National Park Service
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Natural Resource Stewardship and Science

North Cascades National Park Service Complex At-Risk Terrestrial Wildlife Species

Focused Condition Assessment 1995–2020

Natural Resource Report NPS/NOCA/NRR—2023/2539





ON THIS PAGE
Peregrine Falcon (*Falco peregrinus*) by the Skagit River, Ross Lake National Recreation Area.
PAUL DEBRUYN / WASHINGTON DEPARTMENT OF FISH AND WILDLIFE

ON THE COVER
Lower Curtis Glacier and Mount Shuksan in North Cascades National Park Service Complex.
JASON RANSOM / NPS.
Inset photos left to right: Fisher (*Pekania pennanti*) JASON RANSOM / NPS, Northern Spotted Owl (*Strix occidentalis caurina*) JASON RANSOM / NPS, Gray Wolf (*Canis lupus*) CASCADES CARNIVORE PROJECT / NPS / WASHINGTON STATE UNIVERSITY.

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Executive Summary

The North Cascades Ecosystem is one of the largest and most intact wilderness areas in the contiguous United States. It spans 34,965 km^2 across the U.S.-Canada border between central Washington State and southern British Columbia and is bisected north to south by the Cascade Mountain range. The North Cascades National Park Service Complex (hereafter, the Park) lies in the heart of the ecosystem and is comprised of North Cascades National Park (2,044 km^2), Ross Lake National Recreation Area (NRA) (473 km^2), and Lake Chelan NRA (251 km^2). The large wild landscape of the Park is bisected east-west by the North Cascades Highway (Washington State Route 20), which is paralleled by a large electric transmission corridor connecting to hydroelectric power stations and dams in the Skagit River watershed at Gorge, Diablo, and Ross reservoirs. Additional infrastructure throughout the Skagit River corridor and Stehekin Valley includes campgrounds, visitor centers, and housing, as well as the Environmental Learning Center, a developed campus adjacent to Diablo reservoir. The trail system in the Park includes 560 km (348 mi) of hiking trails, along with several climbing and mountaineering routes. Over 75 mammal species, >200 bird species, 27 fish species, and 17 reptile and amphibian species share this ecosystem, along with >2600 plant and fungus species.

This report focuses on terrestrial wildlife species in the Park that are federally listed under the Endangered Species Act of 1973 (ESA), state-listed under Washington Administrative Code, and/or designated by NPS as Management Priority species. We provide a detailed synthesis of information around NPS records from 1995–2020 for nine bird and nine mammal species and one mammal Order (Chiroptera [bats]), as well as a suite of invertebrate pollinators in the Park where data are available. Information for each species includes life history information, occurrence in the Park, protective status, trends when known, a summary of known threats, and a summary of conservation and research needs. We also provide brief summaries for an additional five bird and two mammal species, and one taxa group (woodpeckers) where data are more limited.

Terrestrial wildlife species in the Park are potentially susceptible to a variety of threats, ranging from climate change impacts to habitat and behavior (e.g., changes to temperature, precipitation, phenology of resources, competition), atmospheric deposition of nutrients (e.g., nitrogen, phosphorus, sulfur) and pollutants (e.g., ozone, methylmercury, microplastics, other bioaccumulative toxins), disease (e.g., avian influenza, West Nile virus, white-nose syndrome), to direct impacts from human presence (e.g., infrastructure, recreation, direct killing). Many of these threats can affect presence, distribution, and survival of wildlife species and populations in the ecosystem.

Anthropogenic threats to terrestrial wildlife include pollutants and contaminants to air, water, and natural sounds. Water quality is generally good in the Park, but atmospheric deposition contributes a variety of contaminants, as does leaching from old mines and other ground-based sources. Water impoundment and water-level manipulation are also known to influence bioaccumulation pattern, trophic transfer, and health risk of some heavy metals in food webs. Air quality in the Park is also generally good, but deteriorates when westerly winds transport pollutants from developed areas along Puget Sound, as well as with smoke from wildland fires. Noise pollution is variable across the Park,

and anthropogenic noise (typically aircraft, watercraft, and automobiles) registered 11–59.3% of time across three sites in 2015. Such noise was most often in the 35 dBA range but was recorded as high as 52 dBA at two sites. Terrestrial wildlife responses begin at noise levels of approximately 40 dBA, and most often are reported for species that rely on vocal communication, such as birds.

Emergence and transmission of disease is a persistent threat to wildlife species, and in some cases directly relates to human activities and connectivity of wildlife populations. Thousands of known viruses are currently circulated among wild mammals, and climate change is expected to inflate the rate of transmission across species. Wildlife diseases known to occur in the Pacific northwest include, but are not limited to: avian influenza, West Nile virus, rabbit hemorrhagic disease in lagomorphs, hoof disease, bluetongue virus, and chronic-wasting disease in ungulates, parvovirus in canines and mesocarnivores, *Mycoplasma ovipneumoniae* (pneumonia) in wild sheep and goats, white-nose syndrome in bats, and severe acute respiratory syndrome coronavirus 2 (SARS-CoV-2) in multiple taxa but especially acute cases in felines and mustelids. Diseases will likely continue to emerge through time and the effects of their occurrence on populations may be difficult to detect.

Human activities and infrastructure can also directly impact wildlife species in the Park. Infrastructure associated with the Skagit River Project originally inundated 50 km^2 of wildlife habitat in what is now Ross, Diablo, and Gorge reservoirs. The reservoirs now provide nesting habitat for bald eagles and osprey, as well as provide important migratory stopover habitat for waterfowl, such as western grebe. These reservoirs also likely affect microclimates, influence riparian habitat with fluctuating pool levels, contribute to erosion of terrestrial habitat, accumulate heavy metals and microplastics, and may limit the ranges of some terrestrial species within the Park. The operation of the dams also influences downstream hydrology on the Skagit River, with direct implications on salmon production and consequent winter food availability for piscivorous species such as bald eagles. Throughout the Skagit River corridor, hydroelectric infrastructure, NPS infrastructure such as campgrounds and facilities, and the Environmental Learning Center all emit artificial lights which can disrupt migration and travel of birds, including marbled murrelet. Transmission lines and power stations along the Skagit also introduce the threat of bird strike and electrocution.

Recreation is known to affect wildlife in a variety of ways, including impacts on physiology, behavior, and demographics (decreased survival and reproduction) at both the individual and population level. Annual visitation to the Park has increased dramatically in recent years, with an estimated 982,000 visitors in 2020. Most visitors to the Park arrive on developed roads. The impacts to wildlife from developed roadways include potential disruption of wildlife travel, habituation and/or toxicity from road deicers and automobile byproducts, impacts to reproduction from noise, and death from vehicle strike. In summer 2020, traffic counters logged 255,582 vehicles travelling into the Park from the west and 147,460 vehicles travelling into the Park from the east on the North Cascades Highway. Most vehicular strikes to wildlife are not documented, but from 1995–2020, NPS and Washington Department of Transportation documented individuals of at least 16 species killed in the Park. The most commonly detected carcasses were black-tailed or mule deer, likely due to their conspicuous size, but other mammal species detected included black bear, bobcat, coyote, and porcupine. Bird species were also detected dead on the road, including hawks, owls, grouse, and

common raven. Impacts of roads on wildlife in the Park are largely unquantified and represent an important need for future research.

Direct killing of wildlife during European settlement of the North Cascades severely impaired the ability of some species to persist through time. From 1826–1867, The Hudson's Bay Company shipped >50,000 carnivore skins from around the North Cascades Ecosystem, including 15,350 gray wolves, 11,187 Canada lynx, 1,249 wolverine, 8,706 red fox, 12,464 fisher, and 3,188 grizzly bear. This ultimately resulted in extirpation of some species, like fisher and grizzly bear, with other species remaining very rare today. Purposeful direct killing of wildlife in the Park today is limited to licensed hunting of some species (typically deer and black bears) in Ross Lake NRA and Lake Chelan NRA, which includes traditional use by local Tribes. Illegal killing of wildlife (poaching) does occur intermittently. Wildlife species may also be euthanized by Park managers as a last resort during conflict management operations.

The management and conservation of wildlife in the Park is mandated by the NPS Organic Act of 1916, with guidance on protection, recovery, and stewardship of at-risk species provided in NPS Management Policies. Regulatory protections for Park wildlife are established in the Code of Federal Regulations, with authority granted to the Park Superintendent to establish supplemental protections through the Park Superintendent's Compendium. Examples of statutes included in these supplemental protections are the authority to implement short-term public closures for wildlife protection, which may include active nest or den sites, or situations where public safety is of concern (e.g., a large predator with prey on a trail or near a campsite), prohibition of artificial lights for wildlife viewing or hunting, prohibition of trapping, requirements for use of bear-resistant food storage containers, and regulations pertaining to use of pack animals and dogs.

The Park Wildlife Program guides the focus of wildlife monitoring and research conducted by NPS staff and implements management actions. These actions require consultation with the U.S. Fish and Wildlife Service regarding ESA-listed species, and with Washington Department of Fish and Wildlife regarding state-listed species. The mission of the Park Wildlife Program is "protect, conserve, and restore wildlife species and their habitats through an ethic of stewardship that values the well-being of individuals, populations, and communities. The [Park] Wildlife Program will provide scientific leadership, scholarship, and active collaboration with partner agencies, Tribes and First Nations, and non-governmental organizations, toward stewardship of wildlife resources at an ecosystem scale, in a paradigm of continuous change that is not yet fully understood, based on best available science, accurate fidelity to the law, and with consideration of the diverse cultural values of humans that share the North Cascades Ecosystem". Throughout this report, needed conservation actions are presented by species. A reference table of best practices and required mitigations for management activities around wildlife is also presented.

Acknowledgments

Thank you to all of the researchers and resource managers who helped track down missing datapoints and helped to find voucher records. Erin Colclazier at Hamer Environmental, L.P. graciously formatted and provided their marbled murrelet survey data for the Park, as well as provided the elusive photograph of that species nesting. Sincere thanks to the many peer reviewers for providing expertise and insights into improving this report: J. Garnett and T. Frederick (U.S. Fish and Wildlife), J. Lewis and S. Fitkin (Washington Department of Fish and Wildlife), R. Siegel (Institute for Bird Populations), Sophia Rice (Lummi Tribe), and Leslie Parks (Swinomish Indian Tribal Community). Thanks to Rachel Wolstenholme (NPS Pacific West Region) for providing administrative review of this report. Species assessed in this report all live within the North Cascades National Park Service Complex, which includes the traditional lands of the Confederated Tribes of the Colville Reservation, Lummi Tribe, Nlaka'pamux Nation, Nooksack Indian Tribe, Sauk-Suiattle Indian Tribe, Stillaguamish Tribe of Indians, Sto:lo Nation, Swinomish Indian Tribal Community, Upper Skagit Indian Tribe, and Yakama Nation.

Chapter 1. Resource Setting

Introduction

The North Cascades Ecosystem is one of the largest and most intact wilderness areas in the contiguous United States. It spans 34,965 km^2 across the U.S.-Canada border between central Washington State and southern British Columbia and is bisected north to south by the Cascade Mountain range (Figure 1). The North Cascades National Park Service Complex (hereafter, the Park) lies in the heart of the ecosystem and is comprised of North Cascades National Park (2,044 km^2), Ross Lake National Recreation Area (NRA) (473 km^2), and Lake Chelan NRA (251 km^2). Congress established this federal complex in 1968 (82 Stat. 926), to: "…preserve for the benefit, use, and inspiration of present and future generations certain majestic mountain scenery, snowfields, glaciers, alpine meadows, and other unique natural features in the North Cascade Mountains of the State of Washington…" and to "…provide for the public outdoor recreation use and enjoyment … [and] for the conservation of the scenic, scientific, historic, and other values contributing to public enjoyment of such lands and waters" (NPS 2013). Congress subsequently designated 2,568 km^2 of the Park as the Stephen Mather Wilderness in 1988, and the Secretary of Interior designated an additional 14 km^2 as wilderness in 2012.

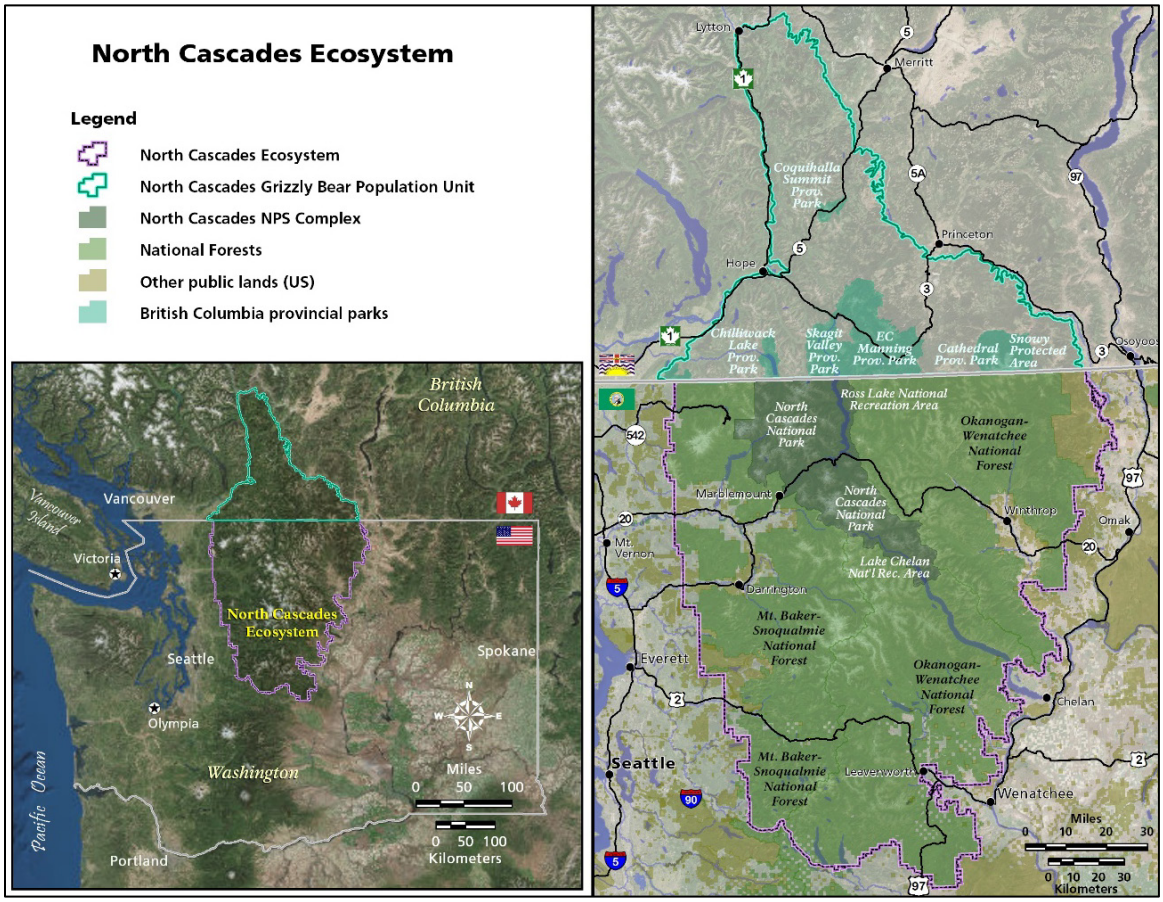


Figure 1. Location of North Cascades National Park Service Complex within the North Cascades Ecosystem.

The large wild landscape of the Park is bisected east-west by the North Cascades Highway (Washington State Route 20), which is closed for roughly five months during winter, from Ross Lake to beyond the east Park boundary. The North Cascades Highway is paralleled by a large electric transmission corridor connecting to hydroelectric power stations and dams in the Skagit River watershed at Gorge, Diablo, and Ross reservoirs (Figure 2). A small stream diversion at Newhalem Creek also provides hydroelectric power. Seattle City Light owns and operates hydroelectric infrastructure known as the Skagit River Project, which collectively totals 77.8 km^2 of area in the Park. These areas include multiple buildings, housing, paved areas, native and non-native vegetation, and visitor services. The National Park Service (NPS) has developed and operates a Visitor Center near Newhalem and developed campgrounds at Newhalem, Goodell Creek (one conventional campground and two campgrounds for groups), Gorge Lake, and Colonial Creek along the North Cascades Highway. The North Cascades Institute operates the Environmental Learning Center, a developed campus adjacent to Diablo reservoir (Figure 2).

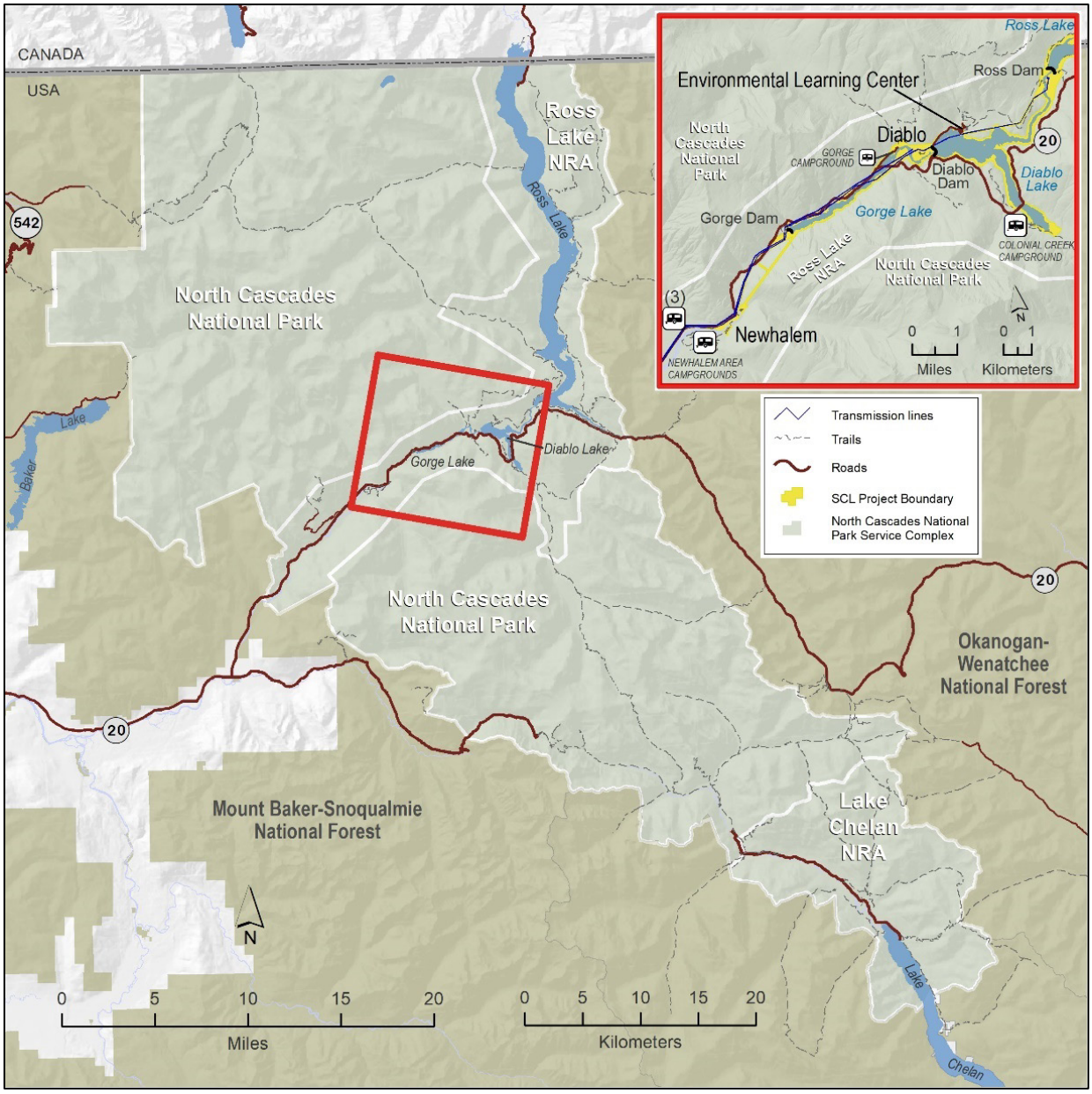


Figure 2. Location of roads, major trails in the North Cascades National Park Service Complex, and location of developed infrastructure along the Skagit River corridor.

The three Skagit reservoirs and Lake Chelan provide hydroelectric power for surrounding communities and Seattle, as well as transportation routes for visitors to access and recreate in the Park. Lake Chelan, the third deepest natural lake in the United States, was dammed in the 1920s, raising the lake level by 6.4 m. The community of Stehekin lies at the north end of Lake Chelan and is accessible only by ferry or hiking. Stehekin is populated by approximately 85 residents who live on private lands along the lower Stehekin River Valley. The NPS maintains and operates a visitor center, staff housing, maintenance facilities, and the historic Buckner Orchard in Stehekin. During summer, the Stehekin Road is open north for 21 km (13 mi) from the ferry landing to its current

terminus at the Pacific Crest Trail (PCT). Due to the limited number of vehicles in Stehekin and no connecting roads, traffic is limited to summer shuttle buses, NPS and resident vehicles, and bicycles.

Most visitors to the Park come for recreation and use the roads, campgrounds, and trails while they pursue hiking, skiing, and climbing. Ross Lake and Lake Chelan NRAs are popular destinations for boating and fishing during summer months, and both NRAs experience limited hunting during applicable seasons. The trail system in the Park is comprised of 560 km (348 mi) of hiking trails, along with several climbing and mountaineering routes. These trails provide the only access into the interior of the Park, and most are limited to use during summer when areas are free of snow. Two National Scenic Trails traverse the Park: the Pacific Northwest Trail (PNT) and the Pacific Crest Trail (PCT). Approximately 99 km (62 mi) of the PNT traverses the Park from the southern end of Copper Ridge to Ross Lake, around Ross Lake, and to the Park boundary with the Okanogan-Wenatchee National Forest. The PCT traverses the Park for roughly 29 km (18 mi), along the upper Stehekin River and Bridge Creek. Each summer, 700–800 PCT hikers cross through the Park, making it one of the busiest trails in the ecosystem.

Biogeography

(adapted from the North Cascades NPS Complex Resource Condition Assessment [Hoffman et al. 2015])

The Park is divided into two major biogeographic zones: the temperate marine west slope of the Cascades and semi-arid continental east slope of the Cascades. A third transitional zone is created in the lower elevations of the Skagit watershed by the Cascade-Pacific, Skagit (west of the Cascade crest), and North Cascade orographic divides, which splits flow north to the Fraser River and south to the Skagit River. This zone is characterized by an intermediate climate between the mild, wet conditions of the west side and the semi-arid conditions of the east side of the Cascades.

Elevation ranges from 25 m in the western lowland maritime forest to peaks exceeding 3,200 m at Mt. Baker and Glacier Peak in the Cascade Range. The mountains are home to over 700 glaciers, which form the headwaters of the Skagit, Nooksack, Snohomish, Stillaguamish, and Skykomish Rivers. The western slope is strongly influenced by a temperate maritime climate, which delivers precipitation averaging roughly 400 cm/yr. This supports extensive mesic lowland and subalpine forests composed predominantly of Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), Pacific silver fir (*Abies amabilis*), and western red cedar (*Thuja plicata*), as well as over 300 species of non-vascular plants such as mosses and lichens. Riparian zones host deciduous forest consisting primarily of alder (*Alnus* spp.), cottonwood (*Populus* spp.), and willow (*Salix* spp.). To the east, the glaciers feed several tributaries of the Columbia River, including the Methow, Stehekin (Lake Chelan), Entiat, Wenatchee, and Cle Elum Rivers. The Park receives average precipitation of 130 cm/yr just east of the Cascade crest, but orographic effects reduce precipitation to an average of only 25 cm/yr on the easternmost edge of the ecosystem (Raymond et al. 2014). Consequently, alpine meadows transition rapidly down the eastern slopes to semi-arid continental forests dominated by Douglas-fir and ponderosa pine (*Pinus ponderosa*) and lowland valleys composed of shrublands and grasslands.

Moving and still water is abundant in the Park, with snow and glacial runoff feeding thousands of waterfalls and streams, which feed large rivers, reservoirs, and lakes. This abundance of water, combined with a diversity of landforms, soil types, range of elevations, slopes, and aspects, create the environment to support a wide variety of habitats and biological communities in the Park. There are >75 mammal species, >200 bird species, 27 fish species, and 17 reptile and amphibian species in the Park, along with >2600 plant and fungus species. Status and trend for many of these species is unknown in the Park, but long-term monitoring of birds indicates that during breeding season surveys from 2005–2014, nearly all of 39 modeled species were found to exhibit stable to increasing populations (Ray et al. 2017). Bird species that were estimated as declining included mountain chickadee (*Poecile gambeli*), olive-sided flycatcher (*Contopus cooperi*), northern flicker (*Colaptes auratus*), and dark-eyed junco (*Junco hyemalis*). Raptor monitoring in the Park has historically focused on species such as peregrine falcon (*Falco peregrinus*), bald eagle (*Haliaeetus leucocephalus*), and northern spotted owl (*Strix occidentalis caurina*) which are detailed later in this report. Likewise, mammal monitoring has focused largely on at-risk species such as Canada lynx (*Lynx canadensis*), gray wolf (*Canis lupus*), wolverine (*Gulo gulo*), fisher (*Pekania pennanti*), hoary marmot (*Marmota caligata*), and American pika (*Ochotona princeps*); all of which are detailed later in this report.

The Skagit River system hosts all five native anadromous salmon species and is one of the few watersheds within the Puget Sound Basin that is managed for natural production of salmon. Carcasses of spawning salmon are an important resource for bald eagles (Rubenstein et al. 2019), and other species such as black bear (*Ursus americanus*), coyote (*Canis latrans*), and bobcat (*Lynx rufus*) have been observed feeding on these resources in the Park. All of the mountain lakes in the Park were historically devoid of fish, but many now support introduced populations of rainbow trout (*Oncorhynchus mykiss*), cutthroat trout (*Oncorhynchus clarkii*), Eastern brook trout (*Salvelinus fontinalis*), and golden trout (*Oncorhynchus aquabonita*).

Scope and Methodology

This report focuses on terrestrial wildlife species in the Park that are federally-listed under the Endangered Species Act of 1973 (ESA) (P.L. 93-205, as amended), state-listed under Washington Administrative Code 220-610-110, and/or designated by NPS as Management Priority species. The Management Priority designation includes federally- and state-designated species, as well as species that may be exhibiting apparent local declines, species facing overt threats from emerging issues (e.g., climate change or disease), and species that require focused management attention due to human activities. In this report, we do not assess species that have been designated as Management Priority species solely due to management needs around human activities or values and whose populations are not considered at-risk (e.g., black bears). We also do not assess mountain goats (*Oreamnos americanus*), which are a Management Priority species. A separate report on mountain goats will follow comprehensive surveys planned for 2022–2023.

This report provides a detailed synthesis of information around NPS records from 1995–2020 for nine bird and nine mammal species and one mammal Order (Chiroptera [bats]), as well as a suite of invertebrate pollinators in the Park, for which the Park has sufficient data. We also provide brief

summaries for an additional five bird and two mammal species, and one taxa group (woodpeckers) where data are limited. There are currently no at-risk reptile species in the Park. We also do not assess amphibians here and defer those taxa to assessments of aquatic organisms. We include all records for birds, mammals, and invertebrate pollinators that have been evaluated as *Probable* or *Confirmed* by NPS. *Probable* records include observations by NPS staff, professionals, and public of a species where 1) a detailed description is provided matching that species, 2) the animal is in appropriate habitat for that species, and 3) physical evidence provided is suggestive but not definitive (such as a photograph of tracks whose identity can't be unequivocally confirmed). *Confirmed* records include observations by NPS staff, professionals, and the public that are supported by voucher evidence, which may include: definitive photographs, genetic samples, a carcass, radio-telemetry movements, or acoustic/sonogram recordings. In some special cases, records have been coded as *Confirmed* without a voucher, if the record was provided by a professional with expertise in the species (e.g., a researcher may have captured a species and released it without a photo, but the identification was incontrovertible). All maps in this report distinguish *Probable* vs. *Confirmed* locations, noting that when both occurred in the same area, *Confirmed* status is displayed.

While maps are provided for general location of survey efforts and detections in the Park, precise locations are not disclosed. Instead, we depict locations as 10 km^2 cells that include any locations of a species within that area. Under Title 54, U.S.C. § 100707, NPS protects the locations of at-risk species found within NPS-managed lands in order to prevent harassment or death of these precious resources. Additional laws and policies around treatment and/or protection of at-risk species can be found in the NPS Management Policies (2006) and the Park Superintendent's Compendium (NPS 2023).

Terrestrial At-Risk Wildlife Species

There are 20 bird, 13 mammal, and two invertebrate pollinator species, as well as one mammalian Order (Chiroptera: bats) that are currently considered at-risk species in the Park and meet the criteria for this assessment (Table 1). Of the bird species, 17 of 20 have *Confirmed* records in the Park during the assessment period of this report (Table 2). The three species without *Confirmed* records include two vagrant (outside of their normal range) species: American white pelican (*Pelecanus erythrorhynchos*) and ferruginous hawk (*Buteo regalis*), and the latter was a single record that occurred before this assessment period. The third bird species without a *Confirmed* record is marbled murrelet (*Brachyramphus marmoratus*), which has been putatively detected by their diagnostic patterns of flight during radar surveys. The only visual record of marbled murrelets during this assessment was not accompanied by a voucher photograph, so the highest level of record status remains as *Probable*. Detailed information about marbled murrelets can be found in later in this report.

Table 1. Terrestrial at-risk wildlife species occurring in North Cascades National Park Service Complex. All species shown have been designated by the National Park Service as Management Priority species. State and federal status is shown for species that are also listed under Washington Administrative Code 220-610-110 (State) or the Endangered Species Act of 1973 (Federal). The Washington State Wildlife Action Plan also designates Species of Greatest Conservation Need (SGCN) (WDFW 2015).

Taxa Group	Common Name	Scientific Name	State	Federal
Bird	American three-toed woodpecker	*Picoides dorsalis*	–	–
Bird	American white pelican	*Pelecanus erythrorhynchos*	Sensitive, SGCN	–
Bird	Bald eagle	*Haliaeetus leucocephalus*	SGCN	Delisted
Bird	Black-backed woodpecker	*Picoides arcticus*	Candidate	–
Bird	Clark's Grebe	*Aechmophorus clarkii*	Candidate, SGCN	–
Bird	Clark's nutcracker	*Nucifraga columbiana*	–	–
Bird	Common loon	*Gavia immer*	Sensitive, SGCN	–
Bird	Ferruginous hawk	*Buteo regalis*	Endangered	–
Bird	Golden eagle	*Aquila chrysaetos*	Candidate, SGCN	–
Bird	Harlequin duck	*Histrionicus histrionicus*	SGCN	–
Bird	Marbled murrelet	*Brachyramphus marmoratus*	Endangered, SGCN	Threatened
Bird	Mount Rainier white-tailed ptarmigan	*Lagopus leucura rainierensis*	SGCN	Proposed Threatened
Bird	Mountain chickadee	*Poecile gambeli*	–	–
Bird	Northern goshawk	*Accipiter gentilis*	Candidate	–
Bird	Northern spotted owl	*Strix occidentalis caurina*	Endangered, SGCN	Threatened
Bird	Olive-sided Flycatcher	*Contopus cooperi*	–	–
Bird	Osprey	*Pandion haliaetus*	–	–
Bird	Peregrine falcon	*Falco peregrinus*	SGCN	Delisted
Bird	Pileated woodpecker	*Dryocopus pileatus*	–	–
Bird	Sandhill crane	*Grus canadensis tabida*	Endangered, SGCN	–
Bird	Western grebe	*Aechmophorus occidentalis*	Candidate, SGCN	–
Bird	White-headed woodpecker	*Picoides albolarvatus*	Candidate, SGCN	–
Insect	Propertius duskywing	*Erynnis persius*	SGCN	–
Insect	Western bumble bee	*Bombus occidentalis*	SGCN	In Review
Mammal	American pika	*Ochotona princeps*	SGCN	–

Table 1 (continued). Terrestrial at-risk wildlife species occurring in North Cascades National Park Service Complex. All species shown have been designated by the National Park Service as Management Priority species. State and federal status is shown for species that are also listed under Washington Administrative Code 220-610-110 (State) or the Endangered Species Act of 1973 (Federal). The Washington State Wildlife Action Plan also designates Species of Greatest Conservation Need (SGCN) (WDFW 2015).

Taxa Group	Common Name	Scientific Name	State	Federal
Mammal	Canada lynx	*Lynx canadensis*	Endangered, SGCN	Threatened
Mammal	Cascade red fox	*Vulpes vulpes cascadensis*	Endangered, SGCN	–
Mammal	Fisher	*Pekania pennanti*	Endangered, SGCN	–
Mammal	Gray wolf	*Canis lupus*	Endangered, SGCN	Endangered
Mammal	Grizzly bear	*Ursus arctos horribilis*	Endangered	Threatened
Mammal	Hoary Marmot	*Marmota caligata*	–	–
Mammal	Keen's myotis	*Myotis keenii*	Candidate, SGCN	–
Mammal	Myotis spp.	*Myotis* spp.	–	–
Mammal	Northern bog lemming	*Synaptomys borealis sphagnicola*	SGCN	In Review
Mammal	Townsend's big-eared bat	*Corynorhinus townsendii townsendii*	Candidate, SGCN	–
Mammal	Western gray squirrel	*Sciurus griseus*	Threatened, SGCN	–
Mammal	Western spotted skunk	*Spilogale gracilis*	SGCN	–
Mammal	Wolverine	*Gulo gulo*	Candidate, SGCN	Proposed Threatened

Table 2. Occurrence, trend, and nature of records for at-risk terrestrial wildlife species occurring in North Cascades National Park Service Complex 1995–2020. Note: “most recent Park record” date for some species is 2021 due to timing and interval of surveys.

Common Name	Occurrence	Type of Occurrence	Trend in Park	Confirmed Reproduction	Voucher Type	Most Recent Park Record	Number of Park Records
American three-toed woodpecker	Present	Resident	Unknown	Yes	Photograph	2018 (Confirmed)	47
American white pelican	Incidental	Vagrant	NA	No	None	2004 (Probable) [d]	1
Bald eagle	Present	Resident	Stable	Yes	Photograph, Genetic	2021 (Confirmed)	4425
Black-backed woodpecker	Present	Resident	Unknown	No	Photograph	2018 (Confirmed)	34
Clark's grebe	Incidental	Vagrant	NA	No	Photograph	2017 (Confirmed)	2
Clark's nutcracker	Present	Migratory	Stable to Declining	No	Photograph, Genetic, Telemetry	2021 (Confirmed)	196 [c]
Common loon	Present	Resident/ Migratory	Unknown	Yes	Photograph, Genetic	2020 (Confirmed)	100
Golden eagle	Present	Migratory	Unknown	No	Photograph	2019 (Confirmed)	35
Harlequin duck	Present	Migratory	Stable	Yes	Photograph, Genetic	2021 (Confirmed)	306
Marbled murrelet	Present	Migratory	Unknown	No	Radar	2021 (Probable)	64
Mount Rainier white-tailed ptarmigan	Present	Resident	Unknown	Yes	Photograph	2020 (Confirmed)	32
Mountain chickadee	Present	Resident	Declining	No	Photograph	2019 (Confirmed)	46
Northern goshawk	Present	Resident	Unknown	Yes	Photograph	2020 (Confirmed)	63
Northern spotted owl	Present	Resident	Declining	Yes	Photograph	2019 (Confirmed)	27
Olive-sided flycatcher	Present	Migratory	Declining	No	Sonogram	2020 (Confirmed)	74
Osprey	Present	Migratory	Stable	Yes	Photograph	2021 (Confirmed)	176

[a] Photographed near Park boundary.

[b] Genetic analysis required to identify this species.

[c] Does not include radio-telemetry data from individuals documented in the Park.

[d] Not expected in the Park due to range/habitat.

Table 2 (continued). Occurrence, trend, and nature of records for at-risk terrestrial wildlife species occurring in North Cascades National Park Service Complex 1995–2020. Note: “most recent Park record” date for some species is 2021 due to timing and interval of surveys.

Common Name	Occurrence	Type of Occurrence	Trend in Park	Confirmed Reproduction	Voucher Type	Most Recent Park Record	Number of Park Records
Peregrine falcon	Present	Migratory	Stable	Yes	Photograph, Genetic	2021 (Confirmed)	82
Pileated woodpecker	Present	Resident	Stable	Yes	Photograph	2020 (Confirmed)	211
Sandhill crane	Incidental	Vagrant	NA	No	None	2008 (Confirmed) [d]	1
Western grebe	Present	Migratory	Declining	No	Photograph	2019 (Confirmed)	17
White-headed woodpecker	Present	Resident	Unknown	No	Photograph	2015 (Confirmed	3
Propertius duskywing	Present	Resident	Unknown	No	Photograph	2020 (Confirmed)	1
Western bumble bee	Present	Resident	Declining	No	Photograph	2016 (Confirmed)	>3
American pika	Present	Resident	Stable	Yes	Photograph	2021 (Confirmed)	886
Canada lynx	Present	Resident	Unknown	No	Photograph, Genetic, Telemetry	2020 (Confirmed)	130 [c]
Cascade red fox	Unknown	Unknown	Functionally Extirpated	No	Photograph [a]	2001 (Confirmed)	3
Fisher	Present	Resident	Increasing	No	Photograph, Genetic, Telemetry	2021 (Confirmed)	76
Gray wolf	Present	Resident	Increasing	Yes	Photograph, Genetic, Telemetry	2021 (Confirmed)	175 [c]
Grizzly bear	Unknown	Unknown	Functionally Extirpated	No	Photograph [a]	1990 (Confirmed)	0
Hoary marmot	Present	Resident	Declining	Yes	Photograph	2021 (Confirmed)	601

[a] Photographed near Park boundary.

[b] Genetic analysis required to identify this species.

[c] Does not include radio-telemetry data from individuals documented in the Park.

[d] Not expected in the Park due to range/habitat.

Table 2 (continued). Occurrence, trend, and nature of records for at-risk terrestrial wildlife species occurring in North Cascades National Park Service Complex 1995–2020. Note: “most recent Park record” date for some species is 2021 due to timing and interval of surveys.

Common Name	Occurrence	Type of Occurrence	Trend in Park	Confirmed Reproduction	Voucher Type	Most Recent Park Record	Number of Park Records
Keen's myotis	Unknown	Unknown	Unknown	No	None	None [b]	0
Myotis spp.	Present	Resident, Migratory	Unknown	Yes	Photograph, Acoustic, Genetic	2021 (Confirmed)	502
Northern bog lemming	Probably Present	Resident	Unknown	No	Specimen	1982 (Confirmed)	0
Townsend's big-eared bat	Present	Resident	Unknown	No	Photograph, Acoustic	2001 (Confirmed)	2
Western gray squirrel	Present	Resident	Stable	Yes	Photograph, Genetic, Telemetry	2021 (Confirmed)	1729
Western spotted skunk	Present	Resident	Declining	No	Photograph	2017 (Confirmed)	356
Wolverine	Present	Resident	Increasing	Yes	Photograph, Genetic, Telemetry	2021 (Confirmed)	61 [c]

[a] Photographed near Park boundary.

[b] Genetic analysis required to identify this species.

[c] Does not include radio-telemetry data from individuals documented in the Park.

[d] Not expected in the Park due to range/habitat.

There are two invertebrate pollinator Management Priority species confirmed as present in the Park: Propertius duskywing (*Erynnis persius*) and western bumblebee (*Bombus occidentalis*), though both have only rarely been observed (see Chapter 3). Of 13 mammal species, grizzly bear (*Ursus arctos horribilis*), Cascade red fox (*Vulpes vulpes cascadensis*), northern bog lemming (*Synaptomys borealis sphagnicola*), and Keen's myotis (*Myotis keenii*) were not *Confirmed* in the Park during this assessment period (Table 2). Grizzly bear was confirmed adjacent to the Park as recently as 2015, but both grizzly bear and Cascade red fox are considered functionally extirpated at the present time. Functionally extirpated for grizzly bears is defined as not meeting the definition of a population: "two or more reproductive females or one female reproducing during two separate years" (USFWS 2022a). Likewise, with only a single detection of a Cascade red fox, it is possible a small number of individuals may persist, but no reproduction has been detected. Northern bog lemming is unlikely to be casually detected and status is currently unknown (see Chapter 5). Keen's myotis can be classified only by genetic analysis, and status based on existing survey data is insufficient to determine presence. Of note, recent taxonomy of Keen's myotis indicates that this species may not be separate from western long-eared myotis (*Myotis evotis*) (Lausen et al. 2019), though official classification under Washington State administrative code currently retains Keen's myotis as a separate species (WDFW 2023).

Abundance and trend for populations of Management Priority species is difficult to estimate, especially for rare species. Extensive annual surveys of landbirds (Ray et al. 2017, 2018) resulted in 5,170 point counts of 57,767 individual birds (131 species) from 2005–2020. From those data, trend for 39 species could be robustly modeled for the period 2005–2014, and resulted in quantitative estimates that informed stable, increasing, or declining trends for some species (Table 2). Likewise, nesting surveys for bald eagle, peregrine falcon, and common loon generated data to help inform general trend based on annual detections and nest success (Table 2) (Christophersen and Ransom 2022). Focused surveys on harlequin duck also gave rise to data that could inform general trend (Rine et al. 2022). For most other bird species, trend is unknown.

Studies focused on some individual mammal species, such as American pika and hoary marmot, have provided data to estimate trend. Other species such as Canada lynx, fisher, gray wolf and wolverine have been surveyed, but generally occur at such low densities over a large geographic area that estimating abundance or trend is difficult. Detections of these four species, however, have increased dramatically over the last decade (Figure 3). Some of this distribution is certainly attributed to increased effort and improvements in monitoring technology, but some is attributed to an increased presence of these species. Fishers, for example, were extirpated in the Cascades until reintroduction efforts released 89 individuals onto the landscape during 2018–2020 (Lewis et al. 2022). Similarly, gray wolves in Washington have been increasing in abundance at an average rate of about 25% per year and their range is expanding annually (WDFW et al. 2022). The first confirmed reproduction (presence of young) of gray wolves in the Park was detected in 2020.

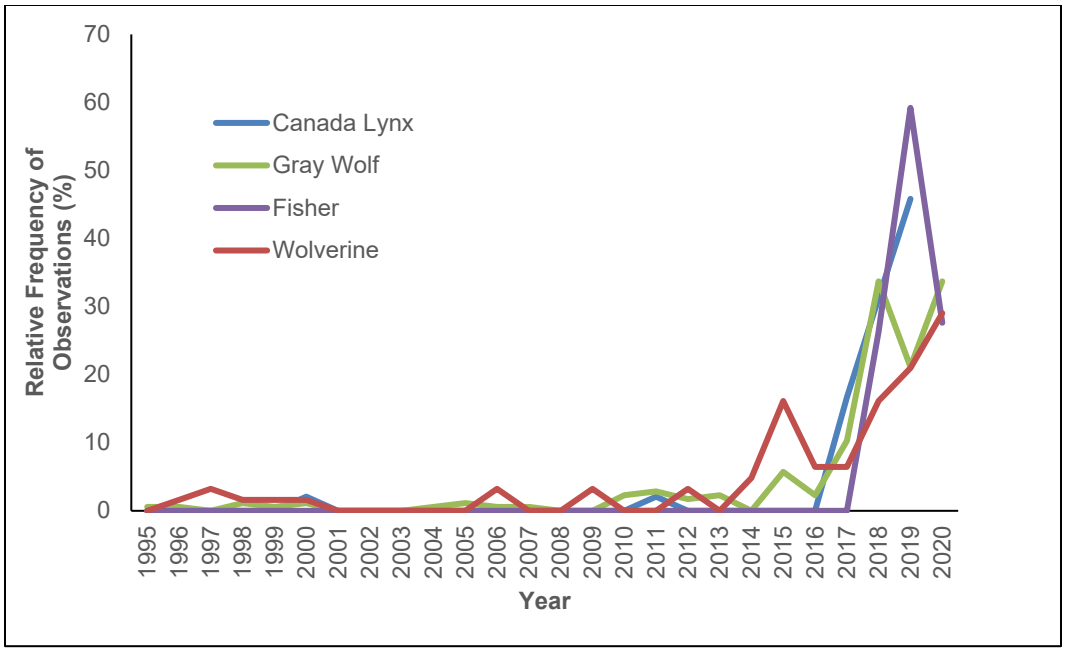


Figure 3. Relative frequency of total confirmed and probable detections of Canada lynx (*Lynx canadensis*), gray wolf (*Canis lupus*), fisher (*Pekania pennanti*), and wolverine (*Gulo gulo*) in North Cascades National Park Service Complex, 1995–2020.

Detections of some species, such as western spotted skunk (*Spilogale gracilis*), have declined to near zero across survey efforts. Western spotted skunk was the most frequently detected mammal during 2003–2004 forest carnivore surveys (Christophersen et al. 2005), after which they were detected only once each year in 2014, 2016, and 2017. Concurrently, the first striped skunk (*Mephitis mephitis*) record in the Park occurred in 2016 near Stehekin, and since then striped skunks have been detected 49 times in the western and southern portions of the Park, ranging from Hozomeen to Stehekin, as well Big Beaver Valley. The combined nature of decreasing western spotted skunk detections and increasing potential competition has led us to suspect that western spotted skunks are in decline. For many species in the Park, trend is simply unknown; however, detailed descriptions of occurrence are provided for each species throughout this report.

Resource Threats Overview

Terrestrial wildlife species in the Park are potentially susceptible to a variety of threats, ranging from climate change impacts on habitat and behavior (e.g., changes to temperature, precipitation, phenology of resources, competition), atmospheric deposition of nutrients (e.g., nitrogen, phosphorus, sulfur) and pollutants (e.g., ozone, methylmercury, microplastics, other bioaccumulative toxins), disease (e.g., avian influenza, West Nile virus, white-nose syndrome), and direct impacts from human presence (e.g., infrastructure, recreation, direct killing). Many of these threats can affect presence, distribution, and survival of wildlife species and populations in the ecosystem.

The biota of the North Cascades is believed to have changed very little over the last several millennia (Mierendorf 1986), and climate change represents one the largest sources of uncertainty for contemporary wildlife species in the Park (see Bellard et al. 2012, Groffman et al. 2014). Wildlife

species are directly subject to changes in the plant communities on which they rely, and plant communities are directly affected by climate (Stephenson 1990, Churkina and Running 1998, Nemani et al. 2003). Most climate change projections for the Park over the next century suggest it is likely to experience warming temperatures, drier summer months and wetter winter and spring months, decreased snowpack, and an increased number of disturbance events (Raymond et al. 2014). Geologic disturbances (e.g., landslides, floods, snow avalanches) could reorganize landforms and alter availability of habitat and some food resources for wildlife (see Rubenstein et al. 2019, for example). As glaciers disappear, freshwater influx into watersheds may decline and influence associated riparian plant communities (Hoffman et al. 2015). Ultimately, vulnerability of wildlife species to climate change is a product of their exposure (i.e., how much the climate will change where a species lives), sensitivity (i.e., how much change a species can endure before responding), and adaptive capacity (i.e., the ability of a species to change to reduce its exposure or sensitivity) (Lawler et al. 2014).

Across their range, some wildlife species may shift their range or abundance while others may be impacted by changing phenology and complex trophic interactions as climate changes through time (Gilman et al. 2010, Thackeray et al. 2016, Pacifici et al. 2017). Changing climate may result in altered food availability for some species, as well as changes to interspecific relationships, predator-prey dynamics and competition, disease and parasite prevalence and distribution, and invasive species colonization (Kareiva et al. 1993, Schneider et al. 2002, Lawler et al. 2014).

Anthropogenic sources of threats to terrestrial wildlife include pollutants and contaminants to air, water, and natural sounds. Air quality in the Park is generally good, with the national park being classified as a Class I airshed and the NRAs being classified as Class II airsheds. Class I airsheds are afforded the highest level of protection under the Clean Air Act of 1963 (42 U.S.C. § 7401), and Class II airsheds generally represent parts of National Forest System lands that are not classified as Class I and may receive greater amounts of anthropogenic pollution (Hoffman et al. 2015). Two major factors contribute to deterioration of air quality in the Park: the prevailing westerly winds that cross large urban-industrial areas of the Puget Sound lowlands, and dense smoke from wildland fires throughout the Park and adjacent national forests (Hoffman et al. 2015). How air quality specifically impacts wildlife species in the Park is unknown.

Water quality is generally good in the Park, but atmospheric deposition contributes a variety of contaminants, as does leaching from old mines and other ground-based sources. Water impoundment and water-level manipulation are also known to influence bioaccumulation pattern, trophic transfer, and health risk of some heavy metals in food webs (Sang et al. 2019). NPS staff monitor water quality in the Park in on-going scientific studies, but one monitoring effort focuses on insects: dragonfly (multiple species) larvae are sampled annually in two lakes, Thunder Lake and Pyramid Lake, to quantify the amount of mercury present within animal taxa. These data help inform an Integrated Impairment Index for the Park, which classifies the potential health risks for fish, wildlife, and humans in a more community-based measure than would be provided by data from a single species (Eagles-Smith et al. 2020). In 2020, the Park geometric mean mercury concentration was estimated as 94.5 ± 6.7 ng/g dry weight (NPS 2020). That year, Thunder Lake was classified as “low

risk" and Pyramid Lake was classified as "moderate risk" on the Integrated Impairment Index. Notably, necropsies of some piscivorous birds found dead in the Park have detected a variety of heavy metals, as well as at least one polychlorinated biphenyl (PCB). In the U.S., PCBs were commercially manufactured from 1929 until production was banned by the Toxic Substances Control Act of 1976 (P.L. 94-469), but PCBs can persist in the environment. As many wildlife species are higher trophic consumers of aquatic species, bioaccumulation of contaminants and toxins is of concern. Analyses of tissues from bull trout (*Salvelinus confluentus*) and rainbow trout in Ross Lake in 2007 and 2012 found that PCBs, 4,4'-Dichlorodiphenyldichloroethylene (DDE: a common break-down product of the pesticide dichlorodiphenyltrichloroethane [DDT]), polybrominated diphenyl ethers (PBDEs: a class of organobromine compounds used as flame retardants), and polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs: long-lived polyhalogenated organic compounds often produced by chloride-related manufacturing) were present at low levels, as were concentrations of chromium, copper, selenium, and zinc (Seiders and Deligeannis 2018).

Anthropogenic noise is a growing threat to wildlife, and the NPS established a Natural Sounds Program in 2000 to help protect natural resources and visitor experience in parks. Extrinsic noise levels at three sites in the Park were monitored in 2015, including Ruby Arm, Boundary Camp, and Cascade Pass (Brady 2017). Noise pollution was variable across the Park, and anthropogenic noise (typically aircraft, watercraft, and automobiles) was detected at all sites. Such noise was most often in the 35 dBA range, but was recorded as high as 52 dBA at two sites. Of note, extrinsic sounds were audible 59.3% of the time at Ruby Arm (due primarily to vehicle and watercraft noise), 7.6% of time at Cascade Pass (due to aircraft), and 11% of the time at Boundary Camp (due to aircraft). Terrestrial wildlife responses have been documented beginning at noise levels of approximately 40 dBA, and most often are reported for species that rely on vocal communication, such as birds (Shannon et al. 2016). Such impacts from anthropogenic noise range from altered individual behavior to changes in entire ecological communities, and have been shown to impact birds, mammals, fish, reptiles, amphibians and invertebrates (Shannon et al. 2016).

Emergence and transmission of disease is a persistent threat to some wildlife species, and in some cases directly relates to human activities and connectivity of wildlife populations. Thousands of known viruses are currently circulated among wild mammals, and climate change is expected to inflate the rate of transmission across species (Carlson et al. 2022). Predominant wildlife diseases known to occur in the Pacific northwest include avian influenza, which has periodic global resurgences related to domestic poultry production (Shriner et al. 2016) and can spread rapidly among gregarious birds such as waterfowl, and then to raptors (Hénaux and Samuel 2011). During the 2022 outbreak of avian influenza, the U.S. Department of Agriculture reported over 58 million detections of the virus in domestic birds, as well as in 5,900 wild birds from 49 states (Sheridan et al. 2023). Avian influenza has also been documented in mammals (often fatal), with 110 cases reported in 15 species during 2022 (Sheridan et al. 2023). The first cases of avian influenza infecting grizzly bears also occurred in 2022 (Hall 2023). West Nile virus also occurs in birds in the Pacific northwest and has been credited with large-scale population declines in several bird species across their range (LaDeau, et al. 2007). Mammals in the region are susceptible to a variety of viruses and diseases that can spread rapidly and often cause fatalities, such as rabbit hemorrhagic disease (GI.2/RHDV2/b

virus) in lagomorphs (Lankton, et al. 2021), hoof disease (Han and Mansfield 2014), bluetongue virus (Maclachlan 2011), and chronic-wasting disease (Williams 2005) in ungulates, *Mycoplasma ovipneumoniae* (pneumonia) in wild sheep and goats (Dassanayake et al. 2010), parvovirus in canines and mesocarnivores (Steinel et al., 2001), and severe acute respiratory syndrome coronavirus 2 (SARS-CoV-2) in multiple taxa but especially acute cases in felines and mustelids (Gryseels et al. 2021). White-nose syndrome is specific to bats and is known to be spread from bat-to-bat contact as well as from contamination of shoes when humans enter hibernacula. Diseases will likely continue to emerge through time and the effects of their occurrence on populations may be difficult to detect. For example, the first amdoparvovirus specific to skunks was recently detected in striped skunks near Vancouver, British Columbia (Canuti et al. 2017), and it is unknown how that parvovirus may be affecting sympatric western spotted skunk populations.

Human activities and infrastructure can also directly impact wildlife species in the Park. Infrastructure associated with the Skagit River Project originally inundated 50 km^2 of wildlife habitat in what is now Ross, Diablo, and Gorge reservoirs (Jarvis 1992). This included 26 km^2 of upland forest habitat, 20 km^2 of riparian habitat, as well as wetland, shrub, and old growth forest habitat. The reservoirs now provide nesting habitat for bald eagles, and osprey (*Pandion haliaetus*), as well as important migratory stopover habitat for waterfowl, such as western grebe (*Aechmophorus occidentalis*). These reservoirs also likely affect microclimates, influence riparian habitat with fluctuating pool levels, contribute to erosion of terrestrial habitat, accumulate heavy metals and microplastics, and may limit the ranges of some terrestrial species within the Park. The operation of the dams also influences downstream hydrology on the Skagit River, with direct implications on salmon production and consequent winter food availability for piscivorous species such as bald eagles. Throughout the Skagit River corridor, hydroelectric infrastructure, NPS infrastructure such as campgrounds and facilities, and the Environmental Learning Center all emit artificial lights which can disrupt migration and travel of birds, including marbled murrelet. Transmission lines and power stations along the Skagit also introduce the threat of bird strike and electrocution.

Recreation is known to affect wildlife in a variety of ways, including impacts on physiology, behavior, and demographics (decreased survival and reproduction) at both the individual and population level (Dertien et al. 2021). Annual visitation to the Park has increased dramatically in recent years, with an estimated 982,000 visitors in 2020 (NPS 2022). During that year, 71,730 overnight stays were recorded in frontcountry campgrounds and concession lodging, and 45,325 overnight stays were recorded for backcountry camping (NPS 2022). While much of this recreation involved hiking on developed trails in a somewhat predictable pattern for wildlife to encounter, some deviations of use can have profound effects from even single disturbances. In 2018, for example, illegal climbing activity resulted in a route being bolted across the face of a peregrine falcon eyrie and the birds have not been observed nesting at that site again. Frontcountry impacts to wildlife can also be profound, with improper food storage at campsites leading to habituation and food conditioning of bears and other wildlife, sometimes resulting in their death. The extent and nature of recreational impacts to wildlife in the Park are largely unquantified and represent an important need for future research.

Most visitors to the Park arrive on developed roads. The impacts to wildlife from developed roadways include potential disruption of wildlife travel routes (Forman and Alexander 1998, Baker and Leberg 2018), habituation and/or toxicity from road deicers and automobile byproducts (Siegel 2007, Beckmann et al. 2010), impacts to reproduction from noise (Shannon et al. 2016), and death from vehicle strike (Garriga et al. 2012). In summer 2020, traffic counters logged 255,582 vehicles travelling into the Park from the west and 147,460 vehicles travelling into the Park from the east on the North Cascades Highway (NPS 2022). One small access road, Cascade River Road, also leads from the North Cascades Highway near Marblemount, WA, to the popular Cascade Pass trailhead. During summer 2020, traffic counters on that road logged 8,174 vehicles (NPS 2022). Most vehicular strikes to wildlife are not documented, but from 1995–2020, NPS and Washington Department of Transportation documented individuals of at least 16 wildlife species killed in the Park. The most commonly reported carcasses were black-tailed or mule deer (*Odocoileus hemionus*), likely due to their conspicuous size, but other mammal species detected included black bear, bobcat, coyote, and porcupine (*Erethizon dorsatum*). Bird species were also reported dead on the road, including hawks, owls, grouse, and common raven (*Corvus corax*). Impacts of roads on wildlife in the Park are largely unquantified and represent an important need for future research.

Direct killing of wildlife during European settlement of the North Cascades severely impaired the ability of some species to persist through time. From 1826–1867, The Hudson's Bay Company shipped >50,000 carnivore skins from around the North Cascades Ecosystem; including 15,350 gray wolves, 11,187 Canada lynx, 1,249 wolverine, 8,706 red fox, 12,464 fisher, and 3,188 grizzly bear (Rine et al. 2018). This ultimately resulted in extirpation of some species, like fisher and grizzly bear, and other species remain very rare. Purposeful direct killing of wildlife in the Park today is limited to licensed hunting of some species (typically deer) in Ross Lake NRA and Lake Chelan NRA, which includes traditional use by local Tribes. Trapping is prohibited in the Park (NPS 2023). Illegal killing of wildlife (poaching) does occur intermittently. Wildlife species may also be euthanized by Park managers as a last resort during conflict management operations, and some species may be killed during research projects (typically limited to insects).

Resource Stewardship

The management and conservation of wildlife in the Park is mandated by the NPS Organic Act of 1916 (P.L. 64-235), with guidance on protection, recovery, and stewardship of at-risk species provided in NPS Management Policies (NPS 2006), and the Park Superintendent's Compendium (NPS 2023). NPS (2006), Section 4.4.2.3 provides specific guidance in management of threatened and endangered plants and animals on NPS-managed lands:

"The Service will survey for, protect, and strive to recover all species native to national park system units that are listed under the Endangered Species Act. The Service will fully meet its obligations under the NPS Organic Act and the Endangered Species Act to both proactively conserve listed species and prevent detrimental effects on these species. To meet these obligations, the Service will

- cooperate with both the U.S. Fish and Wildlife Service and the NOAA Fisheries to ensure that NPS actions comply with both the written requirements and the spirit of the Endangered Species Act. This cooperation should include the full range of activities associated with the

Endangered Species Act, including consultation, conferencing, informal discussions, and securing all necessary scientific and/or recovery permits;

- undertake active management programs to inventory, monitor, restore, and maintain listed species' habitats; control detrimental nonnative species; manage detrimental visitor access; and reestablish extirpated populations as necessary to maintain the species and the habitats upon which they depend;
- manage designated critical habitat, essential habitat, and recovery areas to maintain and enhance their value for the recovery of threatened and endangered species;
- cooperate with other agencies to ensure that the delineation of critical habitat, essential habitat, and/or recovery areas on park-managed lands provides needed conservation benefits to the total recovery efforts being conducted by all the participating agencies;
- participate in the recovery planning process, including the provision of members on recovery teams and recovery implementation teams where appropriate;
- cooperate with other agencies, states, and private entities to promote candidate conservation agreements aimed at precluding the need to list species; and
- conduct actions and allocate funding to address endangered, threatened, proposed, and candidate species."

NPS Management Policies (NPS 2006) Section 4.9 also specifies that "The Service will take action to prevent or minimize all noise that through frequency, magnitude, or duration adversely affects the natural soundscape or other park resources or values, or that exceeds levels that have been identified through monitoring as being acceptable to or appropriate for visitor uses at the sites being monitored." To this end, NPS maintains a Natural Sounds monitoring program in the Park and sound disturbance thresholds have been established to minimize disturbance to at-risk species (Appendix A).

Multiple other Park plans include aspects of resource management with implications for wildlife; including, the North Cascades National Park General Management Plan (1988), Lake Chelan NRA General Management Plan (1995), Ross Lake NRA General Management Plan (2012), the Park Resource Management Plan (1999), the Park Wilderness Management Plan (1989), the Park Fire Management Plan (2007), the Park Mountain Fishery Management Plan/EIS (2008), the Park Invasive, Non-native Plant Management Plan/EIS (2012), Agreement Regarding Establishing New Rock Climbing Routes in the Skagit Gorge Area (2010), and the Stehekin River Corridor Implementation Plan/EIS (2012). The Park also uses the North Coast and Cascades Network Vital Signs Monitoring Report (Weber et al. 2009) as guidance for natural resource planning and management.

Through the Park Superintendent's Compendium (NPS 2023), authority is granted as 36 CFR §2.2 to provide additional protections for wildlife in the Park. Examples of statutes included in these laws are the authority to implement short-term public closures for wildlife protection, which may include active nest or den sites, or situations where public safety is of concern (e.g., a large predator with prey on a trail or near a campsite), prohibition of artificial lights for wildlife viewing or hunting,

prohibition of trapping, requirements for use of bear-resistant food storage containers, and regulations pertaining to use of pack animals and dogs.

In day-to-day stewardship practice, all proposed work in the Park related to maintenance, facilities, construction, research, and management implementation is evaluated by an Interdisciplinary Team comprised of staff from each Division (Resource Management, Visitor and Resource Protection, Facilities and Maintenance, Visitor Services, and Administration). This team reviews compliance with all applicable laws and policies before initiation of any work. For wildlife, this may include identification of specific threats to wildlife and their habitats and required mitigations before the work is initiated (see Appendix A), informal or formal consultation with U.S. Fish and Wildlife Service (USFWS) for species protected under the ESA, and consultation with Washington Department of Fish and Wildlife (WDFW) for species protected under State law. Some routine work is detailed in programmatic plans, such as the Routine Trail Maintenance Plan 2021–2030, for which USFWS has issued a Biological Opinion (USFWS 2022b).

Wildlife research investigations proposed by external institutions, agencies, Tribes, and organizations are reviewed through the Park research permit program and must include Institutional Animal Care and Use Committee (IACUC) approval from the NPS when the work involves potential impacts to wildlife species. Likewise, wildlife research conducted internally by Park biologists that involves capture and handling of wildlife requires a NPS IACUC-approved protocol.

The Park Wildlife Program guides the focus of wildlife monitoring and research conducted by NPS staff and implements management actions. The mission of the Park Wildlife Program is "protect, conserve, and restore wildlife species and their habitats through an ethic of stewardship that values the well-being of individuals, populations, and communities. The [Park] Wildlife Program will provide scientific leadership, scholarship, and active collaboration with partner agencies, Tribes and First Nations, and non-governmental organizations, toward stewardship of wildlife resources at an ecosystem scale, in a paradigm of continuous change that is not yet fully understood, based on best available science, accurate fidelity to the law, and with consideration of the diverse cultural values of humans that share the North Cascades Ecosystem" (Park Wildlife Program Strategic Plan). Core elements of the Wildlife Program include 1) at-risk species recovery and restoration, 2) climate resiliency, 3) wildlife stewardship and management, 4) scientific leadership, and 5) human resource excellence.

Chapter 2. Birds

Bald eagle *(Haliaeetus leucocephalus)*

The bald eagle (*Haliaeetus leucocephalus*) is a massive, broad-winged raptor, widely distributed throughout North America (Figure 4). Adult bald eagles are easily identifiable with a characteristic white head and tail with dark brown body and wings. Their legs and hooked bill are bright yellow. Bald eagles can measure up to 1 m long and have a wingspan ranging from 1.8–2.3 m (Buehler 2000). Immature birds have a mostly dark head and tail with varying amounts of mottling throughout the body and wings. Immatures acquire adult plumage at 4–6 years of age (Buehler 2000). Both sexes have similar plumage, but are dimorphic, with females growing larger than males. Bald eagles can be found throughout Washington State but are more common in the maritime climate west of the Cascade Range where they can be seen along coasts, lakes, wetlands, and rivers during nesting and wintering periods (Smith et al. 1997). Contemporary presence and breeding of bald eagles in the Park has been confirmed (Figure 5).



Figure 4. Bald eagle voucher record NOCA38079, photographed by Jason Ransom/NPS, December 17, 2014, near the Skagit River, Ross Lake National Recreation Area, WA.

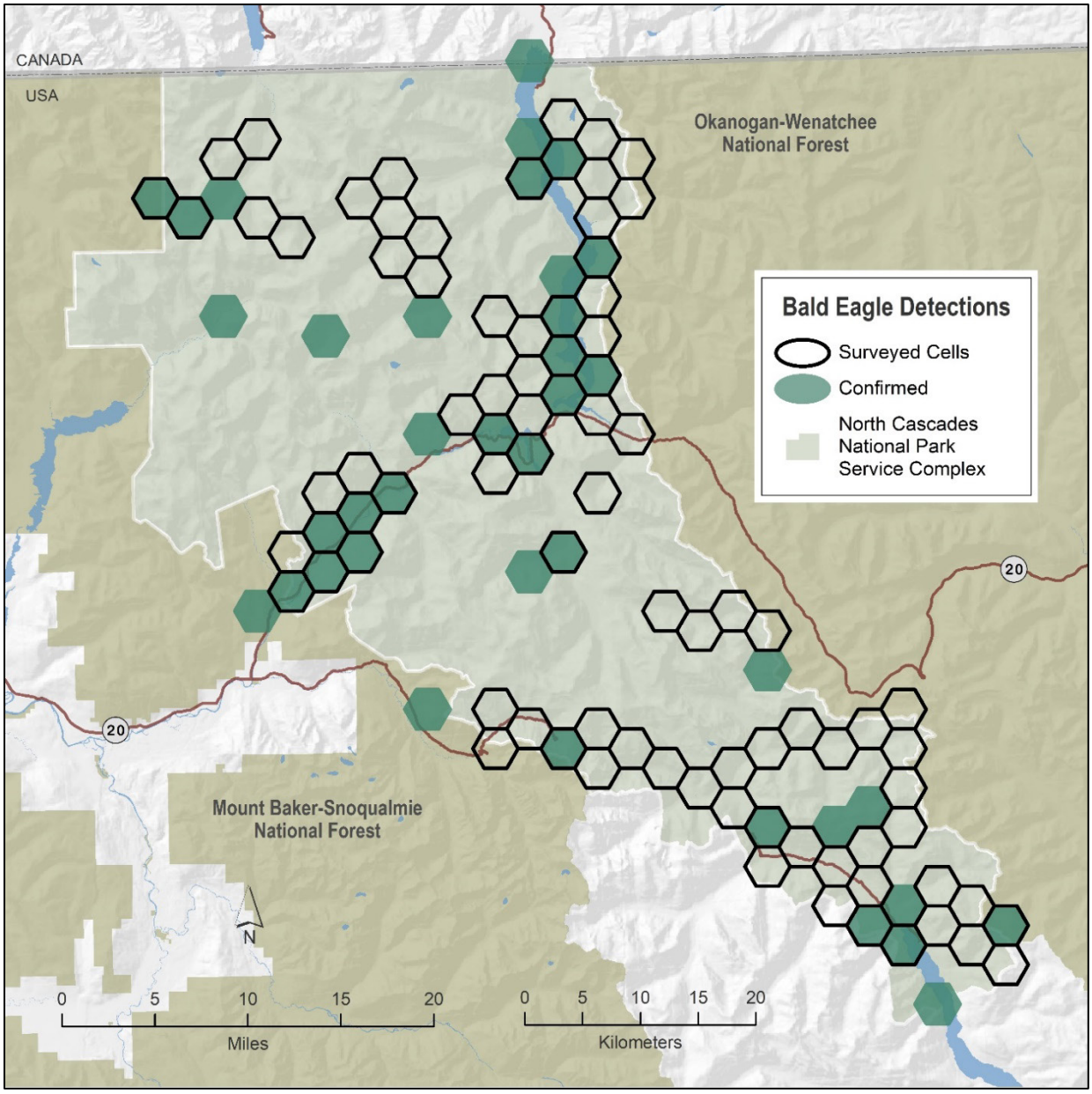


Figure 5. Survey locations, and bald eagle (*Haliaeetus leucocephalus*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 3). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 3. Bald eagle (*Haliaeetus leucocephalus*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2018	Total count of individuals on the Skagit River	Weekly during 6–12 weeks of winter	Dunwiddie and Kuntz II, 2001; Rubenstein et al. 2019
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2005–2020	Nest site monitoring (visual observations) at 3–4 nests	2–4 times per nest each year, May–August	Christophersen and Ransom 2022
2005–2020	Landbirds point counts and incidental detections at 1,222 points (visual and acoustic observations) [a]	Annual or alternate year transects, May–July	Holmgren et al. 2015–2017, 2019–2021; Ray et al. 2017, 2018; NPS unpublished data

[a] Surveys were not designed for optimal detection of this species.

Since the arrival of European settlers, bald eagles have faced declines due to habitat loss, reduction of food sources, and persecution by humans. Over the last two centuries, the decrease in carrion from such species as bison (*Bison bison*), and the reduction in spawning anadromous fish populations may have contributed to long-term eagle declines. Development along coasts and lakes also reduced nesting and roosting habitat. Eagles were historically considered a pest species and regularly shot, trapped, and poisoned due to a largely erroneous belief that they posed a threat to lambs and other small stock animals. States would pay bounties for eagle carcasses, along with a variety of predator species, leading to widespread culling of both bald and golden eagles (Stalmaster 1987, Stinson et al. 2007).

The use of the DDT in the 1940s precipitated even more dramatic declines in eagle populations. Nest failure due to weakened eggshells and direct mortality in adult eagles was rapid, and by 1963, survey efforts found fewer than 700 breeding pairs in the lower 48 states (Stalmaster 1987). In the 1970s, the United States banned the use of DDT and some other environmental contaminants and enacted a number of legislative actions protecting eagles. Bald eagle populations began to rebound, and territories occupied by pairs in the lower 48 states increased from an estimated 791 to 9,789 between 1974 and 2007 (USFWS 2007a): Washington saw a 707% increase in occupied territories from 1981–2005 (Stinson et al. 2007).

A review of all known bald eagle territories in the Washington Species Data Management System indicates that the number of nesting territories has increased by an average of 28 per year since 2005 when the species was down-listed in the state to Sensitive. As of 2015, the total number of known territories in the state was 1,334, but this total reflects the cumulative number of sites and not the number that are known to be active in any particular year (Kalasz and Buchanan 2016). With continued federal protections meant to sustain the population, the species is expected to continue as an important and thriving part of the state's natural diversity for the foreseeable future.

The Park supports large congregations of wintering bald eagles that come to feed on spawning salmon, particularly along the Skagit River. Winter bald eagle surveys occur between November and March and have been conducted in the Park since 1982 (Dunwiddie and Kuntz 2001, Rubenstein et

al. 2019). The average weekly count of bald eagles during winter on the Skagit River from 1982–2016 was 180 birds (range = 54–371) (Rubenstein et al. 2019). These data provide insight into the demographic and population trends of wintering eagles in the Park. Wintering eagle populations in Washington can be three or four times those of the breeding population, with migrants arriving from Alaska, Canada, Montana and California (Stinson et al. 2007).

During the breeding season, bald eagle observations have historically been relatively rare in the Park, with most breeding activity taking place along lower elevation rivers, lakes, and estuaries outside of Park boundaries (Smith et al. 1997). Nesting surveys in the Park date back to 1986 when interagency helicopter flights were conducted to identify nesting territories and report productivity in the Skagit and Baker River watersheds. A similar interagency helicopter survey was conducted in 1988 in the Stehekin River watershed. No bald eagle nests were observed during either of the surveys.

Although there have been some past occurrence records of summer bald eagle activity in the Park, actual known nesting has only recently been documented (Christophersen and Ransom 2022). An increase in bald eagle activity during the breeding season was noted along the reservoirs and rivers of Ross Lake NRA and Lake Chelan NRA. Subsequently, these observations led to more intensive nesting surveys with the first nest site confirmed in 2015 near Weaver Point in Lake Chelan NRA. Annual nesting surveys continued and the total number of confirmed bald eagle nests in the Park has since increased to four, based on surveys conducted through 2020 (three in Ross Lake NRA and one in Lake Chelan NRA), producing a total of six offspring across the 6-year period (Christophersen and Ransom 2022).

The Park reported an average nest territory occupancy rate of 47% for the 6-year period from 2015–2020 (Christophersen and Ransom 2022). This is notably less than a statewide survey in 2005 that reported an occupancy average of 75% (Stinson et al. 2007). The successful nesting rate in the Park was 50%, and a minimum level of 45% nest success has been suggested as the threshold for populations to remain stable (Sprunt et al. 1973). Bald eagle nest productivity in the Park averaged 0.95 young per occupied nest, which is less than the average 1.2 young per nest reported for the rest of the United States (Millsap et al. 2016). Although the time series is short, the percent of occupied nests and nest success both declined from 2015–2020 for reasons that are unclear.

Habitat Requirements

The most critical characteristics of bald eagle nesting habitat include the presence of large mature trees that are close to open water and buffered from intense human activity (Watson and Pierce 1998, Stinson et al. 2007). Bald eagles build some of the largest of all bird nests and require large trees that have structural characteristics capable of supporting the weight of the nest. Their massive stick nests often measure >1.5 m in diameter and 0.6–1.2 m in height and may weigh >100 kg. Nest trees are often the dominant or co-dominant individuals in the forest stand, offering easy access on approach and a sweeping view of the surrounding landscape (Watson and Pierce 1998). Grubb et al. (1975) found that most nest trees in coastal Washington are old-growth Douglas-fir and Sitka spruce (*Picea sitchensis*). Nests trees in the Puget Sound Basin are often mature grand fir (*Abies grandis*) and black cottonwood (*Populus balsamifera*) (Watson and Pierce 1998). Although far fewer in number, nests in

eastern Washington are most often associated with ponderosa pines and black cottonwoods (WDFW unpublished data).

An important component for wintering bald eagle habitat includes suitable communal night roosts. These roost locations occur in stands that may not always meet the strict definition of old-growth forests, but they do contain at least some remnant large trees that are preferred by roosting eagles and can include small patches of forest with old trees in suburban areas (Anthony et al. 1982, Watson and Pierce 1998, Hansen et al. 1980). Roost trees are generally located near areas that provide a rich food source such as anadromous fish or high concentrations of waterfowl. Forest stand composition at communal roosts in the Pacific Northwest are similar to tree species used for nesting, including ponderosa pine, mixed-conifer, Douglas-fir, black cottonwood and western larch (*Larix occidentalis*). The number of eagles using a roost site can be highly variable with some containing as many as 400 individuals on any given night (Crenshaw and McClelland 1989). Although the adaptive strategies of communal roosts are not well understood, they tend to occur in locations that are protected from the wind by vegetation or terrain, which are thought to provide a more favorable thermal environment requiring less energy expenditure (Anthony et al. 1982). These sites may also aid in the establishment of social functions and locating food (Stalmaster 1981, Keister 1981).

Diet

The bald eagle's primary diet consists of fish throughout most of its range, but they are adept predators and are known to take a variety of mammals and birds (especially waterfowl), as well as depredating nests and feeding on carrion. Watson (2002) reported bald eagle diet consisting of 78% fish, 19% birds, and 3% mammals. Bald eagles have also been known to steal prey from mammals and other birds and have occasionally been observed hunting cooperatively (Stalmaster 1987, Stinson et al. 2007). Bald eagles overwintering in Northwest Washington are particularly dependent on chum salmon (*Oncorhynchus keta*) and other salmon species in the fall and early winter (Stinson et al. 2007) and rely more heavily on waterfowl in mid- to late-winter (Elliott et al. 2011); carrion is also consumed during winter (Stalmaster 1987, Watson 2002).

Reproduction

Bald eagles generally begin breeding in their sixth year and can form lifelong pair-bonds between mates (Stalmaster 1987, Jenkins and Jackman 1993). Bald eagles may attempt to breed at three or four years of age if food is particularly abundant and competition for resources is scarce (Gerrard et al. 1992, Buehler 2000). A breeding pair of bald eagles occupy and defend a territory that includes any active nest and may also include one or more inactive nests that are built or maintained but not used for nesting in a given year. In coastal areas of Washington, courtship and nest building or decorating begins as early as December, with egg laying by March, and hatching by the end of April. Anecdotal information suggests there is about a 2-week lag period from coastal nesting chronology to interior Park nesting dates. A nesting pair generally produces between one and three eggs per nest with two eggs being most common (Buehler 2000). Both adults participate in the incubation stage lasting for about 35 days, with young fledging in 75–90 days, during July and early August (Watson 2006). Fledged young usually remain close to the nest, dependent on the adults for food until they disperse approximately six weeks later. The entire breeding season from initial activity at a nest to

young dispersal is about six months. Bald eagles are long-lived with the oldest known wild bald eagle living at least 28 years and the oldest captive eagle living to 47 years of age (Stinson et al. 2007).

Protective Status

Bald eagles were one of the first species to be listed under the Endangered Species Preservation Act of 1966. Protection was carried over with the enactment of the current ESA in 1973 and expanded to include Washington and the rest of the lower 48 states in 1978. Bald eagles were de-listed from the federal ESA in 2007 (USFWS 2007a) but remain under federal protection by virtue of the Bald and Golden Eagle Protection Act, the Migratory Bird Treaty Act (MBTA), and the Lacey Act. The NPS has designated the bald eagle as a Management Priority species within the Park.

The bald eagle's inclusion in the ESA, and its subsequent recovery, has been celebrated as a major victory for environmental legislation although current numbers are still significantly lower than estimates for pre-European settlement populations (Buehler 2000). Recent data from North America indicate that population growth between 1966 and 2013 was 5.37% annually, and modeling indicates that population growth across the range is projected to continue for another 10 to 20 years until the total population stabilizes at around 228,000 birds (Kalasz and Buchanan 2016).

Known Threats

Habitat degradation and disturbance from shoreline development have been identified as major threats to bald eagle nesting and roosting habitat (USFWS 1994). Loss of habitat is also increasingly at risk from high severity fire (Dellasala et al. 1998). Factors that may impact nesting productivity (number of young produced per territory) may include human disturbance, food availability, weather, contaminants, and climate change (Dzus and Gerrard 1993, Dykstra 1995, Buehler 2000). Disease outbreaks, such as West Nile virus and avian influenza, can also disproportionately affect raptors.

Human Disturbance

Human activity, either through landscape disturbance or the presence of boats on water, is negatively associated with nest and roost site-selection and preferred forage area (Stinson et al. 2007, Livingston et al. 1990). However, as with many raptors, the degree of observed impacts of disturbance on nesting bald eagles has varied widely between studies, and even individual eagles, with some individuals showing remarkable tolerance for human activity, and successfully nesting in urbanized areas. This may result from a combination of tolerance from repeated exposure to humans, coupled with a shortage of nesting locations. Bald eagle nest site fidelity appears to increase when the population nears carrying capacity, despite increasing human encroachment in an area, particularly when suitable unoccupied nest sites are limited (Watson et al. 1999).

While some eagles appear to tolerate varying levels of disturbance, they tend to nest away from human activity when alternative nesting options are available. For nesting birds, the distance from disturbance appears to be the most impactful, with both vertical height of the nest and horizontal distance from human activity positively correlated with nesting success (Watson 2004). The distance and duration of disturbance has also been indicated as an important factor for recreational boating impacts on nesting success, leading managers to recommend boat exclusions within 100 m of nests,

and the implementation of no-stopping-zones in the proximity of nest trees (Grubb et al. 2002). Eagle presence has been positively correlated with the distance of human development from shorelines, and negatively correlated with pedestrian use on beaches and near-shore boat traffic (Buehler et al. 1991). Camping within 100 m has been observed to impact bald eagle behavior, reducing the quantity of prey fed to nestlings by nearly 30%. Some individual eagles appeared to habituate to campers within 100 m, but habituation did not carry over between other events, with each new encounter requiring a new habituation period (Steidl and Anthony 2000).

Watson and Pierce (1998) found that the presence of human structures closer than 60 m of nests had a negative impact on incubation times. Incubation was the most influential factor on nest success, with failed nests incubated an average of 14 min/hr less than successful nests. On average, successful nests had been exposed to <50% the rate of disturbance from pedestrians, aircraft noise, and other human activities than unsuccessful nests. Pedestrian activity was the most frequent disturbance, and pedestrian use within 200 m of a nest had a negative effect on success (Watson and Pierce 1998). Excluding pedestrians within this distance threshold and maintaining visual barriers between pedestrians and nests appear to be effective mitigations to pedestrian disturbance (Watson 2004).

Stalmaster and Kaiser (1998) found that recreation (foot traffic and motorboats) on the Skagit River decreased eagle feeding, with >40 recreational events/hr associated with a reduction of bald eagle feeding time of nearly 90%. On weekends, when recreational use of the river is highest, eagles fed 30% less, with individual eagles showing varying levels of tolerance to human presence. When surveyed, recreational users rarely believed that their activities had a negative impact on eagles (Stalmaster and Kaiser 1998).

Increased corvid populations, often associated with human activity, may also have a negative impact on the productivity of potential bald eagle nesting habitats in or near developed areas, and may keep eagles that could otherwise utilize suitable habitat from breeding successfully. Nest abandonment and nesting failure has been reported following intense crow harassment, although this interaction has not been well studied (Thompson 1998).

Food Availability

The abundance and availability of anadromous fish species are important factors that influence the productivity, phenology, and population dynamics of bald eagles (Hansen 1987, Willson and Halupka 1995). Reduced salmon resources may force bald eagles to depend on other prey species that are less beneficial, which may expose them to increased anthropogenic threats and mortality (Millsap et al. 2004).

Rubenstein et al. (2019) analyzed the relationship between chum escapements, eagle numbers, and flood events in the Skagit River with data spanning from 1982 through 2016. Results showed that eagle counts had a strong positive relationship between chum escapement and a negative relationship with flood events. Peak chum runs are happening earlier each season, and post peak floods are interacting mainly with carcasses, washing them off accessible sand bars and effectively removing salmon as a carrion resource for eagles (Rubenstein et al. 2019). Winter flow management from hydroelectric dams that raises water levels soon after salmon spawning may have the same effect of

flushing salmon carcasses back to sea, degrading availability of this important winter food source for bald eagles.

Weather

Rainfall and cold temperatures during the critical spring egg laying, incubation and hatching months have been identified as potential drivers of reduced reproduction in bald eagles and other raptors (Gerrard and Whitfield 1979). Frequency of heavy rain has been reported as having a much greater impact on raptor nestling survival than the total amount of precipitation recorded during the rearing period (Anctil et al. 2014). Although weather was not investigated thoroughly within the scope of monitoring efforts in the Park, it could be a major factor in explaining low productivity in some years.

Environmental Contaminants

Although DDT has been banned for decades, it still persists on the landscape, and has been detected in fish from the Columbia Basin and Lake Chelan, with sources originating from eroding stream banks and nearby orchards (Burgoon and Cross 2005, Lyndal et al. 2013). As predators at the top of the aquatic food chain, possible bioaccumulation and trophic level increase could pose a threat to bald eagle health.

Elevated levels of mercury and lead have been detected in bald eagle tissues across the United (Stinson et al. 2007, Cruz-Martinez et al. 2012). An adult bald eagle was recovered near the Cascade River (just outside of the Park), after being electrocuted on power lines in 2019. Toxicology screening of liver tissue from this eagle revealed elevated levels of iron (1400 ppm), and detectable levels of mercury (9.8 ppm), methyl mercury (2500 ppb), manganese (4.1 ppm), molybdenum (1.2 ppm), zinc (43 ppm), and copper (8.9 ppm), but no detection of lead, arsenic, or PCBs. Statewide from 1998–2008, half of the injured or dead bald eagles recovered and admitted to the Washington Raptor Rehabilitation Program had elevated levels of lead present in their system (Kalasz and Buchanan 2016).

An emerging and pervasive environmental contaminant, microplastic, is also a potential threat to bald eagles and other raptors through bioaccumulation (Wagner et al. 2014). Microplastics are plastic fragments smaller than 5 mm that may pervade the environment through atmospheric deposition. These microplastics are of great ecological concern in aquatic systems because they can be easily ingested and accumulate as indigestible matter throughout a food web. Study of this issue in aquatics organisms is ongoing in the Park. Primary microplastics analyses (tissue digestions and ultraviolet detection) of the gastrointestinal tract from the 2019 bald eagle was conducted at the University of Alaska in 2020. No microplastics were detected from this sample. While encouraging, this small sample size and basic analysis does not rule out the potentially serious risk that bioaccumulation of microplastics in raptors poses.

Electrocutions

Fatalities from power line electrocutions in Washington appear to be relatively rare, with an estimated 15–25 bald eagle mortalities each year. This can vary by type and density of electric power infrastructure. In year 2000, Whatcom, Skagit and Island counties reported about 14 bald eagle

fatalities that were caused by electrocution (Stinson et al 2007). One mortality was recovered in Skagit County adjacent to the Park in 2019. Transmission lines appear to be less of an electrocution hazard than distribution lines, because their size, spacing, and construction makes it unlikely that an eagle will make two points of contact at once. Electrocutions occur most frequently in areas where large concentrations of eagles coincide with distribution lines. Efforts from power companies to retrofit raptor-safe designs and protocols in areas of higher raptor density and electrocution incidence may be beneficial in mitigating mortalities (Stinson et al. 2007).

Climate Change

During the next 20 to 40 years, the climate of the Pacific Northwest is projected to change significantly, with potential consequences for forest ecosystems. Predicted changes include warmer, drier summers, and warmer, wetter autumns and winters, resulting in diminished snowpack, earlier snowmelt, and an increase in extreme heat waves and precipitation events (Littell et al. 2009, Salathé et al. 2010, Mote and Salathé 2010, May et al. 2018). Notably, an extreme heat wave impacted western Washington in late June 2021, and regional raptor nest monitoring reports from camera-monitored nests included fledgling mortalities from this single weather event.

One of the largest potential effects on Pacific Northwest forests is likely to come from an increase in fire frequency, duration, and severity (Littell et al. 2009, Westerling et al. 2006). From a long history of fire suppression, fuel accumulations in fire-adapted forests have increased considerably and converted fire regimes from periodic low-intensity surface fires to high-intensity crown fires (Agee 1993). The buildup of fuels has increased the risk for catastrophic fires that could potentially alter or completely devastate bald eagle nest sites and communal roost areas. Active management can help to mitigate risk through the removal of ladder fuels and slash around the base of nest trees in combination with thinning and prescribed burns at appropriate landscape scale (Dellasala et al. 1998, Agee and Skinner 2005).

Throughout the Pacific Northwest, the warming effects of climate change threatens to drastically reduce fish distribution and viability (Beechie et al. 2013). Changes in water temperatures, stream flow and timing, and altered land uses and fire regimes are projected to have cumulative impacts to foraging, migration and overwintering fish habitats (Mantua and Raymond 2014). These climate patterns may influence the abundance and accessibility of salmon carcasses that wintering bald eagles in the Puget Sound area depend on as a primary food resource. Riparian restoration strategies have greatly improved salmon habitat, but it remains unclear if restoration efforts will be enough to offset increasing climate change impacts in individual stream basins, especially at higher elevations where there are greater snow-rain transitions as well as accelerated loss of glacial ice (Battin et al. 2007, Bond et al. 2015).

Habitat Loss and Alteration

Habitat loss and alterations are the most pervasive threats that contribute to a decrease in nesting trees or slow the development of large trees capable of supporting a nest, thus reducing the potential for bald eagle nesting territories in Washington. Degradation or permanent loss of habitat is often due to anthropogenic actions such as timber harvest and development, and natural disasters such as windstorms, floods and wildfire. Communal night roost habitat is also susceptible to these same

threats. Over the last hundred years, the Puget Sound Basin has lost nearly half of its estuarine wetlands, mainly due to human development. Loss of bald eagle nesting habitat is widespread, with three million people now occupying shoreline lands on the Washington Pacific coast, and along the shores of Puget Sound and Hood Canal. Trends in habitat loss and additional land use changes will likely continue with the human population of Washington topping 7.7 million in 2021 and projected to reach 9.8 million by 2050 (Washington State Office of Financial Management 2021).

Conservation and Research Needs

Information on the population status of bald eagles and other raptors is important for the adaptive management of Park resources. As top avian predators, they act as sentinels of larger ecological change, as they are especially vulnerable to habitat degradation, environmental contaminants, human disturbances, and a changing climate. New threats to raptors and ecosystems emerge on a routine basis, and many are conflated with changing climate. Unexpected changes such as the massive influx of Park visitors stemming from broad societal response to the SARS-CoV-2 pandemic can quickly change the success or failure of raptor nesting in any given year. Earlier snowmelt, longer growing season, earlier plant flowering and invertebrate emergence, and shifting migration phenology could all influence food availability and breeding synchronicity with resource availability. These same biological shifts also portend earlier, and more frequent, visitor access to sensitive nesting areas in the Park. Management of resources for Park managers is complex, interconnected, dynamic, and often time sensitive. Some risks, such as wildland fire, are mediated by real-time information connecting fire crews with locations of sensitive resources like raptor nests, that need specific types of protection. This is accomplished through annual (or more frequent) updates to the Park Resource Advisors Guide. Some risks, such as environmental contamination through atmospheric deposition, require regional, continental, or global scale mitigations to address. Other environmental contaminants may arise locally and be traced and mitigated at the source. One major limitation of work performed in the Park is the lack of information concerning prey abundance for bald eagles, especially during the nesting season. This has not historically been conducted due to limited resources; however, some important long-term monitoring data on fish abundance could provide insight. This type of rigorous monitoring conducted simultaneously with nesting surveys could help emphasize the relevance of such data for understanding the primary drivers of bald eagle and other raptor ecology.

Specific conservation needs for bald eagles in the Park include mitigations for aircraft use near active nests, nest protection measures during wildland fire events, visitor use management around active nests, anti-strike devices on electrical transmission lines in flight routes around nests and primary hunting areas, hydroelectric flow regulation that acts to preserve salmon carcasses during winter, and focused monitoring of environmental contaminants, heavy metals, and microplastics in both eagles and their prey. Eagle carcasses should also be tested for West Nile virus and avian influenza. Routine management, facilities, and infrastructure operations conducted in the Park can affect bald eagles, but adherence to noise and disturbance guidelines around nests may help mitigate local impacts (Appendix A).

Clark's nutcracker (*Nucifraga columbiana*)

Clark's nutcracker (*Nucifraga columbiana*) is a long-lived, year-round resident in high elevation conifer forests throughout montane ecosystems in the western U.S. and Canada, including the Pacific Northwest (Figure 6) (Tomback 1998). Nutcrackers specialize on masting conifer seeds, foraging on immature and mature seeds, then relying on caches for overwinter survival and for feeding young during the early spring breeding season (Mewaldt 1956, Tomback 1998). Nutcrackers are important seed dispersers for at least ten conifer species in the western U.S. and are an obligate mutualist of whitebark pine (*Pinus albicaulis*); whitebark pine depends on nutcrackers for dispersal of their wingless seeds (Tomback 1978, Hutchins and Lanner 1982, Lorenz and Sullivan 2009). In the North Cascades Ecosystem, nutcrackers forage on whitebark pine, ponderosa pine, Douglas-fir, and western white pine (*Pinus monticola*). Each individual annually stores an estimated 32,000 to 98,000 seeds in tens of thousands of below- and above-ground locations (3–10 times their overwintering nutritional needs), and unretrieved seeds have the potential to germinate (Vander Wall and Balda 1977, Hutchins and Lanner 1982, Tomback 1982, Lorenz et al. 2011). Clark's nutcrackers are ecosystem engineers, moving seeds up to 32.6 km (20 mi), across latitude and elevation, and into recently disturbed areas, such as wildfire footprints (Tomback et al. 2001, Lorenz et al. 2011).



Figure 6. Clark's nutcracker ID171445, photographed by Jason Ransom/NPS, February 11, 2018, near Winthrop, WA, after banding and fitting with a radio transmitter.

Clark's nutcrackers are jay-sized corvids, weighing 106–161 g, with moderate sexual size dimorphism (Tomback 1998). They are light to medium gray, with glossy black wings and tail, and white around the eyes, on the forehead, chin, vent, base of tail, tips of secondaries, and outer rectrices (Tomback 1998). They have co-evolved with whitebark pine, and co-evolved features include their

long, decurved bill which allows them to efficiently open cones and extract seeds, their expandable sublingual pouch in which they can transport up to 150 seeds, their strong flight capabilities, and their excellent spatial memory, enabling them to remember thousands of seed cache locations (Bock and Balda 1973, Tomback 1977, Vander Wall et al. 1981, 1981, Balda and Kamil 1992). Nutcrackers have numerous vocalizations, and their common, distinctive "caw-caw" can be regularly heard throughout their range (Mewaldt 1956, Tomback 1998).

Clark's nutcrackers are not territorial (Lorenz 2009, Schaming 2016b). They move over a large area, with 95% kernel home ranges of satellite-tagged nutcrackers in Washington ranging from 7–156 km^2 (2.6–60 mi^2) during the seed harvest season, and 14–255 km^2 (5.5–98 mi^2) during the non-harvest season (Schaming et al. in prep). Ranges expand and contract between seasons and years, and in Washington, whitebark pine cone crop (low, medium, high) significantly predicted home ranges and core areas (95% and 50% kernel, respectively) during both harvest and non-harvest seasons; the largest harvest season home ranges occur during low cone crops, and the largest non-harvest season home ranges follow moderate cone crops (Schaming et al. in prep). Based on breeding season point count surveys at North Cascades and Mount Rainier National Parks from 2005–2014, nutcrackers are declining in Mount Rainier National Park but remain stable in North Cascades NPS Complex (Ray et al. 2017). However, the number of nutcrackers observed annually in the Park fluctuates drastically, and variation is not explained by whitebark pine metrics alone (Ray et al. 2020). The low and fluctuating numbers of Clark's nutcrackers in the Park may be related to the high whitebark pine cover east of the Park: the birds may move into or through the Park in only some years, based on availability of whitebark pine and ponderosa pine cones in the Park and in the region (Ray et al. 2020).

Nutcrackers are also facultative partial migrants, moving regionally to search for food, and emigrating further distances in years with widespread food shortage (Davis and Williams 1957, 1964, Vander Wall et al. 1981, Schaming 2016a). Recent satellite-tracking of nutcrackers east of the Park showed that the birds did not emigrate from the region, even in one year with low or absent whitebark pine cone crops in local stands (Schaming et al. in prep). In contrast, in Lorenz's (2009) radio-tracking study, 21 radio-tagged individuals emigrated out of the study area following large whitebark and ponderosa pine cone crops, moving up to 75 km (29 mi); large flocks of emigrants were also observed between September 2006 and spring 2007 (Lorenz 2009, Lorenz and Sullivan 2009). Individuals in the Greater Yellowstone Ecosystem were recently observed to exhibit classic migratory behavior—during one autumn with a moderate whitebark pine cone crop, a majority of satellite-tagged nutcrackers emigrated to Utah, up to 650 km (404 mi) away, and overwintered there (Schaming 2016a). The majority of these emigrants returned to the Greater Yellowstone the following year (Schaming 2016a). It is unknown how common long-distance movements are in different regions, what the specific triggers are, or why some nutcrackers emigrate while others do not.

Because habitat loss, including the decline of whitebark pine, is reducing availability of seed resources, Clark's nutcrackers are an emerging conservation concern. Nutcracker behavior is still poorly understood, in large part due to their remote habitat and landscape-scale movements, as well

as their low detectability during traditional multi-species bird surveys (Tomback et al. 2001, Lorenz and Sullivan 2010, Schaming and Sutherland 2020). There is sparse information about the status and trends of Washington populations, and because a high proportion of nutcracker research has been conducted outside of the Pacific Northwest, it is unknown to what extent information about nutcracker behavior and trends from these other locations is valid for Washington nutcrackers.

Little research has been conducted on Clark's nutcrackers in the Park. Annual breeding season point count surveys for landbirds have been carried out in the Park since 2005 (Figure 7) (Ray et al. 2017, 2020). Clark's nutcrackers have been observed throughout the year by staff and visitors, but were detected on formal surveys only at elevations and dates when whitebark pine seeds were available, suggesting that nutcrackers breeding or otherwise spending much of their annual cycle outside the Park may have travelled into the Park to forage (Ray et al. 2020). Nutcrackers range over large areas and fly up to 32.6 km (20 mi) from their summer home range to harvest whitebark pine seeds (Lorenz et al. 2011). Breeding in the Park has not been confirmed. Of seven nutcrackers satellite-tagged outside of Winthrop, WA in 2018, the ranges of three extended into the Park. Ongoing nutcracker trapping, satellite-tagging and satellite-tracking is planned for the Park in 2022–2023. Finally, a long-term nutcracker survey project is underway in Washington's Cascades to determine how Clark's nutcracker occupancy relates to habitat and other environmental variables through data collected with acoustic recording units; preliminary results are pending (Taracka 2021).

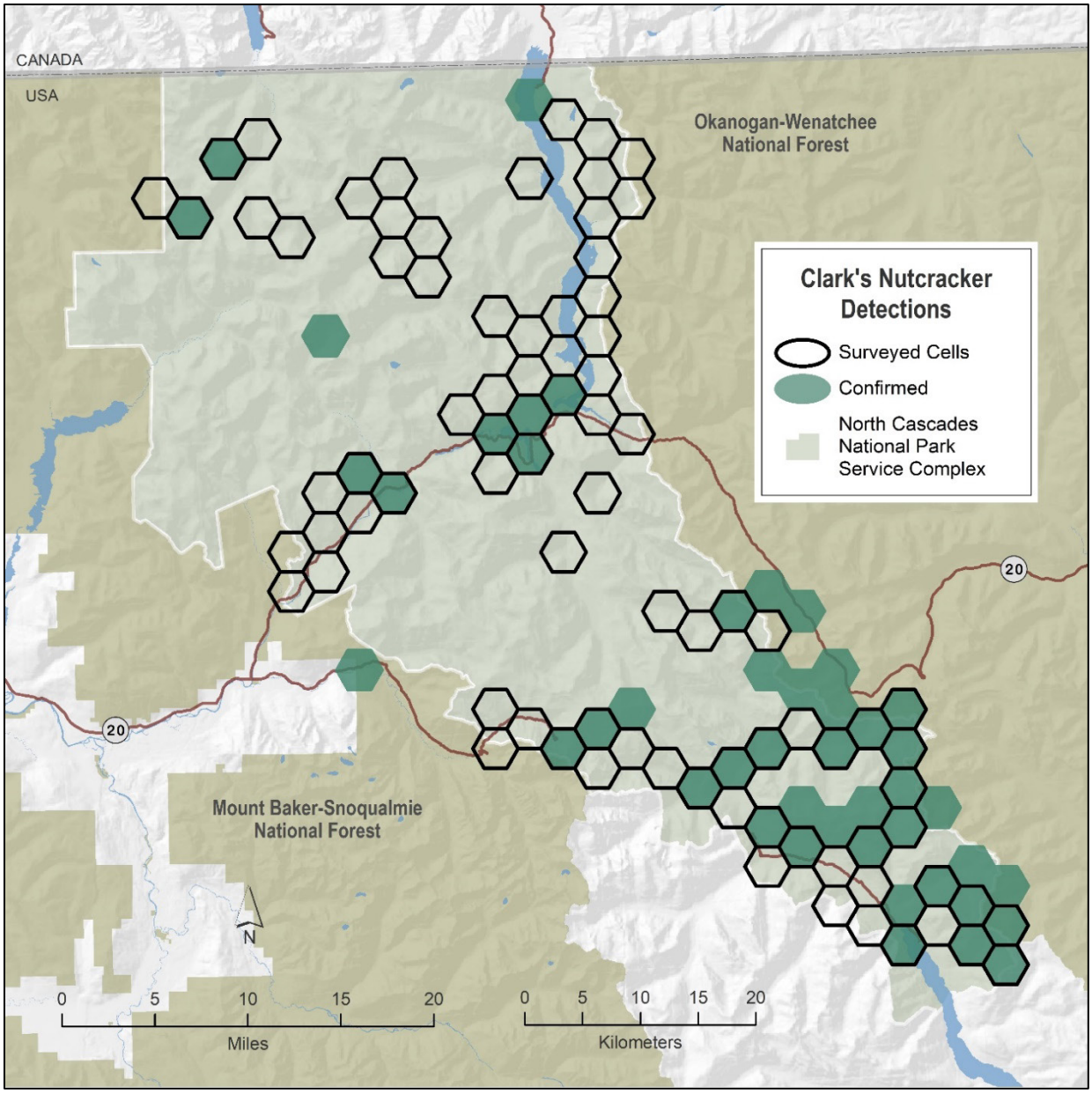


Figure 7. Survey locations, and Clark's nutcracker (*Nucifraga columbiana*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 4). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 4. Clark's nutcracker (*Nucifraga columbiana*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2005–2020	Landbirds point counts and incidental detections at 1,222 points (visual and acoustic observations)	Annual or alternate year transects, May–July	Holmgren et al. 2015–2017, 2019–2021; Ray et al. 2017, 2018, 2020; NPS unpublished data
2018–2020	Radio-telemetry locations for 7 individuals	20,766 points across 7 individuals, all months	T. Schaming and NPS unpublished data

Habitat Requirements

Nutcrackers primarily inhabit conifer habitat in high montane forests, but may move through lower-elevation forests, particularly during winter (Tomback 1998). They rapidly switch food resources based on what is available, moving from one conifer habitat to another once cones are depleted (Tomback 1978, Vander Wall 1988, Lorenz and Sullivan 2009). In the North Cascades Ecosystem, resident nutcrackers selected home ranges containing mixed forests and either whitebark pine or ponderosa pine (Lorenz 2009, Schaming et al. in prep). Optimal nutcracker habitat includes multiple conifer seed sources.

In one Rocky Mountain study, nutcracker occupancy is associated with whitebark pinecone presence, but not cone density, and in two studies, nutcrackers regularly occurred in stands with low cone production (Maier 2012, Schaming and Sutherland 2020). Two other Rocky Mountain studies suggested 1,000 cones/ha and live whitebark pine basal areas of >2–5 m^2/ha is necessary for a high likelihood of nutcracker occurrence (McKinney et al. 2009, Barringer et al. 2012). Whitebark pine abundance on the landscape may be more important than stand level basal area (Barringer et al. 2012, McLane et al. 2017, Schaming and Sutherland 2020), and a high probability of nutcracker occurrence is predicted to occur when the landscape contains at least 125 km^2 (48 mi^2) of cone-bearing whitebark pine habitat within 32.6 km (20 mi) of a location (Schaming and Sutherland 2020).

Whitebark pine stands with alternative seed food sources within 3.2 km (2 mi), the median diameter of the nutcracker breeding season home range, may be preferred when selecting home ranges (Schaming 2016b, Schaming and Sutherland 2020). In the Greater Yellowstone, an optimal habitat mosaic includes whitebark pine and moderate levels of Douglas-fir (Schaming and Sutherland 2020). Clark's nutcrackers selected for Douglas-fir, but not whitebark pine in their breeding season home ranges, but Douglas-fir was primarily used when in a habitat mosaic with whitebark pine (Schaming 2016b, Schaming and Sutherland 2020). Douglas-fir may be particularly important for non-foraging purposes, but Williams et al. (2020) determined that in Colorado, limber and ponderosa pine stands were used primarily as seed resources. Clason (2017) suggests that the northern range of whitebark pine is limited indirectly by Douglas-fir; models predicted that sustainable local nutcracker occupancy, and hence availability for whitebark pine seed dispersal at this northern edge, is only possible when Douglas-fir is present to provide an alternative food source. Health of higher elevation

whitebark pine as well as lower elevation ponderosa pine and/or Douglas-fir stands adjacent to high-elevation whitebark pine habitat may be essential to maintaining a stable nutcracker population. Whitebark pine, Douglas-fir, and ponderosa pine habitat are all present in the Park and common just east of the Park; stands both inside and adjacent to the Park should help support the nutcrackers that use Park habitat.

Diet

Nutcrackers' primary food is conifer seeds, and seed sources vary geographically (Tomback 1998). In Washington, nutcrackers primarily forage on whitebark pine, ponderosa pine, Douglas-fir and western white pine, with whitebark pine and Douglas-fir being the primary food sources in the Park (USFS Region 6 2019, Ray et al. 2020). Nutcrackers rapidly switch food sources based on availability and seed maturation and move over the larger landscape tracking food sources (Tomback 1978, Vander Wall et al. 1981, Vander Wall 1988, Lorenz and Sullivan 2009, Schaming 2016b, Williams et al. 2020). Ponderosa pine and Douglas-fir seeds in stands to the east of the Park likely help support the nutcracker populations that use the Park (Ray et al. 2020). Nutcrackers also forage heavily on invertebrates, and are opportunistic, eating small mammals, carrion, nestlings and eggs, male cones and buds (Giuntoli and Mewaldt 1978, See Tomback 1998, Schaming 2016b).

Nutcrackers forage on immature seeds, then when seeds mature, both eat and cache seeds for later retrieval (Hutchins and Lanner 1982, Tomback 1982). Cache size ranges from 1–24 seeds, and are placed both above and below ground, in both dense treed and open areas (Hutchins and Lanner 1982, Tomback 1982, Dimmick 1993, Lorenz et al. 2011). Many caches are placed within harvest stands, but in Washington, nutcrackers have been observed caching seeds up to 32.6 km (20 mi) from the harvest location (Hutchins and Lanner 1982, Dimmick 1993, Lorenz et al. 2011). Nutcrackers in the Cascade Range cache most seeds centrally within home ranges, selecting low-elevation forests for caching (Lorenz et al. 2011).

Reproduction

Few nutcracker nests have been located and monitored (Schaming 2015). Nutcrackers nest from mid- to late-January through mid-June, depending on the location (Tomback 1976, Campbell et al. 1997, Schaming 2015). In Washington, they have been documented breeding mid-February through May (Lorenz 2009). Nutcrackers are unusual as they feed nestlings seeds cached the previous autumn, allowing them to begin breeding before insects emerge in the spring, even when there is deep snowpack (Mewaldt 1956, Schaming 2015). Nutcrackers exhibit population-wide failure to breed, and nonbreeding years followed low whitebark pine cone crops in the Greater Yellowstone Ecosystem (Schaming 2015). It is unknown how common it is for population-wide nonbreeding in different ecosystems, or whether the frequency of years is changing as habitat declines.

Nutcrackers lay 2–6 eggs, with 2–4 most typical (Schaming personal observation, see Tomback 1998). They will renest after nest failure, but only raise one successful brood, likely because parents continue feeding young until late summer (Schaming personal observation, see Tomback 1998). There is no evidence of territoriality during the breeding season. (Lorenz 2009 final report, Schaming 2016a).

Protective Status

As of 2018, the International Union for the Conservation of Nature (IUCN) assessed the global status of Clark's nutcrackers to be of least concern because of their large range, large population, and the population trend appearing stable (BirdLife International 2018a). NatureServe lists nutcrackers as globally secure, but status ranges from imperiled to secure depending on the state, and is "apparently secure" in Washington, with the last review in 2016. Both whitebark pine and nutcrackers are monitored by NPS as part of the long-term "Vital Signs" program used for identifying trends in natural resources on NPS lands (Fancy et al. 2009). The NPS has designated the Clark's nutcracker as a Management Priority species within the Park.

Evidence suggests that local nutcracker populations are declining in multiple parts of their range, primarily attributed to the widespread decline of whitebark pine and limber pine (*Pinus flexilis*) habitats (Tomback et al. 2001, McKinney et al. 2009, Barringer et al. 2012, Ray et al. 2020). However, despite the high mortality of whitebark pines in the North Cascades Ecosystem, nutcrackers are still relatively common in the region's subalpine ecosystems (Ray et al. 2017, 2020, Taracka 2021). There is evidence, based on annual bird surveys, that the nutcracker-whitebark pine mutualism is vulnerable to disruption in the Sierra-Cascades region (Ray et al. 2020). Nutcrackers have declined in Mount Rainier National Park, corresponding with significant increases in whitebark pine crown mortality and blister rust infections in the Park, indicating either a true population decline or a shift in local range (Ray et al. 2020). On the other hand, nutcracker counts were highly variable in North Cascades NPS Complex (Ray et al. 2020).

Known Threats

Habitat loss and alteration

Whitebark pine, a preferred and important nutcracker food source, is rapidly declining throughout its range due to widespread and increasing infection by non-native white pine blister rust (*Cronartium ribicola*), mountain pine beetle outbreaks (*Dendroctonus ponderosae*), decades of fire suppression, and the interaction of these factors with climate change (Tomback et al. 2001, Gibson 2006, Tomback and Achuff 2010, Keane et al. 2017). Blister rust is the primary direct cause of whitebark pine decline in the Park and Mount Rainier National Park, with 65% and 82% of whitebark pines infected, respectively, in 2016 (Rochefort et al. 2018). Between 2004 and 2015, 21% and 43% of the monitored whitebark pines died in the two parks, respectively (Rochefort et al. 2018). Evidence suggests the decline of whitebark pine is leading to declines in local nutcracker populations (Tomback et al. 2001, McKinney et al. 2009, Barringer et al. 2012, Ray et al. 2020), and, due to the obligate dependence of whitebark pine on nutcrackers, lower nutcracker density could decrease whitebark recruitment, leading to a positive feedback loop and local declines in both species.

Climate Change

Climate change is threatening whitebark pine directly through potential effects on growth, mortality and regeneration, and indirectly through increasing frequency, intensity and duration of blister rust infections, mountain pine beetle outbreaks, and wildfires (Romme and Turner 1991, Logan and Powell 2001, Running 2006). Loss of whitebark pine, as well as other habitat due to wildfires and

insect outbreaks, decreases food sources and habitat required by nutcrackers. Climate change could also shift the availability and diversity in insects, an important nutcracker food.

Conservation and Research Needs

Information on nutcracker population status, habitat use, habitat selection, foraging behavior, and reproductive success, particularly as associated with variation in habitat characteristics (e.g., whitebark pine health, cone crop, local and landscape density, and co-occurring conifer species) is needed to better assess stability of the local nutcracker population, and to inform management strategies in the Park and throughout the North Cascades Ecosystem. Ongoing Clark's nutcracker surveys in the Park, in whitebark pine, ponderosa pine and Douglas-fir stands adjacent to the Park, and throughout the North Cascades Ecosystem, would allow for a better understanding of the annual variation in regional nutcracker population density and local habitat drivers of occurrence. Surveys in the Park and adjacent forests would enable understanding of whether seasonal migration is occurring, and hence a better understanding of the connection between the Park and other public and private lands, as well as the level of importance of these adjacent forests to the heath of nutcrackers in the Park.

Surveys can be conducted in-person or remotely through the use of passive acoustic recording devices. Current bird point count surveys are conducted between late May and late July (Ray et al. 2020), but during the spring there is low nutcracker detectability (Schaming and Sutherland 2020). Nutcracker detectability increases through summer and fall (Schaming and Sutherland 2020), and additional surveys later in the year would increase our understanding of nutcracker use of the Park. Nutcrackers detected during the spring breeding season would document the possibility of a breeding population in the Park. Surveys in July document nutcrackers before emigrants inflate or deflate the population, and additional surveys during the fall seed harvest season would allow a better understanding of the use of seed resources within the Park, as well as the potential for seed dispersal when whitebark pine seeds are mature (Lorenz and Sullivan 2010, Schaming and Sutherland 2020). There are limitations to survey data due to nutcrackers' large range, the varying detectability of nutcrackers, and the difficulties in quantifying abundance for breeding bird populations, but these surveys would provide a useful baseline and measure of variability between years, allowing for assessing whether there is a stable population of nutcrackers in the Park, detection of possible population declines, and to help in reevaluating the relationship between nutcrackers and habitat metrics when the environment changes and management strategies are implemented.

A more detailed investigation of the local spatial extent over which habitat metrics influence nutcracker occurrence is necessary to determine the scale at which to implement management strategies (McKinney et al. 2009, Barringer et al. 2012, Schaming and Sutherland 2020). Satellite-tracking Clark's nutcrackers in the Park would provide information on where nutcrackers go when they move out of the Park, the proportion of time they use the Park versus other lands, and which habitat types they are using within the Park. Such tracking would also allow for documentation of the birds' foraging microhabitat, movement patterns, habitat selection, and reproduction presence and success. Additionally, it is unknown how human disturbance affects nutcracker behavior, particularly whether winter recreational activities affect nesting behavior (Tomback 1998).

Fine scale mapping of whitebark pine in the North Cascades Ecosystem is needed, and accurate, ground-truthed whitebark pine maps in the region would be highly valuable. Whitebark pine trees are only surveyed intermittently in the Park (Ray et al. 2020), and systematic, regular, largescale whitebark pine and ponderosa pine health, mortality and cone crop surveys would allow for more accurate predictions of the vulnerability of Park nutcrackers, and a better assessment of what is driving fluctuations of nutcracker populations in the Park. Because Park nutcrackers likely use adjacent forests, these surveys should also cover nearby U.S. Forest Service and Washington Department of Natural Resources lands. In addition, it is unknown if nutcrackers will remain in the North Cascades Ecosystem if whitebark pine declines, or if the availability of ponderosa pine stands is adequate to maintain the species. Models evaluating the likelihood of nutcrackers abandoning the region as a function of habitat should be considered.

Finally, whitebark pine restoration is likely an essential element of long-term nutcracker conservation. Assessing condition of whitebark stands; reducing risk of fire and mountain pine beetle attacks; gathering seeds; growing seedlings, running blister rust screening trials, then planting seedlings; identifying blister-rust resistant “plus” trees; implementing silvicultural treatments to reduce competition and fire risk for trees; and monitoring the efficacy of the restoration actions long-term are possible adaptive management measures (Keane et al. 2012).

Golden eagle (*Aquila chrysaetos*)

The golden eagle (*Aquila chrysaetos*) is a large, brown raptor that ranges across most of the northern hemisphere (Figure 8). Of six subspecies worldwide, *A. c. canadensis* occurs throughout North America (Katzner et al. 2020). Both male and female adults are completely dark brown apart from their golden nape that extends around the sides of the neck. Their broad wings that stretch up to 2.1 m allow them to soar and glide for extended lengths of time, and thereby use minimal energy to maintain altitude in updrafts or to travel great distances (Katzner et al. 2020). Golden eagles are an opportunistic predator, feeding mainly on small to medium sized mammals and carrion, but will occasionally feed on birds, eggs, and fish. Golden eagles utilize a variety of habitats for breeding including montane meadows, grasslands, shrub steppe, and conifer forests, but generally avoid densely forested areas. Nesting is generally on cliff ledges with access to open areas for foraging (Kochert et al. 2002).



Figure 8. Golden eagle voucher record NOCA39365, captured on a remote wildlife camera (operated by Cascades Carnivore Project, National Park Service, and Washington State University), August 12, 2018, near Purple Pass, Lake Chelan National Recreation Area, WA.

Golden eagles are uncommon to scarce across their range. Populations exhibit a wide variety of migration patterns, from non-migratory to long-distance migratory (Katzner et al. 2020). The non-breeding distribution in North America includes Mexico, all U.S. states, and southern areas of Canada, although it most commonly occurs in western North America. The breeding range encompasses most of Canada and Alaska, the western continental U.S., and parts of northern and

western Mexico. Golden eagles breeding in the highest latitudes of northern Canada and interior and northern Alaska tend to migrate south to wintering grounds, as far away as Mexico. In western North America, individuals nesting south of 55°N are generally less migratory than those nesting north of this latitude (Katzner et al. 2020).

BirdLife International (2021a) estimates a total global population of 100,000–200,000 golden eagles. The IUCN has designated the global population as least concern because of their expansive range and stable population trend. The population in the U.S. is thought to be around 40,000 individuals (USFWS 2016a), with approximately 27,400 individuals occurring in the western U.S. (Good et al. 2007). Recent data suggest that the population in western North America is stable or declining overall (Millsap et al. 2013, USFWS 2016a, Partners in Flight 2020). Count data from some populations in the western states point to long-term declines, whereas populations at northern latitudes (including western Canada) appear to be more stable (McIntyre 2012, NABCIC 2019). The long-lived species naturally has a low reproductive output, so it is likely that anthropogenic stressors (e.g., loss and degradation of habitat, decreases in prey availability) will trigger further declines (Nielson et al. 2016).

This species occurs throughout much of Washington state, but the population size is low (60 known breeding pairs counted in 2005; WDFW 2012a, 2022). It is most common east of the Cascade Crest, especially in the Okanogan highlands at the transition between montane and shrub-steppe ecosystems. Of the 300 documented breeding territories in Washington, over 80% are east of the Cascades. Because of the prevalence of closed-canopy coniferous forest in western Washington, they are thought to be uncommon to rare in this region (Singh et al. 2016, WDFW 2012a, 2022), although 46 territories were identified in 2017 (Hansen 2017). A recent study by Watson and others (2020) provided evidence for depressed reproduction of golden eagles in Washington state, which the authors attributed to declines in preferred prey species (leporids and sciurids). Their analysis of surveys conducted between 1977 and 2014 at 301 known territories across the state showed low occupancy in both eastern (51%) and western Washington (45%) compared to occupancy in Oregon, Utah, and Alaska, in addition to a 22% decline in nesting success in eastern Washington during that period.

Although the Park falls within potential breeding range for golden eagles, they are not frequently seen, with only 35 records from 1995–2020 (Figure 9, Table 5). Only one potential nest was observed, in 2018. In the Park, golden eagles are more likely to occur on the eastern slope of the Cascade Range, in drier forests with more open habitat for foraging. Observations in the Park are too sparse to infer trends in local abundance, and with a wide distribution and large home range (up to 250 km^2/pair), the Park does not represent a significant portion of the suitable habitat available to golden eagles in Washington (Kochert et al. 2002, Hoffman et al. 2015). Historic observations in the Park have occurred primarily in September, which indicates the importance of the area as an autumn migration route.

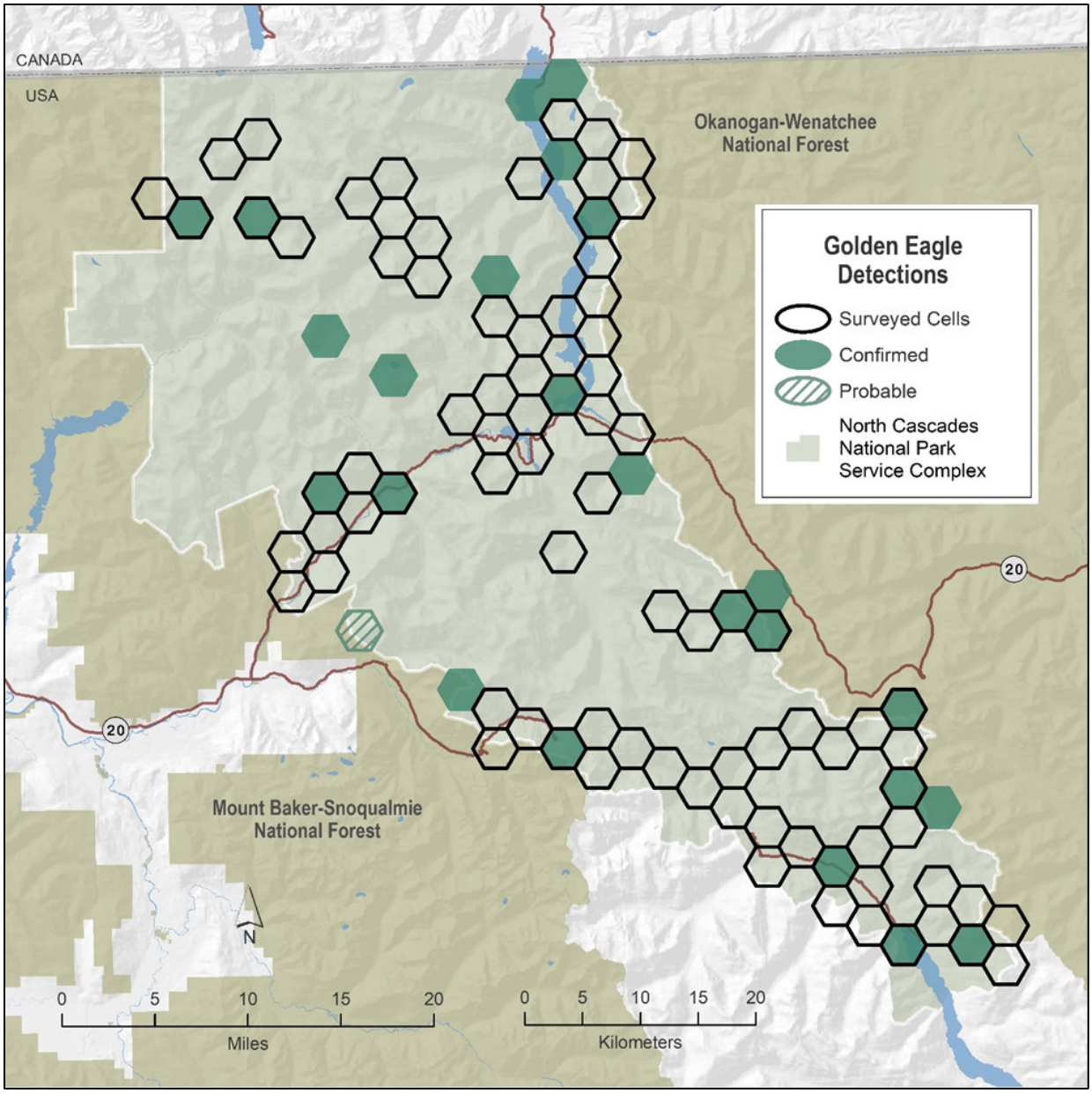


Figure 9. Survey locations, and Golden eagle (*Aquila chrysaetos*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 5). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 5. Golden eagle (*Aquila chrysaetos*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2005–2020	Landbirds point counts and incidental detections at 1,222 points (visual and acoustic observations) [a]	Annual or alternate year transects, May–July	Holmgren et al. 2015–2017, 2019–2021; Ray et al. 2017, 2018; NPS unpublished data

[a] Surveys were not designed for optimal detection of this species.

Habitat Requirements

Golden eagles generally associate with steep, cliffy terrain, high alpine zones, arid plateaus and canyonlands, dry open forests, shrub-steppe and grassland communities as well as the transition zones to forested habitat (Marzluff et al. 1997, WDFW 2022). In western Washington, territories have been found above treeline and in mature and old-growth forests that were no more than 500 m from open fields or expansive, recent clearcuts (Bruce et al. 1982). These areas offer three key features required by golden eagles: open foraging areas with ample prey, protection of nests from predators and weather, and topography that creates thermal or orographic updrafts that are essential for efficient flight (Katzner et al. 2020).

Golden eagles usually forage in various types of open habitat including shrub- and grasslands, meadows, talus fields, rock outcrops, recently burned areas, clearcuts, and wetlands where medium-sized birds and mammals can be more easily spotted and attacked (Marzluff et al. 1997, WDFW 2022). Nielson and others (2016) found strong positive relationships between occupancy in the western U.S. and elevation, wind speed, and solar radiation, and a strong negative relationship with the extent of forest cover. Recent telemetry data from the eastern U.S., however, have shown the use of forested landscapes is more common than previously recognized (Miller et al. 2017). In western Washington, golden eagle occupancy is positively associated with elevation and topographic relief and negatively associated with high canopy closure. These relationships likely reflect the avoidance of human development at lower elevations and their association with clearcuts and open fields where they can more easily hunt (Hansen 2017). Golden eagles generally avoid urban areas, agricultural areas with high levels of habitat fragmentation, and high human population density (Fischer et al. 1984, Craig et al. 1986, Craig et al. 2018).

Their large stick nests are usually built on cliff ledges or rocky outcrops that offer a wide view of the surrounding landscape as well as localized updrafts to subsidize flight from the nest (Beecham 1970, Kochert et al. 2002). Proximity to foraging areas, protection from weather and extreme temperatures, and elevation are also known determinants of nest site suitability (Camenzind 1969, Morneau et al. 1994). Depending on the availability of these features, golden eagles also nest in large trees, such as cottonwood (*Populus* spp.) and Douglas-fir, or human-made structures such as transmission towers and power poles. In eastern Washington, 87% of all surveyed golden eagle nests were on rocky substrates (e.g., steep cliffs, outcrops, talus slopes) and 13% were in trees (Watson et al. 2014). In western Washington, nests are more commonly placed in large trees that border clearcuts or on

various substrates above treeline (Bruce et al. 1982). Golden eagles nest annually and alternate between 2–3 different nests in their territory (Watson and Whalen 2003).

Diet

Golden eagles are considered generalists and opportunistic predators. The large raptor is a highly effective predator and is known to kill or scavenge on large animals such as mountain goat and bighorn sheep (*O. canadensis;* Zettergreen 2006, Katzner et al. 2020), but subsists primarily on medium-sized prey between 0.5 and 4.0 kg (Bedrosian et al. 2017). A meta-analysis of golden eagle diets across western North America showed that 84% of their prey were medium-sized mammals, 15% were birds, and the remainder were reptiles, fish and insects. At this geographic scale, leporids (jackrabbits, other hares, or cottontail rabbits) were the primary mammalian prey, followed by sciurids (ground squirrels, marmots, or prairie dogs). Avian prey are most often gallinaceous birds (grouse, ptarmigan, pheasants, and partridge) and waterfowl (Bedrosian et al. 2017). Prey outside the range of 0.5–4.0 kg are energetically expensive to hunt, so the availability of optimal-sized prey is a key factor influencing foraging behavior, reproductive success, and distribution of golden eagles (Steenhof and Kochert 1988, Moss et al. 2012, McIntyre and Schmidt 2012, Schweiger et al. 2014)

In eastern Washington, golden eagles consume yellow-bellied marmots (*Marmota flaviventris*) at high frequencies in addition to ring-necked pheasant (*Phasianus colchicus*), Chukar (*Alectoris chukar*), rock doves (*Columba livia*), and carrion (Marr and Knight 1983). In western Washington below 1,219 m (4,000 ft) elevation, the most common prey for nesting golden eagles appears to be mountain beaver (*Aplodontia rufa*), sooty grouse (*Dendragapus fuliginosus*), and small mammals (Hansen 2017). In this region where canopy closure is relatively high, golden eagles rely on open fields and early-seral forest stands where mountain beaver and grouse occur at higher abundances (Neal and Borrecco 1981, Hansen 2017). At higher elevations within the Park, hoary marmot (*Marmota caligata*), Columbian ground squirrel (*Urocitellus columbianus*), snowshoe hare (*Lepus americanus*), and grouse species are likely consumed by nesting golden eagles.

Reproduction

Golden eagles are long-lived, generally monogamous, and reproduce slowly (i.e., *k*-selected). The birds are capable of reproducing before adulthood at age five but do so at low frequencies in most populations. The timing and period of courtship, nest selection, and nest refurbishment may be different for resident and migratory golden eagles, with residents often taking longer to prepare for egg laying (Katzner et al. 2020). In Washington, egg laying occurs between mid-February and mid-April (Hayes 2015) but the timeframe is highly dependent upon prey abundance and local weather severity (Steenhof et al. 1997). The average clutch size in the western U.S. is two eggs (n = 332, range = 1–3 eggs; Katzner et al. 2020). The incubation period is usually about 42 days (Katzner et al. 2020), during which the males often relieve females of incubation duty for two short periods per day (Collopy 1984). In Washington, eggs usually hatch between late-March and late-May (Hayes 2015) and chicks remain in the nest for an average of 64 days (range = 45–77 days; Steenhof et al. 2017). Sixty-three percent of nests in western North America produced just one fledgling, 35% produced two fledglings, and only 2% produced three fledglings (Katzner et al. 2020).

Nesting success is highly dependent upon the combination of preferred prey abundance and weather conditions (Steenhof et al. 2017), as it is with many bird species. In years of low prey abundance, pairs often forego reproduction because female body mass and energy reserves are not sufficient to produce eggs. Likewise, cold weather prior to nesting may preclude reproduction altogether or may reduce the probability of nesting success if eggs are laid (Tjernberg 1983, Steenhof et al. 1997, McIntyre and Schmidt 2012). In both southwest Idaho and interior Alaska, the annual reproductive output was influenced most strongly by the percentage of pairs that initiated nesting compared to nest failure rates and brood sizes (Steenhof et al. 1997, McIntyre and Schmidt 2012, Katzner et al. 2020).

Protective Status

Although golden eagles are not federally listed under the ESA, the U.S. provides federal protection under the MBTA and the Bald and Golden Eagle Protection Act. Both laws prohibit the take (which includes disturbance), possession, and handling of eagle parts, nests, and eggs without a USFWS permit. A small number of permits may be issued to take eagles for falconry or for the religious purposes of Native Americans (USFWS 2016a, 2016b). Golden eagles are also legally protected by most U.S. states in which they occur (Katzner et al. 2020). In 1991, concerns about the small population in Washington prompted classification as a state candidate species for possible listing according to Washington Administrative Code 232-12-297. It is also considered a Species of Greatest Conservation Need under the Washington State Wildlife Action Plan (WDFW 2022). Golden eagles are a Management Priority species in the Park.

Known Threats

Human-caused mortality

Various anthropogenic sources account for the majority of reported golden eagle deaths in the U.S. Humans were responsible for >70% of reported deaths, with accidental collisions accounting for the most mortality (27%), followed by electrocution (25%), poaching (15%) and lead poisoning from gun-injured prey consumption (6%) (Franson et al. 1995). Historically, shooting of golden eagles was a major source of mortality, as they were implicated in predation of domestic sheep. Illegal shooting is still occurring at unknown levels. Because adult survival is the most important demographic variable for population stability in golden eagles, anthropogenic mortality is highly relevant to population status and trend (Millsap et al. 2022).

Recreation and other types of human activities near nests have been identified as a significant threat to golden eagle territory occupancy and reproduction, with the potential to impart population-level effects (Kochert et al. 2002, Davis 2019). Studies from across the globe have demonstrated lower occupancy and breeding success in golden eagle territories where both motorized and non-motorized human disturbances occurred at higher rates (Steenhof et al. 2014, Spaul and Heath 2016). Non-motorized recreation near nests, such as camping and skiing, have been associated with reduced territory occupancy and parental care of nestlings (Steidl et al. 1993, Kaisanlahti-Jokimăki et al. 2008). Two studies in western North America documented that human disturbance caused 46–85% of golden eagle nesting failures (Camenzind 1969, Boeker and Ray 1971). Conversely, helicopter flybys between 100–800 m distances from cliff nests only evoked disturbance responses from 2% of nesting golden eagles (Grubb et al. 2010).

Exposures to crop pesticides, anticoagulant rodenticides (AR), mercury, and lead from ammunition have all been detected in eagles and implicated in eagle mortality (Kochert et al. 2002). Use of organochloride pesticides (e.g., DDT) during the mid-20th century caused major declines in several raptor species, but golden eagles in western North America were not as widely exposed given their diet of mainly terrestrial mammals rather than birds and fish (Kochert 1972, Reidinger and Crabtree 1974). Lead toxicosis was the second most important cause of mortality of golden eagles recovered in Washington between 2000–2013 (17.5% of deaths), and lead exposure is expected to be widespread in the state as a result of killing or scavenging gun-wounded prey (Watson and Davies 2015). Eleven of 16 golden eagles from the Columbia Basin also had elevated levels of lead and most had background levels of mercury in their blood samples, although not enough to express clinical signs of poisoning (Watson and Davies 2015). Eagles are also commonly exposed to AR compounds used to kill rodents in the U.S. Seventy-seven percent (n = 13) of golden eagle livers tested for AR showed signs of exposure; however, only one of these individuals had fatally high levels and it appears that AR toxicosis is generally rare in eagles (Niedringhaus et al. 2021).

Golden eagles are the most frequently electrocuted raptor in the U.S. (Lehman et al. 2007), and this is believed to be one of the primary causes of anthropogenic mortality, killing an estimated 504 golden eagles annually (USFWS 2016a). In a study of raptor electrocutions in the rural west, golden eagles were electrocuted 2.3 times more frequently than bald eagles and were the most frequently electrocuted raptor species (Harness and Wilson 2001). Young eagles are disproportionately susceptible to electrocutions, with 66% of incidents involving juveniles or sub-adults. The structures that were most commonly associated with eagle mortalities were banks of 1- and 3-phase transformers, which are only present on a small fraction of power poles and indicate transformers should be a significant focus for reducing eagle electrocutions (Harness and Wilson 2001). In the 1980s, the Avian Powerline Interaction Committee was formed to investigate the effectiveness of collisions and electrocution mitigation methods (APLIC 1996). There have been cooperative efforts between power companies, biologists, and government agencies to promote and implement raptor safe designs and retrofits in power supply infrastructure. Collisions with vehicles, fences, and wind turbines are also a significant source of anthropogenic mortality (USFWS 2016a).

Habitat loss and alteration

Declines of some golden eagle populations in the western U.S. have been attributed to the reduction in preferred prey species from habitat loss and conversion (Kochert and Steenhof 2002). In the Park, climate change has the potential to affect golden eagle reproduction and adult survival by altering montane communities that include high-quality prey species (i.e., hoary marmot, snowshoe hare). Observed abundance of hoary marmots within the Park has declined by 74% from 2007–2016, likely due to reduced winter snowpack that insulates burrows during hibernation (Christophersen 2012, Johnston et al. 2020). Snowpack in Washington has declined by 30% since 1955 and is expected to lose another 38–46% by 2050 (Roop et al. 2019). Climate change is also expected to increase weather severity and local temperatures that affect prey populations as well as nest microclimates that ultimately reduce nest success and brood survival (Steenhof et al. 1997, Wiens et al. 2018, Kochert et al. 2019, Watson et al. 2020).

The growing frequency of high-intensity wildfires also has the potential to affect golden eagle reproduction and adult survival by destroying nesting and foraging habitat (Kochert et al. 1999). For example, severe fires that occurred in eastern Washington in 2014 and 2015 destroyed approximately 45 golden eagle territories (Watson et al. 2020). Conversely, severe fires that destroy forest stands may only have a short-term adverse effect on golden eagle foraging habitat because the early seral landscape would likely increase prey abundance and hunting opportunities for golden eagles (Marzluff et al. 1997, Parkins et al. 2018).

Conservation and research needs

Information on golden eagle population status, movements, resource selection, and exposure to contaminants is important for the adaptive management of Park resources. Despite the considerable body of research on this species, there are still substantive gaps in our understanding of how golden eagles will respond to climate change, increased human activity, energy development, changing environmental contaminants, and wildland fire (Katzner et al. 2020). In the Park, most golden eagles transit the ecosystem during autumn migration, and it is unknown how frequently these migrants hunt at low altitudes around the Skagit River Project transmission line corridor; however, Seattle City Light (2022) found no evidence of collision or electrocution in a 2021 study of golden eagles.

Specific conservation needs for golden eagles in the Park include monitoring raptors during migration, and mitigations for low altitude aircraft use near potential nest habitat in summer, and around mountain passes in late summer through autumn (Appendix A). Anti-strike devices on electrical transmission lines in migratory routes are warranted because with such closed canopy forest on the west slope of the Cascade Crest, artificial openings such as a transmission right-of-way may be selected by golden eagles as the only available hunting habitat at low elevations. Routine collision and electrocution monitoring should be conducted along transmission line corridors. Eagle carcasses should also be tested for West Nile virus, avian influenza, and other emerging diseases, as well as environmental contaminants (especially lead and ARs). Any recovered carcasses should also be genetically sampled to identify relatedness to other eagles, which may help better define migration routes (Kochert and Steenhof 2002).

Harlequin duck (*Histrionicus histrionicus*)

The harlequin duck (*Histrionicus histrionicus*) is a sea duck known for its striking plumage and unusual breeding ecology. Contemporary presence and breeding of harlequin ducks in the Park has been confirmed (Figure 10). Harlequins are highly social, often congregating in groups and mixed flocks on saltwater wintering grounds where they forage near shore on intertidal invertebrates, fish, and roe. Two disjunct populations of harlequin ducks inhabit the Pacific and Atlantic Coasts. While these populations appear isolated, no subspecies are currently recognized. The Pacific population can be found in western North America from Oregon to Alaska along coastal habitats during wintering seasons, breeding inland along the same latitudinal range as far east as Montana and Wyoming. The Pacific population can also be found along coastlines in eastern Asia. The Atlantic population breeds in eastern Canada, Iceland, and southern Greenland, wintering along the coast of Iceland, southern Greenland, and from southeastern Canada through New England (Robertson and Goudie 1999). Washington populations can be found in the Olympic, Cascade, and Selkirk Mountains. Historically, populations were observed in the Blue Mountains but it is unknown if they persist there (Schirato 1994).

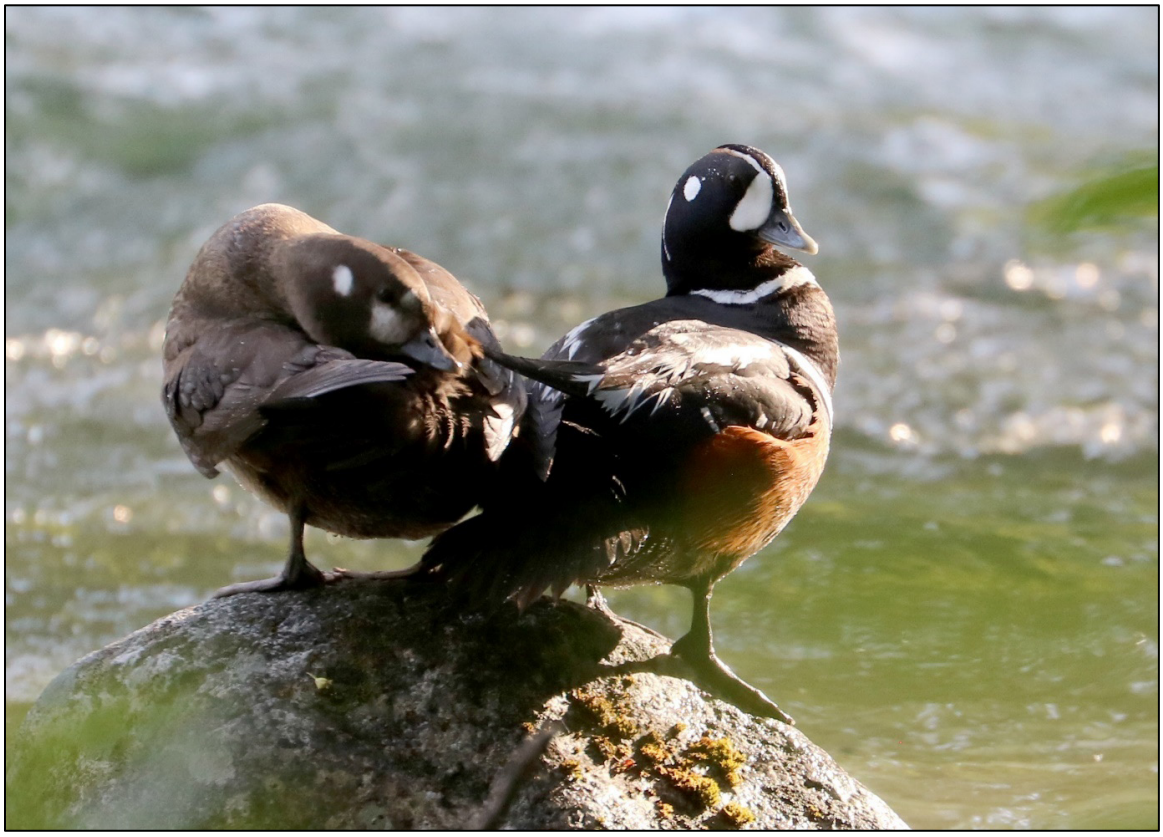


Figure 10. Harlequin duck voucher record NOCA24168, photographed by Jason Ransom/NPS, May 21, 2017, on the Stehekin River, Lake Chelan National Recreation Area, WA.

While the global population of harlequin ducks is apparently stable (IUCN 2019), the Atlantic population (1,500–2,000 individuals) has experienced an overall decline since the late 1970s

(USFWS 1998a) due to numerous factors such as habitat degradation in breeding and wintering areas. Some Atlantic populations were stabilizing in the late 1990s (i.e., Maine, USFWS 1998a). Estimates for the Pacific population at that time were comparatively robust (est. 1 million individuals; USFWS 1998a), with the majority of individuals inhabiting the coastal waters of Alaska. Population estimates decrease with latitude (USFWS 1998a). Previous estimates of Washington's harlequin duck population have indicated there may be an overall population size between 1,500 (Schirato and Sharpe 1992) to 3,000 individuals and 300–400 breeding pairs (Cassirer et al. 1996). Surveys of wintering populations in Washington between 1993 and 2002 indicated that densities were stable region-wide, but that a significant decline was observed for the northern Puget Sound (Whidbey and Camano Islands) cohort during this period (Nysewander et al. 2005). Because data on breeding grounds are lacking, population estimates of wintering birds may be a better gauge of overall population trends in the Pacific Northwest. Based on Christmas Bird Count data from the National Audubon Society (National Audubon Society 2002) in coastal Oregon and Washington, there has been an overall increase in population since 1955, but a significant decrease since 1985 (Wiggins 2005).

Harlequin ducks have been present as breeders on many streams in the Park, but relative abundance across the Park's waterways is unknown (Figure 11, Table 6). In 1990, NPS resource management staff initiated presence-only pair and brood surveys of the lower 12-mile portion of the Stehekin River. Over the next 15 years, surveys on this and 12 other potential breeding streams continued; however, survey efforts, methods, and detectability were heterogeneous and therefore have limited inference for abundance estimates and other statistical analyses. Surveys on the Stehekin River accounted for the highest instantaneous count of pairs (n = 16; 1993) and broods (n = 4; 1990 and 1998) (Rine et al. 2022). In total, harlequin ducks have been observed on 22 separate 2nd to 6th order streams between 1968 and 2017, including 8 of the 13 stream reaches that were actively surveyed by Park staff (Figure 11) (Rine et al. 2022). They were not detected during surveys or incidentally on Stetattle Creek, Lightning Creek, Panther Creek, Bacon Creek, or the North Fork of Cascade River. Nest locations along any Park stream have yet to be identified. Due to their use of fast-flowing rivers and streams, sensitivity to human disturbance, and limited human access to most Park stream reaches, harlequin ducks are not commonly observed in their breeding habitat.

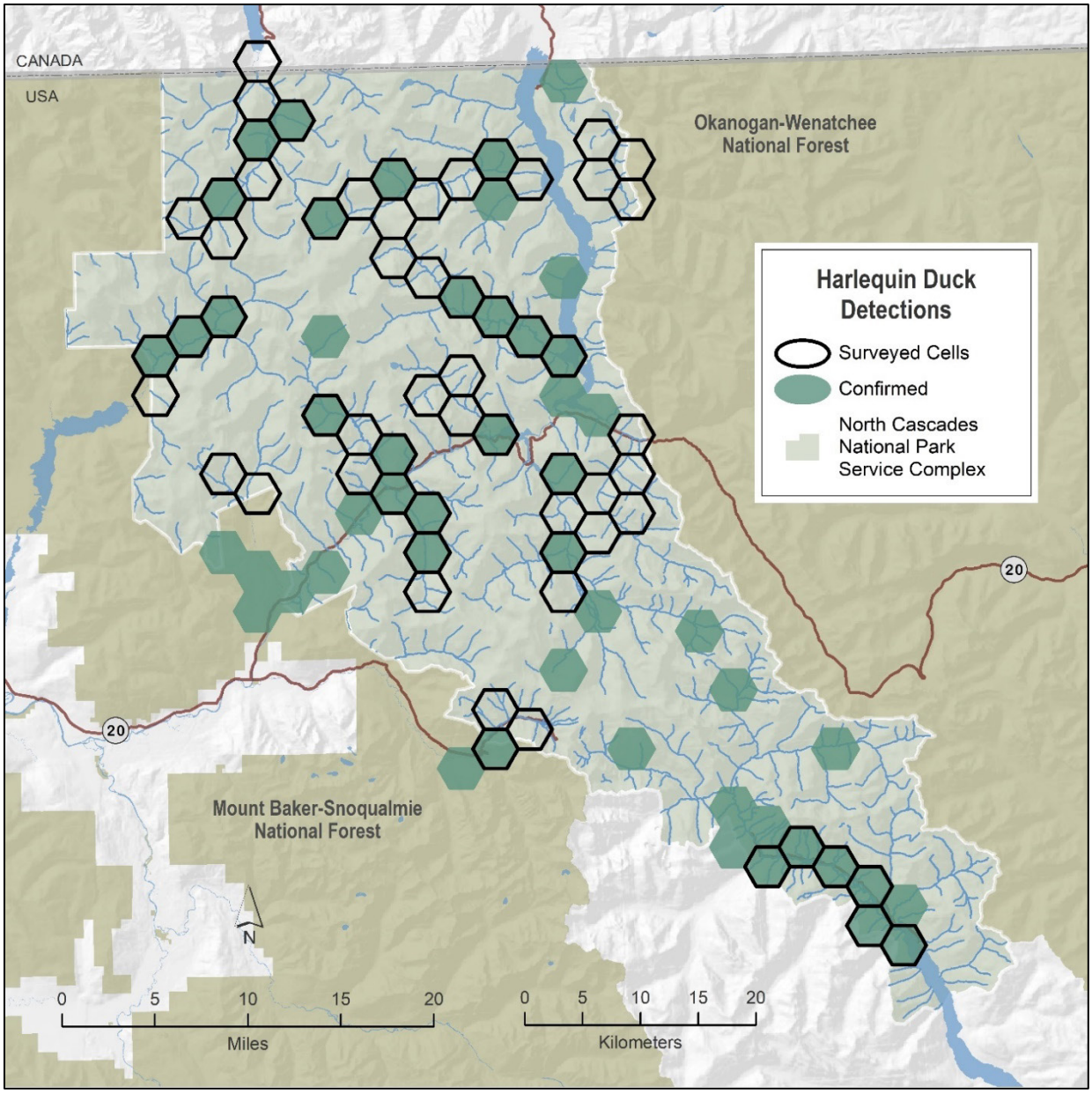


Figure 11. Survey locations, and Harlequin Duck (*Histrionicus histrionicus*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 6). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 6. Harlequin Duck (*Histrionicus histrionicus*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
1995–2017	River course transects (visual observations), wading or on shore at 13 sites	1–2 surveys each site during a given survey year	Rine et al. 2022
2016–2018	eDNA water sampling at 32 points	1–3 samples per point	NPS unpublished data

Habitat Requirements

Non-breeding harlequin ducks are typically observed in shallow, intertidal habitats close to shore (Crowley 1994, Esler et al. 2000) where they dive to forage on benthic invertebrates. Habitat features associated with harlequin duck presence on the Pacific coast include an offshore reef within 500 m and a stream within 200 m (Esler et al. 2000). The wintering demographic is also associated with certain substrate types in large intertidal areas that include gravel, cobble, and bedrock (Rodway et al. 2003) where their principal invertebrate food items are found (Gaines and Fitzner 1987).

Breeding harlequin ducks can utilize either coastal estuarine sites or 1st–5th order inland mountain streams (Bruner 1997, Robertson and Goudie 1999). Research has identified a number of specific habitat features that may be used to predict harlequin duck occurrence, but their importance varies somewhat across studies. A study in the Oregon Cascades showed that bedrock substrate was the only feature significantly and positively correlated to harlequin densities, while other variables like habitat type, stream velocity, water depth, and substrate particle size were not important in the model (Wright et al. 2000). Conversely, an analysis of large-scale habitat characteristics across breeding stream reaches in Washington revealed that elevation, stream gradient, summer stream flow, proportion of surrounding area with conifer forest, and landform characteristics were most effective at determining occupancy (Singleton and Long 2018). Specifically, stream reaches at moderate elevations and those with a higher proportion of gentle gradients (<5%) and mean historic summer flows between 158 and 695 cubic feet per second increased the probability of harlequin duck detections (Singleton and Long 2018). Harlequin duck productivity has shown a negative relationship with stream flow levels, perhaps because high flows disturb or destroy nests, and because rearing young in rougher water may be difficult (Cassirer and Groves 1994, Crowley 1994).

Nest sites of the Pacific population are less well-characterized than those of the Atlantic, but have been observed directly on the ground, on rocks, in cliff cavities, and in woody debris piles (Bent 1925, Jewett 1931, Campbell et al. 1990, Thompson et al. 1993). One nest on the Olympic Peninsula was located in the cavity of a big leaf maple (*Acer macrophylum*; Cassirer et al. 1993), and another was observed within a woody debris cavity adjacent to an anastomosing river within Mount Rainier National Park (Morris et al., unpublished data). Most nests within the Oregon Cascades are characterized by overhead vegetation within 0.75 m of the nest, regardless of substrate, forest type, and seral stage (Bruner 1997).

After hatching, young harlequin ducks typically require slower moving water such as side channels, oxbows, and beaver ponds (Kuchel 1977) due to their smaller size, high buoyancy, and inexperience foraging in swift water (Crowley 1994). Brood rearing reaches tend to exhibit more stable in-stream and bank features, greater riparian canopy cover, and a higher abundance of invertebrate biomass (Kuchel 1977, Cassirer and Groves 1994, MacCallum et al. 2016). Predator avoidance appears to be an important factor in selecting brood-rearing habitat. The composition of protective habitat features varies geographically and may include those associated with higher flows (i.e., greater channel overhang; Machmer 2001, MacCallum et al. 2016).

Harlequins in the Park were most often observed in fast-flowing riffle and rapid habitat with boulder and cobble substrate, as well as slower pool features (Rine et al. 2022). A mature riparian overstory and vegetative overhang were present at a greater proportion of occupied sites than those with young vegetation. NPS data suggests harlequin ducks may prefer breeding streams with less woody debris (e.g., bridge, collapsed bridge, drift) with the exception of ramps, which tend to be favorable loafing objects (Rine et al. 2022). Similar to results from other studies (Wallen 1987, Robertson and Goudie 1999), most detections of harlequin ducks in Park streams were made at sites further removed from human activities (Rine et al. 2022).

Diet

In shallow coastal wintering habitat, harlequin ducks dive beneath the surface to forage on various macroinvertebrates, amphipods, and occasionally small fish and roe (Dzinbal 1982, Robertson and Goudie 1999). On breeding streams across their Holarctic range, harlequins generally forage on benthic aquatic insects, either consuming a wide range of species or specializing on certain taxa (e.g., Plecoptera, Ephemeroptera, *Dicosmoecus* caddisflies; Bengtson 1972, Wright et al. 2000, McCutchen and Ydenberg 2013). Surveys of benthic aquatic insects in some harlequin breeding streams within the Park have shown high relative abundances of preferred prey taxa such as Ephemeroptera, Plecoptera, and Trichoptera (Rine et al. 2022). Fish consumption on breeding streams is apparently rare but has been observed in the central Cascades of Oregon (Wright et al. 2000).

Reproduction

Pair formation occurs on the wintering grounds starting in October, and by the end of December most females are paired (Robertson et al. 1998). Pair-bonds are long lasting, and pairs can be seen together on both breeding and wintering grounds (Robertson and Goudie 1999). While there can be an excess of males on breeding streams (Bengtson 1966, Cassirer and Groves 1989), sexually mature harlequin ducks commonly arrive on breeding streams in pairs that were previously formed on wintering grounds before upstream migration (Gowans et al. 1997, Robertson et al. 1998). Pairs arrive on breeding grounds in late March through early May (Cassirer and Groves 1991). Females initiate nesting from late April to early June (Kuchel 1977, Diamond and Finnegan 1993, Bruner 1997). Shortly after females begin incubation, males depart and return to coastal saltwater habitats for the duration of the year. This life history trait eliminates opportunities for re-nesting if clutch failure occurs. Mean incubation is 28 days (range 27–29 days, Bengtson 1966). Observed brood sizes on Park streams ranged between one and six, with a mean brood size of 2.83 (± 2.13 SD) ducklings (Rine et al. 2022). In Oregon, average clutch size was 5.2 + 1.2 SD (n = 21; Bruner 1997).

There were an estimated 399 breeding pairs in Washington in 1996 (Cassirer et al. 1996). There do not appear to be breeding chronology records specific to Washington apart from what has been observed on Park streams via active staff surveys and incidental sightings. Seasonal timing of distinct demographic groups observed on Park breeding streams generally align with those in the Oregon Cascade Range (Bruner 1997). Breeding pairs are typically observed on Park streams between late April and late May, with post-breeding and non-breeding males returning to coastal habitat by early June (Rine et al. 2022). Females that either did not mate, or experienced nest failure, remained until mid-July through late August. Successfully nested females remained through late August.

Protective Status

Harlequin ducks are considered one of the most imperiled ducks in North America, despite the fact that they are not a species of concern on a global scale. The global population is listed as a Species of Least Concern by the IUCN and is otherwise considered to be stable (NatureServe 2021). The ducks received relatively little scientific attention before the listing of the Atlantic Canadian population as endangered in 1990. That population is currently listed as a Species of Special Concern, with additional provincial listings as either "endangered" or "vulnerable" (COSEWIC 2013). The listing in Canada spurred investigations into protections for the Pacific and Atlantic populations in the early 1990s. A petition was submitted to the USFWS under the ESA in 1994 to list the congruent Atlantic population as "threatened" or "endangered". Rulings in 1995 and 1998 by Region 5 (Northeast U.S.) of USFWS concluded the ruling was "not warranted" because of lack of immediate threats to population viability. Federal protection under the ESA has yet to be established for any U.S. population. Harlequin ducks are not listed in Washington State but are considered a Species of Greatest Conservation Need in the Washington State Wildlife Action Plan. Breeding populations in Oregon and Washington are listed as "sensitive" by the Interagency Special Status/Sensitive Species Program (ISSSSP). The NPS has designated harlequin ducks as a Management Priority species within the Park.

Known Threats

Conservation concern for harlequin ducks stems from increasing disturbance on both wintering and breeding grounds. Their life-history as long-lived species with delayed reproduction and low recruitment make them vulnerable to population declines and slow recovery from impacts (Goudie et al. 1994). A high level of breeding site fidelity may make them more vulnerable to depressed reproductive success following breeding habitat loss or disturbance (Wiggins 2005). Because harlequin ducks take a number of years to reach maturity, they are slower to rebound from threats; therefore, great care must be taken to prevent a negative impact on the population.

Land Use and Recreation

Timber harvest is likely the human activity with the greatest potential to impact western populations at breeding sites (Breault and Savard 1991). Harlequin ducks are more frequently found in streams within old-growth stands (Cassirer and Groves 1994), and at higher breeding densities in areas that have not been harvested for timber (Freeman and Goudie 1998). Activities that alter riparian nesting habitats and debris load in rivers, such as logging, can impact invertebrate communities and water quality (Robertson and Goudie 1999). Hydroelectric dams may also be of concern as they alter

streamflow characteristics and inundate potential nesting areas, although the impacts of both logging and hydroelectric projects have not been fully investigated (USFWS 1998a). Disturbance from fishing and rafting operations may impact site use in breeding streams, and harlequins have been observed abandoning or avoiding stream and river sections with elevated human use (Robertson and Goudie 1999). Harlequin ducks appear to avoid infrastructure and human traffic and are found more frequently at sites greater than 50 m from trails and roads (Cassirer and Groves 1994). Harlequin ducks are also subject to bioaccumulation of environmental toxins and microplastics because they are higher trophic consumers in aquatic habitats such as reservoirs.

On wintering grounds, harlequin ducks are vulnerable to entanglement in fishing equipment and oil spills (Robertson and Goudie 1999). Oil spills are of particular concern since large numbers of harlequins can be concentrated in small areas, representing significant portions of local populations. Populations impacted by the Exxon Valdez oil spill in 1989 experienced declines, and reduced survival was observed in these populations nearly a decade after exposure (Wiggins 2005). While the Salish Sea is home to over 10,000 overwintering harlequin ducks, disturbance of wintering habitat and prey base from increasing shipping traffic is cause for concern.

Hunting at wintering sites remains legal in Washington, where take is currently limited to one harlequin duck per season. Due to their relatively tame disposition, harlequins are an easy target for duck hunters. Hunting was probably the most important contributor to earlier declines in eastern populations.

Climate Change

Climate change is considered a threat to breeding and wintering habitat conditions. Warmer inland temperatures are expected to advance the timing and rate of snowmelt, resulting in earlier and more severe flow events on breeding streams (Hansen et al. 2014). High flow events affect streams in a number of ways that can negatively impact multiple harlequin duck life history phases, chiefly by increasing sediment load, washing out nests, and dislodging invertebrate prey from stream substrates. Egg laying and incubation on Park breeding streams is thought to occur between late-April and late-May before peak flows in June. More frequent and intense events may reduce foraging success for all demographics on breeding streams, while earlier such events may reduce body condition of females prior to egg laying and may ultimately lower nesting success (Hansen et al. 2014).

Changes in water temperature and chemistry (e.g., salinity, alkalinity) are projected to occur in both freshwater and marine systems, affecting physiology of harlequin ducks and their prey base. Increasing stream temperatures and related changes to flow characteristics have been cited as a possible cause of breeding range contraction in southern Oregon (Doerr 2013). Changes in distribution, abundance, and phenology of invertebrate food availability may occur under climate change conditions.

Conservation and Research Needs

Standardized population monitoring is needed across the range of harlequin ducks. The low densities of ducks on breeding streams result in effort-intensive surveys that yield low (30–40%) detection probability (per survey; Jarvis and Bruner 1997, Singleton and Long 2018); consequently, six

surveys of a given stream may be required to yield 90% detection probability (Singleton and Long 2018). Environmental DNA (eDNA) assays are a more sensitive monitoring tool that can identify trace quantifies of DNA from aquatic species and could therefore significantly increase detectability of harlequin ducks on breeding streams. The feasibility of employing this method in Park streams is currently being analyzed and should be considered throughout the range of harlequin ducks to standardize monitoring.

Past surveys on Park streams include presence-only habitat data, but accurately estimating habitat selection and suitability will require data collection from a range of occupied and unoccupied sites. The relative importance of food availability and other breeding stream characteristics (e.g., vegetative cover, instream conditions) to site selection is not well understood despite the summaries provided in a number of studies (Bruner 1997, Wright et al. 2000, Wiggins 2005). Aquatic invertebrate sample data at Park survey sites is lacking and should be added to the suite of other key habitat data in order to estimate current and future habitat potential. These data would also inform management of Park waterways while complementing other habitat data collected from other land management units across the Cascades.

Migratory connectivity for harlequin ducks breeding in Washington is largely unknown. Surveys of major Salish Sea wintering grounds have been conducted (Nysewander et al. 2005), but we do not understand the predominant migration patterns that connect these coastal areas to inland breeding streams; therefore, it is difficult to evaluate the extent to which environmental conditions across life history phases contribute to population trends. This is complicated by the variable migration distance from Pacific coasts to inland nesting habitat, which reaches as far as the Rocky Mountains: for example, a radio-tagged harlequin from a study in Glacier National Park was detected on the Stehekin River in the North Cascades (L. Bates, personal communication). Utilizing techniques such as radio-tagging or banding on breeding grounds, followed by resighting, hunter reports, or radio telemetry on wintering grounds, will be essential in advancing our understanding of a wider range of ecological factors contributing to the status of harlequin ducks using Park breeding streams.

Long term monitoring of bioaccumulation of environmental toxins and contaminants in high trophic level consumers such as piscivorous birds and mammals is needed throughout the Park. Analysis methods for emerging contaminants such as microplastics are still being refined, while analyses of most heavy metals and PCBs in vertebrate tissues are well defined. Such monitoring is most critical at the large water impoundment features of Gorge, Diablo, Ross reservoirs and Lake Chelan, because water impoundment and water-level manipulation are known to influence bioaccumulation pattern, trophic transfer, and health risk of some heavy metals in food webs (Sang et al. 2019). Where harlequin ducks are nesting in the Park, management, facilities, and infrastructure operations should adhere to noise and disturbance guidelines established by NPS (Appendix A).

Marbled murrelet (*Brachyramphus marmoratus*)

The marbled murrelet (*Brachyramphus marmoratus*) is a small northern Pacific diving seabird in the family Alcidae and is named for its intermingling white and black non-breeding plumage. The breeding plumage is dark brown dorsally with white ventral feathers tipped with brown, giving the bird a mottled appearance (Figure 12). Marbled Murrelets are unique among seabirds and other Alcids in that they do not nest in colonies on shoreline and nearshore areas, but instead are solitary nesters in inland old-growth forests (Nelson 1997, McShane et al. 2004).



Figure 12. Marbled murrelet on a nest, photographed by Hamer Environmental, L.P., near H.J. Andrews Experimental Forest, Oregon.

Marbled murrelets are found on the North American Pacific coast, generally within 5–8 km from shore, from the Aleutian Islands of Alaska to central California (Nelson 1997, Ridgely et al. 2007). Populations from mainland Alaska to northern California are genetically distinct from populations at the northern (western and central Aleutian Islands) and southern (central California) peripheries of the species' range (Friesen et al. 2007, Bloxton and Raphael 2009). Their wintering distribution in Washington includes the outer coast and the southern Salish Sea (Strait of Juan de Fuca and Puget Sound; Desimone 2016). The greatest densities of breeding marbled murrelets in Washington are found in northern Puget Sound, the Strait of Juan de Fuca, and the northern outer coast (Miller et al. 2006, Lance and Pearson 2016, Falxa et al. 2016), with large adjacent tracts of mature forest used as nesting areas.

The global population of marbled murrelets was estimated at 947,500 birds, with 91% in Alaska, 7% in British Columbia, and 2% in the federally listed range (Washington to central California; McShane et al. 2004). Between 2000 and 2021, annual population estimates for the federally listed population ranged from 12,700 to 24,100 murrelets (McIver et al. 2023) and the 2022 population estimate for the Salish Sea, Washington, was 3,797 (95% CI = 2,781–4,829) murrelets (McIver et al. 2023). Count data from federally listed populations between 2000 and 2021 estimated a declining annual population trend of −4.6% in Washington and improving trends in Oregon (1.8%) and California (3.8%) (McIver et al. 2023). McIver et al. (2023) reported the regional population declined by roughly 20% between 2001 and 2021.

Documented marbled murrelet nesting areas in Washington occur at elevations ranging from sea level to 1,280 m (Desimone 2016). Nesting habitat in Washington includes mature coniferous forests near freshwater bodies, generally within 64 km (40 mi) of marine shorelines, and at least as far as 84 km (52 mi) inland (Hamer et al. 1995, Madsen et al. 1999). Flight detections of murrelets have previously been documented 113 km (70 mi) from the sea (Huff et al. 2006), and probable radar detections in 2021 in the Park were as far as 122 km (76 mi) from the nearest marine waters (Hamer Environmental 2022).

Although most of the Park is farther from marine waters than murrelets typically select for, habitat modeling of 56,118 ha around the Ross Lake, Diablo Lake, and Gorge Lake reservoirs found that 47% of the area constituted potentially suitable habitat (Hamer Environmental 2022). Empirical data on presence of murrelets are scarce because they are extremely difficult to detect when they are not at sea. During the nesting season, marbled murrelets typically move between foraging sites at sea and their inland nest site under the cover of darkness, most often travelling within the hour before sunrise. Marbled murrelets fly faster than nearly all other bird species their size, and murrelet-type radar targets in the Park were detected at flight speeds ranging 65.6–153.1 km/h (41–95 mi/h) (Hamer Environmental 2008, 2022). Because of this unique set of characteristics, marbled murrelet surveys in nesting habitat are initially conducted using marine radar (Hamer et al. 1995), and when probable targets are detected, auditory/visual surveys are then used to attempt to confirm species identity and locate nesting birds.

Radar detection surveys in and adjacent to the Park were conducted in 2008, at Bacon Creek, Baker River, Chilliwack Lake, Irene Creek, Thornton Lakes, and along the Skagit River via Washington State Route 20 (Hamer Environmental 2008) (Figure 13, Table 7). Those surveys detected 59 murrelet-type targets, 40 of which were detected just before sunrise making them more likely to be marbled murrelets as opposed to another fast-flying species. Follow-up auditory/visual surveys were conducted at select sites in 2009, and while observers with night-vision goggles documented possible murrelet detections at Chilliwack Lake and Thornton Lakes, no birds or nests could be confirmed. Radar surveys on Gorge, Diablo, and Ross Lakes were conducted in 2021: 63 murrelet-type targets were detected and 42 of those were detected just before sunrise (Hamer Environmental 2022). All probable murrelets in 2021 exhibited directional flight consistent with travelling between two points, as opposed to circling or landing behavior that might indicate a nearby nest. Simultaneous auditory/visual surveys of those areas in 2021 did not confirm any murrelets but did confirm other

non-target species and could rule those species out as the putative murrelet targets. From 1995–2020, only one record exists for a probable visual detection of marbled murrelets in the Park. This observation by NPS staff in 2017 was of a pair of putative murrelets together near Roland Point on Ross Lake, but no photographs were taken to document the occurrence.

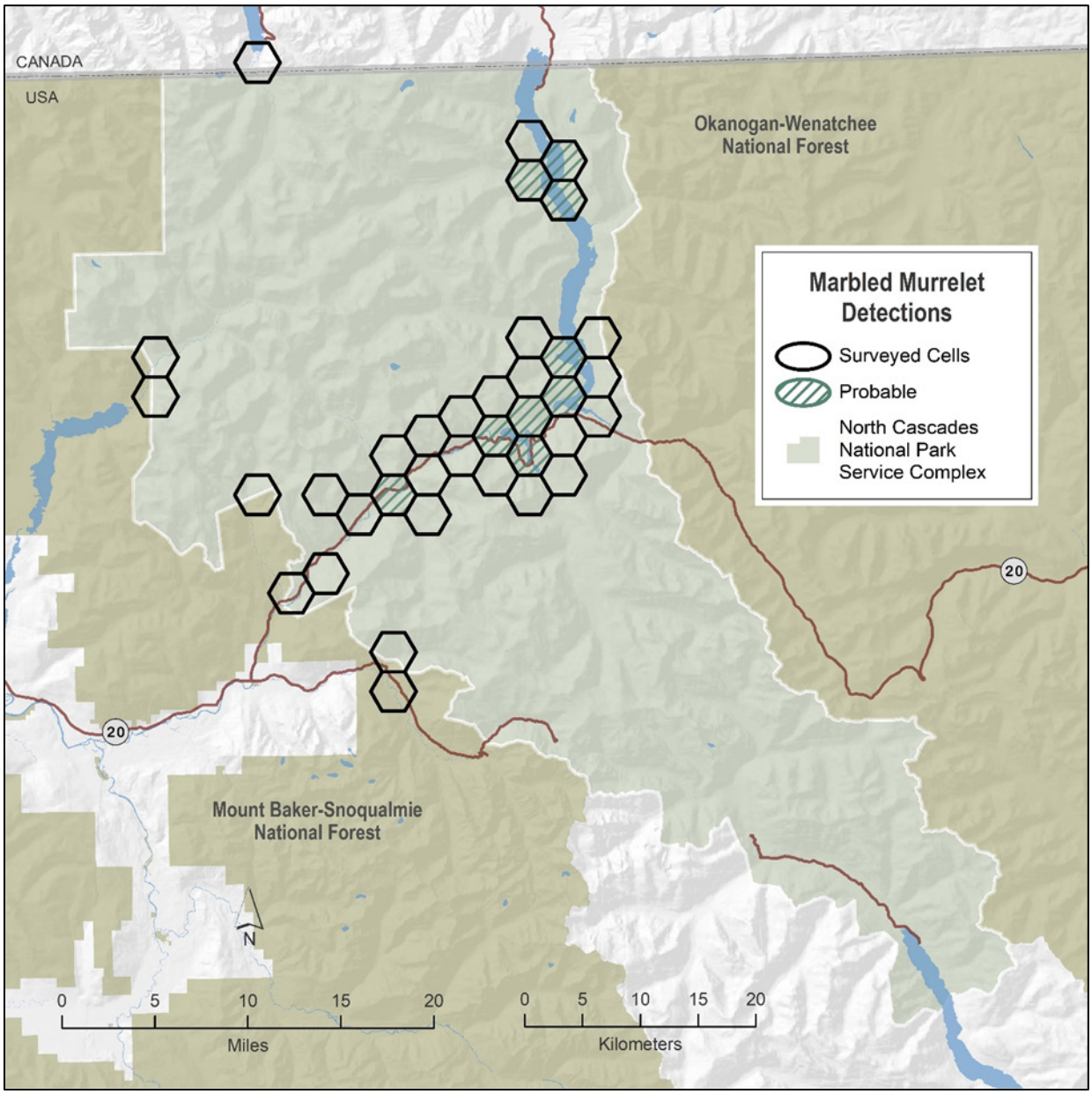


Figure 13. Survey locations, probable marbled murrelet (*Brachyramphus marmoratus*) detections in North Cascades National Park Service Complex, 1995–2021 (Table 7). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 7. Marbled murrelet (*Brachyramphus marmoratus*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
2008	18 radar surveys across six sites.	Once per point	Hamer Environmental 2008
2009	65 visual and acoustic surveys across six sites.	Once per point	Hamer Environmental 2009
2017	NPS staff observation	Incidental	NPS unpublished data
2022	46 radar surveys across nine sites, followed by visual and acoustic searches	Once per point	Hamer Environmental 2022

Nesting Habitat Requirements

Marbled murrelets are unique in that they are primarily a nearshore ocean forager but nest up to at least 84 km (52 mi) inland, typically in late-successional conifer forests (Nelson 1997). Forest stands are usually dominated by Douglas-fir, western red cedar, western hemlock, and Sitka spruce. Nests are typically greater than 10 m above ground and appear to be chosen based on the structural qualities that provide suitable nesting platforms, such as large limbs (approx. 30–75 cm wide), broken tops, mistletoe infections, vegetative cover obscuring the nest site, and high moss coverage that support nesting platforms on which a single egg is laid (Hamer and Nelson 1995, Wilk et al. 2016). The presence of nesting platforms is considered the most important characteristic of quality habitat, whereas tree size seems to be a less important factor (Nelson 1997, Huff et al. 2006). Documented nest sites are also correlated with the lower portion of slopes, proximity to streams or waterways, and larger forest patch sizes (USFWS 1997, Huff et al. 2006). Murrelets are known to occasionally nest on the ground where conditions emulate a traditional tree limb nest, such as one that was found on a cliff ledge on the northern Olympic Peninsula (Wilk et al. 2016).

Suitable nesting habitat is structured with adequate height and depth to provide cover from predators and moderate the microclimate around the nest (Raphael et al. 2002, Meyer et al. 2004, Huff et al. 2006). Although these characteristics are generally associated with mature forests, suitable nesting habitat is occasionally found in younger stands that contain residual older trees (Hamer and Nelson 1995, Huff et al. 2006). Murrelets appear to select for more highly dispersed nests where there is ample habitat available but will nest more closely together if there is limited space (Nelson 1997). In some areas, murrelets change nesting locations between breeding years (Burger et al. 2009) but have shown strong nest site fidelity in some areas, sometimes re-using the same nest (Hébert and Golightly 2006, Bloxton and Raphael 2009).

Diet

Marbled murrelets forage in marine waters, generally 2–8 km (1.2–5 mi) from shore, diving below the surface to obtain forage fish and large pelagic invertebrates (Ralph et al. 1995, Nelson 1997, Piatt et al. 2007, Hebert and Golightly 2008, USFWS 2009). Primary forage fish prey includes northern anchovy (*Engraulis mordax*), Pacific herring (*Clupea pallasii*), surf smelt (*Hypomesus pretiosis*), Pacific sand lance (*Ammodytes hexapterus*), juvenile salmon (*Oncorhynchus* spp.), and juvenile

rockfish (*Sebastes* spp.), which collectively comprise 60–100% of the diet. Invertebrates are commonly ingested by adults in the non-breeding season but rarely used to feed chicks. This prey group includes krill (Euphausiacea), mysid shrimp, squid (*Loligo* spp.), and large pelagic amphipods (Nelson 1997, Piatt et al. 2007). Parents usually choose a relatively large, energy-dense fish, such as immature herring, anchovy, larger sand lance, and smelt, to carry back to their chick (Carter and Sealy 1987). Murrelets have been observed occasionally resting and foraging in larger freshwater lakes near marine areas (Carter and Sealy 1986).

To pursue and capture prey, murrelets dive to depths that typically range 20–50 m in relatively shallow, sheltered marine waters, using their wings for propulsion. Adults are commonly found foraging pairs, even when one parent stays on an active nest, and in the northern part of their range they may gather in small flocks or interspecific aggregations to forage. Pairs often dive simultaneously, which may serve to increase foraging efficiency through cooperative strategies and to maintain the pair bond across breeding and nonbreeding seasons (Strachan et al. 1995).

Reproduction

Pairs of murrelets are seen together throughout the year and are known to pair bond for multiple breeding seasons (McFarlane Tranquilla et al. 2003, Huff et al. 2006). Courtship and mating activities begin on the saltwater in January and February, and a single large egg (nearly 25% of female body mass) may be laid sometime between late April and early August. Chicks hatch after a 30-day incubation period, during which parents alternate on 24-hour incubation shifts. Chicks remain on the nest for 27 to 40 days before fledging to the ocean (Nelson 1997). During the rearing period before fledging, murrelet parents often make multiple (2–8) daily trips to the ocean in order to feed their chick (Nelson and Hamer 1995, Nelson 1997, McShane et al. 2004). Parents are known to brood for only 1–2 days after the chick hatches but then spend most of their time at sea and flying inland to feed the chick. The male is the primary provisioner for the chick, especially during the last half of the rearing period (Bradley 2002).

Reproduction and nesting success are influenced by a variety of factors, the most of important of which are thought to be the presence of quality nesting habitat near shorelines, the adequate availability of prey, and nest predation (Marzluff et al. 2004, Malt and Lank 2007). On the Olympic peninsula, poor nest initiation and success rates (12.6% and 20%, respectively) in 2004–2008 were primarily attributed to the abandonment of nests prior to hatching or to nestling starvation, indicating low food availability (Bloxton and Raphael 2009). Food availability was also proposed to be a limiting factor for a central California population, where murrelets spent a greater amount of time foraging during poor reproductive years (Peery 2004). Local ocean conditions influence the quality and abundance of prey and are therefore likely to have a significant impact on the reproductive success and timing of murrelets, but the mechanisms are poorly known (Nelson 1997, McShane et al. 2004). Speckmann et al. (2000) observed earlier and more successful breeding in an Alaskan murrelet population during a year with higher spring temperatures, attributing this relationship to increased zooplankton and forage fish productivity. Because murrelets sometimes initiate nesting for a second time after nest failure, earlier breeding probably imparts a population-level reproductive benefit (McFarlane Tranquilla and Bradley 2003). Regardless of timing and environmental factors,

females are not likely to be in breeding condition in successive years due to the high energetic cost (Hull et al. 2001, Peery 2004, Bradley et al. 2004).

Protective Status

The marbled murrelet was federally listed as threatened south of its Alaskan breeding range under the ESA in 1992. Under Washington State law (WA Administrative Code 220-610-010), marbled murrelets are designated as endangered and are protected from possession, harassment, hunting, and killing. Marbled murrelets are also designated as a threatened species in Canada (Rodway 1990, British Columbia FLNRORD 2018). Habitat loss, exposure to increased predation due to habitat fragmentation, and marine threats from oil spills and gill net mortalities were cited as reasons for listing. The marbled murrelet's use of old growth forest as nesting habitat prompted its inclusion as a species of concern in the Northwest Forest Plan, which has been the basis for much of the conservation action to preserve marbled murrelet nesting habitat (USFWS 1997). Marbled murrelets are listed as a Management Priority species in the Park.

In 1996, the USFWS designated nearly 3.9 million acres of critical habitat for nesting marbled murrelets in Washington, Oregon, and California. Criteria for critical habitat identified in the 1996 ruling include large, continuous blocks of suitable nesting habitat in proximity to marine foraging habitat, that maintain the species' rangewide distribution and are otherwise without adequate protection and management (USFWS 1996). While lands within the Park were not included in this designation due to preexisting and permanent legal protection, abundant suitable habitat is known to exist west of the crest (Hamer Environmental 2008, 2009, 2022). Designated critical habitat abuts large stretches of the western boundary of the Park and is primarily managed by the Mount Baker-Snoqualmie National Forest (USFS). These areas are located within the Cascade River, Baker River, and upper North Fork Nooksack River drainages.

Known Threats

Habitat Loss, Degradation, and Fragmentation

At the time of federal listing in 1992, it was estimated that >80% of historic marbled murrelet nesting habitat had been lost (USFWS 1997). Habitat fragmentation coupled with a naturally low reproductive rate has contributed to murrelet vulnerability over time. Policies and practices in the Northwest Forest Plan sought to minimize potential habitat loss and completely protect any currently occupied nesting habitat on federal land, with the expectation that habitat would develop over time as protected stands matured; however, it is expected that some habitat would still be lost to logging operations and wildfire (Raphael et al. 2018). In 2016, high quality murrelet habitat was measured using satellite imagery to quantify the net changes since the implementation of conservation actions. It showed a 12% loss of suitable nesting habitat across both federal and non-federal lands from 1993 to 2012, with a 27% loss on non-federal lands and 2.2% loss on federal lands. Private lands were the most impacted with a 37% loss in habitat due almost entirely (98%) to logging (Raphael et al. 2016).

There is significant positive association between nest success and distance from forest edge (Nelson and Hamer 1995). As forest patches become smaller and more fragmented, the ratio of edge to interior forest nesting habitat increases, disproportionately reducing high quality nesting habitat and exposing murrelets to a variety of potential threats (Raphael et al. 2018). Forest fragmentation is

shown to have a positive association with corvid densities (Marzluff et al. 2000), the primary murrelet nest predator, and fragmentation could potentially increase densities of other forest predators like northern goshawks (Burger 2002). Human activities like farming, logging, and poor refuse management have bolstered corvid populations (Marzluff et al. 1994), which has likely resulted in higher levels of murrelet nest predation. Reducing habitat edges and minimizing wildlife access to garbage from commercial and recreational activities in murrelet habitat may help to reduce corvid-murrelet interactions (Raphael et al. 2018).

Fragmentation can create a conundrum for breeding murrelets, where reproductive success is limited because pairs of nesting murrelets only have access to small forest patches, forcing them to nest near edges and reuse nesting sites. Near-edge nests are more frequently predated and, once discovered by intelligent corvid predators, are repeatedly predated over successive breeding years (Hebert and Golightly 2006, Raphael et al. 2018). This impact is most evident in the highly fragmented southern portion of the murrelet's range in Oregon and California, with 61% to 81% of nests failing due to predation (Peery 2004).

Predation

Marbled murrelets at all life stages are believed to be at great risk of predation and this threat is likely a significant limitation to populations (Nelson and Hamer 1995, McShane et al. 2004). Adults have cryptic plumage and travel by indirect routes under low-light conditions to avoid detection of aerial predators (Nelson and Hamer 1995). Anthropogenic light pollution can have a wide range of detrimental impacts on wildlife, and especially on birds such as murrelets who rely on the cover of darkness to protect them from diurnal hunters like falcons (Sanderson et al. 2002). Murrelet eggs and chicks also express cryptic coloration that indicates evolution under the pressure of nest predation, which is considered a primary driver of nest failure for the species. Known nest predators include the common raven, Steller's jay (*Cyanocitta stelleri*), sharp-shinned hawk (*Accipiter striatus*), bald eagle, peregrine falcon, and northern goshawk, while potential nest predators include barred owl, great horned owl (*Bubo virginianus*), Cooper's hawk (*Accipiter cooperii*), American crow (*Corvus brachyrhynchos*), and Canada jay (*Perisoreus canadensis*) (Nelson and Hamer 1995, McShane et al. 2004). Arboreal mammals, such as squirrels and other rodents, martens, and fishers, are other potential nest predators but these interactions have not been documented (Nelson and Hamer 1995, Bradley and Marzluff 2003). Some forest generalists, such as barred owls and corvid species, thrive within fragmented landscapes and have been increasing in numbers, thereby posing a greater predation risk to marbled murrelets (Nelson and Hamer 1995).

Declines in Prey Abundance and Quality

Because food availability and quality are known to limit marbled murrelet breeding success (Speckman et al. 2000, Peery 2004, Bloxton and Raphael 2009), potential reduction of prey stock in nearshore waters is a major concern (Desimone 2016). Of the multitude of factors contributing to the dramatic alteration of coastal marine ecosystems (e.g., pollution, climate change, habitat destruction), overfishing is thought to be the most pervasive and deleterious (Becker and Beissinger 2006). Forage fish populations in the range of marbled murrelets have undergone overall declines and periodic collapses for over 100 years, with overharvesting concentrated on higher trophic level murrelet prey

species like Pacific sardine, northern anchovy, and market squid. Limited availability for these larger, energy-rich prey has driven marbled murrelets and other seabird species to compensate with lower-quality prey such as sandlance, juvenile rockfishes, krill, and mysid shrimp (Becker and Beissinger 2006). This diet shift has likely played a significant role in the 50% decline in murrelet productivity in the Salish Sea over the last 150 years by preventing or disrupting successful breeding. Specifically, adults may not reach sufficient breeding condition to initiate nesting or may resort to feeding nestlings relatively small, energy-poor food at less frequent intervals, which would likely limit their fitness and survival (Gutowsky et al. 2009).

Climate Change

Marbled murrelets are considered moderately to highly vulnerable to the effects of climate change from probable alterations to marine food sources and the quantity and suitability of nesting habitat. Climate change is expected to warm sea surface temperatures, and alter acidity, nutrient upwelling, and current patterns that are linked to more severe hypoxic events, algal blooms, accumulated biotoxins, and marine food web alterations (Tillmann and Siemann 2011). These phenomena all have the potential to impact marbled murrelet health and reproductive success by direct poisoning or by reductions in prey availability (McShane et al. 2004, USFWS 2009). Warmer sea temperatures in particular have already been linked to reductions in marbled murrelet breeding success by impacting food source (krill and juvenile rockfish) productivity, whereas relatively cool temperatures are associated with greater food abundance and higher murrelet reproductive success (Becker et al. 2007). An interrelated and increasing effect, ocean acidification, inhibits shell calcification for species like zooplankton that are important contributors to the food webs of marbled murrelet prey species (Ruckelshaus and McClure 2007, USFWS 2009).

The amount and suitability of nesting habitat is also expected to be adversely affected by climate change. Climate projections predict a generally warmer and drier trend for marbled murrelet nesting habitat within western Washington's coastal spruce/hemlock zone (Halofsky et al. 2011), which suggests the potential for increased severity and frequency of wildfires (Millar et al. 2006). Intense stand-replacing fires could also further diminish patches of suitable nesting habitat and increase exposure to nest predation. A drier forest climate may also inhibit moss and epiphyte growth that plays a role in forming nesting platforms (Desimone 2016). Fog cover in the northern California portion of the marbled murrelet range is showing a declining trend, exacerbating drought-stress in coastal forests (Johnstone and Dawson 2010, Raphael et al. 2018).

Offshore Fatalities

Marbled murrelet life history characteristics leave them highly susceptible to mortalities from oil spills. Since their primary environment is in nearshore ocean, they are in proximity to shipping lanes and other potential sources of oil exposure for much of their lives. Oil spills have killed hundreds of murrelets, and chronic oil pollution exposure is also a likely cause of mortalities (McShane et al. 2004). Although gill net mortalities were originally considered in the federal listing of marbled murrelets, subsequent actions to protect seabirds has proven largely successful off the coasts of Washington, Oregon, and California, but remain a potential threat in the northern portion of the murrelets range (British Columbia and Alaska). Underwater anthropogenic sound also has the

potential to cause death, harm, or avoidance of areas by murrelets (Smith et al. 2023), as does derelict fishing gear (Good 2009). Collisions in proposed offshore windfarms are also an area of concern for marbled murrelet conservation moving forward (USFWS 2009).

Conservation and Research Needs

Because loss of nesting habitat remains a major driver of marbled murrelet decline, protected areas like the Park that preserve contiguous late-seral forest are critical resources. The Park continues to serve in this role, but little is known about if and where murrelets are selecting for nest sites and how nesting is distributed in the Park. Regular radar surveys are needed through time to monitor rates of occurrence, and simultaneous auditory/visual surveys are needed to detect occupied nest trees. Acoustic Recording Units (ARUs), which are small, autonomous devices that record a range of audio frequencies, may also help provide occurrence data when installed in nesting habitat. Such data collected around Ross Lake, Diablo Lake, and Gorge Lake will be especially helpful to managers who must adaptively manage anthropogenic impacts of noise and light associated with management, facilities, and infrastructure operations, as well as recreation impacts to this highly sensitive species. Mitigating noise and disturbance around marbled murrelet nest sites is required under the ESA and Park management policies (Appendix A).

Mitigations for light pollution along nocturnal flight corridors such as the Skagit River may help reduce depredation and disorientation as murrelets move between nests and the ocean. In the Park, an assessment of lighting installations is needed and potential mitigations at Newhalem, Diablo, Colonial Creek, the Environmental Learning Center, transmission towers and all three hydroelectric dams may be necessary. While murrelets and their avian predators can generally see within the same light spectrum, replacing white lights with ultraviolet or colored lights will not likely provide predator avoidance benefit. Constant white light is known to cause disorientation in birds, often leading to collisions with glass or communication/transmission towers and lines during night travel (Gehring et al. 2009, Lao et al. 2020). However, migrating birds have responded well to intermittent lighting as provided by motion-sensing lights on buildings and strobes on aviation and transmission towers. In such cases, birds that are disoriented by white lights can reorient during the moments of darkness between illuminations. In a study that replaced steady white and red lights on communications towers with flashing lights, there was an estimated 50–71% reduction in bird collision fatalities (Gehring et al. 2009).

Managing anthropogenic food availability to wildlife is important for all species, and access to such foods attracts ravens and crows, who are known to in turn prey on bird eggs and chicks of many species, including marbled murrelets and peregrine falcons. While most of the Park sanitation infrastructure consists of tightly closed bear-resistant trash cans, food storage lockers, and dumpsters, non-NPS facilities in and around the Park may not. Replacing deficient sanitation equipment with wildlife-resistant containers, and providing educational outreach to visitors and staff (including on the importance of cleaning up crumbs of food on the ground in campsites), will help reduce the attraction of crows and ravens, as well as prevent conflict with other wildlife such as bears.

Mount Rainier white-tailed ptarmigan (*Lagopus leucura rainierensis*)

White-tailed ptarmigan (*Lagopus leucura*) are the smallest of three ptarmigan species (subfamily Tetraoninae, grouse) that are endemic to alpine habitats of western North American mountain ranges (Aldrich 1963, Braun et al. 1993). There are five recognized subspecies of white-tailed ptarmigan that are thought to have speciated from ancestral rock ptarmigan (*Lagopus muta*) in the southern Rockies (Braun 1969, Johnsgard 1973): the southern white-tailed ptarmigan (*L. l. altipetens*; southern Rockies), the northern white-tailed ptarmigan (*L. l. leucurus*; northern Canada into Montana), the Kenai white-tailed ptarmigan (*L. l. peninsularis*; parts of Canada into Alaska), the Vancouver white-tailed ptarmigan (*L. l. saxatilis*; Vancouver Island, BC), and the Mount Rainier white-tailed ptarmigan (*L. l. rainierensis*; Cascades of Washington and southern BC; Aldrich 1963, Clarke and Johnson 1990). Mount Rainier white-tailed ptarmigan are present and have been documented breeding in the Park (Figure 14).



Figure 14. Mount Rainier white-tailed ptarmigan voucher record NOCA24299, photographed by Alex Patia, July 10, 2017, near Sahale Arm, North Cascades National Park, WA.

Mount Rainier white-tailed ptarmigan spend spring, summer, and autumn above tree line. Other subspecies of white-tailed ptarmigan can spend their entire lives above tree line but do move downslope during severe storms or when snow conditions are unstable (Hoffman and Braun 1977). White-tailed ptarmigan are in a nearly continuous state of molt to remain cryptic within the visually dynamic landscape (see Figure 14). Their plumage is entirely white during winter and transitions to barring of brown and black feathers with the onset of breeding season, followed by summer plumage

that is more finely specked and overall gray to light gray-brown (Braun and Rogers 1971). Males retain a pure white abdomen and under-tail coverts year-round, while females molt to entirely brown-black-buff coloration during the breeding and post-breeding periods. Their feet are heavily feathered, in part to aid travel over snow (Braun et al. 1993).

Collectively, white-tailed ptarmigan is the only species of ptarmigan that does not have a circumpolar distribution and that is known to exist south of Canada (Aldrich 1963, Henden et al. 2016, USFWS 2021a). The Mount Rainier subspecies is currently distributed across the Cascade Range from Mount Adams in the southern Washington Cascades north to the Fraser Valley in southern British Columbia; however, the exact northern boundary is unknown and possibly extends as far as Lytton, British Columbia (USFWS 2021a). It is believed that the Mount Rainier subspecies is currently distributed across most of its historical range, with the notable exception of the Mount St. Helens area, where the population was extirpated as a result of the volcanic eruption in 1980 (USFWS 2021a). This subspecies was unlikely to have ever naturally inhabited the Cascades south of Mount St. Helens and Mount Adams or the Olympic Mountains, due to a lack of sufficient high elevation dispersal corridors (Clarke and Johnson 1990, Hoffman 2006). An introduction of Mount Rainer white-tailed ptarmigan to the Wallow Mountains in Oregon failed to establish a population.

There do not appear to be any published estimates for population size and demographics, abundance, or trends for the Mount Rainier white-tailed ptarmigan (USFWS 2021a). Skagen (1980) conducted a small study on Sourdough Ridge in the Park and estimated ptarmigan density to be 15.44 birds/ha, but this appears to be the only such information reported for this subspecies. Data on the population status of white-tailed ptarmigan across their range are also scarce, as well as variable. The species is categorized as Least Concern on the IUCN Red List despite the metapopulation's overall slow decline (less than 30% decline over three generations), although conflicting data from Christmas Bird Counts and breeding bird surveys from approximately 50% of its range have suggested a 95.5% decline over 40 years (Butcher and Niven 2007, BirdLife International 2018b). Long term datasets from Colorado have shown contrasting evidence for the population status of southern white-tailed ptarmigan, with increasing, decreasing, and stable trends represented across three different study sites (Wann 2017, Wann et al. 2019). More data are required to evaluate the population status and necessary interventions needed for the species as a whole, and especially for the vastly understudied Mount Rainier subspecies.

The Park and the greater North Cascades region contain the most expansive, high quality, and connected ptarmigan habitat throughout the subspecies' range (USFWS 2021a). Habitat modeling based on recent vegetation, elevation, and snow data yielded approximately 634 km^2 (245 mi^2) of potential summer ptarmigan habitat across the Park's landscape, but the extent of use across this area is unknown (USFWS 2021a). In the last 25 years, there have only been 31 confirmed observations of Mount Rainier white-tailed ptarmigan at locations within and adjacent to the Park boundary, and 27 of these observations have occurred since NPS and the Institute for Bird Populations began long-term landbird monitoring in 2005 (Figure 15, Table 8). It is unknown how ptarmigan are distributed across the Park or how the population is trending. Identification and reporting of this species is likely confounded by the cryptic nature of the species and the ubiquitous presence of sooty grouse in

subalpine areas where they are sometimes mistaken for ptarmigan by Park visitors. Observations of this species reported by the public in databases such as eBird (2022) are difficult to verify without photos, and birders often record their location at trailheads which can be lower-elevation than the actual sighting, making it difficult to discern sooty grouse from ptarmigan by elevation or vegetation community.

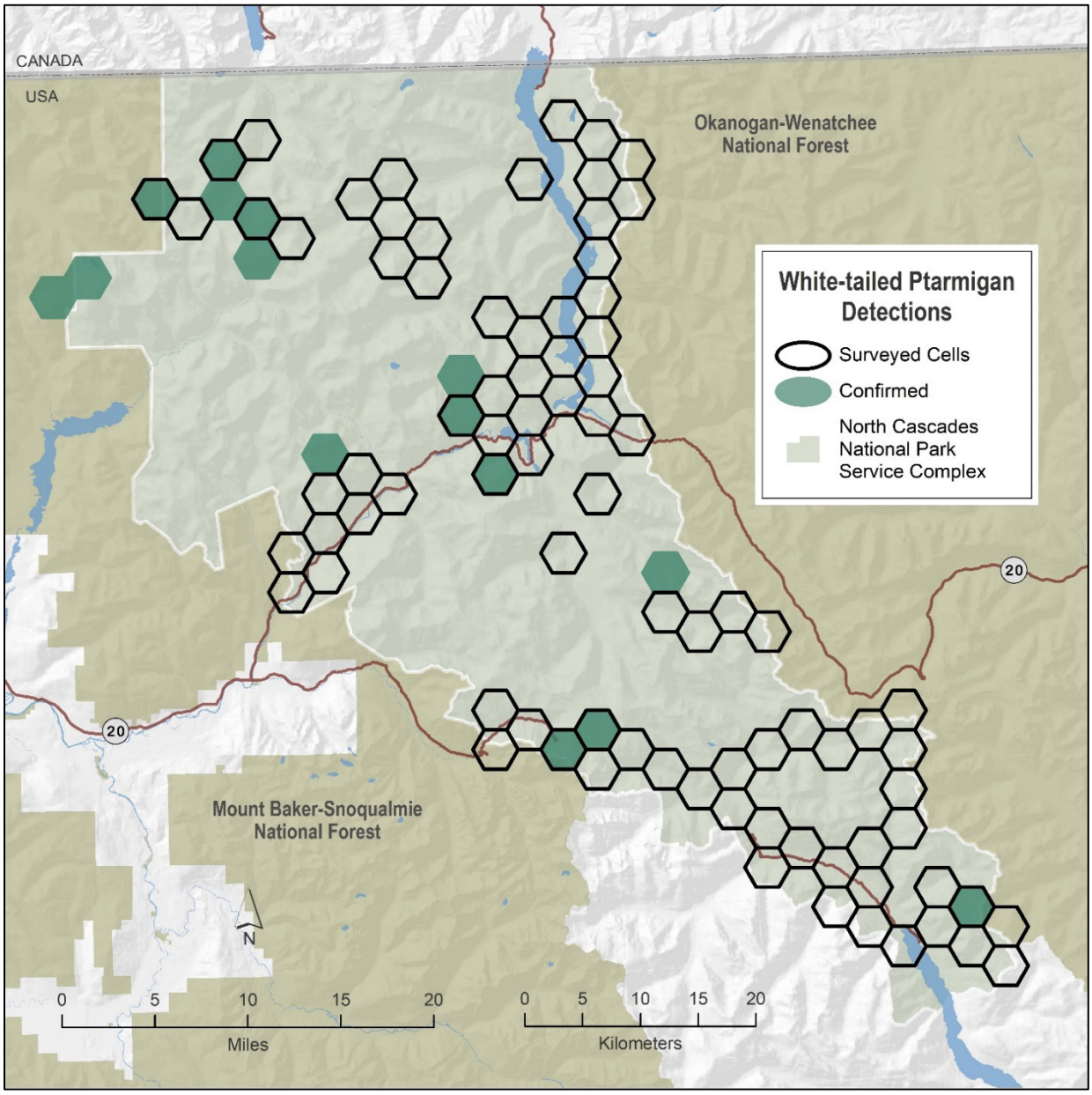


Figure 15. Survey locations, and Mount Rainier white-tailed ptarmigan (*Lagopus leucura rainierensis*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 8). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 8. Mount Rainier white-tailed ptarmigan (*Lagopus leucura rainierensis*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2005–2020	Landbirds point counts and incidental detections at 1,222 points (visual and acoustic observations) [a]	Annual or alternate year transects, May–July	Holmgren et al. 2015–2017, 2019–2021; Ray et al. 2017, 2018; NPS unpublished data

[a] Surveys were not designed for optimal detection of this species.

Habitat Requirements

Habitat use by white-tailed ptarmigan varies seasonally, but their habitat is generally bounded by treeline at the lowest elevations and barren rock and permanent snow at the highest elevations; however, they can also be found along streams and in meadows of the subalpine zone (Braun and Rogers 1971, Herzog 1977, Hoffman 2006). White-tailed ptarmigan migrate short distances, sometimes in flocks, to different habitat areas based on their life history phases and physiological requirements. Habitat use may be grouped into breeding, post-breeding (which includes nesting and brood rearing), and winter.

At the onset of breeding season (late-April to early-May in Colorado), male white-tailed ptarmigan move to higher elevations to establish breeding territories as snow recedes (Braun and Schmidt 1971, Clarke and Johnson 1992). In Colorado, early breeding territories are close to the treeline and shift upslope as snow melts and new herbaceous vegetation becomes available (Hoffman 2006). While little is empirically known about Mount Rainier white-tailed ptarmigan behavior, banded male ptarmigan have been observed moving up in elevation in March to the top of Armstrong Mountain, in north central Washington (Schroeder 2020). Across the species range, white-tailed ptarmigan presence is generally associated with a nearby water source and snowfield edges used for foraging and roosting (Johnson 1968, Skagen 1980, Fedy and Martin 2011). While snow is usually nearby, ptarmigan rarely use the snow patches themselves, especially where the snowpack is deep or extensive (Skagen 1980, Clarke and Johnson 1992). However, WDFW trail cameras have filmed flocks of Mount Rainier white-tailed ptarmigan pecking at snow in twilight and predawn hours (Schroeder 2020).

Breeding and post-breeding habitat use has not been differentiated for the Mount Rainier subspecies, but this dynamic appears to vary in other subspecies depending in part on the availability of water throughout the summer (Frederick and Gutierrez 1992). Mount Rainier white-tailed ptarmigan likely exhibit similar movement patterns to those in the Rocky Mountains, which tend to migrate upslope from breeding areas as snowmelt opens up higher elevation habitat. Abundant rock cover (over 50 percent of the ground cover; Hoffman 2006) appears to be an important summer habitat feature as a refuge from predators and weather (Johnson 1968, Braun and Schmidt 1971). Ptarmigan observed in the Park (Sourdough Ridge) during July and August were associated with stable rocky features and well-developed communities of mountain heather and forbs (e.g., *Cassiope mertensiana*; Skagen 1980), consistent with observations from the Rockies and Vancouver Island (Choate 1963, Martin et

al. 2004, Hoffman 2006). Ptarmigan in Mount Rainier National Park and the Pasayten Wilderness in the Okanogan-Wenatchee National Forest have also been observed during the post-breeding season using boulder fields near persistent snowfields (Schroeder 2020, USFWS 2021a).

Predation risk and microclimate appear to be key factors influencing nest site selection for Southern white-tailed ptarmigan. Hens select nest sites with sufficient cover, often next to a boulder, or otherwise in proximity to other cover types like willow, sedge, and conifer krummholz (Wiebe and Martin 1998). Covered sites have more stable microclimates that mediate large temperature swings, thereby protecting embryos that are sensitive to even short exposures to temperatures above 40°C (Webb 1987). Cover is also necessary to reduce the risk of nest predation, but it may in turn prevent hens from escaping from an approaching predator (Wiebe and Martin 1998). Nest site characteristics in a Rocky Mountain population showed significant seasonal variation, suggesting that microclimate is a greater selective pressure than predation (Wiebe and Martin 1998). Visinoni et al. (2015) found that alpine rock ptarmigan (*L.muta helvetica*) required a trade-off in microtopography and microclimate to meet their requirements for heat dissipation, predator avoidance, and food resources, because no single place in the habitat meets the needs for all requirements at a given time.

Winter habitat use by the Mount Rainier subspecies has not been studied. When snow limits foraging opportunities at higher elevations, southern white-tailed ptarmigan move downslope to wintering areas. Both males and females show high (60%) fidelity to wintering sites, although males generally winter at higher elevations (Braun et al. 1976, Hoffman and Braun 1977, Hoffman 2006). Wintering ptarmigan on Vancouver Island used areas both above and below treeline. Most wintering sites there were located within the Mountain Hemlock Biogeoclimatic Zone, but other forested areas were also occupied. Specifically, ptarmigan used hemlock and cedar forest, alpine bowls, and bare cliffs and rocky outcrops (Martin et al. 2015, USFWS 2021a).

The size of breeding territories is known to vary by season, location (i.e., habitat quality, hunting pressure), and ptarmigan density. As snow melts and exposes new food and cover, territory size expands upslope (Schmidt 1988, Braun et al. 1993). In high quality habitat where resources can be acquired in smaller areas, territories are smaller. In Colorado, male breeding territories on Mt. Evans ranged from 4.9 to 67.1 ha (mean: 28.1 ha ± 18.9 SD; Braun et al. 1993), while those in Rocky Mountain National Park defended territories as large as 19.0 ha (Schmidt 1988). Breeding males rarely use the territory margins except to defend the boundaries from a challenging male. The majority of the time, breeding pairs use much smaller core areas that average about 3.0 ha ± 1.0 SD (Schmidt 1988).

Diet

Diet composition for white-tailed ptarmigan varies regionally and seasonally. In all regions, common food items are willow buds and twigs (*Salix* spp.), sedges, grasses, and rushes (*Poales* spp.), mountain heather, and mosses. Most studies of white-tailed ptarmigan diet come from the Rocky Mountains, where they are known to have a relatively broad summer diet consisting of buds, catkins and other flowers, twigs and stems, fruits, seeds, leaves, and insects (Quick 1947, Weeden 1967, May and Braun 1972, Braun et al. 1993). During other times of the year, especially in the winter, willow is usually the dominant food item (May and Braun 1972). Like other grouse species,

ptarmigan ingest small stones throughout the year to aid mechanical digestion of their tough, fibrous foods. This “grit”, which is stored in the gizzard and consists mostly of quartz stones, may also provide an important source of minerals (May and Braun 1972).

Skagen (1980) observed ptarmigan on Sourdough Ridge (in the Park) selecting mainly for dwarf huckleberry (*Vaccinium deliciosurri*), as well as red mountain heather (*Phyllodoce empetriformes*), black-headed sedge (*Carex nigricans)*, white mountain heather (*Cassiope mertensiana*), crowfoot (*Leutkea pectinata*), Tolmie's saxifrage (*Saxifraga tolmiei*), spiked wood rush (*Luzula spicata*), and mosses. The only other known study of white-tailed ptarmigan diet identified food items from scat samples and found the birds ate primarily willow buds and twigs (92 percent by weight), with sedges and grasses making up the remainder (Weeden 1967). There does not appear to be information on winter forage of the Mount Rainier subspecies, but they likely consume various parts of willow, birch, and alder shrubs that protrude through the snow (USFWS 2021a). Ptarmigan usually obtain their food directly on the ground as well as by flying up into bushes to pick buds and catkins. During the winter, the birds make frequent flights between their roosts and suitable foraging areas with exposed vegetation (Braun and Schmidt 1971).

Reproduction

Pair formation occurs when females migrate upslope into male breeding territories, which occurs from late April to early May (Colorado; Schmidt 1988, Hoffman 2006). Courtship can include male displays of wing dragging, head bobbing, tail fanning, and calling. When concluded, the pair is established and they remain together through the breeding season until the late stages of incubation (usually from late June to mid-July; Braun et al. 1993, Hannon and Martin 1996). White-tailed ptarmigan are primarily monogamous, and while polygamy is uncommon it can occur if a clutch fails (Wittenberger 1978, Hannon and Martin 1996). If the male and female in a pair both return to the breeding grounds, they usually return to the same territory that they originally occupied (Braun and Rogers 1971).

Hens begin nest construction in early to mid-June, during which time the male stands guard while she scrapes out several shallow depressions in the ground and ultimately decides on one that she lines with dried vegetation and soft feathers (Giesen et al. 1980). Hens lay their first clutches of four to nine eggs between early June and mid-July and begin a 22 to 25-day incubation period (Braun et al. 1993, Wiebe and Martin 2000). Hens take short 25 to 30-minute recesses from incubating to forage, which almost invariably occur at dawn and dusk and occasionally during the daytime. Before flying to a feeding area, she will completely shroud the eggs with nearby pieces of vegetation (Giesen and Braun 1979). The male joins her and keeps watch while she feeds continuously until flying back to the nest (Wiebe and Martin 1997). The male continues this daily routine until the eggs are nearly hatched, when he migrates upslope to rocky foraging areas nearer to snowfields where flocks of males and unsuccessful females may gather (Braun 1969).

When chicks hatch in early to mid-July, their eyes are open and dense down covers their bodies and feet (Martin et al. 2015). When all eggs have hatched, the hen and brood leave the nest and do not return. Prior to fledging at 10–12 days of age, chicks depend on the hen for thermoregulation, traveling to foraging areas, and predator protection (Martin et al. 2015). During the brood-rearing

season from mid-July to mid-September, the hen may keep the brood in the breeding territory, or they make their way upslope to separate brood-rearing areas with better climate, moisture, and cover conditions (Braun 1969, Hoffman 2006). Newly-emerged forage near late-lying snowfields is an important food source for older chicks (Hoffman 2006). By late summer, hens and broods have reached higher elevations with these conditions and often use the same areas as other broods and adults. Birds of all demographic classes continue foraging at these high elevations until harsh weather forces the downslope migration (Braun 1969, Schmidt 1988).

Protective Status

At global and national scales, white-tailed ptarmigan are considered to be widespread, abundant, and stable and the species is therefore designated as secure (G5, N5) by the Natural Heritage Foundation (NatureServe 2021) and as Least Concern on the IUCN Red List (BirdLife International 2018b). The Mount Rainier white-tailed ptarmigan subspecies is proposed for listing as a threatened species under the ESA (USFWS 2021b). White-tailed ptarmigan, along with its relatives in the family *Phasianidae,* are not protected in any part of their range by the Migratory Bird Treaty Act (USFWS 2020). At regional scales across the species' range, population status, vulnerabilities, and protections vary. White-tailed ptarmigan are legally hunted in Alaska, California, Colorado, Utah, British Columbia (excluding Vancouver Island), Northwest Territories, and Yukon Territory where the populations are widespread and relatively stable (Hoffman 2006). Geographically isolated populations in New Mexico, Wyoming, and on Vancouver Island are considered sensitive to varying degrees by regional authorities (Hoffman 2006).

The WDFW recognizes the Mount Rainier subspecies inhabiting Washington State to be a "Species of Greatest Conservation Need" in the State Wildlife Action Plan due to their geographic isolation and susceptibility to climate change (WDFW 2015). State authorities do not allow hunting of ptarmigan despite their classification as a game bird; however, British Columbia allows hunting in the North Cascades–West population unit (USFWS 2021a). The USFS does not consider white-tailed ptarmigan as a sensitive species within Washington's National Forests (USFWS 2021a). The NPS has designated Mount Rainier white-tailed ptarmigan as a Management Priority species within the Park.

In response to a 2010 petition to list the southern and Mount Rainier subspecies as threatened under the ESA, the USFWS issued a positive finding that indicated federal protection may be warranted. After conducting separate status reviews for each subspecies, the agency determined that while the southern white-tailed ptarmigan is not at risk of extinction in the foreseeable future, the Mount Rainier subspecies is. The compounding effects of geographic isolation and habitat degradation from climate change will likely drive future population decline of the Mount Rainier white-tailed ptarmigan (USFWS 2021b).

Known Threats

Climate change is the single greatest threat to the viability of Mount Rainier white-tailed ptarmigan and was the only significant stressor identified by the USFWS as grounds for ESA listing. Other factors considered but not deemed critical across the subspecies' range are disease, mining, hunting, grazing and browsing, predation, development and infrastructure, and the invasive willow borer

beetle (*Cryptorhynchus lapathi*; USFWS 2021a). Recreation may pose a threat to individual ptarmigan in localized areas but is unlikely to be a stressor to entire populations at present (USFWS 2021a).

Climate Change

Mount Rainier white-tailed ptarmigan are threatened by climate change on multiple interrelated fronts, which include major impacts to their habitat, food sources, and physiology. The intensity and scope of these combined effects will almost certainly reduce the overall viability of the subspecies. Washington's high elevation habitats are projected to undergo drastic alterations associated with warming rates faster than that of the global average (IPCC 2019). Glaciers and snowpack in the Cascade Range, which provide cool thermal refugia as well as a water sources to ptarmigan and their forage during summer months, have already shown significant declines in surface area within the past several decades. For example, Mount Rainier glaciers have receded by 39.1% between 1896 and 2015 (Beason 2017), while those in the North Cascades have shown a drastic overall decrease of 56% between 1900 and 2009 (Dick 2013) and are projected to decrease by 92–96% by the end of the century (Gray 2019). Similarly, spring snowpack in Washington has declined by 30% since 1955 and is expected to decline another 38–46% by 2050 (Roop et al. 2019).

While bare ground left in place of receded glaciers and snowfields may be colonized by ptarmigan forage plant communities, the successional stage that can support foraging ptarmigan and insect populations for chicks would take 20–100 years to develop (or 5 to 24 ptarmigan generations, assuming a 4.1-year generational period; Bird et al. 2020). Therefore, it is unlikely that the creation of new meadow habitat would keep pace with that lost to both melting ice and treeline advance (Harsch et al. 2009, Stueve et al. 2009, USFWS 2021a). The need to travel greater distances between suitable meadow habitat where ptarmigan breed and forage, and snowfields that serve as water sources, thermal refugia, and post-breeding habitat would therefore result in greater risk of predation and heat stress (Bird et al. 2020). Given that Mount Rainier white-tailed ptarmigan undergo physiological stress above an ambient temperature of 21°C ([70°F] Johnson 1968), projected temperature increases and the reduction of cool summer refugia and meltwater across the range of the subspecies poses a major threat to the greater population in the coming decades (USFWS 2021a).

Winter forage and habitat may also be negatively affected by altered climate regimes. Ptarmigan rely on winter winds to distribute snow into deep banks which serve as insulative and protective roosting sites and as ladders to reach shrub branches for food, as well as to expose winter forage previously covered by snow (Saunders et al. 2007). Declining wind force, which is expected for the Cascade Range as temperatures warm, can therefore reduce adequate winter roost sites and forage availability that ptarmigan need to survive harsh alpine winters (Luce et al. 2013, USFWS 2021a).

Climate change may alter the timing of key environmental transitions (e.g., snowfall, peak plant growth) to which ptarmigan are adapted, resulting in asynchrony and possibly reduced survival and reproductive fitness. Because ptarmigan are adapted to be cryptic across seasons to match the general extent of snow cover, delayed snowfall, early snowmelt, or altered molt timing could make the birds more conspicuous against substrate and result in higher predation risk. In seasonal environments, there is a strong need for animals to couple reproductive events with food resource pulses in order to

meet the energetic demands of both adults and young (Thomas et al. 2001, Wann et al. 2019). The fitness costs of phenological mismatch can be even greater in high-elevation ecosystems where there is a small window for resource productivity and breeding attempts, and where there is an increased risk of extreme weather events (Martin and Wiebe 2004, Martin and Butler 2017). For white-tailed ptarmigan, asynchrony between the post-hatch period and peak plant and insect abundance can affect chick survival, although population-level effects on reproductive success have not yet been demonstrated (Wann et al. 2019).

Species distribution models project all three species of the genus *Lagopus*, including white-tailed ptarmigan, rock ptarmigan, and willow ptarmigan (*L. lagopus*) will experience substantial range declines in British Columbia as a result of climate change (Scridel et al. 2021). White-tailed ptarmigan are expected to experience the greatest range decline of 85.6% by the 2080's.

Conservation and Research Needs

To date, the only detailed data on Mount Rainier white-tailed ptarmigan in the Park come from the small breeding period surveys conducted on Sourdough Mountain in 1980 (Skagen 1980). Virtually all other available biological and ecological data on this species were collected in other regions. It is currently unclear why some areas of apparently suitable habitat lack observational records of the subspecies across its range (USFWS 2021a). The initial basic NPS research need for this species is completion of occupancy surveys of available habitat in the Park, followed by monitoring with a consistently applied protocol over time. Broader research needs include an assessment of demographics, analysis of genetic diversity and connectivity, empirical data collection on seasonal resource selection and dispersal behavior, and empirical measures of reproduction and survival. Additional studies are needed to quantify the availability of climate microrefugia (snowbank edges, stream edges, cold air pockets) that are likely important to ptarmigan and model their ability to mediate the impacts of raising temperatures. Winter resource selection by Mount Rainier white-tailed ptarmigan is unknown and understanding the effects of diminishing snowpack, as well as the impacts of winter recreation, are important steps to developing conservation measures in the Park (USFWS 2021a). Empirical measures of phenology focused on phenological mismatches between reproduction and food resources, and snow cover and plumage would provide deeper insights into potential effects of changing climate through time. Specifically, phenological mismatches between temperature and precipitation and hatching dates could lead to an increase in chick mortality rates arising from cold and wet conditions (Wann et al. 2019). A similar mismatch between plumage color and snow conditions could also decrease survival of adult ptarmigan due to increased susceptibility to predation. When birds have molted white in autumn, and snow has not yet accumulated, they are highly conspicuous to predators and mortality can increase: start date of snow cover was negatively correlated with population growth rates in rock ptarmigan (Imperio et al. 2013). This effect could be accelerated with elevational release of generalist predators, such as coyotes, as high elevation areas become more accessible earlier in the year (Whiles 2021).

While the greatest threats to Mount Rainier white-tailed ptarmigan appear to center around vulnerabilities to changing climate conditions, other anthropogenic threats are also present. Studies on both summer and winter recreation activity impacts on ptarmigan reproductive success and habitat

quality would help inform management and conservation strategies in the popular alpine areas of the Park. Routine management, facilities, and infrastructure operations conducted in subalpine and alpine zones of the Park could impact ptarmigan survival and recruitment. Specifically, helicopter landings on mountain ridges and summits to service radio repeaters and to ferry resource management and fire crews must take steps to mitigate disturbance to ptarmigan, especially during the nesting period of May through September (Appendix A). Spatial modelling of potential future climate refugia is also needed to prioritize where protection from these other (non-climate) threats should be emphasized in recreation management plans, travel plans, and other management plans for the Park.

Northern goshawk (*Accipiter gentilis*)

The northern goshawk (*Accipiter gentilis*) is a relatively large, forest-dwelling raptor that can be found throughout much of the temperate northern hemisphere (Figure 16). Three subspecies of goshawk are known to inhabit North America, but only one subspecies (*A.g. atricapillus*) occurs in Washington. Northern goshawks have medium gray plumage on their back and wings, a white underside with fine gray barring, a dark gray cap with striking white eyebrows, and yellow feet. Their long tail and short, rounded wings make them capable of high speeds and extraordinary agility while maneuvering through the forest to catch bird and mammal prey (Squires and Reynolds 1997). North American goshawks have a broad geographic range and breed throughout most of Alaska and Canada, through New England and the Great-lakes area, much of the western U.S., and south to central Mexico (Johnsgard 1990).



Figure 16. Northern Goshawk voucher record NOCA43681, captured on a remote wildlife camera (operated by National Park Service), March 7, 2020, in the Stehekin Valley, Lake Chelan National Recreation Area, WA.

In Washington, Northern goshawks can be found across much of the forested regions and in many different types of forest stands, although they prefer late seral stage habitat when it is available. Washington Department of Fish and Wildlife estimated there were 338 documented breeding territories as of 2003, but the current number of territories is unknown (Desimone and Hayes 2003, WDFW 2021). About half of the known breeding territories are in the eastern Cascades, followed by

the western Cascades (27%), forested regions of eastern Washington (12%), and the Olympic Peninsula (10%). There are historic records for birds breeding in the Southwestern portions of the state, but current observations show this area contains <1% of current breeding activity. Goshawks in Washington are either year-round residents or seasonal migrants, which move into Washington when prey in their breeding range is scarce. Washington appears to have a lower density of breeding goshawk pairs when compared to the rest of the western U.S. Washington goshawks have been extirpated from historically suitable habitat by human development and large areas of short-rotation timberlands. There are no studies evaluating Pacific Northwest goshawk population status (Desimone and Hayes 2003).

Northern goshawks are considered rare in the Park. Nineteen eBird (2022) observations have been recorded within the Park boundary since 1995 (Table 9). They were detected on 12 occasions during NCCN Landbird Inventory and Monitoring surveys between 2008 and 2018, and on two occasions during northern spotted owl surveys in 1995 (Ray et al. 2018, unpublished NPS data) (Figure 17, Table 9). The small number of observations does not discount the potential goshawk habitat within the Park, as goshawks are generally uncommon throughout their range. The Park contains prime goshawk habitat in its abundant mid-to-high elevation mature conifer forest (Smith et al. 1997). Grouse comprise a significant portion of goshawk prey at northern latitudes (Squires and Kennedy 2006). It is possible that sooty grouse abundance, as well as the abundance of other important prey species, may be linked to goshawk demography in the Park and the surrounding areas (Hoffman et al. 2015).

Table 9. Northern goshawk (*Accipiter gentilis*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2005–2020	Landbirds point counts and incidental detections at 1,222 points (visual and acoustic observations) [a]	Annual or alternate year transects, May–July	Holmgren et al. 2015–2017, 2019–2021; Ray et al. 2017, 2018; NPS unpublished data

[a] Surveys were not designed for optimal detection of this species.

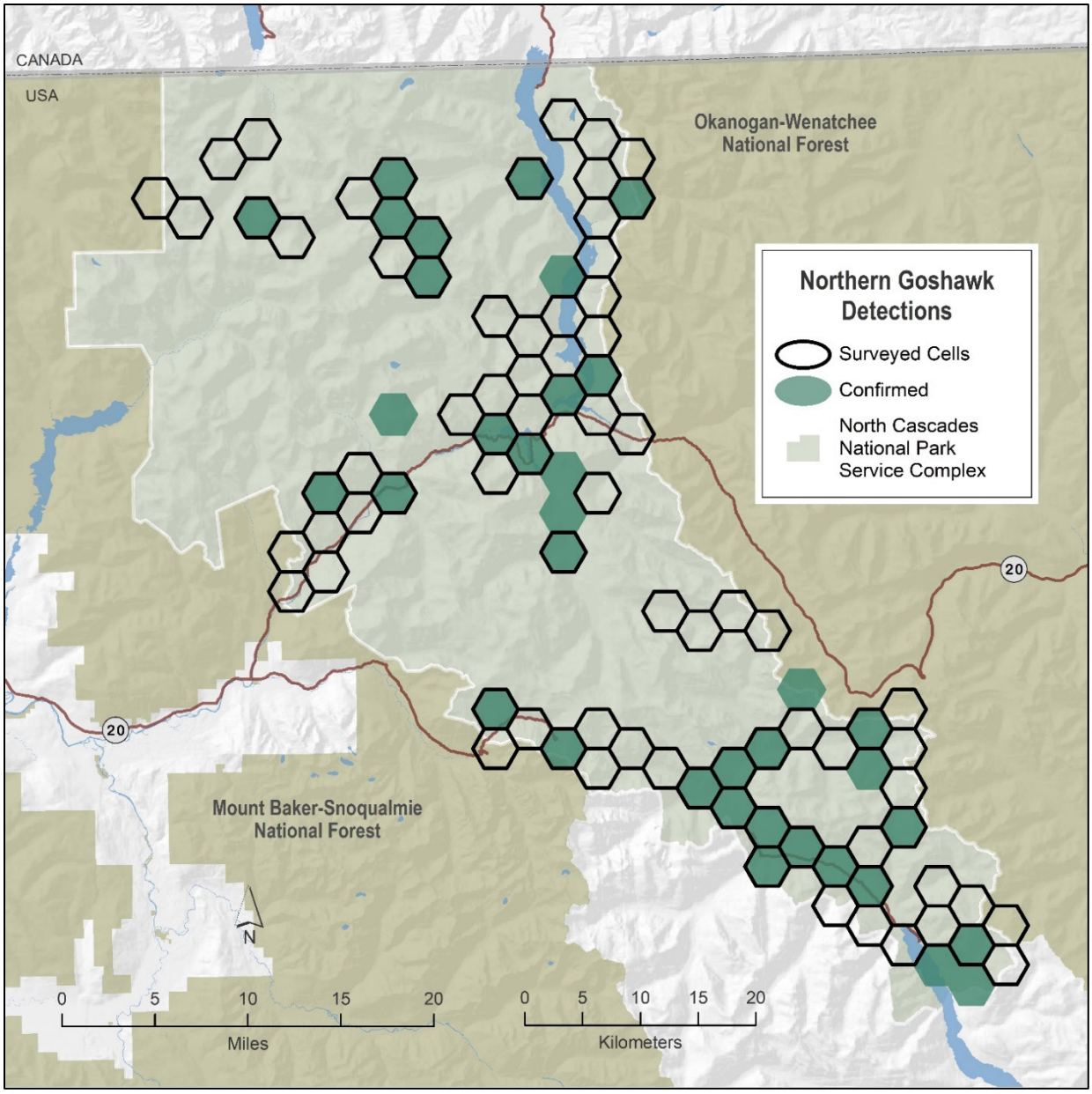


Figure 17. Survey locations, and Northern goshawk (*Accipiter gentilis*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 9). Hexagon survey and detection cells shown are 10 km^2 in size.

Habitat Requirements

In the North America, goshawks strongly select for late-successional forest structures for nesting and foraging but are habitat generalists in terms of tree species across ecoregions. Common features of goshawk habitat include large diameter (over 40 cm diameter at breast height [DBH]) trees with dense spacing and high canopy closure (greater than 40%), multiple canopy layers, and abundant course woody debris (Greenwald et al. 2005). Goshawks often nest in some of the largest trees in a stand, and on gentle to moderate slopes with fairly sparse vegetative groundcover, usually in proximity to a water source. Nests across Washington have been found in a wide variety of mainly

conifer species including Douglas-fir and ponderosa pine but may also nest in large diameter deciduous species such as red alder and aspen (Fleming 1987, Finn 1994, Finn 2000, Finn et al. 2002). Tree species diversity in nesting areas varies widely across studies reporting this metric (Greenwald et al. 2005). Their stick nests are large (up to 1 m diameter) and usually placed in the lower third of the forest canopy close to the bole (Reynolds et al. 1982, Finn 2000). Goshawks will use mistletoe brooms as nesting substrate in areas of heavy mistletoe infection (Moore and Henny 1983, Finn 1994).

For reasons that remain unclear, goshawks maintain one to eight alternate nests within their breeding territory, often switching nests between breeding seasons (Reynolds et al. 2005). Distance between alternate nests is variable. In Oregon, alternate nests were generally separated by 60 m to 90 m (Reynolds and Wight 1978), and in western Washington the average distance between nests was 536 m. In more arid pine and mixed conifer forests, nests were separated by 245–273 m. Loss of goshawk breeding populations could have indirect impacts on other raptor species who sometimes utilize unoccupied alternate nest sites (Desimone and Hayes 2003). Breeding season goshawk home ranges in North America are generally 5.7–35 km^2 (2.2–13.5 mi^2) (Squires and Reynolds 1997).

Foraging habitat is most frequently in mature forest characterized by canopy closure over 60% and high densities of large diameter trees and snags (Bloxton 2002, Greenwald et al. 2005). Prey abundance does not appear to be a limitation of goshawk productivity nor is it an important factor in selecting nesting or foraging sites; rather, goshawks forage where forest structure favors their hunting strategies (Greenwald et al. 2005). While hunting in open areas such as montane meadows has been observed, many studies have demonstrated a strong avoidance of early-seral forest stands that lack adequate flight space beneath the canopy and in open habitat, particularly logged open areas (Greenwald et al. 2005). Foraging areas define the home range of goshawks in the breeding season. Home range sizes in western states other than Washington ranged from 5.7–37.7 km2 (2.2–14.6 mi^2), and the average home range on managed forests in western Washington ranged from 8.4–107.3 km^2 (3.3–41.4 mi^2) (Austin1993, Bloxton 2002). It is likely that in landscapes with reduced late-seral stage forests, home ranges would be larger than average and territories more widely separated (Crocker-Bedford 1998).

Diet

Goshawks are opportunistic hunters, preying on a variety of bird and mammal species, and occasionally reptiles and insects. In Washington, the most frequent prey species were Douglas' squirrel, grouse, and snowshoe hares. Combined, these species represent 54% of goshawk diet on the eastern slope of the cascade crest and Okanagan County, and 41% in the Olympic Peninsula and Western Cascade Range (Watson et al. 1998). Passerines, especially corvids and thrushes, were also a significant source of prey, comprising 28% of prey west of the Cascade Crest, and 18% on the east side (Watson et al. 1998). Habitat requirements of prey species are variable, but a large proportion of prey species are positively associated with large trees, large woody debris, snags, canopy cover, and interspersed openings to provide a diversity of ground cover vegetation (Reynolds et al. 1992). Most goshawk prey occupies the ground-shrub zone (Reynolds and Meslow 1984).

Goshawks are morphologically and behaviorally adapted for hunting in forests (Squires and Reynolds 1997). Goshawks primarily use a sit-and-wait hunting strategy, but do not spend long periods searching from a particular perch; rather, they tend to spend only a few minutes at a perch before repositioning (Squires and Kennedy 2006). This hunting strategy is known as saltatory searching. In western Washington, goshawks appeared to vary foraging strategies by habitat type. Saltatory searching was used 72% of their time foraging overall but used this method 96% of the time when hunting in forest stands over 30 years old. They also spent 13% of their foraging bouts soaring low over substrates, but generally only in early-seral forest stands (Bloxton 2002). As with many raptor species, goshawk males and females cache prey in branches when nests are active, especially during the brood rearing period when chicks need frequent feeding (Zachel 1985).

Fluctuations in prey species appear to drive migratory behaviors over winter, though much is unknown about overwintering behavior in goshawks. Short distance migrations to lower elevations in resident birds appear to be prey-driven. Larger scale irruptions to alternate winter foraging grounds occur, particularly in the northern reaches of the goshawks range, during periods of scarcity in important prey species like grouse or snowshoe hares (Doyle and Smith 1994, Squires and Reynolds 1997). Several studies in Europe have shown that predator-prey interactions can significantly impact goshawk productivity and goshawk populations can significantly impact populations of prey species. This is an important facet of both goshawk management, and management of the species it predates, but there is a lack of studies on goshawk predator-prey interactions for North America (Squires and Kennedy 2006).

Reproduction

Goshawks are monogamous and historic nests are often used. Most goshawk pairs have been observed moving into nesting territories by March through early April, and as early as February (Squires and Kennedy 2006). Some individuals remain near their nest year-round, and in these cases may initiate breeding earlier than migrants (Squires and Kennedy 2006). Goshawks display a variety of courtship behaviors including sky-dance displays above the canopy, playful bouts of chasing (Palmer 1988), a wide range of vocal calling, and conspicuous perching (Lee 1981, Squires and Kennedy 2006).

While timing is highly variable, a clutch of 2–4 eggs (range: 1–5, mean for North America = 2.7 ± 0.88 SD) is generally laid between early April through early June (Reynolds and Wight 1978, Dewey et al. 2003). Incubation lasts approximately 30 to 44 days (Squires and Kennedy 2006), during which time the male may occasionally take over incubation for short periods after he delivers food (Lee 1981). Chicks usually hatch from late May through June and remain in the nest 35 to 45 days while the female does most of the parental care and the male does most of the hunting. One to three chicks fledge between late June and late July (Reynolds and Wight 1978).

In western North America, mean nest success ranges from 76–95%. Productivity, or the number of young fledged per active nest, ranges from 1.2–2.0 young per active nest and 1.4–2.7 young per successful nest (Squires and Kennedy 2006). Productivity estimates are greatest in the northern portion of the goshawk range in interior Alaska and in the Yukon Territory, Canada (McGowan 1975, Doyle and Smith 1994). Productivity is regulated by a wide range of factors including weather,

fluctuating prey populations, habitat quality, human activity, and age of breeding individuals (Squires and Kennedy 2006). Due to an unusually strong La Niña event in 1998–1999, the goshawk population in western Washington experienced high adult mortality and drastically reduced productivity (Bloxton 2002). Disturbance from logging in proximity to nest sites has been observed to cause nest failure and hatchling abandonment. A limited number of nest failures have also been attributed to camping close to nests (Squires and Reynolds 1997).

Protective Status

The goshawk has been proposed for ESA listing multiple times and has been the subject of numerable litigation efforts. In 1998 the USFWS determined that listing goshawks in the western U.S. under the Endangered Species Act was not warranted. The decision was based largely on a lack of evidence that habitat loss or scarcity was limiting goshawk populations, and that habitat limitations were unlikely to limit goshawk populations in the foreseeable future. They also concluded that goshawks are habitat generalists, and not an obligate old growth inhabitant (USFWS 1998b). Subsequent review of this decision agreed that available data were not sufficient to reliably establish population trends, although there is evidence of population reductions among several local studies. The review also acknowledged that goshawks preferentially used old growth forest in nearly every study area, but they have shown enough variability in habitat use that status assessments cannot be based entirely on late-successional forest distribution (Andersen et al. 2005).

The USFWS and USFS list the goshawk as a sensitive species. Reynolds et al. (1992) created a habitat management plan for the southwest United States, delineating recommendations for retaining, restoring, and maintaining forest structure and management practices suitable for goshawk nesting and foraging, and the support of goshawk prey populations (Reynolds et al. 1992, Squires and Reynolds 1997). The effectiveness of this plan is untested and has been met with some skepticism. Aspects of this plan have been adopted in some Pacific Northwest National Forests, but with an emphasis on nesting habitat, and largely omitting foraging habitat requirements (Squires and Kennedy 2006).

In Washington, the goshawk is listed as a state candidate species because of its association with declining late-successional forests. It is also identified as a priority species under WDFW's Priority Habitat and Species Program (WDFW 2021). The IUCN's assessment of the northern goshawk in 2021 designated the species as least concern due to their large range as well as their large and stable global population (BirdLife International 2021b). NatureServe (2021) lists the global goshawk population as stable (G5) but imperiled (G2) in Washington from the extent of habitat alterations, particularly logging.

Known Threats

The greatest threat to northern goshawk populations throughout their range is the degradation of late-seral forest habitat to which they are adapted. Direct or proximal human activity, especially forestry activities, are known to cause acute distress and mortality of nesting goshawks and may have disproportionately large population effects in Washington. Other sources of mortality include pesticides and other contaminants, shooting and trapping, falconry, and invasive species interactions;

however, these factors do not appear to significantly affect goshawk populations (USFWS1998, Squires and Kennedy 2006).

Habitat Loss and Alteration

The effects of timber harvesting on goshawks are not fully understood, but there is evidence to suggest that habitat alterations affect nest site selection, occupancy, and nesting rates (Crocker-Bedford 1990, 1995, Finn et al. 2002). Existing and potential nest trees may be removed, and foraging efficacy may be reduced by the loss of prey habitat or by degradation of foraging conditions (Desimone 1997, Saga and Selås 2012). Increasing fragmentation of mature forest habitat would likely have compounding negative effects on goshawks through greater exposure to inclement weather and interactions with competing and predatory raptors (e.g., red-tailed hawks and great horned owls; La Sorte et al. 2004, WDFW 2021).

A meta-analysis of goshawk habitat associations found that the raptors generally avoid open areas, especially logged early-seral stands younger than 30 years (Greenwald et al. 2005). Many studies have investigated the effects of forest management on goshawk ecology. While patterns are not universal across the breeding range, results generally show that nest and forage site occupancy, and in some cases reproductive performance, can be significantly lower near timber harvest (Crocker-Bedford 1990, Patla 1997, Santangeli et al. 2012, Rodriguez et al. 2016, Blakely et al. 2020). The abandonment of nesting territories is likely triggered above a certain threshold of habitat alteration (Moser and Garton 2009, Santangeli et al. 2012). Goshawks appear to be extirpated from urbanized landscapes and from areas that are moderately developed or are intensively managed for timber (rotations less or equal to 50 years; WDFW 2021).

Across the western U.S., mature and old-growth forests have been reduced to considerably less than 40% of the landscape (Reynolds et al. 1992, Forest Ecosystem Management Assessment Team 1993). Even though implementation of the Northwest Forest Plan in 1994 has slowed the decline of late-seral forests, suitable mature forest habitat has experienced a net loss of 3.4% regionally, with a loss of 1.5% on federal lands primarily from wildfire and a loss of 8.3% on nonfederal lands driven by timber harvest. Across all ownerships, timber harvest accounted for most (63%) of the habitat loss, followed by wildfire (31%). Habitat restoration efforts through the Northwest Forest Plan are projected to provide some increased habitat on federal lands (Davis et al. 2016). These predictions may be optimistic because forest cover projections are based on current harvest rates and wildfire regimes, which may be altered by revisions to forest policy and by a changing climate, respectively (Jones et al. 2016).

Most goshawk populations occur in forests that are naturally maintained by mixed-severity fire regimes, which promote habitat heterogeneity (Squires and Kennedy 2006, DellaSala et al., 2015). A historical emphasis on fire suppression and harvesting of large trees in the Pacific Northwest has resulted in large areas of homogenized forest with significantly increased stand density, reduced tree diameter, and a buildup of ladder fuels beneath the canopy (Lesmeister et al. 2018). Fire suppression, often in concert with timber harvest, can impart direct and indirect negative impacts that include increased risk of severe wildfires, damage by insects and disease, and reduced prey and habitat diversity (Agee 2005). Where lower severity fires have been shown to have neutral or positive effects

on the occupancy and fitness of late-seral forest inhabitants (Reynolds et al. 2006, Wan et al. 2019), near or complete stand replacement from large wildfires destroy mature forest habitat and have a greater potential to impart population-level effects (Lesmeister et al. 2018). Nesting and foraging goshawks are known to avoid areas burned at high intensities (over 64% canopy loss; Ray et al. 2014, Reynolds et al. 2006). Understanding the effects of fire on goshawk habitat is becoming more pressing in light of a changing and warming climate. High severity fires have already become larger and more frequent and are predicted to continue on this trend (Safford and Stevens 2017, Keyser and Westerling 2019).

Human Activity

Goshawks are known to be sensitive to a variety of human activities, but responses may vary depending on the frequency, intensity, and timing of the activity relative to life history phase. Disturbances can result in direct and indirect effects such as physically harming eggs, young or adults, altering habitat or prey populations, and interrupting normal behavior such as incubation and foraging (Postovit and Postovit 1987, Delaney et al. 1999). Loud, irregular noises produced from heavy equipment use (e.g., logging, road and building construction) within or near goshawk territory can cause nest abandonment and failure, even in otherwise suitable forest habitat (Reynolds et al. 1992, Squires and Reynolds 1997). Even irregular events like camping and human intrusion close to nests have caused nest failures and abandonment (Speiser 1992). Other evidence, however, suggests that goshawks can adapt to certain disturbances that are farther away, less severe, and produce regular, relatively benign noise (McGrath et al. 2003, Grubb et al. 2013, McClaren et al. 2015).

More research is needed to tease out the thresholds of human activity that drive reproductive success or failure. Irregular and severe disturbances that occur within the nesting phase prior to fledging are more likely to cause reduced nesting success (Toyne 1997); however, once the raptors choose a nest site, they are typically dedicated to it for the rest of the breeding season since they seldom lay a replacement clutch (Squires and Reynolds 1997). In the Lake Tahoe basin, nesting areas that were frequently occupied by goshawks had less than half the level of human disturbance (including recreational activities) compared to infrequently occupied nesting areas. Correspondingly, reproductive success was significantly higher within the frequently occupied territories (Morrison et al. 2011). Some sources suggest that human activity is not an important factor affecting the long-term survival of North American goshawk population (USFWS 1998b, Squires and Kennedy 2006), but it may have the potential to do so given increasing recreational activities in wilderness.

Conservation and Research Needs

Knowledge of northern goshawk occurrence in the Park is limited to incidental observations and detections during landbirds surveys (which are not designed to survey for goshawks specifically). Focused surveys for northern goshawks would be helpful toward informing management activities in the Park. During nesting season, NPS should survey for goshawks before a management or conservation project in goshawk habitat, and mitigations are established to minimize disturbance of known nests (Appendix A). Incidental disturbance to nesting goshawks is a concern in the Park (see Morrison et al. 2011), with many developed trails and campsites situated within goshawk habitat. Surveys for goshawk along trails and especially near backcountry camps could be used to identify

mitigations, such as short-term area closures around active nests (as authorized under NPS [2020]). Such closures could also be necessary to protect visitor safety, given the aggressive dive-bombing behavior exhibited by nesting goshawks.

Collisions with power lines and structures remains a concern for northern goshawks, and while an uncommon occurrence, a northern goshawk was detected dead at Diablo in 2014 after a window collision with a Seattle City Light building, and another was found in 2020 after a window collision at the Newhalem Visitor Center. Reflective glass mitigations for bird strike (e.g., stickers or anti-strike glass) should be used on large windows of buildings that are situated in goshawk habitat (such as the Skagit River Project infrastructure, Newhalem Visitor Center, Golden West Visitor Center, and the Environmental Learning Center).

Basic research on northern goshawks in the Cascade Range, including information on demography and population trend, goshawk-habitat relationships, impacts of different forms of recreation, and the effects of specific land management practices, is needed to help inform adaptive management strategies (Andersen et al. 2005). Disease and contaminant monitoring should also be conducted on recovered carcasses in the Park.

Northern spotted owl (*Strix occidentalis caurina*)

The northern spotted owl (*Strix occidentalis caurina*) is the largest of the three spotted owl subspecies and inhabits western North America from British Columbia through northern California. Contemporary presence and breeding of northern spotted owls in is Park has been confirmed (Figure 18). Northern spotted owls are primarily nocturnal, foraging mostly between dusk and dawn and occasionally during daylight hours. They are a highly territorial species and will defend their territories mainly with aggressive vocalizations, but rarely through physical confrontations (Courtney et al. 2004, Forsman et al. 1984).



Figure 18. Northern spotted owl voucher record NOCA24142, photographed by Jason Ransom/NPS May 21, 2017, in the Stehekin Valley, Lake Chelan National Recreation Area, WA.

Northern spotted owls occur within mature conifer forests of coastal mountains and the western and eastern slopes of the Cascade Range, from southern British Columbia to northern California. In Washington, this species was historically present in the Puget Lowlands but currently only inhabits the Cascade Range and the Olympic Peninsula, at elevations ranging from sea level to 1,500 m. Densities within the Cascade Range are highest at the southern periphery and generally become increasingly sparse north of Lake Chelan (Buchanan 2016).

The Park lies within the core of the northern spotted owl's range. Spotted owl populations are declining within the Park and across their range, with populations in Washington declining 65–85%

since 1993 (Davis et al. 2022). In 1993, NPS personnel began a 4-year monitoring effort to understand demography and threats facing spotted owls within the Park (Kuntz and Christophersen 1996). This study yielded 11 spotted owl occupied sites, with six pairs and five unassociated individuals. This relatively small sample size did not adequately represent the potential habitat across the Park, so efforts turned toward developing a habitat suitability map using documented spotted owl occurrences in the Park.

All known spotted owl territories were still insufficient to effectively identify potential habitat, so NPS personnel investigated the regional spotted owl habitat suitability model developed for the Northwest Forest Plan (Davis and Lint 2005), which assigns Habitat Suitability Values (HSV) based on a number of habitat characteristics, to see if it sufficiently encompassed the Park's spotted owl habitat. The Northwest Forest Plan's model was compared to habitat metrics from the Park's existing plant community map, and the locations of historic spotted owl territories. It was determined that the habitat map produced with Northwest Forest Plan model was sufficiently consistent with the vegetation map. Of the 11 spotted owl territories, seven of them had HSV values that placed them within the highly suitable range, two were negligibly below the highly suitable HSV score, and two had relatively low HSV scores, but bordered highly suitable habitat. It was determined through this comparison that the Northwest Forest Plan model was sufficiently consistent with Park-specific spotted owl observations to be successfully adopted for habitat mapping within the Park (Wilkerson and Siegel 2007).

During surveys conducted in the Park in 2007 and 2008, of the five historically occupied sites, four were occupied with spotted owl pairs (Siegel et al. 2009). During surveys of the west side of the Cascade Crest in 2009 and 2010, only two non-paired individual owls were documented out of five historically occupied sites (Siegel et al. 2012). Habitat has not diminished significantly within the Park over the last few decades, but barred owl (*Strix varia*) abundance has increased by more than 25% since the 1990s (Seigel at al. 2012) and displacement by barred owls is the primary threat to spotted owls within the Park (Hoffman et al. 2015). The first known hybrid spotted-barred owl in the Park was detected near Big Beaver Creek in 1995. Barred owl abundance and distribution is not well known in the Park, but they are regularly detected by staff and visitors. Nonetheless, site-specific surveys and incidental detections of northern spotted owls continue to confirm their presence in the Park (Figure 19). The last confirmed reproduction was observed in 2017; however, breeding surveys across historic territories have not been conducted in over a decade.

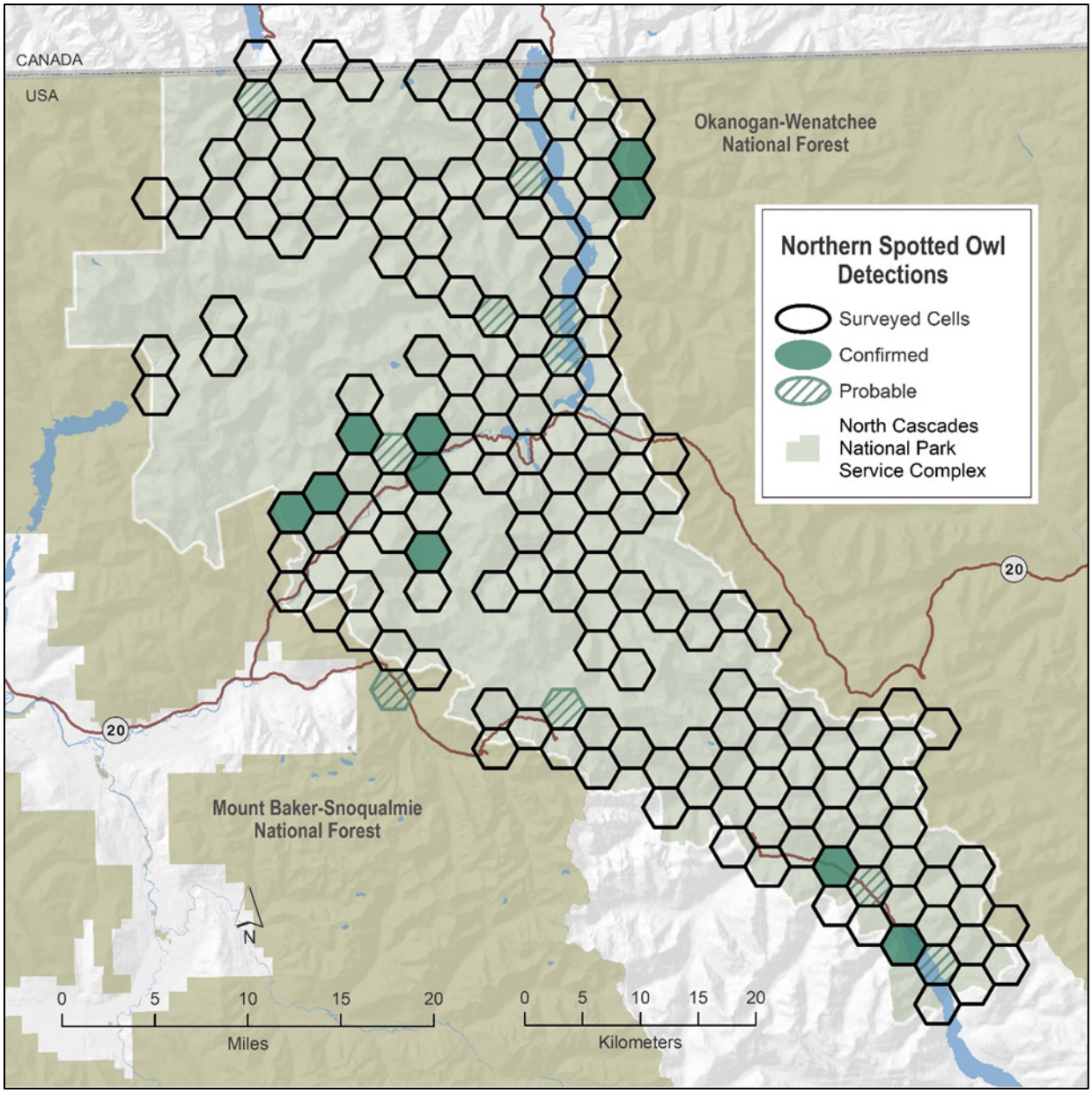


Figure 19. Survey locations, and Northern spotted owl (*Strix occidentalis caurina*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 10). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 10. Northern spotted owl (*Strix occidentalis caurina*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1993–1996	Hoot surveys at 264–562 points in territories	3 times per point each year	Kuntz and Christophersen 1996
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2007–2010	Hoot surveys at points in territories	1 time per site each year	Hamer Environmental, unpublished data; Siegel et al. 2009, Siegel et al. 2012
2005–2018	Hoot surveys along Landbirds transects (across 1,222 unique points)	Annual or alternate year transects, May–July	Institute for Bird Populations and NPS unpublished data
1995–2020	Hoot surveys at 1–2 specific territories (for compliance projects)	3 times per site each year	NPS unpublished data

Habitat Requirements

Northern spotted owls generally associate with old growth forest containing large diameter trees, high levels of structural heterogeneity, and moderate to closed canopy cover (Courtney et al. 2004, Forsman et al. 1984). This forest structure provides roosting and nesting structures in the form of tree cavities and platforms, supports prey abundance (Carey et al. 1992, Forsman et al. 2004), buffers temperature extremes, and provides cover from potential predators (Courtney et al. 2004, Forsman et al. 1984, Sovern et al. 2014). In the Cascade Range, cavity or platform nests are most often located in live Douglas-fir trees, with cavity nests primarily used in mesic forests and platform nests occurring more frequently in drier forests on the east side of the crest (Wilk et al. 2018).

A comprehensive view of spotted owl habitat is more nuanced than simple forest composition and structure. Foraging, nesting and roosting behaviors are associated with different habitat types, and impacted by habitat components such as climate, topography, and vegetation (Glenn et al. 2017). Habitat edge and matrix composition also appear to impact the suitability of spotted owl habitat, with less abrupt delineations of forest edge, through disturbance processes like low severity fire as opposed to stand-replacing fire or clear-cutting, showing a positive association for owl occupancy (Comfort et al. 2016). These influences, as well as the impact of prey abundance and encroaching barred owl populations, provide a more complete picture of suitable spotted owl habitat (Lesmeister et al. 2018).

Diet

Northern spotted owls consume a variety of arboreal, aerial, and terrestrial prey within the forest environment, such as rodents, shrews and moles, lagomorphs, bats, birds, and insects. Compared to the generalist diet profile of sympatric northern barred owls, spotted owls tend to specialize in select mammal prey species; however, both owl species primarily consume nocturnal prey. The most important prey for spotted owls at most study sites in Washington is the northern flying squirrel (*Glaucomys sabrinus*), shown to dominate the diet profile both in frequency and in biomass (Bevis et

al. 1997, Hamer et al. 2001). The bushy-tailed woodrat (*Neotoma cinerea*), snowshoe hair, northern pocket gopher (*Thomomys talpoides*), and American pika are other common prey items across their range (Bevis et al. 1997, Forsman et al. 2001, Hamer et al. 2001). Whether the prey is in a tree or on the ground, spotted owls usually attack by diving from an elevated perch. Excess prey is killed and cached on top of tree limbs, or on the ground beside trees, large rocks, or logs for later consumption (Forsman et al. 1984).

Reproduction

Spotted owls generally remain monogamous through multiple breeding cycles, often mating with a single partner for their entire life. Pairs roost in late winter to early spring and begin to lay eggs in February or March, two to three weeks after copulation (Courtney et al. 2004). In Oregon, an average of 62% of breeding pairs attempted to nest; of these, 81% were successful, with an average clutch size of 2 (range = 1–3). Incubation period is approximately 30 days. During nesting, females incubate and brood young alone, while males provide food during this period until young are around two weeks of age. Owlets typically leave the nest at 34–36 days old (Forsman et al. 1984).

Spotted owl breeding often displays a two-year cyclic pattern, with a high reproduction year followed by a year with few nesting pairs. This phenomenon is not well understood, but there is evidence that reproduction levels are influenced by patterns in prey abundance (Rosenberg et al. 2003), diet composition (White 1996), effects of weather (e.g., Anthony et al. 2006, Glenn et al. 2011a), and energetic costs of reproduction (Franklin et al. 2000). Prey relationships alone have yet to explain the synchronous patterns of spotted owl reproduction, however Rosenberg and others (2003) found a positive correlation between deer mouse abundance and reproductive success (Lesmeister et al. 2018). Prey switching is likely one of many factors confounding the relationship between prey abundance and reproductive performance. Various studies have shown that reproduction is negatively associated with cold wet winters, and nesting seasons and positive associations with summer and fall temperatures. Such results suggest that climate conditions influencing prey abundance could have significant effects on spotted owl reproduction (Glen et al. 2011b).

Protective Status

Northern spotted owls were listed by Washington state as an endangered species in 1988, and are protected from possession, harassment, hunting, and killing (WA Administrative Code 220-610-010). This species was then listed in 1990 under the ESA as federally threatened, primarily due to habitat loss and fragmentation from timber harvest and wildfire, and evidence of declining populations (USFWS 1990, Buchanan 2016). The NPS has designated northern spotted owls as a Management Priority species within the Park. Concern for spotted owl conservation was the major factor in the creation of the Northwest Forest Plan in 1994, which protects mostly large, contiguous stands of late-successional forest in hopes of sustaining and recovering spotted owl populations along with other inhabitants of old growth forest (USFS and BLM 1994). The association of spotted owls with highly valuable old-growth timber has made it a controversial species. Subsequently, the spotted owl has become one of the most studied birds worldwide as land managers struggle to balance the economic impact on the timber industry with the conservation of the species (Dixon and Juelson 1987).

Known Threats

Thirty years of occupancy data show that spotted owl populations have continued to decline at an average annual rate of 3.8% (range: 1.2–8.4%), and at an accelerating rate in some areas despite conservation efforts (Dugger et al. 2016). Populations in Washington are faring worse than in other areas of their range (Anthony et al. 2006), owing largely to habitat loss and alteration, high-severity fires, competition and hybridization with barred owls, changing weather patterns, and loss of genetic variation. The relative magnitude of these impacts varies over time and space, but competition with barred owls is currently known to be the greatest threat to the persistence of northern spotted owls (USFWS 2011). Other threats include diseases (e.g., West Nile virus and *Plasmodium* spp.) and presence of environmental contaminants like anticoagulant rodenticides (Courtney et al. 2004, Learn 2015).

Habitat Loss, Degradation, and Fragmentation

Davis and others (2016) estimated that since the implementation of the Northwest Forest Plan in 1994, suitable nesting/roosting forest experienced a net loss of 3.4% across the species' range, with a loss of 1.5% on federal lands primarily from wildfire, and a loss of 8.3% on nonfederal lands driven by timber harvest. Dispersal habitat declined by 2.3% overall but increased by 2.2% on federal lands. Across all ownerships, timber harvest accounted for the most habitat loss (63%), followed by wildfire (31%). Habitat restoration efforts through the Northwest Forest Plan are projected to provide some increased habitat on federal lands (Davis et al. 2016). These predictions may be optimistic because forest cover projections are based on current harvest rates, which may be altered by revisions to forest policy, and current wildfire regimes which may fluctuate with a changing climate (Jones et al. 2016).

A historical emphasis on fire suppression and harvesting of large trees in the Pacific Northwest has resulted in large areas of homogeneous forest with significantly increased stand density and reduced tree diameter (Lesmeister et al. 2018). These stands may be occupied by spotted owls but often impart direct and indirect negative impacts that include increased risk of severe wildfires, damage by insects and disease, and reduced prey and roost site diversity (Agee 1993). Where lower severity fires have been shown to have neutral or positive effects on spotted owl occupancy and fitness (Wan et al. 2019), near or complete stand replacement from large wildfires destroy spotted owl habitat and may impart population-level effects (Lesmeister et al. 2018).

The effects of habitat fragmentation on spotted owl occupancy vary depending on edge characteristics and the scale of fragmentation (Lesmeister et al. 2018). In southwestern Oregon, forest fragmentation was associated with lower survival and increased home-range size (Schilling et al. 2013). In Washington, increased measures of late-seral forest edge were associated with decreased extinction rates; moreover, colonization potential decreased with more late-seral patches (higher fragmentation) within spotted owl territories (Sovern et al. 2014). The benefit of certain forest edge types has been documented (Zabel et al. 1995, Sovern et al. 2014) and appear to be related to foraging decisions. While their main prey, northern flying squirrels, inhabit closed-canopy mature forest (Forsman et al. 2001), other high-value prey such as bushy-tailed woodrats, snowshoe hair, pika, and gophers thrive in or adjacent to forest openings (Zabel et al. 1995, Sovern et al. 2014).

Spotted owls are associated with natural forest openings and edges that occur around talus, broadleaf-conifer transitions, and low- and mixed-severity wildfire burn areas, but not with abrupt edges at clear-cuts or developed areas (Comfort et al. 2016). Barred owls, on the other hand, are known to thrive in highly fragmented and developed landscapes (Rullman and Marzluff 2014).

Barred Owls

Competition and displacement by encroaching barred owl populations is now recognized as the greatest threat to spotted owl persistence across much of their range (USFWS 2011). The barred owl is a slightly larger member of the *Strix* genus and occupies a similar ecological niche to the spotted owl. Barred owls historically inhabited northeastern North America, but their range has recently expanded to overlap with the entire range of the northern spotted owl (Gutiérrez et al. 2007).

Although the exact mechanism behind the barred owl's expansion to the west remains unclear, they now outnumber spotted owls across their range (e.g., Singleton et al. 2010, Wiens et al. 2011, Dugger et al. 2016). Barred owls have more generalist diets and habitat requirements (Livezey 2007), are more aggressive in their territory defense (Van Lanen et al. 2011), can exist in higher densities, and have higher annual survival rates than spotted owls, making them formidable ecological competitors (Singleton et al. 2010, Wiens et al. 2014). Spotted owl occupancy over the last 20 years has declined by 50% in Mount Rainier National Park and by 60% in Olympic National Park, while barred owls have been increasingly documented within historically monitored spotted owl territories since the first detections in the 1980s (Gremel 2017, Mangan et al. 2019). Monitoring within these federally protected areas with large areas of intact old-growth forest has allowed researchers to clarify the magnitude of impact by barred owl competition.

Successful hybridization between spotted owls and barred owls with fertile offspring is known to occur (Kelly and Forsman 2004). Spotted owls already exhibit signs of inbreeding depression, and hybridizations could be an added concern for the genetic viability of spotted owl populations (Gutiérrez et al. 2007, Funk et al. 2010). Recent genetic studies have shown that the levels of hybridization based on owl appearance may have been overestimated, and that individuals displaying intermediate plumage are actually barred owls that appear to be undergoing local selection for plumage that resembles the spotted owl's (Hanna et al. 2018). Conversely, it is possible that hybridization rates could increase as barred owls become more pervasive (Lesmeister et al. 2018).

Climate Change

Climate change is expected to have several indirect effects on spotted owl populations. The Pacific Northwest is likely to experience warmer, wetter winters with lower snowpack and hotter, drier summers (Parsons et al. 2001). Effects on vegetation and prey abundance will likely be a more severe consequence of this altered climate regime than the direct impact of weather on individual owls. Populations of spotted owls in drier climates will likely experience greater loss of prey abundance, as prey populations experience water and heat stress (Glenn et al. 2011a, 2011b). Projections of vegetation-climate models indicate that the most suitable spotted owl habitat may begin to shift northward and to higher elevations, so a complimentary shift in conservation priorities may be necessary to protect areas with the most potential for future spotted owl persistence (Carroll et al. 2010).

The impact of fire has been the subject of some debate. A recent meta-analysis of 30 years of wildfire and spotted owl data showed no significant negative impacts of fire on owls, and a positive effect on forage habitat, recruitment, and reproduction in low to moderate severity fires (Lee 2018). Another study found that when 5–40% of an owl's home range was severely burned, owls avoided the area (Jones et al. 2020). Both Lee (2018) and Jones et al. (2020) found that owls avoided post-fire landscapes where salvage logging occurred. If climate change continues to increase the frequency and magnitude of wildfires, their impact on spotted owl habitat may be exacerbated over time (Van Mantgem et al. 2009).

Loss of Genetic Variation

Inbreeding due to population bottlenecks is a concern for the genetic viability and survival of some populations of spotted owls, and likely to be of increasing concern as populations decline. Inbreeding is of particular concern in the western Washington Cascades, where it is more frequent than other portions of the spotted owl's range (15% in the Washington Cascades vs. 6.1% in the Olympic Peninsula, 3.5% in Oregon, and 2.7% in California; Miller et al. 2018). Genetic analyses have also presented strong evidence for recent bottlenecks corresponding to population decline in the Washington Cascades, which was not evident in other populations within the species' range (Funk et al. 2010). There is a strong evident selection against reproductive success in inbred birds, with only 6.8% of inbred individuals identified as having successfully bred, as opposed to 27.2% breeding in the non-inbred population. If inbreeding continues to worsen, it is possible that some populations could benefit from translocations to supplement genetic diversity (Miller et al. 2018).

Conservation and Research Needs

Within the Park, where spotted owl habitat is largely protected and intact, conservation efforts are primarily focused on management actions such as fire fuel treatments and mitigating noise during infrastructure activities such as road work and blasting. Such activities are performed in the Park through consultation with USFWS on a project-by-project basis, or through programmatic plans for routine operations. Human disturbance of spotted owls has been shown to induce a glucocorticoid stress response and may affect nesting behavior and success (Hayward et al. 2011). Special attention must be paid to noise and disturbance levels around spotted owl territories during the nesting period of March through September (Appendix A).

Lethal removal of barred owls has occurred in several areas of spotted owl habitat in Washington and has resulted in positive outcomes for spotted owls (Diller et al. 2016, Wiens et al. 2021). Such removals have not been conducted in the Park and are not currently planned. While this action was largely successful under a research design, maintaining a long-term removal program across the North Cascades ecosystem is not a feasible or sustainable operation. The national park system plays an important role in preserving the natural dynamic processes that shape ecosystems, and as such parks have historically been valuable "control" sites in studies that focus on topics such as landscape change, timber management, urban development, and agriculture practice outside of protected areas. This role of preserving a changing ecosystem in national parks is challenged by issues such as barred owl range expansion, where opposing mandates include protection and preservation of at-risk species

while simultaneously monitoring natural ecosystem change. This issue will likely become a more complex problem and involve a more flexible adaptive resource management program in the future.

Longitudinal research on the impacts of fire fuel thinning treatments is needed to better understand how spotted owls use both the managed mosaic forest and how much regrowth is needed for the treated stands to become viable again for owl use (Wan et al 2018). Of note, treatments in currently occupied spotted owl territories may remove important forest floor structure that hosts abundant small mammal prey. Given the rapid declines in this species, it may be prudent to focus fuel management in the Park outside of occupied territories, rather than disrupt the predator-prey dynamics that are already functioning. It also remains unknown how forest management such as thinning disparately impacts spotted owl versus barred owl fitness: it is likely that the more sensitive spotted owl may be negatively impacted while the competing barred owls are provided an advantage (Dugger et al. 2016, Singleton 2015). Evidence presented by Lee (2018) suggests that low- and medium-severity fire may have more benefits than costs to spotted owls, and thinning may not be necessary as a tool to conserve that species.

The abundance, distribution, and population trends of small mammals in the Park have not been well studied, and there are few studies on the relationships between how spotted owl prey are impacted by disturbance, weather, and climate fluctuations. Long-term studies of small mammals in the Park may help inform spotted owl conservation plans through time. Anticoagulant rodenticides (ARs) can have severe impacts on owls through hunting of debilitated small mammals or scavenging of carcasses. The NPS does not use ARs in the Park, but such toxins may be present on private lands in and around the Park. If any carnivores are found dead in the Park, efforts must be made to screen carcasses for environmental toxins and contaminants, and specifically ARs. Any carcasses of spotted owls should also be tested for West Nile virus and avian influenza.

Understanding abundance, distribution, and trends for northern spotted owl in the Park will continue to be a critical need toward conservation of this species. Long-term monitoring may include use of autonomous Acoustic Recording Units to contribute to local and regional occupancy modeling for this species.

Peregrine falcon (*Falco peregrinus*)

Peregrine falcons are medium-sized raptors ranging across all continents except Antarctica, and can be found across all of North America, from the Canadian arctic to Mexico, making it one of the most widespread and best-known raptors (White et al. 2002). They represent an important component of ecosystems, and due to their high trophic level position, often serve as good bioindicators of ecosystem health (Sergio et al. 2008). Peregrine falcons are colloquially known as "duck hawks" in some regions because of their persistence around large migratory flocks of ducks on open bodies of water. They hunt by stooping, or plummeting rapidly down onto their prey, at high speed. With large flight muscles, long pointed wings, tapered tail, and flat head, peregrine falcons are built for fast and powerful flight and can attain stooping speeds of 280–320 kph (173–199 mph), making it the fastest bird on earth (Tekiela 2011). Peregrine falcons present reversed sexual dimorphism, with females generally 15% to 20 % larger and 40% to 50% heavier (White et al. 2002). Adults are 40–50 cm in length and have a wingspan ranging 91–112 cm. Adult males and females have similar markings and plumage including a distinctive dark hood and wide malar stripe or "sideburns" along the cheeks, cream-colored throat, blue barring and yellow soft underparts (Figure 20). Immature falcons are dark brown above with buffy feather edging, vertically streaked underparts, bluish-gray to greenish-yellow soft parts, and a less distinct hood and malar stripe. The average life span of a peregrine falcon is approximately 6–8 years, although some have been reported to live up to 20 years (White et al. 2002). Contemporary presence and breeding of peregrine falcons in the Park has been confirmed (Figure 21).



Figure 20. Peregrine falcon voucher record NOCA40538, photographed by Paul DeBruyn/WDFW, June 27, 2007, near Newhalem, Ross Lake National Recreation Area, WA.

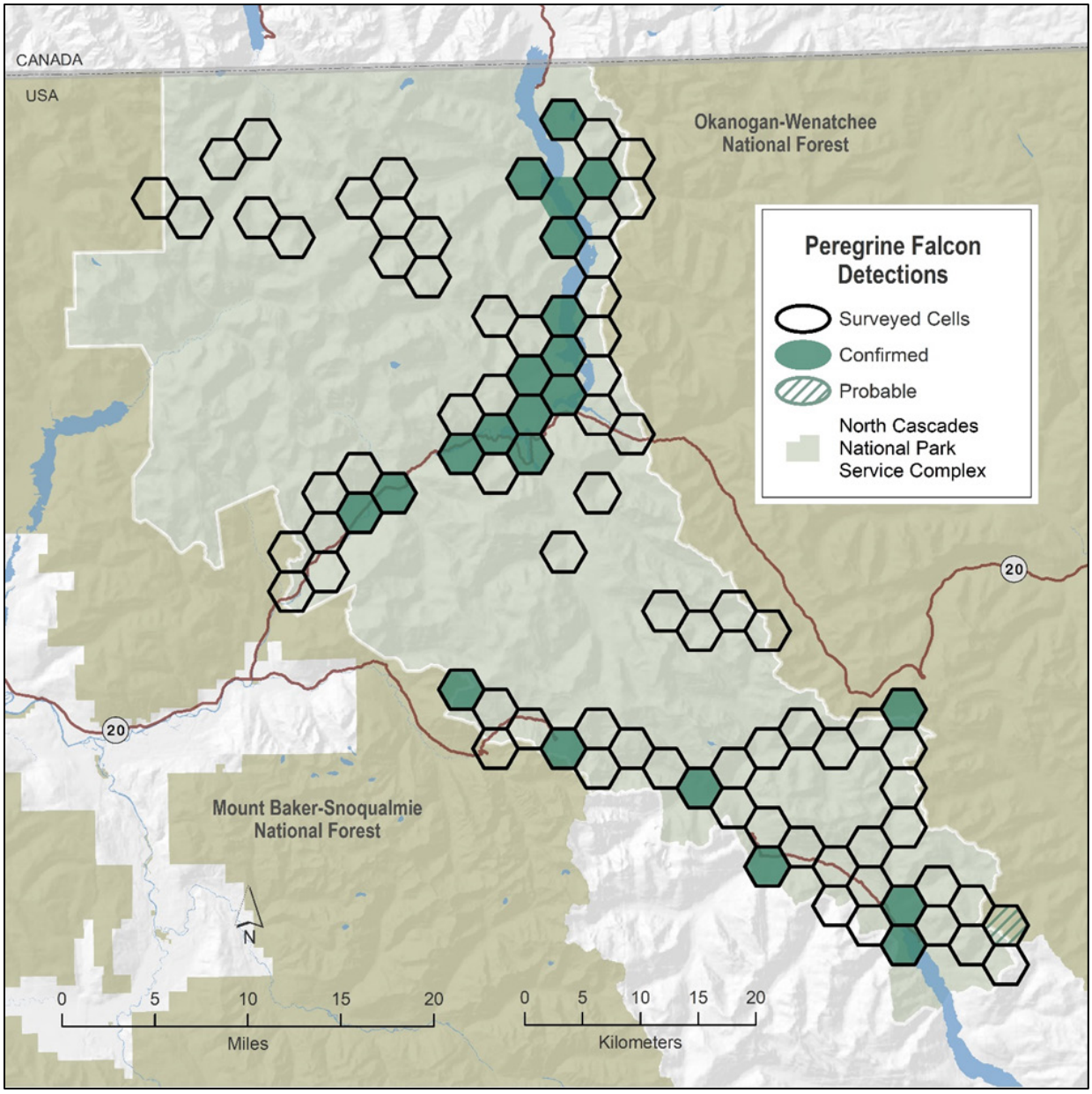


Figure 21. Survey locations, and Peregrine falcon (*Falco peregrinus*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 11). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 11. Peregrine falcon (*Falco peregrinus*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2005–2020	Landbirds point counts and incidental detections at 1,222 points (visual and acoustic observations) [a]	Annual or alternate year transects, May–July	Holmgren et al. 2015–2017, 2019–2021; Ray et al. 2017, 2018; NPS unpublished data
2006–2020	Nest site monitoring at 3–7 nests (visual observations)	3–4 times per nest each year, April–July	Christophersen and Ransom 2022

[a] Surveys were not designed for optimal detection of this species.

The peregrine falcon is taxonomically divided into three subspecies in North America. The largest and darkest colored of the three, the Peale's peregrine (*F. p. pealei*), and the slightly smaller and lighter colored American peregrine (*F. p. anatum*) are known to breed in Washington, while the smallest and lightest colored Arctic peregrine (*F. p. tundrius*) is a migrant or rare winter resident (Varland et al. 2012). Although there are mixed reviews, the Peale's peregrine falcon is most often associated with wet coastal Washington including the Olympic Peninsula outer coast and throughout the San Juan Islands (Dawson and Bowles 1909, Beebe 1960). The American peregrine falcon is known to breed in the Cascade Mountains, the San Juan Islands, major urban areas of the Puget Sound, and farther inland across eastern Washington (Anderson 1993).

Although peregrine falcon breeding occurs throughout much of Washington, the greatest concentration of nesting sites occurs in the western part of the state, primarily in the lowlands of the northern Puget Sound Basin, the San Juan Islands, and the outer northern coast of the Olympic Peninsula. Wintering migrants are also attracted to western Washington where the mild maritime climate and diverse habitat support high densities of prey species during the winter (Anderson and Herman 2005).

Peregrine falcons faced global decline after the introduction of organochloride pesticides in the 1940s, particularly DDT, which caused eggshell thinning and embryonic mortality (Hickey 1969). By the mid-1960s, peregrine falcon populations had plummeted across North America. They had been largely extirpated from the east coast, substantially reduced in the Rockies, and reduced by half in Arctic populations (Fyfe et al. 1976). An inventory published in 1941 reported only 212 nest ledges occupied east of the Mississippi River (Hickey 1942). Subsequently, peregrine falcons were listed as federally endangered in 1970. Later in the 1970s, the United States banned the use of DDT and some other environmental contaminants and enacted a number of other legislative actions protecting peregrine falcons. As recovery efforts showed momentous success and populations began to rebound, peregrine falcons were gradually removed from federal and state endangered species listings but remain protected under other legislative provisions.

In the early 1980s, NPS biologists evaluated 13 areas of potential peregrine falcon nesting habitat primarily in Ross Lake NRA (Bjorklund 1984). Park biologists surveyed the highest rated areas for peregrine falcon activity from the ground beginning in 1985 and by helicopter in 1986 and 1987.

From these surveys, no peregrine falcons or their nests were identified (Bjorklund and Drummond 1987). In 2006, as part of a nationwide monitoring plan to survey a sample of known and suspect peregrine falcon territories following their delisting, helicopter surveys were again conducted within Ross Lake NRA by WDFW biologists. Surveys were restricted to areas of recently discovered nesting activity, recently reported peregrine falcon sightings, and suitable nesting habitat determined to be worth the survey effort. Surveys continued in 2007 and 2009 by WDFW biologists. These efforts resulted in six peregrine falcon territories identified within Ross Lake NRA. Surveys continued from the ground and were conducted primarily by WDFW and Seattle City Light (SCL) staff from 2010–2014. No surveys were conducted in 2008 and 2013. Beginning in 2015, NPS biologists, with WDFW and SCL staff, continued to survey the six known nesting territories and continued through 2020 (Christophersen and Ransom 2022) (Table 11).

Collectively, over the 15-year period from 2006–2020, 3–7 peregrine falcon nesting territories were surveyed annually and had an occupancy rate of 72.5% (Christophersen and Ransom 2022), which is slightly less than 79–82% occupancy reported elsewhere in Washington (Hayes and Buchanan 2002, Vekasy and Hayes 2016). The successful nesting rate for peregrine falcons in the Park was 51%, and other recovering populations ranged 62–83% (Tordoff and Redig 1997, Enderson et al. 2012). Productivity averaged 1.3 young per nest, which is within the range suggested for a sustainable population with potential for growth (1.25–1.5 young fledged per pair) (Grier and Barclay 1988, Wootton and Bell 1992). Since 2009, the number of occupied peregrine falcon territories, successful nests, and peregrine falcon productivity in the Park exhibited an apparent decline (Christophersen and Ransom 2022).

Habitat Requirements

The most common characteristic of peregrine falcon nesting habitat includes the presence of a prominent cliff. To a lesser degree, tall buildings or other man-made structures and cliffs, such as steel bridges and rock quarries, are also used as nesting sites (Cade and Bird 1990). Cliffs and tall structures serve as nesting and perching platforms while offering a sweeping view of the surrounding landscape for vigilant hunting. The cliff or structure also requires adequate ledges or naturally weathered indentations, often referred to as “potholes”, for suitable nesting. Like other members of the falcon family, peregrines do not build nests. Instead, they form a hollow or a “scrape” in soil, vegetation, loose rock or gravel on the cliff ledge. The secluded ledge or pothole feature also minimizes access from potential mammalian predators. The ledge typically has an overhang above that offers protection from heavy rain and other inclement weather (Campbell et al. 1990, Johnsgard 1990). The ledge must be large enough to support a brood of 1–4 young as they develop. Nest sites are typically found in proximity to an open body of water such as a river, lake, marsh or marine waters (Ratcliffe 1993). This type of open landscape offers unobstructed hunting where terrestrial birds and waterfowl can be more easily captured. Factors that limit habitat suitability for peregrine falcons can vary between locations. Primary factors may include limited access to prey, such as shorebirds and passerines, and limited nest site availability (Herman and Willard 1978). With the peregrine falcon’s wide geographic distribution and expansive home range size, the Park does not appear to represent a significant area of suitable nesting habitat (Hoffman et al. 2015).

Peregrine falcons are considered solitary, long-distance migrants (White 1969); however, populations at temperate latitudes and coastal areas are largely resident or engage in short-distance movements (Ambrose and Ulvi 1990, Thelander 1977). They are not known to occur in the Park during the non-breeding season and at least some summer individuals are thought to move short distances from the Cascade mountains to down-valley agricultural and coastal wintering areas (Debruyn, pers. comm). Winter habitat is varied and may include coastal marshes, river valleys, lakeshores, wooded islands and shoreline forests that support high densities of shorebirds, waterfowl, and other small- to medium-sized birds (Anderson and Herman 2005). This may also include habitat that has been altered by human intervention such as open agricultural fields (particularly with standing water), and airports and cities where non-native bird species such as rock doves and European starlings are abundant (White et al. 2002). Another important element of preferred winter habitat is the availability of roost sites. Information from a winter habitat study in the Samish Flats of Skagit County, Washington showed strong site fidelity to roost sites and continuous use of them throughout the winter, with some individuals flying daily over open water to nearby offshore islands between 4 and 14.4 km (2.5–9 mi) from day use areas (Anderson and DeBruyn 1979, Anderson et al. 1980).

Diet

The diet of peregrine falcons is varied and is usually selected in relation to availability. In general, passerines, shorebirds, and waterfowl are the most frequently hunted prey (Ratcliffe 1993), which are often found near cliffs and aquatic features in the vicinity of nest sites. Other components of their diet may include small mammals and insects. Snyder and Wiley (1976) found on average, the peregrine falcon diet consists of 76.7% birds, 3.4% mammals, and 19.8% invertebrates.

Reproduction

Peregrine falcons are generally monogamous and defend their territory as solitary nesters, with some breeding pairs in Washington showing a high degree of nest site fidelity (P. DeBruyn, pers. comm.). Most peregrine falcons begin to mate between two and four years old; however, age at first breeding can vary, even within the same population and among genders. The female peregrine usually lays only one clutch of three or four eggs at 2–3 day intervals (Johnsgard 1990). Anderson (1993) estimated egg laying dates in Washington's San Juan Islands ranged from 1–7 April to 4 May and hatching dates were estimated to range from 5–9 May to 5–6 June. Incubation occurs for 29–32 days with both adults contributing, although there can be variation in time allocated according to the phase of the breeding period. Hunt (1979) found the female spent more time incubating during the early phase of incubation, whereas the male was more involved in the five to 10 days prior to hatching. After hatching, the female participates in the majority of the brooding and feeding of the young and later in the nesting stage when the chicks are more developed both adults hunt and feed the young (Cade et al. 1996). The young continue to increase in bulk, develop feathers and become more mobile around the nest ledge until they are ready to fledge at about 35–42 days of age (Johnsgard 1990, Tekiela 2011). Nesting success and chick survival to fledging is highly variable in the same region between sites and years (Calef and Heard 1979). Fledging dates vary in Washington, but most fledging in the San Juan Islands occurred between the third week of June and the first week of July (Anderson 1993, 1996). Fledging in the Park tends to occur somewhat later and can take place up to 15 July (Christophersen and Ransom 2022). Fledged young continue their dependence on the adults

for several weeks as they gradually learn to hunt on their own and prepare for dispersal. Autumn migration begins by mid-August in the Pacific Northwest and continues through October (Beebe 1960, Anderson et al. 1988, Campbell et al. 1990). This movement corresponds closely to the arrival of migrant flocks of shorebirds and waterfowl throughout the Puget Sound and outer Washington coast.

Protective Status

Because of dramatic population declines resulting from environmental contaminants, the peregrine falcon was listed as a federally endangered species in 1970 under the Endangered Species Protection Act, a precursor to the current ESA. A combination of legislation banning the use of organochloride pesticides, along with reintroduction efforts through captive breeding and release programs, led to the start of peregrine falcon recovery by the late 1970s (Cade et al. 1988). As a result of this remarkable comeback and other recovery goals being met in all recovery regions, the peregrine falcon was removed from the USFWS's list of Threatened and Endangered Species in 1999 (50 CFR §17.46542). They continue to be listed as a federal Species of Concern and remain under the protection of the Migratory Bird Treaty Act (Mesta 1999).

Peregrine falcons were listed as a Washington State endangered species in 1980, with only eight nesting pairs identified statewide. As the population increased and recovery goals were met, they were removed from the state endangered list in 2002 and reclassified as a Sensitive Species. The most recent surveys of peregrine falcon nesting territories in Washington identified 108 territories in 2010, an increase from 91 sites identified in 2006, and 72 sites in 2001. With this significant increase in recovery, peregrine falcons were removed from the Washington State Sensitive Species List in 2016 (Vekasy and Hayes 2016). The species remains classified as "protected wildlife" under state law (WAC 220-200-100) and identified as a Species of Greatest Conservation Need under the Washington State Wildlife Plan. They are considered a Management Priority Species in the Park and known nesting territories continue to be monitored annually for nesting success and productivity.

Known Threats

Human Disturbance

Human disturbance can affect different nesting raptors in different ways. The three reservoirs (Gorge, Diablo and Ross Lakes) and surrounding areas within Ross Lake NRA support large amounts of recreational use, primarily as boating, fishing, hiking and sport climbing opportunities. Much of this activity is in close proximity to, or within, known peregrine falcon nesting territories. Peregrine falcons are known to be adaptable and highly mobile raptors, which allow them to successfully nest in a variety of conditions and subsequently adjust to some human disturbances. Depending on the type of activity, particular individuals, and the period of time it takes for nesting raptors to adjust to the disturbance, some human activities may jeopardize their nesting success, especially during the critical early nesting period. Although there are various suggested management recommendations for protective buffers relating to peregrine falcon nesting, Currie and Elliott (1997) recommended 1 km (0.6 mi.) as a conservative disturbance threshold distance. Numerous recreational motorboats on the reservoirs of Ross Lake pass by peregrine falcon nesting territories within this suggested threshold distance on a daily basis during the busy summer season. Anecdotal observations suggest that

peregrine falcons tend to flush when a boat approaches too close, yet it is unknown if or to what extent this activity may affect nesting success in the Park.

There has been a recent increase in sport climbing activity in the Skagit River gorge within Ross Lake NRA. Human disturbance could be influencing peregrine falcon nesting success where territories overlap with climbing activity. There are currently four approved Climbing Management Units (CMUs) within the Skagit River gorge (36 CFR §1.5: NPS 2023). There are also three known peregrine falcon nesting territories that are on or near developed climbing crags within these CMUs. In addition, two areas that are not authorized CMUs have recently been discovered to contain illegal fixed climbing anchors (Space Wall and Canoehalem): one of which (Canoehalem) has a history of successful peregrine falcon nesting. However, there has been no known peregrine falcon nesting at this crag since it was illegally developed with bolted routes in 2018, and one route was bolted directly through an established peregrine falcon eyrie. This activity suggests that human disturbance may be directly responsible for the lack of nesting at this site since 2018. Other factors such as nearby nesting ravens, which can sometimes be precipitated by human activity, may have also contributed to changes in peregrine falcon nest site selection.

Weather

Rainfall and cold temperatures during the critical spring egg laying, incubation and hatching stages have been identified as potential drivers of reduced reproduction in peregrine falcons (Ratcliffe 1984, Mearns and Newton 1988, Olsen and Olsen 1989). Frequency of heavy rain often has a much greater impact on nestling survival than the total amount of precipitation recorded during the rearing period (Anctil et al. 2014). Although weather was not investigated thoroughly within the scope of peregrine falcon monitoring efforts in the Park, it could be a major factor in explaining low productivity in some years (Christophersen and Ransom 2022).

Environmental Contaminants

Data on bioaccumulation of toxins and environmental contaminants in raptors in the Park are scarce. Three raptors, including two peregrine falcons and one bald eagle and one adult common loon have been recovered and tested for toxins and contaminants. All three raptor carcasses tested negative for West Nile virus and avian influenza. During 2016 peregrine falcon surveys, a fledged juvenile peregrine falcon was found dead at the base of the cliff at the Newhalem eyrie in Ross Lake NRA. Lab analyses of liver, brain, and kidney tissue detected lead (0.70 ppm) and mercury (0.18 ppm), and this bird likely died from starvation/malnutrition. It is unclear if toxicity contributed to premature death of this fledgling, but it does indicate the presence of lead in the local food chain and that peregrine falcons, as higher trophic consumers, are vulnerable. A second peregrine falcon mortality was discovered in 2019 near a cliff adjacent to Ross Dam in Ross Lake NRA. This area has recent peregrine falcon occurrence records, but no pair occupancy or nesting has been confirmed to date. Brain and liver tissue from this adult carcass were analyzed and elevated iron was detected (2,400 ppm). Elevated liver iron concentrations can result from congestion, hemolysis, overexposure to iron, iron storage abnormalities or starvation. Manganese (5.6 ppm), mercury (4.7 ppm), molybdenum (0.51 ppm), zinc (83 ppm), copper (5 ppm), cadmium (0.42 ppm), and methyl mercury (1500 ppb) were also detected in this falcon. This bird was also screeened for 8 PCBs, which are highly toxic

industrial compounds. One PCB was detected, Aroclor 1260 (10 ppm), indicating a likely exposure in the local food chain. Aroclor 1260 may come from a variety of industrial applications but is most commonly used in large capacitors and transformers (National Library of Medicine, 2021). This falcon was also found to have an apparent *Aspergillosis* infection in its lungs, which may have become prevalent following malnutrition.

An emerging and pervasive environmental contaminant, microplastic, is also a potential threat to raptors through bioaccumulation (Wagner et al. 2014). Microplastics are plastic fragments smaller than 5 mm that may permeate the environment through deposition. These microplastics are of great ecological concern in aquatic systems because they can be easily ingested and accumulate as indigestible matter throughout a food web. Study of this issue in aquatics organisms is ongoing in the Park. Primary microplastics analyses (tissue digestions and ultraviolet detection) of gastrointestinal tracts from the 2019 peregrine falcon, 2019 bald eagle and 2019 common loon were conducted at the University of Alaska in 2020. No microplastics were detected in any of these samples. While encouraging, this small sample size and basic analysis does not rule out the potentially serious risk that bioaccumulation of microplastics in raptors poses.

Climate Change

During the next 20 to 40 years, the climate of the Pacific Northwest is projected to change significantly, and with potential consequences for forest ecosystems. Predicted changes include warmer, drier summers, and warmer, wetter autumns and winters, resulting in diminished snowpack, earlier snowmelt, and an increase in extreme heat waves and precipitation events (Littell et al. 2009, Salathé et al. 2010, Mote and Salathé 2010). Notably, an extreme heat wave impacted western Washington in late June 2021, and regional raptor nest monitoring reports from camera-monitored nests included fledgling mortalities from this single weather event.

One of the largest potential effects on Pacific Northwest forests is likely to come from an increase in fire frequency, duration, and severity (Littell et al. 2009, Westerling et al. 2006). The winter of 2014–2015 was a record-low snowpack season, but it is unknown if this factor alone presented a direct relationship with low productivity during the nesting season to follow. However, the reduced winter snowpack was followed by an unusually dry summer with 34 wildfire ignitions in the Park during summer 2015 (NPS unpublished data). Most notably, the Goodell fire actively burned throughout much of the Newhalem and Gorge Lake peregrine falcon nesting territories with high burn severity in some areas. That fire was also near the Diablo nesting territory. The burn intensity resulted in heavy and persistent smoke accumulation in the area. Smoke from two additional wildland fires drifted heavily over three other peregrine falcon territories along Ross Lake, thus all six peregrine falcon nesting territories surveyed were impacted by wildland fires during the late nesting season and may have adversely affected 2015 nesting success, during which no reproduction was documented. Further, the burned area near peregrine falcon eyries in the Skagit River gorge has dramatically reduced the visual buffer that vegetation previously provided, which may potentially cause a greater threat to nesting peregrine falcons from predators and human activities.

Activities that occur during fire suppression, such as aviation operations, vehicles, construction of fire breaks, fire crew camps, smoke and retardant drops could also disturb nesting peregrine falcons

and adversely affect nesting success. Prescribed burns may also cause disturbance to occupied eyries, but generally more control can be exercised through recommendations and best management practices that can help to restrict or minimize impacts.

Habitat Loss and Alteration

In the Park, tree and other vegetation mortality resulting from the 2015 Goodell fire in the Skagit River gorge not only reduced the visual buffer but may have also temporarily affected density and composition of avian fauna that peregrine falcons depend on for prey. It is unknown whether there has been a recent shift in passerine availability or selection, but anecdotal information suggests a recent increase in waterfowl and gull carcasses at the base of the Newhalem peregrine falcon eyrie, in particular. Other peregrine falcon nesting territories within or outside of the Park are also subject to past or future wildland fire habitat alteration.

Loss of wetlands that attract shorebirds and waterfowl could be detrimental to wintering peregrine falcons that depend on these prey populations. Agricultural development has mixed reviews, but likely depends on the level of human disturbance and prey availability (Young 1989).

Conservation and Research Needs

Information on the population status of peregrine falcons and other raptors is important for the adaptive management of Park resources. As top avian predators, raptors act as a sentinel of larger ecological change, as they are especially vulnerable to environmental contaminants, human disturbances, and a changing climate. New threats to raptors and ecosystems emerge on a routine basis, and many are conflated with changing climate. Unexpected changes such as the massive influx of Park visitors stemming from broad societal response to the SARS-CoV-2 pandemic, can quickly change the success or failure of raptor nesting in any given year. Earlier snowmelt, longer growing season, earlier plant flowering and invertebrate emergence, and shifting migration phenology could all influence food availability and initiation of earlier breeding to synchronize with resource availability. These same biological shifts also portend earlier, and more frequent, visitor access to sensitive nesting areas in the Park. Management of resources for Park managers is complex, interconnected, dynamic, and often time-sensitive. Some potential risks can be mitigated with detailed plans through time, such as development of a comprehensive sport climbing management plan to minimize adverse impacts to known peregrine falcon nesting areas. Some risks, such as wildland fire, are mediated by real-time information connecting fire crews with locations of sensitive resources like raptor nests, that need specific types of protection. This is accomplished through annual (or more frequent) updates to the Park Resource Advisors Guide. Some risks, such as environmental contamination through atmospheric deposition, require regional, continental, or global scale mitigations to address. Other environmental contaminants may arise locally and be traced and mitigated at the source. One major limitation to understanding peregrine falcon ecology and conservation needs in the Park is a lack of data on prey abundance. This has not historically been conducted due to limited resources; however, some important long-term monitoring data on fish abundance and landbird trends could provide insight. This type of rigorous monitoring conducted simultaneously with nesting surveys could help emphasize the relevance of such data for understanding the primary drivers of peregrine falcon and other raptor ecology.

Specific conservation needs for peregrine falcons in the Park include mitigations for aircraft use near active nests, nest protection measures during wildland fire events, visitor use management around active nests (especially recreational climbing), anti-strike devices on electrical transmission lines in flight routes around nests and primary hunting areas, and focused monitoring of environmental contaminants, heavy metals, and microplastics in both falcons and their prey. Peregrine falcon carcasses should also be tested for West Nile virus and avian influenza. Routine management, facilities, and infrastructure operations conducted in the Park can affect peregrine falcons, but adherence to noise and disturbance guidelines around nests may help mitigate local impacts (Appendix A).

Chapter 3. Insects

Pollinators

Pollinators are an essential component of healthy ecosystems and functioning agricultural communities. Recently, scientists have documented alarming declines in abundance and distribution of many insect pollinators across the world, and many predict this will continue with changing climate and habitat loss (Forister et al., 2021; Memmott et al., 2007). The complex terrain of the Park provides a diverse array of habitats for pollinators today, which may facilitate future adaptation and survival with changing climate. Although pollinator surveys within the Park have been limited both spatially and temporally, they have documented a highly diverse community of bees, syrphid flies, butterflies, and moths (Figure 22, Appendix B).



Figure 22. *Platarctia parthenos* (Upper left: photograph by Dennis Strenge), *Bombus bifarius* (Upper right: photograph by Jessica Rykken), *Sphaerophoria abbreviata/assymmetrica* (Lower left: photograph by Jessica Rykken), and Hydaspe Fritillary (Lower right: photograph by Regina Rochefort), in North Cascades National Park, WA.

Butterflies

Fifty-nine butterfly species have been documented in the Park from two BioBlitz's (2014, 2016), visitor observations, and repeated surveys by the Cascades Butterfly Project (CBP) (Appendix B) (BAMONA 2022, iNaturalist 2022, Rochefort and McLaughlin 2017). Thirty-seven species were confirmed by CBP surveys; in addition to survey tallies, voucher documentation includes photographs of all species and collection specimens of 13 species. Visitor observations in iNaturalist or Butterflies and Moths of North America (BAMONA) and accompanied by photographs confirmed an additional 12 species. The other 10 species include four species designated as *Probable* because they were documented on CBP surveys close to the Park border, one observation was submitted to the Park without a photograph, and five species were are likely present because their distributional range indicates they have a high probability of being within Park boundaries (Pyle and LaBar, 2018; Jon Pelham pers. comm., April 28, 2021).

The CBP is a long-term, citizen science monitoring project in which butterfly abundances are documented 1–2 times per week during the summer (late June to mid-September). Surveys are conducted using the Pollard Walk method to estimate relative abundance of butterflies (Rochefort and McLaughlin 2017, Rochefort 2017). Ten permanent survey transects were established 2011; two are located in North Cascades National Park, two in Mount Baker-Snoqualmie National Forest, one in Okanagan-Wenatchee National Forest, and five in Mount Rainier National Park. All survey sites are located along maintained trails in subalpine meadows. The primary goal of the project was to document the effects of climate change on subalpine ecosystems using butterflies and plant phenology as indicators of change. The development and distribution of plants and butterflies are directly influenced by temperature and precipitation making them sensitive indicators of environmental effects. We are using data from two sites in the Park (Cascade Pass and Easy Pass) in this report. The Cascade Pass site was established in 2012 and the 1 km route extend from 1,665 m to 1,859 m in elevation. It was monitored 56 times between 2012 and 2020, resulting in 25 species detected, and a total of 2,083 butterflies (Appendix Table B1). The Easy Pass survey route ranges from 1,864 m to 1,996 m in elevation and has been surveyed 60 times 2011–2018, 2020–2021, documenting 32 species and 3,274 individuals (Appendix Table B1).

Volunteer observations of butterflies have been collected during two BioBlitz's: North Cascades 2014 BioBlitz and the National Park Service Centennial BioBlitz in 2016. Additionally, the CBP has requested volunteer observations annually since 2011. Volunteer observations have been recorded in iNaturalist and BAMONA, which are web-based platforms where observers upload their observation data. We have only used data from these sites that has been accompanied by photographs and have either been verified by experts (BAMONA) or has been verified by multiple observers and reach "Research Grade" in iNaturalist. Thirty-three species have been documented in the Park by iNaturalist and 14 species in BAMONA.

Moths

Moth surveys were primarily conducted along the highway corridor of Ross Lake NRA during the 2014 BioBlitz. Dennis Strenge spent two nights in Newhalem using a light trap to sample, collect and identify moths (Appendix Table B2). He confirmed 67 species with specimens and photographed

most of them. David Droppers (Western Washington University) conducted light trap surveys one night at the Newhalem Visitor Center and surveys in the Colonial Creek Campground and recorded 13 additional species, all accompanied by photographs. During the 2016 BioBlitz in Stehekin, Dr. Jamie Strange's (Ohio State University) crew documented one more moth species, *Adela septentrionella*, bringing recorded species count to 81.

Syrphid Flies and Bees

In 2014, Dr. Jessica Rykken (Harvard University) conducted surveys of bees and syrphid flies at 28 sites in the Park between August 4 and 18 (Appendix Table B3, B4). Sampling was conducted at a variety of elevations ranging from 145 m along the Skagit River in Newhalem to 2,010 m at Easy Pass. General sampling areas included both managed and natural landscapes including the Newhalem area, sites near the North Cascades Highway, and Desolation Peak in Ross Lake NRA; Maple Pass, Easy Pass, Cascade Pass, and Monogram Lake in the south unit of North Cascades National Park; and McAlester Lake and South Pass in Lake Chelan NRA.

Syrphid fly and bee identification generally required examination with a microscope, so insect nets and bee bowls were utilized to collect insects prior to washing, drying and pinning specimens for identification and curatorial voucher collections. Bee bowls of three colors (white, blue and yellow) were set out for 6 hours to attract pollinators. Insect nets were used for active surveys at each site (Rykken 2018). Dr. Rykken identified all specimens with assistance in verification or identification of syrphid flies from Michelle Locke (American Museum of Natural History) and Dr. Andrew Young (University of Guelph). Bees were identified by Dr. Rykken with assistance, or verification, as needed from: Robbin Thorp (University of California-Davis), or entomologists from the U.S. Department of Agriculture (USDA) Pollinating Insects – Biology and Systematics Research Unit, Logan Utah: Terry Griswold, Harold Ikerd, James Strange, Jonathan Koch, Schuyler Burrows, and Brian Roznick. Specimens were deposited at the Park, the USDA National Pollinating Insect Collection in Logan, Utah (bees) and syrphid fly specimens were deposited at the National Museum of Natural History, Smithsonian Institution, Washington, D.C.

During Dr. Rykken's surveys, 29 species of syrphid flies were documented in the Park including one invasive, nonnative species (*Syritta pipiens*) (Appendix Table B4). She also documented 36 species of bees including the nonnative honeybee (*Apis mellifera*). Additional surveys for bees were conducted by Dr. Strange and his crew during the 2016 BioBlitz and during surveys in 2013 (Appendix Table B3). They confirmed many species observed by Dr. Rykken but two additional species: the Western Bumblebee (*Bombus occidentalis*) in Stehekin in 2016 and the Purple Miner Bees (*Andrena prunorum*).

Habitat Requirements

Insect pollinators are distributed across all elevations and habitats with the Park. Surveys to date can only describe the types of habitats that have been inventoried, but no habitat within the Park is without insect pollinators. Surveys of bees and syrphid flies found that species diversity increased at higher elevations (Strange et al. 2013, Rykken 2018). Some species were restricted to specific elevation ranges or habitats, but Rykken found a 36% overlap between low and high elevations.

Bumblebees composed 45% of all bee taxa in Rykken's surveys, with the remainder composed of solitary, cuckoo, or other social bees distributed across all elevations.

Butterfly surveys have focused on subalpine meadows, but incidental observations by NPS staff and volunteers have encompassed low elevation forest openings and riparian corridors. Moths are often found near wooded habitat and are more active at night—adding a level of difficulty to inventories with light traps. Park surveys have only been conducted along the forest edges in the Ross Lake NRA corridor during one month in 2014.

Protective Status

There are three invertebrate pollinator species potentially present in the Park that have special status: Propertius Duskywing butterfly (*Erynnis propertius*), Johnson's Hairstreak butterfly (*Callophrys johnsoni*), and Western Bumble Bee (*Bombus occidentalis*). The Propertius Duskywing butterfly is a listed as a Species of Greatest Conservation Need in the State of Washington (WDFW 2015) and is a NPS Management Priority species. It has been documented once in Newhalem and future surveys of dry, open areas near shrublands such as Hozomeen, the east side of Ross Lake, and Newhalem are needed.

Johnson's Hairstreak butterfly is Candidate species in Washington and a NPS Management Priority species. Its range may include the Park. Endemic to Cascadian/Sierran old-growth forests at elevations from 0 to 1,524 m, its host plant is Dwarf Mistletoe (*Arceuthobium campylopodum*) which occurs on old-growth Western Hemlock and Douglas-fir. Surveys for this species have not been conducted in the Park, but relatively recent surveys in Snohomish County detected this species and surveys within the Park are warranted.

Western Bumble Bee is a Candidate species in the State of Washington and is a NPS Management Priority Species. This species is currently under review for federal listing under the ESA, following a 90-day review finding by USFWS (2016c). In 2013, Strange et al. searched sites of historic western bumblebee observations in the Park but could not locate any individuals. In 2016 in Stehekin, Strange et al. did locate several male and female western bumblebees during the Park BioBlitz.

Known Threats

Alarming world-wide declines in insects have been attributed to habitat loss, climate change, parasites, invasive species, and pesticides (National Research Council 2007, Vogel 2017, Forister et al. 2021). High-elevation ecosystems and montane wetlands may be some of the most sensitive areas due to warming temperatures and decreasing snowpacks resulting in drier summer environments (Strange et al 3013, Ryan et al. 2014). These small meadows are often referred to as sky-islands due to their small and disjunct distribution. As tree establishment expands in elevation and wetland hydro-periods decrease, herbaceous pollinator host plants will be displaced and tree stands will become barriers to insect migration. Warming temperatures will also initiate earlier flowering of host and nectar plants and development of butterflies. Climate change represents the most significant threat to pollinators in the Park.

Although most documented effects of parasites, viruses, and pesticides center around honeybees (*Apis* spp.) due to their commercial production, many of these effects are presumed to be similar in wild species and could be present in the Park. Bumble bees can be infested by parasites such as protozoans *Nosema bombi* and *Crithidia bombi*, as well as the tracheal mite *Locustacris buchneri* (National Research Council 2007). Viral threats to bumble bees include emerging threats such as deformed wing virus (National Research Council 2007). Pesticides form a ubiquitous threat to non-target species such as bumble bees and other pollinators, but direct links are nearly impossible to establish because deaths of wild species are rarely reported. Ground-nesting bumble bees are especially susceptible to pesticides applied to lawns (National Research Council 2007). Pesticides such as imidacloprid (Morandin and Winston 2005) and clothianidin (Franklin et al. 2004) have been found to impact foraging and pollinating. Pesticide applications early in the spring and summer may especially impact bumble bee numbers because queens are foraging and colonies are very small during that time period (Thompson 2001).

Conservation and Research Needs

While the threats of habitat loss, climate change, and pesticides to pollinators have been documented, research is still needed to understand the primary drivers, geographic scope, species distribution changes, and ramifications within ecosystems processes (e.g., decoupling of plant-pollinator systems) (Vogel 2017, Forister et al. 2021). The Park is a perfect laboratory for long-term studies to answer many of these questions due to its complex terrain and diversity of habitats and climates, and largely wild landscape.

However, pollinator inventories have only been conducted in a few specific locations and habitats in the Park. Both Rykken and Strange documented new species for the Park even with very limited survey times, underscoring how understudied these taxa are. Rykken (2018) documented three new vouchered species records for North Cascades (*Bombus appositus, B. insularis*, and *B. vandykei*). Syrphid fly and moth lists only exist from 1–2 surveys and Park lists are entirely based on species observed during those surveys. The NPS cannot develop a complete potential list for the Park as there are probably many more species within the Park than currently known and some may present management concerns. A comprehensive invertebrate pollinator inventory with a deliberate sampling scheme targeting elevation and precipitation gradients is needed. This inventory would develop complete species lists, estimate patterns of abundance and associate pollinator species with plant communities or host plants. This should be followed by repeated surveys to understand which species or habitats are most sensitive to changing climates and document population trends.

The CBP focuses on high-elevation areas, looking specifically at seasonal trends in butterfly abundance and phenology as well as host plant flowering patterns. While earlier flowering times have been documented in warmer drier years such as 2015, lower elevation study sites are needed to really answer whether butterfly, bees, and syrphid fly populations are indeed moving up in elevation and latitude (Parmesan 1996, 2006) from lower elevation sites or latitudes. Some inventories and monitoring could be conducted through partnerships with community scientists, web-based platforms, while others will require partnership with academic institutions and long-term monitoring by Park scientists.

Chapter 4. Mammals

American pika (*Ochotona princeps*)

The American pika (*Ochotona princeps*) is a small herbivorous mammal that resides in mountainous regions of western North America, including the Park (Figure 23). Pikas primarily inhabit talus slopes that vary widely in size. These slopes are surrounded by a diversity of herbaceous plants that pikas consume during summer and store in hay piles amongst talus cavities as winter food caches (Smith and Weston 1990). As non-hibernating lagomorphs, pikas have developed thick coats of fur and a high resting metabolism to protect them from the cold, high-elevation winters. During hot summer months, talus habitat helps moderate temperatures and provides shade assisting in pika thermoregulation (Smith and Weston 1990, Beever et al. 2010). The American pika was considered for listing under the ESA in 2010 and petitioned for listing again in 2016 but was not listed because observed extirpations have occurred in low-elevation, marginal habitat rather than across a geographic and elevational range substantial enough to merit listing (USFWS 2010a, 2016d).



Figure 23. Pika voucher record NOCA42630, photographed by Jason Ransom/NPS, July 22, 2017, near Cascade Pass, North Cascades National Park, WA.

Pikas have been surveyed repeatedly throughout the Park for over a decade (Figure 24, Table 12). Field crews led by Bruggeman (2012) conducted transect counts of pikas throughout the Park at 115 talus patches (comprising 30 survey areas) in 2009, resurveyed 58 patches (13 survey areas) in 2010, and resurveyed the same 58 patches (13 survey areas) in 2011. In 2009 and 2010, 18 and five

patches, respectively, were resurveyed two to three times to record potential changes in relative pika abundance throughout the summer. From 2013 to 2015, field crews led by Russello et al. (2015) surveyed pikas (with a focus on genotyping from hair samples) in elevational transects along slopes of Pyramid Peak, Sourdough Mountain, and Thornton Lakes. In 2016, field crews led by Johnston et al. (2019a) resurveyed 58 patches (13 survey areas) from the Bruggeman (2012) surveys and 11 patches (11 sites) from the Russello et al. (2015) surveys.

The Bruggeman (2012) surveys also included 1,786 measurements of vegetation cover, 103 surface and sub-surface manual temperature measurements, and data loggers that recorded temperature every 10 minutes in 27 of the surveyed talus patches. Bruggeman (2012) found interactions of elevation, aspect, longitude, and temperature-buffering patch attributes increased probability of pika use; pikas in low-elevation, south-facing, eastern sites experiencing the greatest heat and cold stress. Bruggeman (2012) also found relative pika abundance to positively correlate with the winter Pacific Decadal Oscillation (PDO) index, snow cover persistence, resource availability, elevation, and aspect while correlating negatively with daily maximum summer temperature. These findings indicated variability in winter weather as well as summer heat stress may have driven pika demographics in the Park.

Findings from Johnston et al. (2019a) indicate American pika abundance trends were driven by snow droughts that resulted in negative changes to forage availability at low elevations, positive changes to forage availability at high elevations, and by exposure to cold stress at mid-elevations (Johnston et al. 2019a). Results showing a one-year lag effect of snow droughts as well as a post-hoc analysis using the normalized difference vegetation index (NDVI) indicated that changes to relative pika abundance were likely related to subsequent changes in fecundity rather than direct, overwinter survival (Johnston et al. 2019a).

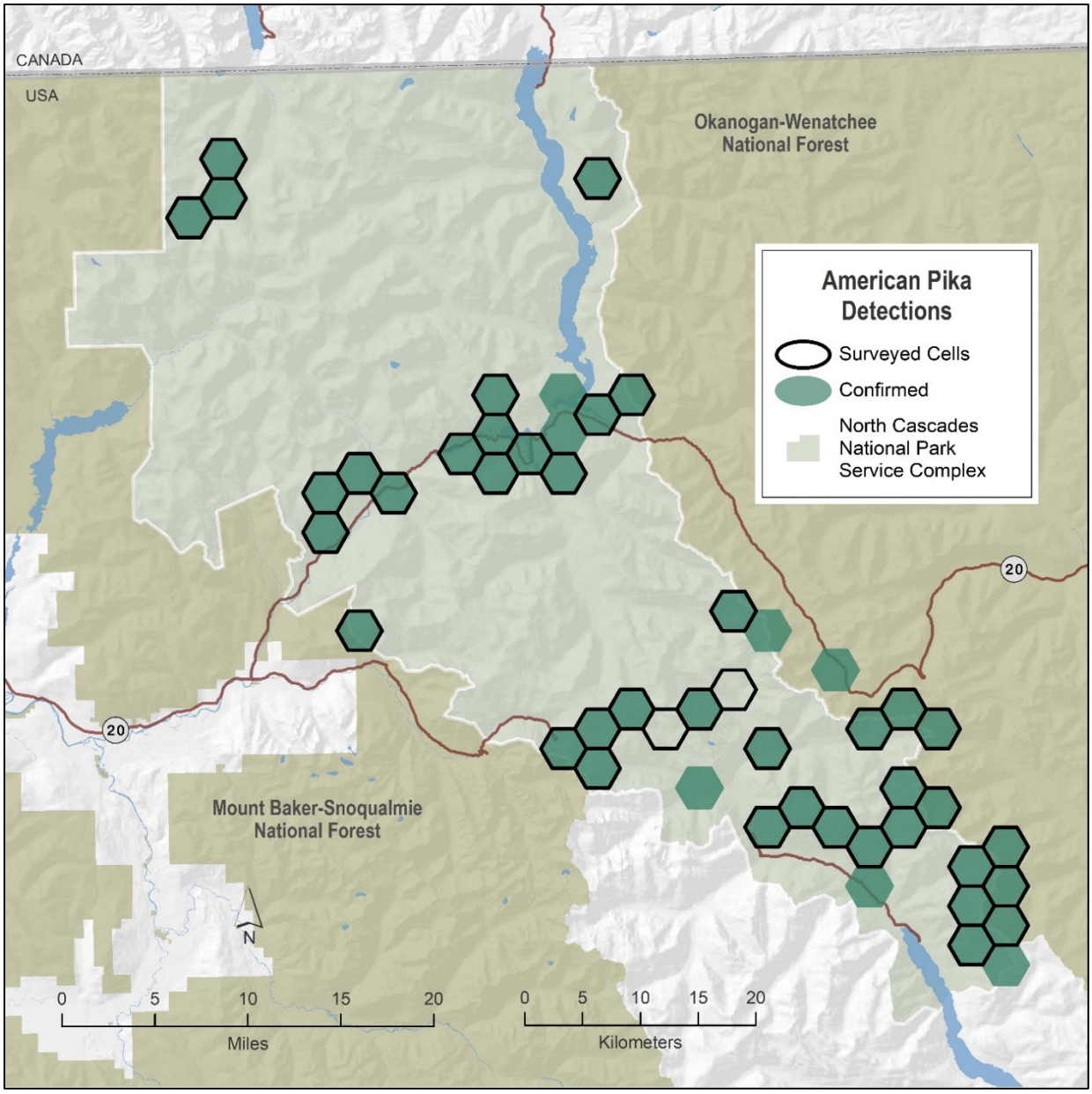


Figure 24. Survey locations, and American pika (*Ochotona princeps*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 12). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 12. American pika (*Ochotona princeps*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2007–2008	Incidental detections at 123 hoary marmot survey points (visual observations)	1–2 times per point, June–September each year	Christophersen 2012
2009–2013	Point surveys at 58–115 points (visual observations)	1–3 times per point, June–October each year	Bruggeman 2014
2013–2016	Point surveys at 11–58 points (visual observations)	1–3 times per point, July–October each year	A. Beers, unpublished data; Johnston et al. 2019a
2015	Point surveys at 13 points (visual observations) at datalogger stations	1 visit per station	NPS unpublished data
2016–2017	Point surveys at 78 points (visual observations)	1 time per point, June–October each year	Johnston et al. 2019a
2018–2019	Incidental detections at 123 hoary marmot survey points (visual observations, trail cameras)	1–3 times per point, June–September each year	Whiles 2021
2020	Point surveys at 13 points (visual observations) at datalogger stations	1 visit per station	NPS unpublished data

Habitat Requirements

American pikas are broken rock obligates most commonly found in talus slopes adjacent to meadows. They primarily live within talus interstices, selecting for boulders with radii between 0.2 and 1.0 m when available, but they also use larger boulders as lookouts and shelters for caches. This species exists in patchily distributed metapopulation structures which reflect the fragmented distribution of their talus-meadow habitat in mountainous landscapes. At the northern end of their range, pikas may be found at elevation ranges from 0 to 3,000 m. At the southern and hotter end of their range they are rarely found below elevations of 2,500 m. Despite current habitat preferences and similarity to other ochotonids inhabiting the Asian steppe, fossil records from the Pleistocene suggest pikas were historically not entirely restricted to talus habitats. Pikas often live in sympatry with other talus-obligate and -facultative rodents from the *Marmota*, *Spermophilus*, and *Tamias* genera. While these sympatric rodents typically dig burrows; pikas do not, but they may dig to enlarge den space within talus interstices (Smith and Weston 1990).

In the Pacific Northwest, American pikas are primarily found in characteristic, high-elevation, broken-rock habitat. However, certain populations thrive in a variety of habitats including the low-elevation Columbia River Gorge, highway-adjacent riprap, and volcanic relic geology (Smith 2020). In the Park, Bruggeman (2012) detected American pikas from 351 to 2,130 m in elevation. Bruggeman (2012) found that talus and large rocks comprised 60.3% of ground cover in surveyed plots; other prevalent types included bryophytes, shrubs, and small rocks, respectively comprising 14.7%, 5.8%, and 5.5% of ground cover.

Diet

American pika diet selection is varied and generalist overall, though some plants are preferred or avoided based on nutritional quality, water content, and toxicity. American pikas also consistently select foods differently throughout the year based on plant phenology. They are central place foragers that exhibit higher fidelity to talus but higher frequency of short foraging bouts into meadows than do other sympatric, talus-living herbivores. Aboveground activity primarily occurs in the morning and late afternoon. Aboveground activity may comprise 30% of the day, and half of this may be spent in surveillance (Barash 1973, Smith and Weston 1990).

Pikas utilize two distinct foraging strategies: feeding and haying. Haying only occurs during summer after breeding and is thought to supplement diet throughout the winter when palatable vegetation becomes scarce. While pikas generally forage within 5 m of talus, they travel significantly farther into meadows when haying than when feeding (Barash 1973, Smith and Weston 1990). In winter, pikas continue foraging, relying especially on cushion plants and possibly lichens that persist close to the talus burrows. Because haypile removal experiments yielded no observed change in pika survivorship, these caches are thought to be necessary only when winters are especially harsh or sustained. Haypile size tends to positively relate to numerous factors; individuals that are male, adult, haying in more abundant meadows, and living in closer proximity to meadows tend to produce larger haypiles. American pikas may also tend to allocate forbs and tall grasses to haypiles more than direct consumption (Smith and Weston 1990).

Reproduction

Like many lagomorphs, American pikas reproduce relatively rapidly and prolifically. They often begin breeding as yearlings, undergo estrus multiple times per year, exhibit postpartum estrus, and generally produce two litters per year. Gestation lasts 30 days. There are typically three individuals per litter especially in characteristic high-elevation pika habitat. Litter size does not tend to vary with age, habitat quality, or litter number but may correspond to average rates of local, overwinter adult mortality. This is because juvenile survival is strongly tied to vacant territory near their place of birth. Weaning success is much lower if a litter has already been produced in the current year, but second litters may succeed when the first of the year did not (Smith and Weston 1990). American pikas are among the fastest developing lagomorph species. Though young spend the first week of life altricial, juvenile pikas tend to open their eyes at nine days of age, are weaned after three to four weeks of age, and begin dissociating from their mother and siblings after four weeks of age. As early as three months of age, pikas reach minimum adult body mass (Smith and Weston 1990).

American pikas maintain territories individually. Adjacent territories are usually occupied by members of the opposite sex, and this pattern persists across generations. The sex ratio of adult pikas is consistently close to 1:1. Territory size does not correlate with sex but is often larger earlier in the summer and correlated with quality and proximity of vegetation. Territory overlap of mate-paired pikas is greatest early in the summer when pikas are breeding (Smith and Weston 1990). American pikas maintain territories with aggression, though actual altercations are not common among familiar conspecifics. Dispersing juveniles, on the other hand, are met with high intraspecific aggression. Juvenile dispersal is also uncommon across non-talus habitats. These challenges lead to a juvenile

dispersal rate of 25%, and the females tend to be the farthest traveled among dispersers (Smith and Weston 1990). Based on historical biogeographical analysis and review of existing observations, Hafner and Sullivan (1995) estimated maximum pika dispersal distance to be 10–20 km and that separation distance between metapopulations is likely 10–100 km.

Protective Status

Despite noting a decreasing population trend, the IUCN listed the pika as "Least Concern" in its 2016 assessment (Smith and Beever 2016). Likewise, NatureServe listed the species as "Secure" globally and in Washington State during the 2016 review of American pikas (NatureServe 2022). This assessment concurs with similar assessments by the USFWS and California Fish and Game Commission, citing a failure to record declines that meet the 80–90% threshold or represent pika trends throughout their range (USFWS 2010a).

In 2010, the USFWS reviewed existing literature and partnered with the National Oceanic and Atmospheric Administration (NOAA) to conduct their own modeling exercises on American pika persistence under climate change. The USFWS ultimately determined that increasing summer temperatures do not pose a significant threat to the American pika in the foreseeable future for three main reasons: 1) recorded (and expected) extirpations have only occurred in low-elevation, marginal habitats rather than across the available elevation gradient within suitable American pika habitat on regional scales; 2) American pikas display considerable behavioral flexibility in mitigating thermal extremes; and 3) assuming 2050 represents the extent of the foreseeable future due to climate models' uncertainty past this point, American pikas are not expected to experience temperature extremes that will result in range-wide declines of adequate significance for species or subspecies listing under the ESA (USFWS 2010a). In 2016, the USFWS determined a petition for reconsideration of the American pika's listing decision did not present substantial scientific or commercial evidence to initiate a status review (USFWS 2016d).

Despite lacking range-wide conservation concerns, the American pika is considered a "Species of Greatest Conservation Need" by the WDFW (2015) due to its high sensitivity to climate change, low reproductive and dispersal rates, and high selectivity for specific habitat characteristics. American pikas are also afforded protection from hunting, trapping, or killing by the Washington Fish and Wildlife Commission (WAC 220-200-100). Overall, information is largely lacking on the Cascade Mountain subspecies of American pika (*Ochotona princeps fenisex*) native to western Washington; no extensive historical records exist from which to derive abundance trends (USFWS 2010a, Smith 2020).

Known Threats

Climate Change

American pikas have a higher resting body temperature than any other North American lagomorph (MacArthur and Wang 1973). This adaptation likely aids in surviving winter low temperatures but comes with the consequence of a relatively narrow temperature optima. Prolonged exposure to temperatures just 3°C above their resting body temperature can cause mortality, leaving American pikas vulnerable to summer heat stress (MacArthur and Wang 1973, Beever et al. 2010). Conversely, pika may be vulnerable to winter cold stress under a drier climate model, with extreme cold coupled

with a lack of persistent snowpack having an apparent negative impact on pika persistence in some areas (Stewart et al. 2017, Rodhouse et al. 2018).

The USFWS (2010a) suggested that because most hydrologic effects of climate change have occurred below 2,500 m elevation, the American pika had not yet experienced these large climatic changes because it mostly occurs above that elevation (Smith and Weston 1990). However, this is not true for the northern half of American pika distribution including the Pacific Northwest (Smith and Weston 1990). In the Park, where pikas inhabit elevations as low as 351 m (Bruggeman 2012), they have likely experienced environmental stressors due to climate change. Johnston et al. (2019a) reported direct and indirect effects of winter weather anomalies on American pika abundance in the Park, and these effects were mediated by elevation. Foraging opportunities, spurred by interactions of abnormal moisture and snow patterns, likely positively affected American pikas at high elevations but resulted in marked declines at low elevations (Johnston et al. 2019a). Meanwhile, American pikas at mid elevations were likely more affected by acute cold stress during snow droughts in winter (Johnston et al. 2019a). Schmidt et al. (2021) found evidence that downslope migration is occurring in the Park, driven by dispersers seeking available territory; this also suggests future range retraction of American pikas here will occur due to extirpations at lower elevations. Nevertheless, American pikas may already mitigate elevation-weather interactions through habitat selection. Bruggeman (2012) found the probability of patch use was positively correlated with talus depth, suggesting that pikas may be utilizing the cool microclimates within interstices of deeper rock layers, as has been suggested elsewhere for the species (Beever et al. 2017). Pikas may also mitigate unfavorable thermal conditions by shifting activity periods to cooler parts of the day and changing their body shape to better thermoregulate (Beever et al. 2017).

Nutritional Landscape Alteration

Livestock grazing is occasionally cited as a threat to American pikas due to competition or trampling of forage resources which may be especially pronounced (and hard to detect afterwards) during drought conditions (USFWS 2010a, Smith 2020). However, relationships between grazing pressure and pika persistence have not been established as causative, and the USFWS (2010a) did not determine grazing to currently represent a significant threat.

Upslope encroachment of native plant species, especially conifers, is an oft-cited concern for the persistence of American pikas and other meadow-dwelling species (Raymond et al. 2014, WDFW 2015). This phenomenon is complex and varies among species and climate variable interactions. Areas projected to experience increases in both temperature and precipitation may see the most pronounced proliferations of woody plant cover (Parmesan 2006). Raymond et al. (2014) posit that the extent of high-elevation meadows, and therefore habitat to species such as the American pika, will certainly decrease under climate change. However, the USFWS (2010a) ultimately determined that native plant encroachment does not constitute a clear, significant threat to pikas.

Invasion of non-native plants occurs and is likely to occur in American pika habitats, but most invasions are not likely to reach and significantly degrade primary, high-elevation pika habitat (USFWS 2010a). Anthropogenic suppression of wildfire has likely helped encourage loss of meadow vegetation to woody plants (USFWS 2010a). Though prescribed fire may be considered as a

management action to help maintain montane meadows, this action is not as relevant in Washington as most American pika range in this state occurs in wilderness areas (Raymond et al. 2014). However, the remoteness of American pika habitat in Washington also aligns with minimal suppression of wildfire by humans, potentially allowing for more natural fire regimes and maintenance of montane meadows (USFWS 2010a, Raymond et al. 2014).

Conservation and Research Needs

Further knowledge is needed on the climate change effects on the woody plant proportion of subalpine and alpine meadows. Lower snow persistence and a longer growing season will likely encourage encroachment of trees which could reduce foraging and vigilance opportunities of high-elevation species. The expected increase in wildfire frequency and severity may mitigate this phenomenon in drier meadows but understanding the extent and net effects of these interacting phenomenon will require more research (Raymond et al. 2014). American pikas possess the ability to survive and even proliferate after moderate intensity wildfires, likely due to the insulative properties of talus and palatability of regenerating vegetation (Varner et al. 2015).

As the climate warms across the range of the pika, a species that relies on cool temperatures for thermoregulation, certain sites have seen population decreases or extirpations in areas like the Great Basin in Nevada, and portions of Colorado, Utah, and California (Beever et al. 2011, Stewart et al. 2017). However, American pikas persist in some of the less hospitable portions of their range such as Bodie, Lava Beds National Monument, and Mono Craters in California, Craters of the Moon National Monument and Preserve in Idaho, and the Columbia River Gorge of Washington and Idaho (Smith 2020). The adaptive resilience of American pikas coupled with the challenges of assessing population trends of an animal that experiences stochastic metapopulation dynamics led Smith (2020) to suggest that American pikas are not under serious threat from climate change on a range-wide scale. Likewise, Schwalm et al. (2016) found that variables related to habitat configuration and connectivity were frequently more influential than climatic variables when modeling American pika distribution. Still, despite complicated and sometimes competing interpretations, studies appear to indicate that proper management of American pikas requires a robust, region-specific population as well as considerations of interactions between climate-driven environmental changes, habitat characteristics at varying scales, and site-specific behavioral plasticity (Beever et al. 2011, Beever et al. 2017, Schwalm et al. 2016; Smith 2020).

Disturbance of talus habitats due to road and trail maintenance, repair, and construction represents a localized threat to pikas. Best practices to mitigate such disturbance includes surveys of project areas to determine presence of pikas and limiting work and noise around occupied patches (Appendix A).

Bats

Twelve bat species have geographic ranges that overlap with the Park, and nine have been confirmed within the Park: California myotis (*Myotis californicus*), western long-eared myotis (*Myotis evotis*, previously distinguished from Keen's myotis; Lausen et al. 2019), little brown myotis (*Myotis lucifugus*), long-legged myotis (*Myotis volans*), Yuma myotis (*Myotis yumanensis*) (Figure 25), big brown bat (*Eptesicus fuscus*), silver-haired bat (*Lasionycteris noctivagans*), hoary bat (*Lasiurus cinereus*), fringed myotis (*Myotis thysanodes*), and Townsend's big-eared bat (*Corynorhinus townsendii*). The two species not detected in the Park to date include western small-footed myotis (*Myotis ciliolabrum*) and western red bat (*Lasiurus blossevillii*) (Christophersen and Kuntz 2003).



Figure 25. Yuma myotis (*Myotis yumanensis*) voucher record NOCA54703, photographed by Jason Ransom/NPS during white-nose syndrome testing, May 27, 2021, in Stehekin, Lake Chelan National Recreation Area, WA.

The little brown myotis, big brown bat, and silver-haired bat are widely distributed across North America, and the remaining nine species are generally confined to the western portion of the continent. Population metrics for many of these species, especially western populations, are notoriously difficult to monitor, but some information is available. The little brown myotis is relatively abundant throughout its range, although eastern North American populations are rapidly declining and may be facing regional species extinctions by 2026 (Kunz and Reichard 2010), thereby prompting IUCN to designate the global population as endangered (Solari 2018). Only sparse and conflicting information exists for the status of silver-haired bats, although the North American

population is thought to be somewhat stable but slowly declining and regionally vulnerable (NatureServe 2021). Other bat species that may occur in the Park are generally considered to be stable across their ranges, although there is evidence that at least some regional populations of fringed myotis, Townsend's big-eared bat, and western red bat are experiencing overall declines (Pierson et al. 1999, Gruver and Keinath 2006, NatureServe 2021). Townsend's big-eared bat is designated as a Candidate for listing as a state endangered, threatened, or sensitive species in Washington.

The primary threat to many species of North American bats, and most certainly the little brown myotis, is a fungal disease known as white-nose syndrome (WNS). The fungus (*Pseudogymnoascus destructans*, or Pd) infects deep skin tissue during hibernation and rouses bats out of their torpor, causing them to expend needed energy reserves when prey resources are scarce. Mortality from this disease may also result from wing deformity, dehydration, and disruptions to breathing and thermal regulation. The little brown myotis is perhaps one of the species hardest hit by the disease in North America, with millions of individuals killed since the emergence of WNS in 2006 (Frick et al. 2010). Both the little brown myotis and Yuma myotis (which has a similar life history) occur in large roosting colonies in the Park during the spring and summer.

In 2016, the first case of white nose syndrome in the western U.S. was confirmed in a little brown myotis in King County, Washington. Since then, over 70 WNS cases have been confirmed on both sides of the Cascades and in four species: little brown myotis, Yuma myotis, western long-eared bat, and fringed myotis. Pd was also found on a silver-haired bat that did not present symptoms of infection (WDFW 2020). There are relatively few data on the overwintering ecology of western North American bat species, but they are generally not known to hibernate in large groups like eastern populations (Perkins et al. 1990, Weller et al. 2018).

Baseline surveys for bats were conducted in the Park from 1998–2001 by Christophersen and Kuntz (2003) using acoustic sampling and captures to assess presence and relative frequency of bat species (methods did not determine population abundance). The surveys covered 32 sites across the Park, situated within three habitat types (subalpine, forest, and riparian) on both the east and west sides of the Cascade Crest (Figure 26, Table 13). Sites ranged in elevation from 159–1685 m (522–5529 ft). Eight of the 12 species were confirmed as a result of that study, but four (fringed myotis, western small-footed bat, western red bat, and Townsend's big eared bat) were not captured or acoustically detected during those surveys. Contemporary acoustic monitoring surveys at various elevations and locations across the Park were also conducted during summers 2019 and 2020. Nine bat species were detected during those surveys, including the first Park record of fringed myotis near Rainbow Creek. Surveys at some abandoned mine mitigation sites also took place during 2019–2020. Townsend's big-eared bat was not detected during any surveys but has been incidentally observed in Stehekin and on the west side of the Park along the North Cascades Highway corridor (Figure 26). A western small-footed myotis was documented just north of the Park complex in the upper Skagit River. If the western red bat and the western small-footed bat occur in the Park, they are extremely rare (Hoffman et al. 2015).

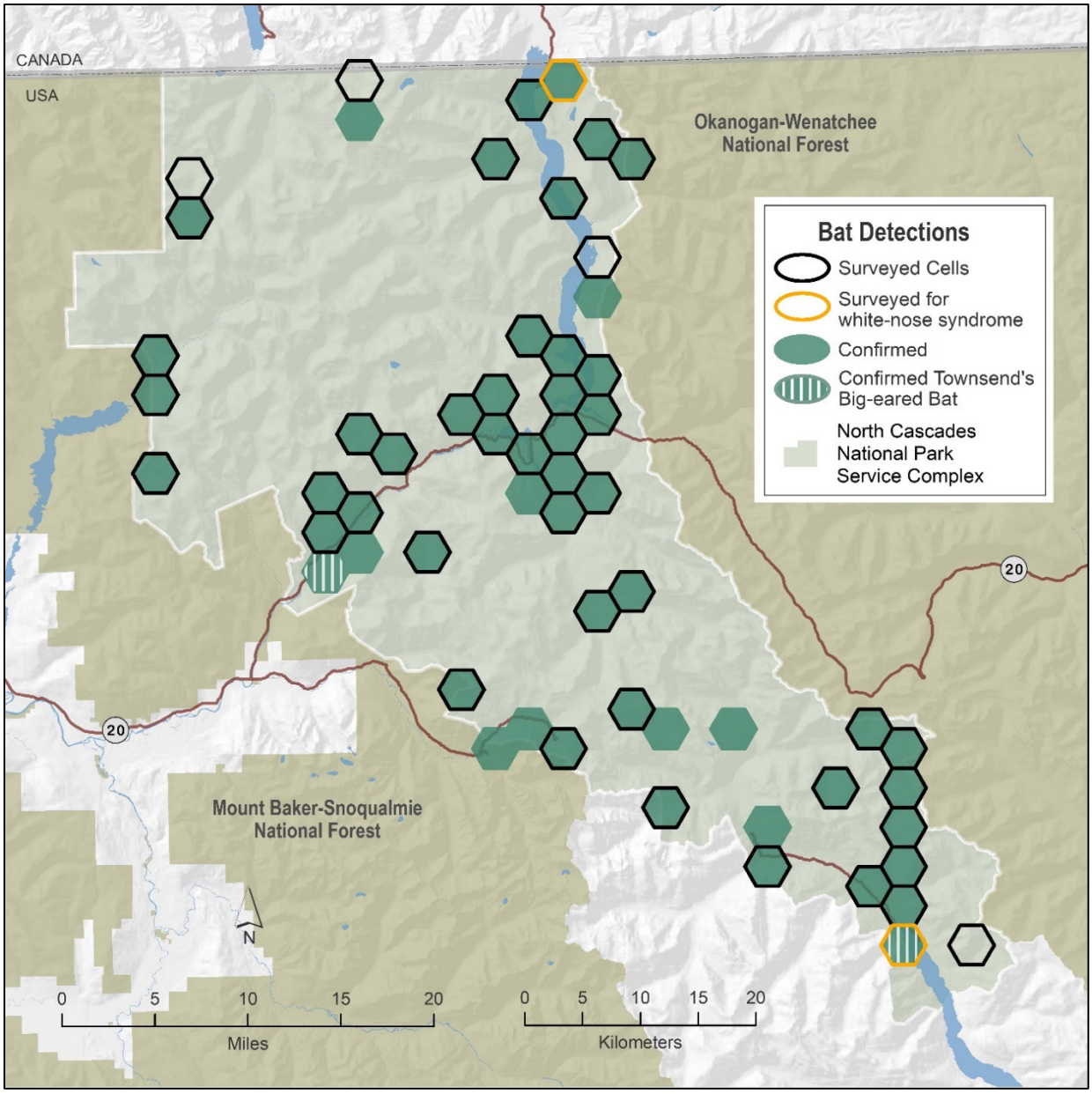


Figure 26. Survey locations, surveyed and incidental bat detections, and white-nose syndrome testing sites in North Cascades National Park Service Complex, 1995–2020 (Table 13). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 13. Bat data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
1998–2001	Acoustic recording devices at 50 stations across habitat types	216 hours, July–September across years	Christophersen and Kuntz 2003
1999–2001	Capture operations at 31 sites	June–September, 372 net hours across years	Christophersen and Kuntz 2003
2003	Visual inspections of 2 abandoned mine sites	1 visit each	NPS unpublished data
2017–2019	Capture operations and exit counts at 2 colony sites (white nose syndrome testing)	2–12 nights, April–August each year	NPS unpublished data
2019–2020	Acoustic recording devices at 90 stations across elevation strata	2–13 nights per point, July–September each year	NPS unpublished data
2020	Acoustic recording and exit counts at 4 abandoned mine sites	1 night each site	NPS unpublished data

Habitat Requirements

The North Cascades provide a valuable source of habitat for bat species through an extensive network of lakes, streams and ponds in old-growth forests. For some Pacific Northwest bat species, population abundance is associated with forest maturity (FEMAT 1993, Hayes 2003). Fringed myotis, long-eared myotis, and long-legged myotis prefer late seral stage forest habitats that offer abundant roost sites (Taylor 1999, Weller 2008).

Snags and large trees, most abundant in mature forests, provide protected roost sites with appropriate microclimates for bats. Open areas over lakes, streams, and rivers in proximity to roost sites provide foraging habitat with abundant insects, unobstructed flight for foraging, and sources of accessible drinking water (Hayes 2003). Environmental parameters vary between species and largely dictate what type of roost is used. In the Pacific Northwest, some species, especially little brown, Yuma, and California myotis, are more likely than others to use human structures like buildings and bridges as roost sites (Senger et al. 1974, Nagorsen and Brigham 1993). Large numbers of little brown myotis and Yuma myotis have been recorded emerging from various buildings throughout the Park (high counts of 425 [2017] at Stehekin, and 2,575 [2018] at Hozomeen). Species such as the western small-footed myotis, Townsend's big-eared bat, and big brown bat, can be classified as roost site generalists that can utilize a variety of natural and human structures. While the Yuma myotis roosts in a wide variety of habitats, it is more closely associated with water than most other North American bats. In Washington, western long-eared and fringed myotis have been documented frequently using various rock crevices (Rancourt et al. 2005, Lacki and Baker 2007), while the silver-haired bat, hoary bat, and long-legged myotis appear to prefer tree roosts such as snags, hollow trees, and exfoliating bark (Campbell et al. 1996, Betts 1998, Lacki et al. 2010).

In their inventory of bat species in the Park, Christophersen and Kuntz (2003) found the greatest species diversity and relative activity within riparian habitat types, compared to forest and subalpine habitats. When comparing acoustic feeding buzz recordings across these habitat types, the data suggested that forest sites were generally used for commuting, while riparian sites were used as foraging areas. Their data also suggests some partitioning of the landscape by species and sex. Little brown myotis appeared to be more prevalent at higher elevations and forest habitats, while Yuma myotis was more often found in riparian habitats. Females were more commonly found at lower elevations than males, and California myotis was never detected in subalpine habitats. These patterns parallel findings from studies at Mount Rainier National Park (Petterson 2001) and elsewhere in the Pacific Northwest (Fenton and Barclay 1980, Thomas and West 1991, Grindal et al. 1999).

Hibernacula have yet to be identified in the Park; however, Pacific Northwest bat species generally hibernate in protected spaces behind bark, in tree cavities and snags, piles of wood, rock crevices, caves, and mines. The North Cascades range has few natural caves that could support large numbers of cave-dwelling bats, but numerous decommissioned mines throughout the ecosystem provide habitat that is actively used by multiple bat species. All species known to occur in the Park either hibernate in the Park or migrate elsewhere for the winter. Myotis species, as well as the big brown and Townsend's big-eared bats, are thought to remain in or near the Park year-round. Hoary bats and western red bats are known long distance migrants between summer and winter ranges. Silver-haired bats are also long-distance migrants, but some do hibernate over winter in the Pacific Northwest (Kunz 1982, Nagorsen et al. 1993).

Diet

Moths (Order Lepidoptera) are an important prey item for many bat species known to occur in the Park. Townsend's big-eared bat and long-legged myotis are known across their ranges as the most heavily reliant on moths, but they feed on a variety of flying insects including beetles (Coleoptera) and flies (Diptera; Barbour and Davis 1969, Warner 1985, Ober and Hayes 2008). Moths are still an important component in the diets of California myotis, hoary bats, silver-haired bats, western long-eared myotis, and fringed myotis, but are not necessarily a staple across their ranges (Kunz and Martin 1982, Reimer et al. 2010). They will consume a variety of other flying insects like beetles, flies, and caddisflies (Trichoptera); some of these species, such as the western long-eared myotis and fringed myotis, are more adept than others at picking non-flying insects off surfaces (Black 1974, Warner 1985, Faure and Barclay 1992). Western small-footed myotis, little brown myotis, and Yuma myotis are considered small flying insect generalists with varying degrees of aquatic (i.e., mayflies, caddisflies, flies) and non-flying insect specializations (Burles et al. 2008, Ober and Hayes 2008). Common diet items for these species can also include small moths, beetles, spiders (Araneae), and true bugs (Hemiptera; Black 1974, Whitaker and Lawhead 1991). Big brown bats are also known as generalists, but with a proclivity for capturing beetles and other flying insects with large body sizes and tough exoskeletons (Freeman 1981, Lacki et al. 2007).

To reduce competition, multiple bat species occupying the same foraging patch utilize different feeding strategies that are broadly categorized as aerial searching (or hawking), gleaning, and trawling (Schnitzler and Kalko 2001); however, bat species occupying the Park generally fall into the

aerial group with the flexibility in some species to glean (Fenton and Bogdanowicz 2002). Aerial foragers hunt flying insects, while gleaners pick off prey from vegetation or the ground (Fenton and Bell 1979, Faure and Barclay 1992). Park bat species fall across a spectrum of these two feeding strategies, and many are known to switch between them depending on prey availability and interspecific competition (Schnitzler and Kalko 2001, Denzinger et al. 2004). Of the moth specialists, for example, Townsend's big-eared bat is more capable of maneuvering around objects in order to glean prey off of leaves and other surfaces, while long-legged myotis is a more rapid and direct flyer that relies on their ability to capture airborne moths over canopy or water and in forest clearings (Black 1974). Other than Townsend's big-eared bat, species known to glean are western long-eared myotis, California myotis, western small-footed myotis, and fringed myotis (Ober and Hayes 2008).

Foraging behaviors and prey preference are strongly linked to morphology, echolocation call structure, and flight patterns (Norberg and Rayner 1987, Barclay and Brigham 1991). Higher frequency, short-duration echolocation calls are best suited for detecting small prey at close range near vegetation clutter and are employed by California myotis, Yuma myotis, little brown myotis, western small-footed myotis, and western long-eared myotis. Lower frequency, long-duration calls are used in less cluttered environments (such as over canopy and open water) and can detect larger prey at longer distances; this general call structure is used by big brown bat, silver-haired bat, fringed myotis, hoary bat, and Townsend's big-eared bat (Schnitzler and Kalko 2001). Like call structure, wing morphology and flight patterns are tightly linked to foraging microhabitat (Norberg and Rayner 1987, Ober and Hayes 2008). Open space foragers like big brown and hoary bats have small pointed wings and a high aspect ratio which are suited for catching airborne prey during fast and agile flight. Bats such as the silver-haired and Townsend's big-eared bat that forage around clumps of vegetation have long wings with rounded tips and average aspect ratios that allows slower flight and greater maneuverability, thereby avoiding objects. Narrow space foragers such as the western long-eared myotis have broad, short wings with heavily rounded tips to provide a hovering, highly maneuverable flight pattern that is well suited to gleaning prey off surfaces (Norberg and Rayner 1987, Norberg 1994).

Reproduction

Bat species in the Park copulate between late summer and early winter, generally after maternity colonies disperse and before torpor/hibernation (O'Farrell and Studier 1980); however, little brown myotis males sometimes mate with hibernating females (Wai-Ping and Fenton 1988). Females store sperm overwinter until ovulation, fertilization and implantation occurs in the spring (Black 1974). Gestation periods are typically around 50–60 days (Fenton and Barclay 1980), and females in northern temperate zones such as the Park give birth in June and July. Females typically give birth to only one pup; however, silver-haired and hoary bats are known to birth two pups (Maser et al. 1981, Kunz 1982, Nagorsen and Brigham 1993). Young begin to fly between 2–4 weeks and continue nursing 1–3 weeks after that (Caire et al. 1979, Milligan and Brigham 1993). Myotis females typically forego breeding their first year, but female Townsend's big-eared bats are sexually mature their first autumn (Herd and Fenton 1983). Females of at least some species (little brown and Yuma myotis) are known to delay or forego reproduction entirely in exceptionally cool, wet years (Grindal et al. 1992). Reproductive lifespans are relatively short for silver-haired bats (12 years; Schowalter

and Dorward 1978), but many species continue breeding for at least 12 years (little brown myotis; Keen and Hitchcock 1980) and even over 20 years (Townsend's big-eared bat, western long-eared myotis; Tuttle and Stevenson 1982, Perkins et al. 1994).

Pregnant and nursing females often form maternity colonies around the time of fertilization but can otherwise roost alone. Reproductive females of most Park species form small to moderately sized colonies in sheltered spaces. Hoary bat females roost alone and with their young in the lower canopy foliage of trees, and rarely roost in sheltered spaces like trees, caves, and buildings (Willis and Brigham 2005). Western long-eared and small-footed myotis females either roost alone or in groups up to 30 individuals (Rancourt et al. 2005), while California myotis usually form small colonies that roost in a variety of spaces, even on the ground (Nagorsen and Brigham 1993, Brigham et al. 1997). Fringed myotis, long-legged myotis, Townsend's big-eared bats, and big brown bat all form maternal colonies of 12 to several hundred females, although Townsend's big-eared bats are regularly seen roosting alone. These species all prefer roost sites in large diameter tree spaces, although fringed myotis shows an affinity for rocky substrates (Nagorsen and Brigham 1993, Baker and Lacki 2006, Lacki and Baker 2007). Large maternity colonies in the Park are formed by little brown and Yuma myotis females, which are often found roosting together. Little brown bat maternity groups consist of 12–1,000 individuals (Nagorsen and Brigham 1993, Falxa 2007), and Yuma myotis numbers range from 500 to over 4,000 (Betts 1997, Falxa 2008). Human structures such as buildings and bridges are preferred roosting sites for little brown bats and Yuma myotis (Weller et al. 2018) and allow for the largest colony formations, but these species are also known to roost alone in large, living trees near water (Evelyn et al. 2004). Known maternity colonies in the Park usually disperse by October.

Protective Status

No bat species in Washington are federally listed or proposed for listing under the ESA, although Townsend's big-eared bat and western long-eared myotis are federal species of concern statewide, and fringed myotis, long-legged myotis, and western small-footed myotis are species of concern in western Washington only. The little brown myotis was recommended for ESA consideration by Kunz and Reichard (2010) amid its decline due to WNS, and a status review and determination are due in 2023. The U.S. Forest Service manages the Townsend's big-eared bat and western long-eared myotis as sensitive species on national forest units in Washington (Hayes and Wiles 2013).

The Federal Cave Resources Protection Act (FCRPA) of 1988 requires federal agencies to protect and maintain the integrity of "significant" caves on federal lands, which provides protections to nearly all caves. Under the FCRPA, locations of caves are granted confidential status and exemption from the Freedom of Information Act. The statute also places a mandate on federal land managers to protect cave resources when planning projects and developing management policies (Hayes and Wiles 2013).

In Washington, all bats are designated as protected wildlife. Townsend's big-eared bat are classified in Washington as "Species of Greatest Conservation Need" (WDFW 2005). Four species found in the Park (fringed myotis, long-legged myotis, western small-footed myotis, and western long-eared myotis) are Washington "monitor" species that require additional emphasis for management,

surveying, or data collection (Hayes and Wiles 2013). The NPS has designated all bat species within the Park as Management Priority species.

Known Threats

Bat populations around the world are generally declining due to their slow reproductive rates and sensitivity to environmental stress (Jones et al. 2009). Major threats to bat populations include the spread of WNS, climate change, and destruction/disturbance of foraging and roosting sites, and these stressors are expected to become even more acute within the current global paradigm (Sherwin et al. 2012). Other related and ancillary threats include contaminants and use of pesticides on insect prey (O'Shea and Johnston 2009, Hayes and Wiles 2013) and increasing use of wind turbines (Arnett et al. 2008, Arnett and Baerwald 2013).

White Nose Syndrome

The number of species in the U.S. infected with WNS continues to grow, with 12 of the 47 bat species known to be afflicted. These species include little brown myotis, fringed myotis, long-legged myotis, Yuma myotis, western long-eared myotis, big brown bat, northern long-eared myotis (*Myotis septentrionalis*), eastern small-footed myotis (*M. leibii*), tri-colored bat (*Perimyotis subflavus*), the endangered Indiana myotis (*M. sodalis*), the endangered gray myotis (*M. grisescens*), and cave bat (*myotis velifer*; USFWS 2016e). All of these species are insectivorous and have adapted to winter food shortages by hibernating in cold, humid environments like mines and caves.

The disease's initial appearance and rate of spread in Washington has outpaced earlier projections, which predicted an arrival in northern Washington by 2031 (Maher et al. 2012). As of late 2020, no infected bats have been found north of King County, but a guano sample collected from a little brown myotis roost in Everett, WA tested positive for Pd, the fungus that causes WNS. To date, no cases have been detected in British Columbia.

Predicting relative effects of WNS on western bats, especially myotis species, is difficult due to their roosting ecology and the vast, rugged landscapes that need to be searched (Weller et al. 2018). Modeling has suggested that WNS is most likely to occur in landscapes that are topographically diverse, higher in elevation, generally cold, and seasonally variable (Flory et al. 2012). Although these conditions match those found in the Pacific Northwest and especially in the North Cascades, key differences in the roosting behavior of western bat populations may help to prevent the magnitude of decline seen in eastern North America (Weller et al. 2018). High rates of transmission are more likely to occur in larger hibernacula where individuals are tightly clustered (Langwig et al. 2012), which is characteristic of eastern bat populations but relatively rare in the west (Perkins et al. 1990, Weller et al. 2018).

Climate Change

Warmer temperatures and more extreme precipitation regimes that are projected for the Pacific Northwest (Dalton et al. 2013) are expected to impact the availability of foraging and roosting resources for bats (Hayes and Wiles 2013). With warmer and drier summers, insect productivity is expected to increase regionally (Sherwin et al. 2012), but reduced precipitation and stream flow may locally inhibit the availability of aquatic insects that are staple prey for some bat species such as the

Yuma and little brown myotis. The Yuma myotis, which is most closely associated with water than other bat species, would likely be most adversely impacted by a drier landscape (Adams 2010). The small body sizes and hairless membranes of the wings make bats more at risk for evaporative water loss relative to other similarly sized mammals (Webb 1995). Lactating female bats could be faced with reduced water availability, which may reduce reproductive rates (Adams and Hayes 2008). This set of conditions may also increase the availability of some flying insects and increase the number of dry nights suitable for foraging, which may be beneficial for aerial hawking species (Sherwin et al. 2012).

Warmer winters with fewer days below freezing are also expected for the Pacific Northwest (Dalton et al. 2013), which may negatively affect torpor and hibernation. Torpor duration is negatively associated with ambient temperature (Geiser and Broome 1993), whereby an increase in temperature will rouse bats when there is limited food and water availability to replenish fat reserves. Some species, such as California myotis, silver-haired bats, and Townsend's big-eared bats, are more tolerant of winter arousals than others; therefore, disturbances to their wintering ecology may be relatively buffered. Conditions that rouse females out of hibernation may affect the overwintering storage of sperm by females (Racey 1982) and may even trigger early ovulation and pregnancy when insect prey is still scarce (Syme et al. 2001). Temperature-induced disturbances and associated energetic stress may further aggravate WNS effects (Jones et al. 2009, Boyles and Willis 2010).

Habitat Loss

Disturbance to hibernating bats causes loss of the vital fat reserves that support bats through the winter, when insect prey is not available. NPS managers take care to protect hibernacula (winter roost site) by installing gates on cave and mine entrances, as well as controlling human access which can spread Pd from one cave to another (Hoffman et al 2015). Bat gates have been installed at several abandoned mines in the Park, and bats have been documented entering and leaving gated sites (Figure 27). Spelunking is not a recreational activity in the Park. Keen's myotis, which is largely reliant on forest habitat, is susceptible to habitat destruction due to logging (Burles and Nagorsen 2003), and forest-dwelling Townsend's big-eared bat populations forage in canopy and edge habitats, and don't appear to use clear-cuts and early seral habitats, making logging a concern for forage habitat availability (Gruver and Keinath 2006). Loss and alteration of wetland habitat could alter important sources of insect prey and drinking water.



Figure 27. Bat gate installed in 2020 at the abandoned Black Warrior Mine, North Cascades National Park, WA.

Conservation and Research Needs

Long-term monitoring of bat diversity, abundance, trend, and microhabitat use is needed for all bat species in the Park. This has historically been difficult to accomplish because most species in the Park are widely dispersed, and roost in small numbers with low roost site fidelity (Hayes 2003). While myotis colonies in some large summer roost sites have been counted annually in the Park, trend assessment from those data are not possible. Those exit count numbers are confounded with changing roost fidelity and anthropogenic manipulation of ingress and egress points on buildings between time steps. Counts in such structures may be more robust through genetic sampling, mark-resight sampling, or other sampling designs that can better quantify observation bias. It is also unknown how efforts to exclude bats from buildings impacts their overall success in the ecosystem, and if mitigation measures such as bat boxes provide an equitable alternative for roosting habitat in anthropogenic structures. Exclusion work on buildings and roofs should be conducted in winter when bats are likely not using Park buildings, and only after a roost area has been inspected for presence of bats. Routine management, facilities, and infrastructure operations conducted in the Park can affect roosting bats, but adherence to noise and disturbance guidelines may help mitigate local impacts (Appendix A).

Migration routes are poorly understood for bat species in the North Cascades, especially for hoary bats and silver-haired bats (Hoffman et al. 2015). The specific nature and magnitude of threats outside of the Park, such as wind turbine collision, are therefore unknown. Studies on bat migration

using radio-telemetry or bio-markers would likely contribute valuable insights into landscape level threats and may provide opportunities for threat mitigation.

Very few empirical data relating to the ecology of any bat species in the North Cascades exist. Studies on microhabitat use, seasonal resource selection by elevation, and relative use of restored mountain lakes would contribute to better understanding of emerging threats to bat species. Long-term monitoring of bats around riparian corridors in the Skagit River watershed, and the Stehekin River watershed, including Gorge, Diablo, Ross reservoirs and Lake Chelan would contribute to better understanding of how the water impoundments and anthropogenic light sources influence resource selection by bats.

Lastly, long-term monitoring and routine testing of bats for WNS in the Park is critical for identifying outbreaks of this catastrophic disease. Early detection of WNS will allow managers to adaptively manage risk and exposure to other colonies. Sampling can be conducted passively by collecting and testing guano, or directly through capture and swabbing of individual bats. Coordination with regional bat monitoring efforts across agencies and organizations is essential for ecosystem scale response to this disease.

Canada lynx (*Lynx canadensis*)

The Canada lynx (*Lynx canadensis*) is a North American boreal and subalpine forest carnivore and the rarest native felid species in Washington State (Figure 28). Lynx in Washington co-occur with two other wildcat species, bobcats and mountain lions (*Puma concolor*). Lynx are often confused with bobcats but are slightly larger (averaging 8.5–10.0 kg) with an entirely black tail tip, larger paws, longer legs and ear tufts, and fuller facial ruff. They were legally trapped in Washington for their highly-valued pelts prior to 1991 when protections were established due to concern for decreasing population size in the region, but they are still trapped for fur in British Columbia (Lewis 2016).



Figure 28. Canada lynx voucher record NOCA24293, captured on a remote wildlife camera (operated by Cascades Carnivore Project, National Park Service, and Washington State University), August 2, 2018, near Desolation Peak, Ross Lake National Recreation Area, WA.

The range of Canada lynx is strongly associated with its primary prey, the snowshoe hare, and is nearly continuous north of the U.S.-Canada border. The southern margins of both species' ranges extend into the northern contiguous U.S. within Maine, New Hampshire, Vermont, Michigan, and Washington, as well as the northern and central Rockies (Idaho, Montana, Wyoming, Utah, and Colorado; Poole 1997, Anderson and Lovallo 2003) where boreal forest and snowshoe hare distribution is relatively patchy. There were also verified historical occurrences in 24 states within the contiguous U.S., including Oregon, but these observations are considered anomalous and are largely not associated with persistent lynx habitat, but rather with years of population peaks in Canada (McKelvey et al. 2000).

In Washington, Canada lynx were historically abundant throughout mid- to high-elevation conifer forests of the northcentral and northeastern parts of the state from eastern Whatcom County to Pend Oreille County at the Washington-Idaho border (Lewis 2016, McKelvey et al. 2000). There are also a few early records from the southern Cascades near Mount Rainier and Mount Adams as well as the Blue Mountains of southeastern Washington (Taylor and Shaw 1929, Dalquest 1948). Washington established six Canada lynx management zones (LMZs) that represent areas historically occupied by lynx, but today the state's only known persistent breeding population occurs in the Okanogan LMZ. That LMZ is dominated by federal lands managed by the Park, the Okanogan-Wenatchee National Forest, the Mount Baker-Snoqualmie National Forest, and the Loomis State Forest, while the other five LMZs in northeast Washington likely support only a small number of lynx, at least intermittently (Lewis 2016, USFWS 2017a). Augmentation of lynx from British Columbia into the Kettle Range began in 2022, led by the Confederated Tribes of the Colville Reservation, in hopes of re-establishing a more robust lynx population there.

While there are no reliable estimates for the historical and current population sizes of Canada lynx for the lower 48 States, resident lynx remain broadly distributed across the southern extent of their historical range with the exception of some recent extirpations of small populations (USFWS 2017a). Such estimates also do not exist for populations across the six Washington LMZs; however, Canada lynx surveys in 2000 and professional opinion led WDFW biologists to the conclusion that there were certainly fewer than 200 and probably fewer than 100 lynx in the state at that time, with an estimated carrying capacity of 238 lynx (Stinson 2001). The state, and the Okanogan LMZ in particular, probably supports a smaller breeding population than historic levels, primarily due to large wildfires over the last 30 years. These events have cumulatively reduced suitable lynx habitat in the state by >26% between 1996 and 2014, with 97% of the losses taking place in the Okanogan LMZ (Lewis 2016). Lyons et al. (2023) estimated a 66–73% reduction in lynx carrying capacity estimates between 2000 and 2020 based on wildfire impacts to lynx habitat in the Okanogan LMZ.

The Park represents the western edge of core lynx habitat within the Okanogan LMZ. Numerous carnivore surveys have occurred throughout the Park and lynx have been consistently detected through time, although detections are still infrequent (Figure 29, Table 14). There have been 41 confirmed records from Park staff, researchers, and visitors during 1995–2020, including the westernmost detections in the state. Seven Canada lynx detections were also documented on trail cameras during a focused lynx occupancy study during 2016–2017 (King 2019, King et al. 2020a, 2020b), and one radio-collared male lynx traveled in the Park during 2011 (Table 14). Overall, detections in the western parts of the Washington State Route 20 corridor have been rare, and detections on the eastern side of Ross Lake have been more frequent due to the orthographically divided climate of the Cascade Range. Canada lynx regularly utilize the southern portion of the Park (Figure 29).

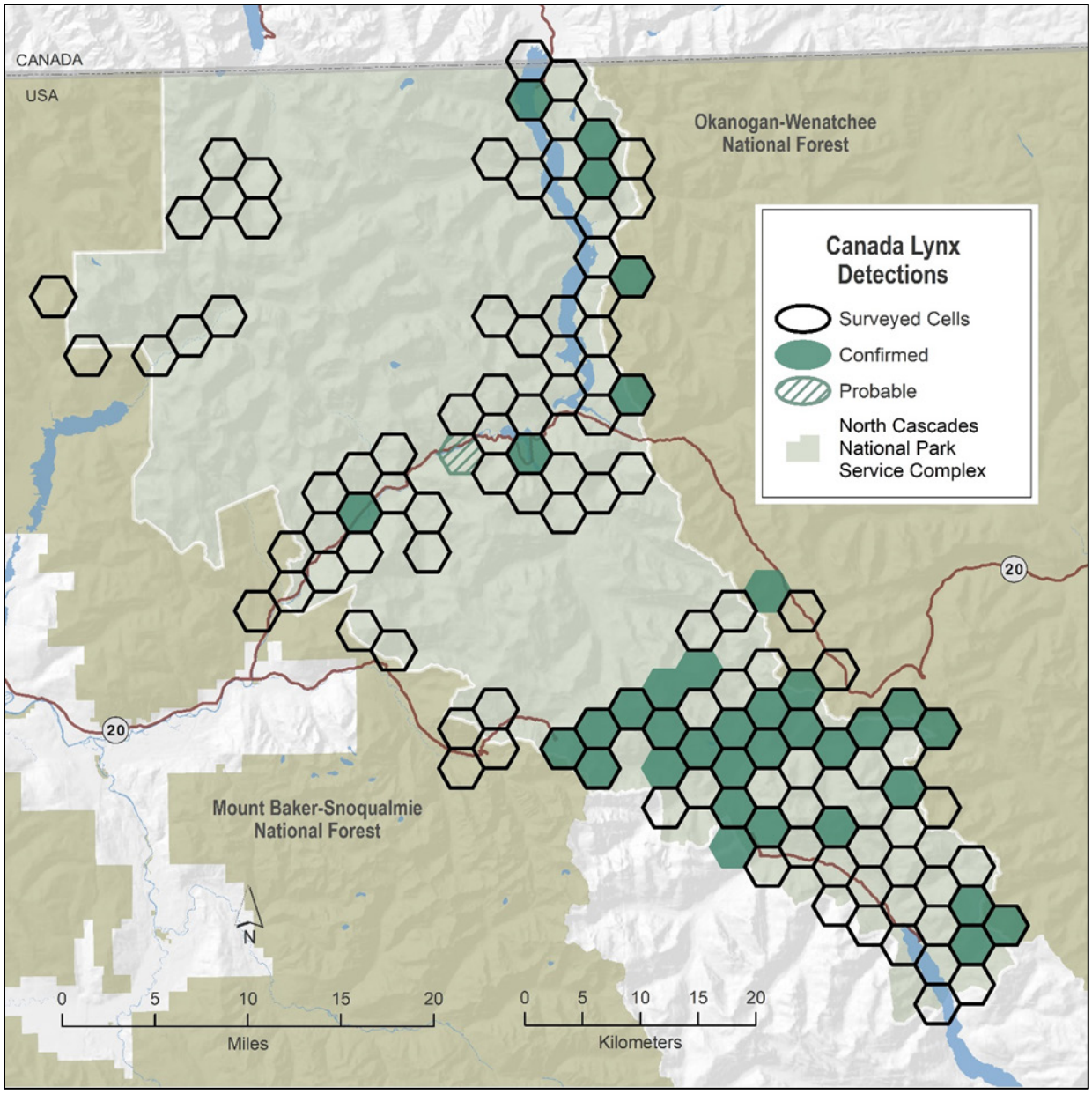


Figure 29. Survey locations, and Canada lynx (*Lynx canadensis*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 14). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 14. Canada lynx (*Lynx canadensis*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2003–2005	Forest carnivore study trail cameras at 24–40 stations	24 h camera stations, February–August each year	Christophersen et al. 2005, Christophersen 2006
2011	Radio-telemetry locations for 1 individual	July–October; 82 locations	WDFW unpublished data
2011–2020	Gray wolf monitoring trail cameras at 8–25 stations	24 h camera stations, October–May each year	NPS unpublished data
2016–2017	Canada lynx study trail cameras at 40 stations	24 h camera stations, June–October each year	King 2019, King et al. 2020a, 2020b
2015–2017	Wolverine study trail cameras at 4 stations	24 h camera stations, November–June	Lewis et al. 2020, R. Long unpublished data
2018–2019	Montane carnivore study trail cameras at 64–66 stations, and scat DNA meta-barcoding	24 h camera stations, May–October each year; 508 total scat samples	Whiles 2021
2018–2020	Fisher study trail cameras at 56 stations	24 h camera stations, all months	Humphries 2022
2020	Black bear study trail cameras at 109 stations	24 h camera stations, July–September	J. Manning and NPS unpublished data

Habitat Requirements

Lynx occur within dense boreal, sub-boreal, and subalpine forests with long winters and persistent deep, powdery snow where their exceptionally large paws and long hind legs provide a competitive advantage over other snowshoe hare predators (Aubry et al. 2000, McKelvey et al. 2000, Hoving 2001). Snow conditions are an important factor influencing fine-scale habitat use and distribution over the landscape during winter months but are also tied to summer habitat use (Squires et al. 2010). In Washington, summer Canada lynx occupancy is more probable where average snow depth during the previous winter is greater (King et al. 2020b).

Occupancy in the North Cascades is most strongly associated with forests dominated by Engelmann spruce, lodgepole pine, and subalpine fir at mid to high elevations (generally between 1400 m and 2150 m). Moderate to dense, multi-layered canopy cover in early seral forests generally supports the highest densities of snowshoe hares as shrubs and young trees provide cover from predators, although they may also occur in older forests with relatively open overstories (Aubry et al. 2000, Hodges 2000, Lewis et al. 2011). Canada lynx select against open areas, recently burned areas, and forests dominated by ponderosa pine and Douglas-fir (McKelvey et al. 2000, Koehler et al. 2008).

Canada lynx den sites in the western U.S. and Canada have been associated with late seral (>200 yr old) forests dominated by Engelmann spruce, subalpine fir, and lodgepole pine (Koehler 1990, Squires et al. 2008). Dens are commonly located in spaces around or under large woody debris such

as root wads or downed logs which are often found in late-successional forests but may also be suitable within mixed conifer-deciduous or in older regenerating stands (>20 yrs since disturbance; Koehler 1990, Mowat et al. 2000, Squires et al. 2008). The amount of woody debris and other denning structures appears to be more important for den site selection than the age of the forest stand (Mowat et al. 2000, Organ et al. 2008), and such habitat is not considered to be a limiting factor at the southern periphery of lynx range (Moen et al. 2008, Squires et al. 2008). Dens must also be close to foraging habitat to minimize time spent away from kittens (Moen et al. 2008, Vashon et al. 2012).

Diet

Throughout their range, Canada lynx are highly dependent on snowshoe hare as their main prey source, which constitutes 50–100% of their diet (Aubry et al. 2000, Mowat et al. 2000). The high degree of specialization on snowshoe hare is evident from the synchronization of the two species' population levels (Krebs et al. 2001). Other prey species include red squirrels and other rodents, grouse, ptarmigan, passerines, and ungulates, but red squirrels typically dominate the diet after snowshoe hare (Nellis et al. 1972, Parker et al. 1983, Hatler 1988, Brittell et al. 1989, Koehler 1990, Stephenson et al. 1990). This general pattern was also reflected in diets of Canada lynx within the Okanogan LMZ, where snowshoe hare was detected in 79–87% of scats collected, and red squirrels were the second most dominant prey species (Koehler 1990, Von Kienast 2003).

Hares tend to dominate lynx diets even in parts of their range with marginally low hare densities and where proportionally more alternate prey are eaten, but persistent breeding populations of lynx are less likely in such areas (Koehler and Aubry 1994, Mowat et al. 2000, Maletzke et al. 2008). While diet variety increases in summer months as different prey become available, supplemental prey is not adequate to compensate for chronically low availability of snowshoe hares (Brand and Keith 1979, WDFW 1993). Nonetheless, alternate prey, in particular red squirrels, may support lynx during stochastically low hare populations and at the southern periphery of their range where hare abundance may be chronically low (O'Donoghue et al. 1997, Roth et al. 2007). Hare densities in Washington generally hover around the low end of the range deemed sufficient to support a persistent lynx population (USFWS 2017a).

Canada lynx use three hunting methods: 1) using temporary "waiting beds" on cliffs or ridges overlooking centers of hare activity, 2) focusing hunts within small areas of hare activity, particularly during low hare periods, and 3) following established hare runways (Brand et al. 1976). Capture success depends on snow conditions influencing lynx movement and the distance between to the hare at the beginning of the chase, as well as the lynx's experience (age) and familiarity with the area (Haglund 1966, Nellis and Keith 1968). After a successful capture, lynx may consume its prey immediately or cache it for later consumption (Nellis and Keith 1968, McCord and Cardoza 1982, WDFW 1993).

Reproduction

In Washington, Canada lynx commonly breed in March and April, and kittens are born between May and mid-June after a 60- to 70-day gestation period. Females usually begin mating in their second year (22 months of age) but may breed at 10 months of age when hares are abundant, while males are not known to breed as yearlings (Koehler and Aubry 1994, Mowat et al. 2000). Average litter size for

Alaska, Canada, and Washington populations is 2.77 *(n*=26; Hatler 1988), but well-fed females have been known to produce between 4 and 6 kittens (O'Connor 1984, Slough and Ward 1990). Females solely attend to kittens at their natal den until they are 6–10 weeks old (Squires et al. 2008, Olson et al. 2011). Kittens stay with their mothers through their first winter as they learn to hunt and capture prey, and thereafter remain closely associated until they disperse in April and May to establish their own home ranges (Mowat et al. 2000, USFWS 2017a).

Perhaps the most important factor influencing lynx reproductive success across most of its range is the availability of snowshoe hares. Lynx population size is highly synchronized in response to that of snowshoe hare populations, which fluctuate in cycles that generally last 10 years (Elton and Nicholson 1942, Krebs et al. 2001). When hares are plentiful and increasingly available, lynx have lower adult and juvenile mortality, higher pregnancy rates, and larger litter sizes that result in rapid population growth (Slough and Mowat 1996, Mowat et al. 2000). As hare populations crash, lynx numbers dramatically decline due to starvation. During this phase, many lynx abandon home ranges to search for food and often die soon after initiating dispersal (Ward and Krebs 1985, Mowat et al. 2000). During the crash phase and when hare populations are low, female lynx produce few or no kittens that are recruited to the winter population (Brand et al. 1976, Slough and Mowat 1996). Those that do survive during these phases may experience nutritive stress that postpones their sexual maturity and lowers their reproductive rate (WDFW 1993).

Protective Status

Lynx in the forested portions of northern states within the contiguous U.S. initially received federal ESA protections as a threatened species in 2000, primarily due to a lack of regulatory mechanisms, low population sizes, and habitat loss and alteration (USFWS 2000). The USFWS published a species status assessment in 2018 that concluded lynx may no longer need ESA protection due to a lack of evidence of a major range contraction or population decline in the contiguous U.S. The agency has not yet released a proposed rule for the delisting. The NPS has designated Canada lynx as a Management Priority species within the Park.

In 2006, the USFWS designated a total of 4,768 km^2 of critical habitat within Minnesota, Montana, and Washington (USFWS 2006). This critical habitat designation was dramatically expanded in 2009 to encompass 101,010 km^2 within Maine, Minnesota, Montana, Wyoming, Idaho, and Washington, then again in 2014 to include anywhere lynx occur within the contiguous U.S. (USFWS 2006, 2014). Critical lynx habitat in Washington is mostly (>90%) managed by the Okanogan-Wenatchee and Colville National Forests under the federal lynx conservation strategy (ILBT 2013) and includes 58% of the Okanogan LMZ (USFWS 2017a). These areas protected under the ESA by requiring evaluation for all actions carried out, authorized, or funded by a federal agency (USFWS 2006).

In Washington, lynx have not been hunted or trapped since 1991. Under Washington State law (WA Administrative Code 220-610-010), Canada lynx are designated as endangered and are protected from possession, harassment, hunting, and killing. This listing in 1993 was largely due to range contractions and substantial habitat loss (Stinson 2001). The Washington Department of Natural Resources (WDNR) developed the Lynx Habitat Management Plan that has been applied to 125,980 acres of state-managed land (WDNR 1996, 2006). The Plan directs DNR and other cooperating non-

federal owners to manage lynx habitat using strategies such as maintaining dispersal routes and patch connectivity, promoting a diversity of forest types and stages that connects denning and foraging sites, and prolonging the persistence of snowshoe hare habitat (WDNR 2006).

Known Threats

The Washington lynx population is at significant risk from multiple acute and ongoing threats, most notably habitat loss and fragmentation primarily from wildfires, various compounding effects of climate change, and demographic issues. Lynx behavior and survival may also be influenced by high levels of recreation (especially in winter), incidental captures in traps, poaching, and vehicle collisions (ILBT 2013, Lewis 2016).

Habitat Loss and Fragmentation

The transition from boreal forest types at the southern extent of the lynx range makes suitable habitat and hare densities naturally patchy, which are further limited by aspect and/or elevation in the west (Koehler and Aubry 1994, Ruediger et al. 2000). High-quality habitat conditions in Washington where lynx have historically occurred are generally fragmented and intersected by landscape features (i.e., valley bottoms, forest openings) and forest types (i.e., Douglas-fir/Ponderosa pine stands) that lynx often avoid (Koehler et al. 2008). Specifically, stands of Engelmann spruce, subalpine fir, or lodgepole pine are often insular or peninsular on the landscape. It is likely that these fragmented landscape characteristics have impeded recolonization of these areas from Canada after disturbances such as wildfires (Koehler et al. 2008).

Widespread catastrophic, stand-replacing wildfires have been the primary cause of significant lynx habitat loss in Washington. Since the mid-1980s, and particularly since 2000, there has been a dramatic increase in large wildfire frequency, duration, intensity, and length of wildfire seasons (Westerling et al. 2006). These changes in the fire-prone landscape of the Okanogan LMZ have contributed to the net loss of high-quality lynx habitat by nearly 26%, a significant reduction (62.8%) in carrying capacity, and possible expansion of female home range size (Lewis 2016, Lyons et al. 2016, USFWS 2017a). Regeneration from severe fires can take 15–40 years to create suitable winter twig forage for snowshoe hares (ILBT 2013, Holbrook et al. 2018), during which time additional fires may impact adjacent habitat and reduce the likelihood that a lynx population will persist (USFWS 2017a). For example, in 2006 alone, multiple large wildfires destroyed 9% (865 km^2) of suitable habitat within the Okanogan LMZ, followed by another 9% loss in the 2015 fire season, and yet another 15% loss in 2020 (Lyons et al. 2023). Future loss and fragmentation of habitat from catastrophic wildfires is a major threat to the Okanogan population, given its small and isolated nature, recent habitat losses, and the expected accelerating effects of climate change (ILBT 2013, Lewis 2016).

Climate Change

Climate change poses a serious threat to Canada lynx and snowshoe hare populations particularly along the southern edge of their range, with evidence already showing a 39% northward contraction from their historical distribution (Laliberte and Ripple 2004, USFWS 2017a, King et al. 2020a, King et al. 2020b). Apart from the direct destruction of habitat by intensifying wildfire regimes, climate change is expected to reduce the availability of suitable lynx habitat by promoting Douglas-

fir/Ponderosa pine forests and by reducing the spatial and temporal availability of persistent, deep, powdery snow (Carroll 2007, Gonzalez et al. 2007, ILBT 2013). A reduction of quality snowpack would likely negate the competitive advantage of the lynx's lower foot-loading anatomy as well as allow the intrusion of potential lynx predators like mountain lions and wolves (Buskirk et al. 2000); whereas, changing phenology of snow and snowshoe hare pelage mismatch may also lead to higher hare depredation rates (Oli et al. 2023). Snowpack loss that has already been documented in the Cascades over the last half of the twentieth century is expected to continue (Mote 2003, Stoelinga et al. 2010). Warmer temperatures and increasing physiological (i.e., nutritive, thermal) stressors may also impact lynx populations by increasing the exposure and virulence of pathogens (Lewis 2016).

Demographic Factors

Because the Washington lynx population inhabits the margin of acceptable habitat and is relatively small, it is at higher risk from stochastic events that affect reproduction, survival, sex ratio variations, immigration, and emigration (Vanbianchi 2015). Lynx occupancy in the Okanogan LMZ relies in part on the distance to source populations in Canada (Schwartz et al. 2002, King et al. 2020a); moreover, genetic and demographic resiliency of the Washington population requires continuous immigration (Koen et al. 2014). Although critical habitat in Washington is broadly connected by continuous forested landscapes to populations in British Columbia (Singleton et al. 2002), the extent and effects of emigration from Canada are unknown (McKelvey et al. 2000, USFWS 2017a). Immigration may be dependent on the lynx population cycle in the core of the range, but how this dynamic affects dispersal into the southern margins, and persistence thereafter, is not well characterized (McKelvey et al. 2000, USFWS 2017a). Dispersal into Washington may also be limited by human development (e.g., cities, highways, railroads) and rivers in southern British Columbia and by the legal harvest of lynx in the province that would remove potential south-bound dispersers (Lewis 2016).

Conservation and Research Needs

Despite almost 50 years of research on lynx-hare cycles, the driving forces of this pattern are not well understood, especially in the context of changing climate in the North Cascades. Understanding population dynamics of both hares and lynx, and how they interact, requires well-informed model parameters. Demographic data on both species is largely absent for the North Cascades. Some of these data could arise from genetic analyses of the population, which may provide insights into sex ratio and reproduction, as well as immigration rates from Canada. Modeling lynx population dynamics with well-informed demographic parameters is also a dynamic process due to the ever-changing fire footprint that modifies connectivity of habitat and size of core patches in the landscape mosaic. Recreational use impacts to Canada lynx have not been studied in depth and quantifying how recreation pressure may affect lynx life history characteristics is critical, especially as the number of visitors in the Park continues to increase rapidly. Of note, Canada lynx have been detected using the North Cascades Highway during its winter seasonal closure. This is also the time that winter recreational use includes snowmobiles on this route.

Specific conservation needs for Canada lynx in the Park include preservation of habitat connectivity to the east and north, genetic monitoring across the Okanogan LMZ, monitoring and management of

backcountry visitor use and motorized winter recreational use, and careful fuel management practices that preserve lynx habitat. Future augmentation of the population may be necessary to maintain genetic diversity. Specific attention to likely lynx presence should be given to early secessional habitat (e.g., burns, windfall, and transmission line right-of-ways) in mid- and high-elevation areas where hare density may be highest. Canada lynx are at risk of vehicular strike, especially along the North Cascades Highway between the Okanogan National Forest and the Park, where habitat is generally contiguous and only bisected by the road. Wildlife crossing structures and/or active animal crossing signals could mitigate lynx deaths in areas where landscape features funnel wildlife into focused highway crossing areas. Mitigating disturbance around Canada lynx den sites is required under the ESA for all management, facilities, and infrastructure operations conducted in the Park (Appendix A).

Cascade red fox (*Vulpes vulpes cascadensis*)

The Cascade red fox (*Vulpes vulpes cascadensis*) is one of three rare subspecies of mountain red fox (Figure 30). These three subspecies live high in the mountains of the American West, in the Cascade, Sierra Nevada, and Rocky Mountains. Together they form the Mountain subclade of the Nearctic clade (Aubry et al. 2009) of red foxes in North America. They are distinct morphologically, genetically, and ecologically from other all red foxes. The Cascade red fox was historically distributed throughout the Cascade Range from southern British Columbia to southern Washington (Aubry 1984) but appears to now be endemic to only a few sites in Washington, primarily in Washington's South Cascades. It occurs naturally at low densities, and is snow-adapted, and thus particularly sensitive to climate warming (Copeland et al. 2010, Sacks et al. 2010).



Figure 30. Cascade red fox voucher record NOCA40654, captured on a remote wildlife camera (operated by U.S. Forest Service), March 22, 2001, near Rainy Pass, Okanogan National Forest, WA.

The Cascade red fox inhabits upper montane forests, subalpine parklands and meadows, and alpine habitats year-round unlike most other species that migrate downslope during harsh mountain winters. In the South Cascades, the Cascade red fox inhabits high-elevation habitat at Mount Rainier National Park, the Gifford-Pinchot and Okanogan-Wenatchee National Forests, the Yakama Nation Reservation, and the Ahtanum State Forest. Sites of current presence include Mount Rainier; Mount Adams, the Indian Heaven, Goat Rocks, William O Douglas, and Norse Peak Wilderness areas; and Ahtanum and Manastash Ridges (Akins et al. 2018). There have been a dozen verifiable observations in the North Cascades (Cascade Range north of I-90) during the past four decades. The remaining known population, which occurs in the South Cascades, appears to be small enough that stochastic

events such as outbreaks of disease, the arrival of novel competitors, increased recreation, or the impacts of climate change could threaten its long-term persistence. In addition, historical habitat loss and fragmentation due to timber extraction and road building may limit connectivity with the north. Hybridization with nonnative lowland red foxes is a possible threat to the integrity of this unique subspecies. Genetic introgression has not been documented in the Cascade red fox to date (Akins et al. 2018) but has occurred rapidly in other montane red fox populations (Merson et al. 2017; Quinn et al. 2019). Efforts to better understand the ecological requirements and threats to the South Cascades population, determine presence throughout the North Cascades Ecosystem, and assess connectivity across the landscape are critical to determining the long-term prospects for this unique carnivore.

Although many survey efforts for montane carnivores have been conducted in the Park, including extensive scat collection targeting Cascade red fox by Park staff and partners, no confirmed detections have occurred in recent times (Figure 31, Table 15). The only confirmed record of this species adjacent to the Park occurred in 2001 near Rainy Pass (Figure 30).

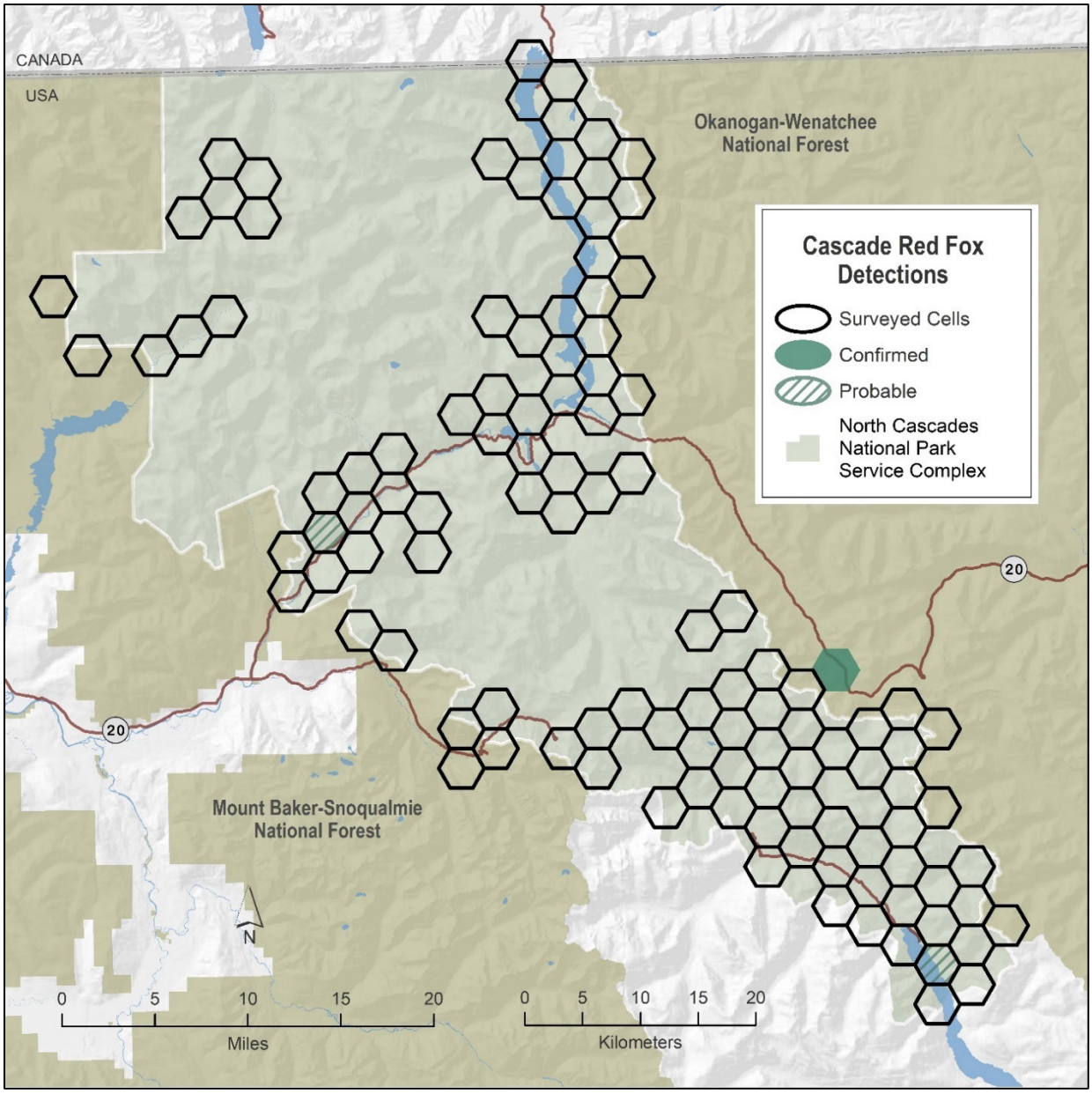


Figure 31. Survey locations, and Cascade red fox (*Vulpes vulpes cascadensis*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 15). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 15. Cascade red fox (*Vulpes vulpes cascadensis*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2003–2005	Forest carnivore study trail cameras at 24–40 stations	24 h camera stations, February–August each year	Christophersen et al. 2005, Christophersen 2006
2011–2020	Gray wolf monitoring trail cameras at 8–25 stations	24 h camera stations, October–May each year	NPS unpublished data
2016–2017	Canada lynx study trail cameras at 40 stations	24 h camera stations, June–October each year	King 2019, King et al. 2020a, 2020b
2015–2017	Wolverine study trail cameras at 4 stations	24 h camera stations, November–June	Lewis et al. 2020, R. Long unpublished data
2018–2019	Montane carnivore study trail cameras at 64–66 stations, and scat DNA meta-barcoding	24 h camera stations, May–October each year; 508 total scat samples	Whiles 2021
2018–2020	Fisher study trail cameras at 56 stations	24 h camera stations, all months	Humphries 2022
2020	Black bear study trail cameras at 109 stations	24 h camera stations, July–September	J. Manning and NPS unpublished data

Habitat Requirements

Historically, Cascade red fox was distributed throughout Washington's Cascades Range and the mountains of southern British Columbia. While occurring on both sides of the Cascade Range as a cursorial predator requiring open habitats to successfully hunt, Cascade red fox prefers open-forest habitats more typically found on the eastern slopes of the range (Aubry 1984, Akins 2017). A lack of recent detections in British Columbia indicates the subspecies may now be restricted to Washington. During the past several decades, a southward range contraction appears to have occurred within Washington as well, as the only known population now occurs in the South Cascades (south of the I-90 corridor). There have been 12 verifiable observations of Cascade red fox in the North Cascades during the past four decades including in the Methow Valley, on the Loomis State Forest, in the southern Sawtooth Range, the Chiwawa River area, south of Stevens Pass, and north of Cle Elum. These detections are sparse and disparate making it difficult to confirm the presence of a population anywhere north of I-90. Thus, the distribution now appears to occur within ≤50% of its historical range in the state (Akins 2017).

Diet

The most comprehensive analysis of diet for the Cascade red fox was conducted four decades ago and reported that mammals comprised half their diet during summer, and the vast majority (89%) of their diet during winter (Aubry 1983). Snowshoe hare, northern pocket gopher (*Thomomys talpoides*), voles and mice (*Clethrionomys gapperi, Phenacomys intermedius*, and *Microtus* spp.), Pacific jumping mouse (*Zapus trinotatus*), and hoary marmot were the most common mammals

found in Cascade red fox scats. Plants (mainly berries; 19.9 %), insects (16.6%, primarily grasshoppers), and birds also contributed to their diet.

The Cascade red fox occurs in cold landscapes that contain deep snowpacks from early winter to late spring (Akins 2017), and during this time, fox forage in part on prey species that inhabit the subnivean zone. Cascade red foxes also rely on larger prey such as snowshoe hares that remain active in the high Cascade mountains during winter (Aubry 1983). During summer, foxes appear to rely on subalpine meadows, parkland, and open upper montane forests to forage for small mammals, insects, and birds; and scavenges on ungulate carcasses such as mountain goats, black-tailed deer, and elk (*Cervus canadensis*).

Reproduction

Reproduction in the Cascade red fox has not been well studied and is not well known. Courtship and mating behavior have been documented on remote wildlife cameras, with presumed courtship behavior documented at the end of February and early March, and mating in early March. Young kits (estimated to be six weeks old) were observed at a den entrance at Mount Rainier National Park in mid-May. Since red fox gestation is typically 53–55 days, these observations together suggest kits are born during mid to late April.

During 2011 to 2016, three litters of 1–3 observed kits were documented at Mount Rainier or the surrounding area, including kits seen as early as May 12th and as late as August 8th (J. Akins, unpublished data). Aubry (1983) documented a subadult (<1 year old) female that, the next summer, was observed at one year of age with a litter of three kits.

Protective Status

The Cascade red fox is listed and an Endangered species by the State of Washington and is a Species of Greatest Conservation Need in the Washington State Wildlife Action Plan due to its endemic status, apparent range contractions, small population sizes, potential threat of climate change, and the limited amount of biological and ecological information available for both Cascade and lowland red foxes in Washington (Aubry 1983, WDFW 2015, Akins 2017, Akins et al. 2018). The USFS lists Cascade red fox as a Regional Forester's Sensitive Species, and NPS has listed Cascade red fox as a Management Priority Species.

Known Threats

The remaining population of Cascade red fox appears to be small enough that stochastic events such as outbreaks of disease, the arrival of novel competitors, increased recreation, or the impacts of climate change could threaten their long-term persistence. In addition, historical habitat loss and fragmentation due to timber extraction and road building may limit connectivity to the north. Hybridization with nonnative lowland red foxes is a possible threat to the integrity of this unique subspecies; and while it has not been documented in the Cascade red fox to date (Akins et al. 2018), it has occurred rapidly in other montane red fox populations (Merson et al. 2017, Quinn et al. 2019). Little research has been conducted to understand the ecological requirements, threats, and factors affecting landscape connectivity of the Cascade red fox.

Climate Change

Climate change is occurring rapidly in the mountains and new climate-related dynamics will likely threaten at-risk carnivores such as the Cascade red fox. Several direct effects of climate change that are evident in montane habitats are likely to affect both Cascade red fox and their prey (Monzón et al. 2011). Warming of montane habitats has resulted in declines in annual snowpack as winter precipitation occurs as rain more often and spring snow melt occurs earlier than previously (Kullman 2001, Mote et al. 2005, Parmesan 2006, Stewart 2009). Changes to the snowpack affects the distribution and abundance of many prey species (Korslund and Steen 2006; Pauli et al. 2013). Snow supports an array of subnivean prey that remain active throughout the winter under a deep, insulating snow layer. Changes within the snowpack can adversely affect prey abundance as well as accessibility for predators as environmental conditions within this subnivean zone are altered (Pauli et al. 2013). When rain-on-snow events occur during winter, the snowpack compresses and hardens, reducing or eliminating access for predators to the subnivean zone. A reduction in snowpack also reduces the amount of insulation value of snow and decreases the temperature in the subnivum, causing prey to suffer higher mortality rates. In addition, many landscape-scale ecological processes are driven by the amount of snow in the subalpine zone including the development of alpine meadows and subalpine parklands (Henderson 1973, Rochefort et al. 1994). Persistent snow cover during a significant portion of the year contributes to the development of the habitat conditions that support many of the prey species on which the fox relies, whereas climatic warming results in meadows being invaded by conifer species (Millar et al. 2004). Warming promotes the upslope migration of conifers and reduces the areal extent of subalpine parklands (Franklin et al. 1971). This encroachment has also been exacerbated by the suppression of wildfires in areas that historically had regular fire intervals or were managed by Indigenous Peoples using fire treatments (Helms 1987, Miller and Halpern 1998, Coop and Givnish 2007, Halpern et al. 2010). In addition, during summer, higher temperatures, associated droughts, and heat stress have resulted in the loss of many mature high-elevation trees, particularly whitebark pine and mountain hemlock (Hansen and Phillips 2015). High-elevation conifers provide cover for prey such as snowshoe hares and are likely critical foraging habitat for foxes. Thus, the loss of these mature high-elevation conifers will likely result in declines in prey availability for the Cascade red fox. Finally, climate change may be causing more abundant carnivores typically found at lower elevations, such as the coyote, to now have greater seasonal access to montane habitats when the snowpack hardens after rain-on-snow events and due to earlier spring snow loss. This will likely increase competition and predation for foxes.

Invasion of competitors

As an intraguild competitor, the coyote is a significant limiting factor for red fox populations elsewhere (Voigt and Earle 1983, Sargeant and Allen 1989, Gosselink et al. 2007). In Washington, coyotes were historically absent or occurred only at low elevations in arid areas east of the Cascade Range (Hody and Kays 2018), where snowfall was low compared to in the mountains. However, three factors may have promoted the expansion of coyotes within and beyond their distribution, including into higher elevations of the Cascades. First, the gray wolf was eliminated from nearly all of its range in the contiguous U.S. by the mid-1900s (McIntyre 1995, Laliberte and Ripple 2004), thereby removing the most significant predator and competitor of the coyote and enabling a dramatic expansion within and outside the coyote's historical distribution. As well, increased winter recreation

may facilitate the upslope movement of coyotes into Cascade red fox habitat as coyotes are able to travel more easily along ski and snowmobile tracks (Gese et al. 2013). Third, reductions in snowpack in the mountains caused by reduced snow fall or increased rain-on-snow events (Cohen et al. 2015) have potentially increased the ease of travel for coyotes at higher elevations where deep snow conditions were more typical (Sheldon et al. 2007).

Coyotes do not possess the same morphological adaptations for deep snow as the Cascade red fox, yet coyotes now occur in montane habitats where they were historically absent (Perrine et al. 2010, C.B. Quinn, University of California, Davis, unpublished data, J.R. Akins, unpublished data). Deep snow likely kept Cascade red fox and coyote populations separated during winter but snow-packing by winter recreationists and below average snowfall in certain regions may be making Cascade fox habitats both more accessible and more suitable to coyotes. During summer, coyotes have always been able to access high-elevation montane habitats, but their abundance was historically much lower and their distribution more limited.

The effects of wolf reintroduction and range expansion on the distribution, behavior, and declining abundance of coyotes has been documented in Yellowstone National Park and Isle Royale National Park (Smith et al. 2003). Several wolf packs now occur in the North Cascades (WDFW et al. 2022), and the ongoing reestablishment of wolves in Washington may reduce coyote populations within the historical range of the Cascade red fox. This reduction in coyote numbers could help facilitate the recolonization or expansion of Cascade fox populations in the North Cascades. However, the population center for the Cascade red fox is currently located in the South Cascades where the gray wolf has not yet become re-established.

Encroachment of Non-Native Red Foxes

Because of the presence of non-native red fox populations in low-elevation areas adjacent to the Cascade Range, it is possible that lowland red foxes could invade habitats occupied by the Cascade red fox, resulting in several negative impacts to Cascade red foxes. These include hybridization resulting in the loss of local genetic adaptations to the high-elevation environment and associated declines in fitness, as well as disease transmission, resource competition, and mortality (Sacks et al. 2011). There is currently no evidence of the introgression of low-elevation red fox genetics in modern Cascade red fox samples; however, surveys that are designed to obtain genetic samples are needed to determine whether this has changed recently in the South Cascades or the extent to which this may have occurred in the North Cascades. Information on the distribution and genetic make-up of non-native foxes in areas adjacent to occupied Cascade red fox habitat would be valuable to assessing this potential threat.

Small Population Size

The Cascade red fox may also be at risk due to its small demographic and genetic effective population size. It is also possible that the population in the South Cascades is largely disjunct from any existing populations in the North Cascades. Akins et al. (2018) indicated that the population of Cascade red foxes is characterized by a small effective population size of 20 foxes, which reflects extremely low genetic diversity. Additionally, based on comparison to the genetic diversity of eight historical Cascade red foxes (Sacks et al. 2010), this population appears to have declined since then.

Based on its current effective population size, this population could be facing inbreeding depression, as may be occurring in a Sierra Nevada red fox population (Quinn et al. 2019). Low genetic diversity additionally can reduce a population's resilience and ability to adapt to environmental changes, such as those anticipated due to climate change.

Fire

One of the largest potential effects on Pacific Northwest forests is likely to come from an increase in fire frequency, duration, and severity (Littell et al. 2009, Westerling et al. 2006). From a long history of fire suppression, fuel accumulations have increased considerably and converted fire regimes from periodic low-intensity surface fires to high-intensity crown fires (Agee 1993). Wildfire may be one of the most significant threats to the Cascade red fox due to its low population size and limited distribution. Large areas of the South Cascades have experienced multiple, overlapping and in places high severity fires in portions of the only known occupied Cascade red fox habitat. In the North Cascades, even larger areas of mid and high-elevation forests have experienced fire during the past several decades. It is unknown whether these areas previously provided habitat for Cascade red foxes or if other threats had already removed foxes from these portions of their historical distribution. In the long term, fire will likely provide open forests that improve opportunities for foxes to forage. In the short-term, it is unclear how recent fires are threatening their small populations.

Habituation at several sites with high human visitation

A small number of habituated and food-conditioned individuals have been frequently observed in subalpine areas where human recreation is high including Mount Rainier National Park, Crystal Mountain and White Pass ski resorts, and on Mount Adams climbing routes. These sites also provide high-quality habitat for the Cascade red fox. Although the National Park Service discourages visitors from feeding any wildlife, some individual foxes have demonstrated clear begging behavior (J. Akins, pers. observations) at these sites. Feeding can negatively affect wildlife because it promotes a dependency on human foods. Habituation places foxes at risk of conflicts with humans and their pets, human-caused mortality particularly vehicular collisions, and disease transmission. Fox kits have been killed along the roadway at Mount Rainier National Park (NPS, unpublished data). While dogs are prohibited from most areas in National Parks, habituation also occurs outside of the Park and their presence in fox habitat may result in the transmission of several diseases from domestic dogs to Cascade red foxes.

Direct Killing

Due to the challenge of accessing the landscapes that the Cascade red fox occupies, poaching, incidental capture by trappers, accidental killing by hunters, and roadkill mortality likely have little impact on their populations. If these impacts become more significant in the future or if the size of the Cascade red fox population becomes substantially smaller, any additional mortality by these or other sources could have significant consequences for population persistence. The Cascade red fox is currently protected from trapping or hunting in Washington.

Conservation and Research Needs

Due to a significant lack of scientific study, there is a poor understanding of the basic ecology or an evaluation of the severity of specific threats and factors limiting their abundance and distribution of

the Cascade red fox, especially in the North Cascades where their presence is little known. New research on their current distribution, population genetic status, threats (especially from climate change and wildfire), and connectivity is critical to evaluating what components of the landscape may be addressed through conservation actions, particularly forest management treatments that improve fire resiliency and promote the connectivity between areas of occupied habitats.

The current understanding of Cascade red fox ecology and conservation is largely informed by research conducted at Mount Rainier National Park four decades ago (Aubry 1983), and a more recent population genetic assessment, habitat suitability model, and connectivity analysis across Washington's South Cascades (Akins 2017, Akins et al. 2018). There is a lack of analysis regarding landscape connectivity between the north and south. Connectivity across the Cascade Range will be critical to supporting a population in the North Cascades, and inferences assessing their habitat requirements in the north need to be evaluated from predictive models based on existing data from the south.

Climate change and its proximate effects will likely impact the Cascade red fox due to its effects on their prey species directly, as well as the landscape itself. A recent habitat modeling effort has provided a critical understanding of their broad habitat requirements and factors affecting gene flow and connectivity in a portion of their distribution (Akins 2017). This study also provides a useful tool to select sites with the greatest potential to provide occupied habitat from the vast potential habitat in the northern portion of their range. Surveys across high-quality habitat are essential to assess the conservation status of the Cascade red fox in the North Cascades—a critical step to understand the long-term prospects for this rare carnivore as habitats shift north in the facing of climate change.

Until their modern distribution in this portion of their distribution is understood, it will be difficult to determine how to protect this rare subspecies. Specific conservation needs for the Cascade red fox in the Park include understanding the level of connectivity and identifying potential barriers to areas where individuals, and hopefully populations, occur to the south and east including the Loomis State Forest and Central Cascades around Steven Pass.

There is currently no evidence that lowland red foxes are present in areas occupied by Cascade red foxes; however, surveys that are designed to obtain genetic samples from surrounding low-elevation sites, along the eastern slopes of the Cascade Range, in the Puget Lowlands, and the Methow Valley, are needed to determine whether this has changed recently in the South Cascades or the extent to which this may have occurred in the North Cascades. Information on the distribution and genetic make-up of non-native foxes in Washington would aid in such an effort. There is a significant lack of critical information on the potential threats they represent and the measures that could be employed to address those threats.

Critically, studies to better understand management actions that would alleviate threats to a robust population in the South Cascades and support connectivity and dispersal to the north are much needed to support recovery of the endangered Cascade red fox.

Fisher (*Pekania pennanti*)

Fishers (*Pekania pennanti*) are large, stocky members of the weasel family, endemic to North America (Figure 32). These forest carnivores historically occurred from the latitudinal tree line in northern Canada, southward through Washington, Oregon, and in California to the southern end of the Sierra Nevada Mountains. They also occurred throughout the Rocky Mountain Range from Canada to northern Utah. In the east, fishers occurred in parts of the mid-west south into the Appalachian Mountains. Fishers were extirpated from much of their range in the U.S. and experienced a range contraction in the southern portions of their historical range following European settlement (Lewis et al. 2012). Historical fur trapping took a heavy toll on fisher populations in the U.S., as did predator control efforts. Habitat loss due to logging, land conversion for farming, and residential development were also responsible for dramatic declines in the fisher's range through time. From the 1920s on, many areas in the U.S. ceased fisher trapping in an effort to protect remaining populations (Powell 1993).



Figure 32. Fisher voucher record NOCA43741, photographed by Jason Ransom/NPS during a reintroduction operation, October 12, 2019, at Baker River, Mount Baker-Snoqualmie National Forest, WA.

Despite decades of protection, populations in the western U.S. have remained small and disjunct (Aubry and Lewis 2003). Fishers have persisted in the Sierra Nevada, and the Klamath Siskiyou region of northwestern California and southern Oregon, and a population in the southern Oregon Cascades is the result of translocations from British Columbia and Minnesota (Aubry and Lewis

2003). In 2020, the southern Sierra Nevada population was federally listed as endangered under the ESA. In Washington, fishers once occurred throughout the Olympic Peninsula, Cascade Range, possibly in southwestern and northeastern Washington, and in the Blue Mountains (Hayes and Lewis 2006). Unregulated harvest, habitat loss and fragmentation, and predator control campaigns beginning in the late 1800s collectively resulted in the decline and extirpation of fishers from Washington by the mid-1900s (Lewis and Stinson 1998). Trapping seasons for fishers in Washington were closed in 1933 (Lewis and Stinson 1998). The last verifiable fisher sighting in Washington was an incidentally trapped fisher on the eastern side of the Olympic Peninsula in 1969 (Lewis and Stinson 1998). No fishers were detected in Washington during extensive surveys in the 1990s, and the fisher was listed in 1998 by the State of Washington as endangered (Lewis and Stinson 1998).

In the Park, 24 unconfirmed observations of putative fishers were received by NPS from visitors and staff from 1971–2001. None of those observations were accompanied by a voucher record and the actual species observed remains unknown. Given phenotypic similarities between certain age classes and sexes of fisher, Pacific marten (*Martes caurina*), and mink (*Mustela vison*), it is not uncommon for the identity of one species to be mistaken for another and thus unvouchered observations of fishers cannot be confirmed. Despite extensive searches through historical range, including extensive carnivore surveys in the Park (Figure 33, Table 16), no confirmed fishers were detected in the Park until after fisher reintroduction efforts began there in 2018.

A large fisher reintroduction project lead by the NPS, WDFW, Conservation Northwest, and Calgary Zoo translocated 170 fishers from British Columbia and Alberta, Canada, to the Washington Cascade Range from 2015–2020 (Lewis et al. 2022). In the North Cascades portion of the project area, 89 fishers (48 females and 41 males) were released from 2018–2020 into the Park and Mount Baker-Snoqualmie National Forest. These fishers subsequently ranged across 15,452 km^2 of the North Cascades ecosystem, as indicated by radio-telemetry locations and confirmed incidental detections on remote trail cameras (Lewis et al. 2022). There are 76 confirmed records of fishers shown in Figure 33, all of which occurred from 2018–2021, during the post-release period of the reintroduction project.

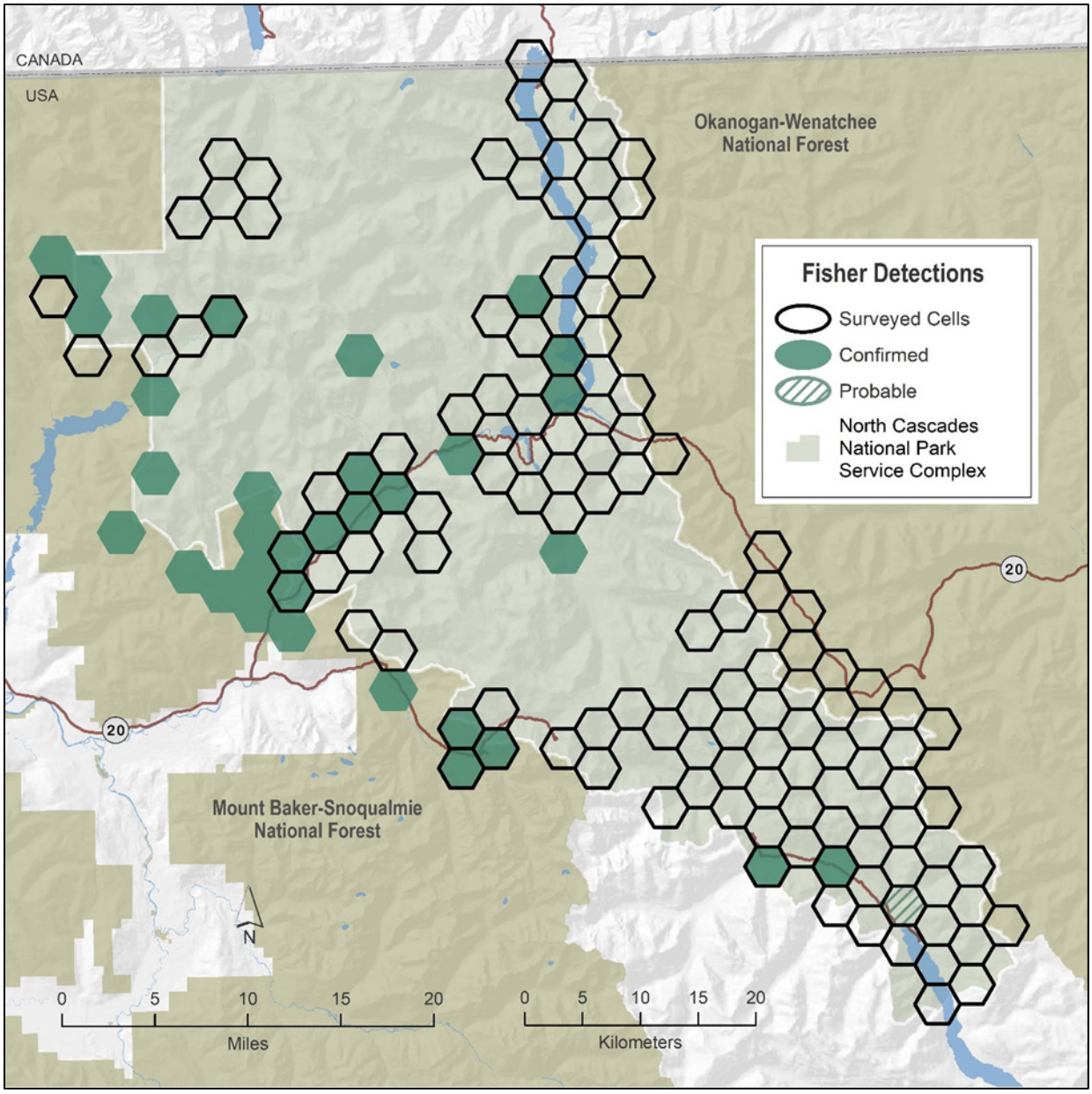


Figure 33. Survey locations, and fisher (*Pekania pennanti*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 16). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 16. Fisher (*Pekania pennanti*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2003–2005	Forest carnivore study trail cameras at 24–40 stations	24 h camera stations, February–August each year	Christophersen et al. 2005, Christophersen 2006
2011–2020	Gray wolf monitoring trail cameras at 8–25 stations	24 h camera stations, October–May each year	NPS unpublished data
2016–2017	Canada lynx study trail cameras at 40 stations	24 h camera stations, June–October each year	King 2019, King et al. 2020a, 2020b
2015–2017	Wolverine study trail cameras at 4 stations	24 h camera stations, November–June	Lewis et al. 2020, R. Long unpublished data
2018–2019	Montane carnivore study trail cameras at 64–66 stations, and scat DNA meta-barcoding	24 h camera stations, May–October each year; 508 total scat samples	Whiles 2021
2018–2020	Fisher study trail cameras at 56 stations	24 h camera stations, all months	Humphries 2022
2018–2020	Radio-telemetry locations for 80 individuals	1–40 points per individual	Lewis et al. 2022
2020	Black bear study trail cameras at 109 stations	24 h camera stations, July–September	J. Manning and NPS unpublished data

Habitat Requirements

Fisher home-ranges can span a mosaic of different forest types, but include high proportions of late-seral, mature forest stands, often positioned with minimal overlap with unforested areas (Raley et al. 2012). Fishers tend to avoid traveling in the open areas within their home range. Based on Parsons et al. (2020), it appears that forest mosaics that include old trees with large woody structure, mixed with young forest and larger prey items, are important for fishers. In general, fishers prefer habitat with a high level of canopy closure, and structural elements including logs, woody debris, large snags, and cavities in large trees. Females require cavities for reproduction, which provide protection from predation and mediate climate (Raley at el. 2012). Heartwood decay cavities are frequently accessed by fishers through openings excavated by pileated woodpeckers (Aubry and Raley 2006).

The availability of large trees is likely limiting for fisher populations, and short-rotation logging removes suitable structures from the landscape (Powell 1993, Hayes and Lewis 2006). Rest sites are habitat features that fishers use to rest between bouts of activity. Rest sites include structural features like logs, snags, ground burrows, raptor and rodent nests, mistletoe brooms, and lateral limbs (Hayes and Lewis 2006). The types of rest structures utilized by fishers may change with environmental conditions, suggesting that rest structures provide security from predators as well as microclimates to reduce thermal stress (Weir et al. 2004).

A study of two populations in California by Zielinski et al. (2004) found that fishers selected rest sites with large trees and dense canopy on steep slopes. Rest trees were often amongst the largest trees in the area, and logs used were also larger than average. Fishers selected sites where canopy cover was uniformly dense and where there was a diversity of sizes and types of structural elements within the forest. Fishers used many different rest areas across their range, suggesting that a large number of suitable rest sites dispersed across a fisher's hunting grounds is optimal. Female fishers preferentially chose cavities as rest sites, and generally used more enclosed sites compared to males. Zielinski et al. (2004) attributed this difference either to females, who are significantly smaller than males, seeking more protection from predation and thermal stressors, or to the predisposition in females to use features similar to den sites. While canopy closure and structural complexity was uniform across both populations, there was habitat-specific variation in rest-site selection, with the population in the more arid habitat of the Sierra Nevada selecting sites within 100 m of water and with an abundance of large oaks, and the coastal population selecting for rest sites in areas with large snags (Zielinski et al. 2004). A meta-analysis by Aubry et al. (2013) across the Pacific States and British Columbia found similar preferences for rest sites; dense overhead cover, steep slopes, high volume of logs and large-diameter trees and snags, and a preference for cooler microclimate (Aubry et al. 2013).

Diet

Fishers primarily hunt for tree squirrels (e.g., *Tamiasciurus* spp.) and snowshoe hares but have a diverse diet of small mammals, birds, reptiles, insects, ungulate carrion, and berries (Hayes and Lewis 2006). They are also known for their unique aptitude at successfully hunting porcupines (Powell 1993). A study of fisher diet in the southern portion of the Washington Cascades Fisher Recovery Area found that snowshoe hares and mountain beaver were the most prevalent species in stable isotope analyses of hair samples (Parsons et al. 2020). Of note in that study, these primary prey species were more closely associated with young forests, while fishers typically prefer older forests for denning and resting. Older forests in that study were more closely associated with chipmunk and small mammal species.

Reproduction

Fisher home ranges in the western U.S. and Canada average 18.8 km^2 for females and 53.4 km^2 for males, with a large variation in reported home range sizes between studies and seasons (Lofroth et al. 2010). A male's home range may expand during the breeding season, and the breeding home range of male fishers can be 2.4 times larger than their home range during non-breeding season (Aubry and Raley 2006). Male fishers have larger home ranges than females (approximately 2.7 times larger, excluding breeding season) (Lofroth et al. 2010). Home ranges tend to increase in size from southern to northern latitudes, which may be explained by the fact that both sexes in the northern reaches of fisher distribution are about 1.5 times larger in body size than their southern counterparts, or that prey abundance is lower in the north and more habitat is required for foraging (Lofroth et al. 2010).

Males visit females during the spring breeding season, but otherwise this species exhibits a solitary lifestyle. Female fishers can undergo delayed implantation, allowing them to carry a fertilized egg, suspended as a blastocyst, for 10–11 months before implantation in the uterine wall the following

spring (Mead 1994). Fishers become reproductively capable at age 1, but generally do not reproduce until age 2, due to delayed implantation (Mead 1994, Powell 1993, Hayes and Lewis 2006). Paragi et al. (1994) found an average reproductive rate of 63% for fishers, influenced by health during autumn and winter, age of the individual, and the availability of prey. One study in Oregon reported a 59% reproductive rate, but only 44% of those fishers successfully raised at least one kit to weaning (Aubry and Raley 2006).

Following implantation and an average of 36 days of gestation, fishers give birth to 1–4 kits in late March to early May, with an average litter size of two kits surviving more than seven days postpartum (Frost et al. 1997). Of note, one female fisher in the North Cascades was documented with four kits at her den, two years after reintroduction (Lewis et al. 2022). That litter size equals the largest litter size documented for fishers in the western U.S. Kits are weaned at 6–8 weeks and have some limited climbing ability at 10 weeks. Females are solely responsible for rearing kits, and provision young with prey in the den tree following weaning. Kits become independent in late summer to early fall (Powell 1993, Hayes and Lewis 2006).

Protective Status

Fishers were listed as endangered in 1998 under Washington State law (WA Administrative Code 220-610-010), and are protected from possession, harassment, hunting, and killing. The western U.S. population became a federal candidate species for protection under the ESA in 2004. Listing of the fisher under the ESA was deemed warranted but precluded, meaning there was sufficient evidence of threat to fisher populations, but other higher priorities take precedent (USFWS 2004). The "warranted but precluded" status was withdrawn in 2016 when the USFWS decided that the previously identified evidence of threat to western fishers was insufficient to constitute an existential threat to the population (USFWS 2016f). The NPS has designated fishers as a Management Priority species within the Park.

Reintroductions

Fisher reintroductions across their historical range began in the mid-1900s, when timber companies sought to mitigate timber losses associated with expanded porcupine populations, which they attributed to the loss of fisher as a porcupine predator (Lewis et al. 2012). Translocation efforts have reestablished or supplemented fisher populations in at least 11 U.S. states and six Canadian provinces since the 1940s (Lewis et al. 2012). Fishers are one of the most successfully reintroduced carnivore species, with a 77% success rate for reintroductions with known outcomes (Lewis et al. 2012).

WDFW published its Fisher Recovery Plan in 2006, which outlined reintroduction efforts to reestablish self-sustaining populations in the Olympic Peninsula, and the northern and southern Washington Cascades (Hayes and Lewis 2006). The recovery effort began with the release of 90 fishers from British Columbia in Olympic National Park from 2008 to 2010 (Lewis 2014, Happe et al. 2017, 2019).

Planning for the Cascades Fisher Reintroduction Project began in 2013 with WDFW's Implementation Plan for Reintroducing Fishers to the Cascade Range in Washington (Lewis 2013). The Park and Mount Rainier National Park led the National Environmental Policy Act process and

completed a Fisher Restoration Plan / Environmental Assessment in May 2015 (NPS 2014). From 2015–2020, that effort translocated 81 fishers (69 fishers (38 females and 31 males) from British Columbia, Canada, and 12 fishers (7 females and 5 males) from Alberta, Canada) into the South Cascades, including Mount Rainier National Park, and Gifford Pinchot National Forest. From 2018–2020, 89 fishers (48 females and 41 males) were translocated from Alberta, Canada, into the North Cascades. Reproduction has been documented in all three Washington fisher reintroduction areas (Lewis et al. 2022). Survival one year after reintroduction was high in the South Cascades (76%), but lower than expected in the North Cascades (42%). Juvenile females represented the highest survival of all ages and across all cohorts, at 83% in the South Cascades, and 55% in the North Cascades. As of the end of 2021, fishers were well distributed across much of the Cascades Fisher Recovery Area. Additional monitoring across the Cascade Range is planned for 2022–2024.

Known Threats

Fishers depend on large, mature trees and snags for den sites, and a high level of structural complexity for resting sites, all of which take hundreds of years to develop if existing structures are lost (Raley et al. 2012). Habitat loss and fragmentation due to timber harvest, landscape conversion and development, and changing fire regimes have likely driven declines in populations and impeded population recovery (Lewis et al. 2012). Fishers continue to be threatened by habitat loss and expansion from human activities that reduce habitat and limit dispersal and connectivity between populations, particularly in more northern populations that require larger contiguous home ranges than southern populations (Lofroth et al. 2010). Low reproductive rates make fisher populations vulnerable to disturbances that impact female survival, reproduction, kit survival, and recruitment. Slow population growth and reported annual survival rates as low as 61% also hinder recovery efforts (Lofroth et al. 2010). The small size of many extant fisher populations, and lack of connectivity between populations, puts fishers at risk for inbreeding depression, which can lead to infertility, higher susceptibility to disease, and less potential adaptability to changing conditions (Hayes and Lewis 2006)

Wildfire and Fuels Reduction

Where fisher populations have been depressed and reduced to small effective ranges, even single fire events can significantly impact habitat availability. For example, in 2017 wildfires in southern Oregon destroyed an estimated 10% of the indigenous fisher population's range, and 3% of the introduced population's range (Barry 2018).

With the potential for increased fire severity due to fuel build-up from historical fire suppression and a warming climate, there has been increasing fuels reduction management in forests, which can impact the quality of fisher habitat. Scheller et al. (2011) created landscape-scale simulations to assess long term impacts of fuels reduction treatments on fisher habitat, comparing the reduced risk of fire to the direct impacts of fuel reduction activities on habitat. The study found that over the long-term there could be a net increase in fisher habitat as a result of fuels reduction, despite the initial disturbance from fuels reduction activities. Truex and Zielinski (2013) investigated fuels reduction activities on a finer scale, looking at impacts of different treatments on fisher foraging and resting habitat separately. That study assessed mechanical and prescribed-burn practices and found that

either treatment on its own did not appear to have a significant impact on fisher resting habitat, but mechanical and fire management treatment together did have a negative effect. There was also a significant impact when prescribed burns occurred in the late-season, but not from prescribed burns in early season. Fuels reduction treatments did not appear to impact foraging habitat (Truex and Zielinski 2013). Zielinski et al. (2004) pointed out that fishers frequently use snags as rest sites, which may be more vulnerable to loss during prescribed fire than live trees. Disturbance from the human activity associated with thinning, fuels reduction, and prescribed fire activities could potentially disturb fishers (Hayes and Lewis 2006).

Predation

Empirical data on natural deaths in fisher populations that were not reintroduced, and are not trapped, are generally unknown, but the reported lifespan for fishers is approximately 10 years (Powell 1993). In a study by Gabriel et al. (2015) investigating fisher mortality from a sample of 167 individuals in California, 70% was due to predation. In that population, 40% of the predation was attributed to bobcats and 38% was attributed to cougars (*Puma concolor*), with the remainder from unidentified felid species. Felid predation comprised 90% of the identified predators, with coyotes and domestic dogs as less significant sources of predation. In the Washington Cascades, 54% (13 of 24) of necropsied fishers were likely depredated and DNA of bobcat, cougar, or both, was confirmed on three of those carcasses (Lewis et al. 2022). Twelve of those 13 depredated fishers were female; likewise, female fishers were also more predisposed to predation in the Gabriel et al. (2015) study. This apparent selection pressure is important considering that small fisher populations are heavily dependent on female survival (Spencer et al. 2011). Of note, there is some evidence that human modification to fisher habitat has increased the likelihood of bobcat predation on fishers (Wengert 2013).

Human-caused Mortality

Gabriel et al. (2015) found that of 167 mortalities they investigated, 2% were from vehicular strikes and another 2% from other anthropogenic causes. Among the 24 necropsied fishers in the Cascades, one female was illegally killed by a trapper, five fishers were hit by cars (but one also had high anticoagulant rodenticide exposure), and two died under suspicious anthropogenic circumstances (Lewis et al. 2022). Gabriel et al. (2015) reported that anticoagulant rodenticides (ARs) were among the most common substances found in fishers that had died from toxicological poisonings. Fishers can be exposed to ARs from ingesting rodents that are incapacitated or killed by household rat poisons, as well as from larger-scale agricultural uses of wildlife pesticides. AR compounds can accumulate in the organs and tissues of fishers and can be fatal. ARs are frequently used in illegal marijuana farms and have been implicated in mortalities of several species.

In California, 85% of fishers tested had detectable amounts of one or more AR compounds in their tissues (Gabriel et al. 2015). There was a 57% increase in mortalities and a 6% increase in exposure to ARs over a three-year period in that study, suggesting either an increase in marijuana cultivation or an increase in rodenticide use at existing sites. Mortalities from ARs were more common in male fishers than females. Exposure was highest in the spring, which coincides with increased ranging in male fishers during the breeding season, likely leading to increased exposure for male fishers

encountering more marijuana farms during spring growing periods, operating at a potentially higher level of rodenticide use (Gabriel et al. 2015). In the Cascades, only four of 11 fishers tested positive for AR compounds and no fishers tested positive for other environmental organic compounds (pesticides, environmental contaminants, drugs and natural products) (Lewis et al. 2022). The levels of AR compounds detected in three Cascades females were an order of magnitude lower than those reported on public lands in California (Gabriel et al. 2012); however, the level of AR compounds detected in one male was at or above levels that Gabriel et al. (2012) reported in fishers that died of AR ingestion. Notably, Chlorophacinone (detection limit 50 ppb) was six times higher in the male Cascades fisher than was reported for a California fisher that died from acute AR poisoning: the male Cascades fisher died after being struck by a vehicle.

Conservation and Research Needs

Several research projects have been completed or are in progress on fishers in the Washington Cascades, including investigations into predator-prey relationships, habitat selection, and AR exposure and mountain beaver relationships (Lewis et al. 2022). As these recently reintroduced fishers establish a self-sustaining population, continued research and monitoring is needed on resource selection and habitat connectivity throughout the ecosystem. Genetic monitoring will also be important through near-term and long-term establishment of the population to determine if heterozygosity remains sufficient to avoid genetic bottle-necking that can occur if only a small number of fishers are breeding and/or the breeding animals are too closely related.

Anticoagulant rodenticides can have severe impacts on carnivores through hunting of debilitated small mammals or scavenging of carcasses. The NPS does not use ARs in the Park, but such toxins may be present on private lands in and around the Park. When fishers are found dead, efforts must be made to screen carcasses for environmental toxins and contaminants, and specifically ARs. Recovered fisher hair or tissue should also be genotyped and matched to the founder population data, which will provide additional information on reproduction and survival in the population.

As fishers are newly returned to the ecosystem, new conservation measures should be incorporated into management practices in the Park and region. Specifically, management practices in forests should preserve large dead snags whenever they do not pose a risk to human safety. Snags with tree cavities made by pileated woodpeckers, or cavities with openings roughly 10–12 cm in diameter, are important denning and resting structures for female fishers. Protection of known fisher dens in such structures should be included in annual updates to the Park Resource Advisor's Guide. Routine management, facilities, and infrastructure operations conducted in the Park can affect fishers, but adherence to noise and disturbance guidelines around dens may help mitigate local impacts (Appendix A). Fuel management practices should also avoid mechanical thinning and forest disturbance in high quality fisher habitat during denning season (February–July) and avoid controlled burns during autumn. Of note, fishers often select mature deciduous tree species such as cottonwood, alder, and maple for denning, when a cavity is available. Specific attention to likely fisher presence should also be given along the interface of mature forest and open or early secessional habitat (e.g., burns, windfall, and transmission line right-of-ways) in low and mid-elevation areas. This transitional zone provides access to both denning and resting structure and preferred prey species. Fishers are

prone to vehicular strikes across their range, and wildlife crossing structures and/or active animal crossing signals could mitigate fisher deaths in areas where landscape features funnel wildlife into focused highway crossing areas.

Gray wolf (*Canis lupus*)

Gray wolves (*Canis lupus*) are the largest of three wild canine species in North America, weighing 26–45 kg. The pelage of gray wolves is phenotypically expressed in black and gray color phases as well as the less common white and tawny color phases. Contemporary presence, pack formation, and reproduction of gray wolves has been confirmed in the Park (Figure 34). Wolves are apex predators in the ecosystems where they occur, due in large part to their tendency to form social packs that are adept at hunting large game together (Sand et al. 2006). Pack sizes vary due to regional population dynamics and diet, but in Washington, packs typically consist of 3–6 individuals (range: 2–10 individuals, n = 33; WDFW 2022).



Figure 34. Gray wolf voucher record NOCA54626, showing four of the six known pack members captured on a remote wildlife camera (operated by National Park Service), November 7, 2020, near Bridge Creek, North Cascades National Park, WA.

Gray wolves were historically common across most of Washington but were largely extirpated by the 1930s due to Euro-American predator control efforts beginning in the mid-19th century (Wiles et al. 2011). Commercial fur trapping in the Pacific Northwest began in the 1820s and over the next four decades resulted in the collection of at least 14,810 wolf pelts throughout the region (Laufer and Jenkins 1989). As human settlements and agriculture increased during this time, efforts to control wolves for the protection of livestock rapidly intensified and broadened to include poisoning wolves, raising prices for wolf skins, and setting lucrative bounties (Wiles et al. 2011).

Since the reintroduction efforts in central Idaho and the greater Yellowstone area in 1995, wolves have expanded into the Pacific Northwest from Idaho and British Columbia (WDFW 2022). The number of sightings in Washington began increasing thereafter, and the first pack (Loup Loup) since the 1930s was confirmed in Okanogan County in 2008. WDFW counted a minimum of 206 wolves, 33 packs, and 19 successful breeding pairs across the state in 2021, marking the 13th consecutive year of overall population growth. A "pack" in that reporting context represents two or more wolves travelling together during the winter. Most individuals and all packs were observed in two of the three wolf recovery areas in the state (Eastern WA and North Cascades), and only lone individuals have yet been observed in the Southern Cascades and Northwest Coast recovery area (WDFW 2021).

Gray wolf observations in the Park have punctuated contemporary records at a regular, but sparse, rate over the course of several decades. Most records through time have been single wolves that were likely dispersers from packs in the north or east. An early exception was apparent pack activity around Hozomeen in the late 1980s and 1990s. After howling responses of adults and pups in the early 1990s, attempts were made to capture and radio-collar a wolf in order to gain insight into their territory and reproduction. Those capture efforts were unsuccessful. Increased wolf activity was then documented again in that area during 2010–2011. Attempts by NPS and WDFW to radio-collar an individual in that pack in 2011 were again unsuccessful, despite a month of trapping effort. The "Hozomeen pack" remains intermittently active at the north end of Ross Lake and Silver Creek area. Consensus of the biologists in 2014 was that the Hozomeen pack territory was likely centered somewhere in southern British Columbia, and thus it was not included in annual reporting on the Washington state map. Wolves continue to travel in the upper Ross Lake area, with tracks of three animals travelling together on the west shore detected in 2015, and a group of three wolves traveling near Lightning Creek in January 2016. A radio-collared male disperser from the Loup Loup pack also traveled on both sides of upper Ross Lake in 2019, before departing into Canada. One other radio-collared male and one female, both from the Teanaway pack, also explored the Park during their dispersals in 2014 and 2015.

Gray wolf detections on the west slope of the Cascades have been rare. In 2018, a male-female pair was documented in an area spanning the western border of the Park in the Baker Lake and Diobsud Creek drainages, marking the first known pack west of the Cascades since extirpation (Figure 35). After the male's radio-collar ceased transmitting in 2019, this pack was re-designated as a single wolf territory by WDFW (2022).



Figure 35. Gray wolf voucher record NOCA39436, showing both members of the Diobsud pack (male on left, female on right) captured on a remote wildlife camera (operated by National Park Service), February 28, 2019, near Baker River, North Cascades National Park, WA.

There have been 175 confirmed or probable gray wolf observations in the Park from 1995–2020, in addition to the four known radio-collared wolves (Figure 35). Despite numerous carnivore survey efforts through time (Figure 36, Table 17), the frequency of wolf observations in the Park only began increasing rapidly in 2017 (Figure 3). Most of that increase in observations can be attributed to activity in the southern part of the Park. Wolves have been regularly detected from Bridge Creek to Horseshoe Basin, and south to Lake Chelan since 2016. In April 2017, a pair of black-colored wolves were detected travelling at Bridge Creek. Two wolves, one silverish black, and one tawny-colored were also frequenting the same area at that time. Single wolves were detected as far south as Lake Chelan.

Based on coat coloration and individual markings, at least 10 different individual wolves were active in the southern portion of the Park from 2018–2020. One gray-colored adult was wearing a non-functioning radio-collar in the Park from at least July 2017 to December 2018, but that individual's identity remains unknown. NPS biologists believe the collar was later dropped. From October 2018 to November 2020, two gray-colored adults were active throughout the southern part of the Park, and three gray-colored young wolves were detected together near McAlester Lake (November 2019): it is unknown if these were dispersers from the two gray-colored adults or not. In November 2020, a clear indication of pack activity coalesced in the Bridge Creek area. The "Bridge Creek pack" was comprised of two adults and four pups in November 2020 (Figure 34). The 2020 pups most likely

dispersed in 2021; after which, wolves were detected in smaller groups. The two adults were travelling with at least one other wolf in April 2021. Because NPS detections generally are not confirmed until early summer and the WDFW wolf report is released in spring, this pack has not been included in state reporting to date. NPS continues to closely monitor this apparent wolf territory.

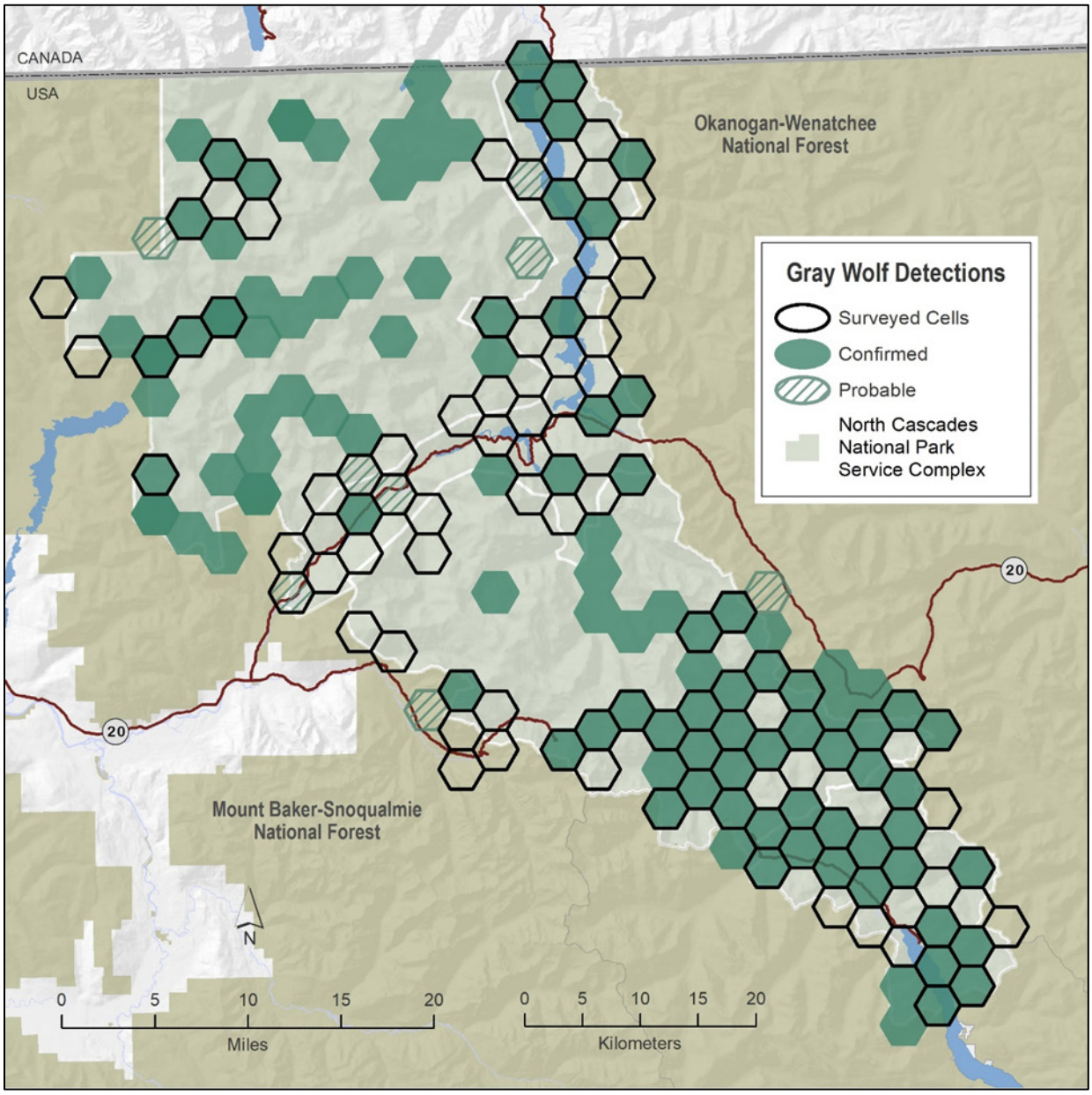


Figure 36. Survey locations, and gray wolf (*Canis lupus*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 17). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 17. Gray Wolf (*Canis lupus*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2003–2005	Forest carnivore study trail cameras at 24–40 stations	24 h camera stations, February–August each year	Christophersen et al. 2005, Christophersen 2006
2011–2020	Gray wolf monitoring trail cameras at 8–25 stations	24 h camera stations, October–May each year	NPS unpublished data
2014–2020	Radio-telemetry locations for 4 individuals	48–166 points per individual	WDFW unpublished data
2016–2017	Canada lynx study trail cameras at 40 stations	24 h camera stations, June–October each year	King 2019, King et al. 2020a, 2020b
2015–2017	Wolverine study trail cameras at 4 stations	24 h camera stations, November–June	Lewis et al. 2020, R. Long unpublished data
2018–2019	Montane carnivore study trail cameras at 64–66 stations, and scat DNA meta-barcoding	24 h camera stations, May–October each year; 508 total scat samples	Whiles 2021
2018–2020	Fisher study trail cameras at 56 stations	24 h camera stations, all months	Humphries 2022
2020	Black bear study trail cameras at 109 stations	24 h camera stations, July–September	J. Manning and NPS unpublished data

Habitat Requirements

Gray wolves are habitat generalists, able to live in forests, deserts, swamps, tundra, coasts, and prairies if their staple prey, large ungulates, are present (Fuller et al. 2003). In western North America, they predominantly inhabit forests and nearby open habitats, although occupancy is strongly influenced by the level of human tolerance/disturbance as well as prey availability (Houts 2000, Wiles et al. 2011). The proximity of occupied habitat to human activity and disturbance varies across the west, but generally they are known to utilize both public and private lands. Washington and Idaho populations primarily use U.S. Forest Service land (Wiles et al. 2011), while most packs in Montana exist on a greater diversity of property types (USFWS et al. 2011).

Packs establish their own territories in which they breed, rear pups, and hunt. During summer, pack activity is concentrated at den sites or selected gathering and resting areas called rendezvous sites. Rendezvous sites are often located in forest openings or wet meadows near the den (Ausband et al. 2010). Packs are known to utilize different areas of their territories from year to year depending on neighboring pack dynamics, as well as the distribution and availability of prey (Wiles et al. 2011). Territory sizes may also change for these reasons but are also known to increase with higher human densities, use of lethal control measures, and terrain ruggedness (Rich et al. 2012). The average territory size in Washington is 500 km^2 (57–1,125 km^2, n = 33; WDFW 2022), similar to those in Montana and Idaho (Ream et al. 1991, USFWS et al. 2011).

Conflict with humans often arises because both species require similar conditions to thrive, namely lower-elevation landscapes with a relatively mild climate and gentle terrain where prey are most abundant, especially in winter months (Boyd-Heger 1997). While protected, public land can serve as important dispersal corridors and refugia from conflicts with humans. The steep and rugged terrain within the Park limits ungulate prey abundance (and some other important supplementary prey such as beaver [*Castor canadensis*]) relative to areas outside its borders, likely reducing the suitability of portions of the Park wilderness landscape for pack establishment.

Diet

Gray wolves are highly adaptive, behaviorally flexible opportunistic predators and scavengers, although ungulates (deer, elk, and moose [*Alces alces*]) make up their staple diet throughout their range. Selection of ungulate and other prey species can vary greatly across seasons, packs, and regions in response to the availability and seasonality of prey, indirect effects of other predators (including humans), weather conditions, terrain, and many other factors (Kunkel et al. 2004, Smith et al. 2004, Stahler et al. 2006). Elk consistently make up substantial proportions of wolf diets across western North America (Wiles et al. 2011), and when moose are abundant, such as in southeast British Columbia, they can provide the greatest summer prey biomass (Stotyn 2008, Wiles et al. 2011). Packs within the same region of Glacier National Park (Montana) diverged substantially in their prey selection: while white-tailed deer is the most common prey for wolves in Glacier National Park, one pack in the same area consumed predominantly moose (61%), followed by elk (Boyd et al. 1994, Kunkel et al. 2004).

Wolves generally hunt in small groups of 2–4 individuals (Mech and Boitani 2003), but some solitary individuals are able to kill adult moose (Thurber and Peterson 1993). There is some evidence that hunting in small groups results in a small advantage in kill rate over solitary hunting (Peterson and Ciucci 2003) possibly by differentiating roles, such as in ambushing and relay chasing (Muro et al. 2011). Group hunting probably serves the additional purpose of allowing offspring to mature their skills prior to dispersing, thereby ensuring greater generational success (Mech and Boitani 2003). Predatory performance increases with experience to around age 4.5 to 5.5 years and subsequently declines with age (MacNulty et al. 2009). Despite group hunting advantages and skill development over time, most hunts are unsuccessful. In North America, measured kill rates average 7.2 kg prey/wolf/day (winter only; Mech and Peterson 2003), roughly equating to 17 elk/wolf/year or one 60 kg deer/wolf every 8.3 days (Wiles et al. 2011). Kill rates vary with season, prey species, and life history dynamics of the pack (i.e., reproductive stage, pack establishment or expansion; Jaffe 2001). For example, annual peak kill rates in late winter and spring coincide with greater energetic requirements of gestating and lactating females, and subsequently decline after pups are weaned in early June (Metz et al. 2012).

Gray wolves have an especially diverse diet in the summer that includes small mammals, birds, fish, invertebrates, and plant matter (Stahler et al. 2006). The small mammal prey documented throughout their range includes beavers, skunks, coyotes, porcupines, bears, snowshoe hares, ground squirrels, and various small rodent species (Boyd et al. 1994, Arjo et al. 2002). Avian prey has included ravens, ruffed grouse, and golden eagles (Boyd et al. 1994, Arjo et al. 2002). Salmon and marine mammals

may be common diet components in coastal regions, namely Alaska and British Columbia (Person et al. 1996, Darimont et al. 2003).

Gray wolf diet in the North Cascades is expected to be similar to that of wolves in the northern Rocky Mountains and predominately consist of deer, elk, and moose in some locations (Wiles et al. 2011). Given the recent and continuing colonization of wolves in the Park, empirical diet data are still sparse. However, fourteen wolf scats collected from 2018–2020 during a Parkwide mountain carnivore study were successfully sequenced for wolf and prey items using DNA meta-coding (Whiles 2021, CCP/NPS/WSU unpublished data). Of the 18 prey items detected, 39% were deer (*Odocoileus spp.*), 17% were mountain goat, 11% were hoary marmot, and the remaining prey items were singular detections of snowshoe, striped skunk, Pacific marten (*Martes caurina*), and domestic cattle (*Bos taurus*).

Reproduction

Male and female wolves disperse from their natal packs at one year of age and older, primarily to search for a mate and establish their own pack and territory or to join another existing pack elsewhere (Boyd and Pletscher 1999, Treves et al. 2009). Individuals typically delay breeding until at least two years of age (Fuller et al. 2003). Mating within the natal pack is strongly avoided (von Holdt et al. 2008), and unless a wolf achieves alpha status it will eventually part from its pack (Mech and Boitani 2003). Breeding within a pack is usually restricted to the alpha male and female, but if non-dominant females breed, in most cases it is with males from other packs (Smith and Ferguson 2005); consequently, packs generally have only one litter on an annual basis (Fuller et al. 2003). The peak of mating season is in mid- to late February (northern Rockies; Boyd et al. 1993), after which the pack searches for a den site prior to parturition in late April (Wiles et al. 2011). Dens are commonly underground burrows at elevated, dry forest sites near freshwater (Trapp et al. 2008) and protected within the central core of pack territories (Wiles et al. 2011).

Litter size is typically four to six pups (Fuller et al. 2003). Survival to adulthood is influenced by pack size and dynamics, disease, and prey availability (Johnson et al. 1994, Mech et al. 2008), and therefore can be highly variable. The canine viral infections known as parvovirus and distemper can cause significant, but short-term, decreases in pup survival rates (Wydeven et al. 1995, Smith and Almberg 2007). Because larger packs (10 or more individuals) are more likely to successfully rear pups (Mitchell et al. 2008), human-caused mortality that reduces pack size can have longer-lasting, intergenerational impacts on productivity (Brainerd et al. 2008).

Protective Status

The federal status of gray wolves in Washington has continued to change since the species was initially designated as endangered under the ESA in 1973. Federal protections for wolves in the eastern third of Washington state were removed in 2011 as part of the delisting of the Northern Rocky Mountain population, but wolves in the western two-thirds of the state maintained their endangered status. In 2019, the USFWS proposed the removal of ESA protections for wolves in the lower 48 states, including those in the western two-thirds of Washington. The ruling was passed in late 2020 and in effect by January 4, 2021, but then was reversed on February 10, 2022. All wolves in the Park are currently listed as endangered under the ESA.

Under Washington State law (WA Administrative Code 220-610-010), wolves are designated as endangered regardless of their federal classification, and are protected from possession, harassment, hunting, and killing. WDFW and tribal governments in Washington are responsible for local classification of gray wolves and will periodically review their status for possible reclassification. State down-listing to "threatened" may occur when two successful breeding pairs are present in each of the recovery regions for three consecutive years. This condition has been met in the Eastern Washington recovery regions but not in the Northern Cascades and Southern Cascades/Northwest Coast regions (WDFW 2022). The NPS has designated the gray wolf as a Management Priority species within the Park.

Known Threats

As in western North America, many gray wolf populations worldwide are currently stable or increasing, largely due to restoration and conservation efforts, stable prey populations, and their exceptional dispersal and adaptive capabilities. This outlook for the species has only recently improved in the Lower 48 states, and due to various anthropogenic threats, the situation has the potential to reverse trend in the near future.

Key factors for wolf survival within a given region include the level of human tolerance, livestock density and protection measures, ungulate availability, disease, and the abundance and connectedness of protected habitat. Susceptibility to climate change effects is considered low, given their wide resource niche and the overall low vulnerability of ungulate prey species to such effects (McKelvey and Buotte 2018). While viability of wolf populations is limited by ungulate availability, these prey populations are not considered at risk (IUCN accessed 18 Dec 2020). Deer, elk, and moose populations in most governances are maintained and regulated for hunting by humans, and their populations are believed to be stable or increasing in Washington and at broader geographic scales (Hoenes et al. 2016). The biggest threats to wolves in the North Cascades are direct killing by humans, habitat alteration, and disease.

Human-caused Mortality

The single greatest cause of historical gray wolf population declines and current mortalities is direct killing by humans (Mech 1970). The suite of motivations behind killing wolves has not significantly changed since early Euro-American settlement in North America, although the overall intensity and primary drivers have evolved. The initial focus of trapping for wolf pelts in the 1820s was largely replaced by the protection of livestock and general fear as human settlements and agriculture increased throughout the region, which continued with intensity until the functional extirpation of wolves in the early 20th century (Wiles et al. 2011). The onset of the environmental movement in the 1970s, as well as federal and state protections for wolves in the lower 48 states, have apparently curtailed unregulated persecution by the public, but in some states regulated hunting, government culling programs, and poaching are still widely practiced (Musiani and Paquet 2004, Treves et al. 2017). Some tribes also hunt or trap wolves on lands managed by their nation, as part of their own wildlife management plans.

Wolf-human conflict stems primarily from depredation of livestock and competition with hunters for ungulate species (Musiani and Paquet 2004). Some government agencies, including WDFW, remove

wolves responsible for livestock depredation on a limited case-by-case basis while emphasizing non-lethal preventative measures such as effective fencing, range riders, night penning, and guard animals (WDFW 2022). Lethal removal was widely used during wolf recovery periods in the northern Rockies and Great Lakes regions. In the northern Rocky Mountain states, 1,517 wolves were killed between 1987 to 2010 with the removal of 7–16% of the population annually since 2002 (Musiani and Paquet 2004). During 2021 in Washington, 30 wolf deaths were documented; two were killed in response to conflict with livestock, 22 were legally killed by tribes on tribal lands, four were struck by a vehicle, and two deaths remain under investigation (WDFW 2022). Collectively, these 30 mortalities represent 14.5% of the minimum known population in 2021 (206 wolves), which is well below the 30% threshold set by the agency. Regulations for lethal removal by agency staff and landowners generally relax as wolf populations grow (Fritts et al. 1992, Bangs et al. 2006).

Liberal hunting and trapping regulations for wolves and other large carnivores are commonly employed by some state agencies as another means to prevent livestock depredation and impacts on ungulate populations. Since removal of ESA protections for the restored Northern Rocky Mountain population, Idaho, Montana, and Wyoming have established open hunting and trapping seasons for gray wolves. Liberalization of hunting regulations has recently occurred in both Idaho and Wyoming. In 2020, Idaho increased annual hunting and trapping harvest limits from 10 to 30 wolves per person. In 2018, Wyoming introduced radical changes to allow any person without a license to kill wolves year-round across 85% of the state. Wolf hunting and trapping regulations in Alaska have also been continuously relaxed over the last 30 years (Ripple et al. 2019), most recently culminating in the 2020 decision to allow lethal take of females and pups from active dens within National Parks and Preserves. Removal of ESA protections across the Lower 48 has spurred other states, such as Wisconsin, Minnesota, and South Dakota, to establish new wolf hunting seasons. The “State-endangered” status of Washington wolves continues to protect them from hunting in the state regardless of changing federal regulations; however, wolves that inhabit tribal lands in the Eastern Washington recovery region are managed by those specific tribal entities and they can issue hunting licenses to tribal members (WDFW 2022).

Poaching has been known to account for a significant proportion of wolf deaths, despite the fact that inherent difficulties in detection and prevention likely produce vast underestimates (Gavin et al. 2010, Smith et al. 2010, Treves et al. 2017). In the northern Rockies, illegal take accounted for the second highest known cause of death (24%) behind legal control as well as a mortality risk of at least 6% in and around protected areas (Smith et al. 2010). In Wisconsin, estimates of wolf deaths from poaching ranged from 39–45%, and at least 80% of these cases were undetected by authorities or unreported by the public (Treves et al. 2017). Liberg and others (2012) estimated that two-thirds of all illegal wolf harvests in Sweden were undetected or unreported, even though these populations are some of the most intensively monitored in the world (Treves et al. 2017). There have been several high-profile poaching incidents in eastern Washington and the North Cascades since wolf recovery began, including the killing of several members of the Lookout Pack in 2010, and there are likely numerous others that go undetected in the region.

Habitat Loss and Fragmentation

Impacts to gray wolf populations from human-wolf conflicts can be intensified through human-derived habitat loss and fragmentation (Mech 1970, Theberge and Theberge 1998). Gray wolves are known for their aptitude to thrive in disturbed landscapes as long as there are certain degrees of human tolerance, relatively low livestock density and/or effective livestock protection measures in place, high ungulate abundance, and protected lands with safe dispersal corridors (Corsi et al. 1999, Mladenoff et al. 1999, Oakleaf et al. 2006). Wolves can thrive within developed and fragmented areas if home ranges have some spatial connectivity to other packs and refugia within protected areas (Haight et al. 1998, Smith et al. 2010).

Lower elevation public land in Washington is relatively fragmented compared to states spanning the northern Rockies wolf population (Wiles et al. 2011); however, a landscape analysis by Singleton et al. (2002) found that the degree of landscape permeability of the Cascades, north-central and northeastern Washington, and interior British Columbia are generally favorable for wolf travel. The authors also identified several dispersal barriers between these high-quality habitat blocks that include the Stevens Pass-Lake Chelan and Okanogan Valley corridors bordering the North Cascades. The four radio-collared wolves that traversed the Park 2014–2019 were able to successfully navigate routes through the rugged landscape, but large mountains and glaciers remain obstacles. Both Ross Lake and Lake Chelan are expansive water bodies that are oriented north-south along their length (i.e., perpendicular to the generally westward colonization of wolves in WA), but they likely do not represent impermeable barriers to wolf travel. Traversing around the lakes, or swimming across them, poses some risk to wolves and may slow colonization of the western slope, but this has not been empirically studied to date.

Disease

Disease is not currently identified as a major constraint on wolf populations in North America but remains a threat to individuals or packs due to local or regional exposure and the uncertainty of future emergence. Two common diseases in dogs and wild canines are canine distemper and canine parvovirus, which spread through fecal-oral contact (Nelson et al. 2012). Canine parvovirus is especially lethal to pups and old adults, and canine distemper can be lethal across all age classes. Exposure to this disease in wolf populations has been pervasive in some areas, based on antibodies detected (Mech and Goyal 1993, Sobrino et al. 2008, Nelson et al. 2012). In the Great Lakes region, canine parvovirus antibodies were annually detected in 13–95% of the population over an 11-year study, and in the northern Rocky Mountain region, 95% of 99 wolves had canine parvovirus antibodies and 24% had canine distemper antibodies over an 8-year study (Nelson et al. 2012). Prevalence of these diseases in wild canids has been associated with sympatric wild canids, domestic and feral dogs, and proximity to human developments across continents and populations (Nelson et al. 2012). Notably, across a 30-year study in the Great Lakes region canine parvovirus reduced pup survival by 40–60% and limited population increase by about 4% (Mech et al. 2008). This type of effect could disproportionately suppress small establishing populations, such as the wolves currently found in the North Cascades (Mech and Goyal 1993, Mech et al. 2008).

Both canine parvovirus and canine distemper can be transmitted from domestic dogs to gray wolves, but both are diseases that most domestic dogs are vaccinated for in the U.S. The likelihood of disease transmission from fully vaccinated domestic dogs is extremely low, but unvaccinated dogs and dogs recently vaccinated with modified-live virus vaccine could pose some risk to wolves. In a study of dogs given a modified-live parvovirus vaccine, 23% shed parvovirus DNA through a 28-day post-vaccination period, though the fecal load of parvovirus DNA in clinically healthy dogs was low (Freisl et al. 2017). Humans with unvaccinated or recently vaccinated companion dogs could expose wolves to diseases where trail networks are shared. The magnitude of exposure and risk to wolves in the Park is unknown. Domestic dogs frequent trails in Ross Lake NRA and Lake Chelan NRA, where on-leash dog-walking is permitted. On-leash dogs are also permitted on the Pacific Crest Trail where it traverses the south unit of the Park. Otherwise, it is not legal to have dogs on backcountry trails within North Cascades National Park, or off-leash on any trails in the Park.

Salmon-poisoning disease (SPD) is another potential threat to wolves that are colonizing areas where tributaries of the Columbia River and rivers south of the Snohomish River contain spawning salmon species. Salmon spawning in these rivers may carry the fluke *Nanophyetus salmincola*, which themselves may contain endosymbionts such as *Neorickettsia helminthoeca*, and *Stellantchasmus falcatus* (Greiman et al. 2016). Naïve consumers of infected salmon carcasses may contract salmon-poisoning disease (Headley et al. 2011, Vaughan et al. 2012, Robbins et al. 2018). SPD is known to affect canines and ursids and symptoms include loss of appetite, vomiting, diarrhea, lethargy, and death. Dogs, coyotes, and foxes may have a death rate from SPD as high as 90% (Foreyt et al. 1987, Headley et al. 2011). Populations of consumers that have co-evolved with the host species causing SPD are less sensitive, but naïve individuals such as widely dispersing wolves from areas without anadromous fish may experience acute symptoms (Robbins et al. 2018). Individuals that survive SPD appear to develop some resistance to future infection. While wolves are generally not considered a species likely to utilize salmon as a primary food source in the North Cascades, they can be opportunistic scavengers and would likely find the carcasses of spawned salmon as they explore new territory. Bobcats, coyotes, and black bears have been observed scavenging salmon carcasses along the Skagit River (NPS, unpublished data), and SPD has been clinically diagnosed in domestic dogs that have scavenged salmon carcasses in western WA. *Stellantchasmus falcatus* is newly arrived in the Pacific Northwest and will likely continue to expand its range as water temperatures warm in the coming decades. The prevalence of *Nanophyetus salmincola* and *Stellantchasmus falcatus* in North Cascades waters has not been well-studied and the risk to wolves has not been quantified.

Conservation and Research Needs

Because wolves have only recently began repopulating the Park, virtually all empirical data on their ecology and life history in the North Cascades are absent. Information on territory size, movements, reproduction, survival, dispersal, den site selection, genetics, diet, and disease exposure would help managers develop adaptive resource management strategies. Empirical data on elk and deer abundance, productivity, and seasonal resource selection in the Park are also lacking. Ungulate abundance and distribution could be a limiting factor for wolf establishment and persistence since these taxa represent the highest quality food source. Because deer are the most abundant ungulate in the Park, it is possible that wolf survival may be lower when compared to areas with high elk

abundance. Smith et al. (2010) found that poor wolf survival was related to mule deer as the primary prey source in the northern Rocky Mountains.

Wolves are in the early stages of repopulating the North Cascades, and new conservation measures should be incorporated into management practices in the Park and region. Specifically, fish in salmon spawning streams should be monitored annually for presence of *Neorickettsia* species and *Stellantchasmus falcatus*, and detections should be mapped through time. Carcasses of dead carnivores found in the Park should be tested for indications of SPD, as well as for canine distemper and canine parvovirus. Tissues from wolf carcasses should also be screened for environmental toxins and pesticides. Wolf hair collected from carcasses or genetic hair-snagging stations used in carnivore research projects should be genotyped and matched to wolves in the WDFW database to help determine the origin of an individual animal.

The effects of recreational activities on wolves are not known in the Park and should be studied; however, the risk of disease from other canines can be immediately mitigated. Domestic dogs visiting the Park should be fully vaccinated for canine distemper and canine parvovirus, and all dog feces should be bagged and placed in an appropriate trash receptable. Wolves are especially sensitive to human activity during denning and pup-rearing season, from roughly March through July. When a wolf den and/or rendezvous site is detected in the Park, a temporary closure should be established in accordance with the ESA and NPS policies. Such closures should provide at least a 1.6 km (1 mile) radius buffer around a den and at least 0.40 km (0.25 mile) around a rendezvous site (Appendix A). Likewise, location and status data for known den and/or rendezvous sites should be annually updated in the Park Resource Advisor's Guide to provide protection guidance during emergency wildland fire operations. Wolves are currently present at a low density in the Park and may not be as prone to vehicle strikes as some other species or under different highway conditions than are present in the Park. Nonetheless, wildlife crossing structures and/or active animal crossing signals could mitigate wolf deaths in areas where landscape features funnel wildlife into focused highway crossing areas.

Grizzly bear (*Ursus arctos*)

The grizzly bear (*Ursus arctos*) is one of three bear species that occur in North America, apart from the American black bear and the polar bear (*U. maritimus*) (Figure 37). Adults are generally large (average size of males in Selkirk Mountains is 161 ± 12 kg) but the species expresses significant variability in size across North American populations (Schwartz et al. 2003, Kasworm et al. 2016). Their hair color varies from blonde to black, often with silver- or creamy-tipped guard hairs that inspired their common name. Although black and grizzly bears can often be mistaken for one another, several unique features can be used to identify grizzly bears. These include the large shoulder hump, rounder and shorter ears, concave (dished) facial profile, and relatively long, prominent foreclaws (Figure 37).



Figure 37. Grizzly bear voucher record NOCA24122, photographed by John Ashley-Pryce near the east gate of Manning Park, British Columbia (north of the Park border) in 2015.

Prior to European settlement, grizzly bears ranged across much of western Canada, the contiguous U.S., and into northern Mexico, but now only occur in parts of Washington, Idaho, Montana, and Wyoming (about 2% of its historical range; Servheen 1999). Its historical range included most of Washington with the presumable exception of the Olympic Peninsula and the Puget lowlands west of the Cascades (Almack et al. 1993), although there is evidence that the bears occurred in the latter region prehistorically (Mustoe and Carlstad 1995). Today, the only extant population in Washington

occurs in and around the Selkirk Mountains in the northeast corner of the state (Pend-Orielle, Stevens, and Ferry counties; Lewis 2019).

The near extirpation from the contiguous states is a result of centuries of extensive direct killing, habitat loss, alteration, and fragmentation, and range restriction (USFWS 1993). To manage remaining populations, six grizzly bear recovery zones were established where the species was either extant or where grizzly bears were thought to have been present in the recent past (USFWS 1993). Two of the six federal recovery areas occur in Washington State (North Cascades Ecosystem [NCE] and Selkirk Ecosystem). Most of the estimated 1,900 bears inhabiting these recovery zones occur in the North Continental Divide Ecosystem (1,068 bears) and the Greater Yellowstone Ecosystem (737 bears) (USFWS 2021c). Small populations occur in the Selkirk Mountains (at least 53 bears) and the Cabinet-Yaak Ecosystem (55–60 bears), and it is believed that very few individual bears currently occupy the Bitteroot-Selway and North Cascades Ecosystems (USFWS 2011, 2016g, NPS/USFWS 2017). The grizzly bear population in British Columbia is widespread and relatively large (15,000 bears), although grizzly bears are considered threatened or extirpated in areas that border the NCE (British Columbia FLNRORD 2020).

The overall population status of the grizzly bear in the greater NCE is unknown but likely comprises less than 10 individuals (Rine et al. 2020). Only one detection of a grizzly bear has been confirmed in the greater NCE in the past 10 years. Another grizzly bear was observed three times (2010–2013). These bears were observed in British Columbia within 20 mi of the international border (Rine et al. 2020). Numerous efforts have been made to detect grizzly bears in the NCE, ranging from genetic hair-snag and scat surveys to baited and unbaited (on trail) remote cameras: no surveys have detected a grizzly bear in the Park from 1995–2020 (Figure 38, Table 18).

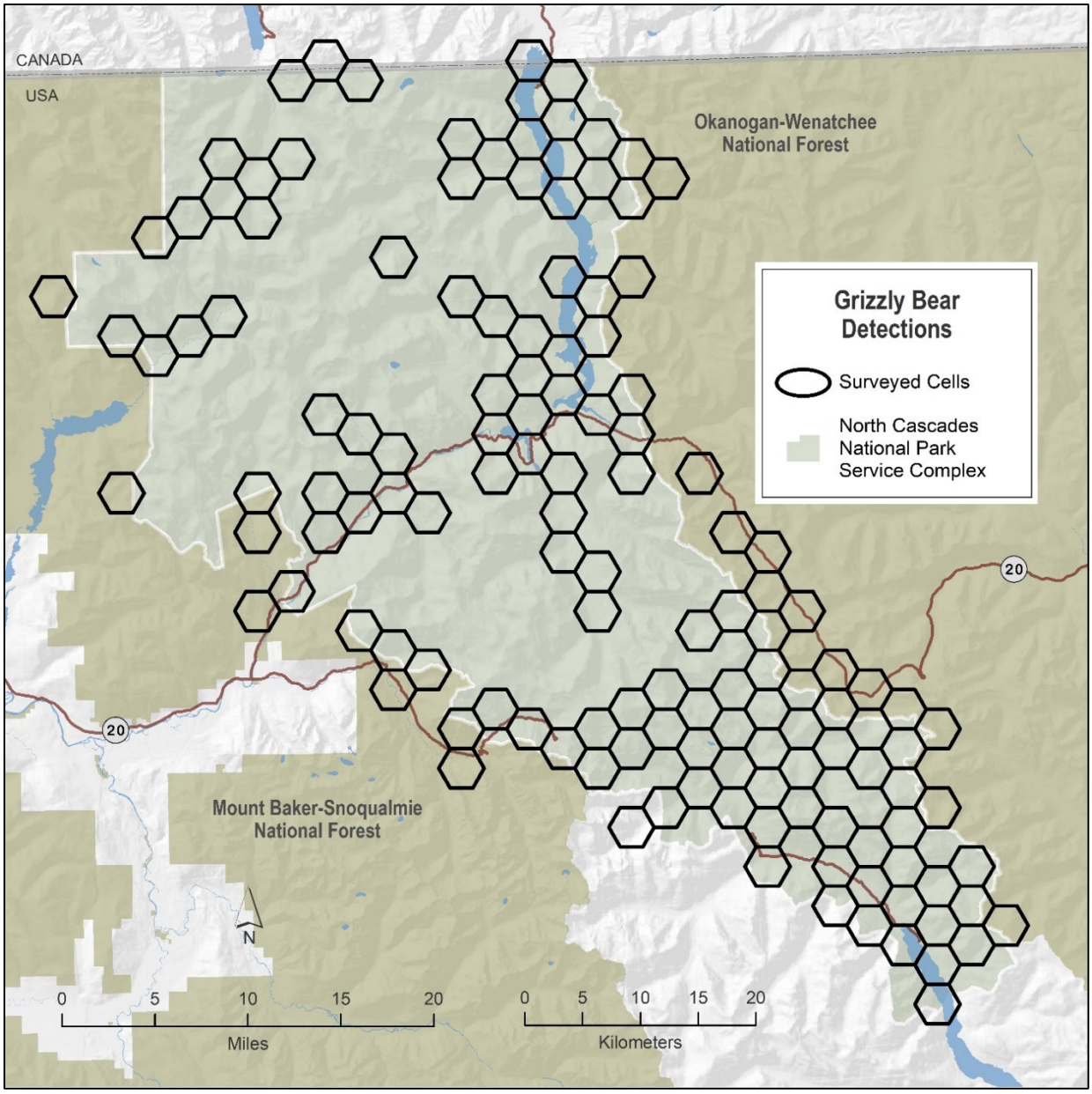


Figure 38. Survey locations, and grizzly bear (*Ursus arctos*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 18). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 18. Grizzly bear (*Ursus arctos*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
1999–2000	Bear genetic sampling at 9–19 stations (hair collection)	24 h genetic stations, May–September each year; 353 hair samples	Romain 2002, Romain-Bondi et al. 2004
2000–2001	Baited camera stations and scat DNA, using dogs trained to locate scat	24 h camera stations (286 nights), August–October 2000, May 2001	NPS unpublished data
2003–2005	Forest carnivore study trail cameras at 24–40 stations	24 h camera stations, February–August each year	Christophersen et al. 2005, Christophersen 2006
2008–2011	Black bear genetic sampling 36–90 stations (hair collection)	24 h genetic stations, May–September each year; 2,866 hair samples	Long et al. 2013
2011–2020	Gray wolf monitoring trail cameras at 8–25 stations [a]	24 h camera stations, October–May each year	NPS unpublished data
2016–2017	Canada lynx study trail cameras at 40 stations	24 h camera stations, June–October each year	King 2019, King et al. 2020a, 2020b
2018–2019	Montane carnivore study trail cameras at 64–66 stations, and scat DNA meta-barcoding	24 h camera stations, May–October each year; 508 total scat samples	Whiles 2021
2018–2020	Fisher study trail cameras at 56 stations	24 h camera stations, all months	Humphries 2022
2020	Black bear study trail cameras at 109 stations	24 h camera stations, July–September	J. Manning and NPS unpublished data

[a] Surveys were not designed for optimal detection of this species.

Habitat Requirements

Grizzly bears are able to inhabit a diverse array of ecosystems including mountainous areas, dry interior forests, tundra, grasslands, wet coastal forests, and some arid ecosystems. Key habitat requirements of grizzly bears are shaped primarily by the availability of high-quality food as well as the extent of human activity and access (Craighead et al. 1995, Mace et al. 1996, Schwartz et al. 2003, Boulanger and Stenhouse 2014). Using predictive spatial models, multiple studies have demonstrated that human infrastructure and activity are more important predictors of grizzly bear density than habitat and food quality due to their vulnerability to human persecution (Mattson and Merrill 2002, Apps et al. 2004, Ladle et al. 2018). Using historical data that approximated human densities and environmental conditions in 1850, Mattson and Merrill (2002) found that grizzly bear occurrence in the contiguous U.S. was less likely where the salmon range was most extensive. Rode et al. (2006) found that even a single human can displace grizzly bears from high-value feeding sites at relatively remote spawning streams in Alaska, and that this sensitivity is profound even at a landscape level. Moreover, it is well-established that grizzly bear populations are less likely to persist

within landscapes where road densities exceed 0.6 km per km^2 (USFWS 1993, Craighead et al. 1995, Mace et al. 1996, Boulanger and Stenhouse 2014). The bears require large, remote areas to establish their expansive home ranges (Singleton et al. 2004, Kasworm et al. 2016).

Grizzly bears choose winter den sites in remote terrain, usually on moderate to steep slopes (Linnell et al. 2000). Grizzly bears are adept at excavating soil and may also use an established cave, cavity, or excavation for hibernation, so denning habitat is not considered to be a limiting resource for most populations (Schwartz et al. 2003). Den sites appear to be selected where large, insulating snowpacks accumulate early and persist through the cold months. These tend to be on slopes just over 30 degrees where both the snowpack and soil are stable through the winter (Harding 1976, Vroom et al. 1980). The aspect of the slope varies with microhabitat conditions, but grizzly bears appear to select den sites on leeward slopes relative to prevailing winds where more snow is deposited. Many dens documented in mountainous terrain were at elevations between 2000 and 2500 m in small glade openings in subalpine herb meadows, subalpine forest, or in avalanche meadows (Vroom et al. 1980, Servheen 1983). Vegetative groundcover associated with these sites included grouseberry-heather, grass-forb meadows, and less often krummholz shrub meadows (Vroom et al. 1980). Pigeon and others (2014) found that dens sites were often in close proximity to high quality spring or autumn food sources.

In the NCE, locations of grizzly bear sightings and other signs are widely dispersed throughout the ecosystem, but encounters are concentrated east of the Cascade crest (Agee et al. 1989, Rine et al. 2020). Agee et al. (1989) found that grizzly bear sightings in the North Cascades were disproportionately concentrated in whitebark pine, subalpine larch (*Larix lyallii*), and open subalpine cover types. This pattern may be a function of whitebark pine seeds as a food source (as demonstrated in the Yellowstone region) and/or the reduced presence of humans at higher elevations where whitebark pines are distributed (Mattson and Merrill 2002). Much of the potential denning habitat in the NCE occurs within protected wilderness areas (Almack et al. 1993).

Diet

Grizzly bears are opportunistic omnivores, which allows them to exploit seasonal food pulses in various biomes. In addition to animal prey and carrion (e.g., mammals, fish, birds, insects), the majority of their diet in many regions is composed of a diverse array of plant materials that include herbaceous vegetation, roots, bulbs, berries, and seeds (LeFranc et al. 1987, Mattson et al. 1991, McLellan and Hovey 1995, Ransom et al. 2018). Using stable isotope samples collected from bears across the Selkirk and Cabinet-Yaak ecosystems between 1984 and 2015, Kasworm and others (2021) found that animal matter made up 12–22% of diets. Diets vary geographically depending on the availability of high-quality resources. Brown bears in coastal areas where human population density is low consume large amounts of nutrient-rich salmon during spawning runs, but do not rely on fish in most interior populations (Hilderbrand et al. 1996). Because the energetic demands of these large animals are high and because food availability varies temporally and spatially, individuals must employ many foraging strategies throughout the annual cycle (Schwartz et al. 2003).

After emerging from their winter dens, grizzly bears inhabiting mountainous areas forage heavily on starchy roots, tubers, and bulbs (e.g., *Hedysarum* spp., glacier lily) and may take advantage of

winter-killed ungulates and other carrion when newborn ungulates are not yet available (Mattson 1997). They transition to a wide variety of gramminoids (i.e., grasses, sedges) and forbs (e.g., clover, cow parsnip, horsetails) as they become more available throughout the summer months (LeFranc et al. 1987, Munro et al. 2006). Insects also become a more important diet item as biomass increases through mid-summer. In late summer and early autumn, the bears exploit sugar-rich berry crops and often increase their meat intake in order to build up fat reserves prior to hibernation (Welch et al. 1997). During this period of excess eating, called hyperphagia, grizzly bears can gain as much as 1.65 kg/day (3.64 lb/day) throughout the 2–4 months prior to hibernation. The thick layers of fat are vitally important as insulation and the sole source of energy during and just after the hibernation period (Craighead and Mitchell 1982, Erlenbach et al. 2014).

The NCE supports many of the plant, animal, and fungi species known to be primary grizzly bear food sources in other ecosystems. Within the NCE, there are at least 2,688 plant and fungus species that grizzly bears have been known to consume, including gramminoids, montane tubers, roots, and forbs (e.g., *Hedysarum* spp., glacier lilies) and forest plants (e.g., cow parsnip, horsetail, clovers, and berry-producing plants (Ransom et al. 2018). The NCE also supports at least 448 species of animal and insect food sources, including deer, elk, moose, carrion, ants, and beetles (Ransom et al. 2018). Trophic ecology of grizzly bears in the NCE is unknown, but it is likely that grizzly bears inhabiting the area would primarily persist on plants, as they do in other populations of interior North America (Almack et al. 1993, Gaines et al. 1994, Roberts et al. 2014, Ransom et al. 2018).

Reproduction

Grizzly bears have a promiscuous mating system facilitated by the wide-ranging movements of males, which has been confirmed by genetic analyses showing that cubs from the same litter can have different fathers (Craighead and Mitchell 1982, Craighead et al. 1998). Mating occurs May through July (peaks in mid-June), when males travel sometimes long distances to copulate with females in estrous. Fertilized embryos, however, are not implanted in the uterus until late fall when the female gains enough weight to survive the winter and nurse up to four cubs for 2–3 months inside the den. Cubs are born in the den between late January and early February after a 6–8 week gestation period (Schwartz et al. 2003).

Age at first reproduction for females depends in part upon their nutritional state but varies from five to seven years of age (Hilderbrand et al. 1999, Mattson 2000, Schwartz et al. 2003). Male grizzlies generally reach sexual maturity earlier, between four and six years of age (White et al. 1998). Females produce litters every two to three years until they cease successful breeding sometime in their mid- to late-20s (Schwartz et al. 2003, Wielgus et al. 1994). Grizzly bears have among the slowest reproductive rates of all terrestrial mammals, due to the combination of late age of first reproduction, small litter sizes, and long intervals between litters (Nowak and Paradiso 1983, Schwartz et al. 2003). Given these factors and lower survival rates of cubs (0.67–0.87) and subadults (0.66–0.95), it may take a single female 10 years to replace herself in a population (USFWS 1993, Schwartz et al. 2003). High adult survival rates are therefore critically important to maintain stability of grizzly bear populations (Lewis 2019).

Protective Status

The grizzly bear was listed as federally threatened within the Lower 48 under the ESA in 1975 (USFWS 1993). A Grizzly Bear Recovery Plan was established in 1982 and revised in 1993 (USFWS 1993). The North Cascades Grizzly Bear Recovery Zone was established as one of six recovery zones where grizzly bears were either extant or were known to have recently occurred in the lower 48 states (USFWS 1993, 1997). Since a 1998 ruling, any grizzly bears that may occur in the NCE would be considered "warranted but precluded" from up-listing to endangered status under the ESA. Specifically, the NCE population is currently in danger of extinction (endangered) but is precluded from this ESA designation due to the lack of recent verifiable detections and other higher listing priorities (USFWS 1998c, 2016g). While additional protections may be largely unnecessary where grizzly bears are not known to exist, the recognition that individuals in the NCE are "warranted" for a higher protective status provides greater incentive for initiating species recovery (Lewis 2019).

Grizzly bears have been designated as a Washington state endangered species since 1980. They are considered threatened or extirpated in areas of southern British Columbia (British Columbia FLNRORD 2020). Hunting grizzly bears is prohibited throughout British Columbia. Grizzly bears are designated as a Management Priority species by NPS.

Known Threats

Grizzly bears are highly adaptive omnivores that can make a living across much of the North American continent, but human activity, primarily in the form of direct killing, has driven many populations to extirpation. Human-driven habitat alteration is another significant threat that often reduces the access and abundance of natural food sources, dispersal corridors, and breeding and denning habitat. Moreover, habitat loss and fragmentation increase the risk of human-bear conflict, often leading to bear deaths. Grizzly bears are expected to be relatively insensitive to many expected outcomes of climate change on food resources (Roberts et al. 2014, Ransom et al. 2023), but effects may be variable and scale-dependent (such as phenology of food availability and distribution of patches). Lastly, disease does not appear to have population-level impacts on grizzly bears (Haroldson and Frey 2009); however, grizzly bears translocated into the NCE may be susceptible to salmon-poisoning disease, an emerging and potentially fatal bacterial infection known to moving north as a function of climate change (Robbins et al. 2018).

<u>Direct killing by humans</u>

Natural sources of mortality (i.e., old age, intra- or interspecific conflict, starvation, den collapse) in grizzly bears is well-documented; however human activity is, and historically has been, the single greatest cause of mortality for grizzly bears throughout their range (Schwartz et al. 2003, USFWS 2011, Hopkins et al. 2014, Kasworm et al. 2016). Human-associated impacts are numerous and varied, but the most significant anthropogenic impact is direct killing. Core motivations behind killing grizzly bears have not significantly changed since early Euro-American settlement in North America, but overall intensity and primary drivers have evolved (Mattson and Merrill 2002). The fur trade in western North America (1810–1872) initiated a precipitous decline of the species across its range as bears were killed for the profit of their pelts and to prevent injury to humans and human

resources (Mattson and Merrill 2002). Between 1826 and 1857, the Hudson Bay Company shipped 3,188 grizzly bear pelts from five trading posts bordering the NCE (Rine et al. 2020). Grizzly bears were generally seen as a threat to human safety and livestock, and therefore an obstacle to westward expansion and settlement; consequently, they were shot, poisoned, and trapped wherever they were found (USFWS 2022a). Increasing human density, livestock husbandry, and the loss grizzly bear habitat in the years since resulted in more frequent lethal interactions until the functional extirpation of many grizzly bear populations, including the NCE.

Grizzly bear killings and range contraction slowed after bears received ESA protections in 1975, allowing some populations to rebound and expand from 2% to 6% of their historical range in the Lower 48 (Haroldson et al. 2020, USFWS 2022a). Despite protections, direct killings continue to take an excessive toll in multiple recovery zones (Wielgus et al. 1994, Wakkinen and Kasworm 2004, Gould et al. 2022). Between 1975 and 1997, 77–85% of 99 mortalities across the interior mountains of British Columbia, Alberta, Washington, Idaho, and Montana were due to intentional human action, including predator control, legal harvest, self-defense, and malicious killings (McLellan et al. 1999). More recently, Kasworm and others (2016) found that 86% of 73 grizzly bear deaths in the Selkirk recovery zone were direct killings due to management removal of conflict bears, unknown human cause, poaching, misidentification by hunters, defense of self and property, malicious killing, and vehicle collisions. McLellan (2015) estimates that in the southern portion of their range, over 80% of grizzly bears older than yearlings are eventually killed by humans, mostly near roads, camps, or permanent settlements.

Habitat loss and fragmentation

Grizzly bears require large contiguous expanses of habitat with limited human influence to support their home ranges, high caloric requirements, denning areas, and dispersal corridors (Mace et al. 1996). Human presence and development severely intensify effects of habitat loss and fragmentation due to the potential for human-bear conflict and human-caused mortality (Craighead and Mitchell 1982, Schwartz et al. 2010). Grizzly bears generally avoid human activity and infrastructure; however, humans travel, recreate, introduce species, dam rivers, and harvest resources at large scales in grizzly bear habitat, making the suite of impacts pervasive (Mattson 1990). Specific habitat-related stressors include motorized access, developed or developing areas (residential and industrial), livestock husbandry, recreational activity, and activities that may disturb dens (USFWS 2022a).

Motorized access has an outsized set of detrimental impacts on grizzly bear individuals and populations. The presence and density of roads is a major influence on behavior and biology by bringing people into grizzly bear habitat, increases the potential for disturbing, displacing, or killing bears through increased noise, activity, vehicle strikes, direct killing or other detrimental human effects. Studies from across multiple ecosystems have shown that these processes alter bears' habitat use, movement patterns, home range selection, demographics, survival, and reproductive rates; however, motorized access management is known to be effective in mitigating these effects (Proctor et al. 2019). In the NCE, 64% of the recovery zone is protected from motorized routes which are almost entirely located outside of the Park (except for the North Cascades Highway and Stehekin Road). Roads in the NCE decreased grizzly bear habitat effectiveness by over 30% in models of

current carrying capacity (Lyons et al. 2018). Trails also bring humans into grizzly bear habitat and have been shown to directly impact carnivore occupancy (Baker and Leberg 2018).

Private land development is a primary driver of habitat fragmentation, human-bear conflicts, and grizzly bear mortality. Negative interactions often arise when bears opportunistically feed on anthropogenic food sources (e.g., livestock species, crops, refuse), but the mere presence of these large carnivores is easily perceived as threatening to human safety and property in many contexts (USFWS 2022a). For this reason, grizzly bear range continues to be defined by the presence of human populations in areas around recovery zones. Human development and rural sprawl not only limit grizzly bear range expansion, but also connectivity with adjacent populations and high-quality habitat. A disproportionate number of human-bear conflicts and grizzly bear deaths occur on private lands, and outnumber incidents on public lands (USFWS 2007b, Schwartz et al. 2010, Proctor et al. 2012); however, significant, population-level detrimental effects of human presence and facilities on protected public land is also well documented. In Yellowstone National Park for example, grizzly bears avoided occupying habitat closer to human facilities where they were management-trapped at a higher rate and where efficient foraging strategies were disrupted. Moreover, subordinate or security-conscious bears (i.e., adult females and subadult males) were displaced into habitat nearer developments by more dominant individuals (Mattson and Knight 1991).

Habitat fragmentation and isolation increase the exposure of grizzly bears to population-level threats, such as decreased demographic or genetic connectivity (USFWS 2022a). Grizzly bears in North America are highly vulnerable to these habitat-driven effects due to their relatively low population densities, slow dispersal and reproductive rates, and susceptibility to human-caused mortality (Proctor et al. 2012). Female grizzly bears are usually the limiting factor for demographic isolation of a population because they disperse at significantly shorter distances over time compared to males, and also have the potential to disperse with offspring to adjacent populations (McLellan and Hovey 2001, Proctor et al. 2004, Proctor et al. 2005). Consequently, females tend to be the focus of demographic and genetic connectivity goals (Proctor et al. 2005, Proctor et al. 2012, McClellan et al. 2021, USFWS 2022a).

Climate change

While it is difficult to predict the long-term outcomes from a rapidly changing climate, the highly adaptable nature and broad diet of grizzly bears may generally lessen the vulnerability of the species to changes in habitat and prey resources. However, alterations in food sources across landscapes are likely and could potentially exacerbate human-bear conflict and mortality as bears attempt to find other food sources. Warmer temperatures, delayed snowfall, and earlier spring melting may shift the timing of den entry and exit, which could also increase the potential for human-bear conflicts in spring and fall. Altered fire regimes may destroy important habitat but could also create high quality habitat in new areas (Roberts et al. 2014, Ransom et al. 2018).

As climate change shapes the NCE, there is considerable uncertainty as to how grizzly bear habitat may change through time (Ransom et al. 2018). Projected increases in growing season length, winter and spring water surplus, summer water deficit, wildfire, and decreases in snowpack will likely lead to substantial vegetation changes by the end of the century. These changes are in turn likely to affect

grizzly bear foraging and denning behaviors, and thus could influence their population dynamics through time. Under a low emissions scenario (Solomon et al. 2007), mountain huckleberry, an important grizzly bear food, is projected to increase its current range across the west by 84–112% by the 2080s. Under a high emissions scenario (Solomon et al. 2007), this species still was projected to increase its current range across the west by 66–78% by the 2080s (Roberts et al. 2014). Ransom et al. (2023) estimated the ecosystem could support a grizzly bear population under several climate change scenarios through the 2080s, with the amount of high quality habitat increasing across all models, scenarios, and time periods, as compared to current conditions. Those models estimated grizzly bear carrying capacity in the NCE could increase to 20–22 bears/1000 km^2 (males and females) (2080s: Ransom et al. 2023) from the previous estimate of 17 bears/1000 km^2 (current habitat conditions: Lyons et al. 2018).

Climate change projections for the North Cascades and across much of grizzly bear range indicate a general uphill migration of species, which may provide more bear foods as well as adequate denning habitat at higher elevations (Roberts et al. 2014). Contrastingly, lower elevations habitats are projected to decrease in summer species richness by as much as 50% by the 2080s (Roberts et al. 2014). In the North Cascades, this equates to a likely scenario of grizzly bears selecting for higher quality habitat that moves farther away from roads and human settlements through time. Some bears may persist at lower elevations, especially as bear density approaches carrying capacity. Notably, decreases in food availability in other parts of grizzly bear range have been related to increased human-bear conflict as bears search for alternate foods (Mattson et al. 1992, Pease and Mattson 1999, Gunther et al. 2004, Costello et al. 2014, Ransom et al. 2018).

Disease

Disease is not currently identified as a major constraint on grizzly bear populations in North America but remains a threat to some populations with the uncertainty of future emergence. Salmon-poisoning disease (SPD) is a potential threat grizzly bears and other carnivores that are colonizing areas where local rivers contain spawning salmon species. In rivers where salmon carry the fluke *Nanophyetus salmincola*, which themselves may contain endosymbionts such as *Neorickettsia helminthoeca*, and *Stellantchasmus falcatus*, naïve consumers of salmon carcasses may contract salmon-poisoning disease (Headley et al. 2011, Vaughan et al. 2012, Robbins et al. 2018). Populations of consumers that have co-evolved with the host species causing SPD are less sensitive, but naïve individuals such as long-distance dispersers from areas without anadromous fish may experience acute symptoms (Robbins et al. 2018). SPD is known to affect ursids and canines and symptoms include loss of appetite, vomiting, diarrhea, lethargy, and death. Individuals that survive SPD appear to develop some resistance to future infection (Ransom et al. 2018, Robbins et al. 2018).

The prevalence of *Nanophyetus salmincola* and *Stellantchasmus falcatus* in North Cascades waters has not been well-studied and the risk to ursids and canines has not been quantified (Greiman et al. 2020); however, grizzly bears translocated into the Cascades are not expected to select for salmon because they will be sourced from populations with non-piscivorous diets (NPS/USFWS 2017). Grizzly bears in other interior ecosystems of North America do not commonly consume fish even when they are abundant (Mace and Jonkel 1986, Hamer and Herrero 1987, McLellan and Hovey

1995, Mowat and Heard 2006, Munro et al. 2006). Bears are opportunistic feeders, however, and grizzly bears may encounter salmon carrion in some areas and would likely capitalize on them as black bears in the North Cascades do. To date, there is no evidence of black bears in the NCE contracting SPD. Under future climate scenarios, water temperatures are projected to warm which will likely cause declines in cold-water salmonids (Mantua and Raymond 2014). As these water temperatures warm, a northward range expansion of the fluke *Nanophyetus salmincola*, and its endosymbiont (*Neorickettsia* spp.), may emerge in the North Cascades. It has already been detected in the Columbia basin drainage and south of the North Cascades. Bobcats, coyotes, and black bears have been observed scavenging salmon carcasses along the Skagit River and Chilliwack River (NPS, unpublished data), and SPD has been clinically diagnosed in domestic dogs that have scavenged salmon carcasses in western WA (Ransom et al. 2018).

Conservation and research needs

Given the low number individuals and slow reproductive rate, grizzly bears in the NCE are functionally extirpated (as defined by USFWS). Despite extensive camera and genetic surveys, only one confirmed grizzly bear has been recorded over the last 10 years across the entire NCE, and no breeding or presence of cubs has been detected. The NCE is isolated from other grizzly bear populations (Gaines et al. 2001, Singleton et al. 2004, Proctor et al. 2012), the nearest occurring to the east in the Granby Kettle Grizzly Bear Population Unit in Canada and the Selkirk Mountains in Washington and BC (Proctor et al. 2012, British Columbia FLNRORD 2020). Grizzly bears may inhabit the remote areas east of the Okanogan River and west of the Granby Kettle Mountains but are limited to a very small number of animals. At present and for the foreseeable future, there are no population pressures in these areas that would cause grizzly bears to expand from the east into the Cascades, and extensive development between the two ecosystems makes successful immigration unlikely.

Because the NCE population has failed to recover naturally, the Grizzly Bear Recovery Plan (USFWS 1993) recommended a range of alternatives be considered under NEPA to determine how best to recover of this population. In accordance with the NCE recovery plan chapter, the NPS and USFWS are developing an Environmental Impact Statement / Restoration Plan (EIS) as joint lead agencies for grizzly bear restoration in the U.S. portion of the NCE (NPS/USFWS 2022). The WDFW is a cooperating agency in this plan, and other partner involvement includes the Southwest British Columbia Grizzly Bear Recovery Working Group (lead by the Okanagan Nation Alliance), British Columbia Ministry of Forests, Lands, Natural Resource Operations and Rural Development, and the Interagency Grizzly Bear Committee (IGBC). This EIS will update and re-evaluate impacts previously set forth in NPS/USFWS (2017).

Hoary marmot (*Marmota caligata*)

The hoary marmot (*Marmota caligata*) is a silver-backed, ruddy-tailed, and otherwise black-furred, semi-fossorial rodent (Figure 39). Hoary marmots select for talus slopes adjacent to alpine or subalpine meadows where they dig burrows and forage. One of the largest members of the Sciuridae family with adult body mass ranging from 3.5–7.0 kg, hoary marmots can lose over 40% of their weight during their 7–8-month hibernation period. Due to the patchiness of suitable habitat in high montane environments, hoary marmots are distributed in isolated colonies spread across the landscape, known as a metapopulation. Colonies typically consist of an adult male, 1–3 females, and 1–10 or more juveniles. In larger colonies, "satellite" subordinate males may also be present and breed. Marmot density ranges from 0.6 to 7.8 individuals per hectare. Mean home range of individuals is 13.8 ha, and mean foraging area is 9.2 ha (Braun et al. 2011). Like other ground squirrels, the hoary marmot is gregarious; colony sizes range from 2–36 individuals (Barash 1989, Braun et al. 2011). Social behaviors are largely (95%) intra-colonial, with the most common behavior being wrestling amongst juveniles (Holmes 1984). Marmots are also known as "whistlepigs" due to their propensity to emit four different types of loud, piercing cries that typically signal a predatory threat to other marmots (Taulman 1977, Braun et al. 2011). Conspecifics respond to alarm calls by becoming alert and either visually scanning the area or retreating to a burrow (Taulman 1977). Hoary marmots frequently exhibit three distinct types of vigilance: looking up while foraging, standing up while foraging to survey surroundings, and perching atop boulders to survey surroundings while basking in the sun (Holmes 1984; Barash 1989). Vigilance intensity is also thought to correlate with level of risk (Holmes 1984).



Figure 39. Hoary marmot voucher record NOCA42629, photographed by Jason Ransom/NPS, July 22, 2017, near Sahale Arm, North Cascades National Park, WA.

Several hoary marmot survey efforts have been conducted in the Park (Figure 40, Table 19). Initially, 128 potential trail-based sites were identified across the Park using a mosaic of 10 different marmot-suitable vegetation classes above 1,220 m, from the 1986 Landsat TM North Cascades Grizzly Bear Ecosystem Evaluation product (Almack et al 1993, Christophersen 2012). Of these 128 sites, a randomly selected and spatially independent sample of 31 sites was established and surveyed with point-transect methods from the hours of 0700–1200. Christophersen (2012) established point count stations every 400 m in either direction of the site until either suitable habitat or the survey window ran out; this design produced 123 point count stations ranging from 2–9 (mean = 4) per site.

In 2006 and 2007, Christophersen (2012) detected 242 hoary marmots at 19 of 31 (61%) survey sites with apparent site abundance ranging from 0–25 (mean = 7.8 ± 1.5 [SE]). Christophersen (2012) identified 61 hoary marmot colonies with this effort. In 2016 and 2017, Johnston et al. (2021) resurveyed 78 point count stations across 19 sites from Christophersen (2012), visiting each station and site once per year. Johnston et al. (2021) detected 377 hoary marmots at 17 of 19 sites across both years and used zero-inflated negative binomial models to analyze changes in abundance and relationships between abundance trends and weather conditions. Johnston et al. (2021) recorded a significant 74% decrease in marmot abundance between 2007–2008 and 2016, but they also recorded decreases between 2007–2008 and 2017 counts which were less severe at some sites and not significantly different. Johnston et al. (2021) interpreted the 2016 decrease as a lagged effect of the

2015 snow drought; their analysis showed that abundance decreased with increasing cold exposure without snowpack, winter weather was more influential than summer weather, the effects of cold exposure were mediated by vapor pressure deficit, and indications that mortality after snow drought was highest in juveniles (fewer 2-year old hoary marmots were counted than would be expected in 2016).

In 2018 and 2019, Whiles (2021) resurveyed 79 point count stations across 21 sites from Christophersen (2012), visiting each station and site 2–5 times per year. Whiles (2021) detected 498 hoary marmots at 20 of 21 sites across both years and identified a minimum abundance of 201 individuals. Whiles (2021) did not analyze abundance trends, but rather investigated predator-prey relationships by conducting behavioral observations of hoary marmots, trail camera surveys for predator presence, and scat surveys for predator diet. Whiles (2021) conducted 1,899 minutes of focal-animal sampling of hoary marmot behavior across 32 stations and 19 sites, 9,930 trap nights from 130 trail camera deployments, and DNA metabarcoding of 413 predator scat samples. Whiles (2021) did not find a relationship between hoary marmot vigilance and indices of predator risk. However, Whiles (2021) found that despite coyote detections being predominately localized to the southeastern portion of the Park, most hoary marmot predation events (7% of scat samples contained hoary marmot DNA) were attributed to coyotes (50%).

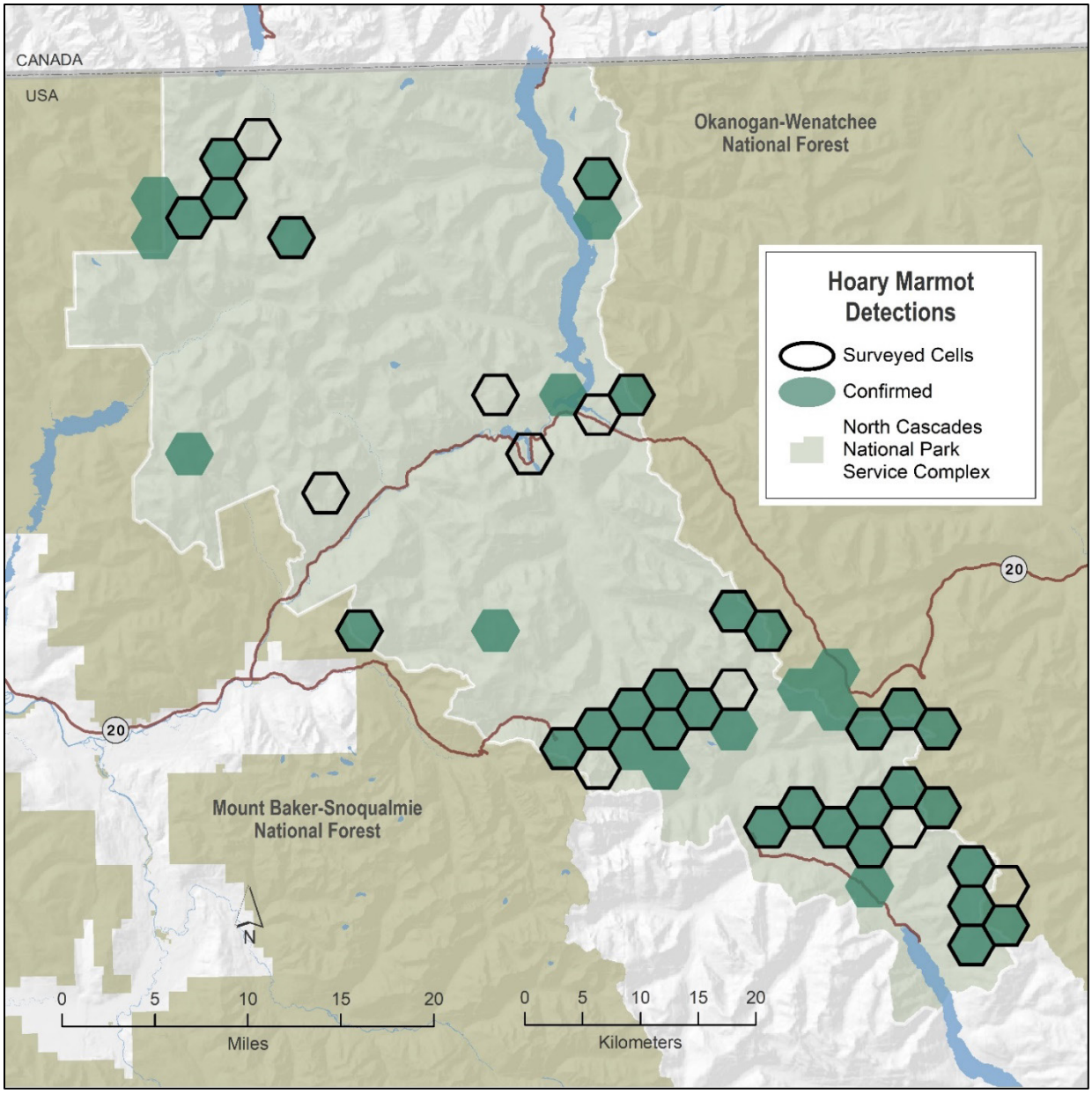


Figure 40. Survey locations, and hoary marmot (*Marmota caligata*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 19). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 19. Hoary marmot (*Marmota caligata*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2007–2008	Point surveys (visual observations) and incidental detections at 123 points	1–2 times per point, June–September each year	Christophersen 2012
2015	Point surveys at 13 points (visual observations) at datalogger stations	1 visit per station	NPS unpublished data
2016–2017	Point surveys 78 hoary marmot survey points (visual observations) and incidental detections	1 time per point, June–October each year	Johnston et al. 2021
2018–2019	Point surveys at 123 points (visual observations), trail cameras, and incidental detections	1–3 times per point, 64–66 camera stations, June–September each year	Whiles 2021
2020	Point surveys at 13 points (visual observations) at datalogger stations	1 visit per station	NPS unpublished data

Habitat Requirements

Hoary marmots primarily select for alpine and subalpine talus slopes adjacent to wet meadows (Barash 1989, Braun et al. 2011). While the talus-meadow association is most prevalent, hoary marmots may utilize a variety of secondary habitats including those dominated by heather and graminoid communities (Reichel 1986). This habitat occasionally contains interspersed krummholz islands as well as snowfields that persist throughout most of the summer, and hoary marmot colonies may shift altitudinally as snowmelt progresses. Hoary marmots are generally found at high elevations relative to the total local relief especially where they are sympatric with yellow-bellied marmots, but they have been recorded near sea level in northern portions of their range (Braun et al. 2011).

Hoary marmots construct and depend on a network of burrows ranging in density from approximately 6–49 burrow openings per hectare. Marmots use different burrows for different purposes including sleeping, nesting and raising young, refuge from predators, and hibernation (Braun et al. 2011). Hoary marmot habitat is therefore limited to areas with suitable substrate for burrow digging; this further supports selection for wet meadows with malleable soil and discourages selection for areas with bedrock or densely packed sedimentary rock substrate (Svendsen 1974). Hoary marmot habitat also requires appropriate boulders. Marmots often incorporate the interstitial space among talus matrices when establishing their burrows, and they rely on outcrops and larger boulders to serve as vantage points from which they can scan their surroundings (Svendsen 1974, Barash 1989).

In the Park, surveys have detected hoary marmots in the elevational range of 992 to 2,195 m (Johnston et al. 2021, Whiles 2021). Christophersen (2012) found talus to be the dominant cover type at 71% of hoary marmot locations, with forb meadow dominating at 21% and sub-shrub or larch cover types at 8%. Christophersen (2012) did not detect hoary marmots where available rocks were

small (< 0.2 m diameter), solely bedrock, or solely outcroppings. Hoary marmot detections generally increased with elevation and most occurred on south to west-facing slopes (Christophersen 2012). In Montana, Turnock et al. (2017) reported higher hoary marmot occupancy in shallower slopes, south-facing slopes, and in areas with shrub cover. Turnock et al. (2017) suggested that although hoary marmots generally forage on herbaceous foods, shrubs could benefit both dietary diversity as well as the insulative properties of snow during hibernation. Likewise, Turnock (2017) suggested that shallow, south-facing slopes likely accumulate more snow in the winter but also melt and uncover vegetation earlier in the year; this interpretation was also reached by Barash (1973) and Christophersen (2012).

Diet

Hoary marmots are generalist herbivores and central place foragers. From burrow networks into adjacent forage patches, hoary marmots make forays that are intermediate in frequency, duration, and distance than sympatric talus-inhabiting mammals. Barash (1973) recorded hoary marmots straying as far as 50 m from the talus when foraging, departing at a rate of 0.16 trips per minute, whereas American pikas made shorter, more frequent forays (5 m, 0.62 trips per minute), and Columbian ground squirrels were the least talus-dependent of the mammals studied (50 m, 0.05 trips per minute). All three sympatric mammals exhibited bimodal, diurnal activity patterns, with peaks of hoary marmot activity near the hours of 0900–0100 and 1800–1900 and less frequent activity from 1200–1600 and approximately 2000–0600 (Barash 1973). Bimodal, diurnal activity patterns, especially with respect to foraging, are well-documented for marmots (Barash 1989).

While hoary marmot diets can be diverse among forbs, graminoids, and shrubs, certain taxa are common throughout their range and populations studied. Likely marmot foods in the North Cascades include arnicas (*Arnica* spp.), buttercups (*Ranunculus* spp.), fireweeds (*Chamerion* spp.), fleabanes (*Erigeron* spp.), green false hellebore (*Veratrum viride*), louseworts (*Pedicularis* spp.), lupines (*Lupinus* spp.), paintbrushes (*Castilleja* spp.), ragworts (*Senecio* spp.), valerians (*Valerian* spp.), windflowers (*Anemone* spp.), and yellow avalanche-lily (*Erythronium grandiflorum*) as commonly eaten forbs, fescues (*Festuca* spp.) and sedges (*Carex* spp.) as common graminoids, and huckleberries (*Vaccinium* spp.), largeleaf avens (*Geum macrophyllum*), and willows (*Salix* spp.) as common shrubs. Other natural foods may include bark, lichens, and mosses. Hoary marmots select for leaves in early summer and flowers in midsummer, but relationships between diet composition and forage availability, biomass, or nutrient content have not been found (Holmes 1984, Braun 2011). Hoary marmots obtain water from standing sources, dew, moisture of forage, and snow, and they obtain salt and other minerals form rocks, soil, and urine. Hoary marmots are occasionally attracted to human-use sites where urine or sweat has accumulated (Braun et al. 2011).

Reproduction

Hoary marmots typically breed biennially, though successive years of both breeding and skipping reproduction have been recorded (Braun et al. 2011). Reproductive skipping is more often seen among subordinate females and those who are mating synchronously with other bigamously-mated females (Wasser and Barash 1983). Litters are spaced 3.3 years on average (Holmes 1984). Sexual

maturity usually occurs by three years of age, with reproductive female ages ranging from 2–9 years (Barash 1989, Braun et al. 2011).

Hoary marmot breeding typically occurs weeks within, or possibly before, emergence from hibernation, with birth occurring underground after a gestation of 25 to 30 days. Litter sizes of two to five have been recorded and tend to be larger if produced earlier in the season. Lactation lasts another 25 to 30 days. Young of the year emerge from the natal den later than the rest of the colony at 3–4 weeks of age; around this time, they are fairly precocial with open eyes, developed pelage, and are nearly finished weaning (Braun et al. 2011).

Mortality rate for young of the year is roughly 70% and can be nearly three times higher for the litters weaned latest in the season (Barash 1989, Braun et al. 2011). Patil et al. (2013) found juvenile mortality to vary widely from approximately 40–95% over five years. Patil et al. (2013) found a negative relationship between juvenile survival and high Pacific Decadal Oscillation Index values, suggesting shallow snowpack and early spring snowmelt is detrimental to hoary marmot recruitment. Juvenile survival may also be tied to size of the social group (Patil et al. 2013). Juvenile dispersal usually occurs in the second year of age, though dispersal of older individuals has been recorded (Braun et al. 2011). Dispersal is nearly three times more likely for males and is positively correlated with presence of new litters and low overwinter mortality (Barash 1989).

Protective Status

On a range-wide basis, the hoary marmot population is reported to be stable, widespread, common in suitable habitat, and lacking serious threats. Therefore, as of 2016, hoary marmots are listed by the IUCN as "Least Concern" and by NatureServe as "Globally Secure" throughout their range (Cassola 2018, NatureServe 2022). NatureServe listed the species as apparently secure in the northern and southern jurisdictions of hoary marmot range, which includes Washington (NatureServe 2022). The State of Washington lists hoary marmots as "Protected" which prevents the species from being legally hunted (WAC 220-200-100).

Though hoary marmots do not currently merit conservation concern throughout most of their range, closely related congeners, the Olympic marmot (*M. olympus*) and Vancouver Island marmot (*M. vancouverensis*) do. The Olympic marmot is listed as "Imperiled" by the state of Washington and NatureServe and is a candidate for the state's endangered species list (WDFW 2015, NatureServe 2022). The Vancouver Island marmot is listed as "Critically Imperiled" by the province of British Columbia and NatureServe and neared extinction with a wild population of less than 30 in 2003 (Environment and Climate Change Canada 2020). Some threats, such as anthropogenic habitat alteration, novel predation regimes, and especially isolated habitat, are likely more pronounced for these congeners than for hoary marmots; however, similar patterns are worth noting in the context of hoary marmot persistence, especially at the trailing edge of hoary marmot range in Washington where shifts in biotic interactions are likely to occur first (Schemske et al. 2009). Given the apparent large declines in hoary marmots and vulnerability to climate change (Johnston et al. 2021), NPS maintains hoary marmots as a Management Priority species in the Park.

Known Threats

Like other high-elevation, montane species, hoary marmots will be particularly affected by a warming climate. Climate change will have direct, abiotic effects on marmot survival (e.g., increased cold exposure during snow droughts while hibernating, as well as indirect, biotic effects such as tree and shrub encroachment into meadow habitats) (Armitage 2013). Predation pressure may also increase if coyote distribution continues to expand and if predators take advantage of decreased snow persistence to access high-elevation areas earlier in the year (Witczuk et al. 2013, Whiles 2021). Though somewhat adaptable, marmots may be particularly susceptible to certain forms of habitat alteration such as logging, though this is less of a concern in the Park and elsewhere in Washington as much of hoary marmot range is protected in designated wilderness (Bryant 1996, Raymond et al. 2014).

Anthropogenic Habitat Alteration

Habitat alteration via forestry management can greatly affect marmot survival, as was especially evident for Vancouver Island marmots in the past several decades. Bryant and Page (2005) found predation to be the most common cause of Vancouver Island marmot mortality between 1992 and 2004. Intriguingly, Bryant and Page (2005) suggest a cascading effect of logging on predation patterns: first, black-tailed deer abundance increased partially in response to increased foraging opportunities in regenerating clearcuts; second, cougar and gray wolf abundance increased in response to increased primary prey (black-tailed deer) abundance; third, especially in times of decreased primary prey abundance, cougars and gray wolves used marmots as alternative prey above historic rates. This dynamic may have been exacerbated by increased predator mobility on logging roads and decreased vigilance efficiency from marmots in regenerating clearcuts (Bryant and Page 2005).

Bryant (1996) found net reproductive value of Vancouver Island marmot colonies to be nearly three times higher in natural versus recently logged habitats. Females had shorter life spans in logged areas and never produced more than one litter, whereas 36% of females in natural areas did (Bryant 1996). Clearcuts may act as ecological traps for marmots as they resemble meadows and support requirements for survival shortly after logging but not after regeneration, consequently failing to produce as many marmot dispersals as are consumed (Bryant 1996).

Though forestry management is very restricted in Washington hoary marmot habitat due to overlap with areas managed as wilderness, it is possible that high-intensity, large scale wildfires could cause "clearcut" effects that negatively affect marmots over time. Conversely, wildfires may act to maintain meadow habitat and prevent tree establishment as discussed in the following section (Raymond et al. 2014).

Climate-Driven Habitat Alteration and Population Effects

Upslope encroachment of native plant species, especially conifers, is an oft-cited concern for the persistence of hoary marmots and other meadow-dwelling species (Raymond et al. 2014). This phenomenon is complex and varies among species and climate variable interactions; areas projected to experience increases in both temperature and precipitation may see the most pronounced proliferations of woody plant cover (Parmesan 2006). Raymond et al. (2014) posit that the extent of

high-elevation meadows, and therefore habitat to species such as the hoary marmot, will certainly decrease under climate change. Though prescribed fire may be considered as a management action to help maintain montane meadows, this action is not very relevant in Washington as most hoary marmot range in the state occurs in wilderness areas. However, the remoteness of hoary marmot habitat in Washington also aligns with minimal suppression of wildfire by humans, potentially allowing for more natural fire regimes and maintenance of montane meadows (Raymond et al. 2014).

To explain hoary marmot declines in the Park, Johnston et al. (2021) evaluated alternative hypotheses of climate-driven forage phenology shifts and adverse winter weather conditions. Models incorporating remote-sensed measures of the normalized difference vegetation index had less support than weather-based models. Johnston et al. (2021) found that snow droughts can reduce insulation during hibernation and lead to greater exposure of hoary marmots to acute cold events in winter. Johnston et al. (2021) also report what appears to be a lagged effect of this phenomenon with particular relevance to juveniles. It is also important to note that the abiotic and abiotic effects of climate change are not mutually exclusive; while winter weather events appear to be a key element in hoary marmot survival, decreased snowpack is also associated with upslope plant encroachment into meadow habitats of marmots (Raymond et al. 2014, Johnston et al. 2021).

Predation

Whiles (2021) detected hoary marmot DNA in 28 of 413 (7%) of predator scat samples collected in 2018 and 2019. Despite coyote detections being predominantly localized to the southeastern portion of the Park in 2018 and 2019, most hoary marmot predation events were attributed to coyotes (50%). Pacific martens (32%), gray wolves (7%), cougars (7%), and black bears (4%) were also recorded consuming hoary marmots; whereas bobcats, Canada lynx, ermines, and long-tailed weasels were not. Whiles (2021) also found that coyote probability of use was negatively influenced by snow persistence, suggesting that coyotes may become more prevalent and influential to hoary marmot survival in the Park as snowpack decreases during climate change.

Though findings from Whiles (2021) can't determine causal relationships, shifts in predation guilds negatively affecting marmots are well documented. Similar to findings in the Park, Witczuk et al. (2013) found that coyotes disproportionately preyed on Olympic marmots and are possibly the driving factor behind the species' decline. Witczuk (2013) cites local extirpation of gray wolves as a contributing factor of coyote invasion of the Olympic highlands. Gray wolves are currently recolonizing the Park which may help to mitigate coyote expansion in the Park. Bryant and Page (2005), as discussed above, inferred predation to be the primary driver of precipitous declines of the Vancouver Island marmot.

Conservation and Research Needs

Because hoary marmot habitat in the Park is largely found in areas with low human accessibility, there is a lack of information concerning this species. Recent studies have begun to address abundance trends and possible abiotic and biotic associations. However, the data collected so far is primarily appropriate for determining site occupancy or, at best, abundance derived from distance-sampling correction methods. Resurveying established sites in the Park, preferably with multiple visits within a season to help account for sampling bias, would provide managers with longitudinal

records of hoary marmot colony persistence (Christophersen 2012, Johnston et al. 2020, Whiles 2021). Perhaps more importantly, intensive demographic studies, as has been conducted for Olympic and Vancouver Island marmots, would yield greater insights for hoary marmot population trends and causes of mortality in the Park (Bryant and Page 2005, Griffin et al. 2008). Further, it is likely that the nutritional ecology of hoary marmots in the Park has yet to be studied at an appropriate scale (Johnston et al. 2020). Abundance data coupled with fine-scale measures of vegetation associations would better elucidate responses related to forage phenology shifts, decreasing palatability of forage under drought conditions, and upslope encroachment of woody plant species into meadow habitats of hoary marmots. The effects of recreational activities on marmot occupancy, persistence, and behavior are currently unknown, though many popular trails in the Park transit occupied marmot habitat. Routine management, facilities, and infrastructure operations conducted in the Park can affect hoary marmots, but adherence to noise and disturbance guidelines around burrows may help mitigate local impacts (Appendix A).

Western gray squirrel (*Sciurus griseus*)

The western gray squirrel (*Sciurus griseus*) is the largest tree squirrel in the Pacific Northwest and is one of three native tree squirrels found in Washington State. Contemporary presence and breeding of western gray squirrels in the Park has been confirmed (Figure 41). Three subspecies have been identified within its range, with *S. g. griseus* present across Washington. Pelage across the back and flanks is steel gray, the belly and throat are pure white, and the tail is gray with white-frosted tips (Figure 41). Western gray squirrels are primarily arboreal and often use connecting tree canopy to travel, although they regularly travel and forage on the ground near trees. Like other tree squirrels, they are generally solitary, but appear to be more wary of human presence and development than other species (Wiles 2016).



Figure 41. Western gray squirrel voucher record NOCA24260, photographed by Jason Ransom/NPS, January 4, 2018, near the Golden West Visitor Center, Lake Chelan National Recreation Area, WA.

Western gray squirrels were once distributed widely throughout low- to mid-elevation areas of Washington, Oregon, California, western Nevada, and northern Baja California (Dalquest 1948, Escobar-Flores et al. 2011). Their current distribution has diminished to scattered and discrete populations, regulated in part by the presence of declining oak (*Quercus* spp.) woodlands (Rodrick 1986). In Washington, the historical distribution of western gray squirrels included the southern

Puget Trough to the Columbia River, throughout the Columbia River Gorge, and at low- to mid-elevations along the eastern slopes of the Cascade Range to Okanogan County (Booth 1947, Larrison 1970, Linders and Stinson 2007). Today, distribution in the state has been reduced to three geographically and genetically distinct populations: southern Puget Trough (Joint Base Lewis-McChord and surrounding areas in Pierce and Thurston counties); Klickitat (Klickitat, Skamania, and Yakima counties); and the North Cascades (Chelan and Okanogan counties; Linders and Stinson 2007). Other small, scattered populations have been observed in parts of Clark (southwestern WA) and Kittitas (central Cascade Range) counties.

The North Cascades population represents the northernmost periphery of the western gray squirrel range and is ecologically unique among the Washington populations in that the region lacks oak trees used for forage and nesting, which extend only as far north as southern Washington (Stein 1990, Stuart 2012). Their expansion outside the range of oaks may have resulted in part from the planting of walnut trees in Chelan and Okanogan counties; however, western gray squirrels are also found in natural habitats in the region (USFWS 2003). In such places like Stehekin, western gray squirrels rely on hypogeous fungus, ponderosa pine nuts, Douglas-fir seeds, and a variety of other sources for forage (Stuart 2012, Johnston 2013).

Sparse information about the status and trends of Washington populations suggest that western gray squirrels were once more numerous and widespread than today but were historically uncommon and in some places locally common (Linders and Stinson 2007). Populations had already declined by the late 1800s and the species was considered rare by 1970 (Wiles 2016). Reductions in abundance and range have been attributed to habitat loss and degradation, altered fire regimes, predation, recreational hunting, disease, highway mortality, competition and predation interactions with introduced wildlife, and continuing genetic isolation (Hamer et al. 2001, Warheit 2003, Linders and Stinson 2007, Vander Haegen et al. 2013). The only statewide population estimate concluded that there were between 469 and 1,405 western gray squirrels between 1994 and 2005, including 705 in the Klickitat population, 42 in the Puget Trough population, and 190 in the North Cascades (Wiles 2016). The North Cascades population extends along Lake Chelan to the lower Stehekin River, and eastward into the Methow Valley.

Fewer data exist for the North Cascades population compared to the other two Washington populations. Surveys were conducted in the Stehekin valley in 2004, which resulted in observations of eight western gray squirrels and 28 nests (Hamer et al. 2005). The most recent survey data (Stuart 2012) resulted in population size estimates of 36 and 476 western gray squirrels in Stehekin and the Methow Valley, respectively. In addition, the effective population size in Stehekin was calculated at 7.2 squirrels compared to the corresponding estimate 95.3 squirrels for the Methow Valley (Stuart 2012). A density estimate for the North Cascades population is not available, but probably does not exceed that of the Klickitat population (0.23 ± 0.08 individuals/ha; Vander Haegen et al. 2005, Stuart 2012), which is considered much lower than estimates in other states (Linders and Stinson 2007). In addition to surveys and radio-telemetry data, there are only 14 confirmed and probable western gray squirrel observations recorded in the Park wildlife database from 1995–2020 (Figure 42, Table 20);

however, this species is frequently observed by staff and residents in Stehekin and often goes unreported.

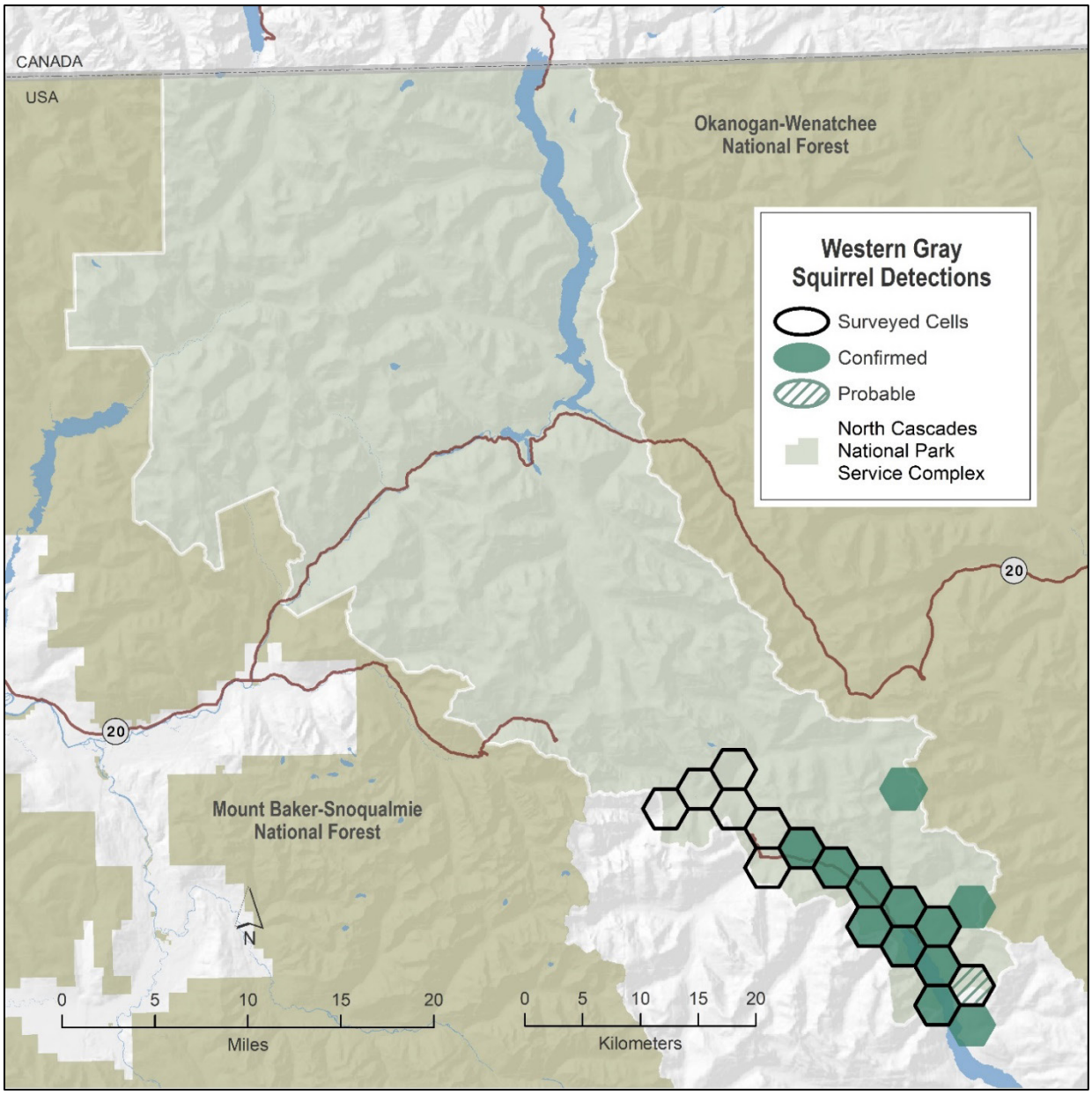


Figure 42. Survey locations, and western gray squirrel (*Sciurus griseus*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 20). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 20. Western Gray Squirrel (*Sciurus griseus*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
1995–2020	Nest occurrence surveys at various points in the Stehekin Valley (visual observations)	1–2 times per site per compliance project	NPS unpublished data
2004	Nest occurrence surveys throughout the Stehekin Valley and genetic hair sampling at 12 stations	September and November: 308 h total search effort; 28 nests	Hamer et al. 2005
2008–2011	Radio-telemetry locations for 24 individuals	≥1 location every 5 days per individual; 1,690 total locations	Stuart 2012, Stuart et al. 2018
2008–2011	Nest occurrence surveys at Buckner orchard and Golden West area	Single search of 7.4 km2; 54 nests	Stuart 2012, Stuart et al. 2018

Habitat Requirements

Forest structure and composition are critical when identifying western gray squirrel habitat suitability. Preferred stands are comprised of groups of trees that form a thick upper canopy among areas with lower canopy cover (Linders and Stinson 2007). Dense canopies are important for efficient travel, escape and cover from predators, and nesting. Ryan and Carey (1995) found that western gray squirrels selected for stands with contiguous tree canopy that facilitates arboreal travel out to at least 60 m from nests. This form of habitat connectivity is essential for dispersing squirrels, which often use riparian zones as movement corridors, especially where dry uplands do not support much tree cover (Linders et al. 2010, Wiles 2016). Sparse ground cover and proximity to water are other common features of stands occupied by western gray squirrels (Hamer et al. 2005, Wiles 2016).

Preferred stands are generally dominated by conifers but intermixed with deciduous trees. Mature, large-seeded mast-producing trees are sources of plentiful food and therefore key components of high quality habitat. Presence of western gray squirrels throughout much of their range (including that in Washington) is associated Oregon white oak (*Quercus garryana*). In the North Cascades where oak is absent, occupied stands consist mainly of Douglas-fir and ponderosa pine with fewer numbers of other species such as bigleaf maple, black cottonwood, and trembling aspen (Hamer et al. 2005, Stuart 2012). In Stehekin, core use areas were composed of 64% Douglas-fir, 14.6% bigleaf maple, and 11.3% ponderosa pine (Hamer et al. 2005). Ground cover tended to be sparse at these sites, yet masting shrubs (*Rubus* spp., Oregon grape, Pacific dogwood) were often present (Hamer et al. 2005).

Western gray squirrels utilize stick nests (dreys) and tree cavities for sleeping, resting, and rearing young. Dreys are constructed with various plant material including twigs, sticks, leaves, and moss, and formed into rounded covered shelters or broad platforms. Covered shelters are used during colder months and for rearing young, while platforms are more suitable for temporary and seasonal use (Ryan and Carey 1995, Linders 2000). Dreys are most often located adjacent to the south facing trunks of large trees, and in the top third of the canopy (Foster 1992, Hamer et al. 2005, Stuart 2012).

Dwarf mistletoe (*Arceuthobium* spp.) brooms are often utilized as nests, and high rates of mistletoe infection can be an important predictor for nest tree selection (Gregory et al. 2010). Tree cavities are valuable for resting and rearing young, and often selected when they are available (Carraway and Verts 1994). One squirrel or a family uses an average of 4.6 nests throughout the year (Stuart 2012), and any given nest may be shared among different individuals (Linders 2000). In the North Cascades, nesting areas are generally characterized by large diameter, tall trees with presence of dwarf mistletoe infection, in addition to dense canopy (Stuart 2012). Park surveys found that 28.6% of the nests within these plots were in ponderosa pines even though they made up only 11.3% of tree species, indicating a preference for ponderosa pines where Douglas firs are more readily available nest choices (Hamer et al. 2005).

Diet

The principal food of western gray squirrels across most of their range is acorns, although hypogeous fungi (truffles and false truffles), conifer seeds, green vegetation, fruit, and insects are also important diet items (Dalquest 1948, Linders and Stinson 2007). Hypogeous fungi are significant dietary components for Washington and other populations (Carraway and Verts 1994, Ryan and Carey 1995, Johnston 2013), with 21 genera eaten in the North Cascades, including *Rhizopogon*, *Geopora*, and *Melanogaster* (Stuart 2012). Plant material could not be identified in fecal samples of Stehekin western gray squirrels; however, other foods consumed in the southern Puget Trough population that also occur in Stehekin include Douglas-fir and ponderosa pine seeds, maple samaras, black cottonwood catkins, and Douglas-fir bark (Johnston 2013). Pine nuts are likely essential for storing body fat and conditioning for winter (Wiles 2016). Foraging for conifer seeds may occur throughout the year but is most intensive in August when cones are developing (Johnston et al. 2019b).

Reproduction

Western gray squirrels are sexually mature at 10–12 months of age, with males taking slightly longer to reach this stage than females (Fletcher 1963, Swift 1977). The breeding period (from copulation to successful rearing) can last from December to June or July. Within this period, copulation can occur anytime from December to late June or July, but in Washington, females are usually pregnant by February or March (Ryan and Carey 1995, Linders et al. 2010). Gestation lasts for about 44 days until parturition sometime between March and July, followed by an approximately 56-day rearing period that in some cases lasts into late August (Foster 1992, Ryan and Carey 1995, Linders et al. 2010). Young are born naked and blind (USFWS 2003). Most evidence indicates that females produce only one litter each year, but in rarer instances, two litters may be possible (Linders 2000, Vander Haegen and Orth 2011). Litter sizes range 1–5 young, with averages of 3.3 ± 0.7 (SD) in the Klickitat population (Vander Haegen et al. 2005) and 3.0 ± 0.5 in the Puget Trough population (Johnston 2013), respectively.

Protective Status

As of 2016, IUCN has assessed the global status of the western gray squirrel to be of least concern because of its supposed large population, widespread distribution, and because it is unlikely to be declining at the rate necessary to qualify for the threatened designation (Lacher et al. 2016). NatureServe has listed the species as globally secure, but imperiled in Washington; however, the last status review was in 1996 (NatureServe 2021).

This species has not yet received federal protection under the ESA; however, the USFWS recognizes the western gray squirrel as a Species of Concern in western Washington and the USFS has identified it as a "management indicator species" and "sensitive species" for oak-pine forest communities (USFWS 2016h). The USFS designation does not provide any official protection or management directive for the species or its habitat in national forests. In 2001, a petition was filed with the USFWS to list the Washington populations as threatened or endangered under the ESA, but after review, the action was denied because the individual or collective populations did not qualify as a distinct population segment within its range, nor did they constitute a significant portion of the range (USFWS 2003).

Western gray squirrels were classified as a threatened species 1993 by the Washington Fish and Wildlife Commission, and this species is currently proposed for up-listing to endangered status. Under the current designation, the species cannot be hunted, trapped, or killed within the state (WAC 232-12-011). Western gray squirrels are also identified as a Species of Greatest Conservation Need under the Washington State Wildlife Action Plan and are identified as a Priority Species under the WDFW Priority Habitat and Species List (WDFW 2015). A periodic status review by WDFW in 2016 recommended that the threatened designation be upheld due to small population sizes, ongoing threats, and a lack of data to indicate a significant change in population status (Wiles 2016). The NPS has designated western gray squirrels as a Management Priority species within the Park.

Known Threats

The most widely cited threats to western gray squirrels across their range include habitat loss and degradation, disease, climate change, continuing genetic isolation, predation, recreational hunting, highway mortality, and interactions with introduced wildlife (Warheit 2003, Linders and Stinson 2007, Vander Haegen et al. 2013, Wiens et al. 2014). Recreational hunting and highway mortality are threats that are less relevant to the Stehekin population, since hunting of this species is not permitted within the state, and highways do not exist within this population's range; however, vehicle strikes in Stehekin have been documented. Barred owls, a known predator of western gray squirrels, continue to expand their range and likely present some degree of threat, but the broader effects of this predator on the North Cascades populations are not well known. Four major threats that are known to (or have the potential to) impart population-level effects on the Stehekin and greater North Cascades populations are habitat loss and alteration, disease, climate change, and continuing genetic isolation.

Habitat Loss and Alteration

The decline of western gray squirrels throughout their range has precipitated mainly from human activities that lead to conifer-hardwood habitat loss, fragmentation, and alteration, such as development, logging, wildfire, and fire suppression (Linders and Stinson 2007). Each of the Washington populations have been adversely impacted by various forms of habitat alteration. Urbanization and development have been the primary drivers of habitat loss in the Puget Trough population, while forest management practices and wildfire have most severely affected the North Cascades population with additional strain from development in Okanogan valley bottoms (Linders and Stinson 2007, USFS 1996, Stuart 2012).

While some forestry practices can improve western gray squirrel habitat, logging can destroy nest sites, fragment tree canopies necessary for travel and escape cover, and diminish or eliminate food sources (Vander Haegen et al. 2004, Linders and Stinson 2007). Logging in Okanogan habitat has emphasized partial cuts where large trees are often removed, thereby having a disproportionately negative effect on canopy density and nesting sites (Linders and Stinson 2007). Variable density thinning is the most suitable method of timber harvest for enhancing western gray squirrel habitat, and involves creating a mosaic of forest openings, thinned matrices, and unthinned patches. This system encourages tree growth to increase mast production while reducing fuel loading as well as maintaining corridors of dense canopy for arboreal travel (Linders and Stinson 2007).

The historical practice of fire suppression has altered forest structure and species composition over time, resulting in increased tree density and associated diseases (e.g., dwarf mistletoe), litter depth, and fuel loading, as well as reduced tree diameter (Agee 1993, Stuart 2012). These conditions generally reduce habitat value for western gray squirrels and heighten the risk of frequent, large scale, catastrophic fires across the landscape that can destroy squirrels and large areas of their preferred habitat (Agee 1993, Graham and Jain 2005). Fire management practices that may benefit squirrel habitat broadly include thinning (especially of forest understories and ladder fuels) and prescribed burning (Stuart 2012).

Disease

Notoedric mange, a highly contagious skin infestation caused by an ectoparasitic mite *Notoedres centrifera*, is the most significant disease known to affect western gray squirrels with the potential to devastate populations (Linders and Stinson 2007). Mange outbreaks have occurred in the Klickitat population since the early 1930s, where infection is the second most common source of mortality after predation (Vander Haegen et al. 2013), but outbreaks have not been reported in either the North Cascades or the Puget Trough populations (Wiles 2016). The highest morbidity rates have been associated with periods of nutritional stress from drought, degraded habitat, or poor mast years (Gurnell 1987, Cornish et al. 2001, Linders and Stinson 2007).

Other diseases and parasites have been observed in western gray squirrels, including western viral equine encephalitis, fleas, ticks and mites, lice, coccidia, ringworm, intestinal roundworms, papilloma, botflies, and coccidiosis (Carraway and Verts 1994, Linders and Stinson 2007). Coccidiosis is the only other disease apart from mange that is associated with high morbidity in western gray squirrels. Most researchers believe that the use of multiple nests by individual squirrels may reduce their exposure to these ectoparasites (Linders and Stinson 2007).

Climate Change

The projected level of impact from climate change on western gray squirrel population is not clear and may depend largely on forest management practices employed in the coming decades. The North Cascades population is especially susceptible to intensified fire regimes that reduce habitat quality over large areas and periods of time (Stuart 2012). Changing climate patterns have likely already increased the frequency and intensity of wildland fires in the region, and such changes are projected to accelerate in the future (Littell et al. 2010). The major wildfires in Okanogan and Chelan counties in 2014 and 2015 catastrophically burned large areas of western gray squirrel habitat due to low

snowpack and hot, dry weather ignited by lightning strikes (Stamper 2014). The scale and magnitude of such fires may be attenuated to some degree by employing beneficial forestry management practices like prescribed burning and forest thinning. Warmer temperatures may also increase the exposure of western gray squirrels to diseases (e.g., mange, coccidiosis) already shown to affect populations in Washington. Despite the potential impact of fires and increasing disease outbreaks, Johnston and others (2012) predicted little range loss and possibly an eastward expansion from Stehekin in the next century under warmer, drier conditions. Elevational expansion of range may also occur under future conditions.

Small Population Size and Isolation

Western gray squirrel populations fluctuate naturally with stochastic events (e.g., mast production, disease outbreaks), but the risk of extirpation can be much greater for small, isolated populations like those in Washington. Such populations usually experience a decline in genetic diversity over time that may lead to inbreeding depression and lower fitness, which may in turn affect their ability to respond and recover from environmental fluctuations. Western gray squirrel populations in Washington are known to have significantly reduced genetic diversity compared to Oregon and California populations by all measures, including extreme reductions in heterozygosity and allelic richness (Warheit 2003). This loss of genetic diversity probably resulted from genetic drift that occurs when alleles are removed from the population but not replaced (Warheit 2003), and in such cases the introduction of individuals may be required to offset the loss of fitness (Reed and Frankham 2003, Linders and Stinson 2007). By the early 2000s, the Puget Trough population was considered at heightened risk of extirpation due to its small size, but an augmentation program in 2007–2012 may have partially counteracted adverse effects (Vander Haegen and Orth 2011). The Stehekin population may also be in need of augmentation at some point, as it has lower genetic diversity and smaller effective population size (N_e = 7.2 squirrels) than the Methow Valley population (N_e = 95.3 squirrels); however, the North Cascades populations collectively have a higher effective population size than previously thought, and there is no evidence of genetic bottlenecks or isolation by distance between them (Griffith et al. 1989, Stuart 2012). Results from the Stuart (2012) study suggested that observed squirrel mortalities in Stehekin were not tied to genetic factors.

Conservation and Research Needs

While some western gray squirrel research has been conducted in the Park, this species is not surveyed or monitored on a regular basis. Repeated surveys through time and across seasons, conducted under a consistent protocol, would provide important information toward conservation of the species in the Park. Such data will also be necessary to inform the state's periodic status assessment, and thus legal designation, of the species. Continued efforts in western gray squirrel habitat modeling may also help inform management and conservation actions through time. Because Johnston (2012) only modeled future presence based on geographic distribution, it remains unknown how habitat and environmental conditions may influence persistence of the species. Insights into the effects of snow depth and seasonal (rather than annual) temperatures may help better understand long-term outlooks for this species.

Fire, fuel management, and the effects of climate change related to fire, are all important considerations for conservation of western gray squirrels. Of note, squirrels in the Stuart (2012) study used fuel-treatment areas in Stehekin, but nested in non-treated areas with high canopy cover and dwarf mistletoe infection. This indicates the importance of maintaining connected canopy continuity throughout mosaic forest where fuel treatments must occur. Additional research could focus on how squirrel reproduction, survival, and distribution are influenced in such mosaic landscapes where fuel reduction treatments have occurred. Management, facilities, infrastructure operations, and fire fuel management activities conducted in the Park can have significant impacts on western gray squirrels simply because of their very small, localized population. Surveys for squirrel presence and nests are required by NPS prior to management operations and adherence to noise and disturbance guidelines around nests may help mitigate local impacts (Appendix A).

Regular genetic monitoring of western gray squirrels in the Park is needed in order to proactively manage decline due to losses of genetic diversity. Such data can be used to inform the need for genetic augmentation in an adequate timeframe that aims at avoiding extirpation of the population. This will also enhance understanding of habitat connectivity and identify habitat corridors between populations of squirrels in the North Cascades. The diet of western gray squirrels in the Stehekin Valley also needs to be better quantified toward understanding the relative importance of specific plant species and the effects of fire fuel reduction activities on those foods. Comparing diversity and relative abundance of plant species found in squirrel diets by Stuart (2012) to vegetation sampling in areas of high and low squirrel use and fuel-reduced and untreated forest would contribute to better management and conservation practice in the Stehekin Valley.

Wolverine (*Gulo gulo*)

Wolverines are the largest terrestrial member of the mustelid family, with adult males weighing up to 18 kg (40 lb) and females reaching 12 kg (26 lb). Contemporary presence and breeding of wolverines in the Park has been confirmed (Figure 43). Wolverines have a circumpolar distribution across the northern tundra, taiga, and forests (Banci 1994), and are among least understood mammals in North America due to the low population density across often remote and inaccessible habitat (Banci 1994, Aubry et al. 2007). Wolverines are one of the rarest carnivores in the contiguous U.S., with a 2016–2017 multi-state survey across Idaho, Montana, Washington, and Wyoming resulting in identification of only 50 individuals (26 female, 24 male) (Lukacs et al. 2020). The total population of the contiguous U.S. was estimated in 2010 at 250–300 individuals (USFWS 2010b). Occupancy surveys in Washington 2016–2017, found that 36% of the 25 sampled cells were used by wolverines (Lewis et al. 2020).



Figure 43. Wolverine voucher record NOCA55744, captured on a remote wildlife camera (operated by National Park Service) February 19, 2022, near Agnes Creek, Lake Chelan National Recreation Area, WA.

Due to their low abundance and secretive life history, our knowledge of wolverine exact range is limited. Self-sustaining populations in the contiguous U.S. exist in boreal forests of the Cascade Range and Rocky Mountains (Aubry et al. 2007), with documented individuals in the Sierra Nevada (Moriarty et al. 2009) and northeastern Oregon (Magoun et al. 2013). There are also historical, but no contemporary, records from the Great Lakes region and New England (Aubry et al. 2007, Copeland et al. 2010). Wolverines underwent significant range contractions, particularly in the southern and

eastern edges, by the mid-1900s (Aubry et al. 2007), and genetic evidence suggests that historic populations within the contiguous U.S. were extirpated by the early twentieth century. These extirpations are likely due to human activity, including mining, high elevation grazing, unregulated trapping and persecution through predator control efforts (Aubry et al. 2007).

The northern and southern wolverine populations in the western states appear to have been discontinuous prior to European settlement; southern populations were genetically isolated from the rest of the North American population for more than 2,000 years (Copeland et al. 2010). These populations, which included the Sierra Nevada and portions of the southern Rocky Mountains, appear to have been extirpated and the isolated genetic lineage eliminated (McKelvey et al. 2014). Other than a single male wolverine, which had dispersed from the Rocky Mountain population (Moriarty et al. 2009), there have been no sightings in California since 1930, and no evidence that California now sustains a viable population of wolverines. Three wolverines were detected in northeastern Oregon during monitoring in and around the Wallowa-Whitman National Forest and Eagle Cap Wilderness between 2010 and 2012, with one individual detected multiple times, making it a likely long-term resident in the area (Magoun et al. 2013). It is unclear if there is an established breeding population in northeastern Oregon, or if the detections are dispersers that will fail to establish residency there (Magoun et al. 2013).

The contemporary wolverine population found in the Cascade Range is the result of more southern range expansion of wolverines from northern populations (McKelvey et al. 2014). The Washington Cascades represents the southernmost confirmed extent of the wolverine's range in the Pacific states (Aubry et al. 2016). Wolverines apparently became rare in Washington during the first half of the 20th century, with no records in Washington State between 1941 and 1964. Records in Washington increased significantly in the later part of the 20th century, with 23 records between 1970 and 1990 (Banci 1994). In 2009–2010 wolverines were detected around Mt. Adams and elsewhere in the southern Washington Cascades (WDFW 2012b). Notably, detections of wolverines in the North Cascades increased from only 0–2 annual detections 1995–2013 to 3–18 annual detections 2014–2020 (Figure 3): it is unknown how much of this change is due to technological advances in detection methodology and increasing number of humans using the backcountry, as compared to increasing wolverine abundance. Park records from direct observations and wildlife surveys contain 18 probable and 55 confirmed reports of wolverine in the Park from 1996–2020, in addition to radio-telemetry locations from 11 individuals (6 males, 5 females) that used the Park during 2007–2015 (Aubry et al. 2016) (Figure 44, Table 21).

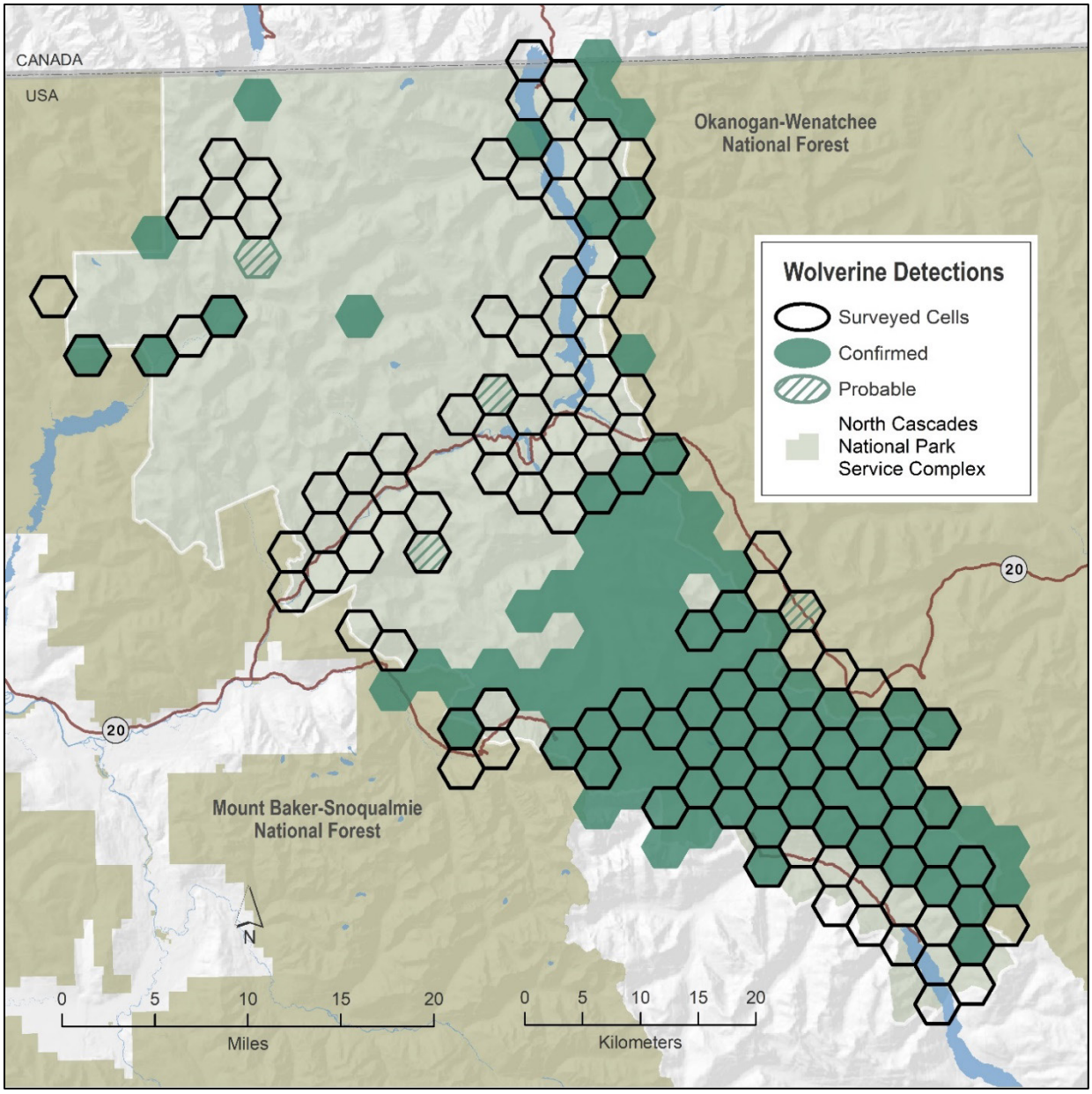


Figure 44. Survey locations, and wolverine (*Gulo gulo*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 21). Hexagon survey and detection cells shown are 10 km^2 in size.

Table 21. Wolverine (*Gulo gulo*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2003–2005	Forest carnivore study trail cameras at 24–40 stations	24 h camera stations, February–August each year	Christophersen et al. 2005, Christophersen 2006
2007–2015	Radio-telemetry locations for 11 individuals	Multiple points per individual	Aubry et al. 2016
2011–2020	Gray wolf monitoring trail cameras at 8–25 stations	24 h camera stations, October–May each year	NPS unpublished data
2016–2017	Canada lynx study trail cameras at 40 stations	24 h camera stations, June–October each year	King 2019, King et al. 2020a, 2020b
2015–2017	Wolverine study trail cameras at 4 stations	24 h camera stations, November–June	Lewis et al. 2020, R. Long unpublished data
2018–2019	Montane carnivore study trail cameras at 64–66 stations, and scat DNA meta-barcoding	24 h camera stations, May–October each year; 508 total scat samples	Whiles 2021
2018–2020	Fisher study trail cameras at 56 stations	24 h camera stations, all months	Humphries 2022
2020	Black bear study trail cameras at 109 stations	24 h camera stations, July–September	J. Manning and NPS unpublished data

Habitat Requirements

Wolverines are active within their home range year-round, with reported estimates for wolverine home range size ranging 87–388 km^2 for females, and 238–1,417 km^2 for males (Inman et al. 2007). During the 2007–2015 telemetry study of wolverines in the North Cascades, the activity areas for 14 individuals (7 female, 7 male) were smallest for females with kits (293–643 km^2), larger for non-reproductive females (535–1,961 km^2), and largest for males (1,149–2,992 km^2) (Aubry et al. 2016). The large home range, in relation to body size, indicates that wolverines exist in a relatively unproductive niche in their winter foraging ground, and large expanses of land are required to find sparse food resources. The exploitation of unproductive landscapes is advantageous because it allows for relatively low interspecific competition for food resources (Inman et al. 2007, Inman et al. 2012, Inman 2013).

Wolverines are well adapted for snow travel with a low, compact body and large feet to aid in travel over snow, and dense fur for protection from extreme temperatures (Copeland et al. 2010). Wolverine records are strongly associated with levels of persistent spring snowpack, coinciding with denning activities that take place in early February to late march (Copeland et al. 2010, Aubry et al. 2007). In the North Cascades study, 91% of telemetry locations were located in areas with snow cover that persisted into late spring (Aubry et al. 2016). Wolverines don't seem to select for a specific vegetation community within their habitat, and distribution appears to be mainly limited by patterns of spring snowpack. This is evident by the restriction of southern populations, which inhabit

warmer climates with more restricted snowpack, to higher elevations, where the northern populations have more expansive patterns of habitat use (Copeland et al. 2010). In the southern portions of the wolverine's range, individuals tend to disperse during summer to cooler regions and microclimates within their home range, suggesting that high summer temperatures may play an important role in habitat selection in warmer climates (Copeland et al. 2010). The "refrigeration hypothesis" suggests that the presence of persistent snowpack is important not just for denning and thermoregulatory purposes, but also to preserve cached food resources for as long as possible (Inman et al. 2012).

Forest structure may be important to wolverines that inhabit forested landscapes. Two densely forested locations where wolverines have repeatedly been detected on trail cameras in the Park, January–April 2018–2020, were at low elevations (412 m and 239 m). Den sites in forested areas are often in structural components like trees and logs, so lack of structural components may be limiting for wolverines. A wide distribution of both potential dens and rendezvous sites are likely important as female wolverines with young kits need to roam in order to find sufficient food (Banci 1994).

Diet

Wolverines are thought to feed primarily on carrion, and are exceptionally strong, with reports of wolverine carrying away moose and caribou (*Rangifer tarandus*) carcasses (Banci 1994). Large mammal carrion is reported to be a critical component of wolverine diet, at least in winter, but they will prey on birds, eggs, insects, small mammals, and fish, and may rely on these prey sources when ungulate carrion is scarce (Banci 1994). In British Columbia, Canada, main diet components of wolverine included moose, caribou, mountain goat, snowshoe hare, beaver, hoary marmot, and porcupine. It is suspected that carrion created by other large predators may play an important role in the wolverine's diet, so the presence or absence of other apex predator species may be a limiting factor for wolverine food availability (Inman et al. 2012). The importance of small prey species in wolverine summer diets is not well understood (Inman et al. 2012), but hoary marmot and porcupine frequency in reproductive female diets was found to be higher than in other demographics (Lofroth et al. 2007). In southern British Columbia and southwestern Alberta, the presence of marmot habitat was one of the strongest predictors of wolverine occupancy, along with protected areas and road density (Lofroth et al. 2007, Kortello et al. 2019).

Reproduction

After three years of age, nearly all female wolverines mate and become pregnant every year, but successful implantation and birth appear to be food-limited (Persson 2005). Low food availability resulted in the loss of pregnancy in about 50% of the non-food-supplemented females in that study. Food-supplemented wolverines bred more frequently and also raised kits to weaning more successfully than non-food-supplemented females. Breeding rates are thus likely driven largely by winter food availability. Persson (2005) suggested that it may take two years for females to store up enough energy to successfully reproduce. Reproduction in wolverines is on a somewhat earlier timescale than many non-hibernating carnivores, allowing them to take advantage of winter-killed ungulate carrion to fuel lactation requirements, and exploit a brief period of summer food abundance for post-weaning growth (Inman et al. 2012). A scarcity of winter forage is likely the cause of pregnancy failure and early litter loss (Inman et al. 2012).

Wolverines undergo delayed implantation during their reproductive cycle, whereby a fertilized egg is suspended in the blastocyst stage for months before implanting in the uterine wall between November and March. Gestation lasts for 30–40 days, and kits are born between January and April. Although litter sizes of up to six kits have been recorded in captivity, generally fewer than three kits survive past den abandonment. Kits are weaned around week nine or 10 and reach adult size at seven months (Banci 1994). A radio-collared female studied by Aubry et al. (2016), only produced one kit at her den in the Park, and a female at another documented den in the South Cascades was observed with two kits (J. Akins, unpublished data).

Wolverine natal dens are comprised of snow tunnels up to 60 m long and are often associated with a large structure like a boulder or log. A minimum of 1.5 m of persistent snow cover is required, and likely provides thermal regulation of the den and protection from predators (Aubry et al. 2007, Banci 1994, Copeland et al. 2010). Kits are often moved from a natal den to a secondary maternal den prior to weaning, but the reasons for these den shifts are not fully understood. Natal and maternal dens are generally abandoned by late April or early May as they begin to be inundated by snowmelt, and the young kits are kept at various rendezvous sites while the female is out hunting (Inman 2013). Rendezvous sites may be in snow caves, under boulders, hollows in trees and logs, in abandoned beaver lodges and bear dens, or anywhere that provides cover. When disturbed, female wolverines have been observed moving kits to an emergency location that appears less suitable than those normally used (Banci 1994).

Protective Status

Wolverines are a Candidate species for state listing as an endangered, threatened, or sensitive species in Washington (WDFW 2012b). The wolverine population in the contiguous U.S. was listed as a Candidate species for protection under the federal ESA in 2010. Federal listing of wolverines was deemed warranted but precluded due to other, higher priorities. The listing decision was based largely on the threat of range contraction due to climate change (USFWS 2010b). The proposal to list wolverines was withdrawn in 2014, after it was concluded that the factors impacting wolverines were not as significant as the earlier decision stated. The withdrawal was subsequently overturned, with a new assessment to determine eligibility for listing under the ESA (USFWS 2016i). Wolverine status under the ESA is currently Proposed Threatened. The NPS has designated wolverines as a Management Priority species within the Park.

Known Threats

Recreation

An increase in backcountry recreation, and the use of snowmobiles and helicopters to access remote areas for winter recreation, is increasing human activity in wolverine habitat outside of most protected areas. Winter recreation coincides with increased thermoregulatory demands of wolverines. During winter, reproductive females must meet additional energetic demands associated with reproduction; therefore, it is possible that winter recreational activities could exceed a disturbance threshold that limits reproductive success (Inman et al. 2007).

Inman (2013) estimated that around half of the highest quality wolverine habitat is privately owned, and could face development in the future, making large protected areas like the Park and its

surrounding National Forest wilderness areas important wolverine refugia. Wolverines are dependent on dispersal across expansive distances, and increasing human activity could limit potential dispersal and gene flow between populations (Inman 2013). Recreation activity such as cross-country skiing and snowmobile use in unprotected areas of southern British Columbia was negatively associated with wolverine occupancy and density as compared to parks (Kortello et al. 2019).

Climate Change

Climate change represents a threat to wolverine habitat and potential connectivity across its range due to wolverines' obligate association with persistent spring snowpack, with particular vulnerability in southern populations where snowpack is already being impacted by climate change (Copeland et al. 2010).

Suitable habitat in the transboundary area of Washington and British Columbia has relatively high connectivity with wolverine habitat in the Cascade Range, but is fragmented by human development and roads, as well as by natural dispersal barriers in the form of the low elevation areas like the Okanagan and Fraser River Valleys (Krosby et al. 2016). Climate-driven range shifts may necessitate more movement by wolverines, and climate change may alter the suitability of existing dispersal corridors for wolverine movement.

Krosby et al. (2016) used conceptual models of wolverine habitat to identify potential climate-driven impacts on wolverine habitat and connectivity. The UKMO-HadCM3 model (hotter, drier summers) they employed showed that large areas of current wolverine habitat would become unsuitable by the year 2080. The climate model also estimated a shrinking snowpack and an upward expansion of treeline and alpine vegetation that would increase suitable habitat at the highest elevations but would result in smaller and more isolated patches of core alpine habitat. All extant habitats in the Rocky Mountains and North Cascades are projected to become unsuitable under this model, and the Okanagan region is projected to experience declines in wolverine habitat, with some remnant fragmented habitat at higher elevations. The projections for the CGCM3.1(T47) model (warm wet spring, summer, and fall) projected a less dramatic range contraction, and retained habitat suitability in North Cascades and coast range, but a loss of all but the highest elevation habitat in the Rocky Mountains. The variation between these models highlights the uncertainty in predicting the impacts of climate change on species distribution (Krosby et al. 2016). Multiple studies have identified high elevation sites as important refugia for wolverine populations as the climate changes and snowpack contracts upward (Copeland et al. 2010, Inman 2013, Krosby et al. 2016).

Diminished extent and duration of snowpack could increase road building and human access in some high elevation areas, resulting in potential negative impacts to wolverine habitat and connectivity. Wolverines avoid areas of human use, and therefore potential climate-driven increases in recreational activities in high elevation areas may adversely impact habitat suitability (Krosby et al. 2016).

Projected increases in summer drought may exacerbate the risk of wildfires and impact patterns in forest pathogens (e.g., mountain pine beetle). Both fire and forest pathogens can aid in the creation of snags and other woody debris structures utilized by wolverines, but at severe levels can be detrimental to habitat quality (Krosby et al. 2016).

Lowered snowpack due to climate change may impact wolverine populations not only by reducing den site availability, but also increasing decay in food caches, potentially increasing competition for food sources as other species shift their ranges to overlap with the wolverine's, and potential impacts on summer foraging grounds, both in alteration of summer forage, and the potential for thermal stress on wolverines in hotter summer temperatures (Inman 2013).

Conservation and Research Needs

The greatest conservation need for wolverines in the North Cascades is identification, development, and protection of habitat connectivity corridors that include modeled changes in future habitat with climate (Krosby et al. 2016). Wolverines require such large expanses of habitat to meet their biological needs that no single protected area, such as the Park, is sufficient to conserve the population. Dynamic projections of habitat and corridors are needed to inform adaptive management measures that can then be implemented through collaborative conservation across the ecosystem.

From a research perspective, better understanding of wolverine physiology and ecology and effects of temperature are important toward understanding wolverine resilience in changing climates (Copeland et al. 2010, Inman 2013). Physiological impacts of climate may include the ability of females to help kits thermoregulate under varying amounts of snow cover, and effects of increased competition (Inman 2013). Timing of summer foods and ability of winter-cached foods to persist are also important to understand through time.

Continued population monitoring is critical for this species and should include genetic monitoring through genotyping of individuals and monitoring of population heterozygosity. While rarely detected in day-to-day Park operations, management actions have the potential to impact wolverines. Routine management, facilities, and infrastructure operations conducted in the Park should adhere to noise and disturbance guidelines in possible den habitat to help mitigate local impacts (Appendix A). Compliance with backcountry and frontcountry food storage regulations is imperative as wolverines can and will obtain anthropogenic food resources when not secured.

Chapter 5. Other NPS Management Priority Species

This chapter provides information on additional at-risk species that have been identified as NPS Management Priority species, but for which Park data and regional assessments may be limited. The nature and extent of threats to some of these species are also not well-defined or could be rapidly changing. None of the species in this chapter are currently listed by the state of Washington or under the federal ESA; however, northern bog lemming (*Synaptomys borealis sphagnicola*) is under review for potential ESA listing (USFWS 2015).

Western Grebe (*Aechmophorus occidentalis*) and Clark's Grebe (*Aechmophorus clarkii*)

Western Grebe (*Aechmophorus occidentalis*) and Clark's Grebe (*Aechmophorus clarkii*) are large piscivorous diving birds, and are characterized by their long thin beaks, red eyes, long neck, and black/dark gray and white plumage (Figure 45). These two species were taxonomically a single species until 1985, when they were split into distinct species. These species are migratory in the Park and typically are observed on Ross Lake and Lake Chelan in late autumn and early winter. Western grebes winter in the Salish Sea and in some large lakes in Washington; whereas Clark's grebes are rarely observed in the Park and typically winter at more southern latitudes.



Figure 45. Western grebe voucher record NOCA24271, photographed by Jason Ransom/NPS, November 6, 2017, on Ross Lake in the Ross Lake National Recreation Area, WA.

Large numbers of western grebes (flocks of 50 to >100) were observed on Ross Lake in the mid-1980s, and a single flock of 50 was observed in 1995. Since then, only 16 observations have been recorded in the Park and most were of single birds. However, a group of 30 western grebes was observed on Ross Lake in 2016. Only two confirmed records of Clark's grebe have been documented

in the Park (1995 on Lake Chelan, and 2017 on Ross Lake), though it is likely other Clark's grebes have been present and were not identified to species.

Western grebe and Clark's grebe are both Candidate species for listing by the State of Washington and both are identified as Species of Greatest Conservation Need under the State Wildlife Action Plan (WDFW 2015). Both species are listed in the Park as Management Priority species. Some of the greatest threats to these species occur in their breeding habitat outside of the Park and include damage to nesting colonies from recreational boating, as well as water fluctuations in reservoirs. Climate change effects (e.g., declines in water pH) and environmental contaminants that affect small fish and invertebrate prey species are thought to be major contributors to population declines in these species. While noise and disturbance guidelines must be adhered to in the Park (Appendix A), nest disturbance is unlikely due to summer distribution of these species. However, small watercraft operators on the Skagit reservoirs and Lake Chelan should be watchful for these birds especially during March–April and October–December: boaters should reduce speed and wake when grebes are observed.

Common loon (*Gavia immer*)

Common loon (*Gavia immer*) is a large-bodied piscivorous diving bird whose breeding plumage consists of a black head and black back with white spots (Figure 46). Most common loons in Washington are observed in the winter when their plumage turns to gray and white. While most common loons in the Park are migratory, one breeding location at Hozomeen Lake has been closely monitored since 1971, and other non-breeding adults persist in small numbers on the reservoirs year round. From 1995–2020, adult loons have been present at Hozomeen Lake almost every breeding season and in most years exhibit nesting behavior, but successful breeding has only been documented twice. Hozomeen Lake trail and all area within ½ mile of Hozomeen Lake is closed to visitor access from April 1 to May 31 each year to protect this sensitive site (NPS 2020). In 2019 and 2020, adult loons were also observed at nearby Willow Lake during the breeding season, but no evidence of nesting was detected. Common loons have been reported 100 times in the Park from 1995–2020, most often on Ross Lake.



Figure 46. Common loon voucher record NOCA24179, photographed by Graham Montgomery/Institute for Bird Populations, May 21, 2016, Lake Chelan National Recreation Area, WA.

Common loons are classified as a Sensitive species by the State of Washington and are a Species of Greatest Conservation Need under the State Wildlife Action Plan (WDFW 2015). Common loons are also listed in the Park as a Management Priority species. Threats to this species include degradation of shorelines from development and agriculture, recreational impacts of boating and fishing (primarily use of lead fishing tackle), fluctuations of water levels on reservoirs that can negatively impact nesting, and climate change impacts to small fish and invertebrate populations (Richardson et al. 2000). Environmental toxins and contaminants pose a bioaccumulation threat for loons given their high trophic position in the food web. Liver tissue from an adult common loon recovered in 2019 on Ross Lake contained manganese (0.79 ppm), iron (130 ppm), mercury (1.9 ppm), zinc (27 ppm), copper (7.4 ppm), and methyl mercury (1200 ppb), although no microplastics were detected (Christophersen and Ransom 2022).

The only known breeding site in the Park is protected annually and noise and disturbance guidance is established (Appendix A). Additional monitoring is needed at Willow Lake annually. Of note, Hozomeen Lake hosts a population of non-native eastern brook trout (*Salvelinus fontinalis*) and Willow Lake hosts a population of Ross Lake rainbow trout (*Oncorhynchus mykiss*). These fish could be eradicated in the future under the Park's Mountain Lakes Fisheries Management Plan (NPS 2008). If eradicated, the food source for loons in these lakes would disappear: the potential effects on loon occupancy and nesting are unknown. The conflicting nature of non-native species eradication and at-risk species conservation will demand a carefully considered implementation and monitoring

plan by NPS. As with western grebes, small watercraft operators on Ross Lake and Lake Chelan should be watchful for loons year-round: boaters should reduce speed and wake when loons are close by.

Mountain Chickadee (*Poecile gambeli*) and Olive-sided Flycatcher (*Contopus cooperi*)

Mountain chickadees (*Poecile gambeli*) and olive-sided flycatchers (*Contopus cooperi*) are small passerines that inhabit boreal and conifer forests in western North America (Figure 47). Both species primarily forage for insects. Flycatchers hunt flying insects in mid-air by swooping from their perch to catch dragonflies, grasshoppers, bees, moths, and flies. During the breeding season, they typically inhabit forest edges or openings such as are found at recent fire burns or wetlands. Mountain chickadees are cavity nesters, often searching out hard-walled tree snags for their nests.



Figure 47. Mountain chickadee, photographed by Jason Ransom/NPS, February 11, 2018, near the eastern border of North Cascades National Park Service Complex, WA.

Mountain chickadee populations have experienced a range-wide decline of about 1% per year, for a cumulative decline of about 39% from 1966–2019 (USGS 2023). Olive-sided flycatcher populations have experienced a range-wide decline of about 79% from 1966–2019 (USGS 2023). Using 3,904 landbird point counts from surveys during 2005–2016, Ray et al. (2018) estimated mountain chickadee and olive-sided flycatchers were two of only four modeled species that were declining in the Park.

Neither mountain chickadee nor olive-sided flycatcher have special status at the state level and are not listed under the ESA. Like all songbirds in the U.S., these species are protected under the federal MBTA. Both species are designated as Management Priority species in the Park. The exact reasons

for decline of these species are unknown, but it is likely that they are vulnerable to changing climate and related factors that impact montane insect populations. Olive-sided flycatchers are also long-distance migrants and may be experiencing habitat loss and other stressors in their South American winter range. Conservation measures in the Park include preservation of tree snags for cavity nesting birds, and adherence to noise and disturbance limits during nesting season (Appendix A).

Osprey (*Pandion haliaetus*)

Osprey (*Pandion haliaetus*) are large piscivorous raptors that select for habitat along bodies of water where fish are readily available. They build large stick nests, and in the Park have established nest sites on Ross Lake and Lake Chelan (Figure 48). Osprey tend not to nest until at least three years of age, and live 20–25 years. This species is migratory and may travel from the Park as far as coastal California, Mexico, Central and South America during the winter.



Figure 48. Osprey voucher record NOCA24094, photographed by Jason Ransom/NPS, April 16, 2015, near a nest site in Lake Chelan National Recreation Area, WA.

Following the ban of DDT use in the United States in 1972, osprey have recovered remarkably, and abundance may now exceed historic levels. This species was never listed under the ESA and does not have special status under Washington state administrative code. However, like all raptors, osprey are federally-protected under the MBTA. The NPS has designated osprey as a Management Priority species in the Park.

Despite positive population trends, emerging issues continue to threaten piscivorous raptors, including microplastics accumulation through the trophic hierarchy (Eerkes-Medrano et al. 2015), and bioaccumulation of toxins such as methyl mercury, lead, and PCBs in lakes and reservoirs (Grove et al. 2009). Fluctuating water levels on managed reservoirs may also affect food availability during nesting season (Christophersen and Ransom 2022).

There are 13 known osprey nesting territories in the Park and occupancy during 1979–2020 averaged 73% (Christophersen and Ransom 2022). Nest success rate in the Park has been calculated at 33%, which is substantially lower than the 68% success rate reported in Idaho and 44% success rate reported in Yellowstone National Park; likewise, productivity in the Park averaged 0.5 young per occupied nest which is below the 1.3–1.8 normal range of productivity rates for the western U.S. (Christophersen and Ransom 2022).

Monitoring and research actions needed for osprey include monitoring of environmental toxins and contaminants through the food chain in reservoirs, better understanding of the effects of fluctuating water levels on osprey food availability and continued monitoring of nest occupancy and success. Noise and disturbance mitigations have been established by the Park during nesting season (Appendix A).

Woodpeckers

Eleven species of woodpeckers are known to inhabit the Park, including four species with Management Priority designation by NPS: American three-toed woodpecker (*Picoides dorsalis*), Black-backed woodpecker (*Picoides arcticus*), Pileated woodpecker (*Dryocopus pileatus*), and White-headed woodpecker (*Dryobates albolarvatus*) (Figure 49). Black-backed woodpeckers are listed as a Priority Species under the Washington State Priority Habitat and Species Program, and as a Candidate for state listing. White-headed woodpeckers are listed as Species of Greatest Conservation Need in the State Wildlife Action Plan, and as a Candidate for state listing (WDFW 2015).



Figure 49. White-headed woodpecker (*Dryobates albolarvatus*) voucher record NOCA24124, photographed by Mandy Holmgren/Institute for Bird Populations, May 12, 2015, near Stehekin, Lake Chelan National Recreation Area, WA.

Woodpeckers have been surveyed in the Park as part of the annual breeding season landbird survey efforts since 2005 (Ray et al. 2018). Of the four Management Priority woodpecker species, only Pileated woodpecker is relatively common in the Park and was detected 181 times during point counts, 2005–2016 (Figure 50). White-headed woodpeckers have only been confirmed in the Park three times. Trend analyses for Pileated woodpeckers in the Park estimated the species as stable, though this was the rarest species for which Ray et al. (2018) attempted to fit a model to and the credible interval around the point estimate for trend was wide. Future estimates may help better refine the trend for this species.

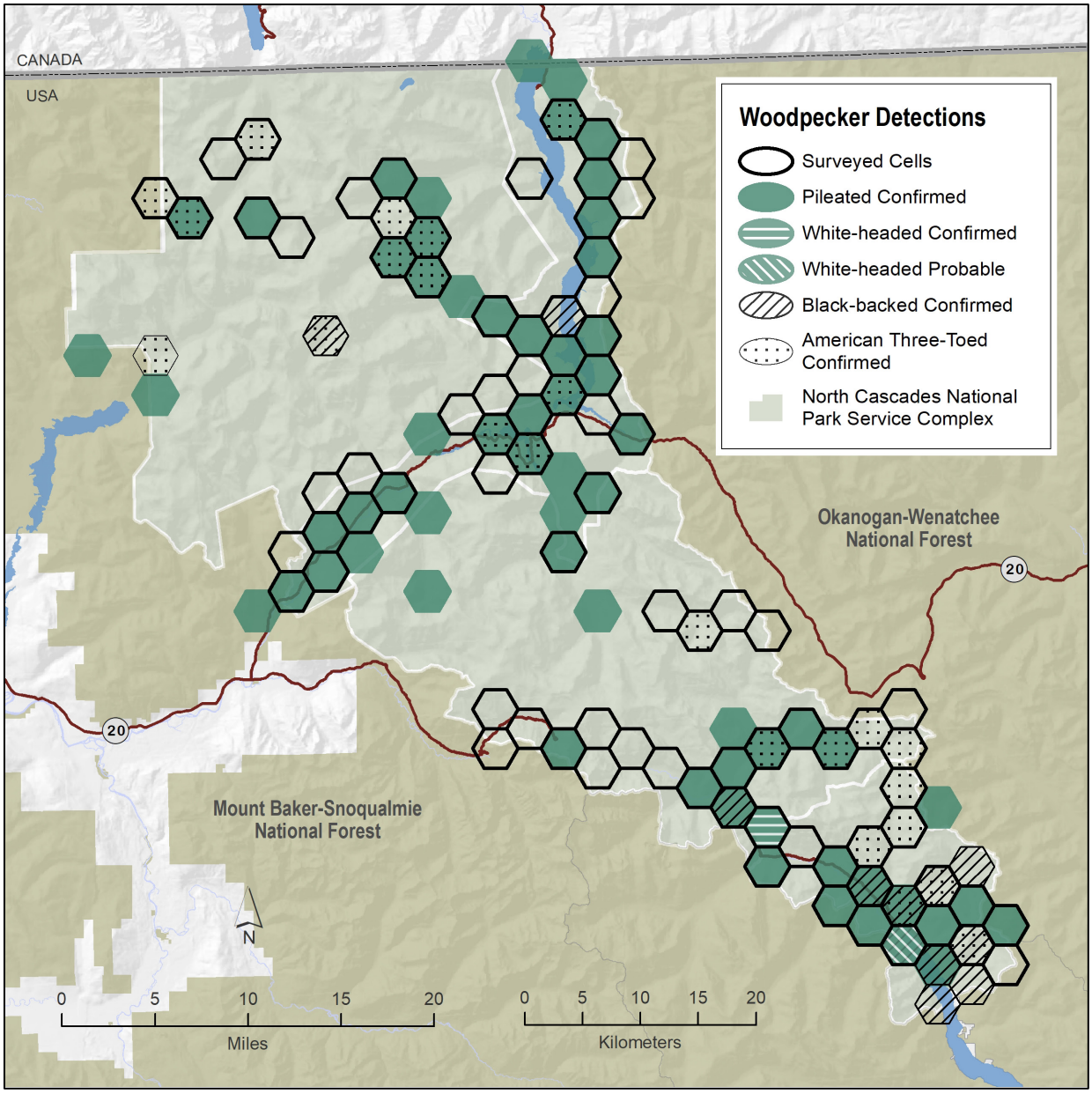


Figure 50. Survey locations, and Pileated woodpecker, White-headed woodpecker, Black-backed woodpecker, and American Three-toed woodpecker detections in North Cascades National Park Service Complex, 1995–2020. Hexagon survey and detection cells shown are 10 km^2 in size.

Most woodpecker species rely on dead and decaying trees to forage for insects and to build their cavity nests. Some species like White-headed woodpecker may be declining specifically due to loss of mature ponderosa pine forest, whereas others, such as Black-backed woodpeckers, rely on fire and large-scale blowdowns for optimal habitat. Major threats include a variety of fire management practices that suppress fire and reduce future habitat for woodpeckers, salvage logging without enough years for wood-boring beetles to become established as woodpecker food (typically up to five years), and reduction of snags in forests. Conservation actions in the Park should include preservation of tree snags whenever possible; notably, many woodpeckers in the Park appear to

select for free-standing limb-free trunks up to 6 m in height and 20–50 cm diameter at breast height (Figure 51). Noise and disturbance guidelines have been established for woodpeckers, during breeding season in the Park (Appendix A).



Figure 51. Example of an active woodpecker nesting snag in Lake Chelan National Recreation Area, WA.

Northern bog lemming (*Synaptomys borealis sphagnicola*)

The northern bog lemming is a small, relatively rare rodent that prefers lush wet meadows with moss or sedge understory. In the western part of its range, it occurs from the subarctic of Alaska and Yukon to northern Washington. The Park sits on the extreme southern edge of the range of this species, and like many climate-sensitive species, the southern edge of range is often the most at risk as warmer climate shifts north. Northern bog lemmings do not hibernate, and instead create tunnels under the snow to travel in winter. Climate change is expected to threaten this species due to their dependence on cold, moist, mossy habitats. These habitat qualities are vulnerable to drying under future climate scenarios that include the warmer temperatures and reduced precipitation.

This species is currently in review for federal listing under the ESA. Northern bog lemmings are identified by the State of Washington as a Species of Greatest Conservation Need under the State Wildlife Action Plan (WDFW 2015). The NPS has designated this species as a Management Priority

species. The only confirmed record of a northern bog lemming in the Park was a single animal that was caught in a snap trap on August 18, 1982, during a small mammal study near Waddell Lake [Park database reference number 1773; SYBO001]. The skull of this specimen is curated in the NOCA museum vertebrate collection (Accession No. 203), which is located in Marblemount, WA.

There have been no contemporary small mammal surveys in lemming habitat in the Park; however, one recent effort by the University of Washington's Burke Museum resulted in capture of a single northern bog lemming during a deployment of 400 box traps near Washington Pass (east of the Park boundary) (Leiser-Miller 2017). Surveys for northern bog lemming should be prioritized in the Park. Mitigations in place include pre-project surveys in lemming habitat and noise and disturbance limitations (Appendix A).

Western Spotted Skunk (*Spilogale gracilis*)

Western spotted skunk (*Spilogale gracilis*) is a small nocturnal skunk with a marbled black and white stripe pattern on their back (Figure 52). They prefer dense understory vegetation in low to mid elevations of the Cascades. Detections of western spotted skunk have declined to near zero across survey efforts in the Park (Figure 53, Table 22). Western spotted skunk was the most frequently detected mammal during 2003–2004 forest carnivore surveys (Christophersen et al. 2005), and then they were only detected once each year in 2014, 2016, and 2017. Concurrently, the first striped skunk record in the Park occurred in 2016 near Stehekin, and since then striped skunks have been detected 49 times in the western and southern portions of the Park, ranging from Hozomeen to Stehekin, as well as Big Beaver Creek drainage. The combined nature of decreasing western spotted skunk detections and increasing potential competition has led us to suspect that western spotted skunks are in decline. It is unknown how sensitive to climate change western spotted skunks are, but secondary impacts of climate change such as emerging diseases may play a role in spotted skunk population dynamics. The first amdoparvovirus specific to skunks was recently detected in striped skunks near Vancouver, British Columbia (Canuti et al. 2017), and it is unknown how that parvovirus may be affecting sympatric western spotted skunk populations.



Figure 52. Western spotted skunk, photographed on a remote trail camera (operated by University of Montana and the National Park Service), August 1, 2019, in the Mount Baker-Snoqualmie National Forest, WA.

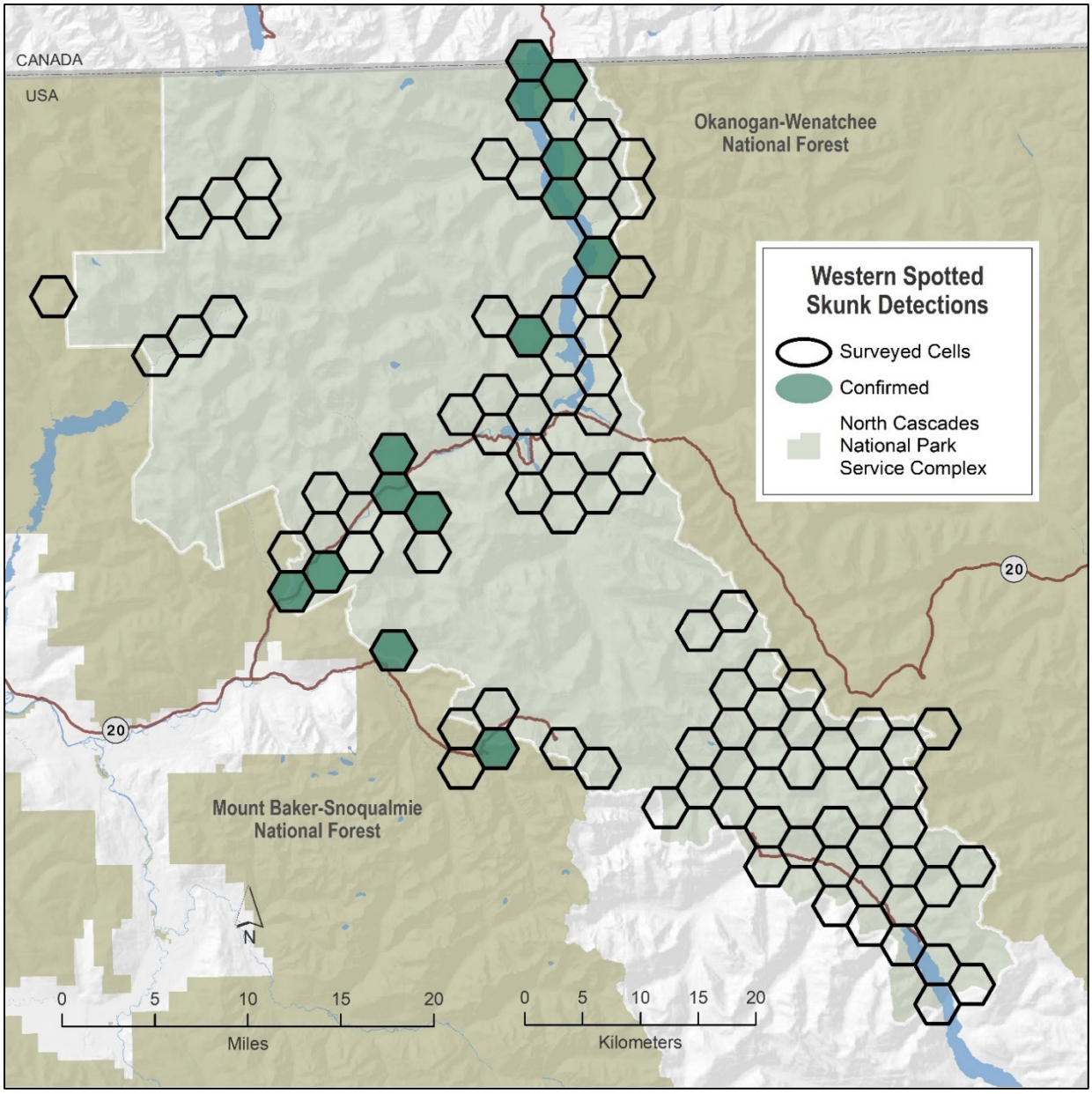


Figure 53. Survey locations, and western spotted skunk (*Spilogale gracilis*) detections in North Cascades National Park Service Complex, 1995–2020 (Table 22). Hexagon survey and detection cells shown are 10 km2 in size.

Table 22. Western spotted skunk (*Spilogale gracilis*) data sources for North Cascades National Park Service Complex, 1995–2020.

Survey Year Range	Type of Data	Interval	Study and Data Reference
1995–2020	NPS staff and visitor observations	Incidental	NPS unpublished data
2003–2005	Forest carnivore study trail cameras at 24–40 stations	24 h camera stations, February–August each year	Christophersen et al. 2005, Christophersen 2006
2011–2020	Gray wolf monitoring trail cameras at 8–25 stations	24 h camera stations, October–May each year	NPS unpublished data
2016–2017	Canada lynx study trail cameras at 40 stations	24 h camera stations, June–October each year	King 2019, King et al. 2020a, 2020b
2018–2020	Fisher study trail cameras at 56 stations	24 h camera stations, all months	Humphries 2022
2020	Black bear study trail cameras at 109 stations	24 h camera stations, July–September	J. Manning and NPS unpublished data

Very little is known about western spotted skunk biology or ecology in the Park. Research and monitoring are needed to better understand presence, distribution, and trend for this species, understand the potentially changing dynamics of interspecific competition, presence and impacts of emerging diseases, and vulnerability to climate change. The NPS has noise and disturbance guidelines in place for this species, noting that communal denning has been documented and could represent substantive impacts should a den be disturbed (Appendix A).

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Appendix A. Sensitive time periods for NPS Management Priority species in North Cascades National Park Service Complex

Table A1. List of select NPS Management Priority species with sensitive time periods around nesting and denning behavior, general management practices aimed at mitigating disturbance, and NPS survey protocols in practice. Required actions for federally-listed species may vary on a project-to-project basis, following Endangered Species Act consultation with U.S. Fish and Wildlife Service. Mitigations for routine NPS operations associated with trail maintenance are defined in USFWS (2022).

Species	Present in Park	High Sensitivity Period (Critical, Early Nesting/ Denning)	Moderate Sensitivity Period (Late Nesting/ Denning)	Entire Nesting/ Denning Period	Minimum Noise and Disturbance Buffer (Nests/Dens)	NPS Monitoring and Management
Bald Eagle	All months; high abundance on Skagit River Nov–Feb	Jan 1–Jun 30	Jul 1–Aug 31	Jan 1–Aug 31	1/4 mi not in line of sight; 1/2 mi within line of sight; 1 mi for blasting/pile driving (Jan 1–Aug 31, 24 hrs). Same for communal roost site Nov 15–Mar 15 (1 hr before sunset to 1 hr after sunrise)	3-visit nest survey protocol (May 1–15; Jun 1–15; Jul 1–Aug 1). Winter survey driving routes every 5 years. Pre-project site visits Jan 1–Aug 31.
Common Loon	All months; increased abundance on reservoirs during migration	Apr 1–May 31	Jun 1–Aug 15	Apr 1–Aug 15	1/4 mi from a nest. Trail closure in place Apr 1–May 31 within 1/2 mi radius Hozomeen Lake	2-visit survey protocol (nesting, presence of young; May 15–Aug 15).
Golden Eagle	All months; increased abundance during migration Aug–Sep around mountain passes	Mar 1–Jun 30	Jul 1–Aug 15	Mar 1–Aug 15	1/4 mile not in line of sight; 1/2 mi within line of sight; 1 mi for blasting/pile driving during entire nesting season	No confirmed nests in Park; High migratory numbers around Sibley Pass and Mt Shuksan. [Note: increased aviation hazard Aug–Sep]

Table A1 (continued). List of select NPS Management Priority species with sensitive time periods around nesting and denning behavior, general management practices aimed at mitigating disturbance, and NPS survey protocols in practice. Required actions for federally-listed species may vary on a project-to-project basis, following Endangered Species Act consultation with U.S. Fish and Wildlife Service. Mitigations for routine NPS operations associated with trail maintenance are defined in USFWS (2022).

Species	Present in Park	High Sensitivity Period (Critical, Early Nesting/ Denning)	Moderate Sensitivity Period (Late Nesting/ Denning)	Entire Nesting/ Denning Period	Minimum Noise and Disturbance Buffer (Nests/Dens)	NPS Monitoring and Management
Harlequin Ducks	Mar–Sep	Apr 15–Jun 30	Jul 1–Aug 15	Apr 15–Aug 15	100 m from known nest Apr 15–Jun 30	Nest surveys on suitable streams. Occupancy survey Apr–May, followed by productivity surveys Jul–Aug. Pre-project site visits Apr 15–Jun 30.
Landbirds (Clark's Nutcracker, Passerines, Woodpeckers)	Some species all months; Migratory species Mar–Sep	Mar 15–Sep 1	Mar 15–Sep 1	Mar 15–Sep 1	Varies by project type: physical disturbance activities (brushing, burning, clearing, pesticides) are generally limited to non-nesting season	Some species nest multiple times in a season, extending the critical nesting period the entire season. See landbirds survey protocol (Siegel et al. 2007). All hazard trees must be inspected for active nest cavities Mar 15–Sep 1. Pre-project site surveys required.

Table A1 (continued). List of select NPS Management Priority species with sensitive time periods around nesting and denning behavior, general management practices aimed at mitigating disturbance, and NPS survey protocols in practice. Required actions for federally-listed species may vary on a project-to-project basis, following Endangered Species Act consultation with U.S. Fish and Wildlife Service. Mitigations for routine NPS operations associated with trail maintenance are defined in USFWS (2022).

Species	Present in Park	High Sensitivity Period (Critical, Early Nesting/ Denning)	Moderate Sensitivity Period (Late Nesting/ Denning)	Entire Nesting/ Denning Period	Minimum Noise and Disturbance Buffer (Nests/Dens)	NPS Monitoring and Management
Marbled Murrelet	Apr–Sep	Apr 1–Sep 4	Apr 1–Sep 4	Apr 1–Sep 4	1/4 mi for typical noise, 1/2 mi for helicopter, 1 mi for blasting/elevated noise during entire nesting season. Artificial light mitigations needed in Skagit flight corridor and appropriate habitat. No aviation in habitat within 2 hours of sunrise and sunset.	Pre-project site visits Apr 1–Sep 4. Hazard trees ≥18 in diameter at breast height require inspection. Note that habitat is considered occupied regardless of detection. Radar survey, followed audio-visual surveys required to confirm presence.
Mount Rainier White-tailed Ptarmigan	All months; subalpine and alpine areas only	May 1–Aug 15	Aug 16–Oct 15	May 1–Oct 15	100 m from nest; 1/4 mi from nest for helicopter; 1 mi for blasting/elevated noise during entire nesting season. Note: chicks with limited mobility are present on ridge crests Sep–Oct.	Pre-project site visits May 1–Oct 15. Surveys conducted as alpine access allows, using audio broadcast along transects, and scat presence on snow fields. Helicopter landing mitigations must be in place (incl. Copper Ridge and Sourdough fire lookouts)

Table A1 (continued). List of select NPS Management Priority species with sensitive time periods around nesting and denning behavior, general management practices aimed at mitigating disturbance, and NPS survey protocols in practice. Required actions for federally-listed species may vary on a project-to-project basis, following Endangered Species Act consultation with U.S. Fish and Wildlife Service. Mitigations for routine NPS operations associated with trail maintenance are defined in USFWS (2022).

Species	Present in Park	High Sensitivity Period (Critical, Early Nesting/ Denning)	Moderate Sensitivity Period (Late Nesting/ Denning)	Entire Nesting/ Denning Period	Minimum Noise and Disturbance Buffer (Nests/Dens)	NPS Monitoring and Management
Northern Goshawk	All months	Mar 1–Jun 30	Jul 1–Aug 31	Mar 1–Aug 31	1/4 mi for typical noise, 1/2 mi for helicopter, 1 mi for blasting/elevated noise during entire nesting season.	Pre-project site visits Mar 1–Aug 1. Nesting survey requires audio callback protocol for presence, and follow-up visit for presence of young. Note: Strong nest defense behavior requires PPE use for site visits (hard hat, eye protection, long sleeves).
Northern Spotted Owl	All months	Mar 1–Jul 15	Jul 16–Sep 30	Mar 1–Sep 30	1/4 mi for typical noise and helicopter; 1 mi for blasting/elevated noise during entire nesting season	3-visit survey protocol (occupied, nesting, presence of young) for hoot surveys, or continuous monitoring using acoustic recording units. ≥ 18" DBH hazard tree requires inspection. Note habitat is considered occupied regardless of detection.
Osprey	Apr–Oct	Apr 1–Jul 15	Jul 1–Aug 15	Apr 1–Aug 15	1/4 mile not in line of sight; 1/2 mi within line of sight; 1 mi for blasting/pile driving, Apr 15–Aug 15	3-visit survey protocol (occupied, nesting, presence of young). Flight mitigations in place around nest structures.

Table A1 (continued). List of select NPS Management Priority species with sensitive time periods around nesting and denning behavior, general management practices aimed at mitigating disturbance, and NPS survey protocols in practice. Required actions for federally-listed species may vary on a project-to-project basis, following Endangered Species Act consultation with U.S. Fish and Wildlife Service. Mitigations for routine NPS operations associated with trail maintenance are defined in USFWS (2022).

Species	Present in Park	High Sensitivity Period (Critical, Early Nesting/ Denning)	Moderate Sensitivity Period (Late Nesting/ Denning)	Entire Nesting/ Denning Period	Minimum Noise and Disturbance Buffer (Nests/Dens)	NPS Monitoring and Management
Peregrine Falcon	All months, but increased abundance Mar–Oct	Mar 1–Jul 15	Jul 16–Sep 1	Mar 1–Sep 1	1/4 mile not in line of sight; 1/2 mi within line of sight; 1 mi for blasting/pile driving Mar 1–Jul 15. Site-specific climbing disturbance at Newhalem set as 200 m within line of sight	3 visit survey protocol minimum (presence, nesting, presence of young). Flight mitigations in place around nest sites. Climbing closures in place pending nest monitoring outcomes.
American Pika	All months	Year round den occupancy	Year round den occupancy	Year round den occupancy	25 m radius from active hay piles and occupied talus patch.	Present year round and do not hibernate. Pre-project site surveys required.
Bats	Some species all months; Migratory species Mar–Oct	Apr 1–Jul 15	Jul 16–Oct 31	Apr 1–Oct 31	Seasonally time sensitive by activity (roofing, bat exclusion mitigations)	WNS surveys Apr 1–May 15; occupancy surveys entire year (visual and acoustic)
Canada Lynx	All months	May 1–Jul 31	May 1–Jul 31	May 1–Jul 31	1/2 mi of known active den during entire denning season. Note: Designated Critical Habitat is in place in Park	Unscented camera trap surveys on trails above 5000 ft. Den location may change and is often undeterminable.
Cascade Red Fox	All months	Feb 1–Sep 1	Feb 1–Sep 1	Feb 1–Sep 1	1/2 mi of known active den during entire denning season.	winter trail camera surveys (multi-species protocol)

Table A1 (continued). List of select NPS Management Priority species with sensitive time periods around nesting and denning behavior, general management practices aimed at mitigating disturbance, and NPS survey protocols in practice. Required actions for federally-listed species may vary on a project-to-project basis, following Endangered Species Act consultation with U.S. Fish and Wildlife Service. Mitigations for routine NPS operations associated with trail maintenance are defined in USFWS (2022).

Species	Present in Park	High Sensitivity Period (Critical, Early Nesting/ Denning)	Moderate Sensitivity Period (Late Nesting/ Denning)	Entire Nesting/ Denning Period	Minimum Noise and Disturbance Buffer (Nests/Dens)	NPS Monitoring and Management
Fisher	All months	Feb 1–Apr 30	Feb 1–Apr 30	Feb 1–Aug 30	1/2 mi of den Feb 1–Jun 30	Hazard trees ≥ 18" DBH, with cavity openings 3–8" diameter need assessment. Wildlife trail camera surveys (interagency protocol in place)
Gray Wolf	All months	Mar 15–May 31 (den)	Jun 1–Jul 15 (rendezvous site)	Mar 15–Jul 15	1 mi of known active den Mar 15– Jul 15; 1/4 mi any rendezvous site May 15–Jul 15	winter trail camera surveys on and off trail
Grizzly Bear	All months: active Apr–Nov (in torpor during winter)	Oct 1–May 31	Oct 1–May 31	Oct 1–May 31	1 mi of active den site Oct 1–May 31. Suspend noisy work within 1 mi of confirmed sighting for 21 days.	Trail camera surveys in summer (multi-species protocol); genetic hair snag stations
Hoary Marmot	All months: active Mar–Oct (in torpor during winter)	Year round den occupancy	Year round den occupancy	Year round den occupancy	Compliance survey before work that may disturbs habitat	Point count visual/audio surveys at colony sites. Pre-project site surveys required.
Northern Bog Lemming	All months (active all months)	Year round den occupancy	Year round den occupancy	Year round den occupancy	1/4 mi from confirmed presence location; no ground disturbance	Genotyping scat or live capture surveys using box traps. Pre-project site surveys required.
Western Gray Squirrel	All months, Lake Chelan NRA only	Jan 1–Apr 14	Apr 15–Aug 31	Jan 1–Aug 31	100 m from nest	Visual surveys for nest structures. Pre-project site surveys required.

Table A1 (continued). List of select NPS Management Priority species with sensitive time periods around nesting and denning behavior, general management practices aimed at mitigating disturbance, and NPS survey protocols in practice. Required actions for federally-listed species may vary on a project-to-project basis, following Endangered Species Act consultation with U.S. Fish and Wildlife Service. Mitigations for routine NPS operations associated with trail maintenance are defined in USFWS (2022).

Species	Present in Park	High Sensitivity Period (Critical, Early Nesting/ Denning)	Moderate Sensitivity Period (Late Nesting/ Denning)	Entire Nesting/ Denning Period	Minimum Noise and Disturbance Buffer (Nests/Dens)	NPS Monitoring and Management
Western Spotted Skunk	All months	Feb 15–Aug 15 (Note: may den in groups ≤20 adults)	Feb 15–Aug 15 (Note: may den in groups ≤20 adults)	Feb 15–Aug 15	1/4 mi of known active den during entire denning season	Trail camera surveys (multi-species protocol)
Wolverine	All months	Oct 1–May 31	Oct 1–May 31	Oct 1–May 31	1/4 mi work and helicopter; 1 mi blasting from active den Oct 1–May 31	Trail cameras, genetic hair snag stations (Interagency survey protocol in place)

Appendix B. Invertebrate pollinators detected in North Cascades National Park Service Complex.

Lists of invertebrate pollinators detected in North Cascades National Park Service Complex, 1995–2020 are presented in Tables B1–B4.

Table B1. List of butterfly species detected in North Cascades National Park Service Complex. Observations from Butterflies and Moths of North America (BAMONA), [https://www.butterfliesandmoths.org/] are verified by subject matter experts, and all iNaturalist [https://www.inaturalist.org/] observations are "research grade".

Family	Species	Common Name	Presence	Evidence				
				Specimen	Photo	Survey	iNaturalist	BAMONA
Hesperiidae	*Erynnis propertius*	Propertius Duskywing	Confirmed	–	–	–	X	–
	Erynnis persius	Persius Duskywing	Confirmed	–	–	–	X	–
	Carterocephalus palaemon	Arctic Skipper	Confirmed	–	–	X	–	–
	Hesperia juba	Juba Skipper	Confirmed	–	–	–	X	–
	Hesperia comma	Common Branded Skipper	Confirmed	–	X	X	–	X
	Ochlodes sylvanoides	Woodland Skipper	Confirmed	X	X	X	–	–
Papilionidae	*Parnassius clodius*	Clodius Parnassian	Confirmed	X	X	X	X	X
	Parnassius smintheus	Mountain Parnassian	Confirmed	X	X	X	–	X
	Papilio zelicaon	Anise Swallowtail	Confirmed	–	–	X	X	–
	Papilio indra	Indra Swallowtail	Probable	–	–	–	–	–
	Papilio rutulus	Western Tiger Swallowtail	Confirmed	X	–	–	X	–
	Papilio eurymedon	Pale Swallowtail	Confirmed	–	–	X	X	–
Pieridae	*Colias eurytheme*	Orange Sulphur	Confirmed	–	–	X	–	–
	Anthocharis julia	Julia's Orangetip	Confirmed	–	–	X	–	–
	Neophasia menapia	Pine White	Confirmed	X	–	X	X	–
	Pieris marginalis	Margined White	Confirmed	X	–	X	–	–
	Pieris rapae	Cabbage White	Confirmed [1]	X	–	–	–	–
	Pontia occidentalis	Western White	Confirmed	–	–	X	X	–

[1] Introduced, non-native.

Table B1 (continued). List of butterfly species detected in North Cascades National Park Service Complex. Observations from Butterflies and Moths of North America (BAMONA), [https://www.butterfliesandmoths.org/] are verified by subject matter experts, and all iNaturalist [https://www.inaturalist.org/] observations are "research grade".

Family	Species	Common Name	Presence	Evidence				
				Specimen	Photo	Survey	iNaturalist	BAMONA
Lycaenidae	*Lycaena heteronea*	Blue Copper	Confirmed	–	–	X	X	–
	Lycaena helloides	Purplish Copper	Confirmed	X	–	X	X	X
	Lycaena mariposa	Mariposa Copper	Confirmed	X	X	X	X	X
	Callophrys gryneus	Cedar Hairstreak	Confirmed	–	X	–	X	–
	Callophrys augustinus	Brown Elfin	Probable	–	–	–	–	–
	Callophrys mossii	Moss' Elfin	Confirmed	–	–	–	X	–
	Strymon melinus	Gray Hairstreak	Confirmed	–	–	–	X	–
	Celastrina echo	Echo Blue	Confirmed	–	–	–	X	–
	Glaucopsyche lygdamus	Silvery Blue	Confirmed	–	X	X	X	–
	Plebejus icarioides	Boisduval's (Common) Blue	Confirmed	–	X	X	X	X
	Plebejus lupini / acmon	Lupine Blue/ Acmon	Confirmed	–	–	X	–	–
	Plebejus anna	Anna's Blue	Confirmed	X	X	X	X	X
	Agriades glandon	Arctic Blue	Confirmed	–	X	X	X	–
Nymphalidae	*Boloria epithore*	Western Meadow Fritillary	Confirmed	–	–	X	X	–
	Boloria chariclea	Arctic Fritillary	Confirmed	–	X	X	X	X
	Speyeria hydaspe	Hydaspe Fritillary	Confirmed	X	–	X	X	–
	Speyeria mormonia	Mormon Fritillary	Confirmed	X	X	X	X	X
	Limenitis lorquini	Lorquin's Admiral	Confirmed	–	X	X	X	–
	Vanessa cardui	Painted Lady	Confirmed	–	–	X	X	–
	Vanessa atalanta	Red Admiral	Confirmed	–	–	X	–	–
	Aglais milberti	Milbert's Tortoiseshell	Confirmed	–	X	X	X	X
	Nymphalis californica	California Tortoiseshell	Confirmed	–	–	X	X	–
	Nymphalis antiopa	Mourning Cloak	Confirmed	–	–	–	X	–

[1] Introduced, non-native.

Table B1 (continued). List of butterfly species detected in North Cascades National Park Service Complex. Observations from Butterflies and Moths of North America (BAMONA), [https://www.butterfliesandmoths.org/] are verified by subject matter experts, and all iNaturalist [https://www.inaturalist.org/] observations are "research grade".

Family	Species	Common Name	Presence	Evidence				
				Specimen	Photo	Survey	iNaturalist	BAMONA
Nymphalidae (continued)	*Polygonia satyrus*	Satyr Comma	Confirmed	–	–	X	X	–
	Polygonia gracilis	Hoary Comma	Confirmed	–	X	X	–	X
	Polygonia faunus	Green Comma	Confirmed	–	X	X	X	–
	Euphydryas editha	Edith's Checkerspot	Confirmed	–	X	X	–	X
	Euphydryas anicia	Anicia Checkerspot	Confirmed	X	X	X	–	–
	Chlosyne hoffmanni	Hoffman's Checkerspot	Confirmed	–	–	–	X	–
	Chlosyne palla	Northern Checkerspot	Probable	–	–	–	–	–
	Phyciodes mylitta	Mylitta Crescent	Probable	–	–	–	–	–
	Phyciodes pulchella	Field Crescent	Probable	–	–	–	–	–
	Erebia vidleri	Vidler's Alpine	Confirmed	–	–	X	–	X
	Erebia epipsodea	Common/ Butler's Alpine	Confirmed	–	–	X	–	–
	Oeneis chryxus	Chryxus Arctic	Confirmed	–	X	X	–	X
	Oeneis nevadensis	Great Arctic	Confirmed	–	X	–	X	–

[1] Introduced, non-native.

Table B2. List of moth species detected in North Cascades National Park Service Complex. BioBlitz 2014 data were collected by D. Strenge and D. Dropper, and BioBlitz 2016 data were collected by J. Strange.

Family (Subfamily)	Species	Common Name	Presence	Evidence			
				Specimen	Photo	BioBlitz 2014	BioBlitz 2016
Adelidae	*Adela septentrionella*	Fairy Longhorn Moth	Confirmed	–	–	–	X
Cossidae	*Acossus populi*	–	Confirmed	X	X	X	–
Crambidae	*Agriphila straminella*	–	Confirmed	–	X	X	–
	Catoptria latiradiellus	Three-spotted Crambid Moth	Confirmed	–	X	X	–
	Catoptria oregonica	Oregon Catoptria	Confirmed	–	X	X	–
	Crambus unistriatellus	Wide-stripe Grass-veneer Moth	Confirmed	–	X	X	–
Drepanidae (Drepanine)	*Drepana arcuata*	–	Confirmed	X	X	X	–
	Drepana billineata	–	Confirmed	X	X	X	–
Drepanidae (Thatirinae)	*Xsyne scripta*	–	Confirmed	X	X	X	–
	Clemensia albata	–	Confirmed	X	X	X	–
	Lophocampa argentata	–	Confirmed	X	X	X	–
	Lophocampa maculata	–	Confirmed	X	X	X	–
	Platarctia parthenos	–	Confirmed	X	X	X	–
Eribidae (Eribinae)	*Drasteria sabulosa*	–	Confirmed	X	X	X	–
Eribidae (Hypeninae)	*Hypena humuli*	–	Confirmed	–	X	X	–
	Hypena palparia	–	Confirmed	X	X	X	–
Eribidae (Lymantriinae)	*Dasychira grisefacta*	–	Confirmed	X	X	X	–
Geometridae	*Biston betularia*	–	Confirmed	X	–	X	–
	Campaea perlata	–	Confirmed	X	X	X	–
	Caripeta divisata	–	Confirmed	X	X	X	–
	Ceratdalia guenata	–	Confirmed	X	–	X	–
	Chlorosea banksaria	Bank's Emerald Moth	Confirmed	–	X	X	–
	Coryphista meadi	–	Confirmed	X	–	X	–

Table B2 (continued). List of moth species detected in North Cascades National Park Service Complex. BioBlitz 2014 data were collected by D. Strenge and D. Dropper, and BioBlitz 2016 data were collected by J. Strange.

Family (Subfamily)	Species	Common Name	Presence	Evidence			
				Specimen	Photo	BioBlitz 2014	BioBlitz 2016
Geometridae (continued)	*Cyclophora pendulinaria*	–	Confirmed	X	X	X	–
	Dysstroma citrata	–	Confirmed	X	X	X	–
	Enypia packardata	–	Confirmed	X	–	X	–
	Epirrhoe alternata	–	Confirmed	X	–	X	–
	Euchlaena johnsonaria	–	Confirmed	X	–	X	–
	Euchlaena tigrinaria	–	Confirmed	X	X	X	–
	Eupithecia cretaceata	–	Confirmed	X	–	X	–
	Eupitheica longipalpata	–	Confirmed	–	X	X	–
	Eupithecia tripunctaria	–	Confirmed	X	–	X	–
	Eulithis xylina	–	Confirmed	X	–	X	–
	Eustroma fasciata	–	Confirmed	X	–	X	–
	Hesperumia latipennis	–	Confirmed	–	X	X	–
	Hesperumia sulphuraria	–	Confirmed	–	X	X	–
	Hydriomena furcata	–	Confirmed	–	X	X	–
	Hydriomena porcelaria	–	Confirmed	X	X	X	–
	Hydriomena renunciata	–	Confirmed	X	X	X	–
	Hypagyris unipunctata	–	Confirmed	X	–	X	–
	Idaea dimidiata	–	Confirmed	X	–	X	–
	Iridposis emasculata	–	Confirmed	X	X	X	–
	Macaria signaria	–	Confirmed	X	X	X	–
	Macaria ulsterata	–	Confirmed	X	–	X	–
	Malacosoma californicum	–	Confirmed	X	–	X	–
	Nematocampa resistaria	–	Confirmed	X	–	X	–

Table B2 (continued). List of moth species detected in North Cascades National Park Service Complex. BioBlitz 2014 data were collected by D. Strenge and D. Dropper, and BioBlitz 2016 data were collected by J. Strange.

Family (Subfamily)	Species	Common Name	Presence	Evidence			
				Specimen	Photo	BioBlitz 2014	BioBlitz 2016
Geometridae (continued)	*Perizoma grandis*	–	Confirmed	X	–	X	–
	Perizoma costiguttata	–	Confirmed	X	–	X	–
	Pero mizon	–	Confirmed	X	–	X	–
	Perro miiiorrisonaria	–	Confirmed	X	–	X	–
	Phyllodesma americana	–	Confirmed	X	–	X	–
	Plagoda phlogosaria	–	Confirmed	X	–	X	–
	Plemyria georgii	–	Confirmed	X	–	X	–
	Prochoerodes forficaria	–	Confirmed	X	–	X	–
	Scopula junctaria	–	Confirmed	X	–	X	–
	Selenia alciphera	–	Confirmed	X	–	X	–
	Spargania magnoliata	Double-banded Carpet Moth	Confirmed	–	X	X	–
	Speranza plumosata	–	Confirmed	X	–	X	–
	Triphosa haesitata	Tissue Moth	Confirmed	–	X	X	–
	Venusia cambrica	–	Confirmed	X	–	X	–
	Xanthorhoe alticolata	–	Confirmed	–	X	X	–
Noctuidae (Acrotinae)	*Acronicta dactylina*	–	Confirmed	X	X	X	–
	Acronicta fragilis	–	Confirmed	X	–	X	–
	Acronicta impleta	–	Confirmed	X	X	X	–
	Acronicta impressa	–	Probable	–	–	–	–
	Acronicta innotata	–	Confirmed	X	X	X	–
	Acronicta vulpina	–	Confirmed	X	X	–	–
Noctuidae (Herminiinae)	*Bleptina caradinalis*	–	Confirmed	X	X	X	–
	Zanclognatha jacchusalis	–	Confirmed	X	X	X	–

Table B2 (continued). List of moth species detected in North Cascades National Park Service Complex. BioBlitz 2014 data were collected by D. Strenge and D. Dropper, and BioBlitz 2016 data were collected by J. Strange.

Family (Subfamily)	Species	Common Name	Presence	Evidence			
				Specimen	Photo	BioBlitz 2014	BioBlitz 2016
Noctuidae (Noctuinae)	*Agrotis vancouverensis*	–	Confirmed	X	X	X	–
	Anaplectoides prasina	–	Confirmed	X	X	X	–
	Anaplectoides pressus	–	Confirmed	X	X	X	–
	Apamea amputatrix	–	Confirmed	X	X	X	–
	Apamea cogitata	–	Confirmed	X	X	X	–
	Aseptis binotata	–	Confirmed	X	X	X	–
	Caradrina morpheus	–	Confirmed	X	X	X	–
	Diarsia esurialis	–	Confirmed	X	–	X	–
	Euplexia benesimilis	–	Confirmed	X	–	X	–
	Euxoa quebecensis	–	Confirmed	X	–	X	–
	Homorothodes communis	–	Confirmed	X	X	X	–
	Homorthodes furfata	–	Confirmed	X	X	X	–
	Homorthodes hanhami	–	Confirmed	X	X	X	–
	Hyppa brunneicrista	–	Confirmed	X	X	X	–
	Hyppa contrasta	–	Confirmed	X	X	X	–
	Lacinipolia comis	–	Confirmed	X	X	X	–
	Lateroiligia ophiogramma	–	Confirmed	X	X	X	–
	Leucania farcta	–	Confirmed	X	X	X	–
	Leucania oxygala	–	Confirmed	–	X	X	–
	Melanchra adjuncta	–	Confirmed	X	X	X	–
	Mythimna oxygala	Lesser Wainscot Moth	Confirmed	X	X	X	–
	Noctua pronuba	–	Confirmed	X	–	X	–
	Ochropluera implecta	–	Confirmed	X	–	X	–

Table B2 (continued). List of moth species detected in North Cascades National Park Service Complex. BioBlitz 2014 data were collected by D. Strenge and D. Dropper, and BioBlitz 2016 data were collected by J. Strange.

Family (Subfamily)	Species	Common Name	Presence	Evidence			
				Specimen	Photo	BioBlitz 2014	BioBlitz 2016
Noctuidae (Noctuinae)	*Phlogophora periculosa*	–	Confirmed	X	–	X	–
	Polia nimbosa	–	Confirmed	X	–	X	–
	Polia piniae	–	Confirmed	X	–	X	–
	Pseudorthodes irrorata	–	Confirmed	X	–	X	–
	Spiramter lutra	–	Confirmed	X	X	X	–
	Sideris maryx	–	Confirmed	X	X	X	–
Noctuidae (Pantheinae)	*Panthea virginarius*	–	Confirmed	X	–	X	–
Noctuidae (Plusiinae)	*Autographa californica*	–	Confirmed	X	–	X	–
	Syngrapha celsa	–	Confirmed	X	X	X	–
Notodontidae	*Oligocentria pallida*	Pale Prominent Moth	Confirmed	X	–	X	–
	Oligocentria semirufescens	Red-washed Moth	Confirmed	X	–	X	–
Notodontidae (Heterocampinae)	*Schizora ipomoeae*	Morning Glory Moth	Confirmed	X	X	X	–
	Schizura unicornis	–	Confirmed	X	–	X	–
Notodontidae (Notodontinae)	*Furcula scolopendrina*	Zigzao Furcula Moth	Confirmed	X	X	X	–
	Gluphisia septentrionis	Common Gluphisia Moth	Confirmed	–	–	X	–
	Pheosia rimosa	Black-rimmed Prominent Moth	Confirmed	X	–	X	–
Notodontidae (Phalerinae)	*Nadata gibbosa*	White-dotted Prominent Moth	Confirmed	X	–	X	–
Notontidae (Pygaerine)	*Clostera albosigma*	Sigmoid Prominent Moth	Confirmed	X	–	X	–
Pyralidae	Unidentified spp.	–	Confirmed	–	X	X	–
	Arta epicoenalis	Snout Moth	Probable	–	–	–	–
	Herpetogramma abdomalis	–	Confirmed	–	–	X	–
	Herpetogramma pertextalis	–	Confirmed	X	–	X	–
Tortricidae	*Choristoneura rosaceana*	–	Confirmed	X	–	X	–

Table B2 (continued). List of moth species detected in North Cascades National Park Service Complex. BioBlitz 2014 data were collected by D. Strenge and D. Dropper, and BioBlitz 2016 data were collected by J. Strange.

Family (Subfamily)	Species	Common Name	Presence	Evidence			
				Specimen	Photo	BioBlitz 2014	BioBlitz 2016
Uraniidae (Epipleminae)	*Calliizia amorata*	Gray Scoopwing Moth	Confirmed	–	X	X	–
Yponomeutidae	*Yponomeuta* spp.	–	Confirmed	–	X	X	–

Table B3. List of bee species (Order Hymenoptera) detected in North Cascades National Park Service Complex. BioBlitz 2014 data were collected by J. Rykken, and BioBlitz 2013/2016 data were collected by J. Strange.

Family (Subfamily)	Species	Presence	Elevation (m)	Evidence			
				Specimen	Photo	BioBlitz 2014	BioBlitz 2013 / 2016
Andrenidae (Andreninae)	*Andrena prunorum*	Confirmed	–	–	–	–	X
	Andrena trevoris Cockerell	Confirmed	1281	–	–	X	–
	Panurginus atriceps (Cresson)	Confirmed	1113–1806	X	X	X	–
	Panurginus ineptus Cockerell	Confirmed	1527–1669	–	–	X	–
Apidae (Apinae)	*Apis mellifera* Linnaeus	Confirmed [1]	1511–1630	X	X	X	–
	Bombus appositus Cresson	Confirmed	1319–1821	X	X	X	X
	Bombus bifarius Cresson	Confirmed	145–2023	X	–	X	X
	Bombus flavidus Eversmann	Confirmed	154–1871	–	–	X	–
	Bombus flavifrons Cresson	Confirmed	1–1934	X	X	X	X
	Bombus insularis (Smith)	Confirmed	725–2023	X	X	X	X
	Bombus melanopygus Nylander	Confirmed	1281–1934	X	X	X	X
	Bombus mixtus Cresson	Confirmed	153–2023	X	X	X	X
	Bombus occidentalis	Confirmed	356	–	–	–	X
	Bombus sitkensis Nylander	Confirmed	145–1745	X	X	X	X
	Bombus sylvicola Kirby	Confirmed	1113–2023	X	X	X	X
	Bombus vandykei (Frison)	Confirmed	215–1880	X	X	X	X
	Diadasia diminuta (Cresson)	Confirmed	153–154	X	X	X	–
	Nomada sp. 1	Confirmed	1113–2054	–	–	X	–
	Nomada sp. 3	Confirmed	1113	–	–	X	–
Colletidae (Colletinae)	*Colletes nigrifrons* Titus	Confirmed	1587–2010	–	–	X	–
Colletidae (Hylaeinae)	*Hylaeus annulatus* (Linnaeus)	Confirmed	1113–2010	–	–	X	–
	Hylaeus modestus (Cockerell)	Confirmed	1–1537	–	–	X	–

[1] Introduced, non-native.

Table B3 (continued). List of bee species (Order Hymenoptera) detected in North Cascades National Park Service Complex. BioBlitz 2014 data were collected by J. Rykken, and BioBlitz 2013/2016 data were collected by J. Strange.

Family (Subfamily)	Species	Presence	Elevation (m)	Evidence			
				Specimen	Photo	BioBlitz 2014	BioBlitz 2013 / 2016
Colletidae (Hylaeinae) (continued)	*Hylaeus nevadensis* (Cockerell)	Confirmed	1322	–	–	X	–
	Hylaeus wootoni (Cockerell)	Confirmed	1322–1537	–	–	X	–
Halictidae (Halictinae)	*Halictus confusus* Smith	Confirmed	145–1635	–	–	X	–
	Halictus rubicundus (Christ)	Confirmed	154–1871	x	X	X	X
	Halictus virgatellus Cockerell	Confirmed	145–1934	–	–	X	–
	Lasioglossum anhypops McGinley	Confirmed	1871	–	–	X	–
	Lasioglossum Dialictus spp.	Confirmed	1–1871	–	–	X	–
	Lasioglossum egregium (Vachal)	Confirmed	504	–	–	X	–
	Lasioglossum Evylaeus sp.2	Confirmed	1871	–	–	X	–
	Lasioglossum Evylaeus sp.3	Confirmed	1564–1745	–	–	X	–
	Lasioglossum Evylaeus sp.5	Confirmed	1745	–	–	X	–
	Lasioglossum niger complex	Confirmed	1452–1871	–	–	X	–
	Lasioglossum nigroviride (Graenicher)	Confirmed	1200	–	–	X	–
	Lasioglossum ?nigrum (Viereck)	Confirmed	1361–2010	–	–	X	–
	Lasioglossum spp.	Confirmed	228–2010	–	–	X	–
	Lasioglossum trizonatum (Cresson)	Confirmed	504	–	–	X	–
	Sphecodes spp.	Confirmed	1–1799	X	X	X	–
	Sphecodes sp. 2	Confirmed	154	–	–	X	–
	Sphecodes sp. 4	Confirmed	1–1821	–	–	X	–
Megachilidae (Megachilinae)	*Anthidium tenuiflorae Cockerell*	Confirmed	2010	X	X	X	–
	Megachile perihirta Cockerell	Confirmed	121–1871	–	–	X	–
	Megachile relativa Cresson	Confirmed	1200–1487	X	X	X	–
	Osmia densa Cresson	Confirmed	145–1527	–	–	X	–

[1] Introduced, non-native.

Table B3 (continued). List of bee species (Order Hymenoptera) detected in North Cascades National Park Service Complex. BioBlitz 2014 data were collected by J. Rykken, and BioBlitz 2013/2016 data were collected by J. Strange.

Family (Subfamily)	Species	Presence	Elevation (m)	Evidence			
				Specimen	Photo	BioBlitz 2014	BioBlitz 2013 / 2016
Megachilidae (Megachilinae) (continued)	*Osmia dolerosa* Sandhouse	Confirmed	1322–1564	–	–	X	–
	Osmia Melanosmia spp.	Confirmed	1113	–	–	X	–
	Osmia paradisica Sandhouse	Confirmed	725–2010	X	X	X	–
	Osmia pusilla Cresson	Confirmed	1527–1793	–	–	X	–
	Osmia tristella Cockerell	Confirmed	1200–1821	–	–	X	–
	Stelis spp.	Confirmed	1821	X	X	X	–

[1] Introduced, non-native.

Table B4. List of Syrphid flies (Order Diptera) detected in North Cascades National Park Service Complex. BioBlitz 2014 data were collected by J. Rykken.

Family (Subfamily)	Species	Presence	Elevation (m)	Evidence		
				Specimen	Photo	BioBlitz 2014
Syrphidae (Eristalinae)	*Blera scitula* (Williston)	Confirmed	1348–2010	X	–	X
	Criorhina coquilletti Williston	Confirmed	1489–1511	X	–	X
	Eristalis hirta Loew	Confirmed	121–2010	X	X	X
	Helophilus fasciatus Walker	Confirmed	1322–1487	X	X	X
	Hiatomyia plutonia (Hunter)	Confirmed	1487–1808	X	–	X
	Sericomyia chalcopyga Loew	Confirmed	1347–2010	X	X	X
	Sericomyia flagrans (Osten Sacken)	Confirmed	1871–1927	X	X	X
	Sphecomyia pattoni Williston	Confirmed	1489–1511	X	–	X
	Sphegina sp.	Confirmed	1511	X	–	X
	Syritta pipiens (Linnaeus) [1]	Confirmed	154	X	X	X
Syrphidae (Syrphinae)	*Chrysotoxum flavifrons* Macquart	Confirmed	1281	X	X	X
	Chrysotoxum ventricosum Loew	Confirmed	1281–2010	X	–	X
	Dasysyrphus intrudens (Osten Sacken)	Confirmed	507–1821	X	–	X
	Eriozona laxus (Osten Sacken)	Confirmed	2010	X	–	X
	Eupeodes americanus/fumipennis/pomus	Confirmed	121–1927	X	–	X
	Heringia rita (Curran)	Confirmed	2010	X	–	X
	Heringia sp.	Confirmed	5–2023	X	–	X
	Lapposyrphus aberrantis (Curran)	Confirmed	1322–2010	X	–	X
	Lapposyrphus lapponicus (Zetterstedt)	Confirmed	505–2010	X	–	X
	Melangyna fisherii (Walton)	Confirmed	1511	X	–	X
	Melangyna labiatarum (Verrall)	Confirmed	1587	X	–	X
	Melangyna umbellatarum (Fabricius)	Confirmed	1361–1927	X	–	X
	Melanostoma mellinum (Linnaeus)	Confirmed	504–1564	X	–	X

[1] Introduced, non-native.

Table B4 (continued). List of Syrphid flies (Order Diptera) detected in North Cascades National Park Service Complex. BioBlitz 2014 data were collected by J. Rykken.

Family (Subfamily)	Species	Presence	Elevation (m)	Evidence		
				Specimen	Photo	BioBlitz 2014
Syrphidae (Syrphinae) (continued)	*Paragus haemorrhous* Meigen	Confirmed	504–1793	X	–	X
	Parasyrphus relictus (Zetterstedt)	Confirmed	507–1956	X	–	X
	Platycheirus ciliatus Bigot	Confirmed	1511	X	–	X
	Platycheirus stegnus (Say)	Confirmed	1322	X	–	X
	Sphaerophoria abbreviata/asymmetrica	Confirmed	1361–2010	X	–	X
	Sphaerophoria abbreviata/asymmetrica/philanthus	Confirmed	1322–2010	X	X	X
	Sphaerophoria sp.	Confirmed	1487–1745	X	–	X
	Sphaerophoria sulphuripes (Thomson)	Confirmed	1361–1871	X	–	X
	Syrphus opinator Osten Sacken	Confirmed	1–1821	X	–	X
	Syrphus ribesii (Linnaeus)	Confirmed	504–2010	X	–	X
	Syrphus torvus Osten Sacken	Confirmed	1372–1821	X	–	X
	Toxomerus occidentalis Curran	Confirmed	243–1113	X	–	X

[1] Introduced, non-native.



Photo: Cascades red fox (*Vulpes vulpes cascadensis*) in Washington's South Cascades. This species, along with the grizzly bear (*Ursus arctos*), is now likely extirpated from North Cascades National Park Service Complex. Photograph taken by Jason Ransom.

The Department of the Interior protects and manages the nation’s natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

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