scolopacidae	*Limnodromus griseus*	Short-billed Dowitcher
Charadriiformes	Scolopacidae	*Limnodromus scolopaceus*	Long-billed Dowitcher
Charadriiformes	Scolopacidae	*Limosa fedoa*	Marbled Godwit
Charadriiformes	Scolopacidae	*Numenius phaeopus*	Whimbrel
Charadriiformes	Scolopacidae	*Phalaropus fulicaria*	Red Phalarope
Charadriiformes	Scolopacidae	*Phalaropus lobatus*	Red-necked Phalarope
Charadriiformes	Scolopacidae	*Tringa flavipes*	Lesser Yellowlegs
Charadriiformes	Scolopacidae	*Tringa melanoleuca*	Greater Yellowlegs

[1] Non-native species.

[2] Endemic species

Table A.2 (continued). Birds of Olympic National Park. This table includes all species ever seen in the park, including vagrant species with single records.

Order	Family	Scientific Name	Common Name
Charadriiformes	Scolopacidae	*Tringa solitaria*	Solitary Sandpiper
Charadriiformes	Stercorariidae	*Stercorarius longicaudus*	Long-tailed Jaeger
Charadriiformes	Stercorariidae	*Stercorarius parasiticus*	Parasitic Jaeger
Charadriiformes	Stercorariidae	*Stercorarius pomarinus*	Pomarine Jaeger
Columbiformes	Columbidae	*Columba livia*[1]	Rock Pigeon
Columbiformes	Columbidae	*Patagioenas fasciata*	Band-tailed Pigeon
Columbiformes	Columbidae	*Streptopelia decaocto*[1]	Eurasian Collared-Dove
Columbiformes	Columbidae	*Zenaida macroura*	Mourning Dove
Coraciiformes	Alcedinidae	*Ceryle alcyon*	Belted Kingfisher
Falconiformes	Falconidae	*Falco columbarius columbarius*	Merlin (tundra subspecies)
Falconiformes	Falconidae	*Falco columbarius suckleyi*	Merlin (coastal forest subspecies)
Falconiformes	Falconidae	*Falco peregrinus*	Peregrine Falcon
Falconiformes	Falconidae	*Falco sparverius*	American Kestrel
Galliformes	Odontophoridae	*Callipepla californica*	California Quail
Galliformes	Phasianidae	*Bonasa umbellus castanea*	Ruffed Grouse
Galliformes	Phasianidae	*Dendragapus obscurus*	Sooty (Blue) Grouse
Galliformes	Phasianidae	*Phasianus colchicus*[1]	Ring-necked Pheasant
Gaviiformes	Gaviidae	*Gavia immer*	Common Loon
Gaviiformes	Gaviidae	*Gavia pacifica*	Pacific Loon
Gaviiformes	Gaviidae	*Gavia stellata*	Red-throated Loon
Gruiformes	Gruidae	*Grus canadensis*	Sandhill Crane
Gruiformes	Rallidae	*Fulica americana*	American Coot
Gruiformes	Rallidae	*Rallus limicola*	Virginia Rail
Passeriformes	Aegithalidae	*Psaltriparus minimus*	Bushtit
Passeriformes	Alaudidae	*Eremophila alpestris*	Horned Lark
Passeriformes	Bombycillidae	*Bombycilla cedrorum*	Cedar Waxwing
Passeriformes	Bombycillidae	*Bombycilla garrulus*	Bohemian Waxwing
Passeriformes	Cardinalidae	*Passerina amoena*	Lazuli Bunting
Passeriformes	Cardinalidae	*Passerina cyanea*	Indigo Bunting
Passeriformes	Cardinalidae	*Pheucticus melanocephalus*	Black-headed Grosbeak
Passeriformes	Certhiidae	*Certhia americana*	Brown Creeper
Passeriformes	Cinclidae	*Cinclus mexicanus*	American Dipper
Passeriformes	Corvidae	*Corvus brachyrhynchos*	American Crow
Passeriformes	Corvidae	*Corvus caurinus*	Northwestern Crow
Passeriformes	Corvidae	*Corvus corax*	Common Raven

[1] Non-native species.

[2] Endemic species

Table A.2 (continued). Birds of Olympic National Park. This table includes all species ever seen in the park, including vagrant species with single records.

Order	Family	Scientific Name	Common Name
Passeriformes	Corvidae	*Cyanocitta stelleri*	Steller's Jay
Passeriformes	Corvidae	*Nucifraga columbiana*	Clark's Nutcracker
Passeriformes	Corvidae	*Perisoreus canadensis obscurus*	Gray Jay
Passeriformes	Emberizidae	*Calcarius lapponicus*	Lapland Longspur
Passeriformes	Emberizidae	*Junco hyemalis*	Dark-eyed Junco
Passeriformes	Emberizidae	*Melospiza georgiana*	Swamp Sparrow
Passeriformes	Emberizidae	*Melospiza lincolnii*	Lincoln's Sparrow
Passeriformes	Emberizidae	*Melospiza melodia*	Song Sparrow
Passeriformes	Emberizidae	*Passerculus sandwichensis*	Savannah Sparrow
Passeriformes	Emberizidae	*Passerella iliaca*	Fox Sparrow
Passeriformes	Emberizidae	*Pipilo maculatus*	Spotted Towhee
Passeriformes	Emberizidae	*Plectrophenax nivalis*	Snow Bunting
Passeriformes	Emberizidae	*Pooecetes gramineus*	Vesper Sparrow
Passeriformes	Emberizidae	*Spizella passerina*	Chipping Sparrow
Passeriformes	Emberizidae	*Zonotrichia albicollis*	White-throated Sparrow
Passeriformes	Emberizidae	*Zonotrichia atricapilla*	Golden-crowned Sparrow
Passeriformes	Emberizidae	*Zonotrichia leucophrys*	White-crowned Sparrow
Passeriformes	Fringillidae	*Carduelis flammea*	Common Redpoll
Passeriformes	Fringillidae	*Carduelis pinus*	Pine Siskin
Passeriformes	Fringillidae	*Carduelis tristis*	American Goldfinch
Passeriformes	Fringillidae	*Carpodacus cassinii*	Cassin's Finch
Passeriformes	Fringillidae	*Carpodacus mexicanus*	House Finch
Passeriformes	Fringillidae	*Carpodacus purpureus*	Purple Finch
Passeriformes	Fringillidae	*Coccothraustes vespertinus*	Evening Grosbeak
Passeriformes	Fringillidae	*Leucosticte tephrocotis*	Gray-crowned Rosy-Finch
Passeriformes	Fringillidae	*Loxia curvirostra*	Red Crossbill
Passeriformes	Fringillidae	*Loxia leucoptera*	White-winged Crossbill
Passeriformes	Fringillidae	*Pinicola enucleator*	Pine Grosbeak
Passeriformes	Hirundinidae	*Hirundo rustica*	Barn Swallow
Passeriformes	Hirundinidae	*Petrochelidon pyrrhonota*	American Cliff Swallow
Passeriformes	Hirundinidae	*Stelgidopteryx serripennis*	Northern Rough-winged Swallow
Passeriformes	Hirundinidae	*Tachycineta bicolor*	Tree Swallow
Passeriformes	Hirundinidae	*Tachycineta thalassina*	Violet-green Swallow
Passeriformes	Icteridae	*Agelaius phoeniceus*	Red-winged Blackbird

[1] Non-native species.

[2] Endemic species

Table A.2 (continued). Birds of Olympic National Park. This table includes all species ever seen in the park, including vagrant species with single records.

Order	Family	Scientific Name	Common Name
Passeriformes	Icteridae	*Euphagus cyanocephalus*	Brewer's Blackbird
Passeriformes	Icteridae	*Icterus galbula*	Northern Oriole
Passeriformes	Icteridae	*Molothrus ater*	Brown-headed Cowbird
Passeriformes	Icteridae	*Sturnella neglecta*	Western Meadowlark
Passeriformes	Icteridae	*Xanthocephalus xanthocephalus*	Yellow-headed Blackbird
Passeriformes	Laniidae	*Lanius excubitor*	Northern Shrike
Passeriformes	Mimidae	*Mimus polyglottos*	Northern Mockingbird
Passeriformes	Motacillidae	*Anthus rubescens*	American Pipit
Passeriformes	Paridae	*Poecile atricapillus*	Black-capped Chickadee
Passeriformes	Paridae	*Poecile rufescens*	Chestnut-backed Chickadee
Passeriformes	Parulidae	*Dendroica caerulescens*	Black-throated Blue Warbler
Passeriformes	Parulidae	*Dendroica coronata auduboni*	Yellow-rumped Warber (Audubon's)
Passeriformes	Parulidae	*Dendroica coronata coronata*	Yellow-rumped Warbler (Myrtle)
Passeriformes	Parulidae	*Dendroica nigrescens*	Black-throated Gray Warbler
Passeriformes	Parulidae	*Dendroica occidentalis*	Hermit Warbler
Passeriformes	Parulidae	*Dendroica petechia*	Yellow Warbler
Passeriformes	Parulidae	*Dendroica townsendi*	Townsend's Warbler
Passeriformes	Parulidae	*Geothlypis trichas*	Common Yellowthroat
Passeriformes	Parulidae	*Oporornis tolmiei*	Macgillivray's Warbler
Passeriformes	Parulidae	*Parula americana*	Northern Parula
Passeriformes	Parulidae	*Setophaga ruticilla*	American Redstart
Passeriformes	Parulidae	*Vermivora celata*	Orange-crowned Warbler
Passeriformes	Parulidae	*Wilsonia pusilla*	Wilson's Warbler
Passeriformes	Passeridae	*Passer domesticus*[1]	House Sparrow
Passeriformes	Regulidae	*Regulus calendula*	Ruby-crowned Kinglet
Passeriformes	Regulidae	*Regulus satrapa*	Golden-crowned Kinglet
Passeriformes	Sittidae	*Sitta canadensis*	Red-breasted Nuthatch
Passeriformes	Sturnidae	*Sturnus vulgaris*[1]	European Starling
Passeriformes	Thraupidae	*Piranga ludoviciana*	Western Tanager
Passeriformes	Troglodytidae	*Cistothorus palustris*	Marsh Wren
Passeriformes	Troglodytidae	*Salpinctes obsoletus*	Rock Wren
Passeriformes	Troglodytidae	*Thryomanes bewickii*	Bewick's Wren
Passeriformes	Troglodytidae	*Troglodytes aedon*	House Wren

[1] Non-native species.

[2] Endemic species

Table A.2 (continued). Birds of Olympic National Park. This table includes all species ever seen in the park, including vagrant species with single records.

Order	Family	Scientific Name	Common Name
Passeriformes	Troglodytidae	*Troglodytes pacificus*	Pacific (Winter) Wren
Passeriformes	Turdidae	*Catharus guttatus*	Hermit Thrush
Passeriformes	Turdidae	*Catharus ustulatus*	Swainson's Thrush
Passeriformes	Turdidae	*Ixoreus naevius*	Varied Thrush
Passeriformes	Turdidae	*Myadestes townsendi*	Townsend's Solitaire
Passeriformes	Turdidae	*Sialia currucoides*	Mountain Bluebird
Passeriformes	Turdidae	*Sialia mexicana*	Western Bluebird
Passeriformes	Turdidae	*Turdus migratorius*	American Robin
Passeriformes	Tyrannidae	*Contopus cooperi*	Olive-sided flycatcher
Passeriformes	Tyrannidae	*Contopus sordidulus*	Western Wood-Pewee
Passeriformes	Tyrannidae	*Empidonax difficilis*	Pacific-slope Flycatcher
Passeriformes	Tyrannidae	*Empidonax hammondii*	Hammond's Flycatcher
Passeriformes	Tyrannidae	*Empidonax oberholseri*	Dusky Flycatcher
Passeriformes	Tyrannidae	*Empidonax traillii*	Willow Flycatcher
Passeriformes	Tyrannidae	*Sayornis phoebe*	Eastern Phoebe
Passeriformes	Tyrannidae	*Tyrannus tyrannus*	Eastern Kingbird
Passeriformes	Tyrannidae	*Tyrannus melancholicus*	Tropical Kingbird
Passeriformes	Vireonidae	*Vireo cassinii*	Cassin's Vireo
Passeriformes	Vireonidae	*Vireo gilvus*	Warbling Vireo
Passeriformes	Vireonidae	*Vireo huttoni*	Hutton's Vireo
Pelecaniformes	Ardeidae	*Ardea herodias*	Great Blue Heron
Pelecaniformes	Ardeidae	*Bubulcus ibis*	Cattle Egret
Pelecaniformes	Ardeidae	*Butorides virescens*	Green-backed Heron
Pelecaniformes	Pelecanidae	*Pelecanus erythrorhynchos*	American White Pelican
Pelecaniformes	Pelecanidae	*Pelecanus occidentalis*	Brown Pelican
Piciformes	Picidae	*Colaptes auratus*	Northern Flicker
Piciformes	Picidae	*Dryocopus pileatus*	Pileated Woodpecker
Piciformes	Picidae	*Melanerpes formicivorus*	Acorn Woodpecker
Piciformes	Picidae	*Picoides arcticus*	Black-backed Woodpecker
Piciformes	Picidae	*Picoides pubescens*	Downy Woodpecker
Piciformes	Picidae	*Picoides tridactylus*	American Three-toed Woodpecker
Piciformes	Picidae	*Picoides villosus*	Hairy Woodpecker
Piciformes	Picidae	*Sphyrapicus nuchalis*	Red-naped Sapsucker
Piciformes	Picidae	*Sphyrapicus ruber*	Red-breasted Sapsucker
Podicipediformes	Podicipedidae	*Aechmophorus occidentalis*	Western Grebe

[1] Non-native species.

[2] Endemic species

Table A.2 (continued). Birds of Olympic National Park. This table includes all species ever seen in the park, including vagrant species with single records.

Order	Family	Scientific Name	Common Name
Podicipediformes	Podicipedidae	*Podiceps auritus*	Horned Grebe
Podicipediformes	Podicipedidae	*Podiceps grisegena*	Red-necked Grebe
Podicipediformes	Podicipedidae	*Podilymbus podiceps*	Pied-billed Grebe
Procellariiformes	Hydrobatidae	*Oceanodroma furcata*	Fork-tailed Storm-Petrel
Procellariiformes	Hydrobatidae	*Oceanodroma leucorhoa*	Leach's Storm-Petrel
Procellariiformes	Procellariidae	*Puffinus griseus*	Sooty Shearwater
Procellariiformes	Procellariidae	*Puffinus puffinus*	Manx Shearwater
Strigiformes	Strigidae	*Aegolius acadicus*	Northern Saw-whet Owl
Strigiformes	Strigidae	*Asio flammeus*	Short-eared Owl
Strigiformes	Strigidae	*Asio otus*	Long-eared Owl
Strigiformes	Strigidae	*Bubo virginianus*	Great Horned Owl
Strigiformes	Strigidae	*Glaucidium gnoma*	Northern Pygmy-Owl
Strigiformes	Strigidae	*Nyctea scandiaca*	Snowy Owl
Strigiformes	Strigidae	*Otus kennicottii*	Western Screech-Owl
Strigiformes	Strigidae	*Strix occidentalis caurina*	Northern Spotted Owl
Strigiformes	Strigidae	*Strix varia*[1]	Barred Owl
Strigiformes	Tytonidae	*Tyto alba*	Barn Owl
Suliformes	Phalacrocoracidae	*Phalacrocorax auritus*	Double-crested Cormorant
Suliformes	Phalacrocoracidae	*Phalacrocorax pelagicus*	Pelagic Cormorant
Suliformes	Phalacrocoracidae	*Phalacrocorax penicillatus*	Brandt's Cormorant

[1] Non-native species.

[2] Endemic species

Table A.3. Fishes of Olympic National Park.

Order	Family	Scientific Name	Common Name
Clupeiformes	Clupeidae	*Clupea harengus pallasii*	Pacific herring
Clupeiformes	Clupeidae	*Sardinops sagax caeruleus*	Pacific sardine
Clupeiformes	Clupeidae	*Alosa sapidissima*[1]	American shad
Clupeiformes	Engraulidae	*Engraulis mordax*	North Pacific anchovy
Cypriniformes	Catostomidae	*Catostomus catostomus*	longnose sucker
Cypriniformes	Catostomidae	*Catostomus macrocheilus*	largescale sucker
Cypriniformes	Cyprinidae	*Mylocheilus caurinus*	peamouth chub
Cypriniformes	Cyprinidae	*Ptychocheilus oregonensis*	northern pikeminnow
Cypriniformes	Cyprinidae	*Rhinichthys cataractae*	longnose dace
Cypriniformes	Cyprinidae	*Rhinichthys osculus*	speckled dace
Cypriniformes	Cyprinidae	*Richardsonius balteatus*	redside shiner
Esociformes	**Umbridae**	***Novumbra hubbsi*[2]**	**Olympic mudminnow**
Gadiformes	Gadidae	*Microgadus proximus*	Pacific tomcod
Gasterosteiformes	Aulorhynchidae	*Aulorhynchus flavidus*	tubesnout
Gasterosteiformes	Gasterosteidae	*Gasterosteus aculeatus*	threespine stickleback
Gasterosteiformes	Syngnathidae	*Syngnathus leptorhynchus*	bay pipefish
Osmeriformes	Osmeridae	*Hypomesus pretiosus*	surf smelt
Osmeriformes	Osmeridae	*Spirinchus starksi*	night smelt
Osmeriformes	Osmeridae	*Spirinchus thaleichthys*	longfin smelt
Osmeriformes	Osmeridae	*Thaleichthys pacificus*	eulachon
Perciformes	Ammodytidae	*Ammodytes hexapterus*	Pacific sand lance
Perciformes	Centrarchidae	*Micropterus salmoides*[1]	largemouth bass
Perciformes	Clinidae	*Gibbonsia montereyensis*	crevice kelpfish
Perciformes	Embiotocidae	*Amphistichus koelzi*	calico surfperch
Perciformes	Embiotocidae	*Amphistichus rhodoterus*	redtail surfperch
Perciformes	Embiotocidae	*Cymatogaster aggregata*	shiner surfperch
Perciformes	Embiotocidae	*Damalichthys vacca*	pile surfperch
Perciformes	Embiotocidae	*Embiotoca lateralis*	striped surfperch
Perciformes	Embiotocidae	*Hyperprosopon anale*	spotfin surfperch
Perciformes	Embiotocidae	*Hyperprosopon ellipticum*	silver surfperch
Perciformes	Gobiesocidae	*Gobiesox maeandricus*	northern clingfish
Perciformes	Percidae	*Perca flavescens*[1]	yellow perch
Perciformes	Pholidae	*Apodichthys flavidus*	penpoint gunnel
Perciformes	Pholidae	*Apodichthys fucorum*	rockweed gunnel
Perciformes	Pholidae	*Pholis laeta*	crescent gunnel

[1] Non-native species

[2] Endemic species (also shown in bold text)

Table A.3 (continued). Fishes of Olympic National Park.

Order	Family	Scientific Name	Common Name
Perciformes	Pholidae	*Pholis ornata*	saddleback gunnel
Perciformes	Pholidae	*Pholis schultzi*	red gunnel
Perciformes	Stichaeidae	*Anoplarchus purpurescens*	high cockscomb
Perciformes	Stichaeidae	*Chirolophis nugator*	mosshead warbonnet
Perciformes	Stichaeidae	*Xiphister atropurpureus*	black prickleback
Perciformes	Stichaeidae	*Xiphister mucosus*	rock prickleback
Perciformes	Trichodontidae	*Trichodon trichodon*	Pacific sandfish
Petromyzontiformes	Petromyzontidae	*Lampetra richardsoni*	western brook lamprey
Petromyzontiformes	Petromyzontidae	*Lampetra tridentata*	Pacific lamprey
Pleuronectiformes	Pleuronectidae	*Platichthys stellatus*	starry flounder
Pleuronectiformes	Pleuronectidae	*Pleuronichthys decurrens*	curlfin sole
Pleuronectiformes	Pleuronectidae	*Psettichthys melanostictus*	sand sole
Salmoniformes	Salmonidae	*Oncorhynchus clarkii*	cutthroat trout
Salmoniformes	Salmonidae	*Oncorhynchus gorbuscha*	pink salmon
Salmoniformes	Salmonidae	*Oncorhynchus keta*	chum salmon
Salmoniformes	Salmonidae	*Oncorhynchus kisutch*	coho salmon
Salmoniformes	Salmonidae	*Oncorhynchus mykiss*	rainbow trout
Salmoniformes	Salmonidae	*Oncorhynchus nerka*	sockeye salmon/kokanee
Salmoniformes	Salmonidae	*Oncorhynchus tshawytscha*	chinook salmon
Salmoniformes	Salmonidae	*Prosopium coulteri*	pygmy whitefish
Salmoniformes	Salmonidae	*Prosopium williamsoni*	mountain whitefish
Salmoniformes	Salmonidae	*Salmo salar*[1]	Atlantic salmon
Salmoniformes	Salmonidae	*Salvelinus confluentus*	bull trout
Salmoniformes	Salmonidae	*Salvelinus fontinalis*[1]	brook trout
Salmoniformes	Salmonidae	*Salvelinus malma*	Dolly Varden
Scorpaeniformes	Agonidae	*Pallasina barbata*	tubesnout poacher
Scorpaeniformes	Cottidae	*Artedius fenestralis*	padded sculpin
Scorpaeniformes	Cottidae	*Artedius harringtoni*	scalyhead sculpin
Scorpaeniformes	Cottidae	*Artedius lateralis*	smoothhead sculpin
Scorpaeniformes	Cottidae	*Artedius notospilotus*	bonyhead sculpin
Scorpaeniformes	Cottidae	*Ascelichthys rhodorus*	rosylip sculpin
Scorpaeniformes	Cottidae	*Clinocottus acuticeps*	sharpnose sculpin
Scorpaeniformes	Cottidae	*Clinocottus embryum*	calico sculpin
Scorpaeniformes	Cottidae	*Clinocottus globiceps*	mosshead sculpin
Scorpaeniformes	Cottidae	*Cottus aleuticus*	coastrange sculpin

[1] Non-native species

[2] Endemic species (also shown in bold text)

Table A.3 (continued). Fishes of Olympic National Park.

Order	Family	Scientific Name	Common Name
Scorpaeniformes	Cottidae	*Cottus asper*	prickly sculpin
Scorpaeniformes	Cottidae	*Cottus confusus*	shorthead sculpin
Scorpaeniformes	Cottidae	*Cottus gulosus*	riffle sculpin
Scorpaeniformes	Cottidae	*Cottus perplexus*	reticulate sculpin
Scorpaeniformes	Cottidae	*Cottus rhotheus*	torrent sculpin
Scorpaeniformes	Cottidae	*Enophrys bison*	buffalo sculpin
Scorpaeniformes	Cottidae	*Hemilepidotus hemilepidotus*	red Irish lord
Scorpaeniformes	Cottidae	*Hemilepidotus spinosus*	brown Irish lord
Scorpaeniformes	Cottidae	*Leptocottus armatus*	staghorn sculpin
Scorpaeniformes	Cottidae	*Oligocottus maculosus*	tidepool sculpin
Scorpaeniformes	Cottidae	*Oligocottus rubellio*	rosy sculpin
Scorpaeniformes	Cottidae	*Oligocottus snyderi*	fluffy sculpin
Scorpaeniformes	Cottidae	*Scorpaenichthys marmoratus*	Cabezon
Scorpaeniformes	Hemitripteridae	*Blepsias cirrhosus*	silverspotted sculpin
Scorpaeniformes	Hexagrammidae	*Hexagrammos decagrammus*	kelp greenling
Scorpaeniformes	Hexagrammidae	*Hexagrammos lagocephalus*	rock greenling
Scorpaeniformes	Hexagrammidae	*Ophiodon elongatus*	ling cod
Scorpaeniformes	Liparidae	*Liparis callyodon*	spotted snailfish
Scorpaeniformes	Liparidae	*Liparis florae*	tidepool snailfish
Scorpaeniformes	Liparidae	*Liparis fucensis*	slipskin snailfish
Scorpaeniformes	Liparidae	*Liparis rutteri*	ringtail snailfish
Scorpaeniformes	Liparidae	*Polypera greeni*	lobefin snailfish
Siluriformes	Ictaluridae	*Ictalurus natalis*[1]	yellow bullhead

[1] Non-native species

[2] Endemic species (also shown in bold text)

Table A.4. Amphibians of Olympic National Park.

Order	Family	Scientific Name	Common Name
Anura	Bufonidae	*Bufo boreas*	Western toad
Anura	Hylidae	*Pseudacris regilla*	Pacific tree frog
Anura	Leiopelmatidae	*Ascaphus truei*	tailed frog
Anura	Ranidae	*Rana aurora*	red-legged frog
Anura	Ranidae	*Rana cascadae*	Cascades frog
Caudata	Ambystomatidae	*Ambystoma gracile*	northwestern salamander
Caudata	Ambystomatidae	*Ambystoma macrodactylum*	long-toed salamander
Caudata	Ambystomatidae	*Dicamptodon copei*	Cope's giant salamander
Caudata	Plethodontidae	*Ensatina eschscholtzii*	Ensatina
Caudata	Plethodontidae	*Plethodon vandykei*	Van Dyke's salamander
Caudata	Plethodontidae	*Plethodon vehiculum*	western red-backed salamander
Caudata	**Rhyacotritonidae**	***Rhyacotriton olympicus***[1]	**Olympic salamander**
Caudata	Salamandridae	*Taricha granulosa*	Rough-skinned newt

[1] Endemic species (also shown in bold text)

Table A.5. Reptiles of Olympic National Park.

Order	Family	Scientific Name	Common Name
Squamata	Anguidae	*Elgaria coerulea*	northern alligator lizard
Squamata	Iguanidae	*Sceloperus occidentalis*	Western fence lizard
Squamata	Charinidae	*Charina bottae*	rubber boa
Squamata	Colubridae	*Thamnophis ordinoides*	northwestern garter snake
Squamata	Colubridae	*Thamnophis sirtalis*	common garter snake

Appendix B. Threatened, endangered, candidate, and sensitive species found in Olympic National Park.

Table B.1. Threatened, endangered, candidate, and sensitive species found in Olympic National Park.

Order	Scientific Name	Common Name	State Status	Federal Status
Mammals	*Canis lupus*[1]	Gray wolf	SE[2]	FE
	Pekania pennanti	Fisher	SE	FC
	Thomomys mazama	Mazama pocket gopher	ST	FSC
	Marmota olympus	Olympic marmot	SC	–
	Corynorhinus townsendii	Western big-eared bat	SC	FSC
	Myotis keenii	Keen’s myotis	SC	–
Birds	*Fratercula cirrhata*	Tufted puffin	SC	FSC
	Strix occidentalis caurina	Northern spotted owl	SE	FT
	Brachyramphus marmoratus	Marbled murrelet	ST	FT
	Haliaeetus leucocephalus	Bald Eagle	SS	FSC
	Aquila chrysaetos	Golden eagle	SC	–
	Gavia immer	Common loon	SS	–
	Peregrine Falcon	Peregrine falcon	SS	FSC
	Chaetura vauxi	Vaux’s swift	SC	–
	Dryocopus pileatus	Pileated woodpecker	SC	–
Fish	*Novumbra hubbsi*	Olympic mudminnow	SC	–
	Prosopium coulteri	Pygmy whitefish	SC	FSC
	Oncorhynchus tshawytscha	Puget Sound chinook	SC	FT
	Oncorhynchus keta	Hood Canal summer chum	SC	FT
	Oncorhynchus mykiss	Elwha and Greywolf steelhead	–	FT
	Oncorynchus nerka	Ozette Lake sockeye	SC	FT
	Salvelinus confluentus	Bull trout	SC	FT
Amphibians	*Plethodon vandykei*	Van Dkye’s salamander	SC	FSC
	Bufo boreas	Western toad	SC	FSC

[1] Extirpated since 1920 from OLYM.

[2] SE = State endangered, ST = State threatened, SC = State candidate, SS = State sensitive, FE = Federally endangered, FT = Federally threatened, FC = Federal candidate, FSC = Federal species of concern.

Appendix C. Assessment questions and focal resources for the Olympic National Park Natural Resource Condition Assessment.

The following questions and focal resources were distilled from preliminary scoping meetings and subsequent discussions among Natural Resource Management Division staff at OLYM prior to soliciting funding proposals. The primary questions and resource topics were provided in a Funding Opportunity Announcement from the NPS to potential project applicants. Three questions were designed to elicit 'big picture' analyses of selected ecosystem functions and health that are of high management significance to the park's staff:

1. What is the current health of the Olympic ecosystem and in what ways does it differ from the way it was before European settlement?
2. What are the ecosystem effects of harvest, poaching, or of declining sensitive species?
3. What species or ecosystem functions are threatened by non-native invaders?

The following brief discussions convey the general resources of concern for which there was known data.

1. What is the current health of the Olympic ecosystem and in what ways does it differ from the way it was before European settlement?
    a. Large animal populations have been greatly impacted through removal of the gray wolf as top predator. Species formerly controlled by wolves include elk and deer which have been studied.
    b. Meso-predator release caused by wolf removal has had cascading effects through the trophic food web with impacts on other species including coyotes and mustelids. Some meso-predator studies are in progress.
    c. Elk populations have had strong effects on forest vegetation. The park has a history of vegetation mapping, plant inventory, and some forest dynamics plots. Two 0.5-ha experimental elk exclosures have been in place for decades. Our current "Landscape Dynamics" program based on Landsat imagery may also be useful.
    d. A lively literature has developed surrounding trophic effects caused by the reintroduction of wolves in Yellowstone. A survey of that literature and a thoughtful analysis of Yellowstone "before" condition could reveal conditions currently in effect at Olympic.
    e. National parks do not exist in a vacuum. The surrounding social and political milieu are potential limiting factors in how park management views its management options.

The park's neighborhood demography and economic setting contribute to our "current condition."

2. What are the ecosystem effects of harvesting, poaching, or of declining sensitive species?
    a. This question addresses issues of harvest, poaching, and past wildlife management practices.
    b. Five species of salmonids constituting 70 distinct populations occur in the park. All of the populations are considered sensitive and five are currently listed as 'Threatened' under the Endangered Species Act. Despite significant benefits provided by the current removal of hydropower dams on the Elwha River, sport fishing, tribal co-management, and global issues like climate change are continuing causes of concern.
    c. The endemic Olympic marmot seems to be in decline and has been the subject of ten years of recent study. Marmot monitoring is ongoing.
    d. Northern spotted owls are likewise in decline. Barred owls are increasing in numbers and aggressively displacing spotted owls. Study of reproductive success and area occupied by spotted owls has been underway for ten years and is ongoing.
    e. Marbled murrelets are also in decline and have been inventoried in the park.
    f. Fishers were extirpated from the park in the early 1900's. Ninety animals were released as part of a reintroduction project beginning in 2008. Monitoring is ongoing.
    g. Amphibians are declining in the park. The park has been a regularly visited "apex" site used in the USGS Amphibian Research and Monitoring Initiative (ARMI).
    h. Harvesting plants for various purposes has been a continuing issue for park management. These include harvest of salal (Gaultheria shallon), mosses, and ferns for flower arrangements and horticulture. Illegal timber harvest and mushroom picking are impacts whose true extent are largely unknown.
3. What species or ecosystem functions are threatened by non-native invaders?
    a. The park has been home to mountain goats introduced for sport hunting purposes in the 1920s before the park was designated. The animals are not native to the Olympic Peninsula. The goats have been controversial since studies conducted in the 1970s and 80s showed that they were increasing in number and having deleterious effects on mountain vegetation. The goats continue to be an active concern and a detailed history exists on their management.
    b. Barred owls are non-native invaders displacing northern spotted owls as noted in (2d) above.

c. Noxious weeds: over 200 species of non-native invaders are actively being tracked. An exotic plant management team works in the park each year. But this is an ongoing dilemma.

d. Outside impacts from airborne contaminants including persistent organic pesticides, mercury, and nitrogen deposition.

e. Wildlife diseases appear to be moving into Olympic animal populations with increasing frequency, possibly related to changing climate.

Appendix D. Participants of NRCA project scoping meeting- 24 October 2014.

Table D.1. Participants of NRCA project scoping meeting- 24 October 2014. The following members of OLYM and USGS staffs (alphabetical order) participated in a scoping meeting held at OLYM headquarters to refine priority assessment questions and resources as identified in Chapter 3.

Name	Title and Affiliation
Sam Brenkman	Chief Fisheries Biologist, OLYM, Port Angeles, WA
Catharine Copass	Vegetation Inventory Project Coordinator, National Park Service, North Coast and Cascades Network, Port Angeles, WA
Pat Crain	Chief Fisheries Biologist, OLYM, Port Angeles, WA
Tonnie Cummings	Air Resources Specialist, National Park Service, Pacific West Region, Vancouver, WA
Jeff Duda	Research Ecologist, USGS, Western Fisheries Research Center, Seattle, WA
Amy East[1]	Research Geologist, Pacific Coastal and Marine Science Center, Santa Cruz, CA
Steve Fradkin	Coastal Ecologist/Limnologist, OLYM, Port Angeles, WA
Jerry Freilich	Research Coordinator, OLYM, Port Angeles, WA
Patti Haggerty	Geographic Information Specialist, USGS, Forest and Rangeland Ecosystem Science Center, Corvallis, OR
Patti Happe	Wildlife Branch Chief, OLYM, Port Angeles, WA
Roger Hoffman	Geographic Information Specialist, OLYM, Port Angeles, WA
Kurt Jenkins	Research Wildlife Biologist, USGS, Forest and Rangeland Ecosystem Science Center, Port Angeles, WA
Louise Johnson	Natural and Cultural Resources Management Division Chief, OLYM, Port Angeles, WA
Steve Perakis[1]	Supervisory Research Ecologist, USGS, Forest and Rangeland Ecosystem Science Center, Corvallis, OR
Carrie Phillips	Forest and Coastal Research Manager, USGS, Forest and Rangeland Ecosystem ScienceCenter, Corvallis, OR
Todd Suess	Deputy Superintendent, OLYM, Port Angeles, WA
Andrea Woodward	Biologist, USGS, Forest and Rangeland Ecosystem Science Center, Seattle, WA

[1] Telephone participant

Appendix E. Chronological summary of historical observations of vegetation, river morphology, and signs of elk in the Olympic Mountains from 1860-1964.

All text is quoted from the original documents. Annotated comments added are identified in brackets [].

1. 1860. Suckley, G., and J. G. Cooper. 1860. The natural history of Washington Territory and Oregon [title abbreviated]. Balliere Brothers, New York. [available digitally in google books].
    - Elk are found in the Rocky, Cascade, and Coast ranges of mountains. They are, perhaps most abundant on the last mentioned chain throughout its whole course through Oregon and Washington Territories. In the latter they are especially abundant on the headwaters of the branches of the Chehalis river, and also upon the northern slope of the Coast Range [Olympic Mountains], back of Port Discovery and Sekwim bay. Near the last locality they are very abundant during the winter, being driven down by the snows on the mountains. They run in large droves, following well beaten trails, and at that season are an easy prey to the hunter. In January, 1857, two men in the vicinity of Sekwim Bay killed eleven fine elk in one day.
2. 1878. Morse. E. Notes on the History and Resources of Washington Territory, Unpublished manuscript, Bancroft Library, University of California, Berkeley. [a summary of the 1878 Watkinson Expedition—Skokomish-Quinault, based on diaries from Melbourne Watkinson] Pages 13-17 *in* Lien, C. 2001, Exploring the Olympic Mountains: Accounts of the Earliest Expeditions 1878-1890. The Mountaineers Books, Seattle, WA. 484 pp.
    - [page 14, Skokomish-Quinault Divide, Sept 6] They journeyed westward, over the divide, amid grand and beautiful mountain scenery, with snow all around and little prairies scattered along their route. Through these prairies the elk had made their trails, the same as a herd of cattle…]
    - [page 15, Quinault River, Sept. 9] In some places the bars are one quarter of a mile wide, with great drift piles on top of them.
    - [page 16, Quinault River, Sept. 11] This river is as large or larger than the Stillaguamish. Its banks are much more regular below the lake [Lake Quinault] than above it. The bottoms are very wide and fertile from the lake down. […] These river bottoms, below the lake, appear to be subject to overflow, only in the high freshets of winter and spring. Some of the bottoms are, probably, rarely or never subject to overflow. The river is so very crooked that it is, probably forty miles from the lake, by its channel, to salt water, while, in a straight line, it does not, in all probability exceed over one half that distance.
3. 1885. O'Neil, J. P. O'Neil's Expedition: Record of his trip back of Port Angeles in 1885. Seattle Press, July 16, 1890. Pages 255-266 in Lien, C. 2001, Exploring the Olympic Mountains: Accounts of the Earliest Expeditions 1878-1890. The Mountaineers Books, Seattle, WA. 484 pp.
    - [page 264, specific location unknown, general route from upper Cameron Basin to headwaters of Dosewallips or possibly to the North flank of Mount Anderson] While

traveling in the valley you come every now and then to what I called an elk yard, the winter home of the elk. These yards are sometimes hundreds of acres in extent. The trees are denuded of their bark, the bushes cut down and the ground as trampled as the picket ground of a cavalry troop. They seemed to be always found on the southern slope of a ridge or mountain, and so hemmed in that they are, to a great extent, protected on all sides.

4. 1889. Banta J. J. Unpublished diary of J. J. Banta. Pages 182-193 *in* Lien, C. 2001, Exploring the Olympic Mountains: Accounts of the Earliest Expeditions 1878-1890. The Mountaineers Books, Seattle, WA. 484 pp.
   - [page 185, mouth of the Queets river to 8 miles from coast] We think the bottom land will average one mile wide. It is level, and rich, does not overflow, is timbered with Cottonwood and Alder.
   - We took dinner about 8 miles [up the river…] fine land covered with small Cottonwood and timber. Large Island on East side. Covered with Alder.
5. 1889-1903. Morgenroth, C. 1991. Footprints in the Olympics: an autobiography by Chris Morgenroth, Olympic Peninsula pioneer. Edited by Katherine Morgenroth Flaherty, Ye Galleon Press. Fairfield, WA [Note: original field notes of C. Morgenroth written 1889-1903 were published in 1991]
   - [page 32, referencing his first journey looking for a homestead in the Bogachiel Valley, 1890] […] we continued on up river through several broad, park-like bottoms. At one of these, we walked into a great herd of over two hundred elk. This apparently was a favorite stomping ground as the benchland had been well browsed and there was open grassland near the river. This area seemed very beautiful to me with giant trees, bigger than I had ever imagined, growing all around.
   - [page 46, referencing circa 1889-1897, general statements about the west-side drainages] The rivers that drain the western side of the Olympics traverse the level valley floors for some thirty to forty miles after leaving the mountains on their way to the ocean. Broad gravel bars that form in the middle of the rivers are channeled with small streams. Young alders, willow, and cottonwood trees grow on these gravel bars causing great piles of logs to be windrowed during high water every little ways as one proceeds up river. This forces the river to meander from year to year and from one side of its banks to the other. For this reason the early settlers like myself who were located on the fertile river benches, built our cabins well back from the edge of the stream.
   - [page 81, referencing circa 1900] The level Hoh Valley averages a mile wide. The river, averages a quarter mile wide, meanders around broad sand and gravel bars, some a mile long. The waters of the Hoh are a milky color due to the action of the melting glaciers that grind the rock to a sediment called “rock flou.r. It is believe the name “Hoh” is a derivation of the Indian word “Ohalet” which means “white water”, and that the name was given to the river prior to 1893 by the tribe that bears the same name and resided for a time at the mouth of the river on the Pacific Ocean.

- [page 82, referencing circa 1890, Hoh Valley above the South Fork confluence] Elk trails were going in and coming out everywhere and the moss-covered ground made walking easy, especially through areas where elk had browsed.
- [page 89, referencing 1903, South Fork Hoh Valley] In 1903 with three companions I set out to explore the South Fork on the Hoh River. We followed this glacial stream to its source, which is about twelve miles above where it forks with the main stream. I found the South Fork valley not as wide as the main Hoh Valley and the slopes were covered with a heavy stand of large sized trees and low undergrowth, much of which had been killed out by over-grazing of elk. This made our travelling comparatively easy. We were confronted with many slides that extended from the top of the mountains to the valley floor. [note overgrazing reference at what was supposed to be the nadir of the elk population. Supports statements that remote areas of the park were less influenced by overhunting from 1890s to 1905 than more accessible areas]
- [page 108] On the cool north fringe of the Olympic Mountains on the banks of the Soleduck River near the river's source is a group of hot springs. […] The Indians had stumbled upon these hot springs that bubbled forth from the ground far up the valley, centuries ago while hunting the great herds of elk that frequented the river valley in the fall and winter months […]
- [page 183] Before the white man came, the Olympic Peninsula was covered with extensive forests estimated to contain 150,000,000,000 board feet timber. […] Much of this primeval forest remains in the upper valleys and slopes of the western flowing rivers. Many trees tower 300 ft into the air and measure ten to fourteen ft in diameter […] Underbrush is very dense in the western regions near the ocean and the straits and includes salal, huckleberry, salmonberry, vine maple, devil's club and others. Big-leaf western maple, cottonwood, and alder grow along the river bottoms in the western "Rain Forest" areas. [note mention of cottonwood]

6. 1890. Anon. 1890. Many Rich Mines: A trip through the Olympics reveals great wealth. Seattle Post-Intelligencer, August 14, 1890. Pages 203-209 in Lien, C. 2001, Exploring the Olympic Mountains: Accounts of the Earliest Expeditions 1878-1890. The Mountaineers Books, Seattle, WA. 484 pp. [journal of member of Pratt/Thompson Parties the summer after Press Expedition]
    - [describing North Fork of Quinault just above confluence with East Fork] The valley has fine bottom lands about five miles wide, overgrown with alder and vine maple, but not much timber. […] The bottomlands continued [writer is ascending up North Fork from confluence] wide and rich, but were all cut up by the river.
7. 1890. Barnes, C. A. Captain C. A. Barnes' narrative of the Press Expedition published in the Seattle Press, July 16 1890. Captain Barnes's Narrative. Pages 35-108 in Lien, C. 2001, Exploring the Olympic Mountains: Accounts of the Earliest Expeditions 1878-1890. The Mountaineers Books, Seattle, WA. 484 pp.
    - [page 50, Elwha Valley, near Goblin's Gate] For the next mile the mountain was a veritable elk pasture. The ground was bare of snow and covered with Oregon grape, affording

excellent grazing, and it would be difficult to find a spot large enough to place a camp kettle which did not have the impressions of hoofs. Some of the tracks were not over a couple of days old. It was evident that a large herd of elk was not far off […]

- [page 58, Elwha Valley, head of Geyser Valley] Then our course took us again along the bottom lands through tangled thickets and fern, through which in some places we had to cut our way, so dense did it become. At this place we found a large tract of old brule or burnt timber. The burn had occurred so long ago that around the fallen trees young trees had grown in a thicket almost impassable. The logs lay upon the ground in every direction.
- [page 60, Lillian] The elk trail here was three or four ft wide, cut deep into the steep slope of the mountain side. The elk had followed it for centuries […]

8. 1890. Fisher, H. Lt. O'Neil's Exploration of the Olympic Mountains. Pages 370-468 in Lien, C. 2001, Exploring the Olympic Mountains: Accounts of the Earliest Expeditions 1878-1890. The Mountaineers Books, Seattle, WA. 484 pp.
    - [Headwater basin of upper Duckabush, August 5] Elk trails were numerous here and I set out to prospect the most favorable route [...] where timber grew it was like parks, generally clear of brush, with long narrow vistas here and there. The earth was inclined to be moist, and the many tracks everywhere with no general trail was indicative of a favorite feeding ground and the natural home of the Elk.
    - [Upper Queets Basin out the Queets River] [Sept 22] After climbing out of this [gulch], I discovered fresh elk tracks and followed them, which led me to the Queets, directly below where it left the canyon [head of the floodplain, presumably upriver from Hee Hee Creek]. [...] the bottom though very narrow was level, and beautiful. Number less elk had kept the under growth down, and walls whose summits were invisible rose upon either side of the stream.
    - [Sept 23; downstream] This [river] had now grown to a beautiful stream and I was more charmed with its character as I proceeded down it. I had given up the bank and had since some time kept upon the gravel bar wading from side to side as occasion required.
    - [Sept 24th, downstream] At one point to day I passed a large log jam that had completely blocked the stream. The water had overflowed, cutting a new channel to the west, and returned to the old channel one mile below. The stream was gradually growing deeper as it gathered more water, and I was continually wet with fording, and wading at least one half of the time. Many times I attempted to take short cuts across the bends but grew sick of the brush, and was glad to reach the water again.
9. 1890. Gilman, C. A. 1890. Farms in Olympics. Seattle Post-Intelligencer, May 28, 1890. Pages 170-171 in Lien, C. 2001, Exploring the Olympic Mountains: Accounts of the Earliest Expeditions 1878-1890. The Mountaineers Books, Seattle, WA. 484 pp.
    - [page 170] Four large rivers, the Quinault, Queets, Hoh, and Quillyhute run from the easterly part of the mountains westerly to the ocean. […] Upon various sections of these rivers both in the mountains and west of them, are very fine bottom lands from one to four miles in width,

timbered but lightly with very small cottonwood or brush, and very cheaply cleared for farms [note: the path of the 1889 expedition from Grays Harbor to Strait of Juan de Fuca explored primarily the Quinault, then stayed near the coast for crossing the Queets and Hoh.]

10. 1890. Gilman, C. A. 1890. Unknown no longer. Seattle Post-Intelligencer, June 5, 1890. Pages 171-175 in Lien, C. 2001, Exploring the Olympic Mountains: Accounts of the Earliest Expeditions 1878-1890. The Mountaineers Books, Seattle, WA. 484 pp.
    - [page 172, describing Quinault Valley from Reservation to Lake Quinault] the country is described as very fertile, with numerous fine bottom lands, broken by an occasional wooded ridge. The bottoms were covered with cotton and alder, with dense underbrush of vine maple and salmon berry.
11. 1890. Meany, E. S. The Olympics: An account of the explorations made by the Press Explorers. Seattle Press, July 16, 1890. Pages 22-34 in Lien, C. 2001, Exploring the Olympic Mountains: Accounts of the Earliest Expeditions 1878-1890. The Mountaineers Books, Seattle, WA. 484 pp.
    - [note: throughout the journals there are references to both very dense vegetation (jungle like growth) that made travel difficult, and open, more parklike vegetation that was easier to travel through. Here and throughout I limit quotes to those that specifically mention effects of elk, river channel characteristics, or detailed descriptions of vegetation.]
    - This region was known to abound in large game, since the valleys putting up in to the unknown region teemed with herds of elk, with cougars, bear and deer. Hunters have shipped in loads of elk horns and hides from the Quilayute region, and bear and cougar skins are common in the huts at the mouth of that river [Quilayute].
    - [page 31] Three or four miles before coming to Alexander river [Rustler Creek] the expedition crossed the river on a tree—the river has now become too large to ford—and found a good elk trail and splendid travel to Alexander River. From Alexander River to Lake Quinault take the west bank if the traveler has any means of crossing. […] If he cannot cross above he will find an opportunity of doing so at the double island, shown on the map, five miles below Alexander River[...][other mentions of islands].
    - [page 32][on the Quinault Valley] the upper valley above the lake, is 16 miles long, and from one to two miles in width. […] The soil is, in general, an alluvial loam, rich with decaying vegetable matter […] The vegetation is very prolific. Besides the dense growth of large conifers, there is a great scattering growth of maple and alder of large size. A dense underbrush covers all the ground, consisting of salmon berry, raspberry and other similar bushes. The land is rather difficult to clear. The river is navigable by canoes as far as Alexander River [Rustler Creek]. There are no formidable rapids. The only dangers to navigation are the drift piles, but these are easily removed.
    - [page 99] This stream, which we named Alexander River [Rustler Creek] is large but very shoal and rapid […] We crossed it without difficulty and found ourselves in a large open bottom, enabling us to see up and down the valley

- [page 100, near present day North Fork Quinault Ranger Station] One hundred yards below camp we forded a slough and traveled for a short distance on a bar. Recrossing to the mainland we came to a large island, over a mile long and about a mile wide. We crossed a deep stream separating it from the mainland, and traveled the length of the island through a dense jungle of underbrush.
- [page 100, near confluence with East Fork Quinault] We continued all day down the river, through dense underbrush consisting of a tangled thicket of salmon berry bushes, vine maple, and all other usual small growth that can be imagined […and farther downriver, below the forks…] The bank of the river becoming now so dense with underbrush as to be almost impenetrable […]
- [page 104, near bunch Creek above Lake Quinault, Press party split on opposite sides of river after rafting accident involving a logjam]. The boys had the best of the road, their side being for a large part along sandy bars. Our side was covered with thick woods, dense with underbrush, and cut with sloughs. The salmon berry bushes and bramble held us so that we did not make more than a mile in three hours of hard work.

12. 1897. Merriam, C. H. 1897. Cervus roosevelti, a new elk from the Olympics. Proceedings of the Biological Society of Washington. XI: 271-275.
    - At the time of my visit to the Olympics the later part of August the elk had been recently driven out of the upper Hoh and Soleduc canyons by Indians, and the numerous tracks seen were 10 days or 2 weeks old. Well-beaten trails followed the crests of the higher ridges and traversed the principal valleys. Many of these trails, with little labor, can be made available for horses and afford almost the only means of penetrating the region.
13. 1917. Murie, O.J., 1917 Olympic mountains field notes. Field notes on file at Olympic National Park, Port Angeles, WA
    - [Murie, hired to trap wolves during the winter of 1916-17 in the Elwha Valley, left handwritten notes indicating wolves were scarce. He trapped no wolves, but saw a few tracks. Murie relates the following observation from Billy Everett from days past] […] wolves were [then] numerous. He once saw a band of about 30. He has shot and dropped quite a few […] He says they used to have regular trails. Often a large band would howl, making quite a din in the night.
14. 1918. Bailey, V. Report on Investigation of elk herds in the Olympic Mountains, Washington. Unpublished report on file Olympic National Park. Port Angeles, WA
    - [Hoh Valley] For a distance of about 15 miles along the river bottoms they [elk] had largely cleaned up the salmonberry, willows, blueberry, elder, and devil's club, but vine maple remains in abundance and has been well browsed. By going back into the side gulches or lower down the river the elk would find an abundance of their best winter food.
    - the loss of last spring could undoubtedly have been largely prevented by gently scattering the elk or urging them down the valley past the three little ranches that seem to be the only

obstruction to their old-time extension down the valley. [suggesting that the concentration in the park is unnatural]

15. 1918. Riley, S. 1918. Memorandum to District Forester. Unpublished memo on file at Olympic National Park, Port Angeles, WA
    - [Hoh Valley] The river, a free flowing stream with high flood periods, when the flow carries an immense amount of cutting material, is continuously changing its channel. Therefore, those lands formed by the action of the river may be cut away at any time. […] A good big percent of the valley floor is occupied by the river and there is every reason to believe this area will increase as the timber is cut from the watershed […]
    - The heaviest loss [of elk] where the recent increase has taken place is on the Hoh River above the settlements within the National Forest, upon the Clearwater, the Queets, and Quinault, also within the National Forest. The food within these limited areas, consisting of salmonberry, willows, alder, vine maple, black stemmed and licorice ferns, not extensive at best, has become extremely scarce under excessive use in recent years. These areas where the elk have congregated and starved of late are clearly defined or traceable by the absence of these plants upon which they feed when driven down from the higher lands by snow. The chief ones of these are salmonberry, elder, and black stemmed fern.
    - One very interesting feature of the Olympic elk is the extreme limitation of their movements. […] This tendency of localizing their movements and limitation of available feed within given areas would appear to be the best reasons for provision to utilize a given number of the animals in those localities […] The solution of the difficulty, therefore, would appear in causing the animals to feed over a greater area.
    - Wolves and cougar, which are continuously trapped by the settlers, appear to be the elk’s only enemies. The loss from these two predacious animals is not extensive, as the wolves are extremely scarce and the cougar, which are not plentiful by any means, seek the deer which are plentiful rather than the elk.
    - Elk have always been described as a grass-eating animal, particularly has this been emphasized for the animals of the Rocky Mountain region. One peculiar feature of the situation in the upper Hoh Valley is that, while there is a grass there eagerly sought by horses or cattle, the elk do not eat it, according to the observation of the forest officers and some of the rangers living in the region.
16. 1922. Webster, E. B. 1922. State and forest officials make study of conditions of elk herds of the Hoh. Unpublished, Olympic National Park. 3 pp.
    - [there is a] pronounced shortage of late winter or early spring feed for the elk of the upper Hoh valley, resulting, this past spring, in the loss of starvation of approximately forty animals [elk], or a possible two per cent [of the herd].
    - There was a great abundance of feed in the lower Hoh region, extending quite up to Spruce […] and in all this great region there are almost no elk trails nor signs of elk. [Also describes a vast area of blowdown which was] impenetrable by either elk or deer.

- Soon after leaving Spruce [heading up-valley], on the river trail, one notes a scarcity of browse. There is no salal, no salmon berry, no thimble berry—practically nothing save vine maple and huckleberry. The former has been trimmed up as far as elk can reach; the later cut down until there is little but stubby bushes remaining […] Further upriver the condition becomes more acute, the timber having an open, park-like appearance, having been thoroughly cleaned to the height of about seven ft.
- Why the elk, on these occasional years when the snow lies deep and late, should not continue on down into the country of abundant food, proved a puzzle to the writer [Webster]…The answer was forthcoming at Spruce, when attention was directed to the fact that everyone on the river kept from 'two to thirty' dogs. The elk will simply not run the gauntlet of the hounds, preferring to remain above Spruce and starve to death instead.

17. 1922. Webster, E. B. 1922. Swinging 'Round the Circle'. Western Out-of-Doors. Report on file at Olympic National Park, Port Angeles, WA
    - But on the Hoh, conditions are entirely different. There is a great abundance of feed in the lower Hoh region, extending quite up to Spruce, as John Huelsdonk's ranch is known, and in all this great region there are almost no elk trails nor signs of elk. On either side of the river valley proper, often over a mile in width, there extends a great strip of benchland, practically level and only two or three hundred ft above the level of the stream. Formerly the elk traveled up and down this benchland, now a mass of wreckage from the big storm, miles on miles of piled up trees, impenetrable by either deer or elk [reference to impacts of the 1921 'Big Blow'].
18. 1933. Anonymous. National Park Service memo on file at Olympic National Park, Port Angeles, WA
    - According to the best information we are able to get, and from our own observation, the elk herds seem to have risen in number some time following the creation of the Monument to a high point from which the numbers have gradually diminished. There has been over use of certain small parts of the range, particularly on the Hoh River, a distance of ten miles or more from the Olympic Ranger station down somewhat below the Jackson Ranger Station. This over use seems to have reached the peak several years ago at which time practically all of the browse used by the elk on the floor of the valley was destroyed. Carrying capacity of this range has been very materially reduced, and it would not now support more than 45 per cent of the elk that could normally winter there. It appears not to be so heavily used now, but the comeback will be slow even if the range has better protection in the future due to the fact that much of the browse has been literally destroyed. Starvation of a few animals seems to have occurred in this area about 1930. Since that time there has doubtless been a considerable reduction in the elk herd, which has used the Hoh Valley during recent years.
19. 1933. Skinner, M. P. 1933a. Report on elk conditions in Olympic Mountains to Boone and Crocket Club, NY. Unpublished report on file at Olympic National Park, Port Angeles, WA
    - The total of 6,000 elk seems to have remained stationary for some time.

- [on primitive conditions] Undoubtedly there were more elk in primitive times. Suckley (1958-60) says they were especially abundant on the peninsula in 1855. Probably there were as many as 25,000; possibly as many as 40,000. Their range was greater and their forage had not then been destroyed over two thirds of the best areas as it has now. For 6,000 elk, the present range seems sufficient, and the present amount of browse ample for all but the severest years.
- John Huelsdonk […] began making reports fifteen years ago, or more, that there were too many elk, and that they were starving to death. All other authorities, although some of them found an over-browsed appearance just as I did, ridiculed the stories of starvation. Siegried, ranching on the Bogachiel, next to the Hoh, said the outcry was "all bunk" and made so to attract hunters. […] A rancher farther down the Hoh Valley told that, if there was any shortage of food above, it was because the ranchers there had been grazing cattle. At one point where I found elk food scarce, the whole areas looked as if it had ben grazed by domestic stock in the past and had not yet recovered.

20. 1933. Skinner, M. P. 1933b. Report on Roosevelt elk, Olympic Peninsula, Washington. Unpublished report on file at Olympic National Park, Port Angeles, WA
    - [page 8] When I reached the Olympics I very soon found that the question of elk food was of the first importance because reports of elk starvation influenced the opening of the hunting seasons […] Evidently, the reports of food shortage had come largely from the Hoh Valley […] All localities other than the Hoh, reported plenty of forage for the elk. I found Mr. John Huelsdonk, of Spruce, Hoh Valley, all worked up about the subject […] To hear his [description], one would think all the elk above his ranch were dying of starvation […] While I found signs of over-grazing in the Hoh, I was inclined to blame the presence of domestic cattle, sheep, and goats. Furthermore, I found that there were numbers of maples growing there, and that the elk were so fond of their leaves that they gathered from far and near to feast on them.
    - [page 17] Huelsdonk also "spoke to the abundance of wolves, cats and cougar in former years compared with the few that exist now." […] The millions of salmon that used to feed the wolves, cats and bears are gone, destroyed by the canners. Hence it may be that some predators are forced to kill mammals now that they would not molest if they had salmon.
21. 1934. Cliff, E. P. 1934. Memorandum on the Olympic Elk. Unpublished report on file at Olympic National Park, Port Angeles, WA
    - [This report was not available for examination. The following quotes are attributed to E.P. Cliff (1934) by C. C. Newman (1953)]. The forage in the canyon bottoms and the lower benches and slopes in the Hoh River valley from the Lewis Ranch up the main canyon to the mouth of Glacier Creek and up the South Fork for at least nine miles is very seriously overgrazed. […] The shrubs show the cumulative effect of overgrazing year after year. The vine maple has been browsed so heavily that the foliage within reach of the elk has been thinned out materially and in many instances a distinct game line can be seen […] the huckleberry brush which is perhaps the dominant undershrub on the timber covered slopes

and the benches is in an extremely impoverished condition […] The salmonberry thickets which are so much in evidence on the lower reaches of the river have been almost completely eliminated in the upper Hoh bottoms. Only occasional plants remain and these are subject to severe grazing. The only place where salmonberry has escaped is where it is growing in inaccessible places such as in log jams or on stumps and logs […] the same thing is true of the highly palatable elderberry […] The willows on the river bars have been so heavily browsed that many of them have been killed off and most of the remaining plants support only a hedgelike tuft of foliage near the base of the stalks […] The bigleaf maple, Acer macrophyllum, and cotton wood are browsed readily when within reach of the elk. Mr. Murie and I watched carefully for reproduction of these trees and found none in the heavily browsed areas.

22. 1934. James R. F. 1934. Unpublished notes on file at Olympic National Park, Port Angeles, WA (Early Wildlife Reports)
    - Most of the territory is heavily timbered and has a dense growth of underbrush which forms an almost impenetrable jungle. Scattered through this area are sections miles in length commonly known as blowdowns, caused by violent wind storms [probably seeing the 1921 "big blow"]. It is nearly impossible for man to penetrate these areas since the size of the timber is such that it is an all day job to make two or three miles.
    - This country at the present time is well populated with predatory animals namely the cougar and wild cat [bobcat].
    - On certain watersheds the entire browse has been cleaned out in the lower valley and yet areas within a mile of this overgrazed location have never been grazed at all. [Note: At this time of peak population numbers, there are reports of both "impenetrable jungles" associated with blowdown, and areas where "entire browse has been cleared out"]
23. 1934. Murie, O. J. 1934. Preliminary report on the elk of the Olympic Peninsula. Unpublished report on file at Olympic National Park, Port Angeles, WA
    - As a result of winter conditions and limited winter range areas within the mountains, some of the valley bottoms are greatly over-browsed. This is particularly true of the Hoh, Queets and Quinault watersheds. On the Hoh River, early settlers describe heavy thickets of salmon berry all the way up past the boundary of the National Monument as being the condition forty years ago [1894]. Today, salmon berry is scarce above the forks of the Hoh. The same condition is duplicated to some extent on the Queets and Quinault.
    - Vine maple shows heavy browsing on the upper reaches of these rivers, and the same is true of huckleberry. These are often browsed so heavily that the limbs are stubby and broom-like and forage production is, as a result, very low. The limbs are browsed as high as the animals can reach. Unfortunately, after the vine maple and huckleberry have withstood winter browsing, the elk and deer continue to eat the leaves in the spring and early summer, which gives these plants little chance to recover.
24. 1935. Murie A. Wildlife of the Olympics: Special Report to the National Park Service, Unpublished report on file at Olympic National Park, Port Angeles, WA

- [page 4] The elk are most plentiful along the higher reaches of the rivers, although some do occur even down as far as the ocean. …From what we know of elk in other areas, the present distribution, with the main herds in the higher country, is probably a typical primitive condition. […] When elk were scarce in the Hoh Valley some thirty years ago, settlers living at Spruce [just outside the current park boundary] hunted their meat twenty miles up the river near the Olympus Station, the elk being more plentiful higher up the valley. Possibly poaching had decimated their number farther down the valley. […] The Hoh valley has a surplus of elk, and the vegetation is much overbrowsed. In the Bogachiel Valley, separated from the Hoh only by a low ridge, elk are less abundant, and the vegetation is in better condition.
- [Hoh River, page 8] In winter the elk are somewhat evenly distributed along the valley floor from a point about eight miles below the Jackson Station to a point two miles above Olympus Station, where the valley begins to narrow.
- [Queets River, page 8] I found heavy browse utilization as far up as Harlow Creek and it was reported that the browsing was heavier beyond Harlow Creek.
- [Quinault Valley, page 9] On the east fork of the Quinault, Ingnar Olson states that most of the elk winter above Graves Creek, and on the North Fork the elk are most plentiful on the Rustler River: […] "During hard winters the elk come down as far as Lake Quinault in large numbers […] many of the cows have calves below Graves Creek and later move up the valley."
- [page 11] Broadleaf maple is highly relished […] Tips of twigs often are chewed down to where the stem is over ten millimeters in diameter.
- [page 11] Willow occurs but sparingly in all the regions visited. On the bars near Olympus Station some low-growing willows had been heavily browsed.
- [page 16] Cattle on the elk range are in rather direct competition with elk for food….at the present time eight head of cattle are ranging for several miles up the Queets Valley above the Kelly Ranch. There are very few elsewhere.
- It is said by some that there were more elk ten years ago than there are today. Thirty years ago they were scarcer than they are today
- [page 17] In many places the browse on the winter range is heavily utilized and might be termed overbrowsed. On the Elwha there are many such places; red huckleberry, in particular, was adversely affected, some bushes having died apparently from overbrowsing […] The vegetation in the Hoh Valley is heavily utilized from a point about one mile above Olympus Station to a point seven miles below Jackson [Hoh] Station, a stretch of approximately seventeen miles.
- [page 18]Along the Queets River, between the Kelly Ranch and Harlow Creek, I found conditions similar to that along the Hoh River. Overbrowsing between Harlow Creek and Pelton Creek is reported to be even worse. […] On the Quinault River overbrowsing is most severe on the East Fork […] the range on the North fork is reported to be in fair condition

[…] for the upper Bogachiel Valley […] the browsing is light. Except for the Elwha, Quinault, Queets, and Hoh watersheds the browsing has not been excessive.

- [page 18]I believe that overbrowsing can take place here without any serious damage to the vegetation or soil, to such an extent that the elk will be automatically controlled by food supply […] My feeling now is that such a process is normal, a primitive course of events, and therefore to be permitted. The Olympus elk picture differs from that of the Yellowstone in being a more normal one.

25. 1935. Murie, O. 1935. Report on the elk of the Olympic Peninsula. Unpublished report on file at Olympic National Park, Port Angeles, WA [report on march 20th-May 19th field trip to Olympics]

    - [page 5] In earlier times elk were not nearly as plentiful. Several settlers on the Hoh River, for example, state that when the first ones came into that part of the country, some 40 years ago, elk were scarce. At that time one could hunt all day, and then possibly fail to get an elk. I was told that even on the Bogachiel, where elk are not nearly as plentiful as on the Hoh, they have increased in numbers in the last 14 years. […] It is entirely possible that in still earlier times the elk were numerous. The possibilities of game cycles or other fluctuations in numbers will not be discussed here.
    - [On Salmonberry, page 6] is very palatable and is probably the first to disappear when elk become overabundant. It may possibly by the “key species” in judging range conditions […] Salmonberry is brittle and the thickets are apparently readily trampled and broken up when numerous elk invade them. […] Originally salmonberry grew abundantly up the Hoh Valley at least as far as Olympus Guard Station, and although they have been grazed out by elk, there are still traces in the upper valley
    - [On bigleaf maple, page 6] where elk are plentiful, I saw no reproduction of the maple.
    - [on willow, page 7] Mention should be made of the willow. It occurs sparingly on the river bars and some other favorable locations, but is eaten down persistently by the elk, and probably also the deer. Apparently willows do not have much chance to spread on these elk ranges.
    - [on Elder, page 7] it is eaten down to a big diameter […] Consequently, it is rather easily destroyed, but persists as individual bushes on high stumps or logs, out of reach of the elk.
    - [On the Hoh River, page 12] The Hoh River is the best for a study of an overbrowsed range. Above the Lewis Range salmonberry has disappeared. John and Henry Heulsdonk and other early settlers declare that in early days, when elk were scarce, salmonberry thickets flourished all the way up at least as far as Olympus Guard Station. Today one has to look sharply to find the few traces of salmonberry still persisting in the river valley above a point short distances above the Lewis Ranch. […] Red huckleberry is badly overbrowsed. Much of it is broom-like, much eaten down to a club-like form, and some is dead […] Vine maple, the staple food is also very heavily browsed […] Even the ferns, particularly elk fern, Blechnum spicant, appear to be relatively scarce.

- [on the Queets] The Queets is overstocked on the upper portion […] about as far down as Tshletshy Creek. […] However, the Queets is not overbrowsed to the extent that the Hoh is.
- [On cougars, page 18] The cougar is the most effective predator on the Peninsula and the one which has aroused the greatest interest in the elk situation. […] It is interesting to contemplate that just when the campaign to exterminate the cougar from the Olympic Peninsula has become most severe, the elk have become so numerous on the Hoh River that they are dying as a result of overstocking and the deer have become so numerous on the Elwha that very few elk can be carried on that range. With a surplus of game, the only remedy seriously considered is the complete extermination of the cougar. […] There can be but one conclusion. We are not justified in exterminating the cougar on the Olympic Peninsula.

26. 1936. Murie, A., Mammals of the Olympic Wilderness. Unpublished report on file at Olympic National Park, Port Angeles, WA. [note much overlap between this report and the previous report]
    - In several watersheds, namely the Hoh, Quinault, Queets, and Elwha, the range might be termed overbrowsed. […] the Hoh Valley is perhaps the best example of overbrowsing. Here salmonberry and elderberry, reported to have been abundant in the early days, have practically disappeared over a seventeen mile stretch where the elk winter. Only a few of these shrubs were seen on high stumps and logs out of reach of the elk and deer. […] Although the vine maple has been browsed to the extent that the tips of the branches within reach have become broom-like, the plants themselves continue to flourish. As most of the tree is out of reach its vitality is little affected. […] the main damage to the range seemed to be the elimination of salmonberry and elderberry. Erosion was not noticeably increased. […] Although there is more and a greater variety of food farther down the valleys adjacent to the wintering areas, the elk fail to take advantage of it. Force of habit seems to keep them in a restricted section of the range.
    - It is quite possible that overbrowsing is part of the natural course of events in this region. There may be a normal cycle of scarcity and abundance of elk with corresponding changes in forage conditions. With a decreased food supply, a reduction of the elk might result either directly from food shortage or indirectly through higher incidence of disease and parasites or a lowering of reproductive rate.
    - In the Olympics, except for the disappearance of the wolf, the natural predators of the deer and elk, so far as we know them, are still part of the fauna. Bears are present in moderate numbers […] The most effective predator on the deer and elk is the cougar. […] Just how many elk a cougar will kill in a year is not known.
27. 1938. Murie, A. Personal letter from A. Murie to L. Sumner. On file at Olympic National park, Port Angeles, WA
    - In the Olympics before any artificial control is exercised I believe we should give the cougar time to come back. I hope the present area in the monument is large enough to give the cougar the necessary protection. I doubt that conditions in the Hoh will get much worse than

at present, or what do you think? If conditions do not get worse, wouldn't it be better to put up with some overbrowsing than messing around controlling the animals. I, myself, am not sure what I think about this. The cougar is our hope and may it come back. […] Enlarging the monument in my opinion might make it easier to save the cougar to kill the elk.

28. 1938. Sumner, L. 1938. Special report on elk in Olympic National Park—summer 1938. Unpublished report on file at Olympic National Park, Port Angeles, WA

    - [Bogachiel, page 2] the browsing can be described as quite heavy from just above Hyak Guard Station to below the Bogachiel Guard Station. […] Some of the damage around the Hyak Guard station may have been caused by domestic stock but it was evident that elk were largely or wholly responsible elsewhere.
    - [Hoh Drainage, page 3] Dr. Murie well describes the heavy over-browsing in the Hoh Valley along the seventeen mile stretch between Jackson Station and the Olympic Station. Elderberry and salmonberry were reported by him as largely gone from the area at that time, and these conditions still prevail. The vine maple has been browsed back as high as the animals can reach and reproduction of young growth must be hampered or prevented.
    - [Queets Drainage, page 3] Elk browse has been rather heavy in places above the mouth of Tshletshy […] and from there, at least, up to the mouth of Hee Hee Creek […] Elk damage in this area, while not as severe, apparently, as in the Hoh is about as bad as in the Bogachiel Valley.
    - [Quinault Drainage, page 4] There has been a rather strong recovery of vine maple and salmon berry since last year. Although some of this recovery may be due to migration of the elk to higher elevations for the summer, some of it also may be due to the fact that large numbers of elk were killed here during last year's open season.
    - [page 5] Probably the wolf was one of the most important controlling factors in former times but since it is now nearly or quite extinct on the peninsula an essential natural check to overpopulation has been removed. Unless some substitute for this now absent controlling factor is provided, serious destruction of certain plants and even their total elimination in certain places through inability to reproduce will no doubt occur.
    - [page 5] Cougars now constitute for the elk population the only important natural predator. Just how important a part is played by cougars in controlling elk is unknown. […] Intense local opposition to the cougar has existed in the past but Mr. Macy believes that he has brought about a discernable change of sentiment […] as a result of his talks. Continued activity in this direction might help the elk more than anything else that we can do at present for it is probable that if persecution of the cougar were eliminated these animals would be of great assistance in solving an elk over-population which threatens to be a serious problem for the Service.
    - [page 6] From the evidence at hand, the starvation factor has operated to control the elk more extensively than any other (except possibly hunting) population control […] If the starvation condition which threatens the Olympic elk represented a strictly natural condition, we might be justified in letting nature take its course but since there is little doubt that the elimination

of the wolf and the persecution of the cougar have contributed to this condition we are scarcely justified in a thorough-going ‘hands off’ policy. Man must take the responsibility for eliminating these predators, therefore, he cannot avoid the responsibility for the elk over population.

29. 1939. Schwartz, J. 1939. The Olympic elk study. U.S. Forest Service, Olympia, WA. Olympic National Park Unpublished Report, Port Angeles, Washington.
    - [page 20] It is interesting to note that the largest elk herds on the peninsula occur on the drainage supporting the most extensive stands of vine maple.
    - [page 31] Huckleberry […] has been severely overbrowsed on portions of the more heavily populated drainages of the west side of the peninsula […] it is usually broomlike and weakened in vigor and often contains considerable dead wood. In fact, on certain localized areas […] it is not uncommon to find a large proportion of bushes dead or in a dying condition.
    - [page 34] Salmonberry is one of the first important browse species to show signs of range saturation and overutilization […] It will practically disappear from a range that is persistently over utilized.
    - [page 61] The Bogachiel is a good example of a west-side drainage that is understocked with elk. In striking contrast to the overgrazed Hoh […] such palatable plants as salmonberry, devil’s club, elk fern, and huckleberry grow in great abundance. […] because it is a shorter and narrower valley and does not contain the extensive vine maple bottoms, its potential carrying capacity can never be as great.
    - [Hoh, page 65] Salmonberry, elderberry, and devil’s club, all highly palatable, had been almost eliminated above the forks of the river, except in isolated places where they were growing on upturned tree roots, stumps, or high banks out of reach of elk […] vine maple had been so persistently eaten back that most branches within browsing reach had been greatly weakened in vitality and contained but a small fraction of normal twig growth. A distinct elk line was evident on the vine maple and trees throughout the area. [Schwartz noted that heavy die off during winter 1937 and hunting seasons of 1936-37 created a marked improvement in the forage recovery].
    - [Seasonal Utilization on Hoh Drainages, page 66] [Schwartz noted that winter twig use on vine maple decreased from 90%-40% from 1936-1940, while winter use of huckleberry decreased from 80% to 50%]
    - [Queets, page 69] a portion of the Queets has also been over utilized. This area, commencing on the lower portions of Tshletshy Creek, extends up the main valley to approximately Kilkelly Creek […] the general condition and utilization of huckleberry and vine maple are similar to that on the Hoh.
    - [Quinault, page 72] Overbrowsing is much in evidence on the Quinault but has not progressed as far or as uniformly here as on the previously discussed drainages. [Schwartz notes that overgrazed areas ranged from four miles below Graves Creek to Fire Creek on the

East Fork and from Big Creek to Rustler Creek on the North Fork]. [Schwartz also notes that percent twig use of vine maple ranged from 60-80% in 1936 and 50-75% in 1938. Percent utilization of huckleberry ranged from 30-80% from 1936-1938, and percent twig utilization of salmonberry ranged from 50-80% all years.]

- [page 130] cougars have been hunted intensively during recent years and consequently their number has been reduced, but there is evidently still a fair number throughout the peninsula. The role these large cats play in elk and deer mortality is imperfectly known, however detailed analysis of 28 cougar scats […] indicated that 8.8% contained elk."
- [page 140] [Schwartz notes that 1187 elk killed in three hunting seasons from 1936-38; 268 in 1936, 811 in 1937 and 108 in 1938. The majority (507) were killed in the Hoh Valley. Approximately 240 elk were killed in the Quinault and 200 in the Queets.]
- [page 141] on portions of the north fork of the Hoh and on the Quinault, sufficient reductions were secured […] to affect a marked improvement in forage conditions by the summer of 1938. The concentration areas on the south fork of the Hoh and the Queets River were but slightly affected [because of the remoteness of these concentration areas].

30. 1939. Sumner, L. 1939. Special report on wildlife conditions in the Olympic National Park with special reference to elk-winter of 1938-39. Unpublished report on file at Olympic National Park, Port Angeles, WA.
    - [page 3] In the vicinity of the Happy Four Shelter [about five miles up the Hoh Valley from the current Hoh Visitor Center] a large band of elk apparently has remained for some time. Rather severe browsing of vine maple is visible along the trail and for a considerable distance off the trail. […] Salmonberry and huckleberry are similarly eaten back where they occur […] salal is virtually absent. Alder, evidently, is not a preferred food species for elk […]
    - [page 4] Although extensive destruction of browse occurred in the Hoh Valley where the elk had congregated in large numbers […] in other parts of the valley where the animals were not so abundant the browse was only moderately eaten and in some cases was practically untouched. The wide maintained trail used by man affords the best route of travel for elk […] with the result that the browse along the trail is consumed more than in other places. […] In the vicinity of the Olympus shelter […] the vine maple was surprisingly little browsed, especially away from the trail amid tangles through which the elk evidently found it slightly more difficult to travel. […] it was obvious that in general the shortage of browse was not acute. […] In the area described there seemed to be scores of acres of this almost untouched food supply
    - [page 5] Although the vine maple is in no danger at present this may not be true of the big-leaf maple [The writer looked carefully at all times for young big-leaf maple trees. None was found.]
    - Mr. Voorhies, an observant rancher in the Quinault Valley, stated that he has seen a few young big-leaf maples in the Quinault but not many. He said that sometimes these trees spring up on steep slopes above the river valleys where such slopes had been recently denuded of conifers by snow slides.

31. 1940. Henson, J. 1940. Great elk herds once kept Lake Ozette area free of underbrush. Port Angeles Evening News. February 15, 1940.
    - Gathered on the lake shore with us was Charles Keller who has lived at and near the lake ever since he was a kid and has cruised over almost every foot of the entire area. […] Keller said that from the best information he can gather, at one time there were many hundreds of elk around the lake [Ozette]. There were so many that instead of the dense underbrush that is there now the vegetation had been browsed down like it is on the upper Hoh River. About fifty years ago [1890] hide and teeth hunters entered the district and killed the animals off […]
    - After the first big slaughter of half a century ago, the area that had been over-browsed started to grow up to underbrush again, it was contended, and now all the lake shore and the adjacent country is a complete thicket except where some rancher has hacked a clearing for his home.
32. 1943. Dixon, J. S. 1943. Report on status of wildlife along western boundary of Olympic national park. Unpublished report on file at Olympic National Park, Port Angeles, WA
    - [page 12] On both the Queets and the Quinault the overbrowsing is due to the combined browsing of both domestic stock and native elk so that it is difficult to say with certainty just how much is due to cattle and to elk. It is obvious at Harlow Creek and Spruce Bottom on the Queets that much of the browsing is done by domestic cattle said to belong to John Andrews. I counted 33 head of cattle on the Queets inside the park between the boundary and Harlow Creek. Fresh tracks and droppings indicated that about 100 cattle now range inside the park along the Queets and I estimate nearly as many range in the park along the Quinault. […] Trespass by domestic cattle was noted inside the park boundary near the Jackson [Hoh] Ranger Station. All along the Queets from the park boundary up to and above Harlow Creek trespassing cattle were seen in numbers. At Graves Creek Ranger Station on the east fork of the Quinault trespassing cattle were numerous and to a considerable degree on the north fork of the Quinault also. I consider this unauthorized trespass by domestic stock within the park as the greatest present menace to native wildlife in Olympic National Park.
33. 1951. Sumner, L. The Olympic Wildlife Problem: Field Survey and Recommendations. Unpublished report on file at Olympic National Park, Port Angeles, WA
    - [page 22] The elk range has improved since 1936, when many animals starved, particularly on the Hoh, but the reasons for this improvement [are poorly understood]. It might be caused by the greater annual kill by hunters, which reached 2,000 last year […]
34. 1952. Sumner, L. 1952, The Olympic Wildlife Problem: Second Season Field Survey. Unpublished report on file at Olympic National Park, Port Angeles, WA
    - [page 6] The elk range and the animals continued to be in good condition, with no repetition of the overuse and starvation that characterized the period around 1934. The writer feels, on the basis of intermittent studies made since 1935, that at present these animals are in little or no danger of exceeding the carrying capacity of the range.

35. 1953. Newman, C. C. 1953. A preliminary report on elk, cougar, bear, coyote, and the browse conditions in the Olympic National Park. Unpublished report on file at Olympic National Park, Port Angeles, WA
    - [Elwha] With the present relatively low population, combined with the facts that during severe winters the elk can, and do, move out of the park into better browse and also where they are subjected to hunting, there is no immediate damage being done to park vegetation. Heavy populations in the past have left their mark, and the effect can still be seen. Huckleberry, vine maple, and salmonberry are greatly reduced when compared with the present supply of Oregon grapes and salal which is low in palatability.
    - [Hoh]The description that Cliff gave for the Hoh in 1934 fits pretty well the conditions existing today. […] Evidently huckleberry (V. ovalifolium) has become further reduced as seldom can it be found on the slopes and benches where it once was common.
    - [Queets] The upper Queets valley is the most severely over-browsed valley in the park. Some of the other valleys can be termed ‘heavily browsed’, but with the upper Queets the term ‘overbrowsed’ must be used. Going upstream from about Harlow bottom the overbrowsing becomes progressively worse until in the vicinity of Pelton Creek, all woody shrubs except alder are gone or reduced to a remnant. […] there probably should be no more than a maximum of 300 elk for the Queets until the range has had a chance to recover.
    - [Quinault] The free ranging of domestic livestock has been a problem in the past and will very likely continue as one. [...] Up the river, above the range of the good blooded Angus cattle, the elk forage plants are in only fair condition. There is so much interference from man’s activities lower down the valley that a true picture cannot be obtained.
36. 1954. Newman, C. C., 1954. Special Report on the Roosevelt elk of Olympic National Park. Unpublished report on file at Olympic National Park, Port Angeles, WA
    - [disclaimer that 1953 report was preliminary; some conclusions have been changed after additional field work]
    - I have noted very little use of the alder as a food at any time of the year.
    - Much evidence is available that indicates heavy use of vine maple in years past […] Reproduction is practically nil.
    - [bigleaf maple] Reproduction is very limited
    - [cottonwood] very few cottonwoods escape being trimmed and barked if blown down […] No reproduction of this plant has been observed. However, there probably is some in protected areas.
    - Elderberry is all but gone from the average elk range in the park.
    - The huckleberry forms a limited portion of the elks diet. […] It is overbrowsed in most of the valleys. Reproduction is primarily confined to top of logs or in other inaccessible places.
37. 1957. Newman, C.C. 1957. Annual Report. 1957. Elk. Unpublished report on file at Olympic National Park. Port Angeles, WA

- Domestic cattle, feeding on park lands on the Hoh and Quinault during the spring and summer, are taking much forage that would otherwise be available for elk in the fall and winter. This continuous and excessive use of the range (spring and summer by cattle and fall and winter by elk) amounts to its unwarranted deterioration. The unauthorized grazing and browsing by the livestock also is having its effect on the physical condition of the elk.

38. 1958. Newman, C. C., 1958. Final Report. Roosevelt elk: Olympic National Park. Unpublished report on file at Olympic National Park, Port Angeles, WA

    - [page 3] As a result of the past two good winters, the population has increased to where it is probably approaching 5,000.
    - [page 7] There are four areas in the Park that suffer from grazing by unauthorized livestock. The two most critical of the four are the Hoh and the Quinault. [Author also references cattle at Big Flat in the South Fork Hoh]. The Humes Ranch-Anderson Ranch area on the Elwha suffers from unauthorized use of pack stock that does not belong to the Park Service.
    - [Summary, #1] The population of the Roosevelt elk on the Olympic Peninsula does not fluctuate a great deal because the factors affecting them are rather constant.
    - [Summary, #2] The vegetation inside the fenced plots grows rapidly and luxuriantly when the elk are excluded. This is especially true with the ferns, grasses, buttercup and salmonberry. Even though many other plants respond rapidly when the elk are prevented from feeding on them, there is much evidence that the range is not severely overused. Woody browse plants are very scarce in certain places on some watersheds, but as has been pointed out in previous reports, elk do not depend on this type of forage during periods of deep snow as they can then feed on fallen branches of Douglas fir, hemlock and cedar.
    - [Summary, #7] The only effective predator on elk is the cougar. There is a fair cougar population and they are important in keeping the elk population in check.
    - [Summary, #20] Evidence derived from the exclosure studies points up the fact that the Roosevelt elk create and maintain a habitat suitable for their needs. Intensive use of an area maintains it in meadow form. Once the feeding herds are excluded from an area, there is a spectacular growth that makes the rain forest appear as a very dense jungle. The jungle-like growth is not preferred by the elk.
    - [Conclusion] Information developed in this study indicates that the elk herds are numerous in certain areas, and are causing an over-browsed look to some plants. Even though this heavy use does occur, it is fairly sure that the range is not in such a poor condition that a reduction in elk is necessary.
    - [Conclusion] Erosion is usually of great concern on overstocked areas, however, on the west side, where the high populations occur, there is always enough big timber, down logs, moss and other debris to prevent erosion even if the browse plants are greatly reduced. And it appears that large concentrations of elk in the valleys cause the establishment and maintenance of park-like meadows with a good turf of herbaceous plants

- [Conclusion] It is accepted that the elk population in the Olympic National Park is rather stable and it seems that some contributing factors are: a mature forest, rapid and regular seasonal growth of forage plants, even pressure from predators, and natural die-off.

39. 1964. Royce, R. 1964. Annual wildlife report narrative. Unpublished report on file at Olympic National Park, Port Angeles, WA
    - In July, Mr. John Schwartz, United States Forest Service, Portland, Oregon, revisited for the first time the Park elk exclosure plots that he installed in the late 1930's. Mr. Schwartz felt that the forest vegetation around the Hoh exclosures had not materially changed and that conditions were about the same as he remembers them some 25 years ago.

Appendix F. Summary of hatchery data.

Table F.1. Summary of hatchery data.

River	**Hatchery Data sources**
Elwha	• Regional Mark Processing Center database • Hosey and Associates 1988
Queets	• Regional Mark Processing Center database • Bureau of Indian Affairs. 1977. Fisheries Management Report FY 1977. Quinault Department of Natural Resources and Economic Development. Bureau of Indian Affairs. Contract No. 14-20-0500-5021 • Bureau of Indian Affairs. 1980. Fisheries Management Report FY 1979. Quinault Department of Natural Resources and Economic Development. Bureau of Indian Affairs. Contract No. 14-20-0500-5021 • Quinault Indian Fisheries Division. 1975. Final Report Fisheries Management Contract (14-20-0500-4280). Table 22. • Quinault Indian Nation. 1976. Fisheries Management Final Report FY 1976. Contract No. 14-20-0500-4693. Quinault Department of Natural Resources and Economic Development. • Scott, J.B., and W.T. Gill. 2008. *Oncorhynchus mykiss*: assessment of Washington State's steelhead populations and programs. Available online at http://wdfw.wa.gov/publications/00150/wdfw00150.pdf. • QIN. Email to Olympic National Park from Tyler Jurasin
Quinault	• Regional Mark Processing Center database • Washington Department of Fish and Wildlife, Steelhead Historical Database, available online at http://wdfw.wa.gov/publications/00150/oly_pen_esu.pdf • Scott, J.B., and W.T. Gill. 2008. *Oncorhynchus mykiss*: assessment of Washington State's steelhead populations and programs. Available online at http://wdfw.wa.gov/publications/00150/wdfw00150.pdf. • Bureau of Indian Affairs. 1977. Fisheries Management Report FY 1977. Quinault Department of Natural Resources and Economic Development. Bureau of Indian Affairs. Contract No. 14-20-0500-5021 • Bureau of Indian Affairs. 1980. Fisheries Management Report FY 1979. Quinault Department of Natural Resources and Economic Development. Bureau of Indian Affairs. Contract No. 14-20-0500-5021 • Hiss et al. 1982. • Hiss, J.M., M.L. Paiya, and R.S. Boomer. 1984. Quinault National Fish Hatchery Salmon and Steelhead Coded Wire Tagging Results • Quinault Indian Fisheries Division. 1981. 1981 Preseason Management Report for Spring and Summer Fisheries on the Quinault and Queets Rivers. 57 p. • Quinault Indian Fisheries Division. 1975. Final Report Fisheries Management Contract (14-20-0500-4280). Table 22. • Quinault Indian Nation. 1976. Fisheries Management Final Report Fy 1976. Contract No. 14-20-0500-4693. Quinault Department of Natural Resources and Economic Development.

Table F.1 (continued). Summary of hatchery data.

River	Hatchery Data sources
Hoh	• Regional Mark Processing Center database • Hoh River Tribal Fisheries Program. Annual Report NO. 5, Fish Year 1980. April 1981 Contract No. P00C-1420-6230. 248 p. • Bureau of Indian Affairs. 1977. Fisheries Management Report FY 1977. Quinault Department of Natural Resources and Economic Development. Bureau of Indian Affairs. Contract No. 14-20-0500-5021 • US v Washington (1973, Table 30; and Appendix IV 1958-1972) • Bureau of Indian Affairs. 1977. Fisheries Management Report FY 1977. Quinault Department of Natural Resources and Economic Development. Bureau of Indian Affairs. Contract No. 14-20-0500-5021 • OLYM Files • Hiss et al. 1986 (early smolts are Cook Creek, Lake Quinault, and Bogachiel stocks; late smolts are Hoh native stock) via Bill McMillan
Quillayute	• Regional Mark Processing Center database • US v Washington (1973, Table 30; and Appendix IV 1958-1972) • Data from Randy Cooper, WDFW (OLYM Files) • Data from Mike Gross, 2012 via Bill McMillan (OLYM Files) • Myers, J.M., R.G. Kope, G.J. Bryant, D. Teel, L.J. Lierheimer, T.C. Wainwright, W.S. Grant, F.W. Waknitz, K. Neely, S.T. Lindley, and R.S. Waples. 1998. Stauts review of Chinook salmon from WA, ID, OR, and CA. NOAA Tech. Memo. NMFS-NWFSC-35, 443 p. • 1977/78-2003-04 obtained from Steelhead Historical Database

Appendix G. Summary statistics of years planted, numbers released and stock origins of hatchery fish released into selected Olympic Peninsula rivers.

Table G.1. Summary statistics of years planted, numbers released and stock origins of hatchery fish released into selected Olympic Peninsula rivers.

River	Stock	Period (yrs planted)	Total released	Mean annual release (range)	Hatchery stock origin by species (Based on hatchery sources in Appendix F)
Hoh	Spring Chinook	1960 – 1985 (6 of 26)	302,066	50,344 (660 – 100,000)	Big Soos, Deschutes, Dungeness, Hoh
	Fall Chinook	1958 – 1985 (12 of 28)	940,111	78,343 (960 – 333,000)	Big Soos, Deschutes, Dungeness, Hoh
	Chinook (Unspecified)	1987	94	94	Big Soos, Deschutes, Dungeness, Hoh
	Fall Coho	2005 – 2009 (4 of 5)	40,756	10,189 (3685 – 16,127)	Dungeness, Green River x Quinault, Hoh, Quinault, Skagit, Sol Duc, Willapa
	Coho (Unspecified)	1959 – 1991 (20 of 41)	2,493,645	124,682 (5330 – 521,469)	Dungeness, Green River x Quinault, Hoh, Quinault, Skagit, Sol Duc, Willapa
	Winter Steelhead	1959 – 2014 (54 of 56)	4,114,999	76,204 (14,200 – 227,115)	Hoh, Bogachiel, Quinault, Cook Creek, Chalaat Creek, Chambers Creek
	Summer Steelhead	1959 – 1983 (24 of 25)	939,779	39,157 (16,100 – 124,994)	Hoh, Bogachiel, Quinault, Cook Creek, Chalaat Creek, Chambers Creek
	Steelhead (Unspecified)	1984	217	217	Hoh, Bogachiel, Quinault, Cook Creek, Chalaat Creek, Chambers Creek
	Fall Chum	1976 – 1980 (2 of 5)	268,000	134,000 (3000 – 265,000)	Hoh, Quilcene, Quilcene x Quinault, Walcott Slough
	Chum (Unspecified)	1978 – 1981 (3 of 4)	758,000	262,667 (125,000 – 368,000)	Hoh, Quilcene, Quilcene x Quinault, Walcott Slough

Table G.1 (continued). Summary statistics of years planted, numbers released and stock origins of hatchery fish released into selected Olympic Peninsula rivers.

River	Stock	Period (yrs planted)	Total released	Mean annual release (range)	Hatchery stock origin by species (Based on hatchery sources in Appendix F)
Queets	Spring Chinook	1976, 2007	280,823	140,412 (72,953 – 207,870)	White, Cowlitz, Cook Cr, Deschutes, Green + Samish, Green, Queets, Quillayute, Quinault + Salmon, Quinault, Quinault x Queets, Salmon, Sol Duc Springs
	Fall Chinook	1976 – 2014 (40 of 40)	9,491,847	237,296 (17,732 – 757,900)	White, Cowlitz, Cook Cr, Deschutes, Green + Samish, Green, Queets, Quillayute, Quinault + Salmon, Quinault, Quinault x Queets, Salmon, Sol Duc Springs
	Chinook (Unspecified)	1979	204,189	204,189	White, Cowlitz, Cook Cr, Deschutes, Green + Samish, Green, Queets, Quillayute, Quinault + Salmon, Quinault, Quinault x Queets, Salmon, Sol Duc Springs
	Fall Coho	1999 – 2014 (14 of 16)	10,427,540	744,824 (22,634 – 1,191,624)	Christmas Creek, Clearwater + Sol Duc, Clearwater, Clearwater Tributaries, Cook Creek, Cook Creek x Salmon River, Fork Creek, Green River x Cook Creek, Matheny Creek, Miller Creek, Queets, Queets Tributaries, Quillayute, Quinault + Queets, Quinault, Quinault x Green, Quinault X Queets, Quinault x Quilcene, Quinault x Willapa, Salmon, Sol Duc, Toutle, Walcott Slough, Willapa x Cook Creek, Willapa x Quinault
	Coho (Unspecified)	1974 – 2012 (28 of 39)	23,053,269	823,331 (170,100 –3,133,648)	Christmas Creek, Clearwater + Sol Duc, Clearwater, Clearwater Tributaries, Cook Creek, Cook Creek x Salmon River, Fork Creek, Green River x Cook Creek, Matheny Creek, Miller Creek, Queets, Queets Tributaries, Quillayute, Quinault + Queets, Quinault, Quinault x Green, Quinault X Queets, Quinault x Quilcene, Quinault x Willapa, Salmon, Sol Duc, Toutle, Walcott Slough, Willapa x Cook Creek, Willapa x Quinault
	Winter Steelhead	1978 – 2014 (36 of 37)	5,258,310	146,064 (665 – 219,831)	Clearwater Tributaries, Cook Creek, Queets + Clearwater, Queets Tributaries, Queets, Quinault + Queets, Quinault + Salmon, Quinault, Quinault + Cook Creek, Salmon, Quinault x Queets
	Steelhead (Unspecified)	1984-2002 (10 of 19)	15,109	1511 (230-2828)	Clearwater Tributaries, Cook Creek, Queets + Clearwater, Queets Tributaries, Queets, Quinault + Queets, Quinault + Salmon, Quinault, Quinault + Cook Creek, Salmon, Quinault x Queets
	Chum (Unspecified)	1975-1980 (4 of 6)	2,867,500	716,875 (300,000 – 990,000)	Walcott Slough

Table G.1 (continued). Summary statistics of years planted, numbers released and stock origins of hatchery fish released into selected Olympic Peninsula rivers.

River	Stock	Period (yrs planted)	Total released	Mean annual release (range)	Hatchery stock origin by species (Based on hatchery sources in Appendix F)
Quillayute	Spring Chinook	1972 – 2008 (36 of 37)	13,628,952	378,582 (21,599 – 1,843,384)	Cowlitz, Cowlitz x Umpqua, Dungeness, Quillayute + Sol Duc, Sol Duc, Big Soos Creek, Bogachiel, Deschutes, Elwha, Finch Creek, Green River, Hood Canal Hatchery, Issaquah Creek Hatchery, Lake Creek, Lake Creek x Winfield Creek, Nemah x Cook Creek, Nemah x Lake Quinault, Quillayute, Sol Duc Hatchery, Sol Duc
	Fall Chinook	1971 – 1994 (22 of 24)	6,168,697	237,258 (14,000 – 978,888)	Cowlitz, Cowlitz x Umpqua, Dungeness, Quillayute + Sol Duc, Sol Duc, Big Soos Creek, Bogachiel, Deschutes, Elwha, Finch Creek, Green River, Hood Canal Hatchery, Issaquah Creek Hatchery, Lake Creek, Lake Creek x Winfield Creek, Nemah x Cook Creek, Nemah x Lake Quinault, Quillayute, Sol Duc Hatchery, Sol Duc
	Chinook (Unspecified)	1973 – 2014 (37 of 42)	8,784,635	237,422 (13,300 – 975,150)	Cowlitz, Cowlitz x Umpqua, Dungeness, Quillayute + Sol Duc, Sol Duc, Big Soos Creek, Bogachiel, Deschutes, Elwha, Finch Creek, Green River, Hood Canal Hatchery, Issaquah Creek Hatchery, Lake Creek, Lake Creek x Winfield Creek, Nemah x Cook Creek, Nemah x Lake Quinault, Quillayute, Sol Duc Hatchery, Sol Duc
	Fall Coho	1974 – 2009 (4 of 36)	666,991	166,748 (518 – 433,800)	Bear Creek, Big Quilcene, Big Soos, Bogachiel, Cowlitz, Dickey, Dickey Tributaries, East Fork Dickey, Dungeness, Elk Creek, George Adams, Hemphill + Fuhrman, Hemphill + Mill Creek, Hoh, Humptulips, Lake Creek, Lake Creek X Sol Duc, Lonesome Creek, Queets, Quillayute + Sol Duc, Quillayute, Quillayute Tributaries, Quinault, Satsop Springs, Skagit, Sol Duc, Sol Duc x Bogachiel, Swamp Creek, Washington-General
	Summer Coho	1973 – 2014 (38 of 42)	14,956,053	393,580 (7800 – 2,526,399)	Bear Creek, Big Quilcene, Big Soos, Bogachiel, Cowlitz, Dickey, Dickey Tributaries, East Fork Dickey, Dungeness, Elk Creek, George Adams, Hemphill + Fuhrman, Hemphill + Mill Creek, Hoh, Humptulips, Lake Creek, Lake Creek X Sol Duc, Lonesome Creek, Queets, Quillayute + Sol Duc, Quillayute, Quillayute Tributaries, Quinault, Satsop Springs, Skagit, Sol Duc, Sol Duc x Bogachiel, Swamp Creek, Washington-General

Table G.1. Summary statistics of years planted, numbers released and stock origins of hatchery fish released into selected Olympic Peninsula rivers.

River	Stock	Period (yrs planted)	Total released	Mean annual release (range)	Hatchery stock origin by species (Based on hatchery sources in Appendix F)
Quillayute (continued)	Coho (Unspecified)	1954-2014 (60 of 61)	48,276,844	804,614 (11,550 – 2,900,673)	Bear Creek, Big Quilcene, Big Soos, Bogachiel, Cowlitz, Dickey, Dickey Tributaries, East Fork Dickey, Dungeness, Elk Creek, George Adams, Hemphill + Fuhrman, Hemphill + Mill Creek, Hoh, Humptulips, Lake Creek, Lake Creek X Sol Duc, Lonesome Creek, Queets, Quillayute + Sol Duc, Quillayute, Quillayute Tributaries, Quinault, Satsop Springs, Skagit, Sol Duc, Sol Duc x Bogachiel, Swamp Creek, Washington-General
	Winter Steelhead	1933 – 2014 (57 of 82)	9,166,240	160,811 (15,020 – 918,350)	Bogachiel/Chambers Creek, Cook Creek, Hoko, Lonesome Creek, Quillayute, Quinault, Sol Duc, Chehalis, Quillayute, Skykomish, Sol Duc, Washington General, Washougal
	Summer Steelhead	1982 – 2014 (33 of 33)	1,608,914	48,755 (10,461 – 97,324)	Bogachiel/Chambers Creek, Cook Creek, Hoko, Lonesome Creek, Quillayute, Quinault, Sol Duc, Chehalis, Quillayute, Skykomish, Sol Duc, Washington General, Washougal
	Steelhead (Unspecified)	1977 – 1980 (3 of 4)	107,592	35,864 (19,800-52,800)	Bogachiel/Chambers Creek, Cook Creek, Hoko, Lonesome Creek, Quillayute, Quinault, Sol Duc, Chehalis, Quillayute, Skykomish, Sol Duc, Washington General, Washougal
	Chum (Unspecified)	1964 – 1986 (4 of 23)	535,894	133,974 (7000 – 274,000)	Finch Creek, Minter Creek, Walcott, Washington General
Quinault	Spring Chinook	1976 – 1978 (3 of 3)	693,288	231,096 (68,288 – 430,000)	Cowlitz, Cook Creek, Deschutes, Finch Creek x Cook Creek, Fork Creek, Hoh x Cook Creek, Hoh x Quinault, Lake Union, Lake Union x Cook Creek, Lake Union x Queets, Lake Union x Quinault, Nemah, Nemah x Cook Creek, Nemah x Deschutes, Queets, Quillayute, Lake Quinault, Quinault, Quinault x Green, Quinault x Hoh, Quinault x Hoodsport, Quinault x Nemah, Quinault x Queets, Quinault x Samish, Quinault x Willapa, Willapa, Willapa x Cook Creek

Table G.1 (continued). Summary statistics of years planted, numbers released and stock origins of hatchery fish released into selected Olympic Peninsula rivers.

River	Stock	Period (yrs planted)	Total released	Mean annual release (range)	Hatchery stock origin by species (Based on hatchery sources in Appendix F)
Quinault (continued)	Fall Chinook	1915 – 2014 (69 of 100)	51,035,974	739,652 (1810 – 2,236,238)	Cowlitz, Cook Creek, Deschutes, Finch Creek x Cook Creek, Fork Creek, Hoh x Cook Creek, Hoh x Quinault, Lake Union, Lake Union x Cook Creek, Lake Union x Queets, Lake Union x Quinault, Nemah, Nemah x Cook Creek, Nemah x Deschutes, Queets, Quillayute, Lake Quinault, Quinault, Quinault x Green, Quinault x Hoh, Quinault x Hoodsport, Quinault x Nemah, Quinault x Queets, Quinault x Samish, Quinault x Willapa, Willapa, Willapa x Cook Creek
	Fall Coho	1915 – 2004 (29 of 90)	9,691,293	334,182 (34,026 – 1,398,000)	Bingham, Cook Creek, Cook x Salmon, Cowlitz, Eagle Creek, Fork Creek, George Adams, Green x Cook Creek, Humptulips, Moclips, Queets, Quillayute, Quinault, Quinault + Queets, Quinault x Queets, Quinault x Quilcene, Salmon, Satsop Springs, Simpson, Skagit x Cook, Skagit x Quinault, Skykomish + George Adams, Sol Duc, Sooes, Willapa, Willapa x Cook Creek
	Coho (Unspecified)	1968 – 2014 (44 of 47)	51,843,372	1,178,258 (4500 – 4,099,805)	Bingham, Cook Creek, Cook x Salmon, Cowlitz, Eagle Creek, Fork Creek, George Adams, Green x Cook Creek, Humptulips, Moclips, Queets, Quillayute, Quinault, Quinault + Queets, Quinault x Queets, Quinault x Quilcene, Salmon, Satsop Springs, Simpson, Skagit x Cook, Skagit x Quinault, Skykomish + George Adams, Sol Duc, Sooes, Willapa, Willapa x Cook Creek
	Sockeye	1916 – 2014 (63 of 99)	193,721,872	3,074,950 (8200 – 18,137,000)	Big Creek, Lake Quinault, Quinault
	Winter Steelhead	1973 – 2014 (42 of 42)	26,227,210	624,457 (22,250 – 1,837,345)	Cook Creek, Eagle Creek, Queets, Quillayute, Lake Quinault, Quinault, Salmon
	Steelhead (Unspecified)	1922 – 2010 (17 of 89)	1,917,573	112,798 (2177 – 478,165)	Cook Creek, Eagle Creek, Queets, Quillayute, Lake Quinault, Quinault, Salmon
	Chum	1970 – 2014 (45 of 45)	69,141,728	1,536,483 (38,559 – 5,939,613)	Big Quilcene, Cook Creek, Elson Creek, Quilcene x Quinault, Quinault, Quinault x Quilcene, Quinault x Wolcott, Wolcott Slough, Wolcott x Cook Creek

Table G.1 (continued). Summary statistics of years planted, numbers released and stock origins of hatchery fish released into selected Olympic Peninsula rivers.

River	Stock	Period (yrs planted)	Total released	Mean annual release (range)	Hatchery stock origin by species (Based on hatchery sources in Appendix F)
Elwha	Winter Steelhead	1958 – 2013 (49 of 56)	4,324,521	88,256 (12,306 – 302,798)	Bogachiel/Chambers Creek, Elwha, Oregon-Eagle Creek National Fish Hatchery, Chehalis, Quillayute, Sol Duc, Washington General, Washougal
	Summer Steelhead	1968 – 2000 (27 of 33)	513,180	19,007 (10,000 – 25,388)	Bogachiel/Chambers Creek, Elwha, Oregon-Eagle Creek National Fish Hatchery, Chehalis, Quillayute, Sol Duc, Washington General, Washougal
	Steelhead (Unspecified)	1993 – 2012 (2 of 20)	390,184	195,092 (161,038 – 229,146)	Bogachiel/Chambers Creek, Elwha, Oregon-Eagle Creek National Fish Hatchery, Chehalis, Quillayute, Sol Duc, Washington General, Washougal
	Summer/Fall Chinook	1916 – 2014 (66 of 99)	104,777,938	1,587,544 (60,648 – 4,265,153)	Big Soos Creek, Dungeness, Elwha, Finch Creek, Issaquah Creek, Sol Duc, Spring Creek
	Spring Chinook	1954 – 1977 (6 of 24)	1,740,738	290,123 (199,200 – 532,647)	Big Soos Creek, Dungeness, Elwha, Finch Creek, Issaquah Creek, Sol Duc, Spring Creek
	Fall Coho	1992 – 2014 (13 of 23)	5,928,675	102,412 (40,763 – 164,062)	Elwha, Big Soos, Dungeness, Satsop Springs, Washington General
	Coho (Unspecified)	1952 – 2006 (40 of 55)	29,232,716	730,818 (3400 – 4,715,720)	Elwha, Big Soos, Dungeness, Satsop Springs, Washington General
	Fall Chum	1920 – 2014 (14 of 95)	517,236	43,103 (1,776 – 105,770)	Big Quilcene, Elwha, Enetai Creek, Walcott Slough, Washington General
	Chum (Unspecified)	1977 – 1997 (12 of 21)	11,904,333	992,028 (17,710 – 2,866,339)	Big Quilcene, Elwha, Enetai Creek, Walcott Slough, Washington General
	Pink	2012, 2014	204,825	104,412 (40,763 – 164,062)	Elwha

Appendix H. Summary of assessment criteria used for each stock based on escapement, terminal run size, and harvest data.

Table H.1. Summary of assessment criteria used for each stock based on escapement, terminal run size, and harvest data. Stocks were also evaluated based on cumulative hatchery releases reported in Appendix G. Data were compiled from a variety of sources contained in Olympic National Park fisheries archives.

River	Stock	Escapement Goal met (yrs of record)	Escapement goal met (last 10 yrs in record)	Mean wild escaped	Mean term. run size	Total harvest (comm. +sport)	Mean Harvest rate % (range)[1]	Peak historical harvest	Other[2]	Condition status and trend
Hoh	Spring Chinook	29 of 38	4	1399	1941	17,840 + 6065	28 (3–55)[a]	unknown	a, b	🔴 ↔
Hoh	Fall Chinook	39 of 41	10	2701	3881	36,349 + 7549	30 (18–52)[a]	2973	c	🟢 ↔
Hoh	Fall Coho	30 of 37	9	4130	5930	70,557 + 8314	33 (5–61)[a]	7495	–	🟢 ↔
Hoh	Winter Steelhead	23 of 39	5	2693	4294	36,298 + 17,133	35 (7–54)[b]	4320	b	🟡 ↓
Queets	Spring Chinook	22 of 45	1	876	899	6244 + 164	15 (0–58)[b]	unknown	a	🔴 ↔

[1] a=hatchery + wild harvest rate; b = only wild harvest rate

[2] a=Recent/current fishing closure; b =high fishing pressure, c=lacks historical baseline; d=genetic concerns with wild broodstock program; e=FAB Overfishing; f=unique wild run; g=extensive hatchery releases; h = large historical catch.

Table H.1 (continued). Summary of assessment criteria used for each stock based on escapement, terminal run size, and harvest data. Stocks were also evaluated based on cumulative hatchery releases reported in Appendix G. Data were compiled from a variety of sources contained in Olympic National Park fisheries archives.

River	Stock	Escapement Goal met (yrs of record)	Escapement goal met (last 10 yrs in record)	Mean wild escaped	Mean term. run size	Total harvest (comm. +sport)	Mean Harvest rate % (range)[1]	Peak historical harvest	Other[2]	Condition status and trend
Queets	Fall Chinook	40 of 47	9	4342	5396	66,117 +6044	32 (8–53)[a]	9249	d	🟡⇩
Queets	Fall Coho	17 of 37	6	6733	8897	289,747 + 14,132	42 (9–68)[a]	10,712	a, e	🔴⇔
Queets	Winter Steelhead	28 of 43	6	4840	7785	140,949 + 34,941	35 (9–54)[b]	13,182	–	🔴⇔
Quillayute	Spring Chinook	12 of 39	0	1165	3063	35,079 + 9571	36 (12–59)[a]	Unknown	–	🔴⇩
Quillayute	Fall Chinook	37 of 39	10	6027	8535	79745 + 12,721	28 (8–61)[a]	4478	–	🟢⇔
Quillayute	Fall Coho	27 of 39	8	9309	15,864	383,874 + 39,102	35 (4–66)[a]	17,322	–	🟢⇔

[1] a=hatchery + wild harvest rate; b = only wild harvest rate

[2] a=Recent/current fishing closure; b =high fishing pressure, c=lacks historical baseline; d=genetic concerns with wild broodstock program; e=FAB Overfishing; f=unique wild run; g=extensive hatchery releases; h = large historical catch.

Table H.1 (continued). Summary of assessment criteria used for each stock based on escapement, terminal run size, and harvest data. Stocks were also evaluated based on cumulative hatchery releases reported in Appendix G. Data were compiled from a variety of sources contained in Olympic National Park fisheries archives.

River	Stock	Escapement Goal met (yrs of record)	Escapement goal met (last 10 yrs in record)	Mean wild escaped	Mean term. run size	Total harvest (comm. +sport)	Mean Harvest rate % (range)[1]	Peak historical harvest	Other[2]	Condition status and trend
Quillayute	Summer Coho	no goal	no goal	866	1505	104,150 + 7298	32 (4–84)[a]	Unknown	f	🔴⇩
Quillayute	Winter Steelhead	33 of 37	9	9691	14,877	228,455 + 173,665	53 (26–78)[b]	10,504	–	🟢⇔
Quinault	Spring Chinook	N/A	N/A	N/A	N/A	4041(C)	N/A	N/A	–	◌
Quinault	Fall Chinook	N/A	N/A	N/A	N/A	217,013 + N/A	N/A	7372	–	◌
Quinault	Fall Coho	N/A	N/A	7454	18,977	578,747 + N/A	49 (8–78)[a]	44,966	h	🟡⇔
Quinault	Sockeye	N/A	N/A	N/A	N/A	N/A	N/A	N/A	–	◌

[1] a=hatchery + wild harvest rate; b = only wild harvest rate

[2] a=Recent/current fishing closure; b =high fishing pressure, c=lacks historical baseline; d=genetic concerns with wild broodstock program; e=FAB Overfishing; f=unique wild run; g=extensive hatchery releases; h = large historical catch.

Table H.1 (continued). Summary of assessment criteria used for each stock based on escapement, terminal run size, and harvest data. Stocks were also evaluated based on cumulative hatchery releases reported in Appendix G. Data were compiled from a variety of sources contained in Olympic National Park fisheries archives.

River	**Stock**	**Escapement Goal met (yrs of record)**	**Escapement goal met (last 10 yrs in record)**	**Mean wild escaped**	**Mean term. run size**	**Total harvest (comm. +sport)**	**Mean Harvest rate % (range)[1]**	**Peak historical harvest**	**Other[2]**	**Condition status and trend**
Quinault	Winter Steelhead	no goal	no goal	3388	6640	266,687 +30,335	48 (15–65)[a]	8712	–	⇩

[1] a=hatchery + wild harvest rate; b = only wild harvest rate

[2] a=Recent/current fishing closure; b =high fishing pressure, c=lacks historical baseline; d=genetic concerns with wild broodstock program; e=FAB Overfishing; f=unique wild run; g=extensive hatchery releases; h = large historical catch.

Appendix I. Estimation of sea otter predation on adult razor clams on Kalaloch Beach, Olympic National Park.

The number of adult razor clams (A_r) consumed by sea otters on Kalaloch beach in any given year was estimated as:

$$A_r = P_{ok} * O_{sc} * D * R_c$$

Where:

P_{ok} = the proportion of the otter population at Kalaloch estimated as the constant 6% of the annual total south coast otter population. Six percent represents a likely overestimation of otter use of Kalaloch, as less than 1% of the of the south coast otter population was observer in the Kalaloch area from 1998-2015.

O_{sc} = the annual number of otters on the south coast, from north of Hoh head to the Quinault reservation.

D = the number of days in a year, the constant 365.

R_c = the number of razor clams consumed per otter per day calculated as:

$$R_c = (W_f / W_{ac}) * P_d$$

Where:

W_f = the total weight of food an individual otter eats per day (the constant 9 kg), estimated as 30% of the 30 kg body mass of an average adult otter (Riedman and Estes 1990).

W_{ac}= The annual average weight (g) of an adult razor clam estimated from the annual average adult shell length, using the equation $log_{10}W = 3.3058 \ log_{10}L - 4.7813$ (Jones et al. 1998), where W = clam weight (g) and L = clam length (mm).

P_d= The proportion of a sea otter diet made up of razor clams, estimated as a constant of 1%. This estimate is two orders of magnitude higher than that estimated by Laidre and Jameson (2006), who observed that razor clams make up less than 0.01% of Washington coastal sea otter diets.

Appendix J. Estimation of trends in territory occupancy dynamics of northern spotted owls and barred owls in Olympic National Park and Olympic National Forest.

Survey Methods

Determination of occupancy of historical territories of northern spotted owls followed established survey protocols (Forsman 1995, Franklin et al. 1996). We used vocal imitations of northern spotted owls to elicit responses from territorial individuals, followed by offers of live mice to determine nesting and reproductive status. Two-person NPS crews visited each site a minimum of three times between mid-March and late July, or until they determined pair, nesting and reproductive status of any spotted owls occupying the site (Gremel 2014). Most surveys were performed during the daytime because of the safety hazards of working off established trails at night. When spotted owls were not found near historic locations, crews searched a 1 – 2 km around the last known location to locate and identify color-banded birds. When owls were found greater than 2 km from a previously activity center (i.e., nest), the decision whether to consider the new locations as a new territory was based on knowledge of the area used by the banded birds at the site in the past, local topography, locations of adjacent territories, and nearest neighbor distances (Gremel 2005, 2014).

Barred owls often respond to calls of spotted owls (Wiens et al. 2011), and field crews recorded all detections of barred owls during surveys of spotted owls at territories historically occupied by spotted owls. Protocols designed to survey spotted owls are less effective for detecting barred owls (Wiens et al. 2011), so annual estimates of occupancy of historical spotted owl territories by barred owls that are not corrected for imperfect detection should be considered minimums (Yackulic et al. 2014, 2015).

Territory Occupancy Dynamics of Spotted Owls and Barred Owls

We used standardized analytical methods applied in regional and range-wide analyses to assess colonization of barred owls into OLYM and associated impacts on territory occupancy dynamics of spotted owls (Forsman et al. 2011, Sovern et al. 2014, Dugger et al. 2016). Specifically, we investigated trends in territory-occupancy dynamics of spotted owls and barred owls separately based on 22 years of monitoring data (1992 – 2013). Recent analyses of site occupancy by spotted owls and barred owls in the Olympic study area used a conditional, two-species occupancy model to determine co-occurrence dynamics of the two species (Dugger et al. 2016). Our intention was not to repeat these analyses, but rather to use single-species site occupancy analyses (MacKenzie et al. 2003) to evaluate spatial patterns of colonization and use of spotted owl territories by each owl species separately. Using this analytical framework, we estimated four parameters for each owl species: site occupancy in the first year of study (initial occupancy in 1992; ψ_{1992}), the probability an occupied site became unoccupied in the subsequent year (extinction; ε), the probability an unoccupied site was re-occupied in the subsequent year (colonization; γ), and the probability of detection (p). We used Theissen polygons (Dugger et al. 2016:62 – 63) to delineate historical territories of spotted owls, which we defined as a 3.6 – 27.3 km^2 patch that represented the cumulative area of use by a spotted owl, or pair of spotted owls, during the study period (e.g., Figure 4.4.1 in Chapter 4.4.1). Actual territory boundaries of individual barred owls or spotted owls may overlap >1 Theissen polygon in different years of the study, so the occupancy parameter (ψ) can be interpreted as the probability of ≥1 spotted owl or barred owl using a given site (Theissen polygon) during the breeding season (MacKenzie et al. 2006). We used this analytical approach to assess time-trends in parameter estimates and potential differences between land ownerships (OLYM, ONF) in occupancy dynamics of each owl species.

We estimated occupancy parameters for each species by applying the same step-wise approach and set of candidate models (Table J.1) to annual survey data collected for each owl species (Olson et al. 2004, Dugger et al. 2011). First, we determined the best model structure for detection probabilities by estimating support for time-dependent effects, including linear (T), log-linear (ln T), and quadratic (TT) trends, as well as a general year-specific effect (t) on among- and within-year detection probabilities. We also examined whether detection probabilities varied with land ownership (OLYM, ONF). We used the most general model for initial occupancy, extinction, and colonization parameters while identifying the best model structure for detection (Sovern et al. 2014, Dugger et al. 2016). Second, we retained the best-supported model of detection probability and proceeded to model initial occupancy, colonization, and extinction parameters with and without the effects of land ownership (area). Third, we examined support for alternative time-related effects on extinction parameters, including a linear time trend (T), a year-specific effect (t), a pseudo-threshold effect (lnT), or no temporal effects (intercept-only). In total, we considered 37 candidate models to investigate the effects of land ownership and time on detection, extinction, and colonization of each owl species (Table J.1). We used information theoretic methods (Burnham and Anderson 2002) to select the best models at each step of our analysis. We evaluated the degree to which 95% confidence intervals of regression coefficients (β) overlapped zero to determine direction, precision, and strength of evidence for effects of interest (time trends, land ownership, and site-specific covariates described below). We used the best overall model from the preceding analysis steps to calculate estimates of year-specific probabilities of site occupancy following MacKenzie et al. (2003), which were estimated as derived parameters in Program MARK.

In the final step of our assessment, we introduced site-specific covariates into the overall top model resulting from the constrained set of detection, occupancy, extinction, and colonization models for each owl species. A barred owl covariate was developed to model the effect of the presence of barred owls on site occupancy dynamics of spotted owls, following Dugger et al. (2011) and Sovern et al. (2014). Here, a year- and site-specific covariate was coded as “1” if a barred owl was detected on the site during any survey and “0” if not detected. Extinction (ε) and colonization (γ) parameters are interval estimates from time i to time $i + 1$, so we investigated the effect of barred owl presence at time i (BO) and barred owl presence on time $i + 1$ (BO1) on these parameters for spotted owls (Olson et al. 2005, Dugger et al. 2011, Sovern et al. 2014). Previous studies of barred owls in OLYM suggest that areas colonized first by barred owls included low-elevation patches of older forest with relatively flat terrain, often with a component of deciduous trees and riparian areas (Gremel 2005). Consequently, we determined support for the effects of: 1) mean elevation, and 2) terrain ruggedness on annual colonization rates of barred owls.

Table J.1. Candidate single-species occupancy models used to examine initial occupancy (ψ_1), local extinction (ε), colonization (γ), and detection (*p*) of northern spotted owls and barred owls in Olympic National Park and Olympic National Forest, Washington, 1992 – 2013.

Measure calculated	**Model**	**Description**
Detection probability (*p*)	*p*(area * t, t)	Full model
	p(area * t, .)	Detection varies between areas and years, but is constant within years
	p(.)	Detection is constant between areas and over time
	p(area)	Detection differs between areas
	p(t, .)	Detection varies among years, but is constant within years
	p(t, t)	Detection varies among and within years
	p(t, T)	Detection varies among years and follows a linear time trend within years
	p(t, lnT)	Detection varies among years and follows a pseudo-threshold trend within years
	p(t, TT)	Detection varies among years and follows a non-linear time trend within years
	p(area + t, .)	Additive effects of area and year on detection, but constant within years
	p(area + t, T)	Additive effects of area and year on detection with a linear time trend within years
	p(area + T, .)	Additive effects of area and a linear time trend on detection, constant within years
Initial Occupancy (ψ)	ψ(.)	Constant initial occupancy
	ψ(area)	Initial occupancy differs between areas
Extinction probability (ε)	ε(.)	Constant colonization probability
	ε(area)	Colonization probability differs between areas
	ε(t)	Year-specific colonization probability
	ε(T)	Colonization probability follows a linear time trend
	ε(TT)	Colonization probability follows a non-linear time trend
	ε(lnT)	Colonization probability follows a pseudo-threshold time trend
	ε(area + t)	Additive effect of area and time
	ε(area * t)	Interaction between area and time
	ε(area + T)	Area effect with additive time trend
	ε(area + TT)	Area effect with non-linear time trend
	ε(area * T)	Interaction between area and linear time trend
Colonization probability (ϒ)	ϒ(.)	Constant colonization probability
	ϒ(area)	Colonization probability differs between areas
	ϒ(t)	Year-specific colonization probability
	ϒ(T)	Colonization probability follows a linear time trend
	ϒ(TT)	Colonization probability follows a non-linear time trend
	ϒ(lnT)	Colonization probability follows a pseudo-threshold time trend

Table J.1 (continued). Candidate single-species occupancy models used to examine initial occupancy (ψ_1), local extinction (ε), colonization (γ), and detection (p) of northern spotted owls and barred owls in Olympic National Park and Olympic National Forest, Washington, 1992 – 2013.

Measure calculated	Model	Description
Colonization probability (Y) (continued)	Y(area + t)	Additive effect of area and time
	Y(area * t)	Interaction between area and time
	Y(area + T)	Area effect with additive time trend
	Y(area + TT)	Area effect with non-linear time trend
	Y(area*T)	Interaction between area and a linear time trend

Results

Occupancy data for spotted owls and barred owls included a total of 264 sampling periods across all years and sites combined during 1992 – 2013 (22 primary sampling periods). Barred owls were detected on <10% of historical territories of spotted owls until 1997, at which point the percentage of territories where barred owls were detected increased to 52% in OLYM and 44% in ONF by 2013.

Spotted Owls

The best occupancy model for spotted owls had strong support from the data, as it accounted for most the AIC$_c$ weight (0.73), and included the effects of general time variation and the presence of barred owls on territory occupancy dynamics. Model selection results did not support a difference in detection rates or local extinction probability of spotted owls between OLYM and ONF (Table J.2). In contrast, we found strong support for the effect of land ownership on colonization rates of spotted owls ($\hat{\beta}$ = 0.78, 95% CI = 0.34 to 1.22). We found strong support for an association between the presence of barred owls and detection, extinction, and colonization rates of spotted owls (Table J.2). We found that the detection of ≥1 barred owl at a spotted owl territory in year $i + 1$ was positively correlated with an increase in the probability of extinction of spotted owls during the interval between year i and $i + 1$, as shown by the strong support for the time-forward effect of barred owls (BO1) on local extinction of spotted owls in our top model ($\hat{\beta}$ = 0.85, 95% CI = 0.37 to 1.33). We also identified strong support for a negative association between the detection of barred owls in year i and colonization probability of spotted owls in year i ($\hat{\beta}$ = –1.29, 95% CI = –2.06 to –0.53). We found no evidence that the influence of barred owls on colonization rates of spotted owls varied with land ownership, as shown by a lack of support for an interaction between the effects of area and barred owls (ΔAIC$_c$ = 2.03; $\hat{\beta}$ = 0.33, 95% CI = –1.60 to 2.26; Table J.2).

Table J.2. Ranking of top models used to examine variation in initial occupancy (ψ_{1992}), extinction (ε), colonization (γ), and detection (p) probabilities of northern spotted owls in Olympic National Park and Olympic National Forest, Washington, 1992 – 2013.

Measure calculated	**Model[1]**	**ΔAIC_c**	**K**	**w_i**	**Deviance**
Initial Occupancy (ψ_{1992})	ψ(.)	0.00	151	0.75	8361.07
	ψ(area)	2.17	152	0.25	8360.91
Extinction probability (ε)	ε(lnT + BO1)	0.00	73	0.93	8347.65
	ε(lnT + BO)	5.29	73	0.07	8352.94
	ε(lnT)	164.64	111	0.00	8428.92
	ε(T)	165.62	111	0.00	8429.90
	ε(TT)	167.23	112	0.00	8429.28
Colonization probability (γ)	γ(area + T + BO)	0.00	74	0.73	8331.38
	γ(area + T + area*BO)	2.03	75	0.27	8331.26
	γ(area + T + BO1)	16.26	74	0.00	8347.64
	γ(area + T)	35.21	72	0.00	8370.90
	γ(area + TT)	127.93	73	0.00	8461.47
Detection probability (*p*)	*p*(t + BO, TT)	0.00	72	1.00	8370.90
	p(t, TT)	169.21	152	0.00	8360.91
	p(area * t, .)	190.77	130	0.00	8433.23
	p(area + t, T)	209.97	152	0.00	8401.66
	p(area + t, .)	214.56	130	0.00	8457.02

[1] Models were ranked according to Akaike's Information Criterion adjusted for small sample size (AIC_c). Model deviance, number of parameters (K) and AICc weights (w_i) are shown for each model. Time effects were modeled as constant (.) varying with years (t), a linear time trend (T), a quadratic time trend (TT), or a pseudo-threshold time trend (lnT). Site-specific covariates include land ownership (area), detection of ≤1 barred owl within the site in year *i* (BO), and detection of ≤1 barred owl within the site in year *i* + 1 (BO1).

Barred Owls

Detection probabilities of barred owls were greater on Olympic National Forest than Olympic National Park ($\hat{\beta}$ = −0.41, 95% CI = −0.60 to −0.22): the probability of detecting ≥1 barred owls when they were present increased from 0.08 in 1992 to 0.25 in 2013 at Olympic National Park, and from 0.11 to 0.33 during this same time period at Olympic National Forest (Table J.4). This difference may reflect the difference in night survey effort between areas, as limited road access in OLYM meant that most surveys were restricted to being conducted during the daytime. The increasing probability of detecting barred owls over time was likely a result of increasing abundance of barred owls in the region, rather than individual barred owls becoming more detectable. The best-supported models for initial occupancy indicated that the probability of use by ≥1 barred owls in 1992 was 0.08 (SE = 0.04, 95% CI = 0.03 to 0.20), and did not vary with land ownership. This model indicated that extinction probability of barred owls remained low and constant during the study period ($\hat{\varepsilon}$ = 0.04, SE = 0.01, 95% CI = 0.04 to 0.22). The best model for extinction probabilities (ΔAIC_c = 0.00; w_i = 0.73) was the null model; models that did include the effects of time or area on extinction probability received poor support from the data (w_i <0.01).

The best model of colonization rates of barred owls without covariates included the effects of area, a linear time trend, and an interaction between area and a linear time trend. This model accounted for 69% of the AIC$_c$ weight and indicated a positive linear time trend in colonization ($\hat{\beta}$ = 0.12, 95% CI = 0.06 to 0.19) that was greater (i.e., steeper slope) at Olympic National Park ($\hat{\beta}$ = 1.97, 95% CI = 0.88 to 3.07) than at Olympic National Forest. When we added the site-specific covariate for topographic roughness to our best-supported model for local colonization of barred owls, a model with a lower AIC$_c$ value was obtained (Table J.3). This model indicated that colonization of historical spotted owl territories by ≥1 barred owl declined with increasing topographic roughness of the site, and the confidence interval for this parameter barely overlapped zero ($\hat{\beta}$ = −0.003, 95% CI = −0.006 to 0.001). We used the top model for barred owls [ψ(.) ε(.) γ(g × T + eleSD) p(g + T)] to calculate year-specific estimates of occupancy parameters. The estimated probability of occupancy by barred owls increased in both study areas by 86 – 88% (Olympic National Park: $\widehat{\psi}_{1992}$ = 0.09 (SE= 0.04); $\widehat{\psi}_{2013}$ = 0.77 (SE = 0.05), Olympic National Forest: $\widehat{\psi}_{1992}$ = 0.09 (SE = 0.04); $\widehat{\psi}_{2013}$ = 0.0.67 (SE = 0.07).

Between 1992 and 1995, the estimated proportion of historical territories of spotted owls that were used by ≥1 barred owl on OLYM increased from 0.09 to 0.29, a 69% increase in occurrence in three years. In contrast, the estimated proportion of historical territories of spotted owls that were used by ≥1 barred owl on ONF increased from 0.09 to 0.12, a mere 25% increase in comparison. This result indicated that use of historical territories of spotted owls by barred owls in a landscape with no timber harvest and relatively little forest management increased at a greater rate than adjacent lands managed by the Forest Service. This result was also consistent with other studies showing the barred owl's use of predominately older forests for nesting, roosting, and foraging activities (Hamer et al. 2007, Singleton et al. 2010, Wiens et al. 2014). The estimated and naïve estimates of occupancy diverged more for OLYM than ONF because detection rates were lower on OLYM (Table J.4).

Table J.3. Ranking of models used to examine variation in initial occupancy (ψ_{1992}), extinction (ε), colonization (γ), and detection (*p*) probabilities of barred owls in Olympic National Park and Olympic National Forest, Washington, 1992 – 2013.

Measure calculated	**Model**[1]	**ΔAIC_c**	**K**	**w_i**	**Deviance**
Initial occupancy (ψ_{1992})	ψ(.)	0.00	88	0.98	3491.47
	ψ(area)	8.04	89	0.02	3497.33
Extinction probability (ε)	ε(.)	0.00	47	0.73	3511.13
	ε(area)	1.94	48	0.27	3510.97
	ε(area + T)	21.21	49	0.00	3528.14
	ε(area * T)	22.11	50	0.00	3526.94
	ε(T)	28.78	48	0.00	3537.81
Colonization probability (ϒ)	ϒ(area * T + eleSD)	0.00	10	0.34	3545.07
	ϒ(area * T)	0.20	9	0.31	3547.29
	ϒ(area * T + eleMN)	0.83	10	0.22	3545.90
	ϒ(area + T)	2.99	8	0.08	3552.10
	ϒ(area + TT)	4.93	9	0.03	3552.02
Detection probability (*p*)	*p*(area + T, .)	0.00	89	0.94	3497.33
	p(T, .)	5.36	88	0.06	3504.88
	p(area + t, T)	17.14	103	0.00	3483.63
	p(t, T)	20.10	85	0.00	3526.15
	p(area)	24.74	88	0.00	3524.25

[1] Models were ranked accordig to Akaike's Information Criterion adjusted for small sample size (AIC_c). Model deviance, number of parameters (K) and AICc weights (w_i) are shown for each model. Time effects were modeled as constant (.) varying with years (t), a linear time trend (T), a quadratic time trend (TT), or a pseudo-threshold time trend (lnT). Site-specific covariates include land ownership (area), mean elevation of the site (eleMN), and standard deviation of elevation of the site (eleSD; a measure of terrain roughness).

Table J.4. Estimates of annual detection probabilities ($\hat{p}$) of barred owls in Olympic National Park and Olympic National Forest, Washington, 1992 – 2013. Estimates are from the best AIC$_c$ single-species occupancy model for barred owls that did not include site-specific covariates.

Year	Olympic National Park 95% Confidence Interval				Olympic National Forest 95% Confidence Interval			
	$\hat{p}$	SE	Lower	Upper	$\hat{p}$	SE	Lower	Upper
1992	0.072	0.010	0.055	0.095	0.107	0.016	0.079	0.144
1993	0.077	0.010	0.059	0.100	0.114	0.016	0.086	0.151
1994	0.082	0.010	0.064	0.105	0.121	0.016	0.093	0.157
1995	0.088	0.010	0.070	0.110	0.129	0.016	0.100	0.165
1996	0.093	0.010	0.075	0.115	0.137	0.016	0.108	0.172
1997	0.099	0.010	0.081	0.121	0.145	0.016	0.116	0.180
1998	0.106	0.010	0.088	0.127	0.154	0.016	0.125	0.188
1999	0.112	0.010	0.095	0.133	0.163	0.016	0.135	0.196
2000	0.120	0.010	0.102	0.140	0.173	0.015	0.145	0.205
2001	0.127	0.009	0.110	0.147	0.183	0.015	0.155	0.215
2002	0.135	0.009	0.118	0.154	0.194	0.015	0.166	0.225
2003	0.143	0.009	0.126	0.162	0.205	0.015	0.177	0.235
2004	0.152	0.009	0.134	0.171	0.216	0.015	0.189	0.246
2005	0.161	0.010	0.143	0.181	0.228	0.015	0.201	0.258
2006	0.170	0.010	0.152	0.191	0.240	0.015	0.213	0.270
2007	0.180	0.010	0.161	0.202	0.253	0.015	0.225	0.284
2008	0.191	0.011	0.170	0.214	0.267	0.015	0.237	0.298
2009	0.202	0.012	0.179	0.227	0.280	0.016	0.250	0.313
2010	0.213	0.013	0.188	0.241	0.294	0.017	0.262	0.329
2011	0.225	0.015	0.197	0.256	0.309	0.018	0.274	0.346
2012	0.237	0.017	0.206	0.271	0.324	0.020	0.287	0.363
2013	0.250	0.018	0.216	0.288	0.339	0.021	0.299	0.382

NPS 149/149534, November 2018

National Park Service
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