SPECIES STATUS ASSESSMENT
for the
ALEXANDER ARCHIPELAGO WOLF
(*Canis lupus ligoni*)

U.S. Fish and Wildlife Service
Region 7, Alaska

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EXECUTIVE SUMMARY

The purpose of this document is to synthesize scientific information relevant to assessing the current and future status of the Alexander Archipelago wolf (*Canis lupus ligoni*). It will be used to inform the U.S. Fish and Wildlife Service's (Service) decision on whether or not the Alexander Archipelago wolf warrants protection under the U.S. Endangered Species Act (Act). As such, it is not a decision document; instead, it forms the scientific basis from which the Service will draw conclusions and make decisions regarding the Alexander Archipelago wolf under the Act.

In this assessment, we review the morphology, taxonomy, distribution, life history, ecology, and population dynamics of the Alexander Archipelago wolf and, as appropriate, its primary prey, black-tailed deer (*Odocoileus hemionus* spp.). We acknowledge uncertainty surrounding the taxonomic status of the wolf, but after careful review of the best available information, we assume for the purpose of this assessment that the Alexander Archipelago wolf is a valid subspecies of gray wolf (*C. lupus*). We then describe current habitat and resource conditions of the Alexander Archipelago wolf and project future conditions by evaluating effects of environmental and anthropogenic stressors to wolves at the individual, population, and rangewide levels. We conclude this assessment by characterizing future status of the Alexander Archipelago wolf using the conservation biology principles of redundancy (ability to withstand catastrophic events), resiliency (ability to withstand stochastic disturbance events), and representation (ability to adapt to changing environmental conditions).

The Alexander Archipelago wolf occurs on the mainland of southeastern Alaska and coastal British Columbia west of the Coast Mountain Range and on larger islands except Admiralty, Baranof, and Chichagof islands and all of the Haida Gwaii, or Queen Charlotte, Islands (Figure ES-1). Its current range, which is similar to its recent historical range, covers roughly 217,000 km^2. These coastal wolves probably interact at low levels with interior, continental wolves via trans-boundary rivers and low mountain passes. Therefore, distribution boundaries of the Alexander Archipelago wolf are porous and are not defined sharply or with certainty; zones of intergradation with interior, continental wolves exist.

We estimated the rangewide population of the Alexander Archipelago wolf to be approximately 850–2,700 wolves with the majority (~62%) occurring in coastal British Columbia where populations have been stable since 2000. In southeastern Alaska (38% of the rangewide population), trend is estimated for only the population occupying Prince of Wales Island and surrounding islands (6% of rangewide population), which constitutes Game Management Unit (GMU) 2. Between 1994 and 2014, the GMU 2 wolf population declined by about 75% (SE=15), although confidence intervals of the point estimates overlap. For the remainder of southeastern Alaska (32% of rangewide population), population trend is not known. Generally, populations of Alexander Archipelago wolf are connected to one another, although some geographical disruptions exist due to the island geography within its range; the GMU 2 population is the most insular population.

The Alexander Archipelago wolf appears to be a habitat and diet generalist, although it exhibits some general preferences. These coastal wolves spend most of their time at elevations below 400 m, probably because abundance of prey typically is higher at low elevations compared to higher elevations. Their diet is highly variable across the range and seasons, but similar to gray wolves, ungulates compose a large portion of it with deer being the most common ungulate species available. The GMU 2 wolf population is more dependent on deer as prey compared to other coastal wolf populations because deer are the only ungulate available in GMU 2; elsewhere, at least two additional ungulate species occur. Consistent with their opportunistic predatory behavior, Alexander Archipelago wolves also consume marine and intertidal species including salmon (*Oncorhynchus* spp.) and marine mammals when and where available.

We identified multiple stressors that may be impacting individuals and populations of the Alexander Archipelago wolf, although most of them have the potential to affect wolves indirectly, not directly. Key stressors examined as part of this assessment include timber harvest, road development, wolf harvest, and climate-related events. Of these, wolf harvest is the only source of direct mortality that may have an impact at the population and rangewide levels. Although road development has little direct effect on wolves, roads provide access for hunters and trappers to areas that otherwise may be inaccessible or difficult to access. Timber harvest and winter severity influence deer habitat capability and abundance, which can impact wolf populations, especially if other ungulate species are not available. We also considered a variety of other stressors such as effects of small populations, oil development, overexploitation of salmon, and hybridization with dogs.

Because many stressors that may be affecting Alexander Archipelago wolves interact with one another, sometimes synergistically, we revised an existing, but outdated model of a hypothetical wolf population in GMU 2, to help determine the relative strength of influence of each stressor and the cumulative impact on wolves. In the model, we explicitly considered timber harvest, frequency of severe winters, and wolf harvest as functions of road development and ocean distance from towns and villages (both measures of access for hunters and trappers). We also used the model to predict wolf population trajectory in GMU 2, the area for which the most data on wolf population dynamics exist, under six scenarios representing possible future conditions. We summarize results of the model in this status assessment and fully describe the model, its assumptions, and outputs in Gilbert et al. (2015).

We then evaluated the relative level of resiliency (low, intermediate, high) of five populations, or group of populations, of the Alexander Archipelago wolf by examining the magnitude of stressors and their known or expected effect on wolves. Those populations included southern coastal British Columbia (Regions 1 and 2), northern coastal British Columbia (Regions 5 and 6), mainland southeastern Alaska (GMUs 1 and 5A), and GMUs 2 and 3 in southeastern Alaska (Figure ES-1). For three populations (i.e., southern and northern coastal British Columbia and GMU 2), we relied on trend information to inform our assessment of their resiliency to stressors, individually and cumulatively. However, for two populations in southeastern Alaska (GMUs 1 and 5A, and GMU 3), we lacked trend information. Therefore, we compared magnitude of stressors to those populations with those of populations for which trend information exists and then assigned a level of resiliency based on degrees of similarity and difference; for these two populations, uncertainty regarding resiliency and future status is greater than for the three populations with existing trend information.

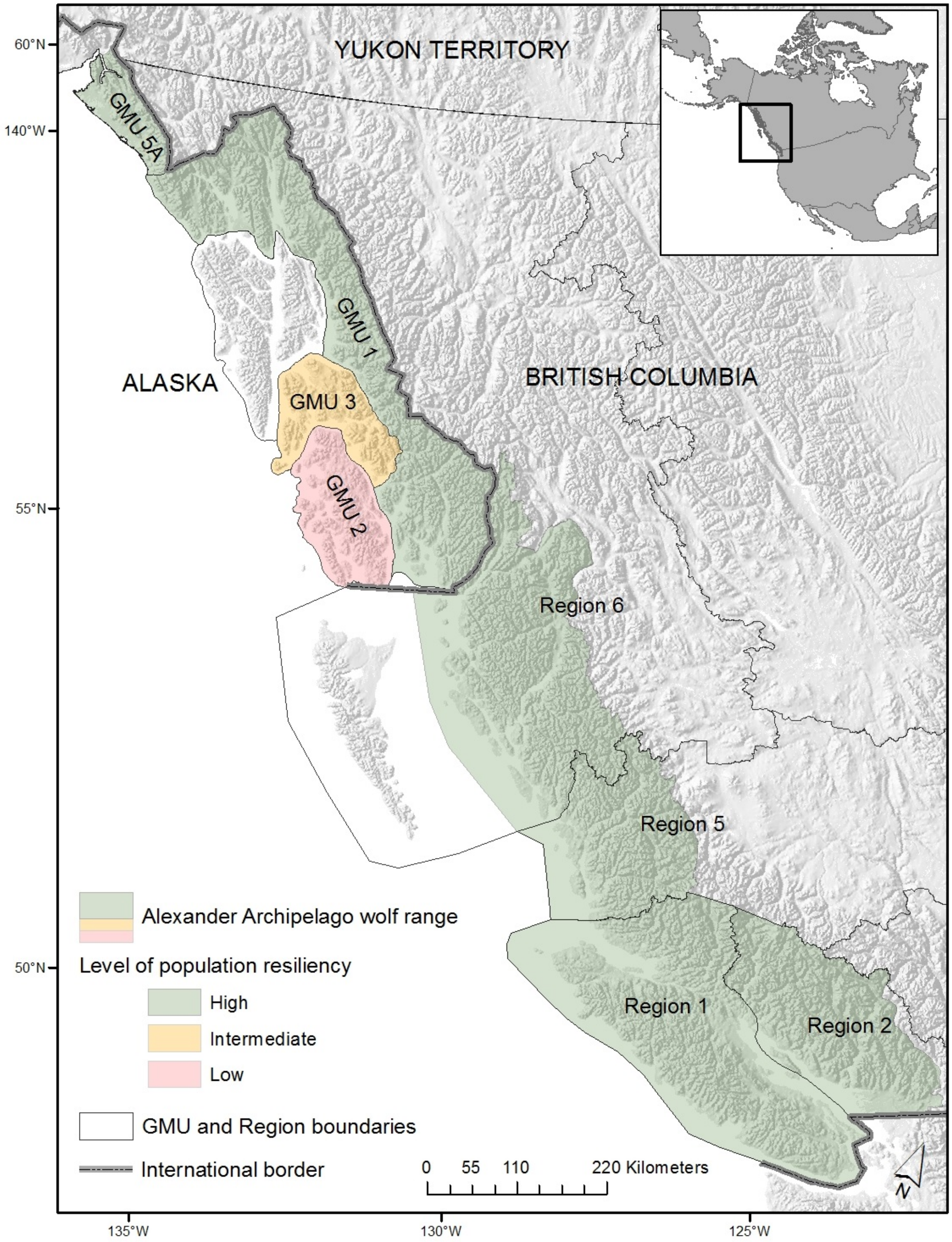

Figure ES-1. Assumed range of the Alexander Archipelago wolf (*C. l. ligoni*), and level of resiliency of individual or groups of populations, which were defined using boundaries of Game Management Units (GMU) in southeastern Alaska and Regions in coastal British Columbia.

Of the five populations evaluated, we found that three of them, composing 80% of the rangewide population, exhibit high resilience to stressors (Figure ES-1). Both populations in coastal British Columbia have been stable since 2000 despite intensive and extensive timber harvest in the southern portion (Regions 1 and 2; 30% of the forest logged) and in the northern portion (Regions 5 and 6; 16% of the forest logged). We attribute their resiliency to the availability of ungulate species other than deer as prey and to apparently sustainable rates of wolf harvest (average reported harvest of <7% of the population annually). Further, these populations likely encounter few disruptions to demographic and genetic connectivity, although we found no estimates of dispersal specific to wolves in coastal British Columbia. Based on similarities in the overall magnitude of stressors and population characteristics, such as ungulate prey availability, with coastal British Columbia, we determined that the Alexander Archipelago wolf population on the mainland of southeastern Alaska (GMUs 1 and 5A) also exhibits high resilience, although we lack trend estimates for this population and therefore are less certain of its ability to withstand stochastic disturbances.

The GMU 2 wolf population, which constitutes 6% of the rangewide population, demonstrates low resilience to stressors, specifically the synergistic effects of wolf harvest and timber harvest (Figure ES-1). Although this population appears to be harvested at sustainable rates (average of 17% of the population annually), unreported harvest contributes substantially to total wolf harvest (38–45% of total harvest) in GMU 2, resulting in unsustainable rates of wolf harvest in some years. High rates of total harvest in GMU 2 have been facilitated by the highest levels of road and boat access for hunters and trappers across the range of the taxon. In addition, approximately 23% of the forest has been logged, likely reducing numbers of deer, the only ungulate species available as prey. The combination of these factors likely has caused an apparent population decline of about 75% (SE=15) since 1994 and, as predicted by our population model, wolf abundance in GMU 2 is expected to decline by another roughly 8–14% of current levels over the next 30 years.

Lastly, we determined that the GMU 3 wolf population in central southeastern Alaska (14% of the rangewide population), shows an intermediate level of resiliency to stressors (Figure ES-1). However, like the population on the mainland of southeastern Alaska (i.e., GMUs 1 and 5A), we lack a trend estimate for the GMU 3 population and therefore, we used a comparative approach. The GMU 3 wolf population has similarities with both a stable population in northern coastal British Columbia (Regions 5 and 6; e.g., level of timber harvest) and with a declining population in GMU 2 (e.g., island geography). For example, in GMU 3, 14% of the forest has been logged, reducing deer habitat capability, although wolves have access to ungulate prey other than deer; these attributes are similar to those in northern coastal British Columbia. Yet, rates of reported harvest in GMU 3 (21% of population annually) are slightly higher than those in GMU 2 (17% of population annually), although we found no evidence indicating that unreported harvest in GMU 3 is occurring at or near the high rates documented in GMU 2; in addition, road and boat access for hunters and trappers in GMU 3 is lower than that in GMU 2. Thus, in considering the evidence collectively, we classified the GMU 3 population as exhibiting an intermediate level of resiliency, in part owing to its island geography.

In conclusion, we believe that the future status of the rangewide population of the Alexander Archipelago wolf likely will be stable or perhaps slightly lower than current levels based on its

resiliency, redundancy, and representation. We found that (1) most (80%) of the rangewide population exhibits high resilience to stochastic disturbance events; (2) multiple populations are distributed across a broad range, demonstrating redundancy for withstanding catastrophic events, although two island populations (i.e., GMUs 2 and 3) constituting 20% of the rangewide population are more insular than the mainland populations; and, (3) as a habitat and diet generalist, the Alexander Archipelago wolf exhibits a high degree of ecological diversity and most populations appear to harbor sufficient levels of genetic diversity with no evidence of genetic bottlenecking; both of these characteristics indicate representation, or the ability to adapt to changing environmental conditions. Owing to predicted declines in the GMU 2 wolf population, it is likely that the rangewide population will decrease in the future, but we expect the overall effect to be minor given that the GMU 2 population constitutes only 6% of the rangewide population, is geographically peripheral to the other populations, and appears to serve as a sink population. Nonetheless, the persistence of the GMU 2 population is desired and requires careful management actions and decisions to ensure its future health.

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Note on typography: This document consists of six chapters, a list of literature cited, and two appendices. Chapters are presented in all capital letters (primary headings); sections are labeled by chapter then section number (e.g., 3.1 refers to Chapter 3, Section 1) and are underlined (secondary headings); and, subsections are denoted by chapter, section number, and subsection number (e.g., 3.1.2. refers to Chapter 3, Section 1, Subsection 2) and are presented in bold typeface (tertiary headings). Primary, secondary, and tertiary headings are listed in the Table of Contents. In some subsections, we found it necessary to include fourth- and fifth-order headings. Fourth-order headings are labeled in italics and underlined and fifth-order headings are presented in standard text; in both cases, the heading is followed by an Em Dash (i.e., "—") and the paragraph begins on the same line. Tables and figures are numbered in chronological order as they appear in the text.

Typography example:
CHAPTER
Section
Subsection
Fourth-order heading.—Paragraph begins.
Fifth-order heading.—Paragraph begins.

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CHAPTER 1: INTRODUCTION

1.1. Purpose and focus of this assessment

The purpose of this document is to synthesize scientific information relevant to assessing the current and future status of the Alexander Archipelago wolf (*Canis lupus ligoni*). We compiled it using the U.S. Fish and Wildlife Service's (Service) guidance on developing a Species Status Assessment (Service 2015. This document will be used to inform the U.S. Fish and Wildlife Service's (Service) decision on whether or not the Alexander Archipelago wolf warrants protection under the U.S. Endangered Species Act (Act). However, we emphasize that this is not a decision document; it contains no conclusions on whether or not the wolf should be listed. Instead, it forms the scientific basis from which the Service will draw conclusions and make decisions.

We collated existing information from published papers, final agency reports and pertinent archived datasets. We also considered information submitted to the Service in the petition (Center for Biological Diversity [CBD] and Greenpeace 2011), during the 90-day finding public comment period (March 31–May 30, 2014), and in response to specific data requests. To the best of our ability, we worked directly with researchers actively studying the Alexander Archipelago wolf or aspects of the ecosystem in which it lives, to include the most current information available. Thus, we aimed to be as comprehensive as possible, but did focus our review on two areas. First, we concentrated on information describing the Alexander Archipelago wolf, drawing on information about the gray wolf (*C. lupus*) and its' subspecies only when necessary (e.g., significant data gap, context). We consulted frequently with Mech and Boitani (2003), a compilation of summaries on wolf behavior, ecology, and conservation throughout the world, and in some sections of this assessment, we refer the reader directly to tables and figures in their publication that list summary statistics for well-studied wolf populations outside of southeastern Alaska and coastal British Columbia. Second, we focused our review on new information collected since 1997 when the Service last reviewed the status of the Alexander Archipelago wolf (62 Federal Register 46710, September 4, 1997).

As part of this effort, we also updated a population model for wolves on Prince of Wales Island (POW) that was originally developed by Person and Bowyer (1997) during our last review of the Alexander Archipelago wolf. This model was used to project future wolf abundance based on hypothetical wolf packs on POW under different resource condition scenarios with variations on levels of timber harvest, road management, wolf harvest, and frequency of winter severity. It was developed with input from key agency partners (e.g., Alaska Department of Fish and Game [ADFG], U.S. Forest Service [USFS]) and from experts on wolves and population modeling. Because of deadlines associated with the Alexander Archipelago wolf listing decision, we wrote this assessment in parallel with a final report describing the population model (Gilbert et al. 2015). These two documents complement one another yet are separate standalone documents. In addition, both documents underwent substantial peer review, as well as agency review; we incorporated comments received during the review process into the final versions of the documents.

1.2. Geographic scope

In this assessment, we summarize scientific information on wolves in southeastern Alaska and coastal British Columbia (Figure 1). Although the taxonomic status of wolves in this region is uncertain at this time (see *Taxonomy* below), we acknowledge that these wolves harbor unique ecological and genetic traits specific to this coastal region and that they appear to constitute a different group compared to continental wolves. Thus, for the purpose of this assessment, we used the Coast Mountain range, which extends 1,600 km from the southwestern corner of Yukon Territory, Canada in the north to the mouth of the Fraser River in the south, as a geographic boundary. We define southeastern Alaska as the area extending from Yakutat in the north to Dixon Entrance in the south, including all islands in the Alexander Archipelago and the narrow strip of mainland eastward to the Coast Mountain range and the Canadian border (Figure 1). We delimit coastal British Columbia to be Dixon Entrance in the north to the Fraser River in the south, including all islands and the mainland west of the Coast Mountains (Figure 1). See *Distribution* below for more detailed descriptions of wolf distribution in this region.

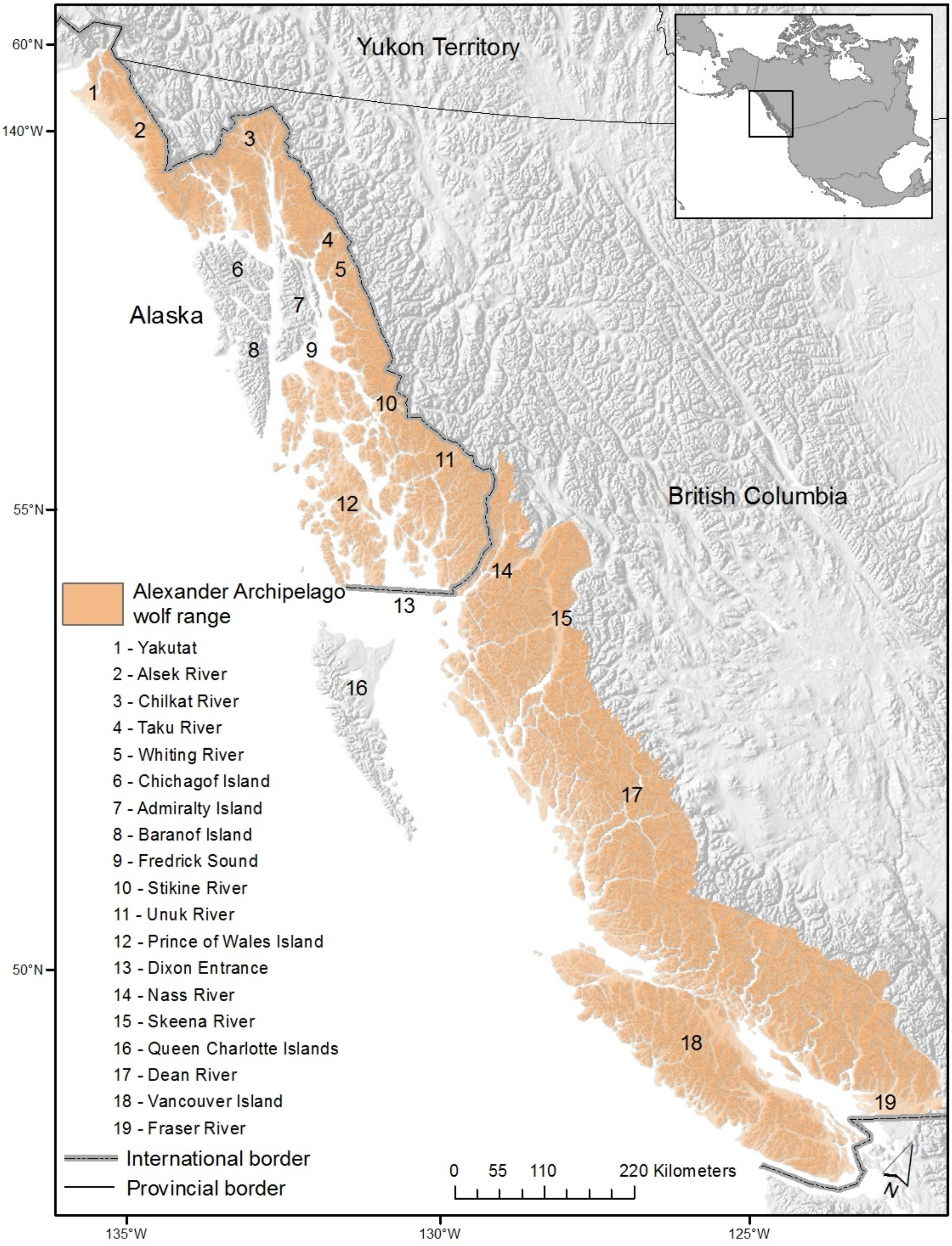

Figure 1. Assumed range of the Alexander Archipelago wolf (*C. l. ligoni*), as reviewed in this assessment, southeastern Alaska and coastal British Columbia.

1.3. Review of previous efforts

Prior to this assessment, the most recent comprehensive compilation of information on the Alexander Archipelago wolf in southeastern Alaska was completed by the Service in 1997 as part of an evaluation to determine whether or not listing was warranted under the Act (62 Federal Register 46710, September 4, 1997). This evaluation was conducted in parallel and in close collaboration with a similar effort led by the U.S. Forest Service (USFS), "The Alexander Archipelago Wolf: A Conservation Assessment" (Person et al. 1996) during the 1997 revision of the Tongass Land and Resource Management Plan (USFS 1997). This USFS-led assessment remains a primary resource for information on Alexander Archipelago wolf in southeastern Alaska.

Since 1997, several other assessments or compilations of information in Alaska were conducted, but none was focused solely on the Alexander Archipelago wolf. For example, MacDonald and Cook (2007) summarized data on taxonomy, status, and distribution of amphibians and mammals, including the wolf, in southeastern Alaska using museum specimens; this publication generally serves as an expanded checklist of species. In addition, Albert and Schoen (2007) completed a conservation assessment and resource synthesis for southeastern Alaska, specifically the Tongass National Forest (hereafter Tongass). This comprehensive assessment includes sections describing status, habitat relationships, and conservation implications for key species of fish, birds, and mammals, including the Alexander Archipelago wolf.

We are not aware of a status assessment targeted to wolves in coastal British Columbia. Darimont and Paquet (2000, 2002) described distribution of wolves on the central and northern coast based on wolf sign including tracks, scat, sightings, and carcasses; the authors also analyzed patterns of distribution in terms of island size, isolation, and distance from the mainland. In 2014, the provincial government published a management plan for wolves in British Columbia (British Columbia Ministry of Forests, Lands and Natural Resource Operations [BCMO] 2014). This plan summarizes the best available information for managing wolf populations, including a threats assessment, in British Columbia, but it is not specific to the coastal region.

1.4. Terminology used in this assessment

Within the range of the Alexander Archipelago wolf, old-growth forests are heterogenous at the scale of watersheds and at finer scales. Some wildlife species select for various attributes of the forest habitat, but these attributes vary from species to species. Likewise, terminology used to describe forest habitat types and attributes also varies across authors, researchers, agencies, countries, etc. Throughout this assessment, we were unable to use a consistent classification system when referring to forest habitat. Generally, we aimed to follow Farmer and Kirchhoff (2007), although we also drew on information included in Alaback (1982). However, when describing results of studies reported by other authors, we chose to retain the terminology used in those publications or reports. We included citations and page numbers throughout the document, allowing the reader to consult with specific studies if desired.

We typically reference two forest habitat types: young-growth and old-growth forests. Young-growth forest (also called second-growth forest) refers to forest stands that have regenerated following timber harvest or large natural events (e.g., wind storms). When relevant, we further defined second-growth forests as: (a) young young-growth to describe forest stands less than 25 years in age; (b) transitional young-growth as forest stands 26–75 years of age; and, old young-growth as 76–150 years of age. Old-growth forests are forest stands in the late successional stage of forest development, typically at least 150 years old. They are defined in many ways, but usually include a mix of tree sizes and ages, ranging from the largest and oldest that a site and the tree species are capable of producing, to young trees emerging through the understory. Old-growth forests also tend to have a significant component of dead trees and downed logs. These forests traditionally have been described based on timber volume (e.g., low-volume old-growth forest). In this document, we refer to productive old-growth forest, which includes all timber volume classes that render a stand commercially viable (i.e., capable of producing at least 8,000 board feet per acre, as defined by the USFS [2008a, p. 7-29]).

CHAPTER 2: DESCRIPTION OF THE ALEXANDER ARCHIPELAGO WOLF

2.1. Physical description

The Alexander Archipelago wolf has been described as being darker and smaller, with coarser and shorter hair compared to continental gray wolves (Goldman 1937, pp. 39–40; Wood 1990, p. 1). However, we are not aware of a recent comprehensive study or examination of specimens that supports this statement.

Like most gray wolves, fur coloration of Alexander Archipelago wolves varies considerably from pure white to uniform black with most wolves having a brindled mix of gray or tan with brown, black, or white. Based on recent harvest records (2008–2011), the black color morph is more common on the mainland of southeastern Alaska (20–30%; ADFG 2012, pp. 5, 18, 24) compared to the southern islands (2%; ADFG 2012, p. 34). On the British Columbian coast, Darimont and Paquet (2000, p. 17) reported that 25% of wolves were black in color and, of the remaining 75% that were gray, 40% had a brownish-red tinge. The pure white color morph appears to be rare throughout the region. For comparison, Adams et al. (2008, p. 8) reported 24% (12 of 51) of wolves captured as part of a research study in the Central Brooks Range, Alaska to be black with the remainder having gray pelts.

In southeastern Alaska, Alexander Archipelago wolves greater than six months old weigh between 22 and 52 kilograms (kg) with males averaging 37.8 kg (SE=0.6) and females averaging 31.2 kg (SE=0.6; Valkenburg 2015). On some islands in the archipelago (e.g., POW) wolves are smaller on average compared to those on the mainland (Table 1; Valkenburg 2015), although these differences are not statistically significant for males (F[2,85]=3.00, p=0.06) or females (F[2,74]=1.30, p=0.28). In all of British Columbia (not just the coastal area), wolves generally weigh between 30 and 50 kg (BCMO 2014, p. 3). For context, in the Central Brooks Range female wolves greater than one year old averaged between 36.9 and 40.1 kg and males were between 42.4 and 47.8 kg (Adams et al. 2008, p. 8). In northwestern Minnesota, average weights of both sexes were lower (females=30.0 kg, males=35.9 kg; Mech and Paul 2008, p. 935) and were more similar to wolves in southeastern Alaska.

Table 1. Weights (kg) of Alexander Archipelago wolves greater than six months old by ADFG Game Management Units (GMU) in southeastern Alaska (Valkenburg 2015; Figure 2).

GMU	Geographic area	Male				Female			
		Mean	SE	n	Range	Mean	SE	n	Range
1	Mainland	36.6	1.5	23	22.7–47.2	30.3	1.3	23	17.2–41.3
2	Prince of Wales Island and surrounding islands	35.7	0.9	17	29.5–43.1	30.6	0.8	20	24.5–36.4
3	Kuiu, Kupreanof, Mitkof, Zarembo, Etolin, and Wrangell islands	39.1	0.8	48	27.7–51.7	32.2	0.7	34	20.9–43.1

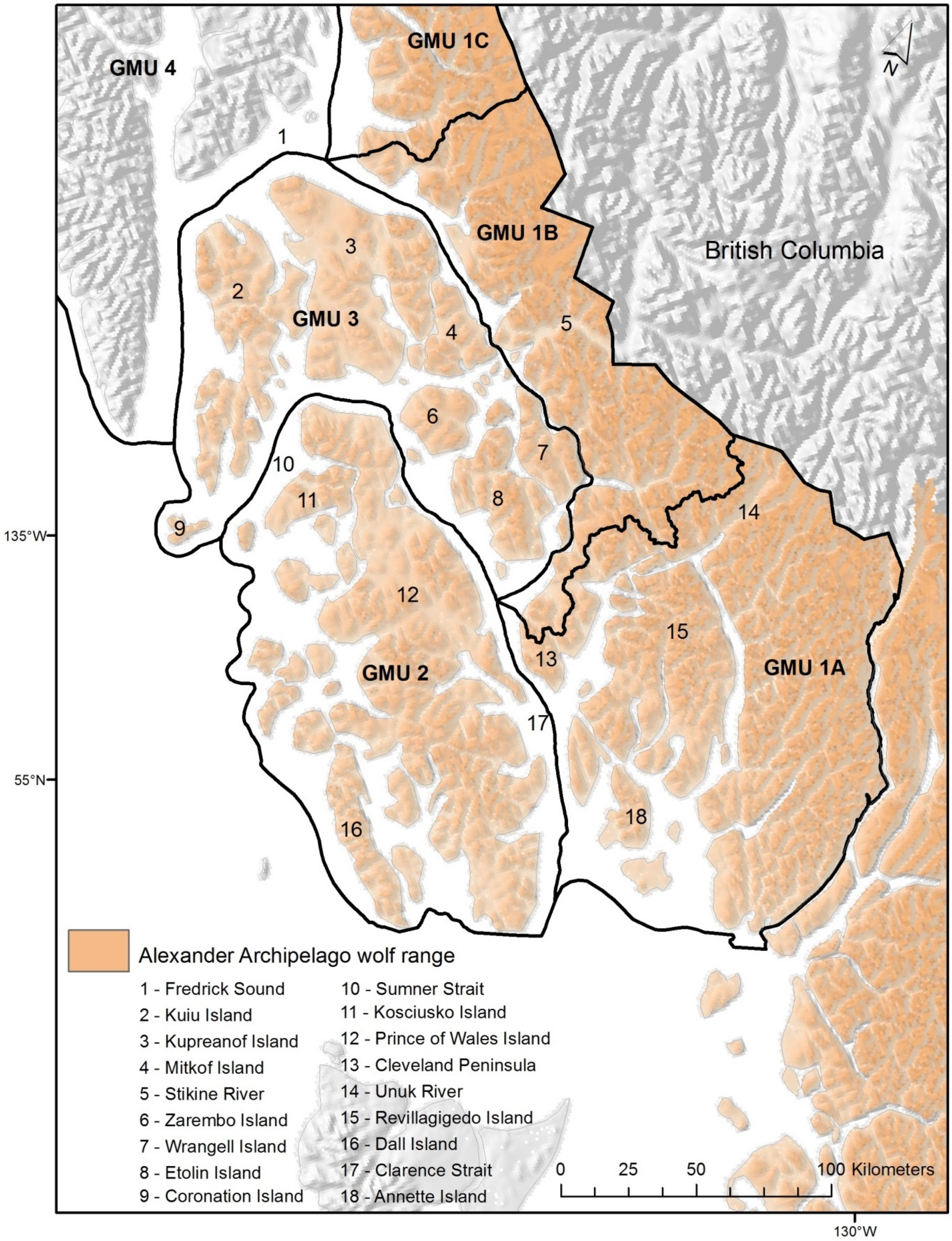

Figure 2. Range of the Alexander Archipelago wolf by Game Management Unit (GMU) in the southern portion of southeastern Alaska.

2.2. Taxonomy

The taxonomy of wolves in North America, including the recognition of *C. l. ligoni* as a subspecies, is a complex topic that has been debated for decades. Chambers et al. (2012) contains the most recent and comprehensive review of the taxonomy of wolves in North America. Although, the authors' interpretation of subspecific groupings has been contested (National Center for Ecological Analysis and Synthesis [NCEAS] 2014; see Uncertainty of taxonomic status), we found the review and references therein to be a valuable resource describing key morphological and genetic analyses. We refer the reader to Chambers et al. (2012) for a detailed account of taxonomy of North American wolves.

For the purpose of this assessment, we assume that the Alexander Archipelago wolf, *C. l. ligoni*, is a subspecies of gray wolf, although we recognize the uncertainty associated with this designation (see *Uncertainty of taxonomic status*). Nonetheless, we believe persuasive evidence exists in the literature suggesting that wolves in southeastern Alaska and coastal British Columbia are an ecological and genetic unit worthy of analysis under the Act. We recognize that zones of intergradation between coastal and interior continental wolves exist and that they probably are dynamic and may be substantial in size, especially in areas where few physical barriers to wolf movement are present (e.g., southern portion of coastal British Columbia) or where major river valleys facilitate movement (e.g., northern portion of southeastern Alaska). Below, we summarize morphological and genetic information on the Alexander Archipelago wolves with an emphasis on recent studies.

2.2.1. Morphometric analyses

Goldman (1937, pp. 39–40) was the first to propose the Alexander Archipelago wolf as a subspecies of the gray wolf (*C. lupus*). He described *C. l. ligoni* as a dark colored subspecies of medium size and short pelage that occupied the Alexander Archipelago and adjacent mainland of southeastern Alaska, northward along the Pacific Ocean coast to Yakutat Bay. Goldman (1944) later described 23 wolf subspecies in North America, including *C. l. ligoni*, and shortly thereafter Hall and Kelson (1952) described 24 subspecies of gray wolf.

As researchers developed more sophisticated and powerful tools for understanding morphological variation (e.g., multivariate discriminant analysis), wolf taxa in Alaska and North America tended toward consolidation. Jolicouer (1959, p. 298) examined pelage coloration and skull measurements of wolves in western North America and deemed it probable that "far too many subspecific designations were in use." Pedersen (1982) then analyzed morphological data from four populations of wolves in Alaska that were recognized as subspecies by Goldman (1944) and proposed two phenotypically distinct subspecific groups, *C. l. ligoni* in southeastern Alaska and *C. l. pambasileus* in interior and southcentral Alaska (pp. 345, 360). Friis (1985) concluded that wolves in southeastern Alaska were morphologically similar to wolves that historically occupied coastal British Columbia, Vancouver Island and the contiguous western United States, although her study included only eight samples from coastal British Columbia. She found "no evidence to justify separation of *C. l. ligoni* as a distinct subspecies confined to the Alaska panhandle," and suggested that perhaps instead the southern boundary of the "*ligoni*" group should be extended southward.

Concurrently, Nowak (1983) suggested a consolidation of wolf taxa in North America into five "groups" based on a review of the literature and bivariate analysis of two skull measurements. Like Friis (1985), he also suggested that wolves in southeastern Alaska showed an affinity to wolves that historically occupied the mountains and plains of the western United States, the Great Lakes region, the Pacific coast of the United States, and southwestern Canada (Nowak 1983). Later, Nowak (1995, p. 375) conducted a more comprehensive multivariate analysis of 10 measurements of 580 skulls of modern adult male wolves and proposed a revised taxonomy for wolves in North America that recognized five subspecies of gray wolf (*C. l. arctos*, *C. l. baileyi*, *C. l. lycaon*, *C. l. nubilus*, and *C. l. occidentalis*). In this taxonomic revision, he combined *C. l. ligoni* with *C. l. nubilus*, a subspecies he believed was formerly distributed from the western coasts of the United States and Canada east to the Great Lakes region and north through central and northeastern Canada (p. 396).

More recently, Chambers et al. (2012) reviewed morphological and genetic analysis of North American gray wolves and concluded that three (*C. l. baileyi*, *C. l. nubilus* [including *C. l. ligoni*], and *C. l. occidentalis*) subspecies are valid with a possible fourth subspecies (*C. l. arctos*), but too few data exist to verify its legitimacy; the authors also determined that *C. l. lycaon* is a full species (*C. lycaon*). In this latest review, *C. l. ligoni* again was grouped with and referred to as *C. l. nubilus* (Chambers et al. 2012, p. 1; see *Uncertainty of taxonomic status*).

We emphasize that our description of morphological analyses contains only the key studies related to taxonomy of wolves in southeastern Alaska and coastal British Columbia. We found others (e.g., Hall 1981), but these studies do not add significantly to the information presented above.

2.2.2. Genetic analyses

Since 1997 when the last status assessment was completed for Alexander Archipelago wolf (62 FR 46710, September 4, 1997), several molecular ecology studies have been conducted on wolves from southeastern Alaska and coastal British Columbia. All of these studies measured neutral genetic variation, which has no direct effect on fitness, but is useful for investigating processes such as gene flow or dispersal. We are not aware of any studies that have examined adaptive genetic variation, or the potential to adapt to local conditions, of wolves in this region.

Researchers have used both mitochondrial DNA (mtDNA) and nuclear DNA markers to investigate neutral genetic variation of the Alexander Archipelago wolf (Table 2). The scope of inference of these studies and their value to informing taxonomy depends on the type of genetic marker used and the spatial and temporal extent of the samples analyzed. Generally, mtDNA is useful for questions related to phylogeny (e.g., evolutionary relationships among groups of organisms, detecting hybridization, tracing maternal lineages) and nuclear DNA (e.g., microsatellite, single nucleotide polymorphisms [SNPs]) is well suited for traditional population genetic studies (e.g., measuring genetic variation, determining population structure; for a review of genetic concepts and tools, see Mills 2013, pp. 33–53). Below, we summarize key genetic studies of wolves in southeastern Alaska and coastal British Columbia that are relevant to taxonomy, specifically *C. l. ligoni*; see Genetic connectivity for a more detailed discussion of genetic structuring and gene flow within the range of the Alexander Archipelago wolf.

Table 2. Key genetic studies of wolves in southeastern Alaska and British Columbia that generated new data (i.e., not meta-analyses or reinterpretations of existing data) using mitochondrial DNA (mtDNA), microsatellite, and single nucleotide polymorphism (SNPs) markers.

		Source of samples		
Study	Type of marker	Southeastern Alaska	Coastal British Columbia	Continental North America
Shields 1995	mtDNA	Yes	No	Yes
Weckworth et al. 2005	Microsatellite	Yes	No	Yes
Breed 2007	Microsatellite	Yes	Yes	No
Munoz-Fuentes et al. 2009	mtDNA	No	Yes	Yes
Weckworth et al. 2010	mtDNA	Yes	No	Yes
Von Holdt et al. 2011	SNPs	No	Yes	Yes
Weckworth et al. 2011	mtDNA	Yes	Yes	Yes
Cronin et al. 2015a	SNPs	Yes	No	Yes

Mitochondrial DNA.—Shields (1995, pp. 6, 11) conducted the first genetic study of the Alexander Archipelago wolf by analyzing mtDNA. Based on a single fixed allelic substitution in 28 of 29 wolves sampled in southeastern Alaska, he concluded that these wolves (assumed to be *C. l. ligoni*) were genetically distinct from wolves in interior Alaska and Yukon Territory, Canada (n=9; pp. 6, 9, 11). Within southeastern Alaska, Shields (1995, pp. 7–8) found considerable genetic variability among wolves, but did not find geographic structure indicating that genetically unique subpopulations of wolves occurred on individual islands in the archipelago. Since this foundational study, three additional studies using mtDNA confirmed that coastal wolves in southeastern Alaska and coastal British Columbia appear to be genetically differentiated from interior continental wolves (Munoz-Fuentes et al. 2009; Weckworth et al. 2010; Weckworth et al. 2011).

Munoz-Fuentes et al. (2009, p. 5) sampled 160 wolves in British Columbia and found eight distinct haplotypes, one of which was absent in inland wolves, five were absent in coastal wolves, and only two were shared; coastal wolves were the only sampled population across northwestern North America to have an endemic haplotype with a frequency greater than 5% (p. 9). Pairwise fixation indices (Φ_{ST}), or the relative magnitude of gene differentiation among subpopulations (Nei 1987, p. 190), between coastal and inland British Columbia was 0.242 and between coastal British Columbia and elsewhere in North America was greater than 0.410 (p. 9). The authors found that habitat (coastal, interior) explained most (21%) of the genetic variation among wolf populations in their sample (p. 8; Figure 3), suggesting that ecological factors may be driving differentiation of wolves. Munoz-Fuentes et al. (2009, p. 9) also determined that coastal wolves were more differentiated from *C. l. occidentalis* (samples from Alaska, Alberta, and Northwest Territories; Φ_{ST}=0.305) and *C. l. nubilus* (historical samples from Leonard et al. 2005; Φ_{ST}=0.550) than *C. l. occidentalis* and *C. l. nubilus* were from each other (Φ_{ST}=0.125).

Weckworth et al. (2010, p. 366) analyzed mtDNA from a large sample of wolves from southeastern Alaska (n=130), interior continental North America (n=173), and Russia (n=4) and reported results similar to Munoz-Fuentes et al. (2009). Seventeen haplotypes were identified with four restricted to Russian individuals, two to wolves from southeastern Alaska, and nine found only in the continental group; two haplotypes were shared between the coastal group of southeastern Alaska and the continental group (Weckworth et al. 2010, p. 367). Subdivision of North American wolves into coastal and continental groups explained 56% of the genetic variation (p. 368; Figure 3). The authors reported mtDNA pairwise population estimates of Φ_{ST} within the coastal group to range from 0.00 to 0.47 and between the coastal and continental groups to be between 0.32 and 0.97 (p. 370).

Lastly, Weckworth et al. (2011, p. 2) conducted the most comprehensive analysis of mtDNA from wolves in southeastern Alaska (n=130) and coastal British Columbia (n=75) and compared results to wolves from continental North America (n=102). Twenty haplotypes were recorded, including some from extirpated regions; three haplotypes were restricted to coastal British Columbia and southeastern Alaska with two of them shared and one found only in southeastern Alaskan wolves. The most common haplotype found in the coastal region was shared with wolves in interior British Columbia (pp. 2–3). When divided into coastal and continental groups, 51% of the genetic variation was explained (p. 5; Figure 3). Pairwise population comparisons indicated some genetic structuring within the coastal group, but the pattern was complex and did not conform to an isolation-by-distance pattern; for example, populations in southeastern Alaska were more similar to Vancouver Island than they were to one another (Figure 3 on p. 6). Generally, however, island populations were differentiated from one another and the mainland, but differences were not always statistically significant (pp. 4–5). The authors concluded that coastal lineages of wolves in southeastern Alaska and coastal British Columbia are distinct from North American continental wolves (p. 5), corroborating mtDNA results of Munoz-Fuentes et al. (2009) and Weckworth et al. (2010).

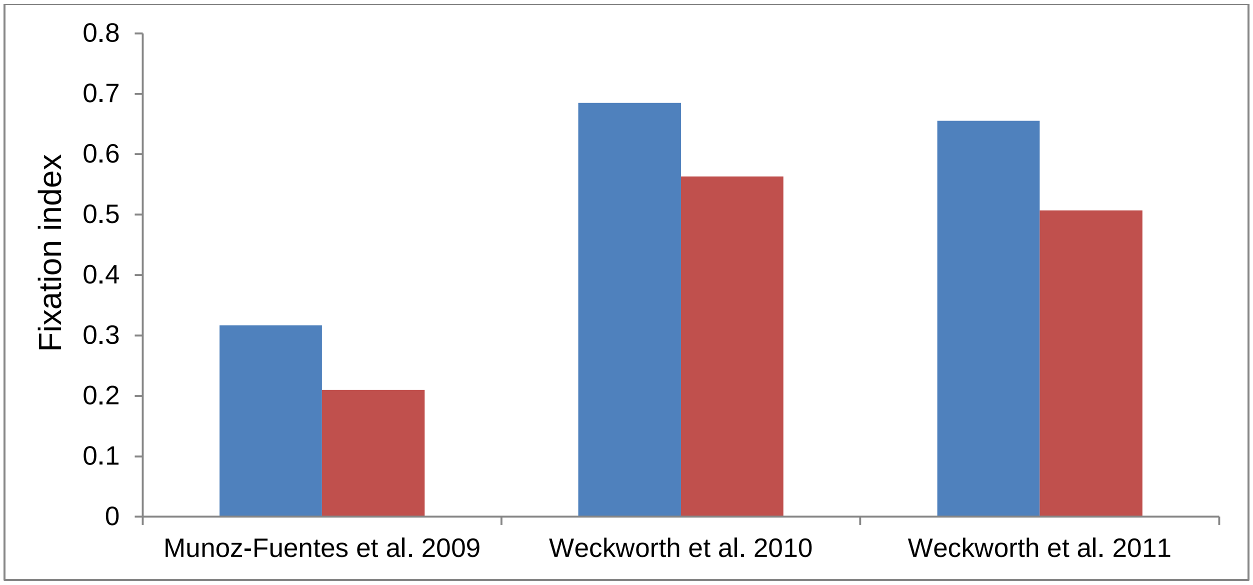

Figure 3. Estimates of Φ_{ST} (blue bars) and Φ_{CT} (red bars) from three studies of mitochondrial DNA variation in coastal and inland wolves from northwest North America. Here Φ_{ST} is the proportion of the total genetic variation that is explained by differences among all populations (overall population structure) and Φ_{CT} is the proportion of the total genetic variation explained by differences between the coastal and continental population groups (between group population structure). Source studies are Munoz-Fuentes et al. (2009, p. 8), Weckworth et al. (2010, p. 369), and Weckworth et al. (2011, p. 5). Results of all three studies demonstrate that most of the genetic variation among wolves sampled was explained by the coastal and continental groupings.

Nuclear DNA.—Weckworth et al. (2005) was the first to examine nuclear DNA of the Alexander Archipelago wolf. Based on microsatellite markers of 101 wolves from southeastern Alaska and 120 wolves from interior continental populations (p. 919), they found that coastal wolves appear to be geographically isolated from continental wolves (p. 924). Mean number of alleles per locus was 5.00 in the coastal group and 7.09 in the continental group; continental populations had a higher frequency of private alleles (4.60 alleles per population) than the southeastern Alaskan population (1.25), although these alleles were restricted to single individuals (p. 921). Within southeastern Alaska, two distinct clusters emerged from an assignment test, one on POW and the other encompassed the remainder of the area (pp. 923, 926). Pairwise estimates of the fixation index (F_{ST}) between POW and three other coastal populations ranged from 0.10 to 0.20, yet F_{ST} values among these three coastal populations were smaller (range=0.02–0.06; p. 923).

Shortly thereafter, Breed (2007) conducted a similar study using microsatellite markers, but included samples of wolves from southeastern Alaska and British Columbia only. The authors considered three “management units” in their analysis, POW (n=45 wolves), mainland coastal Alaska (n=31), and coastal British Columbia (n=42), which served as a control (p. 7). Mean number of alleles ranged from 3.85 on POW to 5.54 in coastal British Columbia (p. 17); POW displayed only two private alleles followed by mainland southeastern Alaska with four and coastal British Columbia with 18 private alleles (79 alleles total across all three units; p. 18). Pairwise F_{ST} values were 0.09 between coastal British Columbia and mainland southeastern Alaska, 0.12 between POW and mainland southeastern Alaska, and 0.15 between coastal British Columbia and POW (p. 19). Assignment tests suggest that two clusters (i.e., management units;

coastal British Columbia and southeastern Alaska) may be more appropriate than three clusters (p. 21).

In recent years, researchers began using SNPs and next-generation sequencing to understand population structure. SNPs are single base variants in DNA that can be used, like microsatellites, to evaluate differences among individuals, populations, and species. Von Holdt et al. (2011) analyzed SNPs of wolf-like species worldwide, including a few samples from coastal British Columbia. Although this study was well beyond the spatial extent of our assessment of Alexander Archipelago wolves, the authors found that wolves on the British Columbian coast formed a genetically distinct population when compared to wolves and their relatives globally (p. 1297, Supplemental Table S5).

Cronin et al. (2015a) led the first genetic study using SNP genotyping that focused on wolves from southeastern Alaska. Their sample included 138 individual wolves from southeastern Alaska (n=6 populations), 35 from British Columbia (although only one wolf from the coastal area; n=1 population), and 132 from continental North America (including eight wolves from New Mexico where *C. l. baileyi* occurs; n=6 populations); additional samples of coyote (*C. latrans*) and dog (*C. familiaris*) were included in the analysis for context (p. 3). Among wolf populations, pairwise F_{ST} values were the lowest comparing populations in Wyoming and Idaho (0.012) and highest comparing populations in New Mexico and GMU 2 (0.390; Supplemental Table 3 to Cronin et al. 2015a). Mean F_{ST} between southeastern Alaska and British Columbia was 0.120 (p. 7). Within southeastern Alaska, F_{ST} values between populations (by GMU) ranged between 0.034 (GMUs 1B and 3) and 0.281 (GMUs 1D and 2; Supplemental Table 3 to Cronin et al. 2015a). The authors stated that genetic variation observed among wolf populations in southeastern Alaska was equivalent to or surpassed variation between other populations in continental North America (p. 8). Generally, results of Cronin et al. (2015a) were similar to other studies of the Alexander Archipelago wolf described above, although interpretations of results differed; most notably, the authors determined that *C. l. ligoni* is not a valid subspecies of the gray wolf (see *Uncertainty of taxonomic status*).

We note that many other genetic studies, meta-analyses, and taxonomic interpretations of wolves in North America and beyond exist, but are not described above (e.g., Carmichael et al. 2007, 2008; Knowles 2010). For brevity, we included only the key genetic studies specific to the Alexander Archipelago wolf.

2.2.3. Other relevant analyses

In addition to morphological and genetic analyses, we reviewed ecological and behavioral traits of Alexander Archipelago wolves and compared them with those of other gray wolf populations. We found that coastal wolves exhibit uncommon behaviors such as feeding on salmon (*Oncorhynchus* spp.) and other marine prey when available (although see *Food habits*, and Adams et al. 2010) and occupying home ranges that include marine habitat, requiring them to swim regularly among islands in the archipelago (Szepanski et al. 1999, p. 331; Munoz-Fuentes et al. 2009, p. 10, and references therein; Stronen et al. 2014, p. 1; see *Ecology* for detailed descriptions). Additionally, researchers have hypothesized that coastal wolves may have evolved to resist disease associated with marine food resources, potentially restricting movements between inland and coastal habitats (Darimont et al. 2003, p. 352; Darimont et al. 2008, pp. 9–

10; also see *Disease*). These traits alone probably would not meet taxonomic standards, but when combined with morphological and genetic analyses they serve as supporting evidence that wolves occupying southeastern Alaska and coastal British Columbia form a distinct group of populations with unusual and shared characteristics.

2.2.4. Uncertainty of taxonomic status

In this assessment, we assume that *C. l. ligoni* is a valid subspecies, although we acknowledge uncertainty associated with this assumption. We emphasize here that uncertainty in subspecies designations nearly always exists, largely because we lack a universally accepted subspecies definition (Haig et al. 2006, p. 1586). For *C. l. ligoni*, we found this to be the case with most of the uncertainty stemming from different interpretations of the same or similar data.

Cronin et al. (2015a, p. 9) concluded that wolves in southeastern Alaska do not comprise a genetically homogenous group and are not genetically isolated from other gray wolf populations and therefore do not qualify as a subspecies. This conclusion was challenged by Weckworth et al. (2015, p. 2) who argued that subspecies should not be defined on the basis of complete reproductive isolation; instead, subspecies should be viewed as groups of populations that are distinguishable and restricted to a geographic region where characters could overlap and interbreeding with adjoining subspecies may occur to a small degree. Weckworth et al. (2015, p. 2) stated that regardless of whether *C. l. ligoni* was recognized as a subspecies or not, a large set of characters (morphological, behavioral, and ecological) and genetic traits demonstrate that coastal wolves are distinctive from interior continental wolves, and that coastal wolf populations harbor a large amount of genetic variation of remaining North American wolf populations. Cronin et al. (2015b, pp. 2–3) responded to Weckworth et al. (2015) defending their conclusion that too much genetic variation exists among wolf populations in southeastern Alaska and between other wolf populations to justify a coastal subspecies of gray wolf.

Similarly, in the most recent meta-analysis of taxonomy of North American wolves, Chambers et al. (2012) proposed five subspecies of gray wolf not including *C. l. ligoni*. Instead, the authors grouped wolves in southeastern Alaska and coastal British Columbia with wolf populations in central and western United States, *C. l. nubilus* (pp. 9, 40–41). Their reasoning was that wolves in southeastern Alaska and coastal British Columbia had haplotypes both unique to the region and shared with historical samples from wolves in Kansas, Nebraska, and the western United States (p. 41). The authors then hypothesized that coastal wolves were a northward extension of *C. l. nubilus* prior to extirpation of that subspecies in inland portions of the western United States (pp. 41–42). Chambers et al. (2012, p. 41) postulated that the large proportion of unique, and apparently extinct, haplotypes in the historical sample of *C. l. nubilus* (Leonard et al. 2005, pp. 13–15) likely exaggerated the measure of divergence between the coastal populations and historical inland *C. l. nubilus* (Munoz-Fuentes et al. 2009, p. 9). The grouping of coastal wolves with *C. l. nubilus* has been contested by several recognized experts that believe wolves in southeastern Alaska and coastal British Columbia are ecologically and genetically distinct and warrant recognition as a distinct group (NCEAS 2014, pp. 10, 14, 17, 47–49, 61). The basis of these contentions generally lies with the lack of a universally accepted definition of subspecies and the history of gray wolf populations in North America. An evaluation of the taxonomy proposed by Chambers et al. (2012) and the arguments against it (detailed in NCEAS 2014) is beyond the scope of this document, yet we acknowledge the uncertainty and continued scientific debate associated with the subspecific designation of wolves in southeastern Alaska and coastal British Columbia.

In summary, we recognize that the science informing taxonomy of wolves in North America is evolving and that some researchers have conflicting opinions, particularly on subspecific designations. However, for the purpose of this assessment, we believe that persuasive evidence exists suggesting that wolves in southeastern Alaska and coastal British Columbia are an ecological and genetic unit worthy of analysis under the Act.

2.3. Distribution

The Alexander Archipelago wolf occurs along the mainland of southeastern Alaska and coastal British Columbia west of the Coast Mountains and on larger islands except Admiralty, Baranof, and Chichagof islands and all of the Haida Gwaii, or Queen Charlotte Islands (Person et al. 1996, p. 1; BCMO 2014, p. 14; Figure 1). Its range is approximately 219,101 km^2 (Appendix I), stretching roughly 1,500 km in length and 250 km in width. The northern, eastern, and southern boundaries of its range are porous and therefore are not defined sharply or with certainty.

In southeastern Alaska, wolves occur throughout the mainland and on most of the islands south of Frederick Sound (GMUs 1, 2, 3, and 5A; Figure 4), excluding Coronation, Forrester, and the smaller, more isolated islands without an adequate prey base (Person et al. 1996, p. 1; MacDonald and Cook 2007, p. 71; Figure 2). Only the largest islands such as POW, Kuiu, Kupreanof, Mitkof, Etolin, Revillagigedo, Kosciusko, and Dall islands likely support wolves consistently over time (Person et al. 1996, p. 1); for example, within GMU 2, only the three largest islands (POW, Kosciusko, and Dall; Figure 2) are known to have been continuously occupied by wolves for more than 20 years (Person and Ingle 1995, p. 10). Wolves were experimentally introduced to Coronation Island in 1960 and 1963, but died out by the early 1970s, presumably due to starvation (Klein 1995, p. 280; Paul 2009, p. 112). MacDonald and Cook (2007, p. 71) report no substantiated records of wolves from any of the islands north of Fredrick Sound, although ADFG has records of them on Douglas Island near Juneau and Sullivan Island near Haines (ADFG 2015a, p. 2). On the mainland, the distribution of wolves probably is limited by icefields and high-elevation rugged terrain, even though they use these habitats occasionally (ADFG 2015a, p. 2); in addition, wolves occur within the six primary river drainages (Alsek, Chilkat, Taku, Whiting, Stikine, and Unuk rivers; Figure 2) that penetrate the Coast Mountains connecting interior British Columbia and southeastern Alaska. Thus, we expect that these areas serve as intergradation zones between the Alexander Archipelago wolf and its continental counterpart.

In coastal British Columbia, wolves occur continuously along the mainland and on all islands (Darimont and Paquet 2002, p. 418; Figure 1), including Vancouver Island where they possibly were extirpated between 1950 and 1970 (Munoz-Fuentes et al. 2010, pp. 547–548). On the mainland, wolves generally are restricted to a narrow coastal zone, but also occupy the few major river systems that connect interior and coastal British Columbia such as the Nass, Skeena, Dean, and Fraser rivers (Figure 1). Alexander Archipelago wolf populations south of the Dean River probably intermix more regularly with other gray wolves than their northern counterparts and therefore the southern part of coastal British Columbia likely is an intergradation zone (Figure 1; Weckworth et al. 2011, pp. 2, 4).

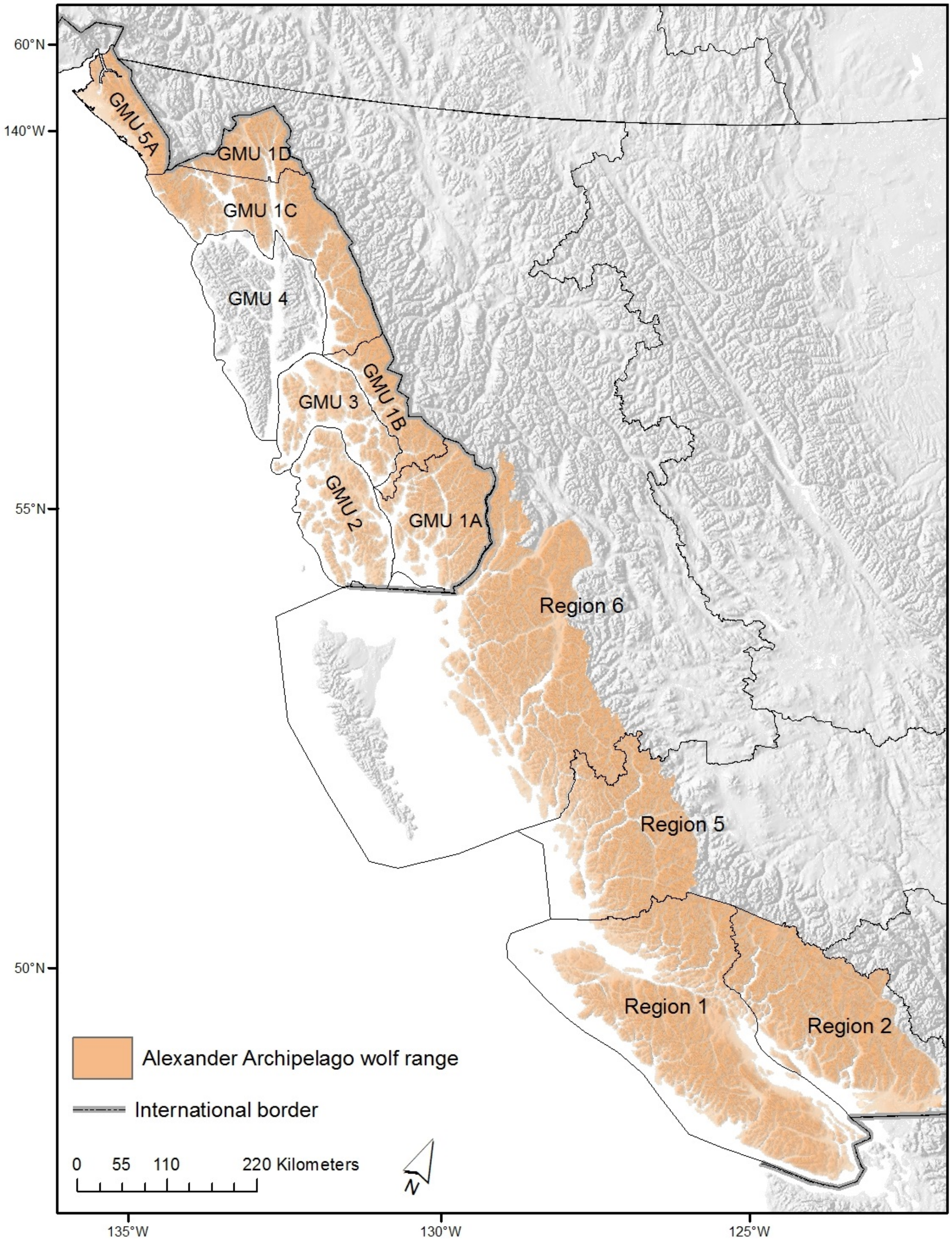

Figure 4. Game Management Unit (GMU) boundaries in southeastern Alaska and Region boundaries in British Columbia that are within the assumed range of the Alexander Archipelago wolf. These boundaries were used in this assessment to define a population of wolves.

CHAPTER 3: LIFE HISTORY AND ECOLOGY (INDIVIDUAL AND WITHIN-POPULATION LEVELS)

For this assessment, we consider a population to be a collection of individuals of a species in a defined area; the individuals in a population may or may not breed with other groups of that species in other places (Mills 2013, p. 3). Therefore, we defined populations of wolves in southeastern Alaska by GMU (including subunits) and in British Columbia by Region because these are defined areas and wolf populations are managed at these spatial scales (Figure 4). For example, GMU 2 comprises one population of wolves on POW and adjacent islands. In this chapter, we describe the life history and ecology of wolves at the individual and within-population levels; see *Chapter 4* for information on dynamics of multiple populations.

3.1. Vital rates

3.1.1. Abundance and trend

Estimating wolf abundance and densities in the temperate rainforests of southeastern Alaska and British Columbia is challenging. Researchers have attempted to generate population estimates using direct methods such as radio-collaring wolves (e.g., Person 2001, pp. 33, 55–70), non-invasive methods such as genetic analysis of hair samples (e.g., ADFG 2014, p. 1), and habitat- or prey-based methods (e.g., BCMO 2014, pp. 5–6). Additionally, managers typically use local knowledge and anecdotal information on wolves in their areas to estimate population size, although these estimates are qualitative and their accuracy is unknown. See Fuller et al. (2003, pp. 165–169) and Boitani (2003, pp. 322–323) for population, density and trend estimates of gray wolf populations worldwide.

Southeastern Alaska.—In southeastern Alaska, the only field-derived, empirical population estimates for wolves exist for POW and the surrounding islands (i.e., GMU 2) where an intensive field study has been conducted almost continuously since the early 1990s. Person and Ingle (1995, p. 11) generated the first estimate by locating Very High Frequency (VHF) radio-collared wolves regularly to determine number of packs, pack size and home range size; they estimated the wolf population on POW and neighboring Kosciusko Island to be 321 (SE=135) during fall 1994, and 199 (SE=111) during spring 1995. Person et al. (1996, p. 11) used a different analytical approach with the same data to estimate the wolf population on POW and Kosciusko as 217 (SE=65) in fall 1994. The authors concluded that the average of the two fall estimates (N=269, SE=80) probably was the “best” wolf population estimate for the two islands (p. 11) and, when extrapolated to nearly all of GMU 2 (8,510 km^2), the population was estimated to be 336 (SE=100) wolves in fall 1994 (p. 12). As part of the same study and using similar methods, in fall 2003, the wolf population in nearly all of GMU 2 was estimated as 326 wolves (SE=75; ADFG 2009, p. 32). Thus, the accepted population estimates for nearly all of GMU 2 were 336 (SE=100) wolves for fall 1994 and 326 (SE=75) wolves for fall 2003.

A decade later with more laboratory and field techniques to study wildlife, researchers developed and tested new methods for estimating wolf abundance on a portion of POW. The goal was to identify a more efficient, cost effective approach compared to radio-telemetry that could be applied broadly in GMU 2 as well as the rest of southeastern Alaska. To allow for reliable comparisons with previous efforts, researchers aimed to radio-collar wolves again, but with downloadable Global Positioning System (GPS) collars as opposed to VHF collars, and to use hair snare traps as a method for collecting wolf hair for genetic analyses. Following a pilot year in 2012, the wolf population in all of GMU 2 (9,069 km^2) was estimated using genetic capture-recapture methods from a sample on a portion of POW to be 221 (95% CI=130–378) wolves in fall 2013 (ADFG 2014, p. 2) and 89 (95% CI=50–159) in fall 2014 (ADFG 2015b, p. 2). Between 2012 and May 2015, too few wolves (n=12) were captured for radio-collaring to permit useful comparisons between techniques (ADFG 2015a, p. 4; ADFG 2015c, p. 10).

Comparison of wolf density and population estimates for GMU 2 across years is possible, but not straight-forward. First, different techniques were used to derive the four available density estimates. Both VHF telemetry and genetic capture-recapture have associated assumptions and potential sources of bias associated with them (e.g., sightability bias, independence of encounters). Second, field efforts in all four years occurred on a portion of POW only and, although the same general study area (i.e., north central POW) was used, the size of the actual study area varied (Table 3), but always overlapped. Third, density estimates initially were not extrapolated to the same area of GMU 2; the 1994 and 2003 estimates were based on an area of 8,510 km^2 and estimates in 2013 and 2014 were based on an area of 9,025 km^2. Recently, researchers performed spatial conversions to allow for comparisons of density and population estimates across years, assuming that the individual point estimates are statistically valid and accurate (Table 3; ADFG 2014, pp. 2–4). When extrapolating density estimates uniformly, it also is necessary to assume that differences in habitat, prey densities, territoriality, and other factors known to influence wolf abundance are the same or reflect the average (i.e., are not disproportionate) across GMU 2. Acknowledging all of these assumptions and caveats, the GMU 2 wolf population declined by 75% (SE=15), or 6.7% (SE=2.8) per annum, between 1994 and 2014, although the confidence intervals of the four point estimates overlap; the steepest decline occurring over a single year (2013–2014) when the population was reduced by 60% (Table 3). We note here that even the lowest density estimate of Alexander Archipelago wolf in GMU 2 (9.9 wolves/1,000 km^2) is not particularly low by most standards for Alaska or elsewhere in North America (Fuller et al. 2003, pp. 167–169).

Table 3. Estimated fall densities of wolves within overlapping study areas of varying size across years and extrapolated population estimates and associated standard errors for Game Management Unit (GMU) 2 (size=9,025 km^2).

Year	Primary estimation method	POW study area		GMU 2 population	
		Size (km^2)	Density (wolves/1,000 km^2)	Estimate	95% confidence interval
1994[1,2]	VHF telemetry	4,014[3]	39.5	356	148–564[4]
2003[2,5]	VHF telemetry	unknown	38.0	345	190–500[4]
2013[6]	genetic capture-recapture	1,683	24.5	221	130–378
2014[6]	genetic capture-recapture	3,280	9.9	89	50–159

[1]Person et al. 1996, pp. 11–12; [2]ADFG 2014, pp. 2–4; [3]As described in Person 2001, p. 31, but size of trap area unknown; [4]Estimates derived from reported standard errors (reported in text) to be consistent with 2013 and 2014; [5]ADFG 2009, p. 32; [6]ADFG 2015b, pp. 1–2.

Outside of GMU 2, few quantitative data on population size, trend, and densities from field studies are available. ADFG Management Reports summarize information from biologists, trappers, hunters, and other resource users, although these reports are based on observations only. With that caveat stated clearly, in the most recent ADFG Management Report (spanning 2008–2011), ADFG (2012, pp. 10, 41) stated that 45–85 wolves in eight packs occupied GMU 1B and 125–385 wolves in 23 packs occupied GMU 3; wolf populations in both of these GMUs are thought to be at higher densities than the early 1990s because of increased moose (*Alces americanus*; GMU 3) and Sitka black-tailed deer (*Odocoileus hemionus sitkensis*; GMU 1B) numbers. In GMUs 1A and 1D, wolf populations apparently are stable (ADFG 2012, pp. 2, 24), and in GMUs 1C and 5A, they are thought to be increasing probably because of increasing moose abundance (ADFG 2012, pp. 18, 48). We reiterate here that these assessments are based on anecdotal information, not survey data or scientific investigations.

Kirchhoff (1994a) summarized density estimates by GMU, but their accuracy is unknown. He reported that Alexander Archipelago wolves apparently attain their highest densities in the southern portion of southeastern Alaska in GMUs 1A and 2, including POW, Revillagigedo Island, and the Cleveland Peninsula, with 16–22 wolves/1,000 km^2 (p. 5); this estimate was lower than that empirically estimated at a similar time for POW and Kosciusko Island (39.5 wolves/1,000 km^2; Table 3), which was comparable with densities for other wolf populations where deer are the primary prey (see Table 1 in Person et al. 1996, p. 13). Kirchhoff (1994a, p. 5) estimated wolf densities on Kuiu, Kupreanof, Mitkof, Wrangell, Zarembo, Etolin, and associated islands (i.e., GMU 3) to be 8 wolves/1,000 km^2, and to be the least on the mainland of southeastern Alaska (5 wolves/1,000 km^2).

We are aware of only one effort to estimate the size of the wolf population as a whole in southeastern Alaska. Using a model linking wolf abundance to habitat capability for deer and other prey (moose, mountain goat [*Oreamnos americanus*]), Suring et al. (1993) estimated that wolves in GMU 2 represent about 37% of the total wolf population in southeastern Alaska, followed by GMU 1 (33%), GMU 3 (28%), and GMU 5A (2%); presented in Person et al. 1996, p. 13). Based on the GMU 2 proportion estimate and their own empirically derived estimates of wolf population size on POW, Person et al. (1996, p. 12) estimated the fall 1994 population in southeastern Alaska to be 908 (SE=216) individuals. The authors noted that the large standard error of this estimate was an underestimate of the true error because the model used to estimate the number of wolves from habitat capability for deer provided no estimate of variance. Using this population estimate for southeastern Alaska, we estimated the number of wolves in each GMU based on the allocations from Suring et al. (1993; Table 4) and applied the overall coefficient of variation to individual estimates to calculate variance. We acknowledge the uncertainty associated with these estimates, but to the best of our knowledge, they represent the only wolf population estimates available for all GMUs except GMU 2 (see Table 3), which we included here for comparison and completeness. We urge caution in interpreting these numbers as absolute because they are based on outdated habitat capability of prey and do not take other factors into account (e.g., wolf harvest and density, territoriality; e.g., Cubaynes et al. 2014).

Table 4. Estimated wolf population size by Game Management Unit (GMU) and subunits (gray shaded area) derived from habitat capability models of deer, moose, and mountain goat developed in the early 1990s, southeastern Alaska (Suring et al. 1993; as presented in Person et al. 1996, p. 13); GMU-specific estimates were based on a total estimate of 908 wolves (SE=216) in southeastern Alaska (Person et al. 1996, p. 12).

GMU	Percent of southeastern Alaska wolf population	Derived population estimate	95% confidence interval	
			Lower	Upper
1 (all)	33	300	160	439
1A	20	182	97	266
1B	8	73	39	107
1C/1D	5	45	24	67
2[1]	37	336	179	493
3	28	254	136	373
5A	2	18	10	27

[1] More recent field-derived estimate available; see Table 3.

British Columbia.—We found only one population estimate specific to coastal wolves in British Columbia, although it is more than 15 years old. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2001, p. 38) estimated that roughly 2,200 wolves occupy the Pacific Ecological Area. However, we found two more recent population estimates for wolves in the entire province of British Columbia, which includes both coastal (*C. l. ligoni*) and interior wolves (*C. l. nubilus*). Based on prey densities (moose, deer) and local knowledge, low and high density areas of wolves were delineated throughout British Columbia (BCMO 2014, p. 5). Then, using density estimates of wolves in other parts of Canada and neighboring Alaska, low density areas were assigned a range of 2–5 wolves/1,000 km^2 and high density areas were estimated to have 5–15 wolves/1,000 km^2 (pp. 5–6). When extrapolated, the provincial population of wolves was estimated as 4,700–11,400 individuals (p. 7). Kuzyk and Hatter (2014) applied a more empirical approach that used ungulate biomass to estimate abundance of wolves in British Columbia. Regional ungulate population surveys were used to estimate biomass, which then was included in a regression model to predict wolf abundance for 2000, 2003, 2008, and 2011 (p. 879). The most recent provincial population estimate was 8,688 (95% CI=5,898–11,760) wolves, indicating a slight but consistent trend upward since 2000 when the estimate was 7,213 (95% CI=4,977–9,696) wolves (p. 881). Because these estimates are more recent, we used them in our assessment after making adjustments to reflect the coastal population of wolves only (i.e., Alexander Archipelago wolves).

By multiplying regional wolf population estimates by the proportion of the region that fell within the range of the Alexander Archipelago wolf, we generated population estimates for coastal British Columbia (Table 5). Using the wolf density approach (BCMO 2014), we estimated that 691–1,688 wolves occupy coastal British Columbia and, using results of Kuzyk and Hatter (2014), we calculated a mean population estimate of 875 wolves (range=597–1,183). We suspect that the latter estimate may be biased slightly high because wolves on the coast primarily eat deer and to a lesser extent mountain goats, which have lower biomass values (0.75 and 1, respectively) compared to moose (biomass value=6), the primary prey item of wolves in interior

British Columbia (Darimont et al. 2004, p. 1871; Kuzyk and Hatter 2014, p. 880). Though, moose are expanding their range into coastal British Columbia (Darimont et al. 2005, p. 235) and have been detected in wolf scats found on the coastal mainland and nearby islands (Darimont et al. 2004, p. 1871). Nonetheless, we urge caution in interpreting these numbers as absolute values, but present them here as general estimates of the size of the wolf population in coastal British Columbia.

Table 5. Estimated population size of the Alexander Archipelago wolf by Region based on estimates of wolf density (BCMO 2014) and ungulate biomass (Kuzyk and Hatter 2014) and adjusted by the proportion of the Region in the coastal portion of British Columbia.

	Proportion of Region along coast	Ungulate biomass method			Wolf density method	
		Mean	Low	High	Low	High
Region 1	1.00	307	218	404	150	480
Region 2	0.83	123	83	170	62	166
Region 5	0.22	200	141	265	94	272
Region 6	0.17	244	156	344	385	770
All of coastal British Columbia		875	597	1,183	691	1,688

Methods to estimate wolf abundance in British Columbia have changed over time and therefore no trend is evident from these estimates (BCMO 2014, p. 25). However, provincial managers believe that wolf populations are stable or increasing based on local knowledge and increased harvest by trappers and hunters; also, they found no contrary evidence of a population decline or significant conservation threat (BCMO 2014, p. 25). Similarly, as stated above, Kuzyk and Hatter (2014, p. 881) reported that provincial wolf populations have increased slightly between 2000 and 2011 using ungulate biomass as an index; in Regions 1, 2, 5, and 6, the same trend has been observed since 2000 (BCMO 2015a). Thus, collectively these findings indicate that the Alexander Archipelago wolf populations in coastal British Columbia are stable or slightly increasing.

Rangewide.—Using the most recent and best available wolf population estimates in southeastern Alaska and coastal British Columbia, we estimate a rangewide population of about 850–2,700 wolves (Table 6). Given the uncertainty in terms of both precision and accuracy associated with the individual estimates, we present an estimated range, which generally corresponds to the estimated 95% confidence limits, for populations or groups of populations within a specific geographical areas. Using the midpoints of these estimates, approximately 62% of the rangewide population of the Alexander Archipelago wolf occurs in coastal British Columbia and 38% inhabits southeastern Alaska.

Table 6. Estimated rangewide population size of the Alexander Archipelago wolf based on the most recent and best available population estimates in Game Management Units (GMU) of southeastern Alaska and Regions in coastal British Columbia. See text for an explanation of derivation, assumptions, and caveats.

Area	Populations	Low	High	Percent of rangewide population based on midpoint
Mainland of southeastern Alaska	GMUs 1 and 5A	170	466	18%
Prince of Wales and surrounding islands[1]	GMU 2	50	159	6%
Middle islands of southeastern Alaska	GMU 3	136	373	14%
Northern coastal British Columbia	Regions 5 and 6	297	1,043	38%
Southern coastal British Columbia	Regions 1 and 2	212	646	24%
Total	All	865	2,687	100%

[1] ADFG 2015b, pp. 1–2

3.1.2. Reproduction

Most wolf packs contain a pair of breeding adults plus other adults that may or may not breed (see *Social organization*). Age of first breeding of the Alexander Archipelago wolf is about 22 to 34 months (Person et al. 1996, p. 8). Sizes of litters range from 1 to 8 pups with an average of 4.1 pups (SD=1.7); new mothers produce fewer pups than older, more-experienced mothers (Person and Russell 2009, p. 216). Although uncommon, some wolf packs fail to exhibit denning behavior or produce litters and, multiple litters within packs of Alexander Archipelago wolves has not been documented (Person and Russell 2009, p. 216). See Fuller et al. (2003, p. 177) for data on litter size and percentage of pups in packs from other gray wolf populations.

Alexander Archipelago wolves use dens from mid-April through early July with peak activity between early May and the third week of June (Person 2001, p. 61; see *Denning* below for habitat description). After early July, most dens are abandoned and pups are located to rendezvous sites typically <1 km from the natal den where they remain until October (Person and Russell 2009, p. 216). At this time, the pups typically are full size, although weigh less than a yearling or adult, and begin traveling with the pack; most disperse the following spring as yearlings.

3.1.3. Survival

Only one study has estimated survival rates of Alexander Archipelago wolves. Based on radio-collared wolves on a portion of POW and adjacent islands (GMU 2), Person and Russell (2008, p. 1545) reported mean annual survival rate of wolves greater than four months old as 0.54 (SE=0.17); survival was lower between 1993 and 1995 (0.45, SE=0.17) than between 1999 and 2004 (0.62, SE=0.16). Survival did not differ between age classes or sexes, but was higher for resident wolves (0.65, SE=0.17) compared to non-residents (i.e., wolves not associated with a pack; 0.34, SE=0.17). None of the non-resident wolves lived more than 86 weeks after radio-collaring or dispersal unless they settled (19% of all non-residents). See Fuller et al. (2003, pp. 176, 179) for annual survival rates of wolves elsewhere in North America.

Of the 55 radio-collared wolves in GMU 2 between 1993 and 2004, 39 of them died from legal harvest by hunters and trappers (n=18), unreported harvest (n=16), and natural causes (n=5; Person and Russell 2008, p. 1545). Thus, 87% of wolves that died during the study were killed by humans. Between 2012 and May 2015, researchers on POW collared 12 additional wolves as part of a study to estimate population size (see *Abundance and trend*); of these wolves, five died from legal harvest, three from unreported harvest, and one from natural causes (ADFG 2015a, p. 4). In addition, one collared wolf has an unknown fate due to technical difficulty (i.e., premature collar release), one has an unknown fate because the collar has not been retrieved yet, and one wolf is alive still (ADFG 2015c, p. 10; ADFG 2015d, p. 2). Thus, during the 3-year period of this ongoing study, 67% of the collared wolves were killed by humans. Both of these studies that involved radio-collaring wolves in GMU 2 took place in a portion of GMU 2 that is roaded and therefore has higher levels of human use compared to unroaded portions, which may have inflated mortality rates. See *Chapter 5* (specifically, *Wolf harvest* and *Road development*) for further description of correlates of mortality of Alexander Archipelago wolves. To the best of our knowledge, estimates of annual survival or mortality of wolves on other islands or the mainland of southeastern Alaska and coastal British Columbia are not available.

3.1.4. Within-population dispersal

Pups that survive to adulthood either remain in their natal pack or disperse (Person et al. 1996, p. 10), here defined as permanent movement of an individual away from its pack of origin. Dispersers typically search for a new pack to join or associate with other wolves and ultimately form a new pack in vacant territories or in vacant areas adjacent to established territories. Hence, dispersal is a critical element of wolf ecology and social biology.

Wolves are capable of dispersing long distances, sometimes hundreds of kilometers (e.g., Fritts 1983, p. 166; Ballard et al. 1987, p. 20; Adams et al. 2008, pp. 10–11), and can quickly re-occupy vacant territories (e.g., Bergerud and Elliot 1986, pp. 1519–1523). Generally, young wolves are more likely to disperse than older ones (e.g., Adams et al. 2008, p. 11) and males are more likely to disperse than females, although females may disperse farther (Ballard et al. 1987, p. 20). Successful dispersal often is short in duration because dispersing wolves are more vulnerable than non-dispersers to hunting and trapping and being killed by other wolves (e.g., Peterson et al. 1984, p. 29).

Within-population dispersal metrics for Alexander Archipelago wolves are available only for GMU 2 where the annual rate of dispersal was 39% (95% CI= ± 23%, n=18) with adults greater than two years of age composing 79% of all dispersers (Person and Ingle 1995, p. 20). Annual rate of survival of dispersing wolves was low (0.16) with most killed by hunters and trappers before settling (Person and Russell 2008, p. 1547); therefore, successful dispersal may be more limited by low survival rates than by actual dispersal capability. Nonetheless, minimum dispersal distances from the point of capture ranged between 13 and 182 km. Two of three wolves captured and radio-collared on Kosciusko Island dispersed long distances; one was located subsequently on the southern end of Dall Island, a minimum distance of 182 km that required at least two swims greater than 350 m each, and the other moved at least 160 km to the south end of

POW (Person and Ingle 1995, p. 23). To the best of our knowledge, these dispersal events are the longest distances documented for the Alexander Archipelago wolf. Stronen et al. (2014) postulated that wolves prefer to disperse to ecological environments similar to their natal habitat. See Fuller et al. (2003, p. 179) for age-specific dispersal rates of gray wolves in North America.

Of considerable relevance to the conservation biology of the Alexander Archipelago wolf is their ability to disperse over water barriers. Wolves introduced to Coronation Island (Figure 2) ultimately starved, yet did not swim 900 m to nearby habitat with abundant food (Klein 1995, p. 280). Person and Ingle (1995, p. 23) reported that none of the dispersing wolves studied by them (n=13) swam to other islands greater than 1 km from POW or dispersed across Clarence or Sumner Straits, which separates POW from other islands in the archipelago and from the mainland (Figure 2). Similarly, none of the wolves on POW that were collared recently with GPS technology, which provides finer temporal resolution of location, has swum more than 1 km (ADFG 2015d, p. 2).

Nevertheless, some evidence exists demonstrating that Alexander Archipelago wolves are capable of swimming large distances, although success probably depends on local water conditions (e.g., tidal current and strength). For example, Darimont and Paquet (2002, p. 418) found wolves on isolated islands in coastal British Columbia that were 5–13 km from other large landmasses. In addition, of three wolves that were radio-collared opportunistically on or near Kupreanof Island in 1999, one was relocated on 17 occasions, all confined to Kupreanof Island, but another was trapped and killed nearly three years later on Revillagigedo Island, roughly 134 km straight-line distance from the capture location (USFS 2015a). Although we do not know the travel route of the dispersing wolf, we know that at some point that wolf must have made at least four water crossings with the shortest being about 2 km in distance. No data on the third wolf were available. More recently, ADFG (2015d, p. 2) photographed a wolf on Shrubby Island, a small island located between POW (~3.5 km) and Zarembo islands (~1.2 km via Bushy Island). These photographs suggest that wolves potentially have the opportunity to move to and from POW if tidal currents and sea state allows, although we do not know the frequency of these movements (see *Connectivity*).

3.1.5. Sex ratio

We found no information describing sex ratios in litters of Alexander Archipelago wolves. Peterson et al. (1984, p. 26) reported a statistically significant male bias in pups captured as part of their study (22 males:10 females), although hunter-killed pups had a female bias (23 males:39 females). In Minnesota, Mech (1975, p. 738) reported a range of pup sex ratios in Minnesota and hypothesized that male-biased litters were more common in saturated, high density wolf populations; otherwise, sex ratio favors females. We found that sex ratio of pups often is not reported in the literature, even though it is an important demographic parameter when considering future population size.

As part of a capture-recapture study of Alexander Archipelago wolves employing genetic markers in a portion of GMU 2, the proportion of females captured via hair board sampling was 0.57 (SE=0.13) in fall 2013 (ADFG 2015b, p. 2). One year later in fall 2014, the proportion was 0.25 (SE=0.11; ADFG 2015b, p. 2), coincident with an observed decrease in abundance (Table 3).

3.2. Social organization

Wolves are social animals that live in packs usually composed of one breeding pair (i.e., alpha male and female) plus offspring of 1–2 years old. Generally the adult parents guide activities of the group with the female leading pup care and defense and the male taking charge of foraging and food provisioning (Mech 1999, p. 1196). Occasionally, unrelated or related wolves are adopted into the pack, but usually the pack functions as a family or a small group of families. The pack is a year-round unit, although all members of a wolf pack rarely are observed together, except during winter (Person et al. 1996, p. 7).

Loss of alpha members of a pack can result in social disruption and unstable pack dynamics. During this time, dominance relationships within and among packs and individuals are re-established, which may lead to higher rates of intraspecific strife and possibly plural breeding, although this is rare (Mech 1999, p. 1200; Packard 2003, pp. 52–56). Pack dynamics are complex and shift frequently as individuals age and gain dominance, disperse from, establish or join existing packs, breed and die (Mech 1999, pp. 1197–1202). The social and reproductive fates of individuals are based mostly on the opportunities presented by these shifting dynamics (Packard 2003, p. 35). Although loss of breeding individuals impacts social stability within the pack, at the population level wolves appear to be resilient enough to compensate for any negative impacts to population growth (Borg et al. 2015, p. 183).

Pack sizes, especially in southeastern Alaska, are difficult to estimate because of heavy vegetation cover. Nonetheless, data are available from three studies. Smith et al. (1987, pp. 4–7) found that pack sizes on Revillagigedo Island range from 2 to 12 wolves and averaged 5.4 wolves (time of year not specified consistently). On POW and Kosciusko islands during the mid-1990s, fall pack size ranged from 2 to 12 wolves, but averaged 7–9 wolves (Person et al. 1996, p. 7). More recently on POW, ADFG (2015d, p. 2) reported similar pack sizes with an average of 7.6 wolves in the fall (SE=0.8, range=1–16) and 4.0 wolves in the spring (SE=0.7, range=1–15). We are not aware of any similar counts for wolf packs in coastal British Columbia. For context, Adams et al. (2008, p. 8) observed mean pack sizes of 6.7 to 9.3 in autumn and 4.3 to 7.1 in spring with packs less than or equal to 17 wolves in the Central Brooks Range, Alaska, and Peterson et al. (1984, p. 25) reported mean pack size in early winter as 11.2 wolves. See Fuller et al. (2003, pp. 165–166, 172–174) for ranges of pack size of wolves in North America.

3.3. Ecology

Most recent information on the ecology of the Alexander Archipelago wolf was generated as part of two primary studies with one centered on the northern portion of POW (early 1990s–mid 2000s, 2012 to present) and the other on the central and northern coasts of British Columbia (early 2000s to present, but more intermittent than POW). Researchers have studied wolves on POW primarily by capturing, radio-collaring and re-locating them throughout the year (e.g., Person 2001); in more recent years, non-invasive sampling (e.g., remote camera systems, hair snares) in addition to more traditional radio-collaring methods has been used, as part of a study to test methods for estimating abundance, to learn about distribution, movements, and space use (e.g., ADFG 2015b). In contrast, researchers in coastal British Columbia have conducted surveys for wolves (or sign of wolves) and studied their ecology using non-invasive methods (e.g., scat analysis; e.g., Darimont and Paquet 2002); to the best of our knowledge no wolves have been radio-collared in coastal British Columbia.

3.3.1. Food habits

Alexander Archipelago wolves are opportunistic predators that eat a variety of prey species, although, like gray wolves (Mech and Peterson 2003, p. 131), ungulates compose most of their diet. One of the apparently unusual aspects of the Alexander Archipelago wolf diet is the seasonal consumption of salmon (15–20% of lifetime diet; Szepanski et al. 1999, p. 327). However, Adams et al. (2010, p. 251) found that inland wolves in Denali National Park, Alaska also ate salmon in slightly lower, but similar quantities (3–17% of lifetime diet) compared to Alexander Archipelago wolves, suggesting that salmon in wolf diet may be common where salmon are available. Further, gray wolves in southwestern Alaska also feed regularly on salmon and other marine mammals when available (Watts et al. 2010, p. 145), indicating that consumption of salmon (and other marine-derived prey) is not unique to the Alexander Archipelago wolf. Regardless, salmon provides a seasonal alternate food source to Alexander Archipelago wolves during a period of year with high food and energy demands (i.e., provisioning pups; Darimont et al. 2008, p. 5).

In this section, we report only the most comprehensive studies of Alexander Archipelago wolf diets that we found during our literature search, but recognize that other smaller studies have occurred (e.g., Garceau 1960). We also note that the biogeography of southeastern Alaska in particular presents difficulties when assessing the diet of wolves in this area because not all prey items are available on all islands (e.g., beaver do not occur on Coronation Island; MacDonald and Cook 2007, p. 27) and because prey species have been introduced to some islands (Table 7). For example, mountain goats were introduced successfully to Revillagigedo Island in 1983 and after several attempts elk (*Cervus canadensis*) are now established on Etolin and Zarembo islands with credible sightings on large nearby islands such as POW, Wrangell, Mitkof, and Kupreanof islands (MacDonald and Cook 2007, p. 188). Given the general importance of ungulates in the diet of wolves (Mech and Peterson 2003, p. 131), presence (or absence) of ungulate species on a specific island or group of islands is particularly relevant. See Peterson and Ciucci (2003, pp. 104–130) and Mech and Peterson (2003, pp. 131–160) for information on wolf food habitats throughout the world, which demonstrate the relationship between breadth of diet and availability of prey, especially ungulates.

Table 7. Generalized distribution of ungulate species by Game Management Unit (GMU) in southeastern Alaska and Region in coastal British Columbia within the range of the Alexander Archipelago wolf. Gray-shaded areas indicate presumed absence on a regular basis.

GMU or Region	Black-tailed deer	Moose	Mountain goat	Elk
	Odocoileus hemionus	*Alces americanus*	*Oreamnos americanus*	*Cervus canadensis*
GMU 1	Present	Present	Present	▨
GMU 2	Present	▨	▨	▨
GMU 3	Present	Present	▨	Present
GMU 5A	Present[1]	Present	Present	▨
Region 1	Present	Present[2]	Present	Present
Region 2	Present	Present[2]	Present	▨
Region 5	Present	Present[2]	Present	▨
Region 6	Present	Present[2]	Present	▨

[1] Small (<50 individuals) introduced population near the village of Yakutat only.

[2] Recent range expansion from inland to coastal habitats; populations are stable or increasing (Darimont et al. 2005, p. 235; BCMO 2015b).

<u>Scat analyses</u>.—A common method for describing wolf diet involves collection and analysis of scat, although results and inference from scat analyses require careful interpretation. Sampling design and protocol can have a strong influence on results, in part owing to the social organization and cooperative hunting by wolves. Additionally, results of scat analyses typically are reported as frequency of occurrence of prey items without accounting for differences in prey size and digestibility, which can be done by converting frequency of occurrence to biomass consumed (e.g., Floyd et al. 1978). Nonetheless, frequency of occurrence can be useful as a general description of wolf diet, but should be interpreted with caution in terms of quantifying it. Recognizing these caveats, in this section we summarize results of scat analyses conducted for the Alexander Archipelago wolf using frequency of occurrence primarily due to the nature of the available data.

The most detailed analyses of Alexander Archipelago wolf food habits based on scat were those by Fox and Streveler (1986), Smith et al. (1987), Milne et al. (1989), Merriam (summarized in Klein 1995), Kohira and Rexstad (1997), Darimont et al. (2004), and Lafferty et al. (2014). Generally, remains of deer had the highest frequency of occurrence in scats found in southeastern Alaska and coastal British Columbia with the exception of the northern mainland, including Glacier Bay where deer are scarce (Table 8); instead, in these areas remains of mountain goats or moose had the highest frequency of occurrence. Beaver (*Castor canadensis*) had the second highest frequency of occurrence in scats found on POW and Revillagigedo islands. Alexander Archipelago wolves also feed on harbor seals (*Phoca vitulina*), mustelids, small mammals, birds, marine invertebrates, and plants. Spawning salmon are taken during the summer and fall by some wolf packs in southeastern Alaska and coastal British Columbia. Thus, consistent with their opportunistic food habits and lack of specialization, Alexander Archipelago wolves seem to eat (or sample) most available prey items within a habitat type or area provided that the cost-to-benefit ratio requirements are met (e.g., prey availability, capture efficiency, nutritional gain, and potential for injury).

Table 8. Percent occurrence of prey remains in wolf scats from various locations in southeastern Alaska and coastal British Columbia (from north [left] to south [right]). Items of infrequent occurrence (<1%) in only one study and unidentified items not included.

Common name	Scientific name	Northern mainland[1]	Glacier Bay[2]	Prince of Wales Island[3]	Revillagigedo Island[4]	Coastal British Columbia[5]
Predominant time of year that wolf scat was collected		Summer	May–July	November–July	Year-round	Summer
Number of scats		78	55	182	329	595
Rodents						
Hoary marmot	*Marmota caligata*	27	0	0	0	0
American beaver	*Castor canadensis*	0	<1	31	20	3
Vole	*Microtus* spp.	0	9	0	0	0
North American porcupine	*Erethizon dorsatum*	0	7	0	0	0
Unidentified rodent	*Rodentia* spp.	10		0	0	1
Lagomorphs						
Snowshoe hare	*Lepus americanus*	0	7	0	0	0
Carnivores						
American black bear	*Ursus americanus*	0	0	8	<1	3
Harbor seal	*Phoca vitulina*	1	<1	0	0	<1
North American river otter	*Lontra canadensis*	0	<1	0	0	4
Pacific marten	*Martes caurina*	0	0	0	0	6
Ermine	*Mustela erminea*	0	0	0	0	6
American mink	*Neovison vison*	0	0	0	0	3
Unidentified mustelid	*Mustelidae* spp.	9	0	17	0	0
Hoofed mammals						
Moose	*Alces americanus*	0	80	0	0	2
Black-tailed deer	*Odocoileus hemionus*	0	0	90	74	63
Mountain goat	*Oreamnos americanus*	53	0	0	0	6
Other						
Marine invertebrates		0	0	0	0	4
Fish and shellfish		4	4	5	<1	8
Unidentified bird		6	11	0	1	6
Vegetation		0	0	0	0	2

[1] Fox and Streveler 1986, pp. 192–193; [2] Lafferty et al. 2014, p. 145; [3] Kohira and Rexstad 1997, pp. 429–430; [4] Smith et al. 1987, pp. 9–11,16; [5] Darimont et al. 2004, p. 1871.

Fox and Streveler (1986, pp. 192–193) examined 124 wolf scats collected between 1973 and 1980 (predominantly during the summer) from four mainland sites, most of which were in alpine, in the northern part of southeastern Alaska. Mountain goat was the most frequent item in the scats from all sites (Table 8). Deer remains were found in 21% of the scats from Chilkat and Herbert glaciers. At Dixon Harbor and Lituya Bay, marmots (*Marmota caligata*) were of secondary importance followed by mustelids and small rodents. The authors believed the low occurrence of deer in the scats reflected low deer density at their study sites.

Smith et al. (1987, pp. 9–11, 16) studied 511 scats from Revillagigedo Island, POW, and Dall Island and found deer in 70% of the scats, followed by beaver in 14% of the scats (Table 8). About 30% of the scats collected in summer contained remains of deer fawns. Beaver remains were about equally represented in the wolf scats during summer and winter. There was considerable variation in the frequency of occurrence of beaver remains in the scats from various packs. The authors observed that wolves fed on salmon during late summer but this use was not recorded in scat collections.

Milne et al. (1989, pp. 83–85) determined contents of 647 scats collected between 1983 and 1985 on northern Vancouver Island. Deer was the most frequent item found, followed by beaver, Roosevelt elk (*Cervus elaphus*), and red squirrel (*Tamiasciurus hudsonicus*). However, elk and deer contributed similarly to biomass of wolf diet. Not surprisingly, adult deer were more frequent in the diet during winter and fawns were more commonly observed in the diet during summer. We were unable to include results of this study in Table 8 because the authors reported relative occurrence instead of frequency of occurrence in their publication.

Klein (1995, p. 277) reported on the contents of 663 scats collected by Merriam on Coronation Island between 1961 and 1986 after wolves were experimentally introduced there in 1960. In the five years following introduction, deer were present in 78–97% of wolf scats, followed by harbor seals, which ranged from 8 to 53% occurrence (Table 9). As deer became less numerous on the island as a result of predation by the introduced wolves, birds, seals, marine invertebrates and small mammals constituted the major food remains in scats. With declining deer numbers, wolves even resorted to cannibalism. As deer declined on Coronation Island, the wolf population declined from a maximum of 13 to one individual. To the best of our knowledge, this study is the only one on Alexander Archipelago wolves that indicates an inability of wolves to maintain high densities in response to a declining deer herd.

Table 9. Percent occurrence by year of prey remains in wolf scats collected during spring and summer from Coronation Island between wolf introduction in 1960 and near extirpation in 1970 (only one wolf survived then; Klein 1995, p. 277).

	1961	1962	1963	1964	1965	1966[1]	1967	1968
Number of scats	146	18	45	77	213	110	44	3
Rodents								
Unidentified rodent	0	0	11	0	3	1	18	33
Carnivores								
Wolf	0	0	0	0	7	10	0	0
Seal	43	48	53	32	8	18	57	33
Hoofed mammals								
Deer	78	89	89	95	97	53	0	33
Other								
Marine invertebrates	1	0	2	0	3	28	23	33
Unidentified bird	2	0	0	8	5	30	25	0
Undetermined item	2	11[2]	27	14	17	66	5	0

[1] Scat collection in February shown; collection in August resulted in only seven samples.

[2] Not examined for bird, rodent, or marine invertebrates.

Kohira and Rexstad (1997, pp. 429–430) examined 182 wolf scats collected between autumn 1992 and summer 1994 on POW. They found deer remains in 90% of the scats and beaver in 31% of the scats; other prey items occurring less frequently included black bear, mink, river otter, small mammals, birds, and fish (Table 8). Significantly higher use of beaver occurred in the winter. Use of fish by wolves was distinctly higher in autumn.

Person et al. (1996, p. 8) combined data from POW (Kohira and Rexstad 1997) and Revillagigedo Island (Smith et al. 1987) and estimated the percentage of prey species by volume in the diet using a regression model. The authors estimated that 77% of the diet of wolves was composed of deer, 14% was beaver, and less than 10% was other prey species. They concluded that wolves inhabiting islands in southeastern Alaska depend on the availability of deer, although only one study (Klein 1995) has actually demonstrated this dependence.

Darimont et al. (2004, p. 1871) identified prey items in 595 wolf scats collected during summers 2000 and 2001 on the mainland and adjacent islands of northern and central coastal British Columbia. Deer was the most frequent item in the scats from the mainland (47%), but was even more frequently found in scats from the islands (75%). On the mainland, mountain goat was the second most frequent prey item (14%) followed by ermine (10%) and on the islands salmon (8%) and birds (7%) were of secondary importance compared to deer. The authors reported a large diversity of prey items (>17 species or groups of species), although deer comprised most of the diet in terms of frequency and biomass (Table 8).

Lafferty et al. (2014, pp. 143–145) studied 55 scats collected during early summer 2010 and 2011 in Glacier Bay National Park. Wolf scats contained on average 1.3 prey items (range=1–3) and by far, moose was the most frequent prey item found (80%; Table 8). This finding was contrary to a similar study conducted in 1993 and only 37 km away farther into the bay; Meikeljohn (1994, p. 8) reported that moose occurred in less than 3% of wolf scats and harbor seal was observed in 41% of wolf scats, although the latter species has declined considerably in recent years (Mathews and Pendelton 2006, p. 167). Lewis and Lafferty (2014, p. 8) also observed wolves feeding on a humpback whale carcass during summer (May–September) 2010 in Glacier Bay, indicating that at times wolves have access to substantial marine subsidies as part of their diet.

Stable isotope analyses.—Since 1997, several studies have assessed Alexander Archipelago wolf diet using stable isotope analyses. This method quantifies the relative proportions of identified food sources in wolf diet by measuring isotopic compositions of wolf tissues and comparing them to their prey. It is a useful technique because it allows for measurement of assimilated nutrients over time as opposed to scat analysis which reveals only an individual's last meal. The temporal inference of stable isotope results is related to turnover rates of the tissue analyzed. For example, hair contains isotopic values of food sources metabolized over a period of months (Darimont and Reimchen 2002, pp. 1640–1641) and bone tissue stores diet history over the individual's lifetime (Chrisholm et al. 1982, pp. 1131–1132). Additionally, stable isotope analysis avoids certain forms of bias inherent in scat analysis because of differences in prey size and digestibility; for example, fish may be more thoroughly digested by a wolf and therefore not detected easily in its scat (Szepanski et al. 1999, p. 328). However, evaluation of stable isotopes requires attention to a different set of assumptions and potential biases (e.g., Gannes et al. 1997, pp. 1271–1276).

Szepanski et al. (1999) was the first stable isotope study of Alexander Archipelago wolves. Using bone collagen of wolves sampled at three sites in southeastern Alaska (Kupreanof Island, POW, and mainland) as well as a sample of wolves from interior Alaska (for context), they found that wolf diet was similar at all three southeastern Alaskan sites. Deer accounted for 44.7–49.0% of the diet, other herbivores (moose, beaver, mountain goats, and voles) for 34.4–36.2%, and salmon ranged from 15.3–20.0% (p. 331). The relative contribution in the diet of wolves from interior Alaska was 9.1%; caribou (55.2%; *Rangifer tarandus*) and moose (35.7%) composed most of the interior wolf diet. Variation among individual wolf diets was higher for southeastern Alaska than for interior wolves (p. 327).

Darimont et al. (2007, 2009), Semmens et al. (2009), and others have used stable isotope analyses of wolf hair, coupled with scat analyses (Darimont et al. 2008) to describe diet of wolves on the northern and central coasts of British Columbia. By sampling for wolf hair before and after molting periods, the authors detected seasonal shifts from a deer-dominated diet in spring and summer to a more varied diet that included larger proportions of salmon in late summer and fall when salmon were available (Darimont et al. 2008, pp. 7–8). Darimont et al. (2009, p. 130) reported estimates of assimilated biomass (%) from three prey groups (deer, salmon, marine mammals) to the diet of wolves on the mainland and inner and outer islands of coastal British Columbia; deer composed most of the biomass of wolf diet on the mainland (82%) decreasing to the outer islands (32%), while both marine mammals and salmon had the opposite pattern, i.e., higher percentages of biomass on the outer islands (52 and 16%, respectively) decreasing to the mainland (20 and -2%). We note here that this finding contradicts that of Darimont et al. (2004, p. 1871) based on frequency of occurrence in scat: deer occurred more frequently in wolf scats on the islands compared to the mainland (see *Scat analyses*). Many of these studies apply stable isotope results to assess individual variation and niche width of populations of wolves in coastal British Columbia (e.g., Semmens et al. 2009, p. 4). We provide a broad overview of their work here because much of it is beyond the scope of this document.

3.3.2. Space and habitat use

Because wolves occupy a variety of habitats across their range in North America, they are considered to be a habitat generalist (Mech and Boitani 2003, p. xv). It is generally believed that the presence or absence of wolves in an area is a function of the availability of their prey and the intensity of human-caused mortality (Mech 1995, p. 273; Mladenoff et al. 1995, p. 286).

Home range size.—In southeastern Alaska, minimum convex polygon (mcp) home ranges of wolf packs on Revillagigedo Island averaged 279 km^2 and ranged from 79 to 447 km^2 (n=7; Smith et al. 1987, p. 15). In the mid-1990s on POW and Kosciusko Island, pack home ranges of VHF radio-collared wolves averaged 280 km^2 with a range of 101–419 km^2 (n=7, estimates based on average of 95% kernel and mcp); core areas (75% kernel and mcp) where wolf activity was concentrated were about 55–60% smaller than total home ranges (Person et al. 1996, p. 7).

In recent years, researchers have equipped wolves with downloadable GPS collars resulting in more locations at finer spatial and temporal resolution, which can lead to larger estimates of home range size. Based on wolves with GPS collars, 95% kernel density estimation (kde) home range size of wolves (n=11) on POW between 2012 and 2014 averaged 535 km^2 (range=292–644 km^2; ADFG 2015d, p. 2), a notable increase compared to wolves collared with VHF technology in the mid-1990s. However, it can be difficult to reconcile differences in data quality between VHF and GPS collars during analysis and therefore comparisons should be done with caution. On the northern mainland of southeastern Alaska, 95% kde pack home ranges were four times as large as those on POW (mean=3,926 km^2, range=2,969–4,884; n=3).

During denning and pup-rearing periods, pack home ranges were about 50% smaller than during other times of year (Person 2001, p. 55). For example, summer home ranges of five packs on POW averaged 100 km^2, where winter home ranges for the same packs averaged 240 km^2 (Person et al. 1996, p. 7). Summer home ranges for wolves on POW were similar to summer home ranges reported for Minnesota where wolves primarily rely on deer for food; however, winter home ranges of POW were substantially larger (Person et al. 1996, p. 7; see Fuller et al. [2003, pp. 172–174] for territory sizes of wolf populations in North America).

Resource selection.—We are aware of two studies with one being a subset of the other that directly examined habitat use of Alexander Archipelago wolves. Person and Ingle (1995) analyzed habitat use in proportion to its availability of radio-collared wolves on POW and Kosciusko Island. They classified forested habitat in their study area into four categories: non-commercial forest, low volume old-growth, high volume old-growth, and young-growth (p. 29). Only diurnal locations of three wolf packs from aircraft were used in the analysis.

Wolves occupied all habitat types in the study area. Analysis of all aerial locations showed that wolves were found in young-growth habitat 7.2% of the time (95% CI= ±3.4%, n=227), low volume old-growth 46.8% of the time (±6.2%), high volume old-growth 9.5% of the time (±3.8%), and non-commercial forest 34.7% (±6.2%) of the time (p. 30). Habitat use differed significantly from availability for all three packs (p. 30). All three packs in the study used high-volume forest in proportion to its availability. Two packs used low volume old-growth stands significantly more than expected and one pack used noncommercial habitat more than expected. All three packs used young-growth habitat significantly less than expected based on its availability. The authors concluded that wolves appeared to be selecting for unlogged habitat types during the daytime (p. 30).

Person (2001) completed a more comprehensive habitat analysis using data described in Person and Ingle (1995). He reported that wolves spent most of their time at low elevation, especially during the denning period, with 50% of radiolocations below 82 m and 95% below 400 m (p. 62). During pup-rearing wolves selected for open-canopy and closed-canopy forests, typically avoiding clearcuts and roads; otherwise, wolves did not appear to select strongly for or against other habitat types in other times of the year except they did show a strong affinity for habitats close to lakes and streams (pp. 62–64). He also found a strong, inverse relationship between home range size of wolves and the proportion of “critical winter habitat for deer” (p. 66), defined as productive old-growth forest less than 250 m in elevation with southern exposure.

Denning.—Alexander Archipelago wolves den in root wads of large living or dead trees in old-growth forests and near freshwater (Person and Russell 2009, p. 211). Person and Russell (2009,

p. 217) examined habitat characteristics of 25 active wolf dens on POW and adjacent islands between 1995 and 2004 and found that wolves select for relatively flat areas near lakes and streams at low elevations. Seventeen of 25 active dens (67%) were adjacent to ponds or streams with active beaver colonies (p. 216). Most used den sites were located farther from logged stands and roads than unused locations, although wolves used areas near clearcuts and roads for denning probably because suitable alternatives were not available (p. 220). The authors speculated that for many wolf packs in their study area, large proportions of the landscape were unsuitable for dens sites owing to logging and topography (p. 222).

We are not aware of any other studies of denning habitat used by Alexander Archipelago wolves. Garceau (1960, p. 487) found a wolf den on Kupreanof Island in 1959; it was located under a decaying western hemlock (*Tsuga heterophylla*) stump within a forest and was about 15 m from the forest edge near a mudflat. Across North America, characteristics of dens vary widely depending on what is available to wolves (e.g., caves, sandy bluffs; see Thiel et al. 1998 and Trapp 2004 for a summary of the various kinds of gray wolf dens).

Wolf-deer habitat relationships.—Alexander Archipelago wolves consume more deer than any other single prey species throughout most of their range (see *Food habits*). Thus, maintaining a viable, well distributed wolf population may depend on maintaining habitat to support a viable, well distributed, and available population of deer (Person et al. 1996, pp. 15–16). Alexander Archipelago wolf abundance may be especially linked to deer abundance and availability on the islands in the archipelago where other ungulate prey species are either lacking or less abundant. In general, wolf populations are expected to track the upward and downward fluctuations in deer populations in some time-lagged fashion (e.g., Van Ballenberghe et al. 1975, p. 40). Therefore, deer habitat will be the primary focus of the following discussion.

Several characteristics of old-growth forest make it valuable winter habitat for deer. High volume old-growth forest stands with multi-layered overstories intercept snow and moderate temperature and wind, creating microclimate favorable for deer (Bloom 1978, p. 108; Kirchhoff and Schoen 1987, pp. 30–32). Owing to the complex canopy structure of old-growth forest types, light penetrates to the forest floor, facilitating production of a diverse understory of shrubs and forbs, including several nutritious forage species for deer (Bloom 1978, p. 110–111; Hanley 1984, p. 4; Parker et al. 1999, p. 21; Hanley et al. 2014, p. 7). In addition, arboreal lichen, which is nutritious deer forage, is available in significant quantities only in old-growth stands (Hanley 1984, p. 8).

Silvicultural practices have affected deer habitat quality in southeastern Alaska and coastal British Columbia. The predominant method of timber harvest in this region has been clearcut logging on a prescribed rotation length of 90–120 years. Clearcut logging of forest stands initiates seral stage development, and continuing timber harvest on an approximately 100-year rotation perpetuates the recycling of early seral stages. This effectively replaces heterogenous older forest with a mosaic of even-aged, young-growth forest stands of various ages. Under natural succession, shrub and herb biomass production increases for up to 10–15 years after timber harvest. Although young clearcuts temporarily produce abundant forage for deer, typically it is of poorer nutritional quality than forage available in old-growth (Hanley et al. 1989, p. 16; Hanley 2005, p. 105). It also is apparent that in some areas, understory plant species may never reappear on sites under intensive even-aged management without subsequent treatment (Alaback 1982, p. 1941; Hanley 2005, p. 104).

Following overstory removal, after 25 to 35 years of stand development, early seral plants (forbs, grasses, and shrubs) give way to young-growth coniferous trees. The canopies of these regenerating forests close over, intercepting sunlight and eliminating most understory vegetation (Alaback 1982, p. 1938), rendering the forests of little use to deer (Wallmo and Schoen 1980, p. 453; Yeo and Peek 1992, p. 257; Person et al. 2009, p. 5). Consequently, the decline in understory corresponds with a depression in deer carrying capacity that extends for many decades until the forest begins to develop old-growth condition at approximately 250 years of age (Alaback 1982, p. 1939). Person (2001, p. 96) described these circumstances as "succession debt," such that forest succession that was initiated by timber harvesting has long-term costs to the ecosystem.

Since 1997, with the realization that most of the even-aged stands harvested between the 1970s and 1990s were entering the closed canopy stage of succession, researchers began evaluating the value of intermediate treatments on even-aged forest stands specifically for deer. Hanley (2005, p. 104) found that thinned stands of 56–190-year-old even-aged forests produced roughly five times as many deer-days use, a metric for quantified value as food for deer, as unthinned stands. He also reported that including red alder (*Alnus rubra*) in regenerating stands facilitates secondary succession and increases the number of deer-days compared to no treatment of the stand (p. 105). However, the benefit of treatments (e.g., thinning, planting of red alder) decreased with increasing age of stand (Hanley et al. 2013, p. 27); in other words, if treatments were applied later (25+ year old stands) food resources for deer were lower than if treatments were done earlier (15–25 year old stands). This finding was relevant to deer because most even-aged stands on the Tongass already are 25+ years old (Sisk 2007, p. 13) and therefore, the value of intermediate treatments for deer in those stands is reduced. In addition, little research has documented whether or not deer select precommercially thinned stands despite the increased forage biomass in those stands.

Deer habitat use.—Many field studies have aimed to understand habitat use and selection of deer within the range of the Alexander Archipelago wolf. Researchers have used traditional methods such as radio-collared deer (e.g., Schoen and Kirchhoff 1990; Yeo and Peek 1992; Farmer et al. 2006; Gilbert 2015), counts of deer pellet groups (e.g., Wallmo and Schoen 1980), counts of deer tracks (e.g., Bloom 1978), and measurements of browse availability and use (e.g., Kirchhoff 1994b), as well as more sophisticated methods such as mark-recapture techniques with DNA from fecal pellets (Brinkman et al. 2011). Farmer and Kirchhoff (2007) proposed an ecological classification of deer habitat to assist managers in managing both habitat and deer populations.

Below, we briefly summarize key findings related to Sitka black-tailed deer and their habitats with an emphasis on recent field studies (Table 10). We refer the reader to Schoen and Kirchhoff (2007), Nelson et al. (2008), and Person et al. (2009) for more detailed overviews of deer and deer research in southeastern Alaska and coastal British Columbia.

Table 10. Select recent (post-1997) studies of Sitka black-tailed deer within the range of the Alexander Archipelago wolf.

Study	Location	Description and key findings
Doerr et al. 2005	Mitkof Island	• Tracked radio-collared deer during winters of deep snow and without deep snow • Deer selected for low elevation (<153 m), southerly slopes close to shoreline (<305 m) • Winter habitat use by deer differs during winters with low and deep snow conditions
Farmer et al. 2006	Heceta Island	• Tracked radio-collared deer to examine linkages between habitat use and fitness • Main causes of death for females and young were predation by wolves and malnutrition; hunting was primary cause of death for males • Use of level terrain was associated with fatalities at all spatial scales analyzed • During snow free months, deer selected for young clearcuts, but also they were at greater risk of death from predation and hunting in this habitat
Person et al. 2009	Meta-analysis	• Estimated survival and habitat selection of deer in winter and summer using data from Mitkof, Admiralty, Heceta, and POW islands • During winters with snow, deer selected old-growth forests on south-facing slopes; old young-growth forests avoided during all seasons • Use of level terrain increased fatalities from wolves, as did high densities of habitat edges
Brinkman et al. 2011	POW	• Developed technique for estimating deer abundance using DNA from fecal pellets • Detected 30% decline during 3-year study; attributed decline to severe winters • Deer densities in managed stands >30 years old were lower (7 deer/km^2) than in managed stands <30 years old (10 deer/km2) and unmanaged lands (12 deer/km^2)
Gilbert 2015	POW	• Tracked GPS-collared female and fawn deer through two mild and one severe winters • Fawn survival was low but variable, driven by black-bear predation in summer and malnutrition in winter; adult female survival was high • Deer habitat selection in summer driven by predation risk and forage availability; snow depth determined selection in winter • Snow interacted with habitat: deer selected for young clearcuts, medium-volume old-growth forest, and against old clearcuts at low snow but trends reversed as snow depth increased

Throughout the year, deer use a variety of habitats ranging from sea level to alpine ridges greater than 1,000 m in elevation. Although some deer migrate from low-elevation, forested winter range to high-elevation summer range in the alpine, other deer remain resident in their home ranges year round. Generally, during spring, summer, and fall, deer focus on maintaining or increasing body condition by selecting habitats with high availability of nutritional browse (Gilbert 2015, p. 88). As winter approaches and snow accumulates, they are increasingly confined to lower elevations and fewer habitats are available to them; for this reason, most research has focused on deer winter habitat use and requirements.

Most studies of deer in southeastern Alaska have found that deer use old-growth forests significantly more than young-growth forests, especially in winter (Bloom 1978, p. 110; Wallmo and Schoen 1980, p. 453; Rose 1982; Schoen and Kirchhoff 1990, p. 374; Kirchhoff 1994b, p. 34). However, some studies have reported that deer use of clearcuts less than 10 years old was similar to that of old-growth (Brinkman et al. 2011, p. 239), even in winter (Yeo and Peek 1992, p. 257; Doerr et al. 2005, p. 326). Doerr et al. (2005, p. 322) found that deer selected for habitats with southerly exposures near shoreline regardless of forest type during the winter, presumably to avoid accumulation of deep snow (Table 10). Gilbert (2015, p. 129) found that deer selected for young clearcuts, against old clearcuts, and against high-volume old-growth at lower snow levels, but decreased selection for young clearcuts and increased selection for old clearcuts and high-volume old-growth as snow depth increased. In addition, Gilbert (2015, p. 130) documented that as local availability of young and old clearcuts increased for individual deer, deer increasingly selected for those habitats (i.e., a functional response), but that deer decreased selection for clearcuts of all ages when old-growth was more available to them as an alternative.

Although winter habitat use of deer may be more ambiguous than previously considered, the effect of periodic, severe winters on deer populations in the region is well documented. Localized reductions in deer numbers have occurred during periods of deep snow (approximately once every 7–10 years) when survival is low (Schoen and Kirchhoff 2007, p. 2; Brinkman et al. 2011, p. 232), especially for fawns (Gilbert 2015, p. 55). In addition, winter severity may be compounded when multiple severe winters occur in sequence (Brinkman et al. 2011, p. 232).

A severe winter can affect deer primarily in two ways: (1) by reducing availability of forage (i.e., snow covers browse) and, (2) by increasing energy expenditure associated with movement (i.e., deep snow is difficult to move through; Parker et al. 1984, p. 474; Parker et al. 1999, p. 5). Hanley (1984) identified three thresholds of snow depths for deer in southeastern Alaska: the depth at which evergreen forbs and herb-layer shrubs become buried (approximately 10 cm), the depth at which deer sink beyond front knee height and energy costs for location increase greatly (25 to 30 cm), and the depth at which tall shrubs become buried (White et al. 2009, p. 484); he recommended that snow deeper than 25 to 30 cm be considered “deep snow”. Similarly, Parker et al. (1999, p. 25) found that when maximum snow depths were >30 cm energy costs associated with movement of an average-sized deer (25–30 cm carpus height) increased significantly. Hanley et al. (1989, p. 29) later suggested that another threshold occurs when snow reaches brisket height (approximately 55 cm) and energy costs for location again increase dramatically.

Snow depth and duration are typically lower in the southern portion of southeastern Alaska than in the northern portion (Figure 9 in Albert and Schoen 2007, p. 15) and this pattern extends southward into British Columbia (Shanley et al. 2015, p. 6). Decreasing amounts of snow also occur as one moves from the mainland westward to the outer islands. Despite the generally lighter snow conditions found in the southern and western edges of southeastern Alaska, deer populations in these areas are still subject to population declines following severe winters (e.g., Farmer et al. 2006, pp. 1404, 1412; Brinkman et al. 2011, p. 232).

The effects of a given quantity of snow on deer depend upon several factors, including physical condition of the deer going into the winter, forage availability during and following the winter, the effectiveness of both thermal and escape cover, and the duration the snow persists on the ground (Wallmo and Schoen 1980, pp. 457–458; Hanley et al. 1989, p. 32; Doerr et al. 2005, pp. 327–329). Population declines after heavy snow events have been attributed primarily to direct mortality from starvation, followed the next year by depression of fecundity among survivors caused by malnutrition (Hanley 1984, p. 13; Gilbert 2015, p. 56). Recovery often occurs within four or five years where ranges are in good condition and predation is not excessive (Klein 1979).

The negative effects of natural patchiness of habitats on deer connectivity in southeastern Alaska and coastal British Columbia are intensified when clearcut logging converts large forest stands into a patchwork of smaller old-growth stands that are isolated or semi-isolated as a result of snow-filled clearcuts (Kirchhoff and Schoen 1987, p. 32; Person et al. 1996, p. 22). Kirchhoff (1994b) stressed the importance of what he termed ‘effective areas’, or the total area of habitat available to deer. If deer can move freely in response to changing snow conditions, effective area would be essentially unlimited. However, should snow accumulation or lack of forage in surrounding managed stands concentrate deer in residual patches of old-growth, effective area may be limiting. During periods of deep snow, deer confined to isolated stands, especially those of forage-poor habitat, consume the available food resources and may suffer higher rates of mortality from malnutrition than deer in unfragmented landscapes (Kirchhoff 1994b, p. 19; Farmer et al. 2006, p. 1412). Further, deer concentrated by deep snow into small patches of winter cover appear to be especially vulnerable to predation by wolves (Farmer et al. 2006, p. 1412; Person et al. 2009, p. 8), which can accelerate declines and delay recovery of deer populations. McNay and Voller (1995, p. 138) reported that forest harvesting, road building, and spatial isolation of winter habitats, may intensify predation on deer populations, impede recruitment, and contribute to an overall loss of population resiliency. Both Schoen and Kirchhoff (1985, p. 99) and Doerr et al. (2005, pp. 326–327) showed that Sitka black-tailed deer generally remain within habitual use areas even if local conditions are adverse. Based on these studies, it appears unlikely that deer will move outside of their home ranges to seek better habitat conditions if important habitat within that home range is degraded.

CHAPTER 4: DYNAMICS OF MULTIPLE POPULATIONS (AMONG-POPULATIONS AND RANGEWIDE LEVELS)

4.1. Connectivity

Connectivity is a broad and vague term referring to movement of individuals among populations and interactions between demes, subpopulations, and populations. Given the naturally-fragmented landscape within the range of the Alexander Archipelago wolf, connectivity among populations is of considerable relevance when assessing the status of the rangewide population. From a conservation perspective, the distinction between demographic and genetic connectivity is important (Lowe and Allendorf 2010, p. 3038). If a lot of interchange among Alexander Archipelago wolf populations occurs, especially between mainland and island populations, then reduction or loss of wolves for any island or island groups poses less risk to the overall population than if interchange of wolves is limited. One major limitation to our understanding of the Alexander Archipelago wolf is the lack of immigration and emigration rates between and among populations and other related processes such as colonization and recolonization. Below, we summarize the available information on demographic and genetic connectivity of the Alexander Archipelago wolf.

4.1.1. Demographic connectivity

Demographic connectivity depends on relative contribution of immigration and emigration compared with within-population vital rates of birth and death; in other words, it does not depend on dispersal or movement alone, but instead considers how migration rates relative to local recruitment affects dynamics within and among populations (Lowe and Allendorf 2010, p. 3039; Mills 2013, p. 177). We are not aware of any field studies that have measured demographic connectivity explicitly or opportunistically and therefore, in this section, we describe among-population dispersal and movements of Alexander Archipelago wolves.

The fact that wolves do not exist on Admiralty, Baranof, and Chichagof islands, even though those islands support high numbers of deer, indicates that wolf movements may be restricted by water barriers. Potential crossing distances to these islands at some locations are not greater than confirmed crossing points elsewhere in the archipelago (Person et al. 1996, p. 4); however, recent glaciation, the shape and distribution of land masses, and tidal currents likely combine with over-water distances to impede successful wolf dispersal. Also, lower densities of wolves on the mainland, a potential source population, may influence chance dispersal events to these islands. In terms of distance, avenues of dispersal are limited, and most of the feasible dispersal routes involve multiple swims. Generally, as the larger straits between land masses become constricted into channels and passes, the greater the influence of tidal currents.

As described above under *Within-population dispersal*, movements of wolves among nearby islands probably are common. Yet, wolves that were radio-collared on POW and Kosciusko Island did not disperse out of the population, which would require at least five swims with the longest being only about 2 km in length. Alexander Archipelago wolves in coastal British Columbia apparently can swim as far as 13 km (Darimont and Paquet 2002, p. 418), although we presume that this lengthy distance is on the extreme end. Nonetheless, we assume that wolves are capable of swimming the short distance from POW to West Island and onward to Zarembo Island (i.e., between GMUs 2 and 3). Using remote camera systems, a wolf was documented on

Shrubby Island between POW and Zarembo Island (ADFG 2015d, p. 2), suggesting that wolves explore and occupy this route, although concrete evidence of successful dispersal is lacking.

The degree of insularity probably varies among island groups. Interchange between POW and nearby islands (GMU 2) and the mainland (GMU 1) may be limited most, in part due to the time required to travel the distance and the low survival rates of dispersing wolves (see *Survival*), and therefore could be inconsequential from a demographic perspective. However, it may be sufficient between the mainland and the islands within GMU 3 via Dry Strait, a relatively narrow waterbody between Mitkof Island and Dry Island that occasionally goes dry during extreme low tides. At least one male wolf that was radio-collared on Kupreanof Island in March 1999 was trapped on Revillagigedo Island in January 2002, providing evidence of movement from GMU 3 to GMU 1A (USFS 2015a).

4.1.2. Genetic connectivity

Genetic connectivity is the degree to which gene flow affects evolutionary processes within populations (Lowe and Allendorf 2010, p. 3042). The consequences of differing levels of genetic connectivity are poorly understood and depend critically on the ecological and genetic history of the associated populations (e.g., Lande 1999, pp. 11–16). Average levels of gene flow are estimated by the number of migrants per generation (N_m) between populations. Unfortunately, it is difficult to measure gene flow over longer time frames directly and therefore indirect methods often are used to estimate it. For example, F_{ST} can be used to estimate N_m, although see Whitlock and McCaughley (1999) for caveats associated with this approach. Another indirect method for assessing N_m is determining the frequency of private alleles (i.e., one found in only one population); for example, if N_m is low, the frequency of private alleles will be high due to mutation.

Since 1997, several genetic studies of the Alexander Archipelago wolf have been conducted (see *Taxonomy* for overview), but not all are relevant for assessing genetic connectivity between and among populations. As noted above, the scope of inference of these studies depends on the type of genetic marker used and the spatial and temporal extent of the samples analyzed. Generally, genetic studies involving nuclear DNA are more informative about contemporary gene flow between and among populations; mtDNA reflects only a single genealogy and maternal inheritance and it does not undergo recombination, making it more useful in phylogenetic studies aimed to resolve questions about evolutionary, historical relationships within and among species. For this reason, we primarily describe gene flow between Alexander Archipelago wolf populations as determined by studies of nuclear DNA, drawing on results of mtDNA analyses only when appropriate. We present results in a similar format as in *Taxomony* above; refer to Table 2 for descriptions of sampling beyond southeastern Alaska and coastal British Columbia.

Of the four mtDNA studies of the Alexander Archipelago wolf, only Weckworth et al. (2011) commented on gene flow among coastal wolf populations. The authors essentially combined data from two of the other studies in coastal British Columbia (Munoz-Fuentes et al. 2009) and southeastern Alaska (Weckworth et al. 2010) and found that only one population (GMU 1C) within the range of the Alexander Archipelago wolf was distinct from all the others (p. 2). Wolves sampled from this population were the only ones to have Haplotype I, which may indicate mixed refugial origins of coastal wolves or contemporary gene flow between interior continental wolves and coastal wolves (p. 3).

We are aware of four nuclear DNA studies with results relevant to gene flow between populations of Alexander Archipelago wolves. Weckworth et al. (2005) sampled wolves from GMU 1A (Mainland Coast South, n=9; Revillagigedo, n=24), GMU 2 (POW, n=42), and GMU 3 (Kupreanof, Mitkof, and Woewodski islands, n=26). Private alleles were found in wolves in GMU 3 as well as both POW and Revillagigedo Island but in each case these alleles were restricted to a single individual (p. 921). Pairwise estimates of F_{ST} between southeastern Alaskan populations ranged from 0.02 (Mainland Coast South and Revillagigedo) and 0.20 (GMUs 2 and 3). The authors concluded that significant genetic structure within southeastern Alaska relative to other populations in the Pacific Northwest, and lack of significant correlation between genetic and geographical distances (i.e., no evidence of isolation by distance) suggest that differentiations of southeastern Alaskan wolves may be caused by geographical disruptions to dispersal and gene flow.

Breed (2007) included wolves primarily from GMU 1A and GMU 2 in southeastern Alaska and from Regions 5 and 6 in coastal British Columbia (considered one population in this paper). Number of private alleles in Regions 5/6, GMU 1A, and GMU 2 (and their frequency) was 18 (0.76), 4 (0.07), and 2 (0.02), respectively (p. 18). Pairwise F_{ST} between Regions 5/6 and GMU 1A was 0.087 and GMU 2 was 0.149, and between and GMU 1A and GMU 2 was 0.122 (p. 19). An interesting aspect of this study was the authors’ ability to describe directionality of gene flow using a genetic assignment test. By estimating and comparing ratios of immigrants to residents, they found that gene flow was mostly uni-directional with most wolves in Region 5/6 being residents and wolves in GMUs 1A and 2 expressing higher degrees of mixed ancestry (pp. 22–23). However, of these three units, GMU 2 was the most differentiated, although it was subtle and gene flow did not appear to be limited (p. 32). The authors concluded that wolf populations in Regions 5/6 serve as a source population for GMU 1A (p. 34).

Stronen et al. (2014) studied genetic differentiation of wolves on the central coast of British Columbia (Region 5) using microsatellite markers to examine data obtained from wolf fecal samples. Their results from 116 individual wolves indicate a genetic cline between coastal mainland and island wolves (p. 8); however, this study was conducted at too small of a spatial scale to detect meaningful population differentiation. Nonetheless, they found that even though wolves have been observed moving between the mainland and islands in their study area, a genetic cline was apparent, perhaps suggesting that natal habitat-biased dispersal may contribute to genetic differentiation (pp. 1, 8).

Cronin et al. (2015a) analyzed SNPs from wolves in six GMUs in southeastern Alaska in which they occur (i.e., GMUs 1A, 1B, 1C, 1D, 2, and 3; only GMU 5A was not represented). The authors conducted many comparisons and estimated mean F_{ST} values among all six GMUs (0.1268, SE=0.0184), among all GMUs except for GMU 2 (0.1147, SE=0.0225), and for GMU 2 versus all other GMUs (0.1511, SE=0.0326; p. 7); they did not find any statistical significance when comparing means (p. 6). Pairwise F_{ST} between populations was lowest for wolves in GMUs 1B and 3 (0.0344) and highest for wolves in GMUs 1D and 2 (0.2811; Supplemental Table 3 in Cronin et al. 2015a). The authors concluded that wolves in southeastern Alaska are not a genetically homogenous group (i.e., some population structure exists) and that although wolves in GMU 2 show a degree of differentiation, as also shown by Weckworth et al. (2005), they are not particularly differentiated compared to the overall differentiation among wolves in other southeastern Alaskan GMUs (p. 8). These data and interpretations were not contested by (Weckworth et al. 2015).

4.2. Population processes

Mills (2013, p. 185) emphasizes that not all populations are equal. Because of varying effects of stressors within and among populations, different populations play different roles in dynamics across the landscape. When a continuous population becomes fragmented by an intervening matrix, four potential outcomes can occur (along a continuum dependent on the natural history of the species and the extent to which dispersal is influenced by the potential barrier): multiple isolated populations, metapopulations, source–sink dynamics, and ecological traps. We posit that these same processes can occur in naturally-fragmented systems such as island archipelagos and that two in particular (metapopulations, source–sink dynamics) may be relevant to the Alexander Archipelago wolf.

Alexander Archipelago wolves probably exist in a metapopulation, defined here as sets of spatially distributed populations among which dispersal and turnover are possible but do not necessarily occur (Harrison 1994, p. 117). However, we lack data from which to test this hypothesis. We include it here to emphasize that, for wolves in particular, both among-population processes and within-population movements are key components of wolf persistence.

Further, some populations of Alexander Archipelago wolf may be strong contributors (i.e., sources) to metapopulation growth while others are drains (i.e., sinks), and still others have no consistent influence on neighboring population units. Breed (2007, p. 34) hypothesized that wolves in coastal British Columbia served as a source population for wolves in southeastern Alaska, specifically GMU 1A, and that hunting of southeastern Alaskan wolves contributed to its role as a sink population. His findings have not been substantiated with demographic data, and doing so would require additional field effort and careful interpretation of population processes.

CHAPTER 5: CURRENT AND FUTURE HABITAT AND RESOURCE CONDITIONS

5.1. Environment

The range of the Alexander Archipelago wolf comprises a narrow strip of mainland and several island complexes found between the Coast Mountain Range of Canada and the Pacific Ocean. The region is dominated by coniferous temperate rainforests, interspersed with other habitat types such as sphagnum bogs, sedge-dominated fens, alpine areas, and numerous lakes, rivers, and estuaries. The largest island complex, the Alexander Archipelago in southeastern Alaska, spans over 500 km in latitude and includes more than 22,000 islands, ranging in size from several m^2 to over 6,000 km^2. South of the Alexander Archipelago, the Haida Gwaii Islands lie approximately 80 km west of mainland British Columbia (no wolves occupy these islands; see *Distribution*). Moving further south, many smaller island groups occur adjacent to the mainland, especially near the outflows of major river systems. At the far southern end of the range, Vancouver Island is separated from the mainland by only a narrow ocean channel at several points.

The topography of mainland southeastern Alaska and coastal British Columbia is rugged with numerous deep, glacially-carved fjords, some of which penetrate the Coast Mountain Range. A narrow band of forest grows between the ocean and steep mountains to the northeast (Albert and Schoen 2013, pp. 775–776). Several major rivers transect the Coast Mountain Range, connecting coastal Alaska to interior British Columbia and Yukon Territory. Outside of these river corridors, glaciers and ice fields dominate the higher elevations, separating the coastal forests from the adjacent inland forest in continental Canada (Albert and Schoen 2007, p. 2).

The climate in southeastern Alaska and coastal British Columbia is generally wet and cool, although with considerable geographic variation. Average annual precipitation varies from 50 to 600 cm near sea level, with more precipitation at higher elevations (Albert and Schoen 2007, p. 2; Tillmann and Glick 2013, p. 22). Summers tend to be drier than winters, when much of the precipitation falls as snow in northern portions of the region and at higher elevations (Tillmann and Glick 2013, pp. 21–22). Generally, temperatures are warmer in the southern portions of the range, and precipitation decreases from west to east (Shanley et al. 2015, p. 5), often resulting in rainshadows on the eastern sides of some of the larger islands (MacKinnon 2003, p. 475).

The coastal forests of southeastern Alaska and coastal British Columbia are part of the temperate rainforest ecosystem that extends along the Pacific coast from northern California to southcentral Alaska. In southeastern Alaska, the lowland forests are composed primarily of western hemlock and Sitka spruce (*Picea sitchensis*), although mountain hemlock (*Tsuga mertensiana*), western redcedar (*Thuja plicata*), and yellow cedar (*Chamaecyparis nootkatensis*) are also present. Currently, these coniferous forests cover roughly 26% of the regional landmass (Figure 5; Shanley 2015). Further south in coastal British Columbia, conifer forest continues to dominate, but Douglas-fir (*Pseudotsuga menziesii*), western redcedar, and other fir species (*Abies* spp.) are increasingly common; about 55–68% of coastal rainforest remains as old-growth, with about 30% logged and reforested, 2% logged and converted to human habitation, and the remainder naturally unforested (MacKinnon 2003, p. 479; Service 2010, p. A-12). With increasing elevation, forests grade into subalpine and alpine vegetation zones; treeline increases in elevation southward, ranging from about 700 to 900 meters (USFS 2008b, p. 3-7; Albert and Schoen 2013, pp. 775–776). At the highest elevations rock, snow, and ice dominate the land surface. In southeastern Alaska, these non-vegetated areas constitute approximately half of the total land surface (Figure 5), particularly along the mainland and on higher mountains on some of the large islands.

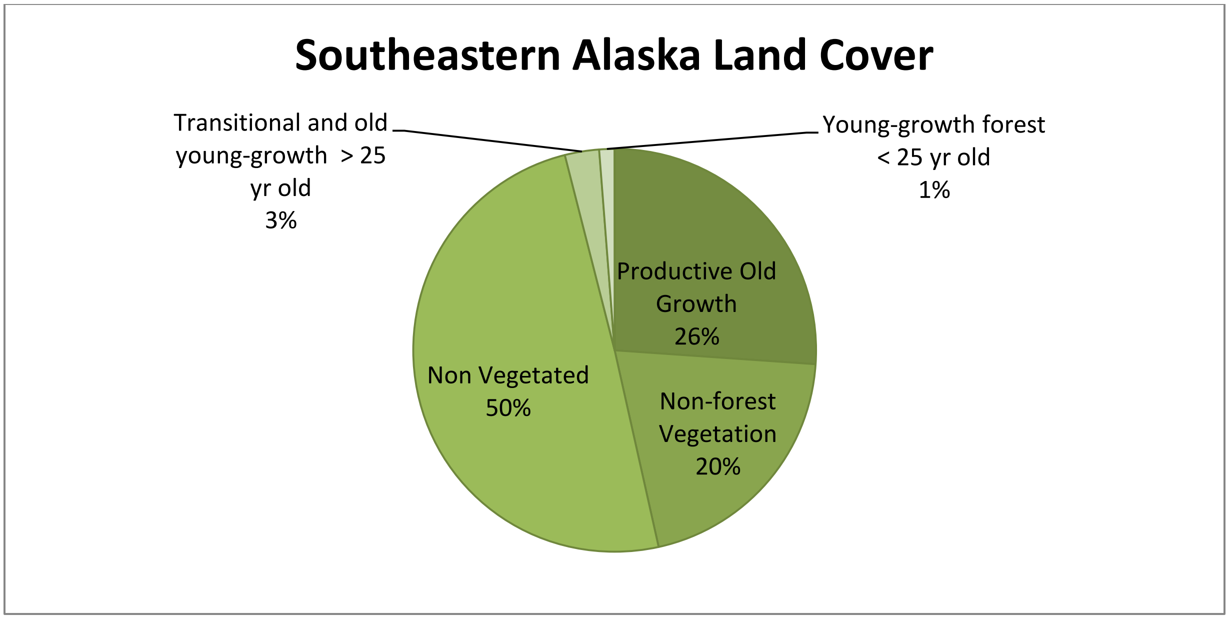

Figure 5. Land cover across southeastern Alaska south of Yakutat (Shanley 2015).

Conifer forests in the northern temperate rainforest are influenced by fine-scale habitat heterogeneity created by underlying geology, topography, and resulting drainage patterns, and various disturbance regimes. Well-drained sites generally have higher forest productivity (often expressed in terms of volume of wood produced per unit area per year, or as a standing volume of wood) compared to poorly-drained sites with deep organic soils, which often produce bogs and fens rather than forest. Forests of intermediate or low productivity occupy transitional ecotones between well-drained productive forests and poorly drained, non-forested areas (Alaback 1982, pp. 1932–1934; USFS 2008b, pp. 3-7–3-8; Albert and Schoen 2013, pp. 775–776). Therefore, local landform diversity and drainage patterns contribute to fine-scale habitat heterogeneity.

Disturbance also plays an important role in the natural fragmentation of the landscape. Fire is rare due to the abundant year-round precipitation, and thus does not play a major role in forest succession at the landscape scale. Instead, wind is the primary disturbance agent, although landslides, avalanches, debris flows, tidal waves, insects, fungi, and disease also influence forest structure and thus contribute to fine-scale habitat heterogeneity. Disturbances generally occur at a small scale, where individual or small groups of trees die or are blown down by wind, creating canopy gaps. Occasional severe wind storms cause extensive damage up to several hundred square kilometers (Nowacki and Kramer 1998, pp.4–14).

5.2. Land ownership

In southeastern Alaska, the majority (76%) of land area lies within the Tongass and is managed by the USFS (Table 11; Shanley 2015). The National Park Service (NPS) is the next largest landholder (about 12% of land area), mostly within Glacier Bay National Park, although there are several smaller parks near Skagway and Sitka (Figure 6). The State of Alaska (State) manages roughly 4% of the land area in southeastern Alaska, with the majority located in the Haines State Forest, which is managed for timber production. Native Corporations established under the Alaska Native Claims Settlement Act (ANCSA) own about 3% of the land area; these lands are managed largely for timber production and are concentrated in the central and southern portions, especially in GMU 2 (Figure 6). All other ownerships, including private, municipal, and tribal reservation lands, amount to about 5%, with land use varying considerably among these landowners.

Table 11. Land ownership and management (km^2; percent in gray-shaded cells) by Game Management Unit (GMU) across southeastern Alaska south of Yakutat Bay and within the range of the Alexander Archipelago wolf (i.e., minus GMU 4; Shanley 2015). Land owners and managers include U.S. Forest Service (USFS), National Park Service (NPS), State of Alaska (State), Native Corporations, and other private, municipal and tribal lands.

	USFS	NPS	State	Native Corporations	Other
GMU 1	33,571	8,430	2,534	263	3,017
	70%	18%	5%	1%	6%
GMU 2	7,232	0	236	1,573	340
	77%	0%	2%	17%	4%
GMU 3	7,133	0	202	226	261
	91%	0%	3%	3%	3%
GMU 4	14,094	0	79	429	573
	93%	0%	<1%	3%	4%
GMU 5A	4,840	2,444	30	79	134
	64%	33%	<1%	1%	2%
Total	66,871	10,874	3,082	2,571	4,325
	76%	12%	4%	3%	5%
Within the range of the wolf	52,777	10,874	3,003	2,142	3,752
	73%	15%	4%	3%	5%

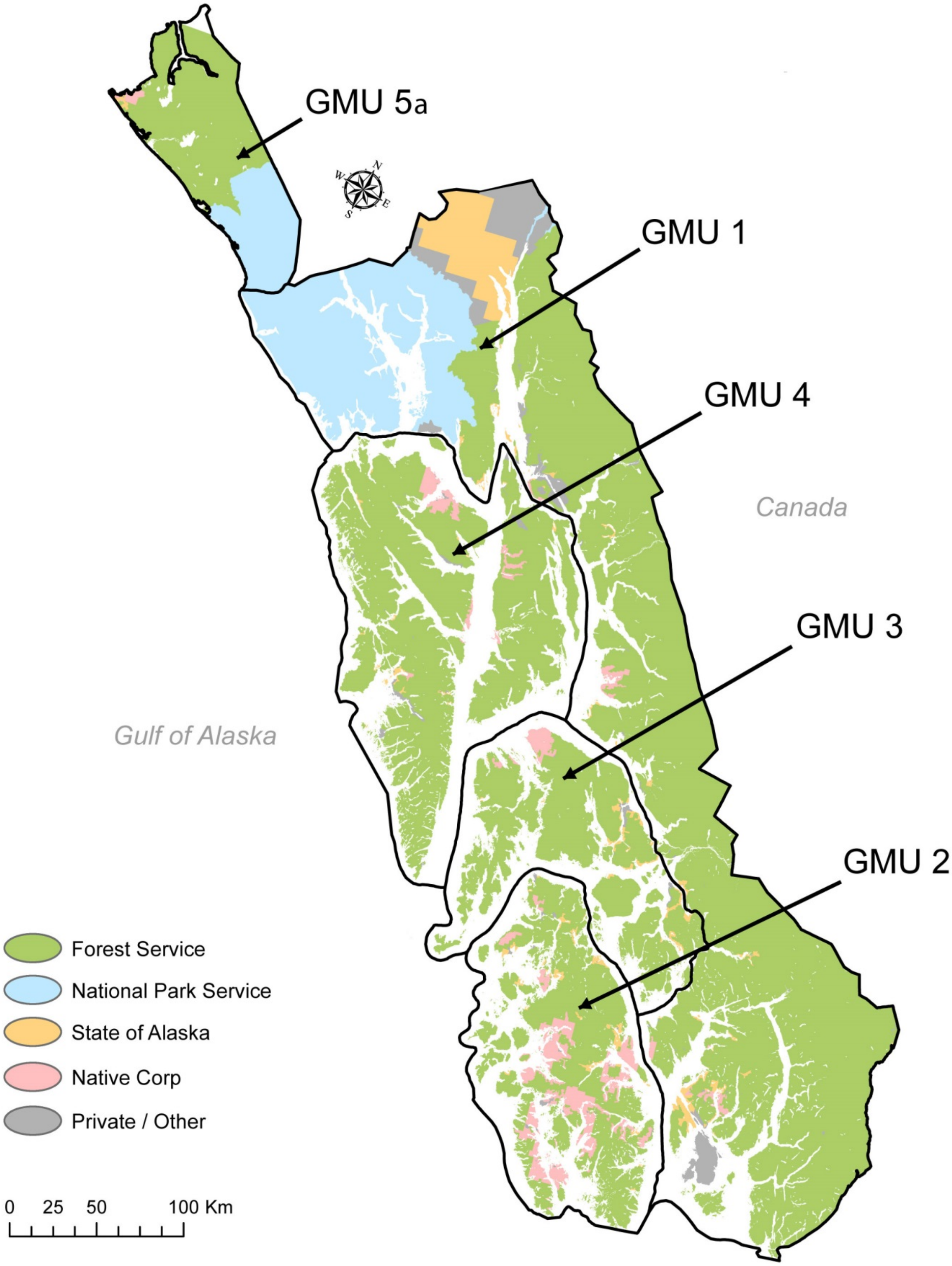

Figure 6. Map depicting land ownership and management across southeastern Alaska south of Yakutat (Shanley 2015). The Alexander Archipelago wolf does not occur in Game Management Unit (GMU) 4.

In British Columbia, most (94%; 895,126 km^2) of the land and forest are owned by the Province of British Columbia (i.e., Crown lands), 4% is privately owned (41,883 km^2), 1% is owned by the federal government (10,371 km^2), and the remaining 1% is owned by First Nations and others (1,349 km^2; British Columbia Ministry of Forests, Mines, and Lands [BCMF] 2010, p. 121). Issues of Aboriginal rights, title, and interest affect land use decisions on public and private land, although little of the total land is owned by First Nations. Roughly 14% of the forests in British Columbia are protected in provincial and national parks, recreation areas, and reserves (BCMF 2010, p. 44).

5.3. Cause and effect analysis

In this section, we describe deterministic stressors that influence environmental conditions experienced by Alexander Archipelago wolves. For the purpose of this assessment, we consider a stressor to be a process or event that may have a negative impact on the target taxon. We use the term deterministic to refer to factors that affect population dynamics in mostly predictable ways, as opposed to stochastic or random factors. We aimed to understand how these stressors may influence the future status of the Alexander Archipelago wolf. A key feature of our analysis was a characterization of the effect of anthropogenic and natural deterministic stressors to wolves at the individual, population, and rangewide levels

GMUs in southeastern Alaska and Regions in coastal British Columbia are both comprised of smaller analysis units, referred to as Wildlife Analysis Areas (WAAs) and Wildlife Management Units (WMUs), respectively. Mean area of WAAs is smaller than WMUs (532 km^2 and 5,624 km^2); for comparison, the home range of Alexander Archipelago wolf is between 500 and 4,000 km^2 depending on location (see *Home range size*). In this section, we conducted analyses at the most appropriate spatial scale given the question at hand and the available data. For example, wolf harvest is managed in southeastern Alaska at the GMU level, so we report harvest statistics by GMU, but we calculated road densities at the WAA level because some GMUs contain water and protected areas (e.g., national parks) that would result in biased or skewed road densities.

For the purpose of our analysis, we considered the future to be about 30 years from present (i.e., 2045). We chose this period of time because it constitutes multiple (roughly 3–6) wolf generations (Fuller et al. 2003, p. 175; Von Holdt et al. 2010, p. 4422) and because it was reasonable in terms of projecting future resource conditions. For example, the Tongass Land and Resource Management Plan (USFS 2008a; hereafter Tongass Forest Plan) is expected to guide management of the Tongass for a period of 10 to 15 years, allowing for 2–3 cycles of plan review and amendment within our defined period of analysis. Further, the Tongass Transition Framework, which outlines a planned shift from old-growth to young-growth harvest, is scheduled through 2040 (Exhibit 6 in USFS 2014a) and therefore is consistent with our time period of analysis for Alexander Archipelago wolf.

5.3.1. Wolf population model

Several deterministic stressors that are known or expected to affect Alexander Archipelago wolves are correlated with one another or interact (e.g., timber harvest and road development). To understand these complex relationships better and to quantify the relative strength of each stressor in isolation, we updated a spatially-explicit population model for wolves based on hypothetical wolf packs in GMU 2, the area for which we have the most data on Alexander

Archipelago wolves and where the three primary stressors on wolves occur at comparatively high levels (compared to other GMUs and to coastal British Columbia) and interact with a fourth stressor. The four stressors (i.e., parameters) included in the model were vegetation condition (i.e., timber harvest), road density, wolf harvest, and frequency of severe winters. Each parameter was assigned 3–5 conditions that described the range of possible future conditions for wolves in GMU 2. We informed the model from 1995 to 2014 with actual data when possible (i.e., hindcasting) and predicted rate of change in wolf abundance between 2015 and 2045 under six scenarios with different combinations of conditions for each of the four stressors (see *Chapter 6*). Our modeling effort allowed us to examine cumulative effects to wolves in GMU 2 within the constraints of the model and therefore under the explicit assumptions of the model. The model also provided an approach to assess response of wolves to each factor individually; in this chapter, we briefly summarize results from our sensitivity analysis, but refer the reader to Gilbert et al. (2015) for details.

To ensure proper interpretation of the model results presented in this chapter, we highlight a few key points about the model. First, although many parameters in the model were informed with empirical data, we modeled a hypothetical wolf population using a wolf pack as a sampling unit; wolf pack status was determined as a function of deer abundance, which was estimated in a sub-model of deer habitat capability based on forest developmental stage and environmental features. Because our model was based on deer habitat capability and not empirical estimates of deer abundance, we present results as percent change in wolf and deer abundance over a specified period of time. Second, model results, including those of the sensitivity analysis, are comparable only to each other. For example, we conducted sensitivity analyses, i.e., where only one variable was perturbed and the remainder was held constant (see Gilbert et al. 2015). In this scenario, wolves (and deer) were predicted to decline and therefore, wolf and deer abundance also declined in the sensitivity analyses; thus, it is critical to interpret the sensitivity analysis results presented in this chapter relative to one another for that particular parameter. In other words, the reader should evaluate the relative change (emphasis added) among conditions, not the absolute values themselves. Third, because the model was developed for wolves in GMU 2 only, the predictions are limited to the GMU 2 population, which is not representative of the other populations of the Alexander Archipelago wolf.

5.3.2. Timber harvest

Timber harvest and associated development has altered the landscape within the range of the Alexander Archipelago wolf more than any other human activity, and probably will continue to do so in the future. Therefore, timber harvest is an obvious deterministic stressor considered in this assessment. In this section, we briefly review timber management and practices in southeastern Alaska and coastal British Columbia, then describe past timber harvest, current conditions on the landscape, and projections for future timber harvest. For simplicity, we assess road development separately, although we recognize that these two stressors are interconnected. We close this section by summarizing the potential effects of timber harvest on Alexander Archipelago wolves and their prey.

Overview of timber management and practices

Southeastern Alaska.—In southeastern Alaska, regulation of timber harvest and associated activities is carried out primarily by the USFS via the Tongass Forest Plan, which partitions the landscape into various Land Use Designations (LUDs). Each LUD allows specified levels of timber harvest and other development activities, ranging from no development to management primarily for timber production, with intermediate levels of development allowed in some LUDs. Under the current Tongass Forest Plan, timber harvest and other development are allowed on approximately 14,000 km^2 of the Tongass, equivalent to 21% of total Tongass land area (USFS 2008c, p. 4). Given the large percentage of the land managed by the USFS (76%), the Tongass Forest Plan is the single most important regulatory/management framework influencing future habitat and resource conditions of the Alexander Archipelago wolf in southeastern Alaska.

Timber harvest on State, private (including Native Corporation), and municipal land is governed by the Alaska Forest Resources and Practices Act. This State law requires retention of unharvested buffers along anadromous fish-bearing water bodies and establishes standards to minimize erosion of soil. These regulations and their implementation are generally less restrictive than the Tongass Forest Plan that applies to National Forest System lands only. Therefore, on State and private lands that are managed for timber production, harvest is often more intensive than on the Tongass. On NPS lands, timber harvest is not permitted and only limited development is allowed.

Across all land ownerships, clearcut logging has been the primary timber harvest method. This method uses ground-based cable yarding systems to move logs to landings where they can be transported by truck to a processing facility, or to a barge for further transport. Clearcutting removes all trees from a logged unit and results in regeneration of an even-aged young-growth stand. Logging costs for a given volume of wood are typically lowest with this method and regeneration of preferred tree species such as Sitka spruce is favored. In some cases, single trees or small groups of trees may be left to provide wildlife habitat or reduce visual impacts (USFS 2008b, p. 3-328).

In recent years, various forms of uneven-aged management have been used as alternatives to clearcutting in some areas. These approaches include group selections and diameter-limit harvests and are best suited for areas where helicopters can be used to yard logs, rather than ground-based cable systems. Costs typically are higher with these “partial harvest” systems than with clearcutting, so higher-value trees often are targeted for harvest to help offset higher costs (USFS 2008b, pp. 3-328–3-329). Harvest also is spread over a larger area to produce the same timber volume that clearcutting could produce. While this methodology does not result in complete removal of tree cover from an area, nevertheless considerable slash and debris can result at the site and persist for some time. This approach results in retention of forest canopy that captures some snowfall (reducing snow accumulation) and increases heterogeneity during stand development, which favors retention of forage plants.

Coastal British Columbia.—In British Columbia, approximately 50% of the timber volume is located on land suitable for harvesting (BCMF 2010, p. 127). Rights to harvest timber on Crown lands, which is most (94%) of the land in British Columbia, are granted to various parties through timber harvesting licenses. As of 2009, roughly half (52%) of the timber harvesting rights were held in long-term licenses, which under most circumstances the rights to harvest timber and associated responsibilities continue indefinitely; medium-term licenses generally confer rights for 5 to 20 years (26% of licenses) and short-term licenses last 1 to 4 years (22% of licenses; BCMF 2010, p. 123). The provincial government is shifting away from long-term licenses (75% in 1999 and 52% in 2009).

Timber harvest on Crown lands is regulated by allowable annual cut quota (as determined by the provincial government) and is subject to the Forest and Range Practices Act passed in 2004. For major forest tenure holders, this Act requires two levels of plans and one of these, the Forest Stewardship Plan (FSP), is submitted to the provincial government for approval (BCMF 2010, p. 25). This plan identifies forest development units within which development can occur, and must provide measureable results or verifiable strategies consistent with government objectives. The government typically consults regional- or watershed-based land use plans, which outline long-term management goals and objectives for public lands and include input from a variety of stakeholders and the public, when evaluating FSPs, as well as existing regulations. The FSP has a 5-year term that can be extended to 10 years. Other requirements include consultation with First Nations, the public, and other resource users. In addition, tenure holders must also prepare a site plan that identifies intended roads, cutblocks, and other site-specific details. These plans do not need to be approved by the government, but must be available to the public on request. On private forest land (about 4% of all forested land), planning is the owner’s responsibility.

Historically, clearcutting has been the most common silvicultural method for harvesting timber in British Columbia with partial harvest composing a small percentage of the total cut. In the late 1990s, silvicultural practices shifted and since then on public land, 44% of the harvested area was clearcut and the remaining 56% was harvested under clearcutting with reserves, variable retention, and other partial cutting systems (BCMF 2010, p. 144). Since 1987, holders of timber harvesting licenses have been required to reforest the areas that they harvest (p. 143).

Past timber harvest and current conditions

Southeastern Alaska.—Commercial logging was initiated in southeastern Alaska in the late 1800s, primarily to encourage local economic growth and support development of mining, fishing, and local communities. In 1955, following completion of a major pulp mill in Ketchikan, industrial-scale logging began, dramatically increasing the rate of timber harvest. From 1909 to 1952, an average of 41 million board feet (mmbf) per year was harvested, increasing to 380 mmbf per year from 1955 to 1995 (Iverson et al. 1996, pp. 7–8; USFS 1997, p. 3-259). Timber harvest then declined to 89 mmbf per year from 1996 to 2004, and further declined to 32 mmbf per year from 2008 to 2013 (USFS 2014b, p. 220).

Across southeastern Alaska, nearly 3,000 km^2 of forest has been logged. Timber harvest was near or above 500 km^2 per decade during the 1960s through the 1990s, peaking in the 1980s when approximately 780 km^2 of productive forest was logged (Figure 7). Recent declines in the rate of logging have been linked to several factors, including changes in market conditions, more restrictive standards and guidelines in the 1997 Tongass Forest Plan, and litigation (Brackley et al. 2006, pp. 4–5, 27; USFS 2012, p. 13); additionally, vast amounts of the easily accessible productive forest has been logged.

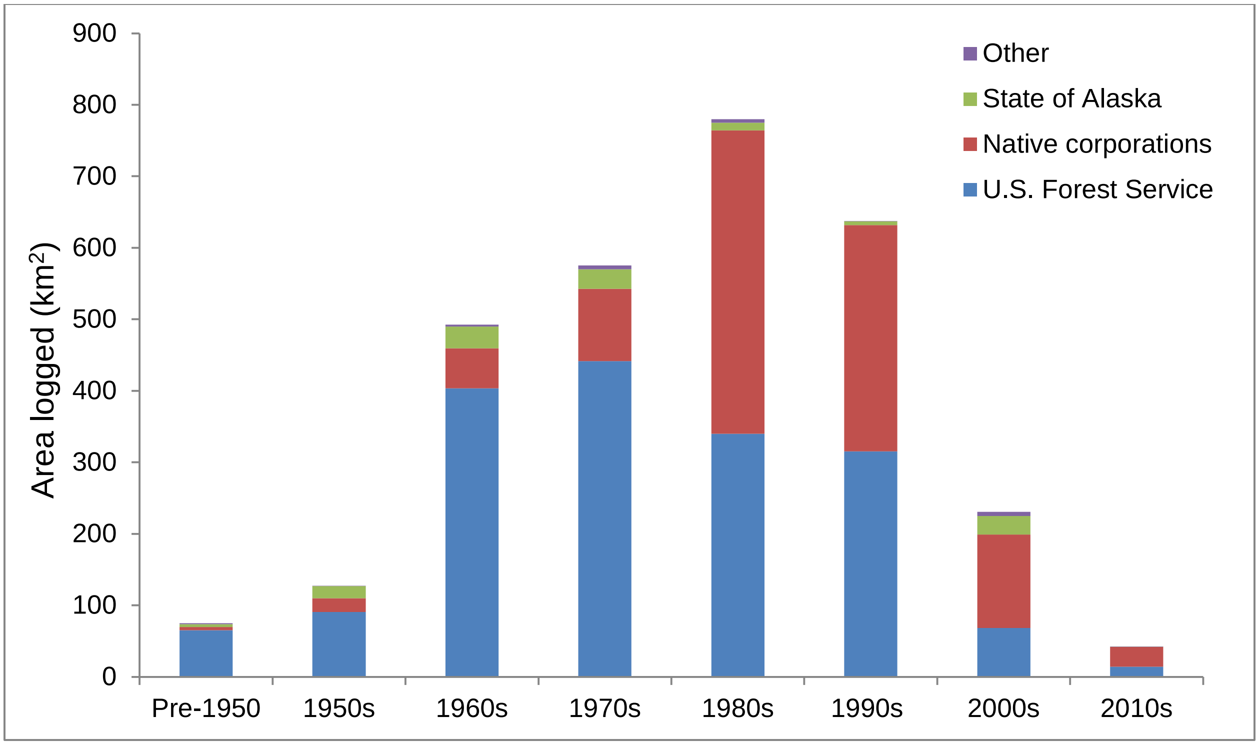

Figure 7. Age distribution of logged forest across all land ownerships in southeastern Alaska (Shanley 2015). Data from 2010s reflects harvest through 2013.

Although most (58%) of the logging in southeastern Alaska has occurred on USFS land, Native Corporations, which own only 3% of the land area, account for roughly one-third of the logging based on area harvested (Table 11, Figure 7). This reflects the higher rates of harvest on lands owned by Native Corporations (56% of their productive forest harvested to date) compared to USFS land (8% of the productive forest harvested; Table 12). These data are based on current ownership of the land and may overestimate the amount of logging accomplished by Native Corporations if young-growth now in Native Corporation ownership was originally logged while managed by the USFS or others. Nonetheless, combined USFS and Native Corporation lands currently account for over 95% of the area logged in southeastern Alaska (Figure 7).

Table 12. Current condition (km^2) of forest stands by land ownership and management, southeastern Alaska (Shanley 2015). Land owners and managers include U.S. Forest Service (USFS), National Park Service (NPS), State of Alaska (State), and Native Corporations.

Land owner or manager	Current forest condition (km^2)			% of forest logged
	Productive old-growth	Young-growth	Total forest	
USFS	19,903	1,739	21,642	8.0
NPS	827	1	828	0.1
State	799	120	919	13.1
Native Corporations	854	1,080	1,934	55.8

Intensity of timber harvest has not occurred evenly across southeastern Alaska (Figure 8). Initially, harvest was concentrated along marine shorelines near mines and towns to support early industry (primarily mining and fishing) and community development. However, after mills were built in Ketchikan, Sitka, and Wrangell in the 1950s, areas designated specifically for timber harvest were targeted in order to supply those mills. As a result, substantial timber harvest occurred on POW, Revillagigedo, and surrounding islands for delivery to the Ketchikan pulp mill, on northern Baranof and eastern Chichagof islands to support the Sitka pulp mill, and portions of Wrangell, Etolin, and Mitkof islands for the Wrangell sawmill. Native corporations have logged on many islands including POW and surrounding islands, Revillagigedo, Kupreanof, Kuiu, Admiralty, Baranof, and Chichagof, as well as portions of the mainland (e.g., Hobart Bay, Port Houghton). In addition, logging has occurred on State lands on the northern mainland near Haines and Yakutat and on islands in the southern portion of the region, including POW, Gravina, and Revillagigedo (Figure 8).

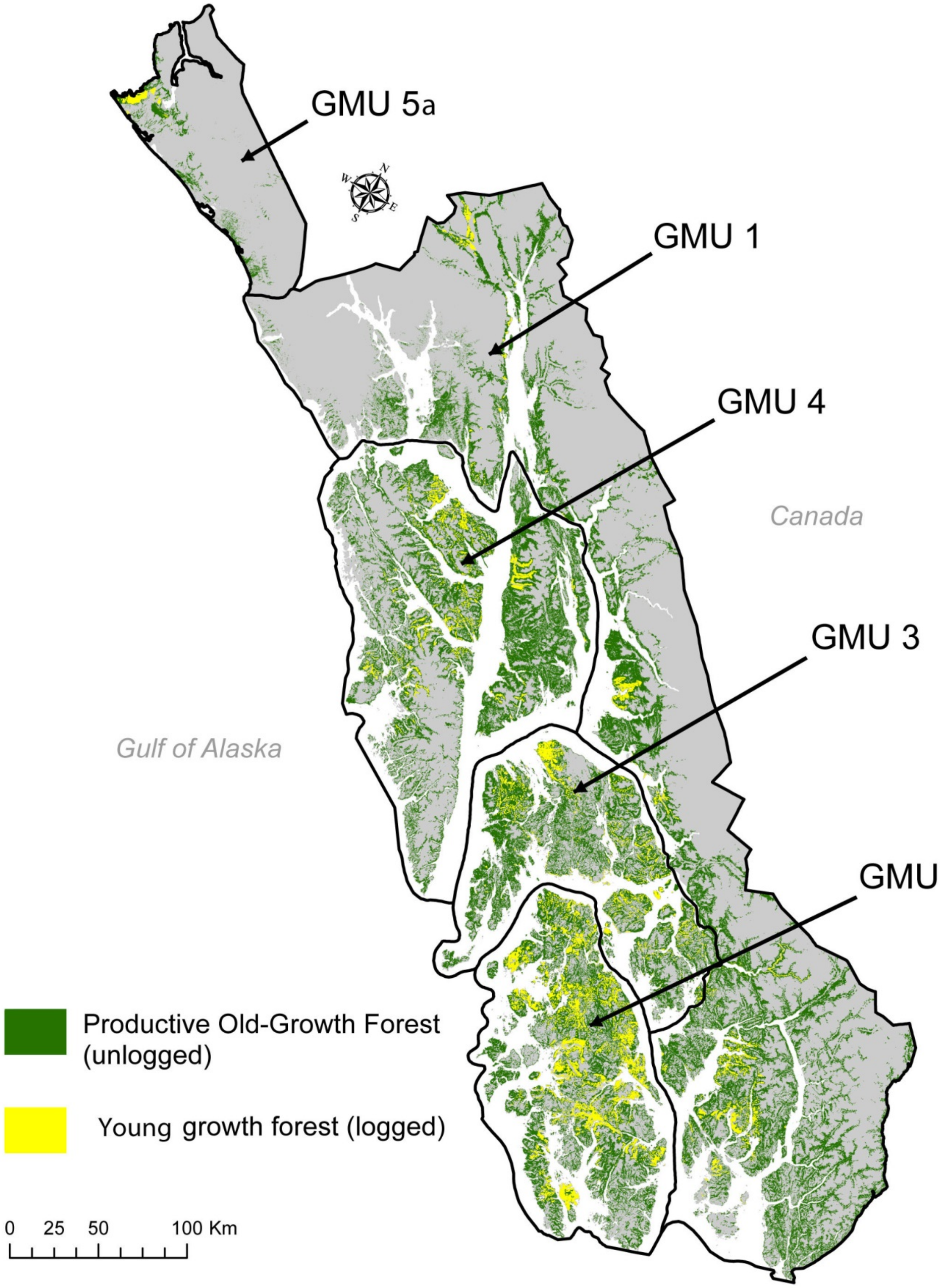

Figure 8. Current distribution of (unlogged) productive old-growth forest and (logged) young-growth forest across southeastern Alaska with Game Management Unit (GMU) boundaries (Shanley 2015). Gray areas are unforested.

Across all of southeastern Alaska, the highest rates of logging (% of productive forest harvested) have occurred in GMU 2 where about 30% of the productive old-growth forest has been logged (Figure 8, Table 13). POW was one of the primary sources of timber for the pulp mill in Ketchikan (which is now closed), as well as a sawmill in Klawock (which continues to operate), in addition to supporting most of the Native Corporation lands devoted to timber production (Figure 6). Overall, logging rates in GMU 2 are at least twice those in all other GMUs and over the range of the Alexander Archipelago wolf in southeastern Alaska (Table 13).

Table 13. Current condition (km^2) of forest stands by Game Management Unit (GMU) and within the range of the Alexander Archipelago wolf (i.e., minus GMU 4), southeastern Alaska (Shanley 2015).

Land owner or manager	Current forest condition (km^2)			% of forest logged
	Productive old-growth	Young-growth	Total forest	
GMU 1	9,800	582	10,382	5.6
GMU 2	5,560	1,639	7,199	22.8
GMU 3	4,177	603	4,780	12.6
GMU 4	6,331	584	6,915	8.4
GMU 5A	512	83	595	13.9
Total	26,380	3,491	29,871	11.7
Within the range of the wolf	20,049	2,907	22,956	12.7

The age distribution of logged stands is of particular importance to deer, the primary prey of the Alexander Archipelago wolf (see *Deer habitat use*). Generally, stands less than 25 years of age are used by deer because they produce abundant forage, but young-growth stands greater than 25 years of age provide little forage for deer due to canopy interception of sunlight and are avoided. These low-forage conditions can last for another 150 years, until natural disturbances or further timber harvest disrupt the uniform structure of the forest canopy (Alaback 1982, pp. 1936–1942). In GMU 2, where the vast majority of timber harvest occurred, harvest rates were high from the 1960s and the 1990s with a notable peak in the 1980s (Figure 9); therefore, most of the young-growth stands in GMU 2 currently are roughly 15 to 55 years of age, with the large 1980s cohort currently entering the old young-growth age that is poor habitat for deer. Although other GMUs were logged over a similar time period (Figure 9), the rate of harvest in those GMUs was considerably lower (Table 13), underscoring the compromised current condition of GMU 2.

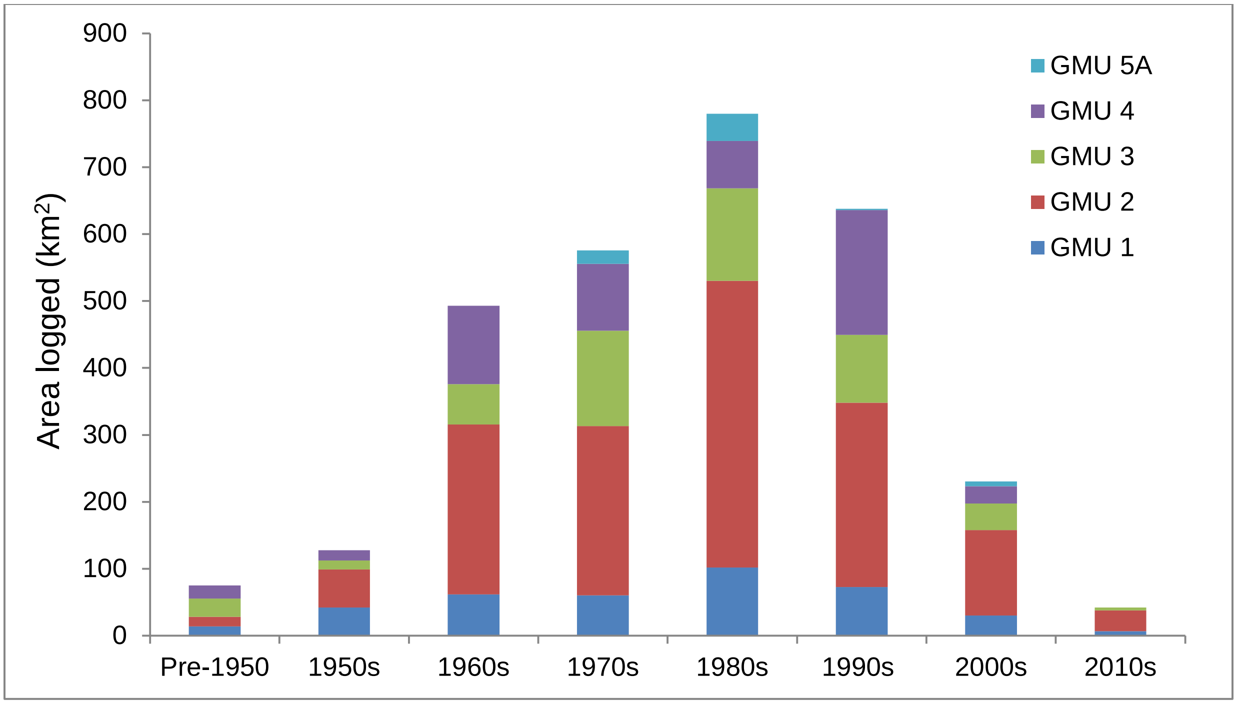

Figure 9. Age distribution of logged forest by Game Management Unit (GMU), southeastern Alaska (Shanley 2015). Data from 2010s reflects harvest through 2013.

Regeneration of a forest stand following logging typically results in dense stands of young trees that compete with each other for light and nutrients. Removal of some trees in these stands (i.e., precommercial thinning), is a common silvicultural practice used to encourage growth in fewer, larger trees, improving lumber quality and reducing time until subsequent harvest. While precommercial thinning also stimulates the growth of understory (i.e., shrub and forb) biomass, it produces dense slash as a byproduct of thinning, which may reduce use of thinned stands by deer and other wildlife. In addition, as the canopy closes in regenerating stands following precommercial thinning, understory vegetation is shaded and declines; in approximately 10 years, most understory browse once more is eliminated unless additional thinning or other intermediate treatments are conducted (Alaback 1982, pp. 1936–1942; USFS 2008b, pp. 3-329–3-330; Hanley et al. 2013, pp. 1–3).

Typical “precommercial” thinning (done before the trees are large enough to be commercially valuable) reduces stem densities (trees per unit area) to a predetermined spacing (e.g., 4 m by 4 m) or to a variable spacing (depending on distribution of dominant trees). Slash produced by thinning usually is left to decompose in the stand where it was cut. Heavy slash accumulations probably interfere with movement of deer (and other wildlife) through thinned stands (Hanley et al. 2013, p. 20), suggesting that the value of thinning to improve deer habitat may be overestimated. Importantly, deer utilization of precommercially thinned stands in southeastern Alaska has not been studied.

Alternative young-growth treatments include pruning (removal of lower branches to improve light penetration and grow knot-free lumber), creation of gaps (removal of small patches of young-growth trees to create openings), and girdling (cutting through the inner bark of live trees to kill them but leave them standing). In some cases, slash may be cut into smaller pieces to improve access through the stand. Planting alder among regenerating conifers has also been suggested as a method for increasing understory shrub and forb production. Some of these alternative treatments are being evaluated in the Tongass-Wide Young-Growth Studies, with early results (4–8 years post-treatment) suggesting that forage production is improved by thinning and pruning, especially in younger stands, by girdling, but not by alder planting (Hanley et al. 2013, pp. 36–37).

Response of understory browse species to young-growth treatments was much lower in treated stands greater than 35 years old than it was in younger stands (Hanley et al. 2013, pp. 24–29). Approximately 43% of the young-growth on the Tongass is older than 35 years (Figures 7, 9). We do not know how much of this has been thinned (and therefore might produce more browse than unthinned stands), but it is likely that future thinning would have limited benefits for deer in most of the older young-growth in southeastern Alaska. Accumulations of thinning slash that limit availability of forage by interfering with deer movement would further reduce the short-term potential forage benefits of stands treated by traditional precommercial thinning (Hanley et al. 2013, p. 20).

The creation of canopy gaps is a promising technique for stimulating growth of the understory to improve forage for deer and other herbivores. Canopy gaps, unlike precommercial thinning, create small openings in the forest that persist for several decades (Alaback 2010, p.16), and thus could improve forage for deer over a longer timescale. In addition, deer utilization of forage in canopy gaps was high (based on browse surveys, Alaback 2010, pp.7–8), indicating that deer use gap habitats. However, canopy gaps currently are not widely implemented, although precommercial thinning is common.

Since 1979, over 400 km^2 of young-growth on the Tongass has been precommercially thinned, primarily to promote timber production, but also with the intention of improving conditions for deer. In recent years, an average of 23 km^2 has been thinned annually (USFS 2008b, p. 3-329). In addition, young-growth treatments designed to improve habitat have been implemented across a limited area, and include 15 km^2 of riparian thinning, 2.6 km^2 of created corridors, 25 km^2 of created canopy gap openings, and 18 km^2 of upland wildlife thinning (USFS 2015b, p. 3). Approximately 26 to 38 km^2 of young-growth are scheduled for precommercial thinning annually between 2015 and 2019 (USFS 2014c).

Sealaska, the regional Native Corporation established by ANCSA, also has an active thinning program that targets young-growth stands at an early age of development, resulting in less slash and higher retention of the understory compared to stands on the Tongass. Since 1993, Sealaska has treated over 200 km^2 of young-growth on their timber lands and, over the next 5 years, the corporation intends to thin approximately 12 km^2 annually. In addition, Sealaska has “pruned”, i.e., removed the lower branches of the trees, to reduce competition and improve understory development, on approximately 17 km^2 of young-growth forest, with plans to prune approximately 2.8 km^2 annualy over the next 5 years (Kleinhenz 2015, p. 1).

Coastal British Columbia.—Between the 1900s and 1990s, timber harvest on public and private lands in British Columbia increased 10-fold, levelling off in the 1990s (BCMF 2010, p. 137).

Beginning in 1949, the government set allowable annual cut limits in an attempt to regulate the growing timber industry on public lands and some private lands. Over the last 10 years, the average total timber harvest across the province was 78 million cubic meters per year and most (89%) came from forests that were regulated by allowable annual cut limits. On these forests, actual harvest typically is below the cut limit; over the last 10 years, average annual harvest was 69 million cubic meters per year, but the allowable cut was 78 million cubic meters per year, roughly 12% below the permissible level (p. 138). However, the provincial harvest in 2005 peaked at 90 million cubic meters per year, which was well above the average annual harvest and the allowable cut. Harvest levels rapidly decreased since then and are now below the average due to market conditions (pp. 140–141).

More than half of the old-growth forest in coastal British Columbia remains intact, although the percentage is much lower in Regions 1 and 2, especially in productive Douglas-fir forests (MacKinnon 2003, p. 483; Figure 10). Using an integrated land cover developed for the transboundary area by the North Pacific Landscape Conservation Cooperative (available at http://www.nplcc.databasin.org), we estimated percent of forest logged in each Region of coastal British Columbia. Based on that data layer, we determined that across all of coastal British Columbia, 24% of the forest was harvested with a larger percentage harvested in Region 1 (34%) than all other regions (Region 2=12%, Region 5=14%, and Region 6=17%). Most of the offshore islands adjacent to the mainland remain unharvested (Figure 10).

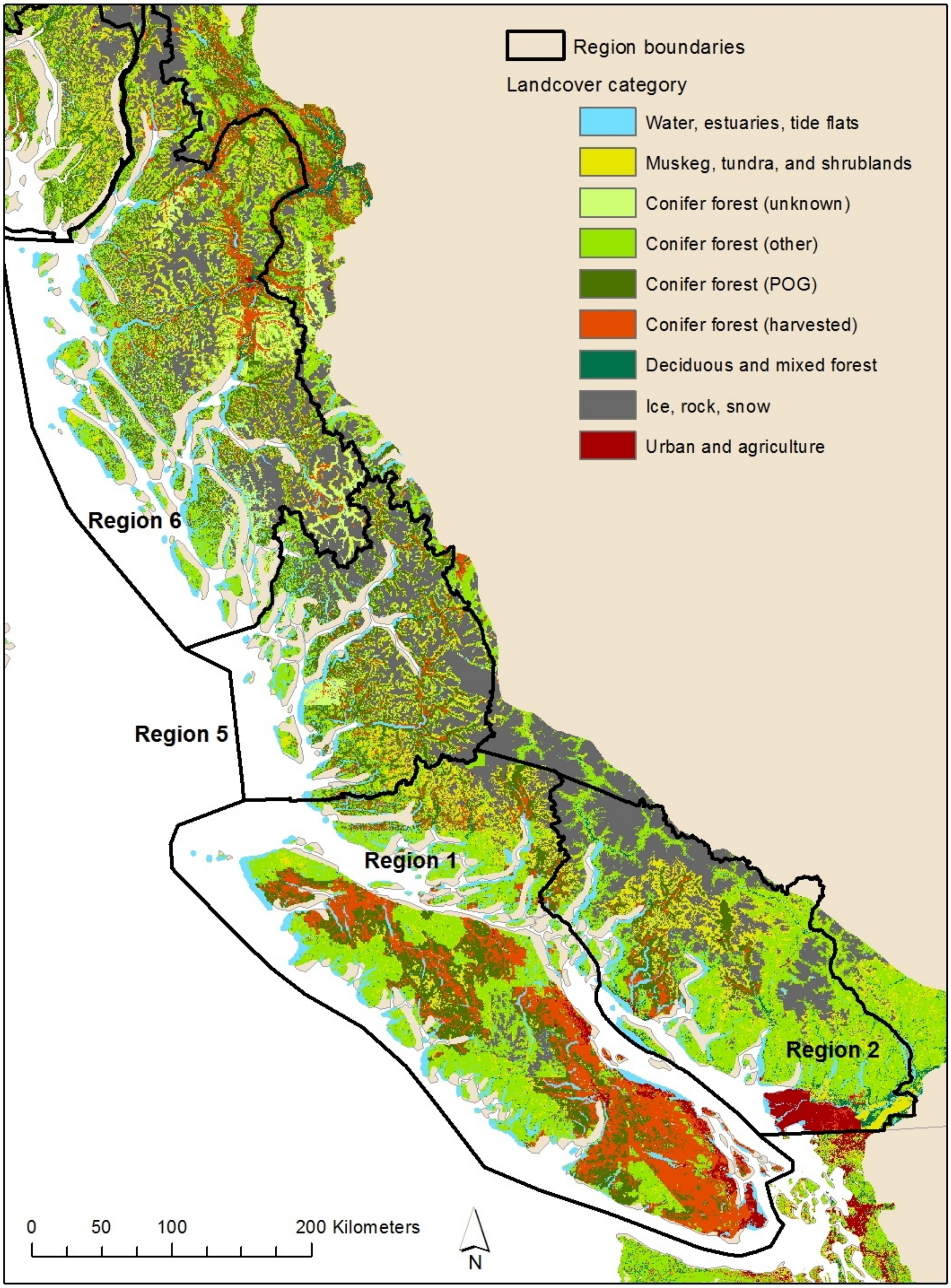

Figure 10. Map depicting current land cover in coastal British Columbia (available at: https://sciencebase.gov/catalog/item/558474dae4b023124e8f5969; accessed July 10, 2015).

Future timber harvest

Southeastern Alaska.—In an effort to meet demand for timber, the USFS aims to keep an adequate volume of timber available for sale to operators in southeastern Alaska (USFS 2008d, pp. G-1–G-3). The current Five Year Sale Schedule and Contract Plan produced by the USFS for Tongass identifies 31 timber sales that will provide an estimated 331 mmbf of timber between 2015 and 2019. Most of this timber is scheduled to be sourced from GMUs 2 and 3 (Table 14; USFS 2014d).

Table 14. Timber volume (mmbf) scheduled to be for sale on the Tongass National Forest between 2015 and 2019 by Game Management Unit (GMU), southeastern Alaska (USFS 2014d).

Year	GMU 1	GMU 2	GMU 3	Total
2015	15.00	24.88	10.00	49.88
2016	10.00	18.00	35.00	63.00
2017	12.00	20.00	44.00	76.00
2018	2.00	36.50	45.50	84.00
2019	28.00	25.00	5.00	58.00
Totals	67.00	124.38	139.50	330.88

Before the USFS can offer timber for harvest, it must first evaluate and disclose the potential effects of proposed logging in an environmental assessment (for smaller projects) or an environmental impact statement (for large projects with more significant impacts), under the authority of the National Environmental Policy Act (NEPA). After this analysis is complete, approval is documented in a Record of Decision and the timber is referred to as “NEPA cleared.” At this point, unless the USFS is challenged legally, the timber may be sold. As of June 2015, timber totaling an estimated 205 mmbf from 12 different NEPA project decisions remained NEPA cleared but unsold (Table 15; Sever 2015). It is difficult to convert this volume of wood accurately to an area that will likely be logged because many of these projects involve partial harvest.

After the timber is sold, it is considered to be “under contract” and may be cut by the purchaser. As of April 2015, the Tongass had 56 contracts in place with approximately 150 mmbf of timber remaining that was not yet cut (Table 15; USFS 2015c). Most (78%) of the projected harvest (including volume under contract, scheduled volume, and NEPA-cleared volume) is located in GMUs 2 and 3 where roughly 64% of the past harvest occurred in southeastern Alaska (Table 13). GMUs 1 and 4 are projected to contribute proportionately less to the total future harvest compared to the past. GMU 5A will continue to play a minor role (Tables 11, 12).

Table 15. Timber volume (mmbf) by Game Management Unit (GMU) and status on the Tongass National Forest, southeastern Alaska, as of June 2015 (USFS 2014d; USFS 2015c). Timber cleared under the National Environmental Policy Act (NEPA) has been reviewed and approved, but not sold. Uncut timber under contract has been sold and may be cut at any time.

GMU	NEPA-cleared volume of timber (unsold)	Uncut volume of timber under contract (sold)	Total projected harvest
1	31.5	4.5	36.0
2	54.9	110.5	165.4
3	76.7	26.6	103.3
4	41.8	8.8	50.6
5A	0.0	0.0	0.0
Total	204.9	150.5	355.3

We acknowledge a large degree of uncertainty associated with planning, selling, and cutting timber on the Tongass. Between 2008 and 2013, an average of 28.29 mmbf of timber was sold, roughly 11% of the annual allowable sale quantity under the current Tongass Forest Plan (267 mmbf). Timber sales on the Tongass often have been delayed during the planning process and due to litigation surrounding individual project decisions. In some cases, timber is offered for sale, but not sold due to a lack of bidders. Thus, with respect to future timber harvest on the Tongass, it is likely that the projected harvest described in this section will not be implemented fully or on schedule; alternatively, sales and projects currently not on the schedule could be prioritized in the future.

We did not find data on future timber harvest planned for private lands. In December 2014, Congress passed legislation authorizing transfer of 18 parcels totaling 283 km^2 of USFS land on the Tongass to Sealaska. Most of this land (approximately 277 km^2; 98%) reportedly will be managed for natural resource development (primarily timber harvest), with the balance to be preserved as cultural sites or used for small economic development opportunities (Sealaska 2014, p. 1).

Across all land ownerships, nearly all timber harvested to date has been from old-growth forests. In July 2013, the Secretary of Agriculture issued a memorandum directing the Tongass to transition to a young-growth-based timber program in 10–15 years (U.S. Department of Agriculture 2013), much sooner than outlined in the Tongass Forest Plan. To implement this direction, the Forest Service currently is developing and evaluating alternatives to accelerate harvest of existing young-growth forest and reduce harvest of old-growth substantially, while providing adequate timber to support existing industry. An amendment of the Tongass Forest Plan is underway and is expected to be completed by the end of 2016 (80 Federal Register 35934, June 23, 2015).

Most of the oldest young-growth forest, which provides the first available harvest under this transition, is located at low elevations along marine shorelines. Under the current Tongass Forest Plan, timber harvest is not allowed in these areas because of their high value to wildlife (e.g., deer winter habitat; see *Deer habitat use*). The USFS is considering alternatives as part of the Tongass Forest Plan amendment process that would permit commercial harvest in these older young-growth stands along the beach. Because these stands tend to be effective at intercepting snow and providing thermal cover and forage for deer during winter, harvest of these low elevation, young-growth stands along the beach may impact deer negatively, particularly in deep-snow winters.

Coastal British Columbia.—Forests with government-regulated allowable annual cuts in British Columbia are forecast to be stable over the long term. However, in the shorter term, some units in the interior part of the province are experiencing widespread tree mortality due to the mountain pine beetle (*Dendroctonus ponderosae*) epidemic. Therefore, the provincial government predicts reduced timber supply from the interior forests and potential increased supply in some coastal forests. For now, the actual timber harvest is occurring at reduced levels throughout British Columbia and allowable annual cut limits and forecasts will be revisited in 2020 (BCMF 2010, p. 141).

Between 2005 and 2100, an additional 17% of old-growth forest was projected to be harvested on Vancouver Island and additional 39% was expected to be logged on the mainland of coastal British Columbia (Service 2010, p. A-12). These rates equate to roughly 1% per year across coastal British Columbia. However, some of this timber volume would be harvested from old young-growth stands. Regardless, based on the information we found, the timber industry in British Columbia is faced with substantial uncertainty in the near future due to market conditions and insect infestations.

Effects of timber harvest on wolves.— The only potential direct effect from timber harvest to Alexander Archipelago wolves for which some evidence exists is the modification of and disturbance at den sites. Person and Russell (2009, p. 220) found that most used den sites were located farther from logged stands and roads than unused locations; on average, dens were located 988 m from logged stands and 1,351 m from roads (p. 217). Wolves never located dens in clearcuts or young-growth forests (p. 221). Although wolves were tolerant of short periods of researcher disturbance, the authors believed that wolves avoided logged areas and roads because of regular and long-term use by humans. To minimize destruction of and disturbance at den sites, the USFS developed standards and guidelines specific to active wolf dens in or near areas where timber harvest is occurring (summarized in *Existing conservation mechanisms*). These efforts are required to be implemented on federal lands in southeastern Alaska only (roughly 74% of land; Figure 6, Table 11). We are not aware of similar conservation guidelines for wolf den sites in coastal British Columbia.

Other potential direct effects from timber harvest may exist, such as loss of rendezvous sites and movement corridors, but we found no evidence suggesting that these effects are impacting individuals or populations of Alexander Archipelago wolf. Nonetheless, we acknowledge that wolves in southeastern Alaska and coastal British Columbia may be impacted directly by timber harvest in undocumented ways, but we have no basis from which to evaluate them.

Throughout their circumpolar distribution, gray wolves exhibit a high degree of plasticity in their use of different habitats. As a species, their presence is thought to be linked to a sufficient supply of prey, primarily ungulates, rather than a preference for specific habitat types. On POW, Person and Ingle (1995, p. 30) found that wolves used young-growth habitat significantly less than expected based on its availability and appeared to be selecting for unharvested forests, presumably due to high cost of movement and low visibility, resulting in poor hunting conditions for wolves. Wolves also preferentially use low elevation areas, i.e., below 400 m (Person 2001, p. 62). Based on these findings and others summarized above (*Resource selection* and *Wolf-deer habitat relationships*), the effect of timber harvest on wolves probably is mostly indirect and is driven by the presence and availability of deer and other prey at low elevations and, for forest-associated prey, in unharvested forests. Therefore, below, we consider potential functional and numerical responses of Alexander Archipelago wolves to habitat modifications from timber harvesting, and to changes in the abundance of deer and other prey.

Deer.—Sitka black-tailed deer are found on all islands within the range of the Alexander Archipelago wolf and the mainland except in GMU 5A (other than a small introduced population; MacDonald and Cook 2007, p. 102; Nelson et al. 2008, p. 7). Many field studies have established that deer selectively use productive old-growth forests in winter, especially during periods of deep snow (e.g., Schoen and Kirchhoff 1990, p. 374), although physical site characteristics may be better predictors of use in some areas and during some winters (e.g., Doerr et al. 2005, p. 322). Early seral stage forests provide some benefits to deer as the result of increased production of shrub biomass, but these benefits are short-term; additionally, this forage may be buried in snow in winter and may be of lower quality for deer (Hanley 2005, p. 105). Once regenerating forests are >25 years old, they provide fewer benefits to deer (e.g., Person et al. 2009, p. 5), and this condition extends for more than 100 years until the understory returns fully (Alaback 1982, p. 1939). Populations of deer in those areas of intensive timber harvest are expected to decline as the result of long-term reduction in the carrying capacity of their winter habitat; for example, in GMU 2, Person (2001, p. 79) predicted a 28% decline in deer between 1995 and 2045. During winters with deep snow, extreme levels of mortality from malnutrition also is predicted, as well as increased susceptibility to wolf predation as deer concentrate in smaller patches of winter range (e.g., McNay and Voller 1995, p. 138; Farmer et al. 2006, p. 1412; Person et al. 2009, p. 8). We describe deer habitat use in more detail above (see *Deer habitat use*).

We are not aware of any long-term data that document declines of deer in southeastern Alaska or coastal British Columbia at the watershed or landscape scale relative to timber harvest, yet projections of long term declines in deer numbers, based on an evaluation forage, are well founded. These projections have been generated using several models developed over the last 30 years aimed to estimate carrying capacity of deer (e.g., Fagen 1988; Hanley and Rogers 1989; Kirchhoff et al. 1990) and more models continue to be developed (e.g., Forage Resource Evaluation System for Habitat; Hanley et al. 2015). The model most often used in conservation planning and management for deer originally was developed by Suring et al. (1993), although it has undergone some revision since then (e.g., Schoen and Kirchhoff 2007, p. 9; hereafter we refer to it as the deer habitat model). This model estimates deer habitat capability, which is an index of carrying capacity, during winter when deer populations are assumed to be most limited. Variables included in the model are elevation, slope, aspect, and stand age and size. The model was not intended to predict actual populations or densities of deer in the future, but instead to estimate the maximum number or density of deer that a specific habitat (or defined area) can support indefinitely. These estimates can be useful in assessing the current and future conditions for deer and therefore wolves, given that deer are their primary prey, assuming that deer habitat capability as expressed in the deer habitat model is a reliable indicator of deer abundance.

Using the latest version of the deer habitat model, Albert and Schoen (2007, p. 31) modeled and mapped deer habitat capability in two time periods (1954 and 2002) across all land ownerships in southeastern Alaska. As of 2002, 79% of the original deer habitat value still remained in southeastern Alaska (p. 2-15). Within the range of the Alexander Archipelago wolf, the highest deer habitat capability was in GMU 2 (47%), followed by GMU 3 (34%), and the remainder in parts of GMUs 1A and 1B (19%); most of the mainland of southeastern Alaska was not included in their analysis because it was considered as generally poor winter habitat for deer (p. 2-16). Between 1954 and 2002, deer habitat capability changed the most (62% of its original value) on northern POW and the least (89%) on southern POW. In GMU 3, habitat capability in 2002 was 77–87% of that in 1954, and in GMUs 1A and 1B it was 85% of its original value. To the best of our knowledge, their modeling effort provides the most current published estimates of deer habitat capability throughout all of southeastern Alaska.

We used the deer habitat model to predict habitat capability of deer in GMU 2 out to year 2045 and then used the output to model the GMU 2 wolf population over the same time period. Our primary goal was to understand the effects of past and future timber harvest on future wolf abundance. Across the range of the Alexander Archipelago wolf, GMU 2 has been disproportionately harvested for timber compared to other GMUs and Regions (Figure 8, Table 13) and therefore, our results should represent the apparent worst-case scenario for deer and wolves. We acknowledge that our wolf population model is subject to the assumptions of the deer habitat model, but in the absence of empirical data on deer abundance, we believe it is reasonable and logical to use deer habitat capability as an indicator of deer abundance.

We estimated deer carrying capacity annually under six possible future vegetation conditions in GMU 2 (Table 16): Steady K (constant carrying capacity beginning in 2015), No future harvest (referring to timber harvest, natural succession only), Transition SG (transition of timber harvest from old-growth to second-growth), Continued OG (continued old-growth harvest; comparable to recent rates of harvest between 2008 and 2014), Increased OG (increased old-growth harvest; comparable to rates of harvest between 1995 and 2000), and Max OG (maximum old-growth harvest; full implementation of the Tongass Forest Plan). We assumed a predation rate of 15 deer/wolf/year as estimated from Szepanski et al. 1999 (p. 331, calculated following Person et al. [1996, p. 42]) and a beginning deer abundance as 75% of the carry capacity in 1995. Our model also included other factors such as predation that may limit deer abundance. See Gilbert et al. (2015) for details about the wolf population model.

Table 16. Description of vegetation conditions and assumptions for rate of harvest by land ownership and management under which we modeled changes in wolf abundance between 2015 and 2045 in Game Management Unit 2, southeastern Alaska. Rate of harvest is in millions of board feet of timber (mmbf). See Gilbert et al. (2015) for more detailed description of vegetation conditions.

Vegetation condition	Land ownership and management					
	USFS	Sealaska Land Finalization Act	Mental Health Land Exchange	State Forest	Other ANCSA[1] Corporation Lands	Total harvested
Steady K	Not applicable – deer carrying capacity in 2015 assumed constant through 2045					None
No future harvest	Not applicable – natural succession only					None
Transition SG	Second-growth and old-growth projects as described in current USFS 5-year Schedule of Activities[2]	28 mmbf/year	No exchange (remains USFS land)	11 mmbf/year	14 mmbf/year	53.0 mmbf/year, plus projects included in USFS Transition Schedule
Continued OG	12.6 mmbf/year from GMU 2[2]	28 mmbf/year	15 mmbf/year	11 mmbf/year	14 mmbf/year	80.6 mmbf/year
Increased OG	47.6 mmbf/year from GMU 2[2]	56 mmbf/year	30 mmbf/year	22 mmbf/year	28 mmbf/year	183.6 mmbf/year
Max OG	69.2 mmbf/year from GMU 2[2,3]	84 mmbf/year	45 mmbf/year	33 mmbf/year	42 mmbf/year	273.2 mmbf/year

[1] Alaska Native Claims Settlement Act.

[2] Assumes that the Big Thorne Timber Sale proceeds as sold.

[3] Assumes that administrative protections applied under the 2001 Roadless Rule are repealed for the Tongass National Forest.

Based on our modeling effort, we found that impacts of past and future timber harvest were greater for deer than for wolves under all six vegetation conditions (Figure 11). Percent of mean population change of wolves between 2015 and 2045 ranged from -4% under Steady K to -10% under Increased OG, while deer ranged from -16% change under Steady K to -26% change under Max OG. Variance associated with all estimates was substantial (Figure 11) due to the stochastic structure of the model. When we held deer carrying capacity constant (Steady K), both wolf and deer abundance still were predicted to decline, indicating that current conditions on the landscape are insufficient for maintaining stable deer and wolf populations. Our results suggest that past timber harvest in GMU 2 will result in declines in deer and to a lesser extent wolves over the next 30 years, and that future timber harvest would exacerbate declines, especially for deer, but would have less of an impact compared to effects of past timber harvest.

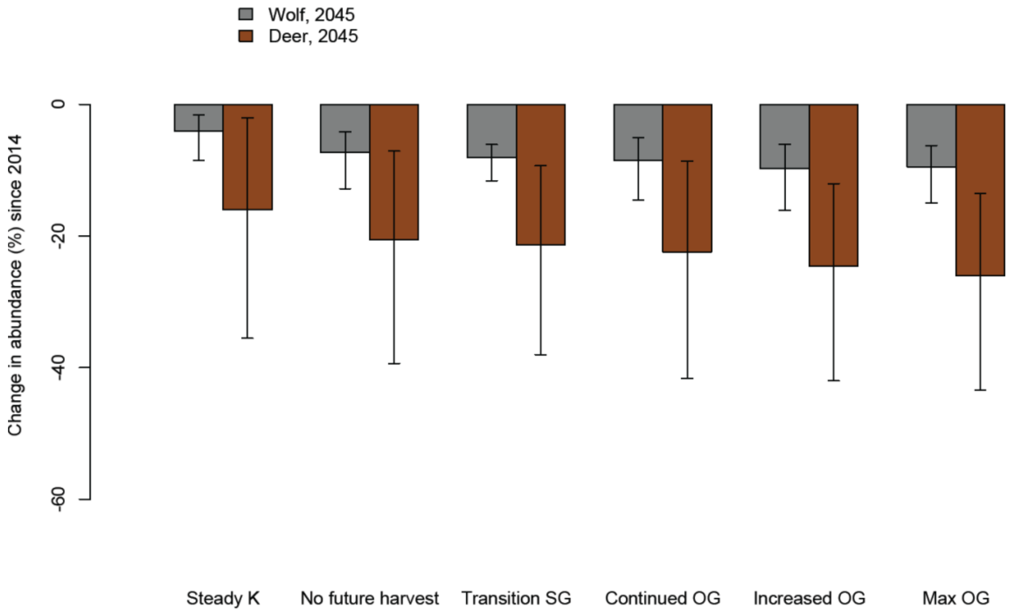

Figure 11. Estimated percent change in mean abundance of wolves and deer under six vegetation conditions between 2015 and 2045 in Game Management Unit 2, southeastern Alaska. Error bars denote 95% confidence intervals.

Our wolf population model assumes a strong positive relationship between deer and wolf abundance, which is supported by information on wolf diet in GMU 2. Although wolves are opportunistic predators and are known to shift their diet based on prey availability (see *Food habits*), the relationship between deer abundance and wolf populations likely will remain. Given expected declines in deer abundance, we explored sensitivity of the model to variation in wolf predation rates. Specifically, we ran the model using 9.5 deer/wolf/year (equivalent to ~28% deer in wolf diet, following calculations by Person et al. 1996, p. 42), 15.0 deer/wolf/year (45% deer in wolf diet, based on stable isotope analysis by Szepanski et al. 1999), 20.5 deer/wolf/year (~60% deer in wolf diet), and 26.0 deer/wolf/year (77% deer in wolf diet, based on scat analysis by Person et al. 1996, p. 42).

We found that percent in mean population change in wolves was highly sensitive to wolf diet composition (Figure 12). When wolf diet was composed of only 28% deer, percent change in wolf abundance over the 30-year period was positive (35%), but when at least 45% of wolf diet was deer, then change in abundance was negative, ranging from -8% to -54% as deer increased in the diet of wolves. Likewise, as wolf predation rate on deer increased, percent change in mean population size of deer responded accordingly, with large changes in deer abundance across wolf diet compositions (Figure 12). Thus, although our underlying model relies on a strong deer–wolf relationship, our results demonstrate that these two species are intricately linked in GMU 2.

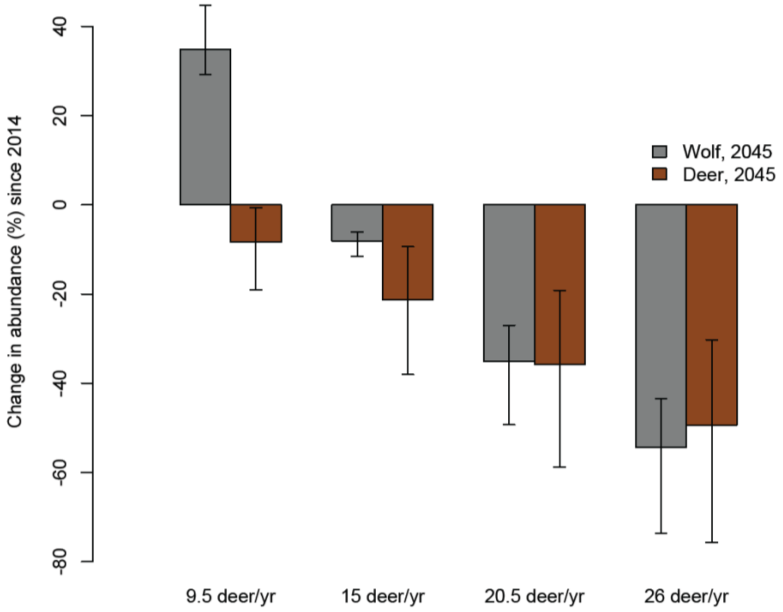

Figure 12. Modeled estimates of percent change in mean abundance of wolves and deer based on four rates of wolf predation on deer between 2015 and 2045 in Game Management Unit 2, southeastern Alaska. Error bars denote 95% confidence intervals.

Mountain goat.—Within the range of the Alexander Archipelago wolf, mountain goats are found naturally only on the mainland (MacDonald and Cook 2007, p. 104). However, recent genetic analyses suggest that they may have occupied Baranof Island (GMU 4) at one time prior to being introduced there in 1923 (Paul 2009, p. 16; Shafer et al. 2011, p. 1261). Additionally, mountain goats were successfully introduced to Revillagigedo Island in 1983 (GMU 1A; Paul 2009, p. 20). Wolves take mountain goats where the two species are sympatric, especially on the mainland where deer densities are low (Fox and Streveler 1986, p. 193).

During winter, snow forces mountain goats to forested areas at lower elevations (White et al. 2011, p. 1740). In old-growth forests, goats typically are found in close proximity to cliffs, which they use as escape terrain from predators, primarily wolves (Fox and Streveler 1986, pp. 192–194). Most of the mountain goats in southeastern Alaska occur on the mainland where the amount of old-growth forest is more limited than on islands and little of it has been harvested (Figure 8, Table 13). Thus, although forests adjacent to cliffs provide critical habitat for mountain goats during the winter, it is unlikely that timber harvest has had or will have a population level effect on mountain goats within the range of the Alexander Archipelago wolf, but some individuals or local populations (e.g., Cleveland Peninsula) may be impacted.

Moose.—Moose migrated down the major river systems from Canada into southeastern Alaska during the early 20th century. All moose populations in southeastern Alaska are natural except for transplanted populations in Berners Bay north of Juneau in Lynn Canal (1958 and 1960) and the Chickamin River (1963 and 1964; MacDonald and Cook 2007, p. 187). Within both coastal British Columbia and southeastern Alaska, moose have expanded their range over the last 50 years and continue to do so (Darimont et al. 2005, p. 235; Hundertmark et al. 2006, p. 331). They now are distributed on the mainland and on Kupreanof, Mitkof, Wrangell, Zarembo, Etolin, and Kuiu islands as well as many of the other smaller islands and apparently a small part of POW (MacDonald and Cook 2007, p. 100). Moose likely are preyed upon by wolves (see *Food habits*), wherever they co-exist.

In southeastern Alaska and coastal British Columbia, moose are associated primarily with riparian and, where available, post-glacial, early successional vegetation types (e.g., White et al. 2014, p. 227). Therefore, although moose habitat is declining in some areas as a result of natural succession, timber harvest has resulted in early successional vegetation types favorable to moose; however, such habitat is ephemeral, diminishing as old second-growth forest (>25 years of age) becomes established. Nonetheless, moose abundance is increasing in British Columbia (Kuzyk and Hatter 2014, p. 880), including the coastal regions (BCMO 2015b), and in southeastern Alaska is thought to be stable or increasing (ADFG 2010, pp. 2, 12, 54, 66, 80), or is being managed through harvest (p. 29). For these reasons coupled with the fact that little timber harvest has occurred on the mainland (Figure 8, Table 13), it seems unlikely that timber harvest has affected moose populations negatively. In fact, given that the biomass per moose is greater than that per deer, it is possible that Alexander Archipelago wolves would benefit from an expanding and growing moose population, although capture efficiency or low rates of moose population growth may negate any realized benefit (Person et al. 2001, p. 264).

Beaver.—Beavers are found on most large islands and throughout the mainland within the range of the Alexander Archipelago wolf. They are taken as prey throughout the year by wolves on POW and Revillagigedo islands where, next to deer, their remains were the second most frequently observed in wolf scats; to a lesser extent, wolves also prey on beavers in Glacier Bay and coastal British Columbia (see *Food habits*). We found no information on whether or not beavers are affected directly by timber harvest in southeastern Alaska and coastal British Columbia.

Pacific salmon.—Salmon spawn in most freshwater streams and rivers throughout southeastern Alaska and coastal British Columbia. Wolves feed on spawning salmon when available, typically during the late summer and early fall. Timber harvest can impact physical stream environments by altering hydrologic regimes, reducing quantities of large wood, increasing width-to-depth ratios of stream channels, and reducing the size of stream sediments; all of these impacts can affect juvenile salmon rearing habitat (e.g., Burnett et al. 2007, p. 66; Keeton et al. 2007, p. 852; Tiegs et al. 2008, p. 4). Although timber harvest can affect physical characteristics of freshwater streams used by salmon, it is less clear whether or not these habitat alterations result in reduced survival, reproduction, or abundance of salmon.

5.3.3. Road development

Within the range of the Alexander Archipelago wolf, road development associated with urban development, timber harvest, and other types of resource extraction has modified the landscape. Some roads are permanent (e.g., federal highways), while others, such as those built to support the timber industry, are temporary. Roads not only alter habitats used by the Alexander Archipelago wolf and its prey, but also, importantly, provide access for subsistence and recreational users to previously unroaded and remote areas. Therefore, following removal of timber (or other natural resources), road management can be controversial because most local individuals and communities prefer to have continued access along temporary roads and can be expensive owing to maintenance and decommissioning costs.

Below, we describe the existing road systems in southeastern Alaska and coastal British Columbia to the best of our ability; few road databases seem to be complete and current. We estimated road densities by WAA in southeastern Alaska and WMU in coastal British Columbia and then summarized results by GMU and Region, respectively. For the purpose of this assessment, we considered all types of roads (e.g., sealed, unsealed) to be roads provided that they offer access to humans using any motorized vehicle (e.g., off-highway vehicle, all-terrain vehicle, snowmachine, passenger vehicle). We also reviewed plans for decommissioning of roads and new construction associated with current and planned timber sales. We urge the reader to be mindful of the fact that plans for road management (e.g., decommissioning, closure, new construction) are difficult to ascertain and are not certain to be implemented.

We acknowledge that timber harvest, road construction and management, and wolf harvest interact as stressors to the Alexander Archipelago wolf and that these interactions can be complex. In this subsection, we assess potential direct impacts of roads to coastal wolves only and separately examine hunter and trapper access afforded by roads as a possible indirect effect to wolves (see *Wolf harvest*). To avoid redundancies, we present results in this subsection with regard to a road density threshold of 0.9 km/km^2, following Person and Russell (2008, p. 1548) and refer to these results when assessing wolf harvest.

<u>Southeastern Alaska</u>.—In southeastern Alaska, roads occur on federal, state, and private lands with the majority of them administered by the USFS. Outside of population centers, nearly all roads originally were built to facilitate logging and forestry-related activities. Many of these roads remain, although their status, accessibility, ownership, and maintenance vary over time. For example, the USFS stores (i.e., closes with a gate or similar temporary barrier) some roads that may be used later; the period of storage must exceed one year for a road to be considered "stored." Similarly, the USFS authorizes construction of temporary roads by contract, permit, or lease for short-term operations, yet at the end of those operations, the road remains and the ownership or responsibility of it is not necessarily clear at times. The USFS and other transportation and land management agencies recognize the need to inventory and manage for the existing road system in southeastern Alaska. Recently, several travel plans have been developed for the region (e.g., USFS 2009; Department of Transportation and Public Facilities 2004), although implementation of these plans is expensive and ongoing.

The USFS adopted a system for describing National Forest System roads based on level of required maintenance and use. Roads categorized as Maintenance Level 1 are stored and used only intermittently (i.e., closed to regular vehicular traffic); Maintenance Level 2 roads are open and used by high-clearance vehicles, but are not maintained; Maintenance Level 3 roads are open, maintained, and suitable for prudent drivers in standard passenger vehicles; and, Maintenance Levels 4 and 5 are maintained regularly and used by all vehicles. The USFS prepared Motor Use Vehicle Maps for all Ranger Districts to inform the public of road status and accessibility based on these maintenance levels. One goal of these maps was to discourage unauthorized road access, although we are uncertain of the effectiveness of their effort. Some “closed” roads are inaccessible to motorized traffic due to removal of culverts, alder planting, or other techniques; however, some roads are closed with gates, which can be circumvented by determined users with all-terrain vehicles or similar. In addition, closed roads often can be used via snow machine in winter, if snow cover is sufficient.

Currently, 10,975 km of roads exist within the range of the Alexander Archipelago wolf (72,930 km^2) in southeastern Alaska, resulting in an overall road density estimate of 0.23 km/km^2 (Albert 2015; Table 17). Most (52%) of these roads are located in GMU 2, followed by GMUs 1 and 3 (22 and 23%, respectively); GMU 5A has few of the roads (3%). However, estimates of road density, which are more meaningful and informative, are greatest in GMU 2 (0.62 km/km^2) and GMU 3 (0.26 km/km^2), and are negligible in GMUs 1 and 5A (0.08 and 0.04 km/km^2, respectively; Table 17). Within GMU 1, subunit 1A has a higher road density (0.14 km/km^2) than all of the other subunits (0.02–0.07 km/km^2).

Table 17. Mean estimates of road density and ranges calculated by Wildlife Analysis Area (WAA) with each Game Management Unit (GMU; Albert 2015). Shaded area indicates subunits of GMU 1.

GMU	Road density (km/km^2)		WAAs	
	Mean	Range	Total number	Percent with road density >0.90 km/km^2
1	0.08	0.00–1.25	74	1%
1A	0.14	0.00–1.25	26	4%
1B	0.02	0.00–0.19	17	0%
1C	0.07	0.00–0.40	26	0%
1D	0.02	0.00–0.08	5	0%
2	0.62	0.00–1.57	31	42%
3	0.26	0.00–1.06	25	4%
5A	0.04	0.00–0.25	7	0%
Total	0.23	0.00–1.57	137	11%

Across all GMUs, road density estimates in WAAs ranged from 0.00 to 1.57 km/km^2. Of 137 WAAs, 62 (45%) had no roads in them, 60 (44%) had estimated densities between 0.01 and 0.90 km/km^2, and 15 (11%) exceeded the 0.90 km/km^2 threshold above which wolf harvest rates can be problematic (Person and Russell 2008, p. 1548; Figure 13). Not surprisingly given the high road densities in GMU 2, most (13 of 15; 87%) of these WAAs are located in GMU 2; one each was located in GMU 1A and GMU 3.

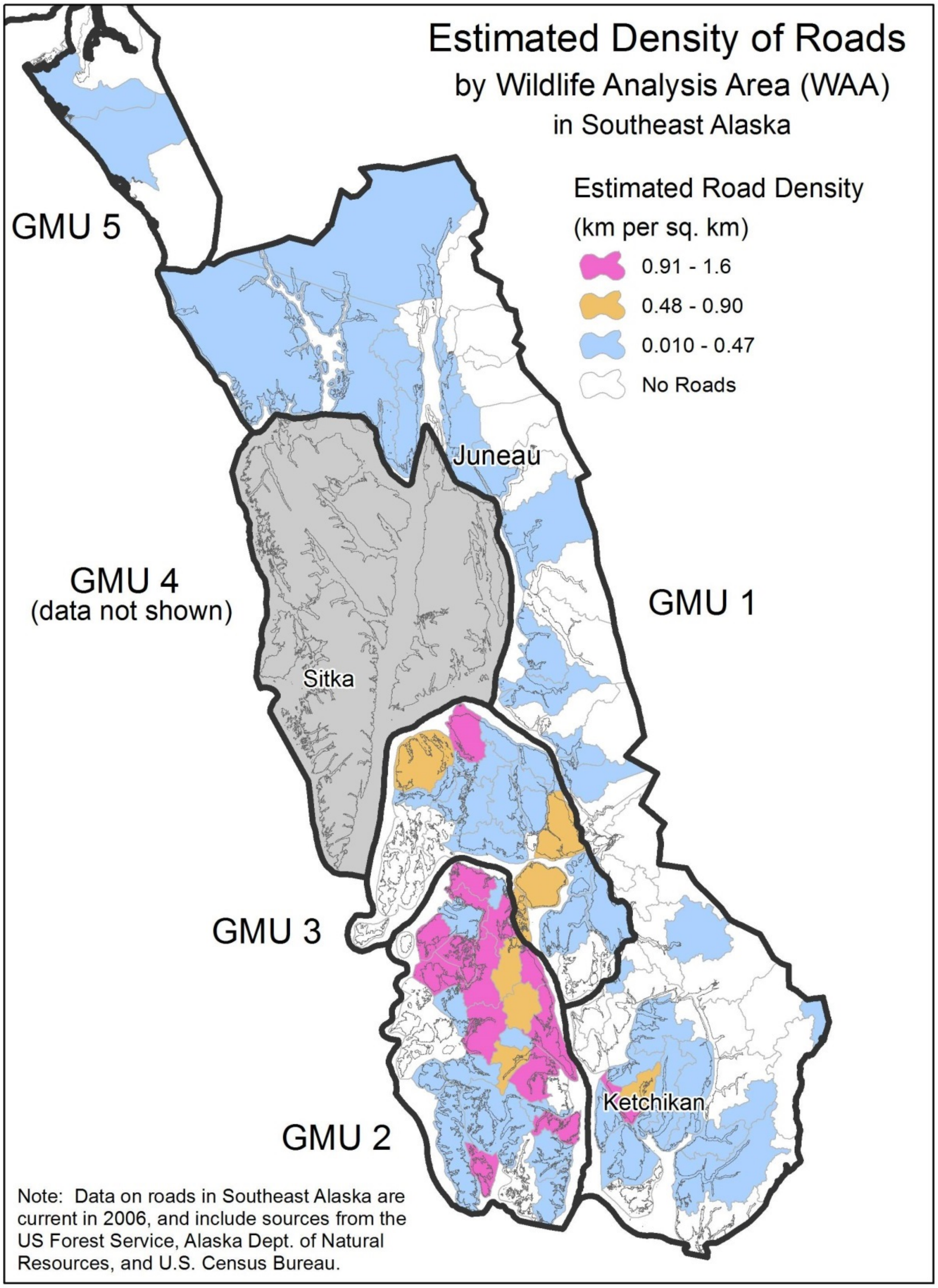

Figure 13. Map depicting road densities estimated by Wildlife Analysis Area (WAA) and presented by Game Management Unit (GMU) within the range of the Alexander Archipelago wolf in southeastern Alaska (not found in GMU 4; Albert 2015). Estimated road densities greater than 0.90 km/km^2 are considered to be problematic for wolves due to high rates of wolf harvest by humans (Person and Russell 2008, p. 1548; see *Wolf harvest*).

The USFS has been working to decommission or permanently close unused roads in southeastern Alaska. In some areas (e.g., GMU 2), the closure of roads is complex and controversial. Maintenance of roads is expensive, but so is decommissioning them. Further, many local residents are accustomed to using some of these roads for subsistence purposes including hunting, trapping, firewood collection, berry picking, etc. In 2009, the USFS drafted the POW Access Travel Management Plan that outlines a schedule for re-categorizing 2,283 km (1,419 miles; ~40% of all roads in GMU 2) of road based on Maintenance Levels 1–5, decommissioning roads, and converting roads to trails (USFS 2009, p. 2). This plan calls for decommissioning 129 km (80 miles) of road, converting 16 km (10 miles) to trail and 357 km (222 miles) to motorized trail, and storing an additional 237 km (147 miles) of road; the remainder of road will be divided into sub categories of Maintenance Level 2. Since 2009, the USFS has made progress on implementing this plan, including storage of 631 km (392 miles) of road and continued efforts are scheduled (Smith 2015).

We expect some new road construction associated with Tongass timber sales that have been sold (but not cut yet), NEPA-cleared, or are in the planning stages. By far, the biggest contributor is the Big Thorne Timber Sale in GMU 2, which requires 74 km (46 miles) of new road construction and 59 km (37 miles) of reconstruction of existing roads (USFS 2013a, p. 1). Other small sales such as Mitkof Island and Navy timber sales in GMU 3 will result in small amounts of new road (<2 km each), temporary road (<8 km), and reconstructed road (8 km). Planned sales such as Saddle Lakes Timber Sale in GMU 1A probably will require some road construction, although the total length should be less than 30 km. The USFS is aiming to shift to young-growth harvest and away from old-growth harvest, allowing for use of existing roads as opposed to constructing new ones in the future.

<u>Coastal British Columbia</u>.—In coastal British Columbia, roads often are described as being “connected” or “unconnected” to the main road system. In total, 67,612 km of road exist within the range of the Alexander Archipelago wolf, resulting in a mean road density of 0.47 km/km^2 (Table 18; Albert 2015). The majority of roads are located in Region 1 (41%) and Region 2 (31%), although Region 6 in the north supports 22% of the total road length; Region 5 has few roads (6% of the total). Mean road density estimates follow a similar pattern with the highest densities in Regions 1 and 2, followed by Region 6, then Region 5 (Table 18).

Table 18. Mean estimates of road density and ranges estimated by Wildlife Management Units and summarized by Region, coastal British Columbia (Albert 2015).

Region	Road density (km/km^2)		WMUs	
	Mean	Range	Total number	Percent with road density >0.90 km/km^2
1	0.53	0.05–1.07	15	27%
2	0.55	0.06–3.03	13	15%
5	0.13	0.09–0.17	3	0%
6	0.30	0.07–0.89	5	0%
Total	0.47	0.05–3.03	36	17%

Across all Regions, estimated road densities ranged from 0.05 to 3.03 km/km^2 (Table 18). Six of 36 (17%) WMUs exceeded the threshold of 0.90 km/km^2 presumed to be negative for wolves; all of these WMUs were located in Regions 1 and 2 and surrounded the large cities of Victoria and Vancouver (Figure 14). One WMU in Region 6 had an estimated road density of 0.89 km/km^2, which was high compared to the other four WMUs in that region (0.07–0.21 km/km^2). We did not find data indicating planned road construction or future road management in coastal British Columbia.

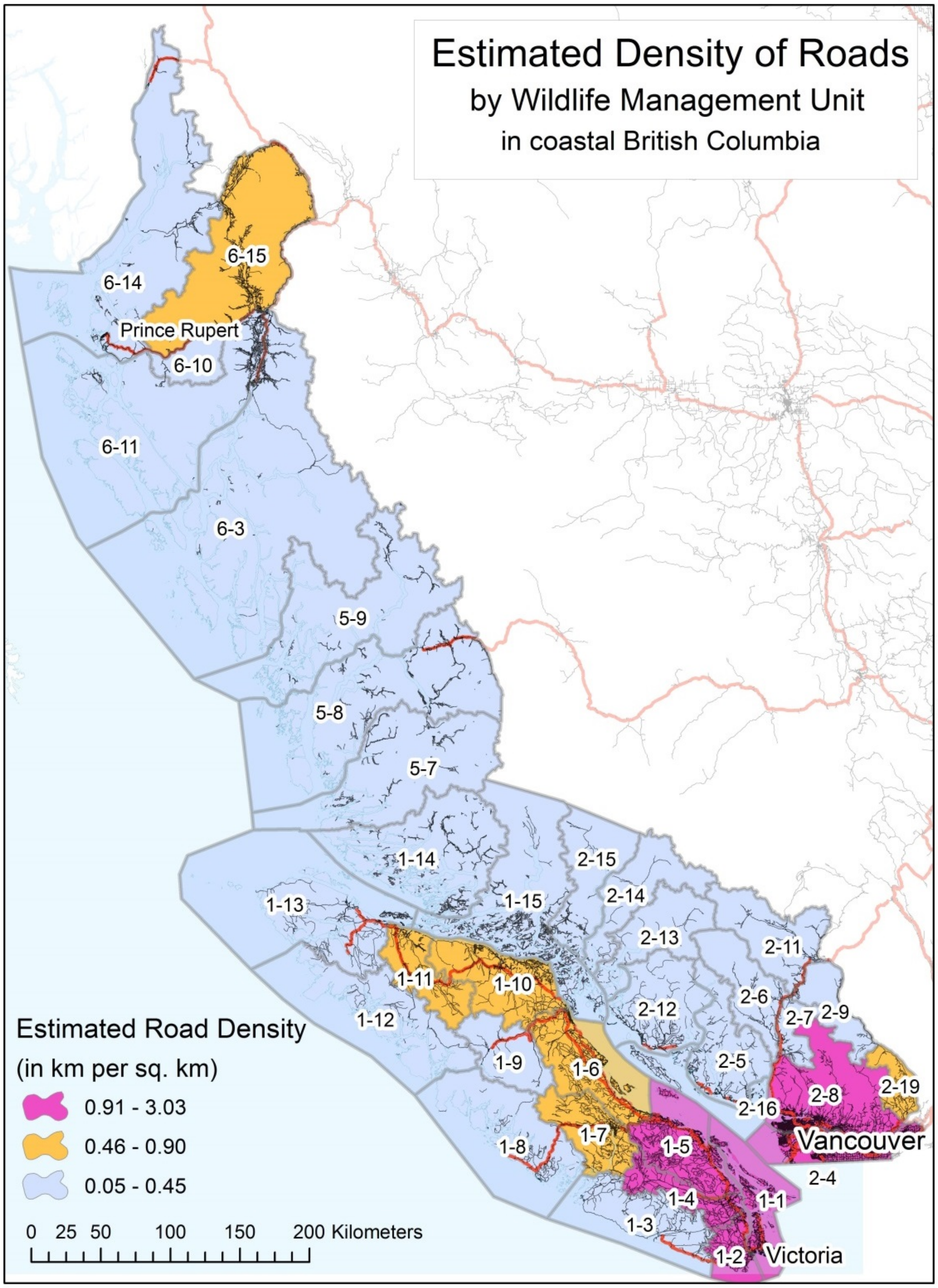

Figure 14. Map depicting road densities estimated by Wildlife Management Unit (identified on map with Region preceding the hyphen) within the apparent range of the Alexander Archipelago wolf in coastal British Columbia (Albert 2015). Estimated road densities greater than 0.90 km/km^2 are considered to be problematic for wolves due to high rates of wolf harvest by humans (Person and Russell 2008, p. 1548; see *Wolf harvest*).

Effects of road development on wolves.—Roads alone are not considered to be problematic for wolves, but instead it is the human access afforded by them that can affect individuals and populations of wolves; we examine road access as an indirect impact to wolves in a separate subsection (see *Wolf harvest*). Here, we consider two potential direct impacts to Alexander Archipelago wolves from road development: alteration of habitat and disturbance at den sites.

The Alexander Archipelago wolf is a habitat generalist, but prefers habitats below 400 m in elevation where most roads are located (e.g., Figure 10 in Person 2001). Gray wolves commonly use roads with little vehicular traffic as travel corridors, especially in winter, and occasionally as territory boundaries (Fritts et al. 2003, p. 301); we postulate that coastal wolves use roads similarly. Based on radio-collared wolves in GMU 2, Person (2001, p. 64) found that wolf packs typically were located nearer to roads than random locations regardless of season, but when the analysis was limited to wolf locations below 100 m in elevation, some packs avoided roads, although others selected for habitats near roads. These inconsistent findings suggest that roads probably are not a strong selection factor for Alexander Archipelago wolves. Furthermore, roads are located in a small percentage of the range of the coastal wolf; we estimate that <4% of the landscape is roaded (total roads=78,587 km), assuming that roads average 100 m in width. Thus, based on our review, we hypothesize that loss or alteration of habitat due to road development probably has little effect on Alexander Archipelago wolves.

Similarly, Person and Russell (2009, pp. 217–219) found that most used den sites of Alexander Archipelago wolves in GMU 2 were located farther from logged stands and roads than unused locations, but that other landscape features such as slope, elevation, and proximity to freshwater had a stronger influence on den site selection. This result was consistent at a fine spatial scale (100 m surrounding active dens) and at a broad spatial scale (1,000 m buffer). The authors reported that wolves denned within landscapes heavily modified by human development, but believed that they did so out of necessity given reduced availability of alternatives (p. 222). Nonetheless, the authors did not report any demographic consequences such as lower reproductive success or pup survival associated with denning closer to roads or logged stands. Therefore, while Alexander Archipelago wolves may prefer to den in unlogged, roadless areas, they will use dens sites in areas near roads and logged stands and, based on the available information, do not experience negative demographic impacts from doing so. More importantly, other natural landscape features appear to be better predictors of den site use of Alexander Archipelago wolves.

In summary, road development has altered a small proportion (<4%) of habitat within the range of the Alexander Archipelago wolf and may disturb wolves using den sites nearby, but we found little information indicating a negative and consistent demographic response of wolves to roads (although see *Wolf harvest*). Although the majority (86%) of roads are located in coastal British Columbia (67,612 km) where populations of Alexander Archipelago wolf are thought to be stable or slightly increasing, comparatively high road densities in GMU 2, especially in some WAAs, may be contributing to the observed apparent decline of that population (see *Abundance and trend*). Thus, based on our review, we believe that road development is not impacting Alexander Archipelago wolves directly at the population level; although some individual wolves may use den sites in undesired habitat owing to close proximity to roads, we found no evidence that breeding effort or reproductive output is being affected.

5.3.4. Wolf harvest

Of the stressors that affect Alexander Archipelago wolves, wolf harvest by humans is the only one that directly results in mortality. Generally, wolf populations in Alaska are managed for long-term sustainable use (ADFG 2015a, p. 6) while providing opportunities for hunting and trapping to the public (Alaska Board of Game 2011, p. 2). In British Columbia, objectives of wolf management are more varied, but include opportunities for cultural, economic, and recreational use (BCMO 2014, p. v).

Mortality of wolves due to human harvest may be compensated for via increases in survival, reproduction, or immigration (i.e., compensatory mortality) or harvest mortality may be additive, causing overall survival rates and population growth to decline. Most studies demonstrate that high rates of reproduction and immigration can compensate for human-caused mortality rates of 17–48% (±8%; Fuller et al. 2003, pp. 184–185; Adams et al. 2008 [29%], p. 22; Creel and Rotella 2010 [22%], p. 5; Sparkman et al. 2011 [28%], p. 5; Gude et al. 2012 [25%], pp. 113–116). However, results of other studies suggest that harvest of wolves by humans are at least partially additive (Murray et al. 2010, pp. 2519–2520), and therefore, sustainable mortality rates may be lower than expected (~22–25%; Creel and Rotella 2010, p. 5). Sustainable rates of human-caused mortality within a wolf population vary considerably based on population characteristics such as age and sex structure, but typically depend on productivity and immigration (Fuller et al. 2003, p. 185). In this regard, each population (or group of populations) is different and a universal human-caused mortality rate does not exist.

For Alexander Archipelago wolves in GMU 2, Person and Russell (2008, p. 1547) reported that total annual mortality >38% was unsustainable and that natural mortality averaged 0.04 (SE=0.05) annually (p. 1545). We did not find any other data on sustainable harvest rates specific to the Alexander Archipelago wolf. Because the biological circumstances of each wolf population are different, we assessed wolf harvest statistics of individual populations relative to 20% and 30% of the estimated population size in a given year. These thresholds were chosen based on findings presented by Person and Russell (2008, pp. 1545–1547), harvest guidelines applied in GMU 2 between 1997 and present (described in more detail below), and on relevant literature for gray wolf (e.g., Fuller et al. 2003, p. 182; Adams et al. 2008, pp. 19, 22). We note here that they are presented as guidelines only to aid in interpreting the wolf harvest data; we do not know what constitutes sustainable harvest levels for most populations of Alexander Archipelago wolf because populations are not monitored regularly.

Management authorities, regulations, and guidelines.—In southeastern Alaska, wolf harvest regulations are set by the Alaska Board of Game for all resident and non-resident hunters and trappers, and by the Federal Subsistence Board for federally-qualified subsistence users on Federal lands. ADFG implements regulations set by the Alaska Board of Game, whereas the USFS implements regulations set by the Federal Subsistence Board. These two management agencies work collaboratively to manage wolf populations and harvest, with public input from the Southeast Alaska Regional Advisory Council.

Although hunting and trapping regulations vary across GMUs (Table 19), generally the hunting season opens on August 1 followed by the trapping season on November 1, and both seasons conclude in late April or May. The bag limit for hunters under both State and Federal regulations is five wolves annually; no bag limit is set for trappers under either set of regulations. In addition, bag limits for hunters and trappers are tallied separately; for example, if a hunter reaches their bag limit of five wolves, the individual can still trap wolves with no bag limit. Across GMUs, all harvested wolves must be sealed, which involves the placement of an official marker or locking tag on the skull or hide by an authorized representative of ADFG.

In GMU 2, wolf harvest is managed differently compared to all other GMUs. First, season length, timing, and sealing requirements are more restrictive (Table 19). Second, since 1997, ADFG and USFS annually determine a combined maximum number of wolves that can be hunted or trapped under either set of regulations (i.e., State and Federal regulations); if the annual harvest guideline is exceeded, both agencies issue an emergency order closing the hunting and trapping seasons. Starting in 1997, the allowable wolf harvest in GMU 2 was set as 25% (~90 wolves) of the fall population estimate. In 2000, the harvest guideline level was increased to 30% of the fall estimate, although the total number of wolves that could be harvested remained as 90 wolves; in 2011, this number decreased to 60 wolves because of suspected declines in wolf abundance. Based on the fall 2013 wolf population estimate (221 wolves; Table 3), wolf harvest for 2014 season was capped at 25 wolves. Since 1997, emergency closures were issued on three occasions (1999 with 96 wolves harvested, 2013 with 57 wolves harvested, and 2014 with 29 wolves harvested). In January 2015, the Board of Game lowered the harvest guideline level to 20% of the fall population estimate because of concerns about the status of wolves in GMU 2, which equates roughly to 18 wolves based on the 2014 population estimate of 89 wolves (Table 3). However, owing to concerns about the GMU 2 wolf population, ADFG and USFS set the harvest cap for the 2015 season at 9 wolves, or 10% of the 2014 population estimate; to date (November 18, 2015), one wolf has been sealed in the 2015 season. It is important to note that ADFG typically does not estimate wolf populations in GMU 2 (or any other GMU) on an annual basis; therefore, annual wolf harvest caps are determined using the most recent estimate (see Table 3) and knowledge of the local biologists, hunters, and trappers, while adhering to the guidelines established by the Board of Game.

In coastal British Columbia, the provincial-based Ministry of Forests, Lands and Natural Resource Operations manages wolf harvest according to an established management plan (BCMO 2014). Wolves can be hunted by residents and non-residents (designated as “big game”) and trapped (designated as “furbearers”). In Regions 1 and 2 (Figure 4), the hunting season extends from September 10 to June 15 and reporting is required within 30 days of the kill; in Regions 5 and 6, the hunting season is from August 1 through June 15 and reporting is not required. The hunting bag limit is three wolves annually. Under trapping regulations, wolves are considered a Class III species, meaning that they generally are not vulnerable to over-trapping and trappers are encouraged to trap these species. Most trapping seasons open in October and close in March; reporting is required in Regions 1 and 2 only within 15 days of the end of the trapping season. Similar to southeastern Alaska, regulations set no limit on the number of wolves that can be trapped in a season.

Table 19. Current hunting and trapping regulations for wolves implemented by the State of Alaska and U.S. Forest Service (with authority from the Federal Subsistence Board). For both management agencies, hunters can harvest a maximum of five wolves (i.e., bag limit) and trappers can harvest unlimited number of wolves (i.e., no bag limit) unless a harvest cap is instituted (e.g., GMU 2; see text). Bag limits for hunters and trappers are tallied separately, if applicable.

GMU	State regulations			Federal regulations		
	Hunting season	Trapping season	Sealing period	Hunting season	Trapping season	Sealing period[1]
1A	August 1–May 31	November 1 – April 30	30 days	August 1–April 30	November 10 – April 30	30 days
1B (south of Bradfield Canal)	August 1–May 31	November 1 – April 30	30 days	August 1–April 30	November 10 – April 30	30 days
1B (remainder)	August 1–April 30	November 1 – April 30	30 days	August 1–April 30	November 10 – April 30	30 days
1C	August 1–April 30	November 1 – April 30	30 days	August 1–April 30	November 10 – April 30	30 days
1D	August 1–April 30	November 1 – April 30	30 days	August 1–April 30	November 10 – April 30	30 days
2	December 1–March 31	December 1–March 31	14 days	September 1–March 31	November 15 – March 31	14 days
3	August 1–May 31	November 1 – April 30	30 days	August 1–April 30	November 10 – April 30	30 days
4[2]	August 1–April 30	Not specified	30 days	August 1–April 30	November 10 – April 30	30 days
5	August 1–April 30	November 1 – April 30	30 days	August 1–April 30	November 10 – April 30	30 days

[1] Only if harvested wolves are transported out of Alaska.

[2] Although wolves do not occupy GMU 4, hunting and trapping regulations allow for harvest.

Hunter and trapper access.—Harvest rates of Alexander Archipelago wolves are associated with access (e.g., Person and Logan 2012, entire). In southeastern Alaska, successful hunters and trappers report type of transportation used when sealing a harvested wolf. Across all GMUs between 1997 and 2014, successful hunters and trappers reported using four types of transportation to harvest wolves: boats (58% of the trips), motorized vehicles including snow machines, all-terrain vehicles, and highway vehicles (28% of the trips), airplane (8% of the trips), and non-motorized transportation (e.g., walking skiing, sledding; 5% of the trips; ADFG 2012; ADFG 2015e). However, transportation preferences varied considerably across GMUs (Figure 15). Generally, hunters and trappers in GMUs with extensive road systems (e.g., GMUs 1D and 2) used motorized vehicles more regularly than hunters and trappers in largely unroaded GMUs (e.g., GMUs 1A and 1B; Figure 16). We did not find similar data for coastal British Columbia.

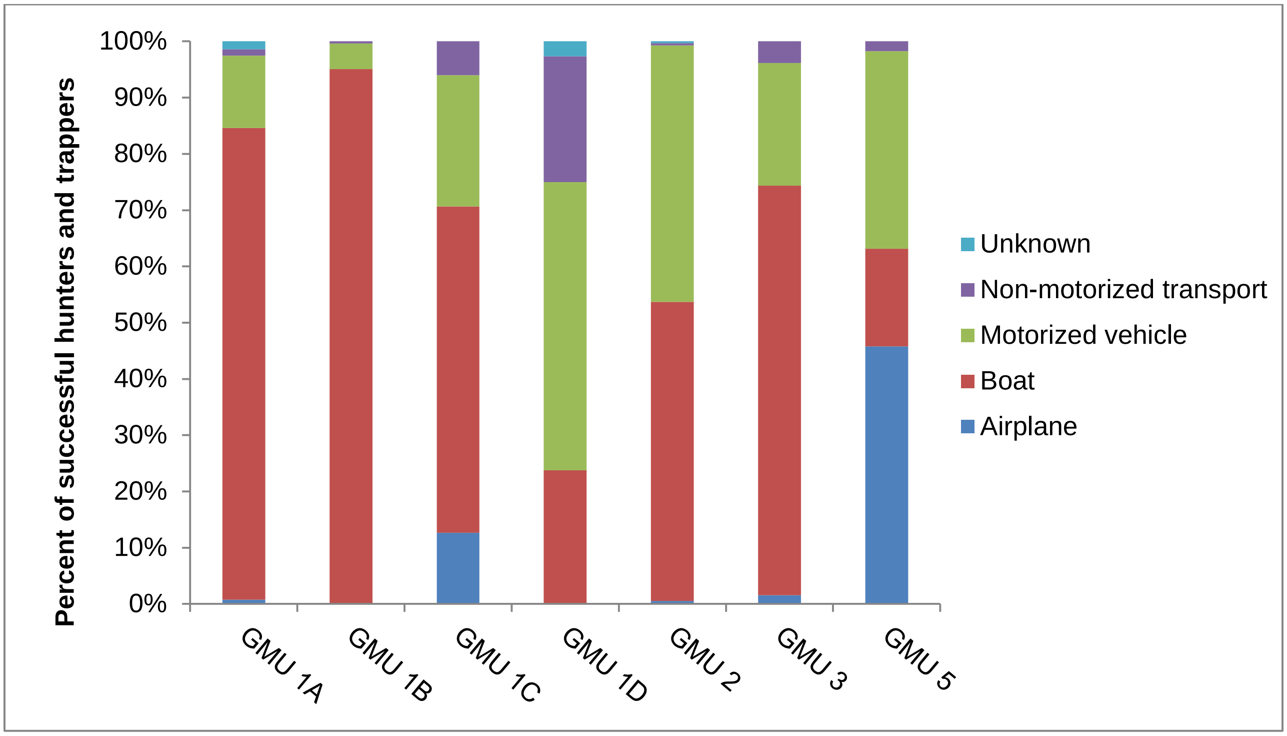

Figure 15. Mean percent of successful hunters and trappers of Alexander Archipelago wolves by type of transportation and Game Management Unit (GMU), southeastern Alaska, 1997–2014 (ADFG 2012; ADFG 2015e).

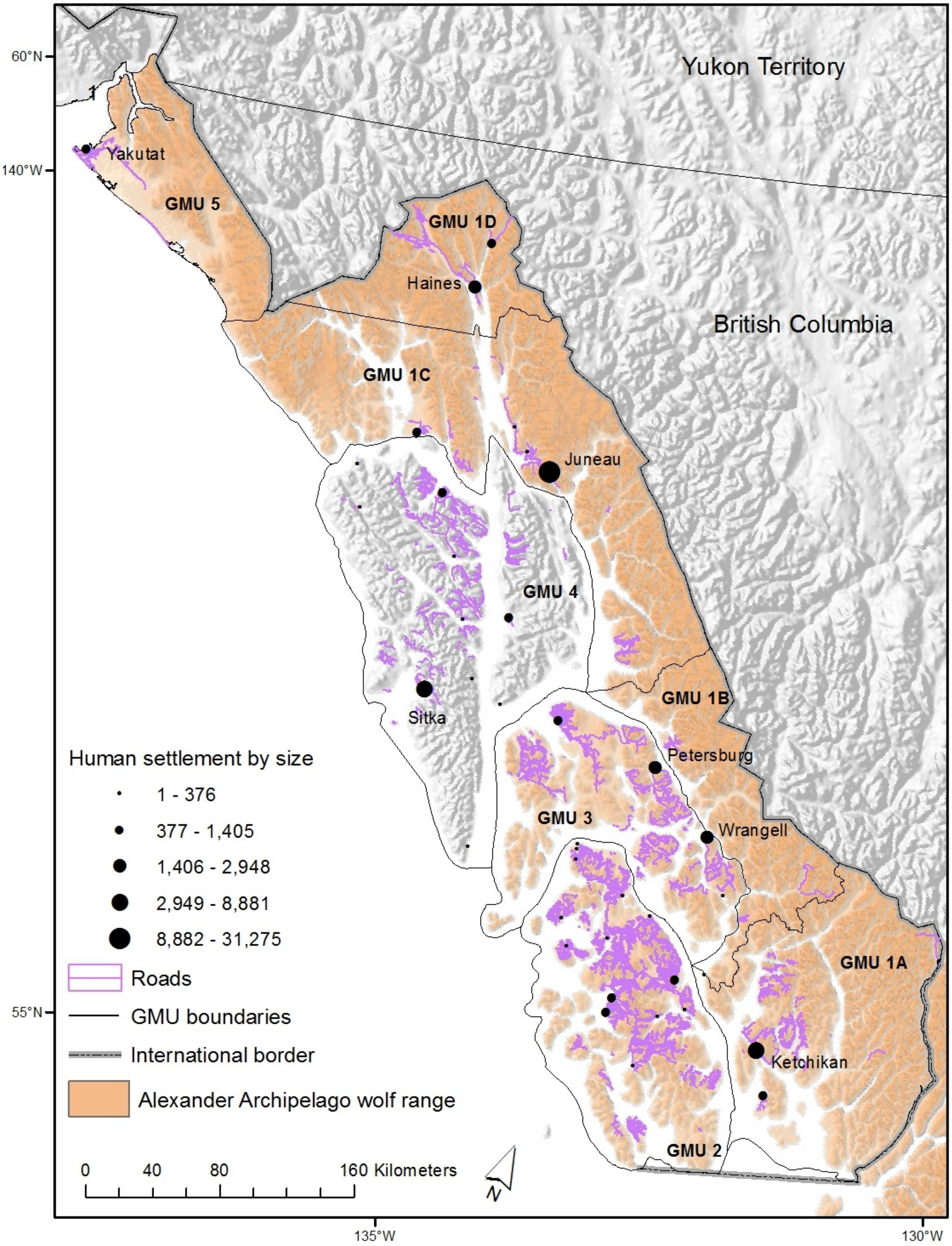

Figure 16. Map of human settlements by population size and roads to demonstrate variation in access (e.g., road, boat) for hunters and trappers within the range of the Alexander Archipelago wolf, southeastern Alaska.

We examined road and boat access in more detail because they served as the majority (86%) of transportation types used by hunters and trappers to harvest Alexander Archipelago wolves in southeastern Alaska where the only available data were collected. We did not find comparable data describing transportation access for coastal British Columbia, but we assume that most hunters and trappers use motorized vehicles given the extensive road system and perhaps boats in the more isolated areas of northern coastal British Columbia.

Road access.—Generally, most studies of gray wolves found that populations do not survive when road densities exceed 1.00 km/km^2 (e.g., Fuller et al. 2003, p. 181 and references therein), although densities of about 0.60 km/km^2 have been recommended as a threshold for wolf persistence (Thiel 1985, p. 405). In some cases, these studies were conducted in areas where legal hunting was not permitted because wolves were protected under the Act, but nonetheless, the thresholds are informative and applicable in areas where wolves are not protected. In recent years as attitudes toward wolves have improved, gray wolves are occupying areas successfully where road and human densities were thought previously to be too high (e.g., Merrill 2000, pp. 312–313, reported wolves breeding in an area where road density was >1.40 km/km^2). On POW, Person and Russell (2008, p. 1548) reported that road density was an important predictor of Alexander Archipelago wolf harvest rates, but when density exceeded 0.90 km/km^2, the relationship deteriorated, suggesting a threshold beyond which further increases in road density had little detectable effect on wolf harvest rates.

We reviewed mean road densities in relation to the threshold of 0.90 km/km^2 in southeastern Alaska and coastal British Columbia (Tables 17 and 18; see *Road development*). In summary, road density is lowest on the mainland of southeastern Alaska (GMUs 1 and 5A; mean=0.08 km/km^2), followed by northern coastal British Columbia (Regions 5 and 6; mean=0.23 km/km^2) and GMU 3 (mean=0.26 km/km^2) where only 0–4% of the WAAs and WMUs exceed the road density threshold. In southern coastal British Columbia (Regions 1 and 2), mean road density is 0.54 km/km^2, largely due to urban areas of Victoria and Vancouver (Figure 14), and 21% of the WMUs have densities greater than 0.90 km/km^2. GMU 2 had the highest road density (mean=0.62 km/km^2) and percentage of WMUs over the threshold (42%). Therefore road access for hunters and trappers is greatest in rural GMU 2, followed by the highly urban regions in southern coastal British Columbia; elsewhere in the range of the Alexander Archipelago wolf, road access is limited at the scale of our analysis (Table 20, Figure 16).

Boat access.—Within the range of the Alexander Archipelago wolf, harvest rates of wolves decrease with increasing ocean distance from towns and villages (Person and Russell 2008, p. 1546), although the relationship can be complicated because hunters and trappers likely choose the most efficient means of transportation to access a particular area. In some cases, a favored area for hunting or trapping could be accessible by road, boat, or more than one type of transportation (e.g., boat and all-terrain vehicle).

We examined boat access by calculating the ratio of shoreline to land area as a proxy of boat access for hunters and trappers of Alexander Archipelago wolves. Although Region 1 in coastal British Columbia has the greatest amount of shoreline, GMU 2 has the highest ratio of shoreline to land area, followed by GMU 3 (Table 20). The remaining areas within the range of the Alexander Archipelago wolf have comparable ratios of shoreline to land area ranging between

0.07 and 0.21, suggesting lower overall boat access to these areas compared to Region 1 and GMUs 2 and 3.

Table 20. Description of road and boat access for hunters and trappers by Game Management Unit (GMU) in southeastern Alaska and Region in coastal British Columbia within the range of the Alexander Archipelago wolf. We summarized road access using mean road density (km/km^2) and percent of Wildlife Analysis Areas (WAAs, southeastern Alaska) and Wildlife Management Units (WMUs, coastal British Columbia) and boat access using total shoreline (km) and the ratio of shoreline to land area, which is presented in Appendix I.

	GMU or Region	Mean road density (km/km^2)	Percent of WAAs or WMUs with road density >0.90 km/km^2	Total shoreline (km)	Ratio of shoreline to land area
Southeastern Alaska	1	0.08	1%	8,669	0.18
	2	0.62	42%	7,644	0.81
	3	0.26	4%	4,880	0.62
	5A	0.04	0%	932	0.12
Coastal British Columbia	1	0.53	27%	10,786	0.23
	2	0.55	15%	2,332	0.07
	5	0.13	0%	4,802	0.21
	6	0.30	0%	8,444	0.19

In considering road and boat access collectively, GMU 2 provides the greatest access for hunters and trappers within the range of the Alexander Archipelago wolf (Table 20, Figure 16). In GMU 3, boat access is somewhat high, but road access is comparatively low and, based on the low percent of WAAs exceeding the road density threshold, also is concentrated. These results are reflective of the transportation used to harvest wolves; in GMU 2, both roads and boats are commonly used, while in GMU 3, which favors boat access, boats are the preferred transportation type (Figure 15). Although road access is high in Regions 1 and 2 of coastal British Columbia, these roads primarily lie within the urban areas of Victoria and Vancouver; further, boat access is much lower than in GMUs 2 and 3.

Reported wolf harvest

Southeastern Alaska.—In southeastern Alaska, reported annual wolf harvest ranged from 103 to 224 wolves (mean=167) between 1997 and 2014 (data summarized from ADFG 2012 and ADFG 2015e). Harvest varied substantially across years and GMUs with no consistent or obvious trends (Figure 17). Following patterns in wolf abundance, annual wolf harvest was lower on the mainland (GMUs 1, including all subunits, and 5A) compared to the islands (GMUs 2 and 3). Across all years, harvest in GMUs 2 and 3 accounted for 63% of all reported harvest in southeastern Alaska.

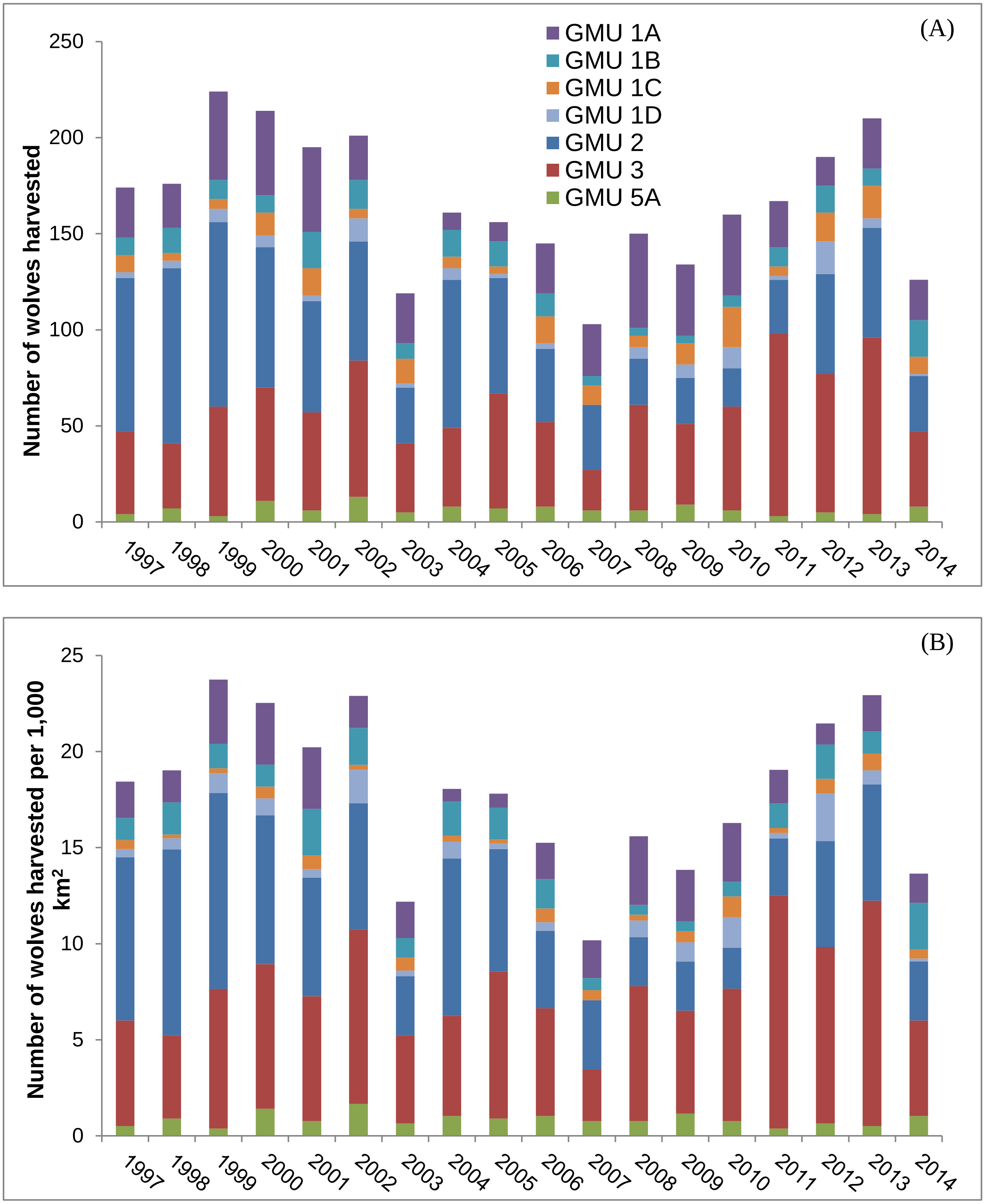

Figure 17. Number of Alexander Archipelago wolves harvested and reported by hunters and trappers (A) by Game Management Unit (GMU) and (B) after accounting for variation in size of GMU (per 1,000 km^2) between 1997 and 2014, southeastern Alaska (ADFG 2012; ADFG 2015e).

Based on the best available population estimates of Alexander Archipelago wolf (Tables 3 and 4), mean reported annual harvest between 1997 and 2014 represented 15–43% of the GMU-specific wolf population (Table 21). In GMUs with more than 50 wolves (i.e., GMUs 1A, 1B, 2, and 3), 15–21% of the population on average was harvested and reported annually. In GMUs with small populations (<50 wolves; i.e., GMUs 1C, 1D, and 5A), harvest rates relative to population size were higher (23–43%), but these values probably are unreliable because ratios of small numbers typically are biased. Thus, combining the wolf populations on the mainland of southeastern Alaska (i.e., GMUs 1 and 5A), mean reported annual harvest of wolves was 19% of the population, ranging between 11 and 27 percent, which is more consistent with the other populations. We emphasize that these values were based on reported harvest only; we address unreported harvest and other sources of human-caused mortality below (see *Unreported harvest*).

Table 21. Mean annual reported harvest of Alexander Archipelago wolves by Game Management Unit (GMU) between 1997 and 2014 relative to estimated population size (ADFG 2012; ADFG 2015e). We combined values across all GMUs on mainland southeastern Alaska (i.e., GMUs 1 and 5A) to avoid biases associated with estimating ratios of small numbers, but present all data here to inform the reader; gray-shaded rows identify reliable estimates of harvest rates relative to estimated wolf population size.

GMU	Population estimate[1]	Number of wolves harvested annually		Percent of population harvested annually	
		Mean	Range	Mean	Range
1A	182	29	9–49	16%	5–27
1B	73	11	4–19	15%	5–26
1C	23	10	4–21	43%	17–91
1D	23	5	0–17	23%	0–74
2	89–356[2]	52	20–96	17%	6–33
3	254	54	21–95	21%	8–37
5A	18	7	3–13	37%	17–72
Mainland (GMUs 1 and 5A combined)	319	62	36–86	19%	11–27

[1] Refer to Tables 3 and 4 for derivation and citations related to population estimates.

[2] Estimated population size varied between 1994 and 2014.

In GMU 2, annual reported wolf harvest is highly variable, but generally has declined (Figure 18), perhaps as a result of lower wolf abundance especially in recent years (see Table 3). In the three years when emergency orders to close the harvest seasons were issued (1999, 2013, and 2014), reported wolf harvest was high (>25% of the population estimated for that year) compared to years when the season remained open for its duration (mean=14%), indicating that the emergency closures were warranted. We emphasize here that we are presenting reported harvest only and unreported harvest can be substantial (38–45% of total harvest in GMU 2; see *Unreported harvest*). When reported and unreported harvests are combined, total harvest likely has been unsustainable in some years.

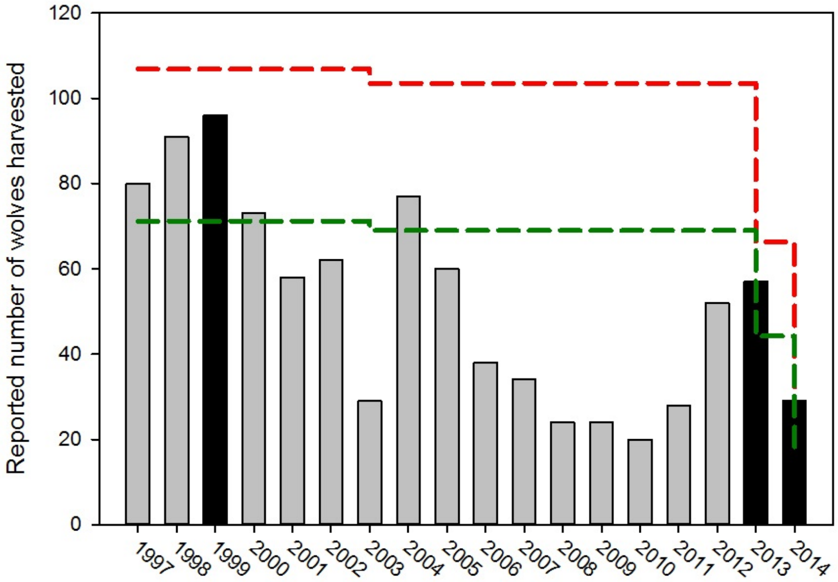

Figure 18. Reported number of wolves harvested by regulatory year in Game Management Unit 2, southeastern Alaska, 1997–2014 (ADFG 2012; ADFG 2015e). The black bars denote the years in which emergency orders were issued to close the hunting and trapping seasons. For context, the green dotted line indicates 20% of the estimated population size in 1994, 2003, 2013, and 2014 (as presented in Table 3) that was current for that year, and the red dotted line denotes 30% of the estimated population size.

Coastal British Columbia.—In the British Columbia portion of Alexander Archipelago wolf range, annual harvest of wolves between 1997 and 2012 from hunting and trapping ranged between 16 and 139 wolves (mean=62) with large variability among Regions (Figure 19; Wolowicz 2015). During this 16-year period, 919 wolves were harvested by hunters while only 76 wolves were trapped. We emphasize that these are minimum values because reporting is not required in all Regions.

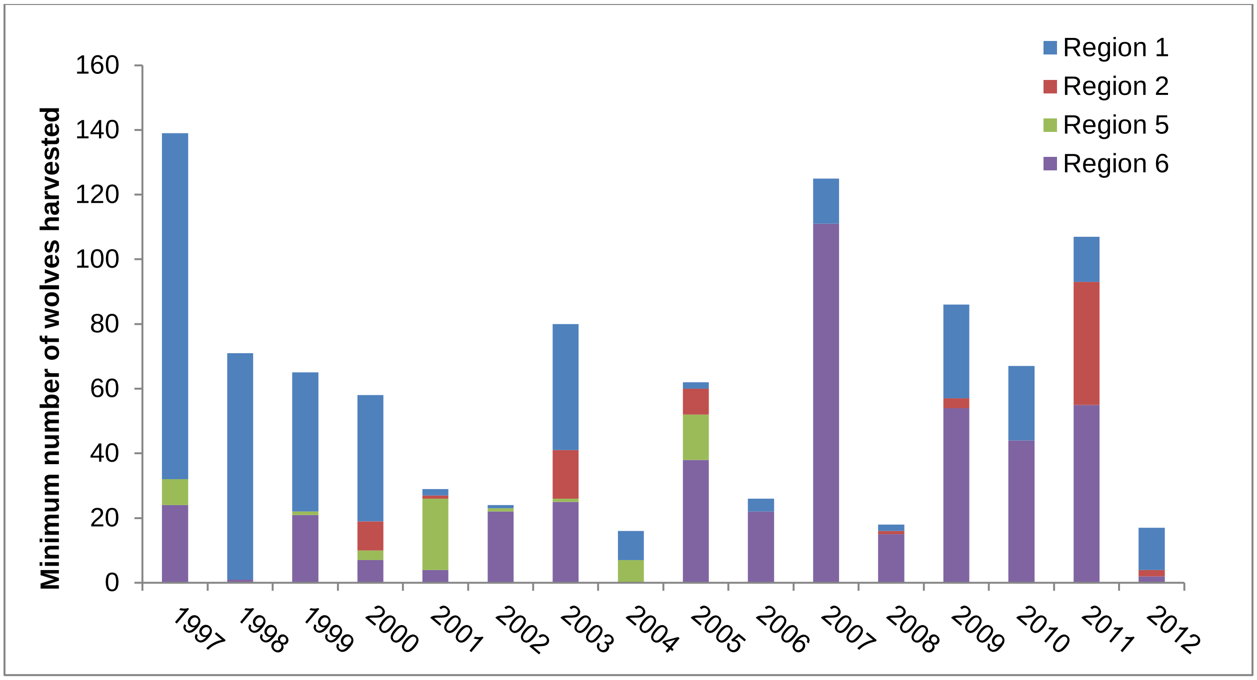

Figure 19. Minimum number of Alexander Archipelago wolves harvested by hunters and trappers by Region between 1997 and 2012, coastal British Columbia (Wolowicz 2015).

In Regions 1 and 2, where a compulsory reporting program is in place, annual wolf harvest was lower in recent years than in the late 1990s. In Regions 5 and 6, we are not able to separate trends in harvest from trends in reporting, although based on the available data, harvest appears to have declined recently in Region 5 (no harvest since 2005), but increased in Region 6, especially in some years (Figure 19). Nonetheless, across all Regions, the mean minimum percent of the population that was harvested annually between 1997 and 2012 was small (2–8%; Table 22), although we recognize that the harvest statistics presented here are incomplete because of the lack of reporting requirements in Regions 5 and 6. Further, we note that the population estimates may be biased high (see *Abundance and trend*) and therefore, the minimum percent of population harvested may be biased low; however, these data represent the only available information, to the best of our knowledge.

Table 22. Region-specific Alexander Archipelago wolf population estimates (specific to coastal area only) and minimum reported wolf harvest between 1997 and 2012 (Wolowicz 2015). These numbers were generated as a guideline and do not reflect absolute values (see *Abundance and trend* for further details).

Region	Population estimate[1]		Mean minimum number of wolves harvested annually between 1997 and 2012 (range)	Mean minimum percent of population harvested annually (range)
	BCMO (2014)[2]	Kuzyk and Hatter (2014)		
1	315	307	26 (1–107)	8% (0–35)
2	114	123	5 (0–38)	4% (0–33)
5	183	200	4 (0–22)	2% (0–10)
6	578	244	28 (0–111)	7% (0–40)

[1] See *Abundance and trend* section for description of methods for generating Region-based population estimates for the coastal areas of British Columbia only based on two sources.

[2] Midpoint of low and high estimates as described in *Abundance and trend* section, following British Columbia Ministry of Forests, Lands and Natural Resource Operations (BCMO) 2014, pp. 6–7.

Unreported harvest (and other human-caused mortality).—In southeastern Alaska and Regions 1 and 2 in coastal British Columbia, hunters and trappers are required to report their wolf harvest, yet not all harvest is reported (e.g., Person and Russell 2008, p. 1545; ADFG 2012, pp. 3, 12, 19, 43). Unreported harvest can result from a hunter or trapper unknowingly harvesting a wolf (e.g., wounded animal that dies and is not recovered, often referred to as wounding loss) or from a hunter or trapper choosing not to report harvest for whatever reason (e.g., killed outside of open season, exceeded bag limit, etc.). If this situation is common, over-harvest of the population can occur, resulting in population decline (e.g., Liberg et al. 2011, p. 1). We cannot distinguish between wolves that were killed and purposefully not reported and those that were killed and unknowingly not reported. Therefore, for the purpose of this assessment, we consider unreported harvest to be a trapping or shooting harvest that was not reported or sealed, but was required to be, regardless of when it occurred (i.e., during open or closed seasons) and includes wounded animals that later died and were not recovered.

Unreported harvest is inherently difficult to document and quantify. Person and Russell (2008, p. 1545) found that between 1993 and 2004, 16 of 34 (47%) radio-collared wolves harvested on POW were not reported. Most of these wolves were shot (13 of 16, 81%), as opposed to trapped, out of season or killed during legal season and not reported (p. 1545). Average annual rates of mortality attributed to legal harvest, unreported harvest, and natural mortality were 0.23 (SE=0.12), 0.19 (SE=0.11), and 0.04 (SE=0.05), respectively (p. 1545), indicating that unreported harvest on POW can be substantial (i.e., 0.45 of total annual harvest).

Between 2012 and 2014, researchers captured and radio-collared 12 wolves on POW (see *Abundance and trend* for more details). Subsequent to collaring, eight of these wolves were harvested; five wolves were harvested and sealed and three were categorized as unreported harvest (ADFG 2015a, p. 4; ADFG 2015c, p. 10; ADFG 2015d, p. 3). Assuming that these three wolves in fact were harvested and not reported and that harvest rates were equal across years, these data suggest that unreported harvest may be slightly lower (3 of 8; 38%) compared to data collected in the 1990s and 2000s by Person and Russell (2008), although the sample size was small (n=8). Nonetheless, total harvest continues to be high and unreported harvest continues to account for a substantial portion of total harvest. We note here that 0.38 is a minimum proportion of unreported harvest to total harvest. Of the remaining four collars on wolves, one collar currently is retrievable (i.e., the signal has not moved recently) but has not been collected yet (ADFG 2015d, pp. 1–2), and therefore, we do not know the circumstances associated with that collar or wolf.

Collectively across these two studies, unreported harvest of radio-collared wolves on POW constitutes an estimated 38–45% of total harvest. We applied these proportions of unreported harvest (0.45 of total harvest between 1997 and 2011, and 0.38 between 2012 and 2014) to reported harvest for a given year to estimate total harvest of wolves in GMU 2 and found that in most years (13 of 18), total harvest exceeded 20% of the population estimated for that year and in some years (9 of 18), total harvest exceeded 30% of the population (Figure 20). In fact, these seemingly high rates (or proportions) of unreported harvest may have contributed to the recent observed decline in the wolf population in GMU 2 (see Table 3; ADFG 2015b, p. 2). We recognize that rates of unreported harvest probably vary across years and therefore assume that the proportions used in our calculations reflect the average within the two time periods.

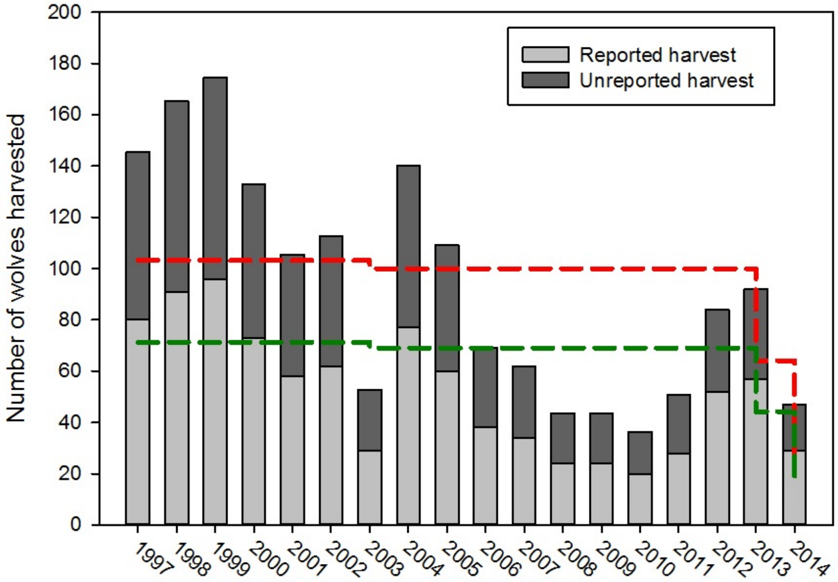

Figure 20. Estimated total number of wolves harvested by regulatory year in Game Management Unit 2, southeastern Alaska, 1997–2014 (ADFG 2012; ADFG 2015e). Unreported harvest was estimated using a rate of 0.45 of total harvest between 1997 and 2011 and a proportion of 0.38 of total harvest between 2012 and 2014 (see text for details and citations). For context, the green dotted line indicates 20% of the estimated population size in 1994, 2003, 2013, and 2014 (as presented in Table 3) that was current for that year, and the red dotted line denotes 30% of the estimated population size.

Outside of GMU 2, we found other reports of documented and suspected unreported harvest of wolves in southeastern Alaska, but not in Regions 1 and 2 in coastal British Columbia. In GMUs 1A and 1C, at least two wolves each have been taken illegally since 1996 (ADFG 2009, pp. 3, 19; ADFG 2012, pp. 3, 19). In GMUs 1B and 3, reported take of wolves is suspected to be below actual take due to poaching (i.e., unreported harvest; ADFG 2012, pp. 12, 43). In GMU 5A, wolves were found dead in snares after the trapping season ended on two occasions (ADFG 2000, p. 41; ADFG 2006, p. 47); in one case, criminal charges were pressed against one of the trappers for failing to salvage a fur animal. Thus, unreported harvest of wolves is occurring throughout most of southeastern Alaska, but outside of GMU 2 we do not know the rate at which it is occurring. We found no information on unreported harvest in Regions 1 and 2 of coastal British Columbia where reporting is required.

We did not apply proportions of unreported harvest estimated for GMU 2 to the other GMUs in southeastern Alaska because GMU 2 has greater boat and road access than all other GMUs (Table 20), which likely results in higher rates of unreported harvest. In addition, although GMU 2 has a small human population size (about 3,400 people), people live in 14 communities that are distributed across GMU 2, facilitating their access to most parts of the unit; in contrast, the human population in the remainder of the GMUs, is more concentrated, leaving most of the units remote and difficult to access (Figure 16; based on 2010 census data, www.census.gov). For these reasons, we believe that GMU 2 is not representative of GMUs 1, 3, and 5, and that applying rates of unreported harvest from GMU 2 to the other GMUs is not appropriate.

In addition to unreported harvest, wolves may be killed accidentally by humans (e.g., vehicle collisions). Since 1996, eight Alexander Archipelago wolves were killed by vehicles in GMU 1A (ADFG 2000, p. 3; ADFG 2006, p. 3; ADFG 2009, p. 5), GMU 1C (ADFG 2006, p. 18), and GMU 5A (ADFG 2000, p. 41). In addition, on rare occasions, wolves can be aggressive, especially if conditioned to human food, and may be pursued by concerned home owners or community members (ADFG 2012, p. 49); we found only one record of two Alexander Archipelago wolves being killed by humans because of increasingly aggressive behavior (Vargas Island, coastal British Columbia; summarized in McNay 2002, p. 5). If a wolf is killed accidentally (or in defense) and is not reported, we consider it to be an unreported, human-caused mortality. It may be difficult to differentiate an unreported harvest event from an unreported, human-caused mortality event, but given the documented high rate of unreported harvest presented by Person and Russell (2008), we believe this distinction is important when assessing the status of the Alexander Archipelago wolf. The intent of these types of events is fundamentally different and should be acknowledged.

Intensive management of black-tailed deer, which includes the culling of wolves with the aim of increasing deer populations and deer harvest by humans, is authorized for GMU 1A (ADFG 2013a) and in GMU 3 (ADFG 2013b). Currently, these programs are inactive, but operational plans exist and could be implemented in the future. If activated, the treatment area in GMU 1A would be restricted to Gravina Island (about 2% of total land in GMU 1A) and all wolves would be eliminated from the treatment area over a 5-year period (ADFG 2013a, p. 6). In GMU 3, the treatment area constitutes 22% of the total land area and is located in the northern portion of the unit including Woewodski, Mitkof, and part of Kupreanof Island (ADFG 2013b, p. 6). Within the GMU 3 treatment area, up to 80% (or ~50 wolves in 5–6 packs) would be removed; duration of the culling effort would be a minimum of five years (ADFG 2013b, pp. 8–9).

Effects of wolf harvest on wolves.—Wolves can compensate for harvest through adjustments in dispersal, reproduction, survival, or a combination of these vital rates, although other factors such as prey availability also may be limiting the population. Evaluating the effect of harvest on wolves requires information about the dynamics of the population, including social structure (e.g., Rutledge et al. 2010, p. 332); for example, if harvest rates are high, wolf density may be lower, resulting in increased prey abundance, which may trigger higher reproduction rates. Therefore, when assessing whether or not rates of harvest, or human-caused mortality, are sustainable, it is useful to understand which factors may be limiting the population and the thresholds at which those limitations apply. In the absence of data on ecological limitations, population trend can be used to evaluate sustainable harvest limits.

The Alexander Archipelago wolf is harvested throughout most of its range (with the principal exception of Glacier Bay National Park), yet we do not understand fully the demographic mechanism by which populations may compensate for harvest. Although individual wolves are affected by harvest, few data exist to assess population- or taxon-level response of Alexander Archipelago wolves to harvest. We found that, in most years, rates of reported harvest relative to estimated population size (mean=≤21% of the population; Tables 21 and 22) were within the sustainable harvest guideline for Alexander Archipelago wolf (~34%; Person and Russell 2008, p. 1547) and for gray wolf populations in continental North America (~20–30%; e.g., Adams et al. 2008 [29%], p. 22; Creel and Rotella 2010 [22%], p. 5; Sparkman et al. 2011 [28%], p. 5; Gude et al. 2012 [25%], pp. 113–116). However, population estimates of Alexander Archipelago wolf are rare temporally and spatially, increasing uncertainty in our estimates of percent of the population harvested. Further, unreported harvest in some areas may be substantial and may be having an undocumented impact on some populations, although outside of GMU 2, we found few data to examine.

In GMU 2, however, wolf harvest likely is contributing to an apparent population-level decline, especially in recent years (Table 3). Although reported annual harvest between 1997 and 2014 constituted <34% of the population (Table 21), when estimated unreported harvest is accounted for, total wolf harvest probably exceeded sustainable limits in most years (Figure 20). We lack a clear understanding of the demographic compensation of the GMU 2 population to wolf harvest, but we assume that the insularity of the population makes it more susceptible to over harvest (see *Connectivity*). Interestingly, however, even though the GMU 2 wolf population recently declined during a period of high total harvest (2013–2014), it appeared to be stable between 1994 and 2003 when reported harvest approached 30% of the population (Table 3, Figure 20). This finding suggests that population estimates are not correct for either period, we are overestimating unreported harvest in the late 1990s, or that the recent population decline is driven by a combination of factors including wolf harvest. We have little ability to predict response of Alexander Archipelago wolves to future harvest, although we can posit that if unreported harvest continues to occur at the documented rates (38–45% of total harvest), we anticipate further population declines of wolves in GMU 2.

To understand the influence of wolf harvest on future population size, we modeled a hypothetical wolf population in GMU 2 under three harvest guidelines (0%, 20%, and 30% harvest caps of the estimated fall population size), assuming a constant rate of unreported harvest (0.42; see Gilbert et al. 2015). Wolf harvest was predicted for each hypothetical pack as a function of road density (road access) and distance via ocean to the nearest human settlement (boat access); see Gilbert et al. 2015 for regression equations. We found that the percent of mean population change for wolves between 2015 and 2045 varied little under the 0%, 20%, and 30% harvest regimes (-7%, -8%, and -8% change in wolf abundance, respectively; Figure 21) because realized harvest rates in the model over the same time period were 3.5%, 10.3%, and 10.5% for the three levels of harvest caps, respectively, even after including unreported harvest. Fall population estimates from the model were high relative to harvest estimates and therefore the model was not informative about the difference in the effects of 20% and 30% harvest caps because these caps were rarely if ever met. However, the model did indicate the following: (1) harvest between 1995 and 2014 affected the GMU 2 wolf population and harvest may explain most of the 2013–2014 observed decline in wolf abundance if unreported harvest was considered; (2) even at low rates of harvest, wolf abundance in GMU 2 probably will decline if other management action(s) are not applied (e.g., reducing access, deer management); and, (3) wolves are projected to decline at about -1% for every 7% increase in wolf harvest even given the current system state.

Modeled harvest rates likely declined quickly after 2014 and were lower than expected until 2045 because harvest rates are determined mostly by road and boat access. In the wolf model, packs in areas with high road densities or easy boat access are quickly reduced or eliminated. These packs are recolonized by dispersers, but they tend to remain at low levels with recurring pack-wide exterminations and therefore contribute little to overall wolf harvest. We did observe a strong influence of wolf harvest levels on deer abundance (-35%, -21%, and -21% change in deer abundance, respectively), especially when no legal wolf harvest occurred (Figure 21), which is expected given the low actual levels of wolf harvest. Our results suggest that wolf harvest did have a strong influence on past changes in wolf abundance based on the population model developed for GMU 2 and that future effects may be similar if these caps are maintained without acknowledging unreported harvest (see Gilbert et al. 2015 for details on this model).

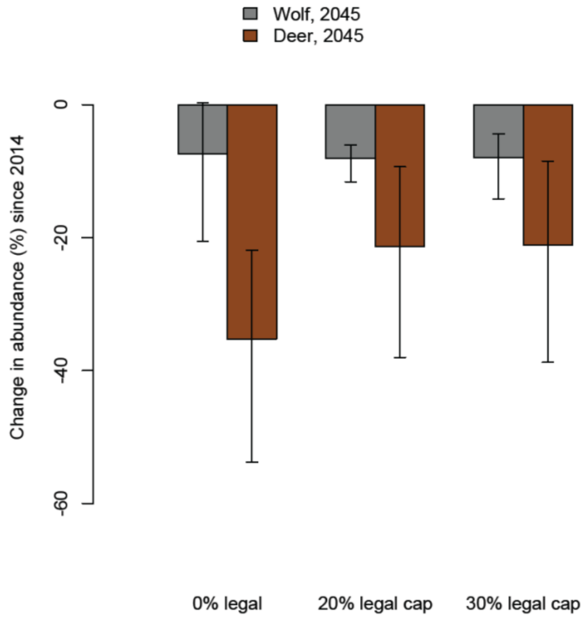

Figure 21. Estimated percent change in mean abundance of wolves and deer under three wolf harvest guidelines (i.e., 0% reported harvest, and 20% and 30% of the estimated fall population size) between 2015 and 2045 in Game Management Unit 2, southeastern Alaska. Error bars denote 95% confidence intervals.

Access and rates of wolf harvest.—Person and Russell (2008, pp. 1546–1548) found that risk of harvest to wolves increased with combinations of road access and open habitats (e.g., muskegs) and decreased with increasing ocean distance from towns and villages. Other habitat features positively influenced harvest risk (e.g., increasing distance from lakes and streams [Person and Russell 2008, p. 1545], and increasing proportion of alpine habitat [Person and Logan 2012, p. 14]), but, high road densities and short ocean distances from human population centers had the greatest effect on wolf harvest rates.

Person and Logan (2012) further evaluated the disproportionately high risk of harvest to Alexander Archipelago wolves in some WAAs compared to others in GMU 2. The authors predicted risk of chronic unsustainable harvest (annual harvest rates ≥3 wolves/300 km^2 for ≥5 years between 1985 and 2009) and pack depletion (annual harvest rates ≥7 wolves/300 km^2 for ≥2 years between 1985 and 2009) and found that over the time period analyzed 19 of 32 (59%) WAAs in GMU 2 were chronically over-harvested and most of those (16 of 32; 50%) also met the criteria for risk of pack depletion (pp. 12–13). Their analysis included only reported wolf harvest, yet data from GMU 2 suggests that unreported harvest can be quite high in some years (see *Unreported harvest*), which would exacerbate the status of wolves in the majority of WAAs on POW with roads being a primary factor facilitating wolf harvest.

Although roads increase risk of harvest to wolves, most wolves in southeastern Alaska, including GMU 2, are harvested by hunters and trappers using boats for transportation (Figure 15; no comparable data for coastal British Columbia). In fact, despite USFS efforts to store some roads on POW since 2009 (see above), emergency closures of the wolf hunting and trapping seasons in GMU 2 were issued in 2013 and 2014 (Figure 18) and the wolf population declined considerably over this period (Table 3), suggesting that their efforts either were not effective or were not substantial enough to reduce wolf harvest. Person and Logan (2012, pp. 22–23, 25) predicted wolf harvest rates under planned road closures by the USFS on POW and found that in most WAAs, closing of those roads only had a modest effect on reducing wolf harvest because planned road closures represent a small percentage of the total road density in those WAAs and therefore access was not be reduced by a meaningful amount. In some WAAs, however, rates of harvest decreased substantially. Therefore, the efficacy of road closures to mitigate for possible over-harvest of wolves in GMU 2 is dependent on the roads selected for closure, timing and duration of the closure, and method by which the road is closed.

We modeled wolf abundance out to 2045 under five possible future road conditions in GMU 2. We considered the following conditions: (1) no change in road densities, (2) planned decommissioning of USFS roads (-2.2% of total road density), (3) mid-level of decommissioning of USFS roads (-27.8% of total road density), (4) maximum level of decommissioning of USFS roads (-38.2% of total road density), and (5) construction of new roads to accommodate the maximum old-growth harvest vegetation condition (estimated using the existing ratio of road density to harvested stands; roughly a 30% increase in current road density). We relied on the POW Access Travel Management Plan (USFS 2009, p. 2) for planned and possible decommissioning rates. Generally, mid-level decommissioning involves closing all roads except those suitable for off-highway and high-clearance vehicles, and maximum-level decommissioning closes all roads except those suitable for passenger vehicles only. We applied these decommissioning rates as well as new road construction uniformly across GMU 2.

We found that abundance of wolves was responsive relative to changes in road densities, especially when compared to response of deer (Figure 22). Under mid-level and maximum road decommissioning conditions, change in percent of mean wolf abundance was positive (4% and 8%, respectively); percent change was greatest for wolves under new road construction (-20%). Deer abundance was less responsive; percent change ranged between -20% and -22% for all of the decommissioning conditions and the no change condition, but was more pronounced under the new construction condition (-30%). These results suggest that future wolf abundance is sensitive to road densities and, in fact, decommissioning of roads at a higher rate than currently planned could result in positive changes in wolf abundance. However, we caution that the predictive relationships used in the wolf population model are simplifications of reality. Importantly, at road densities < 0.90 km/km^2, the regression relationship used in the model (see Gilbert et al. 2015 for details) does not use distance by ocean to community as a predictive variable of harvest, instead including only road density. As a result, continued high harvest via boat in some pack areas with easy shoreline access for hunters may not be fully captured in the model results.

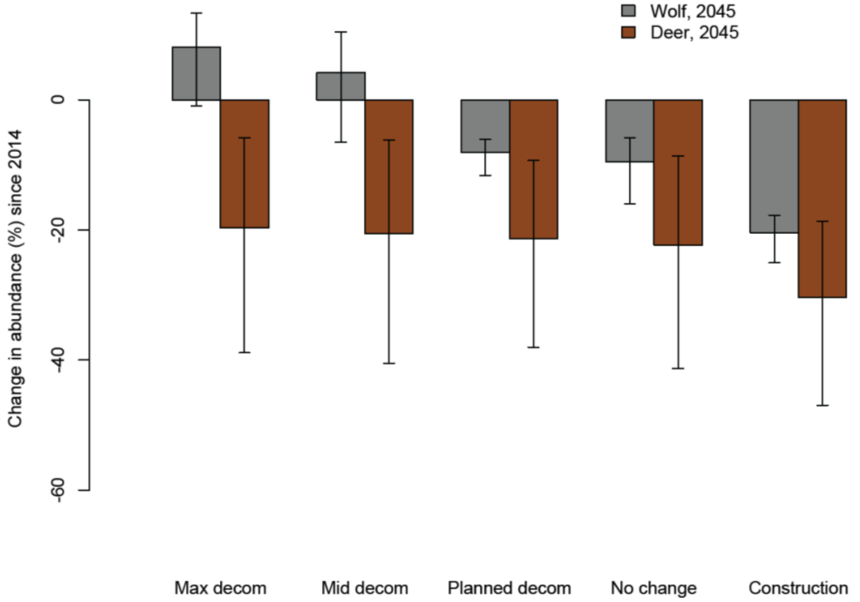

Figure 22. Estimated percent change in mean abundance of wolves and deer under five road conditions between 2015 and 2045 in Game Management Unit 2, southeastern Alaska. Error bars denote 95% confidence intervals.

In our wolf population model, the number of deer harvested annually also is a function of road density because the same roads that provide access to hunters and trappers for wolf harvest also are used by deer hunters. The deer population in GMU 2 is estimated at approximately 60,000 to 75,000 individuals, and hunters legally harvest 2,800 to 3,600 deer annually (Person and Brinkman 2013, p. 155; ADFG 2013c, p. 35), or roughly 4% to 6% of the overall population annually. In addition, ADFG biologists believe that GMU 2 has the highest illegal and unreported harvest rates of deer in southeastern Alaska, largely due to the extensive road system and lack of law enforcement (ADFG 2013c, p. 37). Therefore, using our model, we explored the effect of deer hunting by humans on future wolf abundance. We found that removing legal deer harvest resulted in a 22% change in wolf abundance between 2015 and 2045; under current deer harvest rates, percent change in mean wolf population size was -8% (Figure 23). These estimates do not account for illegal or unreported take of deer.

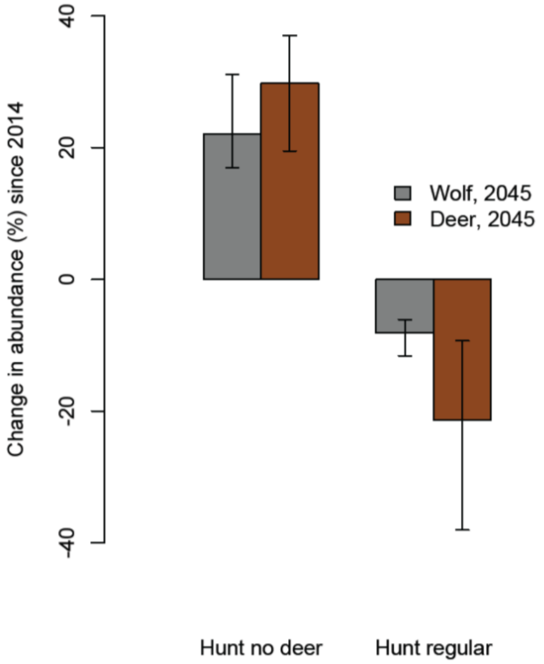

Figure 23. Estimated percent change in mean abundance of wolves and deer when no legal harvest of deer was permitted (Hunt no deer) and when current regulations were allowed (Hunt regular) between 2015 and 2045 in Game Management Unit 2, southeastern Alaska. Error bars denote 95% confidence intervals. Unreported harvest of deer is not included in either condition.

Given the strong influence deer harvest had on future wolf abundance in GMU 2 (Figure 23), we compiled the reported number of deer harvested in 2010 and 2011 (most recent management report available) in other GMUs in southeastern Alaska where reporting is required (data compiled from ADFG 2013c; Figure 24). In mainland GMUs, mean number of deer harvested was low, ranging between 41 (GMU 5A) and 448 (GMU 1C), and was commensurate with low deer densities (ADFG 2013c, pp. 3, 14, 23, 78). In GMU 3, reported deer harvest was 673 and 514, respectively, and has been declining steadily over the years (p. 50). The total number of reported deer harvested in GMU 2 was more than twice that of all other GMUs combined, averaging 3,439 deer over the 2-year period (ADFG 2013c, p. 35). Certainly, deer in GMU 2 are harvested at higher rates than all other GMUs, especially given the comparatively small size of GMU 2 (Figure 24).

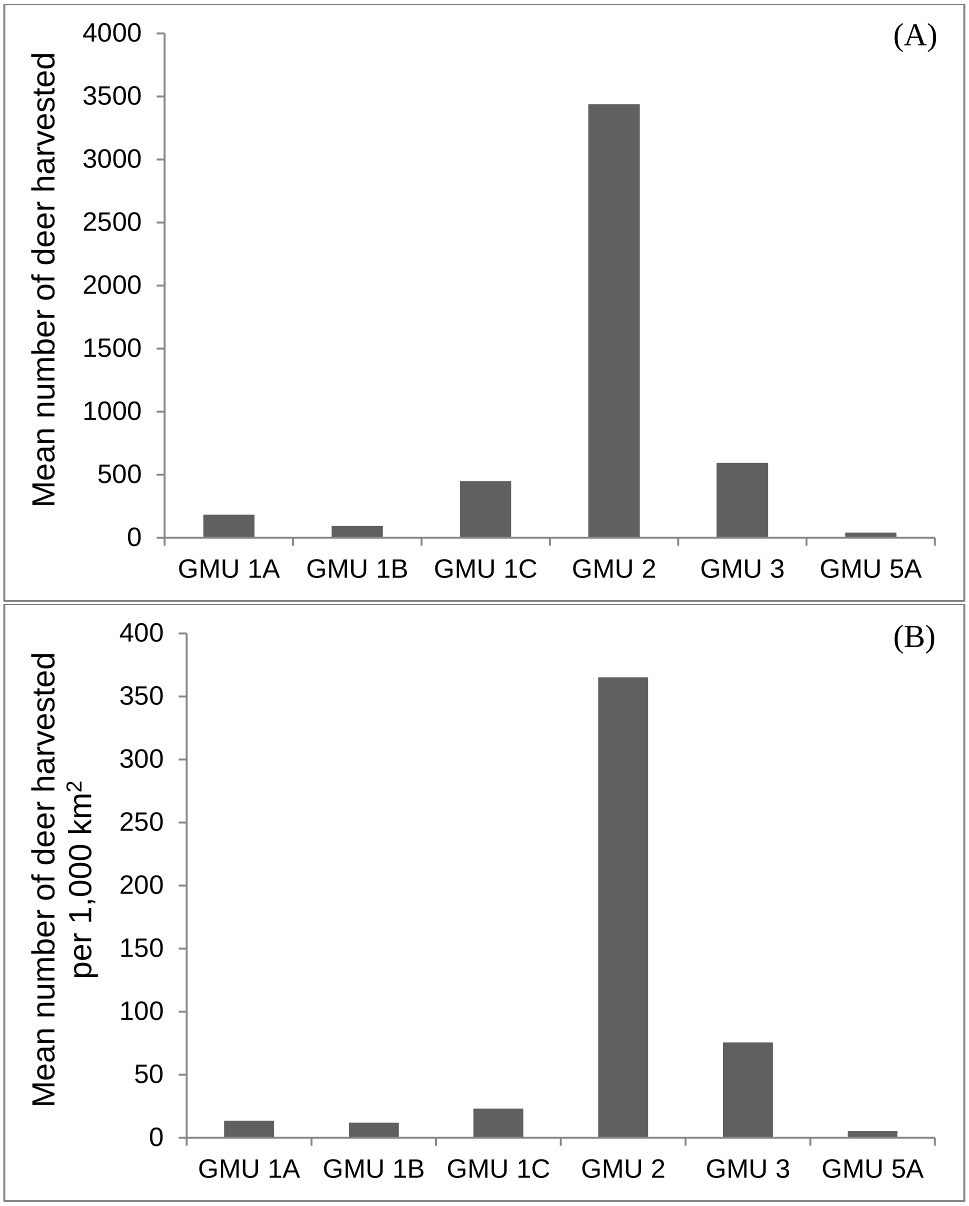

Figure 24. Mean number of reported deer harvested by (A) Game Management Unit (GMU) and (B) after accounting for variation in size of GMU (per 1,000 km^2) within the range of the Alexander Archipelago wolf, southeastern Alaska, 2010 and 2011 (ADFG 2013c). Unit 1B includes both legal and illegal harvest. There is no open season on deer in Unit 1D.

5.3.5. Disease

Several diseases have potential to affect Alexander Archipelago wolf populations. Wolves are susceptible to a number of diseases that can cause mortality in the wild including: rabies, canine distemper, canine parvovirus, blastomycosis, tuberculosis, sarcoptic mange, and dog louse (Brand et al. 1995, pp. 419–422). Many of these diseases impact individual wolves and the social structure of wolves may facilitate rapid spread of some diseases within packs. We are not aware of a disease monitoring program for wolves in southeastern Alaska or coastal British Columbia. Therefore, our assessment of potential disease occurrence and effects on populations of

Alexander Archipelago wolves was based on the prevalence of diseases in domestic dogs and from studies conducted elsewhere on wolves.

Rabies is caused by Rhabdovirid virus and infects all warm-blooded animals. Verified cases of rabies have been documented in wild wolves in other parts of Alaska (e.g., Johnson 1995, pp. 436–437; Weiler et al. 1995, p. 80; Ballard and Krausman 1997, p. 243). Further, rabies has been linked to declines in wolf abundance in Alaska; for example, in northwestern Alaska rabies was a significant factor in the decline of a wolf population and 21% of wolf mortality was attributed to rabies (Ballard et al. 1987, p. 22).

Common vector species of rabies in Alaska are Arctic fox (*Alopex lagopus*), which is not found in southeastern Alaska, and red fox (*Vulpes vulpes*), which is restricted mostly to mainland river valleys. Thus, the most likely vector for rabies in southeastern Alaska is the domestic dog. Prior to 1993, we found no documented cases of rabies in any terrestrial animal from southeastern Alaska (Alaska Division of Public Health 2015). However, since then, three individual bats tested positive for rabies in 1993 (Revillagigedo Island), 2006 (POW), and 2014 (POW), respectively (ADFG 2015f). Likewise, bats are reported to carry rabies in British Columbia (British Columbia Ministry of Environment 2003), but we found no documented cases of rabies in wolves from coastal British Columbia.

Canine distemper is a viral disease usually affecting pups between the age of three and nine weeks of age. Stephenson et al. (1982, pp. 420–421), reported finding wolves from northwestern and interior Alaska seropositive (i.e., positive result in a test of blood serum) for distemper. The low seropositive rate suggests either rare exposure or a high fatality rate. Peterson et al. (1984, p. 31) also reported deaths of two yearling wolves from distemper in 1978 and 1980 on the Kenai Peninsula, Alaska. Although distemper has largely disappeared in domestic dogs as a result of vaccination, rare cases do occur in southeastern Alaska. In 1996, canine distemper was confirmed in a domestic dog treated at the Juneau Veterinary Hospital (New 2015). Canine distemper has been reported in wolves from the Canadian Rockies (Nelson et al. 2012, pp. 71–72) and in Riding Mountain National Park in Manitoba, Canada (Carbyn 1982, p. 108; Stronen et al. 2011, p. 224). We did not find any reported cases of canine distemper from coastal wolves within British Columbia.

Canine parvovirus was discovered during the late 1970s in both domestic dogs and wild wolves. The disease spread rapidly through dog, coyote, and wolf populations in North America, resulting in considerable mortality. Death of captive and free-ranging wolves from parvovirus has been documented (Goyal et al. 1986, p. 1093; Mech and Goyal 1995, p. 567; Johnson et al. 1994, p. 271; Mech et al. 1997, p. 322; Mech et al. 2008, pp. 827–828). For a population of wolves in Minnesota, Mech and Goyal (1995, p. 567) found that the prevalence of the canine parvovirus antibody increased an average of 4% annually during 1979–1993 and reached 87% in 1993; they found a statistically significant inverse relationship between the prevalence of the canine parvovirus antibody and percent change in the wolf population. The authors hypothesized that the wolf population would decline when antibody prevalence consistently exceeds 76%, but below this level mortality from parvovirus was compensatory with natural mortality (pp. 568–569). A 30-year study on the demographic effects of parvovirus on free-ranging wolf populations in Minnesota concluded that compelling circumstantial evidence suggests parvovirus may be a major determinant of the rate of wolf population increase and may restrict further recolonization within Minnesota (Mech et al. 2008, p. 834).

A controlled study of the effect of parvovirus on wolves revealed that 30% of the wolves developed clinical disease symptoms, and 10% would likely have died without supportive care (Brand et al. 1995, p. 421). The crash of the Isle Royale National Park wolf population during 1980–1982 from 50 to 14 individuals and the chronic decline through 1988 may have been related to parvovirus (Peterson et al. 1998, p. 834). This decline coincided with a parvovirus outbreak in domestic dogs in Houghton, Michigan, the main departure point for visitors to the island. Parvovirus presence on the island was confirmed by positive titers in several wolves during the late 1980s (Peterson et al. 1998, pp. 834–835).

Canine parvovirus occurs regularly in domestic dogs throughout Alaska and even with intensive care, high mortality still results. Within southeastern Alaska, canine parvovirus is not common, but some outbreaks have occurred, especially in remote villages that do not have immediate access to veterinarian care (New 2015). Additionally, parvovirus outbreaks in British Columbia have been reported in domestic dogs (Bryan et al. 2011, pp. 14–15). Although documented cases are rare, the transmission of parvovirus from domestic dogs to wild wolves is a conservation concern in southeastern Alaska and coastal British Columbia; vaccination of domestic dogs, control of feral dogs, and surveillance of wolf populations for the presence of positive parvovirus titers are preventive means of control.

Blastomycosis is a fungal disease characterized by granulomatous lesions in various tissues, primarily of the respiratory system in dogs. Blastomycosis has been reported mostly in humans and dogs, but death caused by blastomycosis has been documented in at least one wolf from Minnesota (Thiel et al. 1987, pp. 321–322), and serologic evidence of blastomycosis was found in wolves from Wisconsin (Thiel et al. 1987, p. 322) and Ontario (Krizan 2000, p. 492; Forshner et al. 2004, p. 100). Currently, this disease appears to be limited to the Great Lakes region and the Mississippi River drainage.

Tuberculosis is a disease primarily of cattle and other ungulates, although avian and human forms exist. Sitka black-tailed deer are potential hosts for this disease but we found no evidence of its occurrence in southeastern Alaska or coastal British Columbia. In Canada, the only reports of wolf fatalities related to tuberculosis were from Riding Mountain National Park, Manitoba (e.g, Wobeser 2009, p. 1173).

Sarcoptic mange is caused by the ectoparasitic mite, *Sarcoptes scabei.* In North America, mange is commonly found on red foxes and also occurs on coyotes and wolves throughout their ranges. Wolves with mange usually have severe hair loss, and severe infestations are manifested in crusted lesions and hairless thickened, slate-gray skin over much of the body (Brand et al. 1995, p. 427). Infested animals generally suffer from alopecia, hyperkeratosis, seborrhea, scabs, ulcerations, and lesions (Jimenez et al. 2010a, p. 1120). Severe mange infestations can result in wolf mortality, especially in pups and may play a role in regulating wild canid populations, with the number of cases increasing when wolf populations increase (Todd et al. 1981, p. 727). Sarcoptic mange has been reported in wolves from interior Alaska (Murie 1944, p. 16) and British Columbia (Miller et al. 2003, p. 183).

The dog louse (*Trichodectes canis*) has been reported on free-ranging wolves throughout much of their range in North America (Brand et al. 1995, p. 426; Jimenez et al. 2010b, p. 331). Dog louse can cause skin irritations, matting, and secondary bacterial dermatitis. Although dog louse was documented in domestic dogs, Schwartz et al. (1983, p. 372) was the first to report it in wolves in Alaska (Kenai Peninsula) in 1981. Since then, dog louse has been documented in wolves from the Matanuska-Susitna Valley (Golden et al. 1999, p. 4), interior Alaska (Woldstad 2010, pp. 240–241), and north of the Alaska Range (Gardner et al. 2013, p. 630). Dog louse has also been documented in wolf populations from coastal British Columbia, in areas where wolf densities are high (Hatler et al. 2008, pp. 88–91). We found no evidence that dog louse affects wolves at the population level, but survival of individual pups may be reduced (Brand et al. 1995, p. 426).

Several researchers have hypothesized that coastal wolves in British Columbia may be isolated from pathogens common in other wolf populations by evolving resistance to disease associated with marine food resources (e.g., Darimont et al. 2003, p. 352; Darimont et al. 2008, pp. 9–10; Munoz-Fuentes et al. 2009, pp.1526–1527). The trematode *Neorickettsia helminthoeca*, which is responsible for “salmon poisoning disease” and can be fatal to canids, is thought to concentrate in kidney and muscles of infected fish. Darimont et al. (2003, pp. 350–351) was the first to propose that coastal wolves may have adapted to avoid exposure to *N. helminthoeca* primarily by consuming salmon heads, thereby avoiding infected tissue. Thus, interior-born wolves that disperse to coastal areas may be more vulnerable than coastal wolves to suffer acute symptoms from this disease (Munoz-Fuentes et al. 2009, pp.1526; Stronen et al. 2014, pp. 2, 7). We are not aware of field studies or evidence to test this hypothesis.

The role of disease in limiting wolf populations remains largely unknown. Both canine distemper and canine parvovirus are known or suspected to have affected gray wolves at the population level in other parts of North America (Brand et al. 1995, p. 420 and citations therein). If populations of Alexander Archipelago wolf decline to small numbers or become highly localized, then their vulnerability to disease may increase. Primary defenses against disease include regular vaccination of domestic dogs for rabies, distemper and parvovirus, control of feral dog populations, and preventing the introduction of new diseases. Even though disease is rare in both southeastern Alaska and coastal British Columbia wolf populations, disease outbreaks can have strong influences on the population dynamics of wolves (Bailey et al. 1995, p. 445). Further, Bryan et al. (2011, p. 12) suggested that the potential for introduction of new pathogens and susceptibility of wolf populations to existing pathogens could be influenced by changes in climate and increased economic activities, especially in coastal wolf populations. However, we found few data from which to assess potential impacts of disease to Alexander Archipelago wolves currently or in the future given possible changes in disease dynamics.

5.3.6. Climate-related events

Of the stressors that may be affecting Alexander Archipelago wolves, climate-related events and projected changes in climate is the only one that is primarily stochastic. As discussed above (see *Deer habitat use*), severe winters can strongly affect deer populations, which in turn impacts wolves by reducing available prey. Therefore, it is important to attempt to understand the frequency and influence of severe winters on wolf and deer population dynamics. In this section, we consider possible effects of climate-related events on wolves in southeastern Alaska and coastal British Columbia, primarily by evaluating effects of severe winters on deer populations.

Most studies of deer in southeastern Alaska have investigated their habitat use during the winter when deer populations are thought to be most limited (e.g., Schoen and Kirchhoff 1990; Doerr et al. 2005; Brinkman et al. 2011). These short-term field studies have provided useful information on immediate response of deer to varying degrees of winter severity (summarized in Deer habitat use), but we lack reliable estimates of deer population trends over longer periods of time (>10 years) in southeastern Alaska. ADFG and partners have been conducting deer pellet surveys as an index of deer abundance for decades, but generally these surveys are not done routinely in space or time and, more importantly, recent research has deemed them to be unreliable (Brinkman et al. 2013, p. 444). Without longer term data on trends in deer abundance, our analysis of possible effects of climate-related events on deer populations within the range of the Alexander Archipelago wolf largely was limited to modeling exercises.

A severe winter can affect deer primarily in two ways: (1) by reducing availability of forage (i.e., snow covers browse) and (2) by increasing energy expenditure associated with movement (i.e., deep snow is difficult to move through; Parker et al. 1984, p. 474; Parker et al. 1999, p. 5). Researchers often refer to severe winters and their impacts on deer (e.g., Brinkman et al. 2011, p. 233), but we are not aware of a standard definition of a “severe winter” with regard to the response of deer populations in southeastern Alaska or coastal British Columbia. For example, Farmer et al. (2006, p. 1404) described a moderately severe winter as one when snow accumulation reached 67 cm at sea level by February, with depths exceeding 150 cm at higher elevations, and Doerr et al. (2005, p. 325) considered a winter with deep snow to be one when maximum snow depth was about 85 cm. Person (2001, p. 54) used temperature and precipitation data from local weather stations to estimate that six winters per century may result in general declines in deer numbers in the southern portion of southeastern Alaska.

For the purpose of this assessment, we considered a severe winter to be a combination of >160 cm total winter snowfall (October–March) and >25 cm maximum monthly snow depth in any given month during that winter. In a population model for wolves on POW, Person (2001, p. 54) identified two winters (1969, 1970) as being severe for deer; we used these winters as benchmarks to define a threshold of total winter snowfall (>160 cm). Because deer can be affected by single, extreme snowfall events, we also considered snowfall at a finer temporal scale, i.e., maximum monthly snow depth. Parker et al. (1999, p. 25) found that when maximum snow depths were >30 cm energy costs associated with movement of an average-sized deer (25–30 cm carpus height) increased significantly. During the first winter, fawns weigh ~40% less than adults (Parker et al. 1999, p. 17) and their carpus height is ~10% shorter than that of adults (Parker et al. 1984, p. 481); thus, our criterion of maximum monthly snow depth of 25 cm should account for fawn energy expenditure with locomotion. In addition, most evergreen forbs are covered when snow depths >10 cm (Parker et al. 1999, p. 37) and most larger *Vaccinium* spp. plants are buried when snow >30 cm (White et al. 2009, p. 484).

We modeled the effect of severe winters on deer and wolf abundance in GMU 2 through 2045. Using our definition of severe winter, we determined that two severe winters occurred in GMU 2 over the last 20 years (frequency=0.10) based on data from the Annette Island Weather Station

(National Climatic Data Center, NOAA); this frequency was slightly higher than that estimated by Person (2001, p. 54; frequency=0.06). In our wolf model, we treated winter severity as a binomial variable (i.e., each winter either met the definition of being a severe winter or it did not). We estimated projected frequencies of a severe winter occurring by downscaling regional models of snowpack developed by Littell et al. (2015), following methods outlined in McAfee et al. (2014), to GMU 2 specifically. This approach involved projecting annual precipitation as snow to a 30-year window of 2030–2059 (centered around 2045, total projection length of our population model) using five global climate models, then estimating the percent change in annual snowfall between the future projections (2030–2059) and recent data (1970–1999). The greatest percent change in annual snowfall (-28.6%) among climate model predictions translated to a low frequency of severe winter (0.07) and the lowest percent change in annual snowfall (0%) translated to a high frequency of severe winter (0.10), equivalent to the historical record; the average across all five climate models was a -19.2% change in annual snowfall and served as average frequency of severe winter (0.08; Littell 2015). See Gilbert et al. 2015 for details about the wolf population model.

Results of our wolf population model demonstrated that both deer and wolf abundance were affected by frequency of severe winters, although wolves were affected more than deer (Figure 25). Over the 30-year modeling period, change in abundance of wolves ranged from -6% under low frequency to -13% under high frequency, doubling the effect on wolves with only a small change in the frequency of severe winters (0.07 versus 0.10, respectively). Relative change in abundance was smaller overall for deer, i.e., -20% change under low frequency and -25% under high frequency. This result is consistent with our understanding of the wolf-deer system; as fewer severe winters occur over time, the deer population benefits through increased survival rates, which also benefits wolves through increased deer availability. However, it is likely that the benefits of fewer severe winters would reach capture efficiency limits, as healthy, free-roaming deer are more difficult to catch and kill than under-nourished deer restricted to small patches of snow-free habitat.

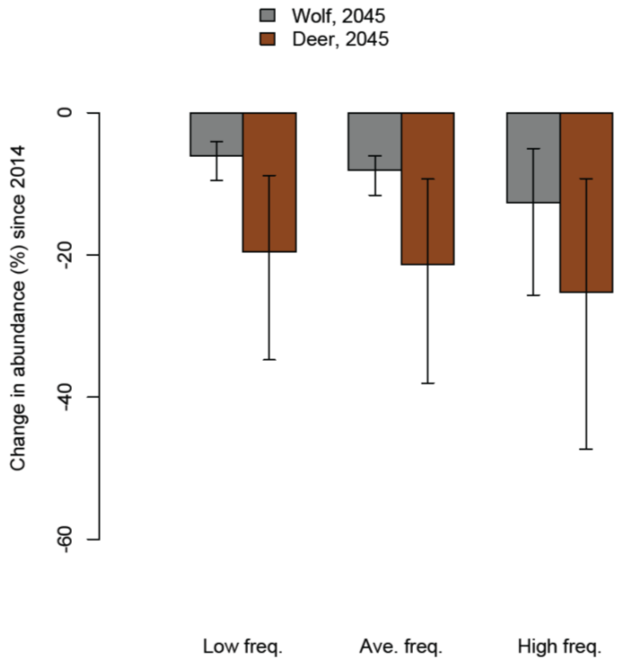

Figure 25. Estimated percent change in mean abundance of wolves and deer under three frequencies of a severe winter occurring (0.07, 0.08, and 0.10, respectively) between 2015 and 2045, as projected by downscaling regional climate models (Littell 2015) to Game Management Unit 2, southeastern Alaska. Error bars denote 95% confidence intervals.

Several researchers recently modeled current and projected climate conditions in southeastern Alaska and coastal British Columbia. Under current climate conditions, snowfall is greatest along the mainland of southeastern Alaska (GMUs 1 and 5A), decreasing as one moves south into northern coastal British Columbia (Regions 5 and 6; Albert and Schoen 2007, p. 15; Shanley et al. 2015, p. 6). In southern coastal British Columbia (Regions 1 and 2), persistent snowfall is rare except at higher elevations. On the islands of southeastern Alaska (GMUs 2 and 3), precipitation as snow is driven largely by local conditions and elevation, but tends to be periodic in frequency (Albert and Schoen 2007, p. 15; Shanley et al. 2015, p. 6).

Based on the average of five global climate models, Shanley et al. (2015, pp. 5–6) projected that precipitation as snow will decrease up to 58% in southeastern Alaska and northern British Columbia over the next 80 years, which should improve winter conditions for deer. McAfee et al. (2014, p. 3944) found that most changes in snow (measured as snow-day fractions, or the percent of precipitation days that receive snow) will occur in February and March, suggesting an earlier onset of spring conditions and longer growing season, which also should benefit deer (although this also could result in a mismatch between peak forage nutrition and birth of fawns as forage plants mature more quickly). Thus, although severe winters affect both deer and wolves at the individual and population levels, future projections indicate that fewer severe winters will occur in southeastern Alaska and coastal British Columbia. These projected future conditions should have an overall positive influence on deer and wolf populations rangewide.

Other climate-related changes that are occurring or are expected to occur within the range of the Alexander Archipelago wolf include reduction or loss of yellow cedar as a result of warmer winters and reduced snow cover (Hennon et al. 2012, p. 156). Decline in yellow cedar occurs at several thousand locations, cumulatively affecting about 2,500 km^2, or <6%, of the forested portions of southeastern Alaska and coastal British Columbia (Hennon et al. 2012, p. 148). Although these changes on the landscape have been observed, we do not know their impact to the Alexander Archipelago wolf. We hypothesize, however, that effects (negative or positive) will be negligible because the wolf is a habitat generalist and an opportunistic predator (see *Resource selection* and *Food habits*). Further, yellow cedar is a minor component of the temperate rainforest, which is dominated by Sitka spruce and western hemlock and neither of these tree species appears to be impacted negatively by reduced snow cover (e.g., Schaberg et al. 2005, p. 2065). In addition, any potential effects on deer as a result of loss of yellow cedar and possible cascading changes in landscape composition are speculative at this point. We are not aware of research that has measured changes in deer abundance with regard to loss of yellow cedar in forests of southeastern Alaska or coastal British Columbia.

We also found evidence that changes in climate are predicted to result in hydrologic changes that may reduce salmon productivity within the range of the Alexander Archipelago wolf (e.g., Edwards *et al.* 2013, p. 43; Shanley and Albert 2014, p. 2). Warmer winter temperatures and extreme flow events are predicted to reduce egg-to-fry survival of salmon, resulting in lower overall productivity. It is unclear whether or not these changes will result in reduced salmon abundance and availability to wolves, but we assume that some wolves will be impacted if that is the case.

5.3.7. Other

In addition to the primary stressors reviewed above, we acknowledge that other factors may influence the status of the Alexander Archipelago wolf at the individual, population, and rangewide levels. We briefly review other possible stressors below.

Endemism.—In the petition to list the Alexander Archipelago wolf, the petitioners raised island endemism as a possible stressor (CBD and Greenpeace 2011, pp. 51, 84–85). An endemic is a distinct, unique organism found within a restricted area or range; a restricted range may be an island, or group of islands, or a restricted region (Dawson et al. 2007, p. 6-1). Although smaller populations are more vulnerable to extinction than larger ones (Lande 1993, p. 921) due to demographic stochasticity, environmental variability and catastrophic events, endemism or “rarity” alone are not stressors. Therefore, we instead considered possible effects associated with small and isolated populations of the Alexander Archipelago wolf.

Based on the best available information, we estimated the current rangewide population to be about 850–2,700 wolves within perhaps 10 or more individual populations, although we have little data to use for defining a population (see *Abundance and trend*). Movement across populations does occur, albeit at low levels (see *Connectivity*), and therefore, none of the populations appear to exist in complete isolation. However, owing to the island geography and steep, rugged terrain within the range of the Alexander Archipelago wolf, some populations are small (less than 150–250 individuals, following Carroll et al. 2014, p. 76) and are more insular (e.g., GMU 2) than others (e.g., coastal British Columbia). These small, partially isolated populations are susceptible to possible negative genetic consequences.

The primary genetic concern of small, isolated wolf populations is inbreeding, which, at extreme levels, can reduce litter size and increase incidence of skeletal effects (e.g., Liberg et al. 2005, p. 17; Raikkonen et al. 2009, p. 1025). We found only one study that examined inbreeding in the Alexander Archipelago wolf. Breed (2007, p. 18) tested for inbreeding using samples from Regions 5 and 6 in northern British Columbia and GMUs 1 and 2 in southern southeastern Alaska and found that inbreeding coefficients were highest for wolves in GMU 1, followed by GMU 2, then by Regions 5 and 6. This finding was unexpected given that GMU 2 is the smaller, more isolated population, yet it indicates that inbreeding probably was not affecting the GMU 2 population at the time of the study despite its comparatively small size and insularity. We found no evidence of historical or recent genetic bottlenecking in the Alexander Archipelago wolf (Weckworth et al. 2005, p. 924; Breed 2007, p. 18), although Weckworth et al. (2011, p. 5) speculated that a severe bottleneck may have taken place long ago (over 100 generations).

Oil development.—Also in the petition, the petitioners listed oil development in coastal British Columbia as a potential stressor to the Alexander Archipelago wolf (CBD and Greenpeace 2011, p. 83). Specifically, the petition identifies a proposed oil pipeline project (i.e., Northern Gateway Project) intended to transport oil from Alberta to the central coast of British Columbia, covering about 1,177 km in length. If the proposed project is approved and implemented, there will be a risk of oil spills on land and on the coast within the range of the Alexander Archipelago wolf. However, we hypothesize that given its dispersal capability and opportunistic food habits, the wolf probably would not be affected negatively by the pipeline project even if an oil spill occurred. Oil development occurs throughout the range of the gray wolf (e.g., Trans Alaska Pipeline System) and is not thought to be impacting wolf populations negatively (e.g., BCMO 2014, p. 11).

Over exploitation of salmon runs.—The petitioners raise concern over the status of salmon runs in coastal British Columbia due to over exploitation and disease transmission from introduced Atlantic salmon (*Salmo salar*; CBD and Greenpeace 2011, pp. 83–84). In coastal British Columbia, only 0–16% of the diet of the Alexander Archipelago wolf is salmon (Darimont et al. 2004, p. 1871; Darimont et al. 2009, p. 130; see *Food habits*). Therefore, we postulate that given the opportunistic food habits of the Alexander Archipelago wolf, reduction or even complete loss of salmon as a food resource may impact individual wolves in some years, but probably would not result in a population level effect. However, loss of salmon in the diet probably would result in a greater dependency on deer and other prey items (see *Food habits*), potentially altering predator-prey dynamics to some extent in affected areas.

Hybridization with domestic dogs.—In the petition, hybridization with domestic dogs was presented as a stressor to the Alexander Archipelago wolf (CBD and Greenpeace 2011, p. 84). Based on microsatellite analyses, Munoz-Fuentes et al. (2010, p. 547) found that at least one hybridization event occurred in the mid-1980s on Vancouver Island where wolves were probably extinct at one point in time, but then recolonized the island from the mainland. Although hybridization has been documented and is more likely to occur when wolf abundance is unusually low, most of the range of the Alexander Archipelago is remote and unpopulated by humans, reducing the risk of interactions between wolves and domestic dogs.

5.3.8. Summary of stressors

In this section, we summarize stressors within the range of the Alexander Archipelago wolf by considering the magnitude (scope, intensity, and immediacy) and level of exposure and possible biological response of wolves to each stressor (Table 23). In our summary, we did not consider interactions or cumulative impacts of stressors, but instead aimed to distill some of the key information presented in this chapter; it is not intended to be a comprehensive list of stressors or their possible impacts to the Alexander Archipelago wolf.

Table 23. Summary of selected stressors within the range of the Alexander Archipelago wolf. Gray-shaded cells include definitions and categories of evaluation factors. We present this table as a summary only and not as a complete list of stressors or their possible impacts; see text for more detailed discussion of each stressor.

Stressor	Possible ecological impacts	Possible biological response	Scope	Intensity	Immediacy	Exposure
Process or event with negative impacts on target taxon	Process or event that occurs as a result of the stressor	Level of response of target taxon due to the stressor	Geographic extent of the stressor	Strength of the stressor relative to the scope	Action time frame of the stressor	Degree of overlap between target taxon and stressor
		Behavioral, Basic need inhibited, Mortality	Localized, Moderate, Widespread	Low, Moderate, High	Historical, Imminent, Future, Ongoing	Low, Moderate, Significant
Timber harvest	Reduced den site availability	Basic need inhibited	Localized	High	Ongoing	Low
	Disturbance at den sites	Behavioral	Localized	Low	Ongoing	Low
	Reduced forage for deer	Basic need inhibited	Widespread	Moderate	Ongoing	Moderate
Road development	Disturbance at den sites	Behavioral	Localized	Low	Ongoing	Low
Wolf harvest	Direct mortality	Mortality	Localized	High	Ongoing	Moderate
	Change in pack dynamics	Behavioral	Widespread	Low	Ongoing	Low
Disease	Increased transmission	Basic need inhibited	Localized	High	Future	Low
Climate-related events	Reduced snowfall, increased rainfall	Behavioral	Widespread	Low	Future	Significant
Endemism (small populations)	Increased inbreeding depression and genetic bottlenecking	Basic need inhibited	Moderate	Low	Future	Low
Oil development	Reduced prey	Basic need inhibited	Localized	Low	Future	Low
Overexploitation of salmon runs	Reduced salmon as prey	Basic need inhibited	Localized	Low	Future	Moderate
Hybridization with dogs	Loss of adaptation and taxon	Behavioral	Localized	Low	Future	Low

5.4. Existing conservation mechanisms

We reviewed relevant existing conservation mechanisms that directly or indirectly benefit, or are intended to benefit, the Alexander Archipelago wolf in southeastern Alaska and coastal British Columbia. We did not evaluate the efficacy of these mechanisms, but instead briefly review their intended purpose and any pertinent limitations to them.

5.4.1. Southeastern Alaska

Tongass Conservation Strategy.—During development of the 1997 Tongass Forest Plan, the Forest Service worked with scientists and managers from a variety of agencies and institutions to design a conservation strategy for old-growth dependent species on the Tongass (hereafter Strategy). Elements of the Strategy were developed after synthesis and consideration of the state of knowledge on many topics, including landscape-scale conservation science, island biogeography, and natural history of several species, including the Alexander Archipelago wolf, and species groups. Primary components of the Strategy include a Tongass-wide network of old-growth habitat reserves linked by connecting corridors of forested habitat, and a series of standards and guidelines that direct management of lands available for timber harvest and other activities outside the reserves. The Strategy developed as part of the 1997 Tongass Forest Plan was retained in the current (2008) Tongass Forest plan with a few modifications. Below, we briefly review key elements of the Strategy with emphasis on provisions directly related to conservation of wolves and their primary prey, deer.

Old-growth reserve network.—The foundation of the Strategy is a series of large, medium and small old-growth reserves, protected from timber harvest and most other human development, and distributed across the Tongass. It is intended to function as a “coarse filter” that provides adequate habitat for most species. Design considerations for reserves specify that they are to be circular rather than linear to maximize interior forest conditions rather than edge conditions, that inclusion of early seral habitats be minimized, and that features such as the largest remaining blocks of productive old-growth, some of the highest-volume remaining forest stands, nesting habitat for Queen Charlotte goshawk (*Accipiter gentilis laingi*) and marbled murrelet (*Brachyramphus marmoratus*), and important deer winter range be included (USFS 2008d, pp. D-6–D-10).

Large reserves are contiguous landscapes of at least 160 km^2, with a minimum of 80 km^2 of productive old-growth forest and 40 km^2 of high volume strata old-growth (defined by canopy coarseness, soils, and aspect). These reserves are spaced no greater than 32 km (20 miles) apart. Many of the 38 large reserves are in non-development LUDs such as Wilderness, Remote Recreation, or Municipal Watershed. Where necessary to achieve spacing requirements, additional large reserves were designated using the Old-growth Habitat LUD. Management prescriptions specify that lands with this designation be managed for maintenance of old-growth forest characteristics.

There are 112 medium reserves of approximately 40 km^2 each with a minimum of 20 km^2 of productive old-growth and 10 km^2 of high volume strata old-growth. They are spaced a maximum of 13 km (8 miles) apart and include a variety of non-development LUDs. All large and small reserves should be linked by corridors of unharvested forest, using existing protected landscape features such as riparian and beach buffers where available.

Small reserves are included in each “value comparison unit” (equivalent to a medium-size watershed or a WAA) that does not already include non-development LUDs over at least 16% of its area. Each of the 237 small reserves should be composed of at least 50% productive old-growth forest.

Matrix management.—Outside of the old-growth reserve network, standards and guidelines apply to the matrix of lands that are open to development on the Tongass. For example, beach and estuary fringe within 305 m (1,000 ft) of saltwater shorelines and riparian habitat along streams are protected as wildlife habitat and movement corridors; these forested buffers benefit deer, especially in winter. Additionally, legacy forest structure standards require retention of residual trees and snags in timber harvest units larger than 0.08 km^2 (20 acres) and in heavily logged watersheds. These standards and guidelines are intended to improve the function of the matrix as a whole and support the reserve network.

For some species, including wolf and deer, the Strategy also includes “fine filter” elements to alleviate species-specific conservation concerns not addressed adequately with the coarse filter reserve network (USFS 2008a, pp. 4-90–4-100). For the Alexander Archipelago wolf, fine filter elements address disturbance at and modification of active wolf dens and elevated mortality of wolves. Buffers of 366 m (1,200 feet) are required around active dens (when known) to reduce risk of abandonment, although if a den is inactive for at least two years, this requirement is relaxed. However, most wolf packs are not monitored, and may switch dens between years, making it probable that many den sites are not identified, monitored for activity, or protected. In areas where wolf mortality concerns have been identified, the Strategy requires development and implementation of a Wolf Habitat Management Program; despite concerns for wolves in GMU 2, no such plan has been developed yet. In addition, when road access and human-caused mortality are thought to be contributing significantly to unsustainable wolf mortality, planning processes associated with road management and wolf harvest regulations are supposed to incorporate measures to reduce risks to wolves in that area. Total road densities of 0.40–0.60 km/km^2 (0.70–1.00 mile/mile^2) are suggested, yet in areas of past timber harvest on POW specifically, road densities exceed this recommended level (Table 17, Figure 13) and recent timber sale decisions will result in even higher road densities in some areas of GMU 2 (e.g., USFS 2013a, pp. 29–30).

The Tongass Forest Plan directs the USFS to provide sufficient deer habitat capability on the Tongass first to maintain sustainable wolf populations, and then to consider meeting estimated human deer harvest demands. Using the deer habitat model (described above in *Timber harvest*), biologists analyze trends in deer habitat capability and other local information to evaluate the extent to which a management area can support wolf populations and human demands for deer. The recommended guideline of deer habitat capability is 7 deer/km^2 (18 deer/mile^2). Many localized areas on POW and elsewhere are currently below this habitat capability guideline due in large part to past timber harvest activities and subsequent forest succession (e.g., USFS 2013b, pp. 3-114–3-115) and implementation of recent timber sale decisions on POW is expected to reduce deer habitat capability further (e.g., USFS 2013a, pp. 26–28). In addition to the deer habitat capability guideline, habitat needs for deer should be considered during project planning and analysis, but no specific standards are identified (USFS 2008a, p. 4-92).

Roadless Rule.—In January 2001, the USFS published a final rule prohibiting road construction and timber harvesting in "inventoried roadless areas" on all National Forest System lands nationwide (hereafter Roadless Rule; 66 Federal Register 3244, January 12, 2001). On the Tongass, the USFS has inventoried 109 roadless areas covering approximately 38,000 km^2 (USFS 2008b, p. 3-444). These roadless areas include approximately 1,200 km^2 of "suitable forest land", which is the land base where timber production is believed to be possible without irreversible resource damage to soils, productivity, or watershed conditions; forests can be adequately restocked; and Tongass management has determined that timber production is an appropriate use (USFS 2008a, p. 7-42; USFS 2008b, p. 3-449). The balance of the inventoried roadless areas (36,800 km^2) are not subject to timber harvest because they are either non-forested, in LUDs that do not allow timber harvest, or on lands too steep, unstable, or otherwise environmentally sensitive to allow logging. Thus, the Roadless Rule effectively protects about 1,200 km^2 from timber harvest.

The Roadless Rule was challenged in U.S. District Court by the State of Alaska in 2001, and in 2003 the USFS and the State reached a settlement. The USFS issued a rule temporarily exempting the Tongass from the 2001 Roadless Rule. This exemption was set aside by the District Court in 2011, which reinstated the prohibitions on roadbuilding and logging. That judgment was overturned by the Ninth Circuit Court of Appeals in 2014, potentially reinstating the exemption, and eliminating the logging and roadbuilding prohibitions. At this time, the Court of Appeals ruling has not been finalized, and additional legal challenges are pending. Therefore, the Tongass is subject to the provisions in the 2001 Roadless Rule (i.e., no logging and roadbuilding is permitted in roadless areas), although the outcome of these legal challenges is uncertain.

5.4.2. Coastal British Columbia

Forest and Range Practices Act.—The Forest and Range Practices Act and its regulations govern the activities of forest and range licensees in British Columbia (see *Timber harvest*). The statutes set the requirements for planning, road building, logging, reforestation, and grazing. The Act does not include provisions specifically for coastal wolves.

Wildlife Act of British Columbia.—The Wildlife Act of British Columbia is the legislative foundation for the interaction of people and wildlife in British Columbia. This Act authorizes the government to declare a species as threatened or endangered. Wildlife is defined as all native and some non-native amphibians, birds, and mammals that live in British Columbia; the gray wolf is included under this Act where it is classified as "big game." It was amended with the Environmental Amendment Act in 2008, authorizing management of alien species and increasing fines for wildlife violations, among other minor changes.

Federal Fisheries Act.—The Federal Fisheries Act provides the regulatory framework for protecting the productivity of commercial, recreational, and Aboriginal fisheries. Fisheries and Oceans Canada, a division of Environment Canada, is the regulatory agency that oversees implementation of this Act. It allows the federal government to manage and reduce threats to the fisheries and the habitat that supports them. Pacific salmon are protected under this Act.

Convention on International Trade in Endangered Species.—The gray wolf is listed as a furbearer and protected under the Convention on International Trade in Endangered Species. Therefore, a permit is required before exporting wolf pelts across international boundaries. For a permit to be issued authorities must determine that such export will not be detrimental to the survival of the species and that specimens to be exported have not been obtained by violation of the laws for their protection.

Regional land use and management plans.—We found over 20 regional- and watershed-based land use and management plans active within the range of the Alexander Archipelago wolf (e.g., Central Coast Land and Resource Management Plan, Vancouver Island Land Use Plan). These land use plans are developed with public and stakeholder input and are considered in decisions pertaining to timber harvest (see *Timber Harvest*). These plans can be found at https://www.for.gov.bc.ca/tasb/slrp/.

CHAPTER 6: CURRENT AND FUTURE STATUS OF THE ALEXANDER ARCHIPELAGO WOLF

The purpose of this document is to synthesize scientific information relevant to assessing the current and future status of the Alexander Archipelago wolf. In this chapter, we summarize information presented in Chapters 2–5 with the goal of projecting future status of the Alexander Archipelago wolf. Owing to the variation in quantity and quality of available data across the range of the wolf, we first assess current status of each population separately and evaluate the potential impact of stressors, individually and cumulatively, to that population by examining its resiliency. We then project future status by considering all populations collectively using the principles of redundancy, resiliency, and representation. For the purpose of this assessment, we define resiliency as the ability of a population or taxon as a whole (whichever is applicable) to withstand stochastic disturbance events; redundancy as the ability of the taxon as a whole to withstand catastrophic events; and, representation as the ability of the taxon as a whole to adapt to changing environmental conditions. Thus, resiliency is assessed at the population level as well as the rangwide level while redundancy and representation are evaluated at the rangewide level only; this approach follows the Service’s framework for conducting a Species Status Assessment (Service 2015).

6.1. Biological considerations

Our concern for the future population status of the Alexander Archipelago wolf is tempered to some degree by the extraordinary resilience of wolves to high levels of take and the activities of humans (Mech and Boitani 2003), and their adaptability to man-altered landscapes (Mech 1995). In fact, as wolf populations recover elsewhere in the United States, wolves are inhabiting areas with higher road densities than earlier thought possible and more open and populated areas. Weaver et al. (1996, p. 964) describe three mechanisms that influence the resilience of large carnivores to disturbances that may affect their persistence: 1) behavioral plasticity in foraging behavior that ameliorates flux in food availability; 2) demographic compensation that mitigates increased exploitation; and, 3) dispersal that provides connectivity between fragmented populations. Wolves, with high potential annual productivity and long dispersal abilities, are considered among the most resilient of carnivores to human activities (e.g., Weaver et al. 1996, pp. 966–968). If food is available and wolves are not unduly persecuted, they can survive in highly altered areas; ultimately, it is human attitudes and values that will limit the number and distribution of wolves in North America (e.g., Mech 1995, p. 275), including southeastern Alaska and coastal British Columbia.

Wolves exhibit behavioral plasticity in foraging behavior by having access to a variety of prey and the ability to switch from one prey species to another depending on their availability and susceptibility to wolf predation. Alexander Archipelago wolves are opportunistic predators and take a variety of prey species, although on islands in southeastern Alaska, most of their diet appears to consist of deer and beaver (Tables 8 and 9). Wolves on POW apparently can kill beavers and other prey throughout the year (e.g., Kohira and Rexstad 1997, p. 430), which certainly could ameliorate the effects of reduced deer availability to some extent. However, it is uncertain, but probably unlikely, that beaver and other prey could sustain high wolf densities on islands in southeastern Alaska in the absence of deer.

Because of their high reproductive rates, wolves are able to capitalize on increases in prey biomass and compensate for increased mortality. Generally, gray wolves can sustain human-caused mortality rates of roughly 17–48% (e.g., Fuller et al. 2003, pp. 184–185; Adams et al. 2008, p. 22; Creel and Rotella 2010, p. 5; Sparkman et al. 2011, p. 5; Gude et al. 2012, pp. 113–116;). Human exploitation of wolves may increase the amount of prey biomass per wolf, which may increase productivity and survival. Under conditions of high prey availability, wolves may lower their age of first reproduction, have larger litters, and a greater proportion of females in an area may reproduce. Provided prey biomass is high enough, wolves are able to rapidly repopulate areas that have been depleted by hunting and trapping (e.g., Ballard et al. 1987, p. 20).

In parts of the range of the Alexander Archipelago wolf, total mortality of wolves through hunting and trapping may be occurring at unsustainable rates. Although wolves are able to sustain high levels of human take, unreported harvest probably has contributed to decline of the GMU 2 wolf population and may be having an undocumented impact on other populations. Recently, for GMU 2, the Alaska Board of Game reduced the cap for hunting and trapping mortality from 30% of the fall wolf population to 20%, demonstrating their ability to respond to changing conditions. However, it remains to be seen whether or not the change in the harvest guideline level will result in population stabilization given the high rates of unreported harvest and the predicted declines in deer habitat capability due to past timber harvest in GMU 2.

Because wolves are capable of dispersing hundreds of kilometers, often across inhospitable terrain, problems associated with inbreeding and genetic variability are uncommon. Alexander Archipelago wolves are known to disperse more than 160 km and swim up to at least several kilometers between islands (Person and Ingle 1995, p. 23); in coastal British Columbia, wolves occupy islands up to 13 km away from the nearest island or mainland (Darimont and Paquet 2002, p. 418). Nonetheless, water barriers may limit the ability of wolves to disperse among some islands in southeastern Alaska especially. Inter-island movements may be sufficient to prevent loss of heterozygosity in populations of Alexander Archipelago wolves, but may not be sufficient to buffer wolves on some islands from declines. Because wolves in southeastern Alaska have been studied little (outside of GMU 2), few data exist on the amount of interchange among wolves on various island groups and mainland, and between southeastern Alaska and British Columbia, although evidence suggests that low rates of migration occur (see *Within-population dispersal*). Generally, although widely debated, 1–10 migrants per generation are necessary to maintain sufficient gene flow among populations; Von Holdt et al. (2010, p. 4423) found that a minimum of 5.4 migrants per generation was necessary for gray wolves in the Rocky Mountains. Within the range of the Alexander Archipelago wolf, POW and the adjacent islands are among the most isolated and therefore the GMU 2 population likely experiences the least amount of interchange with the mainland populations (i.e., GMU 1/5A, Region 1/2, and Region 5/6).

Even in complete or nearly complete isolation though, wolves have demonstrated a remarkable ability to persist at low population levels. For example, wolf populations numbering less than a few hundred individuals have persisted for decades in Isle Royale National Park (Michigan), Riding Mountain National Park (Manitoba), northern Italy, and in parts of Norway and Sweden (summarized in Fuller et al. 2003, p. 190). We recognize that not all of these populations are stable or will persist into perpetuity, but they demonstrate that wolves can persist for many generations at low numbers. Furthermore, reintroductions in the Rocky Mountains demonstrate the capacity for rapid population growth for populations below 100 wolves. Regardless, it is widely accepted that small, isolated populations have a higher probability of extinction than large, connected populations and that as populations become small, they become susceptible to random events and may experience negative genetic consequences.

We are not aware of an established population threshold or minimum viable population estimate for the Alexander Archipelago wolf. Generally, most minimum viable population estimates for gray wolves seem to range between 100 and 500 wolves. For example, the Scandinavian wolf population, which has similar characteristics to some populations of the Alexander Archipelago wolf (e.g., partial isolation and high rates of unreported harvest), is assumed to be secure at a minimum of roughly 150–200 individuals (Liberg 2005, p. 39). In addition, the recovery of gray wolves in the Northern Rocky Mountains called for 100 wolves and 10 breeding pairs for at least three years in each of the three recovery areas (i.e., northwestern Montana, Yellowstone National Park, and central Idaho). It is unclear how these thresholds for other wolf populations apply to the Alexander Archipelago wolf, but they do provide some general guidance.

6.2. Assessment by population

For each population, we assessed range and population characteristics (e.g., % of range, population size, trend) and magnitude and exposure of stressors to the population (Table 24). We used our assessment to project resiliency of the population to future resource conditions, while acknowledging uncertainty. For GMU 2, the population for which the most data exist, we developed a spatially-explicit model to examine future population change of a hypothetical wolf population under several scenarios. For all other populations, our assessment is qualitative and should be viewed as a synthesis of available information and not as a quantitative risk or viability assessment.

Given the paucity of data available on wolves and similarities in stressors, we combined subunits A–D of GMU 1 and all of GMU 5A into one population of wolves that occupies the mainland of southeastern Alaska. For the same reasons, we combined Regions 1 and 2 and Regions 5 and 6 into two populations (southern and northern, respectively) in coastal British Columbia. Therefore, below we evaluate the potential for decline in five wolf populations: mainland southeastern Alaska (GMU 1/5A), GMU 2, GMU 3, northern coastal British Columbia (Region 5/6), and southern coastal British Columbia (Region 1/2). We began with GMU 2 because it is the population for which the most information on wolves exists and it harbors the greatest concentration of deterministic stressors; the remainder of populations is presented from north to south.

Table 24. Summary of range and population characteristics and primary stressors facing populations of the Alexander Archipelago wolf by Game Management Unit (GMU) in southeastern Alaska (SEAK) and Region in coastal British Columbia (BC). Estimates provided reflect the best available information; see text for details on estimate derivation, assumptions, and citations. Gray-shading provided for readability only.

Category		Metric	GMU 1/5A	GMU 2	GMU 3	Region 1/2	Region 5/6
			Mainland SEAK	POW in SEAK	Middle islands of SEAK	Southern coastal BC	Northern coastal BC
Range or population characteristic	Physical	Land area (km^2)	55,672	9,414	7,844	79,074	67,097
		% of wolf range	25%	4%	4%	36%	31%
		% of wolf range <400 m in elevation	18%	9%	8%	35%	30%
		Shoreline (km)	9,601	7,644	4,880	13,119	7,134
		Ratio of shoreline to land area	0.17	0.81	0.62	0.17	0.13
	Demographic	Estimated population size (95% CIs)	318[1] (170–466)	89[2] (50–159)	255[1] (136–373)	429[1] (212–646)	670[1] (297–1,043)
		% of rangewide population	18%	6%	14%	24%	38%
		Change in population size (SE)	Unknown	-6.7%[2] (2.8) per annum	Unknown	Stable	Stable
		Estimated predicted trend	Unknown	~-8–11% in 30 years[3]	Unknown	Unknown	Unknown
		Relative degree of insularity	Low	High	Intermediate	Low	Low
	Ecological	# of ungulate prey species present	3	1	3	3	3
Stressor	Timber harvest	% of forest logged	6%	23%	14%	30%	16%
		% reduction in deer habitat capability	~15%[4]	11–38%	13–23%	Unknown	Unknown
		Projected future logging (mmbf)	36.0	165.4	103.3	17–39%[5]	17–39%[5]
	Road development	Total roads (km)	2,795	5,712	2,467	48,632	18,980
		Mean road density (km/km^2)	0.08	0.62	0.26	0.54	0.23
		% of WAAs/WMUs over threshold	1%	42%	4%	21%	0%
	Wolf harvest	Mean annual reported wolf harvest (range)	62 (36–86)	52 (20–96)	54 (21–95)	15 (1–107)	16[6] (1–111)
		Mean percent of annual reported harvest to population size (range)	19% (11–27)	17% (6–33)	21% (8–37)	7% (1–25)	5%[6] (1–21)
		Estimated unreported harvest	Documented	38–45% of total harvest	Suspected	Unknown	Unknown
	Winter severity	Relative snow accumulation	Highest	Lowest	Intermediate	Lowest	Intermediate

[1] Based on the midpoint of range; [2] Empirical estimates from ADFG 2015b, pp. 1–2; [3] Based on Gilbert et al. 2015; [4] Available only for GMUs 1A and 1B, following Albert and Schoen (2007, p. 31); [5] Estimated percent loss of old-growth forest and mature young-growth forest between 2005 and 2100; comparable estimate for all of southeastern Alaska is 12% (Service 2010, p. A-12); [6] Minimum mean annual wolf harvest; reporting is not required.

6.2.1. Game Management Unit 2 wolf population

GMU 2 constitutes 4% of the range of the Alexander Archipelago wolf and 9% of its range below 400 m in elevation where wolves spend most of their time (see *Space and habitat use*; Table 24). Currently, the GMU 2 wolf population is estimated as 89 wolves (95% CI=50–159), based on the most recent field effort in fall 2014 (ADFG 2015b, p. 2). Using this estimate, the GMU 2 wolf population currently is roughly 6% of the overall estimated population of Alexander Archipelago wolves and 15% of the southeastern Alaska portion only. However, the highest population estimate for GMU 2 was 356 wolves (95% CI=148–564) wolves in 1994; using this value, the GMU 2 wolf population is roughly 18% of the overall population and 25% of the southeastern Alaska population. The wolf population in GMU 2 has declined by 75% (SE=15) since 1994, or an average of -6.7% (SE=2.8) per annum, based on population estimates, although the variance surrounding these estimates is substantial and the confidence intervals overlap.

Using a spatially-explicit model, we predicted future population change of a hypothetical wolf population in GMU 2 under six possible scenarios that involved perturbations of four primary stressors: vegetation, wolf harvest, road density, and frequency of severe winters (Table 25). Each stressor (or parameter) had several conditions, which are briefly described in Chapter 5 and in more detail in Gilbert et al. (2015). We developed the scenarios and defined conditions during a 2-day technical model review workshop with experts on wolf biology and management, population modeling, spatial analysis, and forest management in March 2015. At the workshop, we also received constructive feedback on the model itself; after revising the model based on that feedback and generating the necessary spatial data, we ran these six scenarios, and the sensitivity analyses described above, through the final model.

Table 25. Description of scenarios evaluated using population model to estimate percent change in mean wolf abundance in Game Management Unit 2 between 2015 and 2045. We used Scenario B (*) as a base model to conduct the sensitivity analyses described above in Chapter 5.

Scenario	Parameter	Condition	Description
No Change	Vegetation	No change	Natural succession only.
	Roads	No change	Current road densities.
	Wolf harvest	20% harvest cap	Harvest of 20% of fall population estimate, plus unreported harvest.
	Frequency of severe winter	Predicted average	Frequency of 0.08.
Scenario A	Vegetation	No change	Natural succession only.
	Roads	Planned decommission	Reduction of 2.2% of current road densities.
	Wolf harvest	No legal harvest	Unreported harvest only.
	Frequency of severe winter	Predicted low	Frequency of 0.07.
Scenario B*	Vegetation	Young-growth transition	Harvest of 53.0 mmbf/year[1].
	Roads	Planned decommission	Reduction of 2.2% of current road densities.
	Wolf harvest	20% harvest cap	Harvest of 20% of fall population estimate, plus unreported harvest.
	Frequency of severe winter	Predicted average	Frequency of 0.08.
Scenario C	Vegetation	Continued harvest of old-growth	Harvest of 80.6 mmbf/year.
	Roads	No change	Current road densities.
	Wolf harvest	20% harvest cap	Harvest of 20% of fall population estimate, plus unreported harvest.
	Frequency of severe winter	Predicted average	Frequency of 0.08.
Scenario D	Vegetation	Increased harvest of old-growth	Harvest of 183.6 mmbf/year.
	Roads	No change	Current road densities.
	Wolf harvest	30% harvest cap	Harvest of 30% of fall population estimate, plus unreported harvest.
	Frequency of severe winter	Predicted high	Frequency of 0.10.
Scenario E	Vegetation	Maximum harvest of old-growth	Harvest of 273.2 mmbf/year.
	Roads	Road construction	Construction of 30% more roads above current densities.
	Wolf harvest	30% harvest cap	Harvest of 30% of fall population estimate, plus unreported harvest.
	Frequency of severe winter	Predicted high	Frequency of 0.10.

[1] Million board feet (mmbf).

Of the six scenarios, Scenario A resulted in the smallest percent change in mean wolf abundance between 2015 and 2045 (-5%), although this same scenario also resulted in the largest percent change in mean deer abundance (-33%; Figure 26). Scenario E ranked worst in terms of change in wolf abundance (-20%) and next to last for change in deer abundance (-32%). Overall, Scenario B seemed to serve as the most optimistic model for both wolf and deer abundance, resulting in -8% and -21% change, respectively, as well as the lowest associated variance, and based on the conditions used in this model, it also seems to be the most probable to occur over the next 30 years.

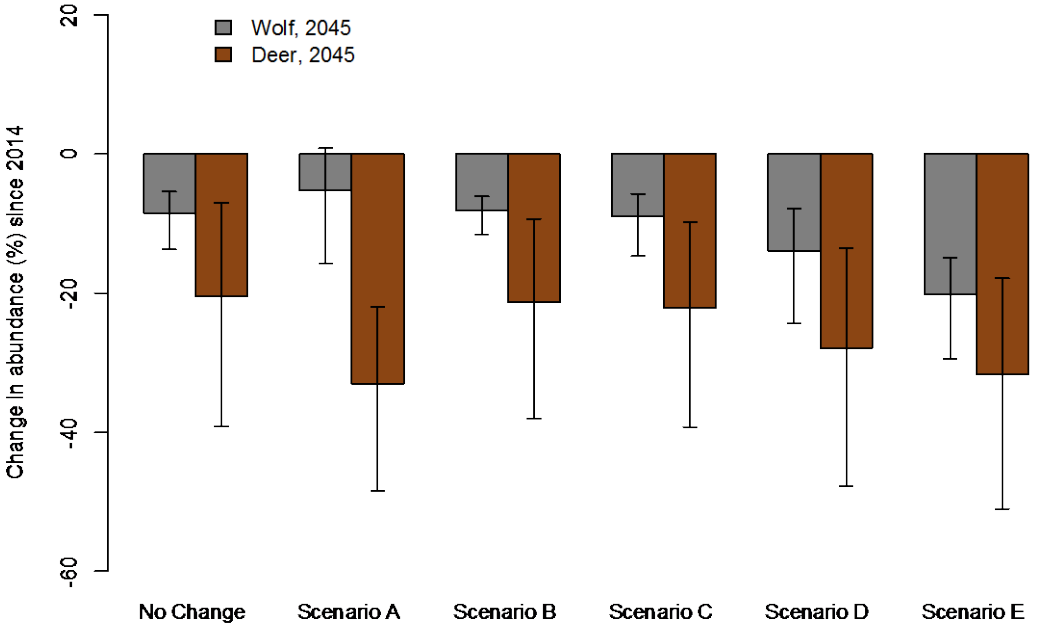

Figure 26. Estimated percent change in mean abundance of wolves and deer under six scenarios with variations of vegetation, wolf harvest, road density, and frequency of severe winter conditions between 2015 and 2045 in Game Management Unit 2, southeastern Alaska. Error bars denote 95% confidence intervals.

Consistent with knowledge of predator–prey dynamics, change in wolf abundance was not uniform over the time period modeled, but instead wolves were responsive to their primary prey, deer. Between 2000 and present, we used actual (empirical) wolf harvest and winter severity data to inform the model. Results demonstrate that following severe winters deer abundance declines sharply and immediately with a corresponding increase in wolf abundance, which then declines several years later in response to low deer numbers (Figure 26). Presently, wolf abundance is predicted to be at a low point, but is expected to increase within the next few years owing to stabilizing deer abundance. Over the 30-year future period, however, both wolf and deer abundance are predicted to decline with a larger percent change occurring for deer than wolves, primarily due to reduced carrying capacity for deer, under all scenarios (Figure 27). See Gilbert et al. (2015) for model assumptions and evaluation of how well they were met.

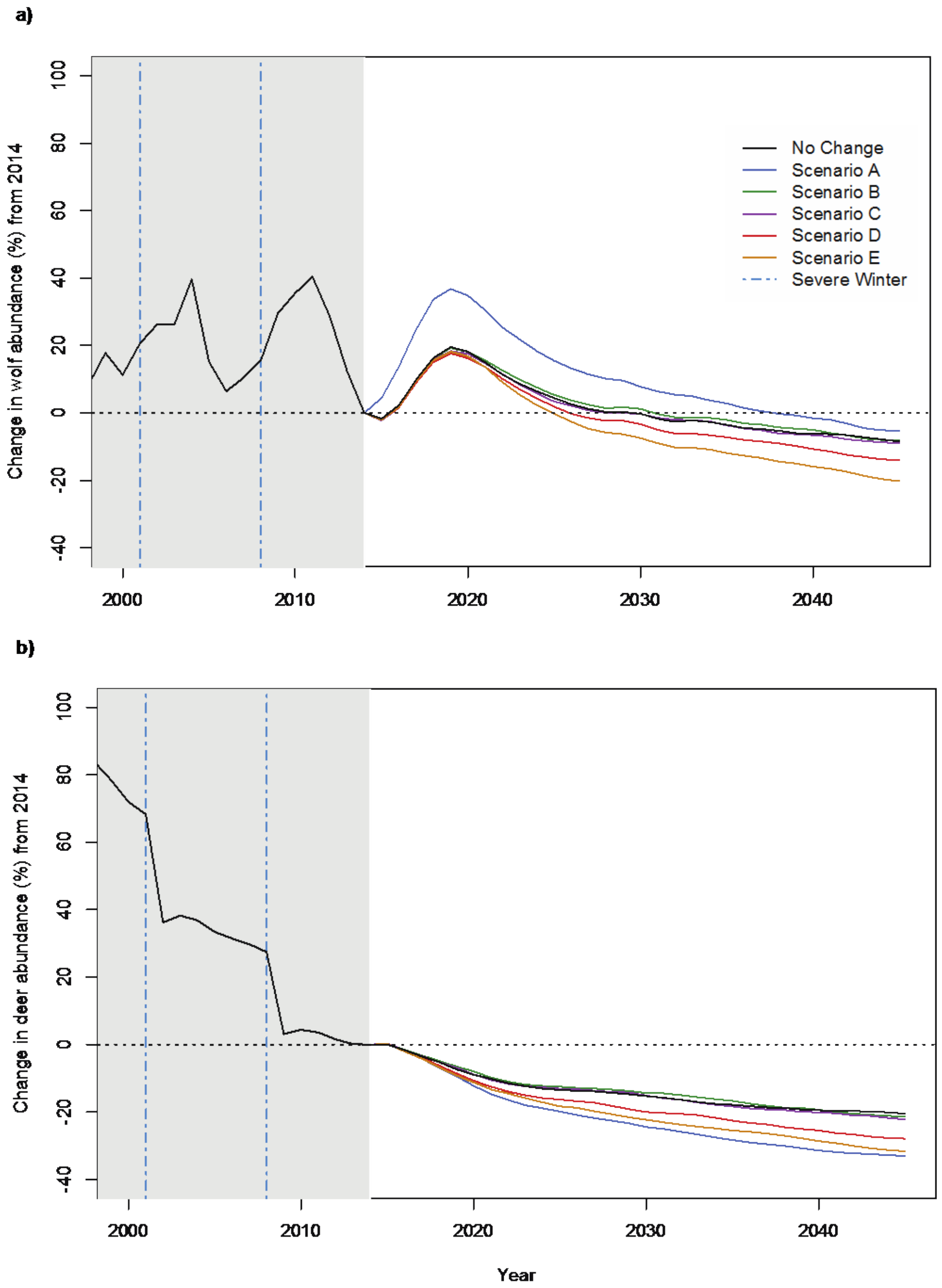

Figure 27. Percent change in (A) wolf abundance and (B) deer abundance across model scenarios from 2014 levels, shown from the year 2000–2045, in Game Management Unit 2. The grey box represents the years in the model (before 2015) where actual severe winters (dotted blue line) and reported wolf harvest were used as model inputs, while the white portion represents mean model predictions. See Gilbert et al. (2015) for details.

The GMU 2 wolf population based on empirical data apparently has declined by about 75% (SE=15) between 1994 and 2014 with the steepest decline occurring over a 1-year period between 2013 and 2014 when the population was reduced by about 60% (see Table 3 for point estimates and associated variance). Although our model results indicate that the current wolf population is at its lowest point since 2000, the steep decline observed over a 1-year period suggests that the GMU 2 wolf population may be at more elevated risk of decline than was predicted by our model. Only a few possible explanations exist that can explain that level of decline between 2013 and 2014, assuming that the trend estimate is reliable, and its inconsistency with our model results. First, actual rates of unreported wolf harvest were higher between 2013 and 2014 than were included in the model (42% of total harvest); we believe this explanation is most likely given our knowledge of unreported harvest in GMU 2 (see *Unreported harvest*). Second, we did not account for unreported harvest of deer in our model, but given the high sensitivity of wolf abundance to reduced deer harvest (Figure 23) and apparent high levels of unreported take of deer in GMU 2 (ADFG 2013c, p. 37), it is possible that actual deer abundance over the last few years was much lower than predicted by the model. Other possible explanations include widespread disease in the population, although no evidence of this situation exists, overall reduced deer availability, and decreased ability of wolves to catch and kill deer.

In summary, based on information regarding primary stressors evaluated with the GMU 2 wolf model, observed population decline, and model-predicted population declines, we project that GMU 2 wolf population likely will decline in the future. Given that this wolf population relies on only one ungulate species as prey (i.e., deer; Table 7), it is especially susceptible to changes in deer abundance. Roughly 23% of the forests in GMU 2 have been logged (Table 13), resulting in a reduction of 11–38% of the deer habitat capability, and another 165.4 mmbf is slated for harvest (47% of the future harvest in all of southeastern Alaska; Table 15). Despite efforts by the USFS and others to improve habitat for deer with intermediate treatments, the majority of previously logged stands has entered an age range (>25 years old; Figures 7, 9) when restoration techniques are less effective. Therefore, we believe that for the short-term (approximately the next 30 years) deer abundance will decline (unless deer harvest is eliminated or sharply curtailed, which is highly unlikely; Figure 23) and wolf abundance probably will respond similarly, but at a slower rate (Figure 27).

However, declines in wolf abundance could be mitigated for through reduced wolf and deer harvest, which both could be managed somewhat by restricting access of hunters and trappers. Current rates of reported wolf harvest appear to be sustainable (mean=52 wolves, or about 17% of the population; Table 21), but a substantial amount of unreported wolf harvest has been documented in GMU 2 (38–45% of total harvest). These high rates of total harvest in GMU 2 are facilitated by greater access for hunters and trappers; GMU 2 has the highest road density (mean=0.62 km/km^2) and ratio of shoreline to land area (0.81; proxy to boat access) across the entire range of the Alexander Archipelago wolf (Table 24). Therefore, we consider the GMU 2 wolf population to demonstrate low resiliency and, as predicted by our model, we anticipate further declines in wolf abundance in GMU 2 over the next 30 years (an average decline of 8–14% of current population), largely owing to reduced deer abundance due to timber harvest, high rates of total wolf harvest, and a combination of these factors.

6.2.2. Mainland of southeastern Alaska wolf population (GMU 1/5A)

The combined population estimate of wolves in GMU 1/5A on the mainland in southeastern Alaska is 318 wolves, ranging between 170 and 466, approximately 18% of the rangewide population of Alexander Archipelago wolves and 47% of the southeastern Alaska portion (Table 24). GMU 1A supports the majority (57%) of the mainland wolf population in southeastern

Alaska. We found no data to estimate the trend of the wolf population in GMU 1/5A. Therefore, we evaluated characteristics of the range and population and the primary stressors that appear to have the greatest relative strength on wolf abundance (informed by the sensitivity analysis conducted using the GMU 2 wolf population model).

Primary stressors to the mainland wolf population in southeastern Alaska (GMU 1/5A) are comparatively low (Table 24). Across the entire area, only 6% (total=665 km^2) of the productive forest has been logged and only 36 mmbf is projected to be logged in the future (10% of planned future timber harvest across all of southeastern Alaska; Tables 14 and 15). Further, given that deer (and wolves) occur naturally at low abundance on the mainland, we anticipate low levels of impact to the wolf population in GMU 1/5A from past and future timber harvest; in addition, wolves in GMU 1/5A have access to ungulate species other than deer (Table 7). Likewise, reported wolf harvest rates between 1997 and 2014 appear to be sustainable across the entire mainland (mean annual harvest=62 wolves, or ~19% of the population), although harvest is higher in the southern portion of the mainland (GMU 1A; Table 21) where wolves are presumably more abundant (Table 4). Unreported harvest of wolves has been documented on a few occasions in GMUs 1A and 1C and suspected in GMU 1B, but has not been quantified in any subunit along the mainland; given the low level of road and boat access for hunters and trappers (Tables 20 and 24), we suspect that rates of unreported harvest in GMU 1/5A also are low compared to GMU 2.

In summary, we found no reliable data to indicate population trend of the wolf on the mainland of southeastern Alaska, yet we believe that resiliency of this wolf population likely is high based on its similarities with populations in coastal British Columbia. Although the mainland receives higher snowfall compared to the islands, snowfall is predicted to decrease in the future, resulting in improved conditions for deer and wolves (see *Climate-related events*). Further, wolves on the mainland have access to ungulate species other than deer and are less insular than island wolf populations. GMU 1A presents the greatest potential for local population decline along the mainland, but even still, in this subunit only 6% of the forest has been logged (Table 13), mean annual reported harvest appears to be sustainable (mean=16% of the population annually; Table 21), and hunter and trapper access is the lowest across the range of the wolf (Table 20). Nonetheless, the value of the mainland to the Alexander Archipelago wolf should not be under estimated; it connects the coastal populations with the interior populations via transboundary river corridors and the southeastern Alaska populations with the coastal British Columbia populations along the rugged coastline. Therefore, high resiliency of the GMU 1/5A population benefits the rangewide population of Alexander Archipelago wolf.

6.2.3. Game Management Unit 3 wolf population

The GMU 3 wolf population is estimated to be 255 wolves, or between 136 and 373 wolves, constituting ~14% of the current rangewide estimated population of Alexander Archipelago wolves and 36% of the southeastern Alaska portion. We found no data to estimate trend of the wolf population in GMU 3. Therefore, similar to the mainland population (GMU 1/5A), we evaluated characteristics of the range and population and the primary stressors that appear to have the greatest relative strength on wolf abundance.

In GMU 3, primary stressors occur at intermediate levels compared to other GMUs and Regions (Table 24). Although only 14% of the forest has been logged in this area (Table 13), reducing deer habitat capability by 13–23% since 1954 (Albert and Schoen 2007, p. 16), nearly 30% of the projected timber harvest is scheduled to occur in GMU 3 (Table 15). However, ungulate species other than deer occur in GMU 3 (Table 7), relaxing dependency of wolves on deer. Rates of reported wolf harvest appear to be sustainable (mean=21% of the population annually) and although managers suspect that unreported harvest is occurring, it has not been confirmed or quantified. We determined that boat and road access is lower in GMU 3 than in GMU 2 (Table 20) and therefore, we postulate that rates of unreported harvest likely are lower also, but not zero; in addition, wolves in GMU 3 probably have greater dispersal capability compared to wolves in GMU 2, which likely improves their ability to compensate for intermediate rates of harvest.

In summary, we have no information on the status or trend of the GMU 3 wolf population, but when the population characteristics and primary stressors are considered collectively, we believe that the GMU 3 population has intermediate resiliency compared to other wolf populations. An operational plan for intensive management for deer in GMU 3 has been developed and involves the culling of ~50 wolves, or 20% of the current estimated population (ADFG 2013b, pp. 8–9; Table 4). Although the program currently is inactive, if implemented the GMU 3 wolf population would be reduced, given that it is the goal of the program, potentially having an effect on the GMU 2 population because GMU 3 provides the most reasonable transit path for wolves to move or disperse between the mainland and GMU 2 (Figure 2). Therefore, maintaining or reducing current rates of wolf harvest in GMU 3 would benefit the rangewide population of Alexander Archipelago wolves; an increase in mortality rates likely would lower immigration rates to GMU 2, which apparently are uni-directional (Breed 2007, p. 22), thereby increasing the vulnerability of the GMU 2 wolf population.

6.2.4. Northern coastal British Columbia wolf population (Region 5/6)

We estimate that the wolf population in northern coastal British Columbia, Region 5/6, is 670 wolves, ranging from 297 to 1,043 wolves, representing ~38% of the overall population of the Alexander Archipelago wolf. Of all of the estimates presented in this section, this estimate is the least precise and may be the least accurate (biased slightly high; see *Abundance and trend* for details). Nonetheless, we generated it using the best available scientific data, to the best of our knowledge. We found that the wolf population in Region 5/6 is thought to be stable or slightly increasing (Kuzyk and Hatter 2014, p. 881; BCMO 2015a), as is true of the provincial wolf population (BCMO 2014, p. 25). Therefore, we assume that stressors facing this population are not having a population-level impact, indicating that the population shows high resilience to the magnitude of stressors currently present.

The magnitude of primary stressors to the Region 5/6 wolf population (northern coastal British Columbia) appear to be low relative to other populations (Table 24), although data on wolf harvest represent minimum values only. Based on voluntary reporting of wolf harvest, we estimated annual minimum take of 16 wolves (5% of the population annually) with higher rates of annual harvest reported in Region 6 (mean=28 wolves) compared to Region 5 (mean=4 wolves) where estimated road density is more than two times lower (Tables 18 and 22). Across Region 5/6, mean road density is 0.23 km/km^2 with only one WMU near the recommended road density threshold (Figure 14, Table 18). Timber harvest in Region 5/6 is lower than in GMU 2 and Region 1/2 and is spatially concentrated (Figure 10); across all of Region 5/6 approximately 16% of the forest has been logged, although additional logging at a loss of about 1% per year is expected in the future if market conditions allow (Service 2010, p. A-12). However, despite these stressors, the wolf population in Region 5/6 has remained stable for the last 15 years.

In summary, we believe that the Region 5/6 wolf population demonstrates high resiliency, as evidenced by its apparent stability since 2000. We hypothesize that access to ungulate species other than deer, an apparently high degree of connectivity with other wolf populations, and low levels of boat and road access for hunters largely are responsible for the current stability and high resiliency of the Region 5/6 wolf population. This conclusion is favorable to the GMU 1 wolf population given that the Region 5/6 population may serve as a source population to GMU 1A (Breed 2007, p. 34).

6.2.5. Southern coastal British Columbia wolf population (Region 1/2)

The southernmost population of Alexander Archipelago wolf in Region 1/2 of coastal British Columbia is estimated to be 429 wolves, ranging between 212 and 646, approximately 24% of the rangewide population. Similar to the Region 5/6 wolf population, the Region 1/2 wolf population has been stable or slightly increasing since 2000 (Kuzyk and Hatter 2014, p. 881; BCMO 2015a). Therefore, we presume that the stressors to the Region 1/2 wolf population are not having an impact at the population-level and that this population is resilient to the current magnitude of stressors.

We found that primary stressors to the Alexander Archipelago wolf in Region 1/2 occurred at low to intermediate levels compared to other wolf populations (Table 24). Although boat access was low compared to GMUs 2 and 3, road access was somewhat high (Table 20), but we attribute the high road densities to the cities of Victoria (Region 1) and Vancouver (Region 2), which likely are not resulting in increased wolf harvest. In fact, unlike Region 5/6, wolf harvest reporting is compulsory in Region 1/2 and, based on those data, mean annual reported harvest composed only 7% of the population annually; we found no reports of unreported harvest occurring in this region. In Region 1, timber harvest has been intensive; 34% of the productive old-growth forest has been logged, but harvest is expected to decline over the next 35 years (Service 2010, p. A-11) presumably because fewer trees are left to harvest. In contrast, in Region 2, only 12% of the forest has been logged. Timber forecasts indicate that additional logging at the rate of about 1% per year may occur by 2100 provided that the market remains stable. Nonetheless, despite the intensive and extensive timber harvest and the high road densities, the Region 1/2 wolf population has been stable over the last 15 years.

In summary, similar to Region 5/6, the Region 1/2 wolf population demonstrates high resiliency in the face of stressors to the population. We postulate that access to alternative ungulate species (other than deer), comparatively low levels of reported wolf harvest, and high levels of connectivity with other wolf populations likely results in a resilient wolf population even though other stressors such as timber harvest occur at a high magnitude. We suspect that of all the populations assessed in this document, wolves in Region 1/2 probably interact more frequently with other gray wolves (i.e., not *C. l. ligoni*, as assumed in this assessment; see *Taxonomy* and *Connectivity*), which may enhance demographic compensation and stability.

6.2.6. Summary of individual populations

In our assessment of each population, we determined that three populations of the Alexander Archipelago wolf demonstrate high levels of resiliency (GMU 1/5A, Region 5/6, and Region 1/2), one population exhibits an intermediate level of resiliency (GMU 3), and one population shows a low level of resiliency (GMU 2; Table 26). For populations in coastal British Columbia, key evidence in determining level of resiliency was the stable (or slightly increasing) population trend even though stressors to those populations were somewhat high (e.g., road development, timber harvest). Similarly, we relied largely on estimates of population change and vital rates (e.g., survival) relative to the magnitude of the stressors to categorize resiliency as being low for this population. In GMUs 1/5A and 3 where trend information is not available, we compared the evidence with populations for which trend information is available (i.e., coastal British Columbia and GMU 2) and found that, for the most part, these populations were more similar to coastal British Columbia than GMU 2. Overall, lack of empirical abundance and trend estimates, as well as low precision of existing estimates, were the primary sources of uncertainty. In addition, we found little information, scientific or observational, on individual movements of Alexander Archipelago wolves across populations.

Table 26. Relative level of resiliency of individual populations of the Alexander Archipelago wolf, southeastern Alaska and coastal British Columbia (order follows north to south).

Wolf population	Level of resiliency	Key evidence	Uncertainties
GMU 1/5A	High	• Low percent of forest logged • Reported wolf harvest appears to be sustainable • Boat and road access is limited • Ungulate species other than deer available as prey • Few disruptions to demographic and genetic connectivity	• No empirical abundance or trend data • Severe winters likely will affect population more so than others • Unreported harvest documented on few occasions, but not quantified • Southern portion of GMU may be sink population
GMU 3	Intermediate	• Intermediate level of timber harvest • Reported wolf harvest appears to be sustainable • Boat access is high, but road access is low • Ungulate species other than deer available as prey • Intermediate insularity of population	• No empirical abundance or trend data • Connectivity may be more limited than for mainland populations due to island geography • Unreported harvested suspected, but not quantified • Distribution and abundance of ungulate prey species are not uniform across GMU
GMU 2	Low	• Reduction wolf population over last 20 years • High percent of forest logged with expected declines in deer • High rates of unreported harvest documented • Boat and road access for hunters and trappers is highest across range of taxon • Deer serve as only ungulate species for prey • High insularity of population	• Confidence intervals of abundance estimates overlap; decline not statistically significant • Abundance estimated in portion of GMU and extrapolated uniformly to entire GMU • Unreported harvest is highly variable and difficult to document • No information on immigration rates; emigration appears to be low
Region 5/6	High	• Wolf population stable since 2000 despite intermediate levels of timber harvest • Ungulate species other than deer available as prey • Few disruptions to demographic and genetic connectivity • Appears to serve as source population	• Assume wolf trend in entire Region is representative of coastal portion • Reporting of wolf harvest is not required; assume harvest is occurring at sustainable levels • No information on movements
Region 1/2	High	• Wolf population stable since 2000 despite high rates of timber harvest • Ungulate species other than deer available as prey • Few disruptions to demographic and genetic connectivity	• No information on impacts of urbanization • Likely to interact with gray wolves (*C. l. nubilus*) from continental North America, which may increase taxonomic uncertainty • No information on movements

6.3. Characterizing future status

In this section, we characterize the future status of the rangewide population of Alexander Archipelago wolf. To do so, we used the conservation biology principles of redundancy, resiliency, and representation (Table 27). Together, these principles and their core autecologic parameters of abundance, distribution, and diversity, comprise the key characteristics that contribute to the Alexander Archipelago wolf's ability to sustain a healthy rangewide population over time.

6.3.1. Redundancy

We defined redundancy as the ability of the taxon as a whole to withstand catastrophic events and evaluated it qualitatively using the geographic scope of the range and the number and spatial distribution of populations (Table 27). The current range of the Alexander Archipelago wolf is broad, encompassing ~219,000 km^2 in area and is similar to its recent historical range. As defined in this assessment, multiple populations (at least about 10) occur within its current range. Although some island-based populations, especially GMU 2, are more insular than mainland populations, they occupy only 8% of the range and 17% of the range below 400 m in elevation where Alexander Archipelago wolves tend to spend time (see *Resource selection*). Therefore, we believe that the Alexander Archipelago wolf has the ability to withstand catastrophic events given its broad range with multiple populations distributed across it and the degree of connectivity among most of those populations.

6.3.2. Resiliency

We defined resiliency as the ability of the taxon as a whole to withstand stochastic disturbance events and assessed it by considering the collective resiliency of the individual populations (Tables 26 and 27). In the previous section, we found that three populations of the Alexander Archipelago wolf demonstrate high resiliency (GMU 1/5A, Region 5/6, and Region 1/2), one population exhibits intermediate resiliency (GMU 3), and one population shows low resiliency (GMU 2; Table 26). We considered the numerical contribution of these populations to the rangewide population and found that overall 80% of the rangewide population exhibits high resiliency, 14% demonstrates intermediate resiliency, and 6% shows low resiliency. Although one population (i.e., GMU 2) of the Alexander Archipelago wolf may not contribute significantly to the resilience of the rangewide population, we conclude that the taxon as a whole has the ability to withstand stochastic disturbance events.

6.3.3. Representation

We defined representation as the ability of the taxon as a whole to adapt to changing environmental conditions and evaluated it by reviewing the breadth of the ecological and genetic diversity (Table 27). Given that the Alexander Archipelago wolf is a habitat and diet generalist with a variable diet across seasons and throughout its range, it exhibits ecological diversity at the rangewide level. In our review, we did not find evidence of unique or rare behaviors specific to wolves within a specific population or group of populations; Alexander Archipelago wolves appear to be highly adaptive to their environment. Although continental wolves harbor higher genetic diversity than their coastal counterparts (e.g., Weckworth et al. 2010, p. 372), we did not find evidence of recent genetic bottlenecking (Weckworth et al. 2005, p. 924; Breed 2007, p. 18; Weckworth et al. 2011, p. 5). The populations in coastal British Columbia exhibit higher genetic diversity compared to those in most of southeastern Alaska, especially GMU 2, and appear to serve as source populations to southern southeastern Alaska (Breed 2007, p. 34). We believe that the Alexander Archipelago wolf has the ability to adapt to changing environmental conditions owing to its high level of behavioral plasticity in foraging, general lack of preference for habitat use, and a comparatively high degree of genetic diversity in the majority of the rangewide population (62% occurs in coastal British Columbia).

Table 27. Characterization of future status of the Alexander Archipelago wolf using the conservation biology principles of redundancy, resiliency, and representation.

Principle	Definition	Metric	Key evidence
Redundancy	Ability of the taxon to withstand catastrophic events	Geographic scope of the range and spatial distribution and number of populations	• Geographic scope of range is large (about 217,000 km^2) and is similar to recent historical range • Multiple populations occur throughout current range indicating high redundancy • One population (GMU 2) is more insular than others, but it occupies only 4% of the range and 9% of the range below 400 m in elevation • Two island-based populations (GMUs 2 and 3) probably experience the least connectivity with other populations; they occur within 8% of the range and 17% of the range below 400 m in elevation • Mainland populations with fewer disruptions to connectivity with one another occupy 92% of range and 83% of range below 400 m in elevation
Resiliency	Ability of the taxon to withstand stochastic disturbance events	Synthesis of resiliency of individual populations	• One population (GMU 2) exhibits low resiliency (6% of rangewide population) • One population (GMU 3) shows intermediate resiliency (14% of rangewide population) • Remainder of populations (GMU 1/5A and coastal British Columbia) demonstrate high resiliency (80% of rangewide population) • Overall, 80% of the rangewide population is exhibits high resilience
Representation	Ability of the taxon to adapt to changing environmental conditions	Breadth of genetic and ecological diversity	• Habitat and diet generalist and high variation in food habits across seasons and range indicate high ecological diversity • Genetic diversity of Alexander Archipelago wolves appears to be lower than for interior continental wolves, but no evidence of recent genetic bottlenecking • Genetic variation exists among populations with the greatest diversity exhibited in populations in coastal British Columbia and lowest in the highly insular GMU 2 population • Representation appears to be high across range of taxon, especially in coastal British Columbia where most of the rangewide population occurs

6.4. Summary

Across its range, the Alexander Archipelago wolf encounters anthropogenic and environmental stressors, although they are not uniformly distributed, nor does the wolf respond similarly to all stressors. Generally, coastal wolves are resilient, feeding on a variety of prey items and using most habitat types throughout their annual cycle. This ecological and behavioral plasticity permits them to endure conditions that other species with narrow biological niches may not be able to tolerate. However, some populations of Alexander Archipelago wolf are more insular than others, lowering their resiliency and increasing their vulnerability to stressors.

In our review, we found that all but one population (i.e., GMU 2) exhibited intermediate or high resiliency to stressors. Collectively, these resilient populations occupy 96% of the range of the Alexander Archipelago wolf and 91% of the range below 400 m in elevation where wolves tend to spend their time; in addition, they comprise 94% of the rangewide population of the Alexander Archipelago wolf. In coastal British Columbia where 62% of the rangewide population occurs, trends in wolf abundance have been stable or slightly increasing since 2000 even though substantial timber harvest has occurred and road access for hunters and trappers is somewhat high. Although uncertainty regarding the status of wolf populations in GMUs 1, 3, and 5A is greater than for those in coastal British Columbia, population characteristics and magnitude of stressors tend to be more similar to coastal British Columbia than to GMU 2, based on the best available information (Tables 24 and 26).

In GMU 2, however, we found that the wolf population demonstrated low resiliency to stressors. In fact, this population apparently has declined considerably from past abundance already and is predicted to decline further over the next 30 years (Figure 27). The GMU 2 wolf population has been disproportionately impacted by timber harvest, which has reduced deer habitat capability, for two reasons. First, wolves in GMU 2 rely heavily on deer as the only ungulate prey species available, and second, rates of timber harvest in GMU 2 are among the highest within the range of the Alexander Archipelago wolf (Table 24). In addition, although reported wolf harvest is occurring within sustainable limits (Figure 18, Table 21), high rates of unreported harvest result in unsustainable total wolf harvest in some years (Figure 20). GMU 2 offers the highest levels of boat and road access to hunters and trappers (Table 20), which facilitates harvest of wolves. Further, the GMU 2 wolf population is more insular than the others, probably due to difficult water crossings and to the geographic position of this GMU; it is not on a transitory pathway, but instead likely is a destination for dispersing wolves. Although this population exhibits low resiliency, it occupies only 4% of the range and 9% of the range below 400 m in elevation and composes only 6% of the rangewide population.

In conclusion, we believe that the future status of the rangewide population of the Alexander Archipelago wolf likely will be stable or perhaps slightly lower than its current status based on its resiliency, redundancy, and representation (Table 27). Owing to predicted declines in the GMU 2 wolf population, it is possible that the rangewide population may decrease slightly, but we expect the overall effect to be minor given that the GMU 2 population constitutes only 6% of the rangewide population, is geographically peripheral to the other populations, and appears to serve as a sink population. Nonetheless, the persistence of the GMU 2 population is desired and requires careful management actions and decisions to ensure its future health.

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APPENDIX I. Land area (km^2) of Game Management Units (GMU) in southeastern Alaska and portion of Regions in coastal British Columbia, within the range of the Alexander Archipelago wolf.

Location	Identifier	Land area (km^2)
Southeastern Alaska	GMU 1	47,904
	1A	13,727
	1B	7,828
	1C	19,451
	1D	6,898
	GMU 2	9,414
	GMU 3	7,844
	GMU 5A	7,768
Coastal British Columbia	Region 1	46,507
	Region 2	32,567
	Region 5	23,194
	Region 6	43,903
Total		219,101