Species Status Assessment for the Coastal Marten (*Martes caurina*) Version 2.2
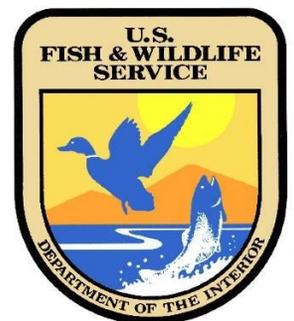

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Species Status Assessment for
Coastal Marten (*Martes caurina*)
Prepared by the
U.S. Fish and Wildlife Service

EXECUTIVE SUMMARY

This species status assessment reports the results of the comprehensive status review for the distinct population segment (DPS) of the Pacific marten (*Martes caurina*) in coastal Oregon and northern coastal California, commonly referred to as the coastal marten, and provides a thorough account of the overall viability and extinction risk for the coastal marten. The coastal marten is a mammal in the weasel family and is native to forests of coastal Oregon and coastal California. They occur primarily in older forests, although there is one remnant population occupying the coastal shore pine forest in the dunes ecosystem of central Oregon.

To evaluate the biological status of the coastal marten both currently and into the future, we assessed a range of conditions to allow us to consider the species' resiliency, redundancy, and representation (together, the 3Rs). Coastal marten needs multiple resilient populations distributed widely across its range to maintain persistence into the future and to avoid extinction. Several factors influence whether coastal marten populations will increase to maximize habitat occupancy, which increases the resiliency of a population to stochastic events. These factors are the connectivity between populations, amount of suitable habitat for establishing home ranges, and amount of habitat that allows for predator avoidance. As we consider the future viability of the species, more populations with high resiliency distributed across the known range of the species are associated with higher overall species viability.

Coastal marten historically ranged throughout coastal Oregon and coastal northern California, but have not recently been detected throughout much of the historical range, despite extensive surveys. The species exists in four small populations and is absent from the northern and southern ends of its historical range. We have assessed the coastal marten's levels of resiliency, redundancy, and representation currently and into the future by ranking the condition of each population. Rankings are a qualitative assessment of the relative condition of occupied habitat based on the knowledge and expertise of Service staff, as well as published reports.

Our analysis of the past, current, and future influences on what the coastal marten needs for long–term viability revealed that there are two factors that pose the largest risk to future viability of the species. These risks are primarily related to habitat loss and associated changes in quality and distribution: decrease in connectivity between populations, and habitat conversion from that suitable for martens to that suitable for generalist predators and competitors, potentially

increasing interactions and subsequent marten injury, mortality, or predation; these are all influenced by vegetation management, wildfire, and changing climate.

The coastal marten faces a variety of risks from loss of habitat, wildfire, and increased predation risk. These risks play a large role in the resiliency and future viability of the coastal marten. If populations lose resiliency, they are more vulnerable to extirpation, with resulting losses in representation and redundancy. Given the uncertainty regarding connectivity between populations, suitable habitat, and increases in predation within the populations, we have forecasted what the coastal marten may have in terms of resiliency, redundancy, and representation under three plausible future scenarios. While we don't expect all assumptions to be met in each scenario, these scenarios are meant to give a range of possibilities for the future. To create this range of possibilities we made the following assumptions about stressors to the populations:

(1) Scenario One (Continuation):

- Central Coastal Oregon – Trapping is legal. Loss of older forest continues to follow current trends on Federal and private lands due to wildfire and vegetation management, proposed restoration on the Oregon Dunes National Recreation Area moves forward potentially affecting currently occupied suitable habitat. There is no change in connectivity across Highway 101.
- Southern Coastal Oregon – Trapping is legal. There is a moderate loss of older forest on Federal and private lands due to wildfire and vegetation management.
- California–Oregon Border – There is little change to older forest on Federal and private lands due to limited impacts of wildfire and vegetation management.
- Northern Coastal California – There is a moderate loss of older forest on Federal and private lands due to wildfire and vegetation management.

(2) Scenario Two (Upper Bound):

- Central Coastal Oregon – There is a ban on trapping. Connectivity improves to the east through implementation of functional corridors across Highway 101, and there is no increase in the rate of loss of older forest above current trends as described in Scenario One. Proposed dune restoration at the Oregon Dunes National Recreation Area does not negatively impact suitable habitat.
- Southern Coastal Oregon – There is a ban on trapping. There is a 25% increase over the rate of older forest lost in Scenario 1 from a shorter fire return interval and drought, based on a moderate emissions scenario. The rate of timber harvest does not increase.
- California–Oregon Border – There is a 25% increase over the rate of older forest lost in Scenario 1 from a shorter fire return interval and drought, based on a moderate emissions scenario. The rate of timber harvest does not increase.
- Northern Coastal California – Assisted dispersal is feasible and is implemented to create one additional population and the Habitat Management Guide is implemented. There is a 25% increase over the rate of older forest lost in Scenario 1 from a shorter fire return

interval and drought, based on a moderate emissions scenario. The rate of timber harvest does not increase.

(3) Scenario Three (Lower Bound):

- Central Coastal Oregon – Trapping remains legal. Proposed dune restoration at the Oregon Dunes National Recreation Area negatively impacts the amount of suitable habitat. There is a 50% increase over the rate of older forest lost in Scenario One from a shorter fire return interval and drought, based on a high emissions scenario. There is no change in connectivity across Highway 101.
- Southern Coastal Oregon – Trapping remains legal. There is a 50% increase over the rate of older forest lost in Scenario One from a shorter fire return interval and drought, based on a high emissions scenario.
- California–Oregon Border – There is a 50% increase over the rate of older forest lost in Scenario One from a shorter fire return interval and drought, based on a high emissions scenario.
- Northern Coastal California – The assisted dispersal study is not feasible. There is a 25% increase over the rate of older forest lost in Scenario One from a shorter fire return interval and drought, based on a moderate emissions scenario.

We examined the resiliency, representation, and redundancy of coastal marten under each of these plausible scenarios (Table ES–1). Resiliency of coastal marten populations depends on connectivity, available suitable habitat, and habitat that allows for predator avoidance. We expect the four extant coastal marten populations to experience changes to these aspects of their habitat in different ways under the different scenarios. We projected the coastal marten's expected future resiliency, representation, and redundancy based on the events that would occur under each scenario (Table ES–2). For these projections, populations in high condition are expected to have high resiliency at that time period; i.e., they occupy habitat of sufficient size and in sufficient numbers to survive stochastic events. Populations in high condition are expected to persist into the future and have the ability to withstand stochastic events that may occur. Populations in moderate condition have less resiliency than those in high condition, but the majority of these populations are expected to persist into the future. Populations in moderation condition are smaller and less dense than those in high condition. Finally, the populations in low condition have low resiliency and are not necessarily able to withstand stochastic events. As a result, they would not be likely to persist in that condition into the future.

Under Scenario One (Continuation) – We would expect the coastal marten's viability to be characterized by declining resiliency, representation, and redundancy. The two Oregon populations would remain in low condition with the Central Coastal Oregon population at risk of extirpation. The two California populations would be in low–moderate condition. The coastal marten would be at risk of losing redundancy and representation of the population adapted to the dune ecosystem.

Under Scenario Two (Upper Bound) – We would expect the coastal marten's viability to be characterized by maintaining representation and redundancy with a slight improvement in resiliency. The two Oregon populations would remain in low condition with the Central Oregon population more resilient to stochastic events. The California–Oregon Border population would remain in low–moderate condition and the Northern Coastal California population would improve to moderate–high condition.

Under Scenario Three (Lower Bound) – We would expect the coastal marten's viability to be characterized by declining resiliency, representation, and redundancy. The two Oregon populations would remain in low condition with the Central Coastal Oregon population at risk of extirpation. The California–Oregon Border population would remain in low–moderate condition and the Northern Coastal California population would decline to low condition. The coastal marten would be at risk of losing redundancy and representation of the population adapted to the dune ecosystem.

Table ES–1. Coastal marten population resiliency under each scenario.

Population	Current Condition	Years into the future	Scenario 1 Upper Bound	Scenario 2 Continuation	Scenario 3 Lower Bound
Central Oregon	Low	15 30 60	Low Low Low – 0	Low Low Low	Low Low – 0 Low – 0
Southern Oregon	Low	15 30 60	Low Low Low	Low Low Low	Low Low Low
CA–OR Border	Low–Moderate	15 30 60	Low–Mod Low–Mod Low–Mod	Low–Mod Low–Mod Low–Mod	Low–Mod Low–Mod Low–Mod
Northern Coastal California	Moderate	15 30 60	Moderate Moderate Low–Mod	Moderate Mod–High Mod–High	Moderate Moderate Low–Mod

Table ES–2. Species Status Assessment summary for the coastal marten.

3Rs	Needs	Current Condition	Future Condition (Viability)

Resiliency: Population (large populations able to withstand stochastic events)	• Connectivity between populations
	• Available suitable habitat for home ranges and reproduction
	• Habitat that allows for predator avoidance.
		• Four populations; Two in Oregon and two in California
		• Population status:
		  o Two low resiliency (OR)
		  o One low–moderate (CA)
		  o One moderate resiliency (CA)
			Projections based on future scenarios:
			• Scenario One – Continuation: Threats continue on current trajectory. 1 population is at risk of being extirpated (OR). One other population experiences a drop in resiliency (CA).
			• See Table ES–1 for other Scenarios

Representation: (genetic and ecological diversity to maintain adaptive potential)	• Ecological representation exists between older, interior forests and coastal shore pine forests. • Genetic variation is unknown.	• Ecological representation is present for both the coastal shore pine forest and interior forest.	Projections based on future scenarios: • Scenario One – Continuation: Populations in the interior forest ecosystem are expected to persist, but the coastal shore pine population is at risk of extirpation. • See Table ES–1 for other Scenarios.
Redundancy: (number and distribution of populations to withstand catastrophic events)	• Multiple populations in each area of ecological representation.	• The coastal shore pine representation area has no redundancy: only one population exists in Central Coastal Oregon. • The interior forest has representation in three populations.	Projections based on future scenarios: • Scenario One – Continuation: The coastal shore pine population is at risk of extirpation; interior forest has one population in low condition and two populations in low–moderate condition. • See Table ES–1 for other Scenarios.

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Chapter 1 Overview

This report synthesizes the biology and status of the distinct population segment (DPS) of the Pacific marten (*Martes caurina*) in coastal Oregon and northern coastal California, commonly referred to as the coastal marten. In 2014, we published a notice in the Federal Register (79 FR 35509) that summarized the taxonomic classification of the subspecies (based on current genetic information) and indicated our intent to conduct an evaluation of a potential DPS of martens in coastal Oregon and coastal northern California relative to the full species classification level. In 2015, we published a DPS analysis (80 FR 18742) concluding that Pacific martens in coastal Oregon and northern coastal California were both discrete and significant and constituted a listable entity referred to collectively as the "coastal DPS of the Pacific marten." Throughout this document when we use the term "coastal marten," we are using this term as shorthand for the coastal DPS of the Pacific marten.

The coastal marten is a medium–sized carnivore that historically occurred throughout the coastal forests of northwestern California and Oregon. The coastal marten has been a candidate for listing under the Endangered Species Act of 1973, as amended (Act), since 2012 (66 FR 54808). The Species Status Assessment (SSA) framework (USFWS 2015a, entire) is intended to support an in–depth review of the species' biology and threats, an evaluation of its biological status, and an assessment of the resources and conditions needed to maintain long–term viability. The intent is for the SSA Report to be easily updated as new information becomes available and to support all functions of the Endangered Species Program from Candidate Assessment to Listing to Consultations to Recovery. As such, the SSA Report will be a living document upon which other documents, such as listing rules, recovery plans, and 5–year reviews, would be based if the species warrants listing under the Act.

This SSA Report for the coastal marten is intended to provide the biological support for the decision on whether or not to propose to list the species as threatened or endangered and, if so, where to propose designating critical habitat (Smith *et al.* 2018, entire). Importantly, the SSA Report does not result in a decision by the Service on whether this species should be proposed for listing as a threatened or endangered species under the Act. Instead, this SSA Report provides a review of the available information strictly related to the biological status of the coastal marten. The listing decision will be made by the Service after reviewing this document and all relevant laws, regulations, and policies, and the results of a proposed decision will be announced in the Federal Register, with appropriate opportunities for public input.

For the purpose of this assessment, we generally define viability as the ability of the coastal marten to sustain populations over time. Using the SSA framework (Figure 1.1), we consider what the species needs to maintain viability by characterizing the status of the species in terms of its resiliency, redundancy, and representation (Wolf *et al.* 2015, entire).

- **Resiliency** describes the ability of populations to withstand stochastic disturbance. Resiliency is positively related to population size and growth rate and may be influenced by connectivity among populations. Generally speaking, populations need abundant individuals within habitat patches of adequate area and quality to maintain survival and reproduction in spite of disturbance.

- **Representation** describes the ability of a species (or in this case a DPS) to adapt to changing environmental conditions. Representation can be measured by the breadth of genetic or environmental diversity within and among populations and gauges the probability that a species is capable of adapting to environmental changes. The more representation, or diversity, a species has, the more it is capable of adapting to changes (natural or human caused) in its environment. In the absence of species–specific genetic and ecological diversity information, we evaluate representation based on the extent and variability of habitat characteristics across the geographical range.

Species Status Assessment Framework

SPECIES' NEEDS

+ / − Current Availability or Condition of those Needs

CURRENT SPECIES' CONDITION

+ / − Future Availability or Condition of those Needs

FUTURE SPECIES' CONDITION
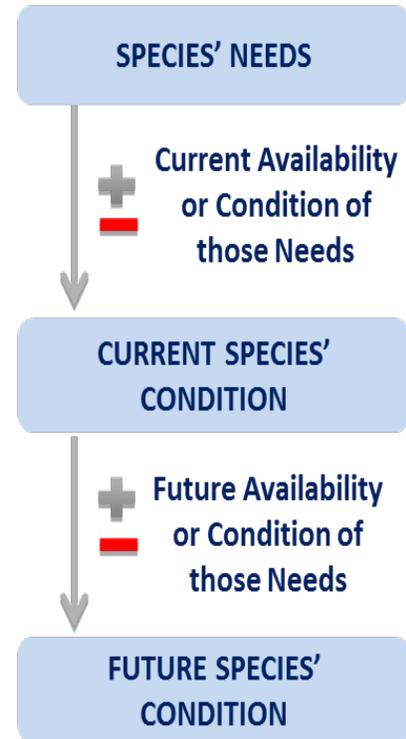
Figure 1.1 SSA Framework

- **Redundancy** describes the ability of a species (or DPS) to withstand catastrophic events; it is about spreading risk among multiple populations to minimize the potential loss of the species from catastrophic events. Redundancy is characterized by having multiple, resilient populations distributed within the species' ecological settings and across the species' range. It can be measured by number of populations, population resiliency, spatial extent, and degree of connectivity. Our analysis explores the influence of the number, distribution, and connectivity of populations on the species' ability to withstand catastrophic events (e.g., rescue effect).

To evaluate the biological status of the coastal marten both currently and into the future, we assessed a range of conditions to allow us to consider the species' resiliency, redundancy, and representation (together, the 3Rs). This SSA Report provides a thorough assessment of biology and natural history and assesses demographic risks, stressors, and limiting factors in the context of determining the viability and risks of extinction for the species.

The format for this SSA Report includes: (1) the resource needs of coastal marten individuals and populations (Chapter 2); (3) reviewing the likely causes of the current and future status of the species and determining which of these risk factors affect the species' viability and to what

degree (Chapter 3) (2) the coastal marten's historical distribution and a framework for determining the distribution of resilient populations across its range for species viability (Chapter 4); and (4) concluding with a description of the viability in terms of resiliency, redundancy, and representation across future scenarios (Chapter 5). This document is a compilation of the best scientific and commercial information available and a description of past, present, and likely future risk factors to the coastal marten.

Chapter 2 Species Needs

2.1 Introduction

This report reviews the biology and status of Pacific marten (*Martes caurina*) populations in coastal Oregon and northern coastal California, and the stressors that may be influencing those populations. Currently, the two extant Pacific marten populations in coastal Oregon are considered a separate subspecies (*M. c. caurina*) from the two extant populations in northern coastal California (*M. c. humboldtensis*; "Humboldt marten"). However, recent genetic research suggests that these four populations represent a single evolutionary unit (or "clade"), indicating a single taxon (refer to the TAXONOMY section below for details). Throughout this report, we refer to the combined populations of the Pacific marten occurring within coastal Oregon and northern coastal California as the "coastal marten." When supporting information only applies to one or more of the four extant coastal marten populations, or to martens that occur outside of coastal Oregon and California, it will be noted as such. We refer to the coastal marten populations in northwestern California as the Northern Coastal California population and the California–Oregon Border population. We refer to the two coastal Oregon populations by their geographic locations; the Central Coastal Oregon population and the Southern Coastal Oregon population (see Figure 3.2).

Research on the ecology of coastal populations of the Pacific marten began in 2000, with an emphasis on a single population found in northern coastal California. Beginning in 2014 systematic surveys and subsequent telemetry work were conducted focused on martens in the central coastal Oregon population. Thus, most of the available information on coastal martens is from these two populations. Regardless, we evaluated all available data, published and unpublished, for Pacific martens within coastal Oregon, and northern coastal California. Where information gaps exist, we rely on Pacific marten information from outside these coastal areas and use published information that appears to best characterize aspects of marten biology that are consistent across the range of the species. Unpublished data from outside these coastal areas are used only in cases where these data fill gaps in the published literature. Finally, biological information from research on two other North American mustelid species that have similar habitat ecologies, the American marten (*Martes americana*) and fisher (*Pekania pennanti*), was used to fill information gaps, when applicable to the coastal marten (*Martes caurina* sspp.).

The historical range and corresponding analysis area for our assessment of the coastal marten includes a 56,705–square–kilometer (km^2) (21,894–square–mile (mi^2)) area that extends from the coastal and inland areas of Oregon south to near Fort Ross in Sonoma County, California (Figure 2.1). We delineated the coastal marten analysis area using the generalized published historical extent of the distribution of the Pacific marten in coastal Oregon and northern coastal California (Grinnell and Dixon 1926, p. 415; Bailey 1936, p. 296; Grinnell *et al.* 1937, pp. 190, 207, 209; Zielinski and Golightly 1996, p. 115; Zielinski *et al.* 2001, p. 480; Slauson *et al.* 2019, p. 32),

broad–scale ecological criteria (Ricketts *et al.* 1999, entire), and natural biophysical features. The 36,348–km^2 (14,034 mi^2) Oregon portion of the analysis area includes all counties west of the Willamette Valley, the Coast Range portions of Lane and Douglas counties, and Coos, Curry, and Josephine counties in the south coast (Figure 2.1). The 20,747–km^2 (8,010 mi^2) California portion of the analysis area includes all or portions of Sonoma, Mendocino, Trinity, Humboldt, Siskiyou, and Del Norte counties (Figure 2.1). For further information on the range of the species see Section 3.1 Historical Range and Distribution (pre–1980).
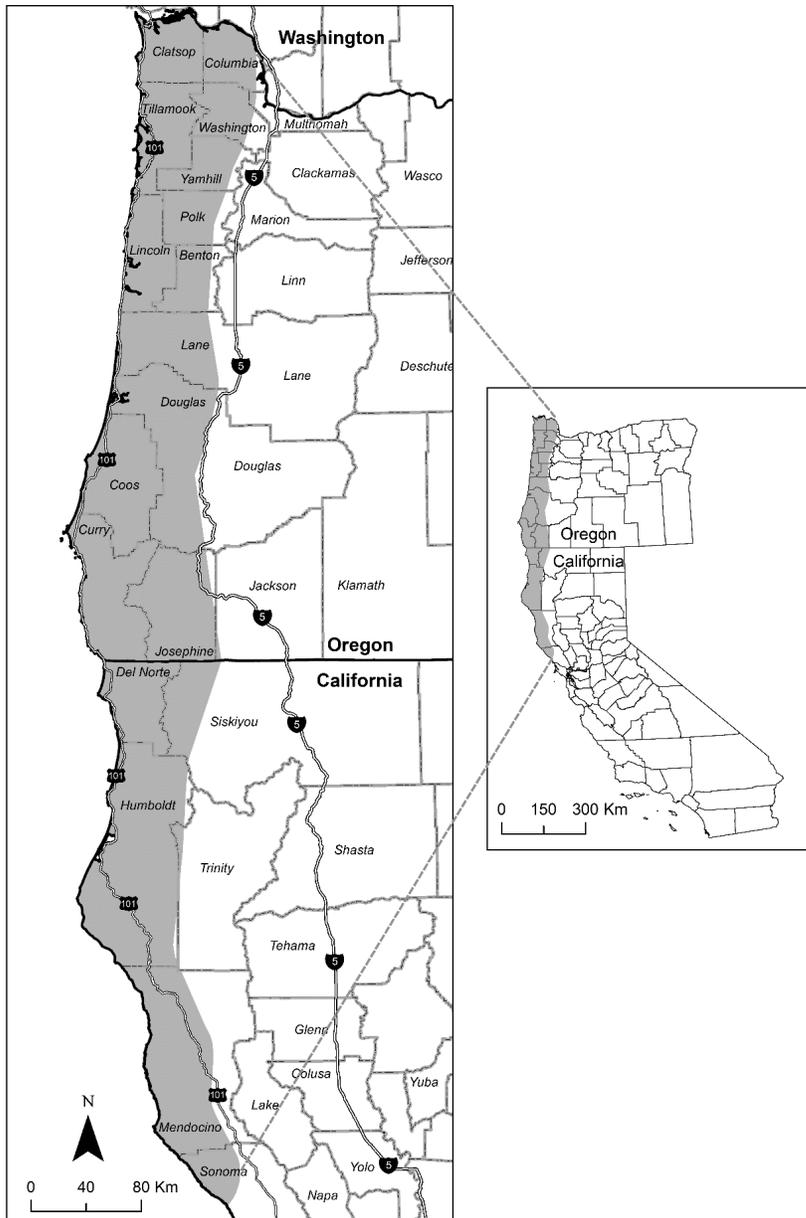
Figure 2.1. Analysis area (aka historical range) for coastal Oregon and northern coastal California populations of the Pacific marten (*Martes caurina*).

2.2 Taxonomy

The American marten (*Martes americana*) was historically recognized as a single species occurring across a broad range in North America, including the boreal forest region, montane coniferous forests, and Atlantic and Pacific coastal forested regions of the United States and Canada (Gibilisco 1994, entire). The Pacific marten (*M. caurina*) was recently split from the American marten based on genetic and morphological differences (Dawson and Cook 2012, entire; Dawson et al. 2017, entire). The Pacific marten occurs largely in montane and coastal coniferous forests west of the Rocky Mountain crest in North America, while the American marten occurs to the north and east of the Rocky mountain crest (Colella et al. 2021, Figure 1). The genetic split between these two species of martens is thought to have originated from the persistence of marten populations in two disjunct glacial refugia during the last glacial period (Dawson and Cook 2012, entire; Schwartz 2020, entire).

Throughout this report we adopt the new species–level nomenclature for the Pacific marten (i.e., *Martes caurina*), but maintain previously described (e.g., Hall 1981, pp. 981–985) subspecies epithets (*Martes americana humboldtensis* is now referred to as *Martes caurina humboldtensis*).

In Oregon, two subspecies of martens have been historically recognized, with *M. caurina caurina* occurring in the Coast Range and Cascades Mountains of central and western Oregon and *M. c. vulpina* occurring in the Blue Mountains of northeastern Oregon (Figure 2.2; Hall 1981, pp 983–985). Schwartz et al. (2020) eludes to three potential subspecies in Oregon including the coastal and cascades, as well as the Blue Mountains which align more closely with martens in Washington (pg, 10). Two subspecies of Pacific marten have been historically recognized in California. The Humboldt marten (*M. c. humboldtensis*) occurs along the northern coast, whereas the Sierra marten (*M. c. sierrae*), is found in the interior mountains of northwestern California, the Cascade Mountains in northern central California, and the Sierra Nevada Mountains of eastern California (Figure 2.2; Grinnell and Dixon 1926, entire; Grinnell *et al.* 1937, entire). Schwartz et al. (2020) eludes to two or three potential subspecies in California with martens between the coast and Sierra as a possible distinguishing group (pg 10). Both historical and contemporary genomic evaluations suggest the coastal martens are distinct and similar between Oregon and California.
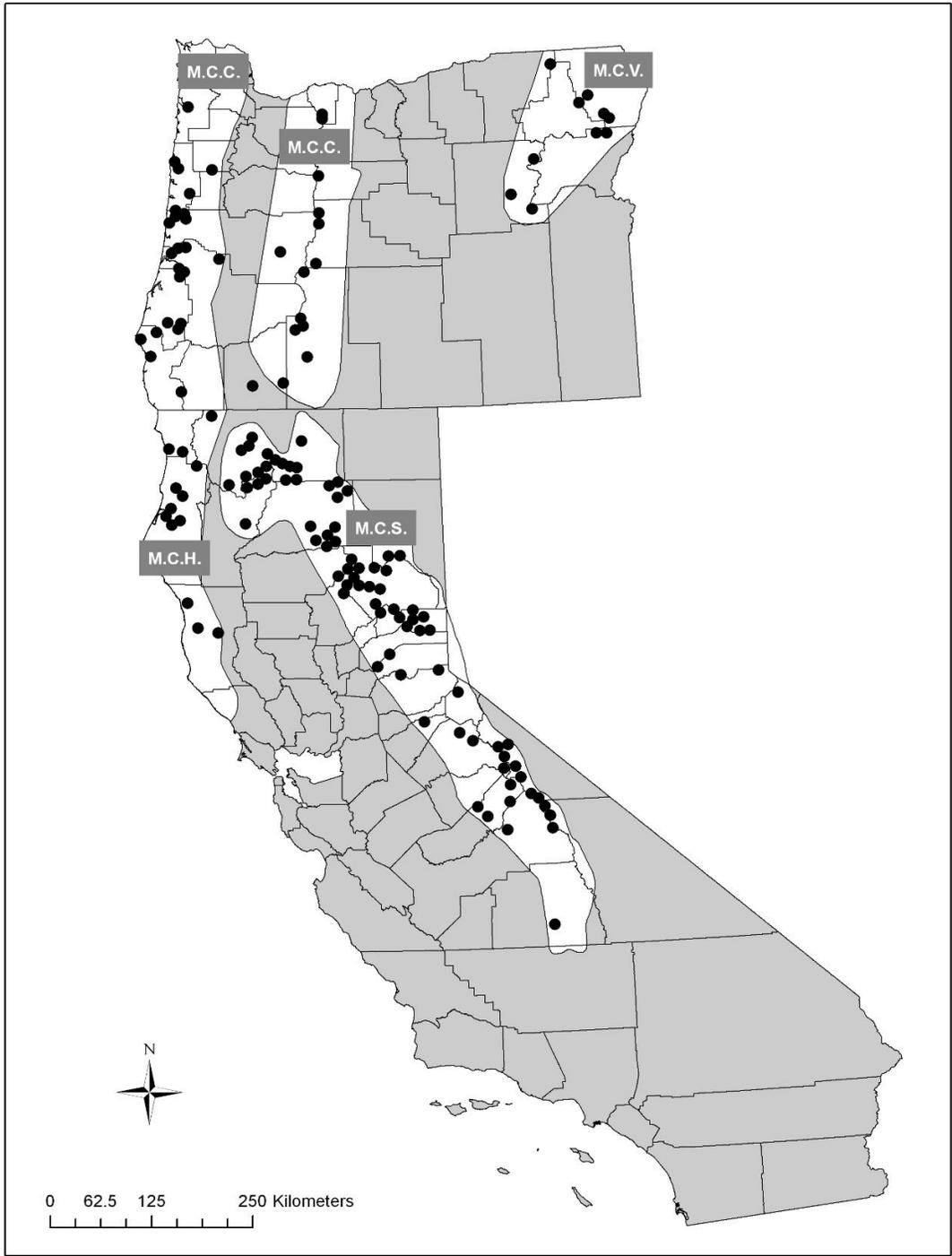
Figure 2.2. Historical range and distribution for the four Pacific marten subspecies occurring in Oregon and California. Range boundaries (white polygons) and historical records of occurrence (black circles) are modified from Zielinski *et al.* (2001, p. 480) which is derived largely from Hagmeier 1956 and Hall 1981. Subspecies: *M. c. humboldtensis* (M.C.H.), *M. c. sierra* (M.C.S.), *M. c. caurina* (M.C.C.), *M. c. vulpina* (M.C.V.). However, recent genetic data indicate that M.C.C. individuals in coastal Oregon are genetically more similar to the M.C.H. subspecies than to the M.C.C. individuals in the Oregon Cascades (Schwartz *et al*. 2020, entire).

The Humboldt marten was historically distributed throughout the coastal coniferous forests of northern California from northwestern Sonoma County northward to Tillamook Bay in coastal Oregon (Grinnell *et al.* 1937, pp. 207–210, Zielinski et al, 2001, pg. 480). Recent phylogenetic analyses using mitochondrial DNA (mtDNA) support the distinctiveness of the Humboldt marten subspecies, based on the presence of distinct haplotypes shared by historical museum specimens and martens currently occupying portions of the historical range in northern coastal California (Slauson *et al.* 2009a, entire; Schwartz et al. 2020, entire). Marten populations in coastal Oregon, which were historically described as *M. c. caurina*, also share these haplotypes, leading Slauson *et al.* (2009a, pp. 1338–1339) to suggest that martens in the Coast Range of Oregon were also *M. c. humboldtensis.* This assumption was validated in 2020 with nuclear DNA (nDNA), revealing that coastal Oregon and northern coastal California marten populations represent a single evolutionary clade, calling into question the separation of the original subspecies range boundaries (i.e., *M. c. humboldtensis* in northern coastal California and *M. c. caurina* in coastal Oregon) at the California–Oregon border (Schwartz et al. 2020, entire). This report synthesizes the biology and status of the distinct population segment (DPS) of the Pacific marten (*Martes caurina*) in coastal Oregon and northern coastal California, commonly referred to as the coastal marten.

2.3 Morphological Description

The current science supports two species of marten, divided into 14 subspecies, inhabiting North America (Colella et al. 2021 entire; Dawson et al. 2017, entire; Lucid et al. 2020). Collectively, North American martens are characterized by the long and narrow body type typical of the mustelid family (Mustelidae; e.g., weasels, minks, otters and fishers): overall brown fur with distinctive coloration on the throat and upper chest that varies from orange to yellow to cream, large and distinctly triangular ears, and a bushy tail that is proportionally equivalent to about 75 percent of the head and body length (Clark *et al.* 1987, p. 2; Powell *et al.* 2003, p. 636). The coastal marten shares most of these physical characteristics with other subspecies of North American martens, but may occasionally differ slightly in pelage color and the size and coloration of the throat patch (Figure 2.3). Compared to the Sierra subspecies (*M. c. sierrae*) of the Pacific marten in northern interior California and the Sierra Nevada Mountains, the coastal marten subspecies is sometimes darker, with a richer golden tone to the under fur, a patchy and smaller throat patch that is more cream colored than orange and yellow (Figure 2.3). The Humboldt subspecies may also exhibit a smaller skull, smaller and less crowded premolars, and a narrower rostrum (snout) than the Sierra subspecies (Grinnell and Dixon 1926, p. 411).
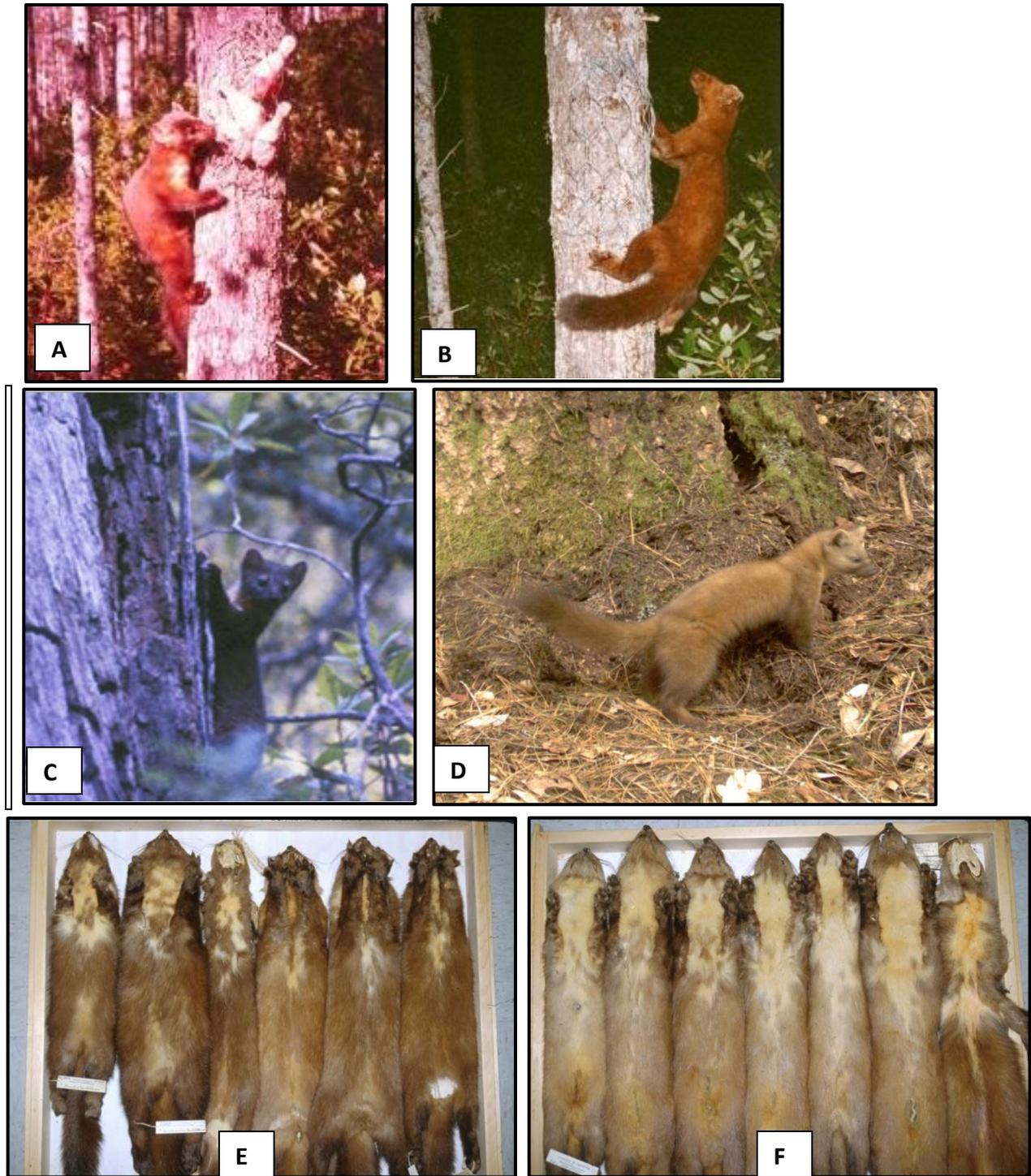
Figure 2.3. Coastal marten photographs from southern coastal Oregon (A and B) and northern coastal California (C and D). Bottom row: comparison of overall pelage coloration and extent of the gular patch on typical historical museum specimens of the coastal marten from northern coastal California (E) and the Sierra marten (F). Specimens are from the Museum of Vertebrate Zoology, University of California–Berkeley, Berkeley, CA, USA.

Grinnell and Dixon (1926, p. 411) also compared pelage coloration and skull and tooth morphology between *M. c. humboldtensis* in northwestern California and *M. c. caurina*; however, the *M. c. caurina* specimen used for the comparison was from central coastal Washington, not coastal Oregon or the Oregon Cascades. Recent genetic analyses suggest that martens from coastal Washington (currently *M. c. caurina*) are genetically distinct from martens in coastal Oregon (also currently *M. c. caurina*) and martens in northern coastal California (currently *M. c. humboldtensis*; Schwartz *et al.* 2020, entire). Comparisons of pelage characteristics and morphological measurements between coastal martens in northern coastal California and coastal Oregon between the Oregon Cascades have not been made to date.

2.4 Life History

2.4.1 Reproduction

North American martens are polygamous, with females solely responsible for raising young. Information on timing of marten mating is largely known from the behavior of captive animals, but is believed to occur from late June to early August, with a peak in July (Markley and Bassett 1942, pp. 606–607). Females typically give birth in March and April (Strickland *et al.* 1982, p. 602) (Figure 2.4). Parturition dates for coastal marten in northern California ranged between March 31 – April 21 with a mean date of April 13 and standard deviation of 7.2 days (Delheimer et al. 2021). Females do not mate until 15 months of age and, due to delayed implantation, will not produce their first litters until they are at least 24 months old (Strickland *et al.* 1982, p. 601). However, not all yearling females produce ova. Thompson and Colgan (1987, p. 831) reported less than 25 percent of yearlings produced ova, and Fortin and Cantin (2004, pp. 228–229) reported a range of 44–76 percent ovulation rate (females >1.5 years) for greater than 183 American martens over a decade.

Not all females of reproductive age give birth in any given year. In Ontario, Thompson and Colgan (1987, p. 831) reported a 50 percent pregnancy rate during years of environmental stress. Of 22 monitored adult Pacific marten females, 5 did not attempt to reproduce (Delheimer et al. 2021). Of coastal martens, the proportion of adult females reproducing was 80% (12 of 15) with age of parturition ranging between two and five (Delheimer et al. 2021). With this limited data available for coastal martens, the observed mean litter size was 1.8 (0.6) kits, which is on the lower range for martens in North America (Delheimer et al. 2021, Table 2).
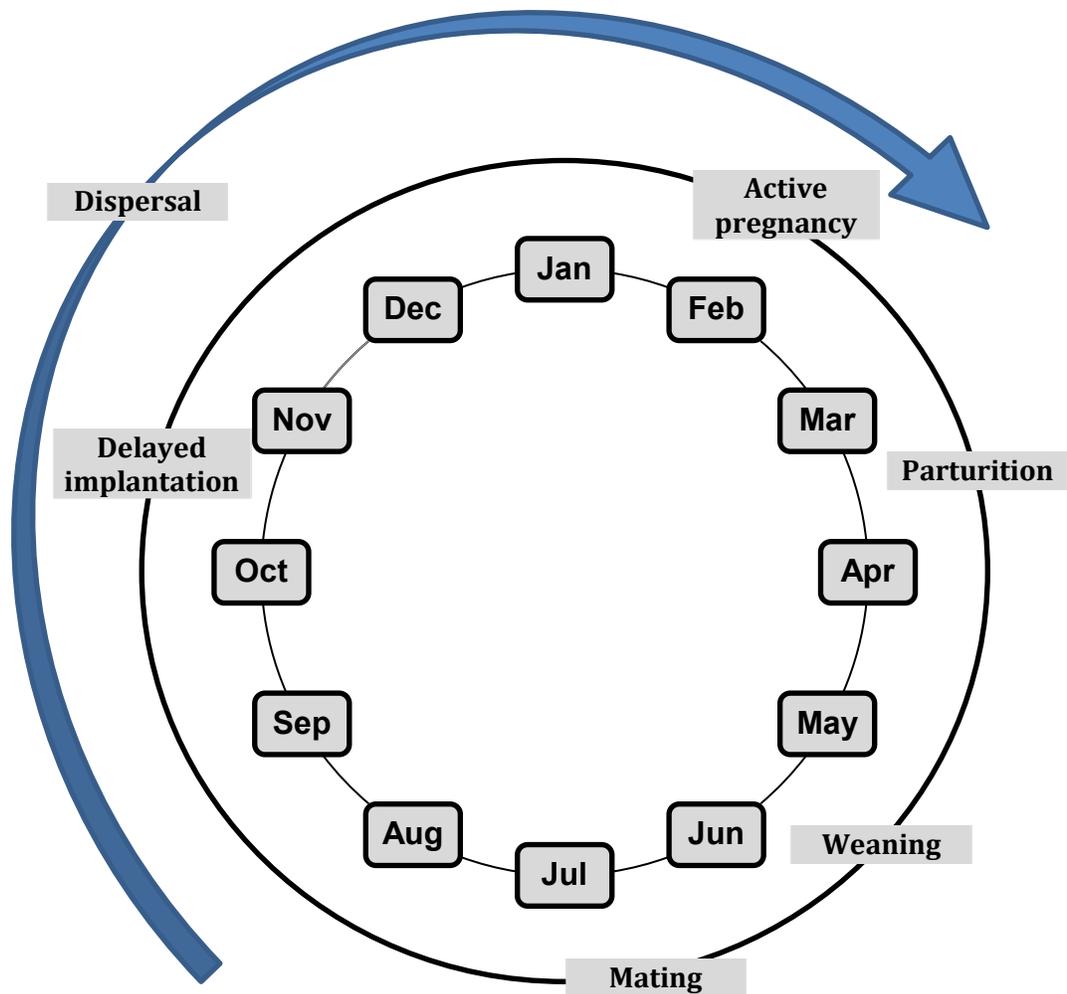
Figure 2.4 Annual reproductive cycle for North American martens (from Slauson et al. 2019, p. 38).

2.4.2 Survivorship and Age Structure

Minimal information is available on marten longevity and age structure. Longevity in North American martens is not well understood because most studies of longevity are from trapped populations where the age structure is truncated and captures are biased towards younger individuals. Few studies of un–trapped populations report age structure. Captive martens are known to reach 15 years of age (Clark *et al.* 1987, p. 3). In natural settings the maximum age is most likely less. Age data on over 1,200 marten carcasses from Oregon during 1986–2015 (342 females, 866 males) shows that four percent of the animals were 6 years or older (18 females and 35 males with maximum ages of 12 and 10 years, respectively) (Broman 2018, unpublished data). Much of these data were from harvested animals, which is likely responsible for the larger proportion of males in the sample. Similar age proportions occurred in a live–trapping research study in northeastern Oregon, where all 34 trapped martens were less than 5 years old (Bull and Heater 2001a, p. 4). Over a 3–year period in California, age structure in an un–trapped

population of 96 Pacific martens in the Sierra Nevada Mountains maintained relatively consistent proportions of yearling and adult age classes, with only a single female observed that was greater than 4 years of age (Slauson 2017). However, of martens captured in the northern Sierras, 10 percent were over 5 years of age, including a male and female that were each 7 years old (Moriarty 2018, pers. comm.). Elsewhere in North America, proportions of older females are somewhat larger; for example, in the Algonquin Region of Ontario, Canada, of 2,076 females trapped, only approximately 10 percent were more than 5 years old (Strickland and Douglas 1987, p. 535).Therefore, the best available data suggest that in the wild, most North American martens, and presumably coastal martens, live less than 5 years. In light of delayed implantation, a small proportion of female martens, perhaps 10 percent at best, are reproducing for more than three years, contributing to a slow reproductive output.

Annual recruitment rates of juvenile cohorts (i.e., the proportion of juveniles that survive to adulthood and become part of a population by establishing a home range) in un–trapped marten populations may depend on three factors: (1) the number of young produced per female that survive to the dispersal stage; (2) the rate of success of dispersing juveniles establishing home ranges; and (3) the effects of annual environmental stochasticity on recruitment (Slauson *et al.* 2019, p. 42). Over a 3–year period in an un–trapped population of Pacific martens in the Sierra Nevada Mountains of California, annual juvenile recruitment represented approximately 40 percent of the total observed population size and was relatively constant over the 3–year period (Slauson 2017, p. 124).

2.4.3 Dispersal

Juvenile dispersal of American martens is generally thought to occur as early as August, although fall, winter, and spring (the year after birth) dispersal periods have been reported (Clark and Campbell 1976, p. 294; Slough 1989, p. 993; Bull and Heater 2001b, p. 9; Johnson 2008, p. 32). Juvenile dispersal in Humboldt martens has been observed to occur as early as August and continue at least until the following summer season (Slauson and Zielinski, unpublished data) (Figure 2.4). No information is available regarding the timing of juvenile dispersal for coastal martens in Oregon, although there is no reason to believe it would be different than that of coastal martens in northern coastal California.

Dispersal has been divided into three phases to understand the decisions faced by juveniles or adults and the potential factors influencing those decisions: (1) searching for new habitat (search); (2) settling in new habitat (settlement); and (3) living in new habitat (residency) (Stamps 2001, entire). Linking the social or habitat–related influences on each phase of the dispersal process is critical to understanding how these factors influence the outcome of dispersal events (Bowler and Benton 2005, entire). Landscape heterogeneity and likelihood of mortality are factors that influence the likelihood of longer-distance dispersal events in martens (Day et al. 2020, entire), with martens making more risky decisions the longer they are dispersing (Day et al. 2019, entire).

While some adult male and female martens leave their home ranges during periods of low prey densities (Thompson and Colgan 1987, pp. 830–831), overall the prevalence of adults leaving their established home ranges is low. Search behavior and the distance juvenile and adult coastal martens can travel are critical to understanding how landscape pattern can affect dispersal (Day et al. 2019, entire). While dispersal distances of more than 70 km (43 mi) have been reported for martens (e.g., Fecske and Jenks 2002, p. 310), this is rare and most studies find that the majority of juvenile martens dispersed <15 km (9.3 mi) (<15 km in Maine by Phillips 1994, pp. 73–75; <5 km in Ontario by Broquet *et al.* 2006, p. 1694; 15.5 km in Alaska and British Columbia by Pauli *et al.* 2012, p. 393; 10.8 km in California by Slauson 2017, p. 143). If you ignore the longest distance (70 km) the average of averages is 7.4 km. An average daily movement for coastal martens in the Oregon dunes was 7.1 km (SD = 5.9 km, marten days = 128, individuals = 6: Moriarty et al. 2017, entire). The longest daily movement of a Pacific marten was 27.2 km (Moriarty et al. 2016). In novel environments, translocated pine martens (*Martes* martes) moved an average of 8.7 km, but up to 103 km from release sites (McNicol et al. 2020). The limited data we have for dispersal events of coastal marten suggest that dispersal distances are similar (K. Slauson 2018, pers. comm.). Landscape conditions greatly influence dispersal (Day et al. 2020, entire). Juvenile martens in logged versus unlogged landscapes in Canada traveled slower, moved shorter distances, and suffered twice the mortality risk (Johnson 2008, pp. 33–36). Another study from Canada demonstrated that the unlogged landscape offered increased foraging efficiency (Andruskiw *et al.* 2008, pp. 2275–2277), presumably resulting in improved physical condition and thus facilitating longer dispersal distances and twice the success rate (25 percent in logged versus 49 percent in unlogged landscapes) of surviving to adulthood (Johnson *et al.* 2009, p. 3365).

Therefore, the best available information suggests that landscape condition (e.g., the spatial distribution of unlogged and logged stands) may have important effects on dispersal dynamics, affecting both the distance dispersers can travel and the success rate they have in establishing home ranges and surviving to adulthood.

2.5 Marten life history needs

This section describes the resources and circumstances needed for individual coastal martens to complete each life stage. Martens have three life stages: they are considered kits from birth until ~6 months of age when they begin dispersing from their natal habitat, juveniles from ~6 months until ~2 years of age when they begin reproducing, and adults after they become reproductive (Strickland *et al.* 1982, p. 601). The needs for each stage are discussed below and summarized in Table 2.1 at the end of this Section. Section 2.5.1 addresses needs of kits and all following Sections pertain to juveniles and adults.

2.5.1 Kit resource needs

Coastal marten kits need the female adult marten as a thermal source and to provide lactation early in development. They also rely on their mother to provide prey and protection from predators. There is no paternal involvement in raising kits (Markley and Bassett 1942, entire). Kits are born in a den which can be any enclosed area that provides shelter from weather and predators. This is most often a large diameter tree (either live or dead) with cavities, but can also include hollow logs, crevices under rocks or roots, log piles, and squirrel nests (Schumacher 1999, pp. 26–28; Slauson and Zielinski 2009, pp. 41–42; Moriarty et al. 2017a, pp. 82–87).

2.5.2 Food

North American martens are considered dietary generalists; however, as mentioned above, their diet changes with seasonal prey availability, and during particular seasons they may become specialists on a few prey species (Zielinski *et al.* 1983, pp. 389–392; Martin 1994, pp. 303–304; Slauson and Zielinski 2017, pp. 1701–1703). North American martens require 15–25% of their body mass in prey daily to meet their metabolic requirements (Gilbert *et al.* 2009; entire; Martin et al. 2020, entire).

Overall, the diet of North American marten species is dominated by mammals, but birds, insects, and fruits are seasonally important (Martin 1994, pp. 298–301). Diet analysis for the coastal marten is currently limited to scats collected from the northern coastal California population from July to November and a recently conducted analysis in central coastal Oregon encompassing three seasons. In a diet analysis of 420 coastal marten scats in the northern coastal California population from July to November, mammals occurred in 93 percent of the scats, berries in 85 percent, birds in 21 percent, insects in 20 percent, and reptiles in 7 percent (Slauson and Zielinski, 2017, entire). Sciurid (squirrels) and cricetid (New World rats and mice) rodents dominated the diet, with the other most frequent prey species being chipmunks (*Tamias* sp.) and red–backed voles (*Myodes californicus*). Other mammalian prey species may include Douglas's squirrels (*Tamiasciurus douglasii*), flying squirrels (*Glaucomys* sp.), red tree voles (*Arborimus longicaudus*), white–footed voles (*Arborimus albipes*), pocket gophers (*Thomomys* sp.), and moles (*Scapanus* sp.). The relative biomass contributed by chipmunks, red–backed voles, Douglas's squirrels, and flying squirrels represented 85 percent of the total mammalian biomass consumed by coastal martens in the summer–fall period (Slauson and Zielinski, 2017, entire). The frequency of berries in summer and fall scats from northern coastal California martens was higher than previously reported in other North American marten subspecies (Martin 1994, p. 298) and was dominated by *Gaultheria shallon* (salal; 33 percent), *Vaccinium ovatum* (evergreen huckleberry; 26 percent), and *Vaccinium parviflora* (California red huckleberry; 23 percent) (Slauson and Zielinski, 2017, entire).

In an analysis of 90 scats across three seasons in the central coastal Oregon populations, martens fed largely on mammals (predominantly voles), birds and berries (contained in 80, 53, and 63 percent of scats respectively). This is in contrast to some California populations of martens that relied more heavily on birds (53 percent vs. 10–20 percent) and on chipmunks (which constituted

<2 percent of scats in Oregon). A large percentage of winter scats contained berries in fall and winter (100 and 86 percent respectively) and mammals were eaten all three seasons with voles being the most common (Eriksson et al, 2019, entire). In general, coastal martens consumed more large passerines and rabbits compared to montane counterparts (Erikkson et al. 2019, entire). In addition, coastal martens consumed more arboreal (red tree and white footed) voles (*Arborimus* sp.) compared to montane martens (Moriarty et al. 2021a, entire).

The availability of prey species to the coastal marten varies seasonally (Slauson and Zielinski 2017, pp. 1703, 1706). Due to the lack of significant snowpack in areas where coastal martens occur, prey are probably more available to coastal martens during the winter than for other North American martens (Slauson *et al.* 2019, p. 36). An important winter prey species for most North American martens, the snowshoe hare, occurs only within the central and northern coastal Oregon portions of the historical range of the coastal marten (Verts and Carraway 1998, p. 142). Therefore, other typical winter prey species, such as resident fruit–eating birds, Douglas's and flying squirrels, and species otherwise unavailable where snow cover occurs, likely play a more important role range wide in the winter diet of the coastal marten.

Many of the key prey species of the coastal marten reach their highest densities in forest stands with structural features characteristic of older forests where the key food resources of those prey species—conifer seed crops and fruiting bodies of ectomycorrhizal fungi—reach their greatest abundances (e.g., Hayes and Cross 1987, p. 543; Carey 1991, entire; Rosenberg *et al.* 1994, pp. 267–268; Carey and Johnson 1995, pp. 340–343; Waters and Zabel 1995, pp. 861–863; Smith *et al.* 2002, pp. 190–197; Luoma *et al.* 2003, pp. 346–347). The density of ericaceous (members of the plant family Ericaceae or heather family) shrub layers has also been shown to be positively correlated with chipmunk density in coastal Oregon and prey diversity and abundance in coastal dune forests in Oregon (Hayes *et al.* 1995, pp. 69–70; Eriksson *et al.,* 2019, entire). Complex physical structure on or near the forest floor is typically provided by dense coarse woody debris and is directly related to predation success for martens (Thompson *et al.* 2012, pp. 212–221). It has been shown for American martens that when this complexity is reduced, predation success declines due in part to the increased vigilance prey exhibit when in the more open habitat (Andruskiw *et al.* 2008, pp. 2275–2277).

2.5.3 Habitat Use

There is no single description of coastal marten habitat and it likely differs within and between each population to an extent. North American martens, including coastal martens are associated with forests with old forest attributes (e.g., multi-layer canopy, large trees, snags, or logs). Coastal martens are found at the highest densities in the coastal dune forests (Linnell et al. 2018, entire). Similar in structure, coastal martens have been located consistently in areas with dense shrubs facilitated by natural openings within the clades of 'serpentine soils' (e.g., magnesic, parasquic, serpentinite types) (Slauson et al. 2007, Gamblin et al. 2019, and Moriarty et al. 2019).

A wide variety of terms have been used to describe older forest stands the martens tend to select when available (e.g., late–successional, old–growth, large conifer, mature, late–seral, structurally complex). Where these terms appear in cited literature, or where specific ages are referred to, we refer to them in this analysis. Otherwise, we use the term "older forest" when collectively referring to these stand conditions. In using the term "older forest" we are not implying a specific stand age that represents coastal marten habitat. Rather, we use the term to represent the mixture of old and large trees, multiple canopy layers, snags and other decay elements, dense understory development, and biologically complex structure and composition often found in forests selected by coastal marten.

Studies drawn from both Pacific and American martens demonstrate that marten select habitat at four primary spatial scales: micro–habitat, stand, home range, and landscape scales. At the micro–habitat scale, martens select specific structures to use for foraging or resting, such as large logs, which they run along searching for prey, or cavities in snags that provide thermal benefits (Taylor 1993, pp. 1–8) and reduce predation risk while resting. At the stand–scale, martens select stands with adequate structural features that provide for one or more life–history requirements (e.g., prey populations, foraging structures, and resting structures) (Slauson *et al.* 2007, pp. 464–465). At the home–range scale, martens position their home ranges to include enough habitat of sufficient quality to provide for year round life history needs (e.g., seasonal prey bases, den sites) and access to mates, while avoiding overlap of home ranges with same–sex individuals (Katnik *et al.* 1994, pp. 604–606; Powell 1994, entire). At the landscape–scale, dispersing individuals select suitable portions of the landscape that do not overlap but that are close enough to allow for metapopulation structure (Johnson 2008, pp. 14–16).

Habitat selection of North American martens appeared to be strongest at the landscape scale (Minta 1999, entire), with a consistent pattern of declines in marten occupancy when 25–30 percent of a forested landscape is regenerating following timber harvest (Thompson *et al.* 2012, p. 217). Although North American martens generally prefer older forests over young regenerating stands, marten habitat use varies across the continent, making generalities difficult to infer at the stand–scale (Thompson *et al.* 2012, entire). The variation in the preference for older forests within home ranges appears to be linked to key aspects of Pacific and American marten ecology, including the abundance of primary prey species, predator distribution, and the development of structural complexity near the ground in different forest types (Thompson *et al.* 2012, pp. 212–221). Consequently, assumptions about stand–scale habitat relationships should be ecosystem specific and not based on general understandings from throughout their ranges.

2.5.3.1   *Micro–scale Habitat: Resting and Denning Habitat*
Resting Habitat: Large–diameter live trees with large horizontal limbs, standing snags with cavities or chambers, and downed hollow logs provide the main types of resting structures for martens in California and Southern Oregon (Spencer *et al.* 1983, pp. 1182–1185; Slauson and Zielinski 2009, pp. 41–42; Moriarty et al. 2017a, pp. 73–78; Tweedy 2018, pp. 22–23). Coastal martens that occupy shore pine forest habitat in Oregon use rest structures that include squirrel

nests in trees (most commonly), bare branches, and basal hollows from old overturned trees (Moriarty 2018, pers. comm.). In coastal California, of 55 rest structures used by coastal martens in the summer and fall, 37 percent were snags, 23 percent downed logs, and 17 percent live trees (Figure 2.5; Slauson and Zielinski 2009, pp. 39–42). Rest structures are used daily by martens between foraging bouts to provide thermoregulatory benefits and protection from predators (Taylor and Buskirk 1994, pp. 253–255). Rest structures used by coastal martens in California averaged 95 cm (37 in.) diameter–at–breast–height (dbh) for snags, 88 cm (35 in.) maximum diameter for downed logs, and 94 cm (37 in.) dbh for live trees (Slauson and Zielinski 2009, p. 40). These woody structures were found in the oldest forest development stages (Slauson and Zielinski 2009, p. 40). Most resting locations—the actual place in the structure the marten used for resting—occurred in tree cavities (33 percent), on platforms (33 percent) created by broken top snags or large live branches, or in chambers (28 percent) created by log piles or rock outcrops (Figure 2.5; Slauson and Zielinski 2009, p. 39). In coastal Oregon and northern coastal California, rest structures providing cavities or chambers likely become seasonally important during the rainy period of the year: late fall through late spring.
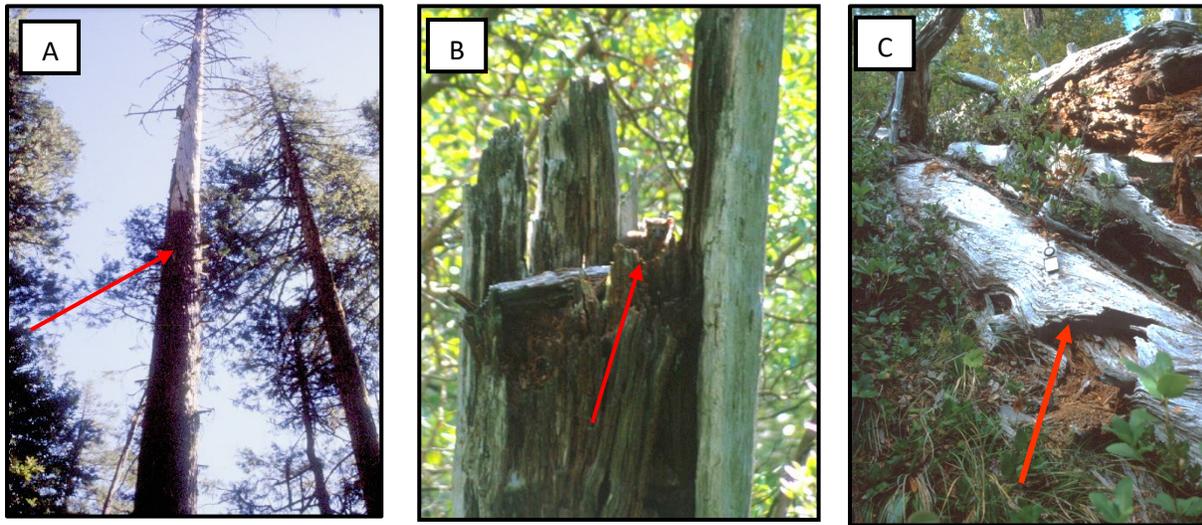
Figure 2.5. Examples of coastal marten resting structures in northern coastal California: **A**. Cavity in Douglas–fir snag. **B**. Platform on broken Douglas–fir snag. **C**. Cavity in downed Douglas–fir log. Red arrows identify the resting location in each structure.

Historically, reuse rates (individuals returning to the same structure more than once) for individual rest structures were considered low, however Sierra martens have shown reuse rates of 79 percent (Moriarty et al. 2017a, p. 95). This is important when considering the elements that martens need for suitable habitat. Selection for rest structure type changes seasonally to meet thermoregulatory needs (e.g., Spencer 1987); rest structure type may also change to accommodate young martens, as evidenced by female martens with kits choosing ground–based rest sites (stumps, logs, and rock piles) more than half the time (Moriarty et al. 2017a, p. 5).

These studies indicate that a diversity of resting structure types are needed to meet seasonal requirements across the home range. Martens typically select the largest available structures for resting and denning (Spencer *et al.* 1983, pp. 1182–1185; Gilbert *et al.* 1997, pp. 138–142).

Denning Habitat: The denning habitat which is used by female martens to give birth to kits are called natal dens, and the subsequent locations where they move their kits are referred to as maternal dens (Buskirk and Ruggiero 1994, p. 17). The most common den structures used by Pacific and American martens are large diameter live and dead trees with cavities (Thompson *et al.* 2012, p. 223). Of over 35 coastal marten den sites found in northwestern California, most were in cavities of live and dead trees and all were in larger–diameter trees compared to the surrounding stand. The snags and live trees had structural features such as complex crowns, large limbs, broken tops, hollow bases, or multiple cavities (Slauson and Zielinski 2009, p. 40; Delheimer et al. 2021, entire). Dens monitored on privately managed forests (33 den sites) were located in a variety of stand ages (Delheimer et al. 2021, entire). In this same study, 50 percent of reproductive females monitored in multiple breeding seasons reused a den from a previous season, whereas 60 percent reused a site within the same season (Delheimer et al. 2021).

Pacific and American martens appear to be more selective of habitat conditions at den sites than at resting sites (Thompson *et al.* 2012, p. 223). Ruggiero *et al.* (1998, pp. 665–669) found that both the characteristics of the den structures and the characteristics of the stands they were found in influenced den–site selection. This is likely due to the importance of suitable foraging habitat in close proximity to den sites, allowing females to simultaneously maximize the energy they gain from foraging during lactation and minimize the time spent away from kits, especially when they are dependent on their mothers for thermoregulation.

2.5.3.2   *Stand Scale Habitat Use*

Martens select forest stands that provide habitat structure supporting one or more life history needs that include foraging, resting, or denning. In addition, stands that provide sufficient structure to reduce the risk of predation, such as dense overhead vegetation and vertical tree boles, are also important and affect marten survival. We discuss the three habitat types coastal martens are detected in below.

Older Forest Habitat: Coastal martens in California most strongly select stands of older, conifer–dominated forests with dense, ericaceous shrub layers and an abundance of large downed logs, and large, decadent live trees and snags (Figure 2.6; Buskirk and Ruggerio 1994, pp. 21–22; Slauson *et al.* 2007, pp. 464–465). Other than the older forests, which are used in proportion to their availability, stands in earlier developmental stages are selected against (Slauson *et al.* 2007, pp. 462–464). These older forests occur on areas of highly productive soils that are most often dominated by Douglas–fir overstories, but also have mature hardwood understories composed of either *Notholithocarpus densiflorus* (tanoak) or golden chinquapin. Shrub layers were generally dense (greater than 70 percent cover), spatially extensive, and dominated by evergreen huckleberry, salal, and *Rhododendron* sp. (rhododendron) (Figure 2.6; Slauson *et al.* 2007, p.

465). The majority of stands with detections of coastal martens in southern coastal Oregon share these same characteristics (Zielinski *et al.* 2001, p. 485).

Similarly in Oregon, coastal martens are strongly associated with areas of expansive and dense shrub cover comprised of primarily salal and evergreen huckleberry with ranges reported from 40->75% shrub cover (Eriksson et al. 2019, entire; Moriarty et al. 2019b, entire). Dominant overstory on non–serpentine soils includes *Picea sitchensis* (Sitka spruce), *Tsuga heterophylla* (western hemlock), and Douglas–fir. Marten sites were also in close proximity to large snags and logs. While martens used older forests, they may be found in forests with smaller diameter trees as long as combined overstory and understory cover remained high (Moriarty et al., 2019, entire).

Marten also select habitat based on prey availability. Many of the key prey species for coastal marten such as red–backed vole, flying squirrel, Douglas's squirrel, and chipmunks exhibit their highest densities in older forest stands that have dense shrub layers (Hayes and Cross 1987, pp. 544–545; Carey 1991, pp. 14–15; Hayes *et al.* 1995, p. 69; Waters and Zabel 1995, pp. 861–862).
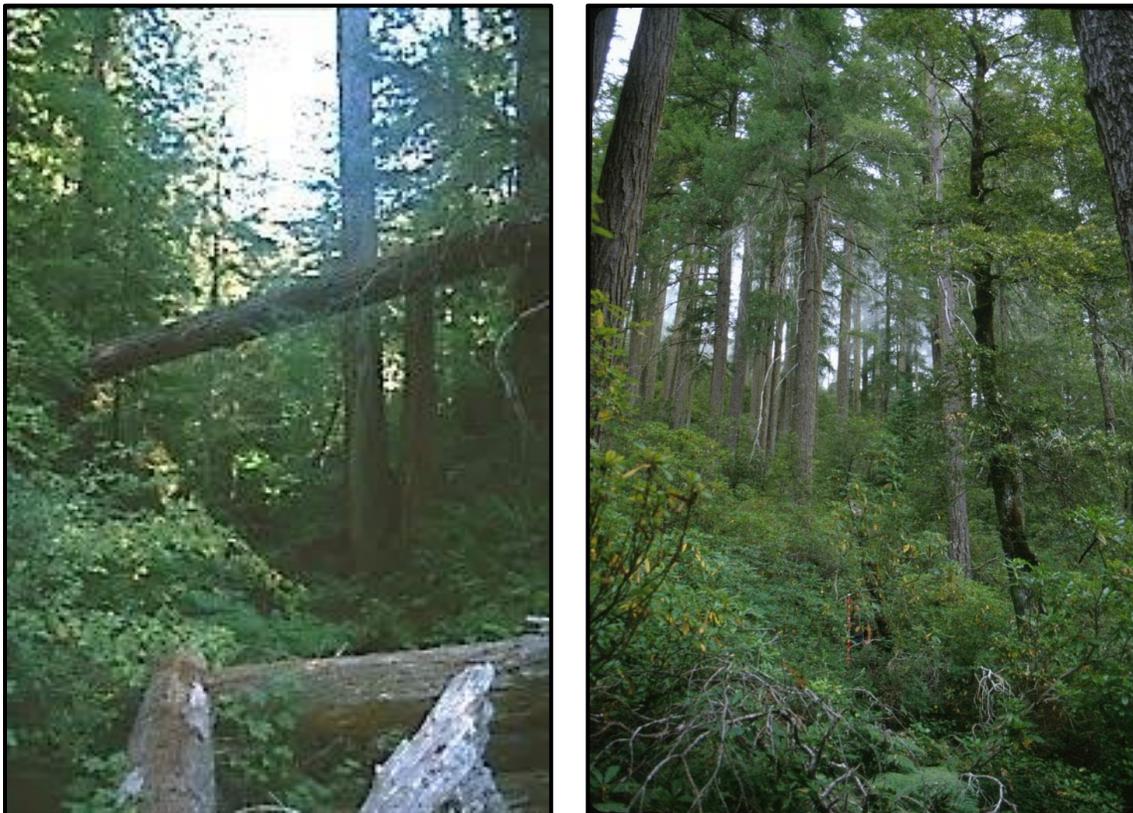
Figure 2.6. Habitat structure within typical coastal marten occupied stands in northern coastal California. Note conifer dominated overstory, dense shrub layers and presence of large diameter dead standing and downed wood.

Serpentine Soil and Vegetation: In addition to older forests on highly productive soils, coastal martens have used forest and shrub dominated habitats occurring on less productive serpentine soils, hereafter called serpentine habitats. These serpentine habitats include areas with conifer–dominated tree overstories, with dominants including *Pinus contorta* ssp. *murrayana* (lodgepole pine), *Pinus monticola* (western white pine), and Douglas–fir, but also including dense (greater than 70 percent cover) shrub layers dominated by *Notholithocarpus densiflorus* (tan oak), *Quercus vacciniifolia* (huckleberry oak), *Notholithocarpus densiflorus* var. *echinoides* (dwarf tanbark), and *Vaccinium parvifolium* (California red huckleberry) (Figure 2.7; Jimerson et al. 1996, pp. A–8 to A–12; Zielinski *et al.* 2001, p. 485; Slauson *et al.* 2007, p. 462; Moriarty et al., 2019, entire). In contrast to the dense older forests used by martens on productive soils, stands used in serpentine soils can include any seral stage and exhibit a variable tree overstory canopy closure ranging from sparse to dense (Figure 2.7; Zielinski *et al.* 2001, pp. 485, 488; Slauson *et al.* 2007, p. 462). Serpentine habitats used by martens also contain dense shrub layers and abundant rocky outcrops, providing chambers that martens use as resting structures because large woody structures are rare in serpentine habitat (Slauson and Zielinski 2009, pp. 39–40). While the distribution of serpentine soils is extensive in southwestern Oregon and northwestern California, martens have only been found in serpentine habitats in the fog influenced portions of their distribution near (less than 30 km (18.6 mi)) the coast, where the increased moisture promotes shrub composition and densities sufficient to meet marten needs (sensu Dawson 1998, pp. 482–483; Slauson et al. 2007, pp. 465–466).

Figure 2.7. Typical stand structures within serpentine habitat occupied by martens in northern coastal California and southern coastal Oregon.

Shore Pine/Dune Habitat: Coastal martens in the central coastal Oregon population occupy shore pine and transitional shore pine/Douglas–fir–hemlock forests at the Oregon Dunes National Recreation Area (Zielinski *et al.* 2001, p. 484, Moriarty *et al.* 2016b, entire, Moriarty et al.

2021b, entire). Forests grow on nutrient–poor sandy soils, dominated by young stands of shore pine and Sitka spruce <70 years old. The dense understory is dominated by *Salix hookeri* (willow), *Myrica californica* (Pacific wax myrtle), and berry–producing ericaceous shrubs such as evergreen huckleberry and salal. These shore pine forests share many of the same characteristics with serpentine habitats, including having a variable tree over–story. However, the common denominator with serpentine and older forest habitats is the presence of dense, spatially extensive ericaceous shrub understories and diverse and abundant prey (Chappell and Kagan 2001, pp. 25, 27, 95, 100; Eriksson 2016, pp. 14–19; Linnell et al. 2018, p.4; Moriarty et al., 2021, entire).

Figure 2.8. Typical stand structures within shore pine/dune habitat occupied by martens in central coastal Oregon.

2.5.3.3 *Home Range Habitat Use*

Pacific and American martens exhibit strong habitat selection at the home range scale, suggesting that this scale of selection most directly influences an individual's fitness (Thompson *et al.* 2012, p. 210). Martens establish home ranges to encompass their year–round resource needs as well as, during the breeding season, access to members of the opposite sex. Theoretically, home range size for a predator is a function of prey density and habitat quality. Smaller home ranges typically represent better habitat conditions. Marten home ranges are often selected to maximize the composition of suitable habitat and minimize low quality habitat (Figure 2.9; Phillips 1994, pp. 59–60). Individual Pacific and American marten home ranges

typically include a high proportion (≥70 percent) of older forest habitat (reviewed in Thompson *et al.* 2012, p. 218). Females, due to their solitary role in raising young, have unique needs and must have access to reliable and nearby prey resources to support the energetic demands of lactation and providing food for kits. In northern coastal California, 97 percent (38 of 39) of a typical female's within–home range resting and active locations occurred in older forests and home ranges averaged 3–4 km^2 (Figure 2.9; Slauson and Zielinski, unpublished data). For males, 30 of 39 (77 percent) within–home range resting and active locations occurred in older forests (Figure 2.9; Slauson and Zielinski, unpublished data).

Home Range Size: There is an inverse relationship between the amount of suitable habitat and marten home range size. As the amount of unsuitable habitat increases, home range size increases (Thompson 1994, p. 276; Potvin and Breton 1997, p. 462; Fuller and Harrison 2005, pp. 715–719). For example, in studies of the home range area of Pacific martens in California and Oregon, the largest home ranges (>10 km^2 (3.9 mi^2) occurred in landscapes which had been extensively logged (Ellis 1998, pp. 35–41; Bull and Heater 2001b, p. 9; Self and Kerns 2001, p. 5).

Home ranges of Pacific martens in the Sierra Nevada Mountains of California in largely unlogged forest landscapes averaged 3–6 km^2 (1.2–2.3 mi^2) for males and 2–4 km^2 (0.8–1.5 mi^2) for females (Simon 1980, p. 97; Spencer *et al.* 2011, p. 798; Moriarty *et al.* 2016a, p. 626). Limited telemetry data from coastal martens in northern California suggests that home ranges for adult males (n =3) are of similar size (3–4 km^2; 1.2–1.5 mi^2) (Slauson, unpublished data). Telemetry work in the coastal dune forest of the central coastal Oregon population revealed the smallest reported home ranges of any North American marten, with mean territory sizes of 0.84 km2 (0.32 mi2) for 3 females and 3.06 km2 (0.4 mi2) for 4 males (Moriarty et al. 2017b, p. 684; Linnell *et al.* 2018, p. 10). Telemetry data and habitat selection analysis at coastal marten detection sites in northern California and southern Oregon reveal that martens select habitat at the home range scale that includes large patches (median >1.5 km^2 (0.6 mi^2)) of older forests and serpentine habitats (Figures 2.9, 2.10.; Slauson 2003, pp. 49–54; Slauson *et al.* 2007, pp. 462–463). Telemetry data from the central coastal Oregon population show martens used areas with high amounts of vegetation cover, averaging 75 percent (Linnell *et al*. 2018, p. 10).

A. Position of an adult female home range in landscape and within home range habitat use.

Old forest (>200 years)
Mature forest (>125 years)
Pole to mid–mature (35–100 years)
Regenerating clear–cut (<35 years)
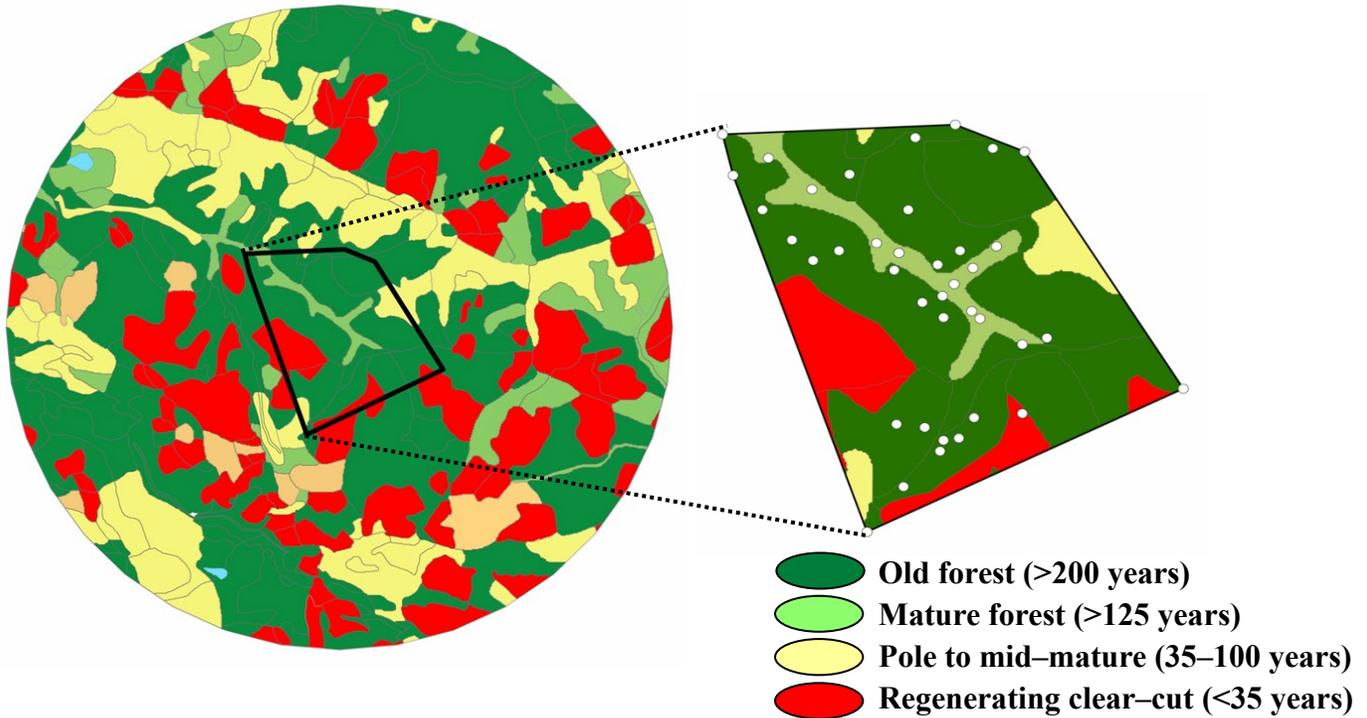
B. Position of an adult male home range in landscape and within home range habitat use.
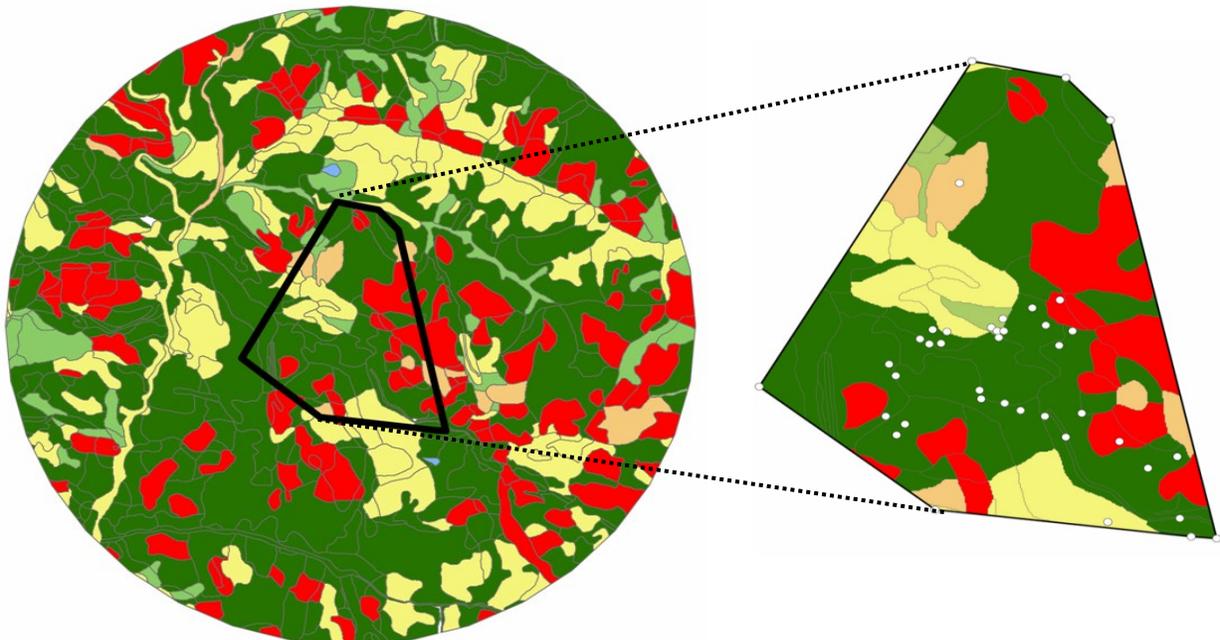
Figure 2.9. Home range position in the landscape, habitat composition, and within–home range habitat use by: (**A**) an adult female, and (**B**) an adult male coastal marten in non–serpentine habitat in northern coastal California. Home ranges represents 100 percent minimum convex polygon in non–serpentine forest habitat. Small white circles on inset maps represent rest sites and "active" sites located by radio telemetry. Source: Slauson *et al.* 2019(a).
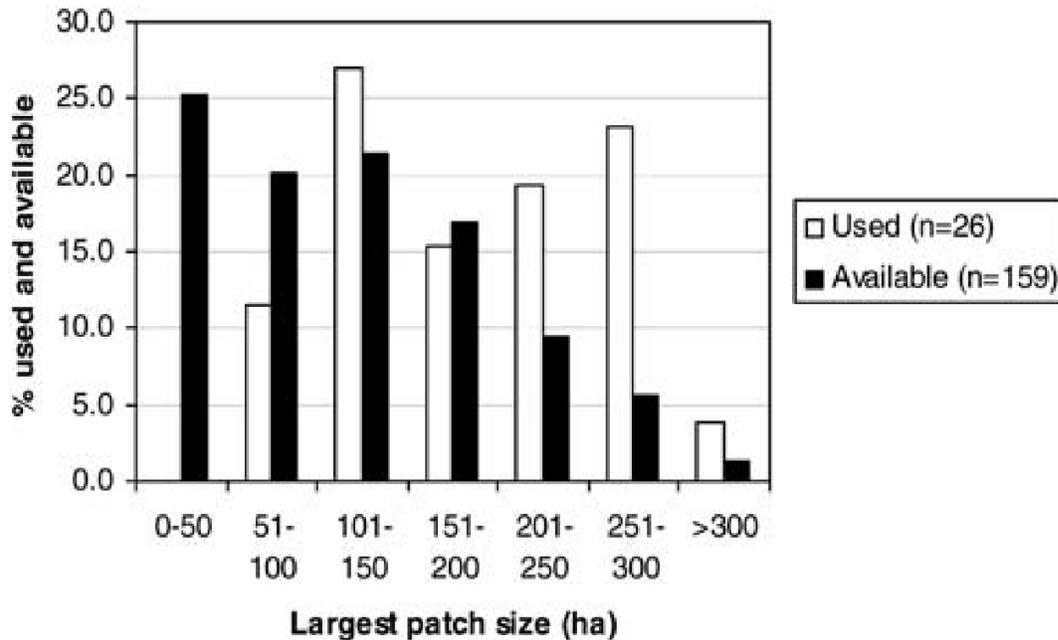
Figure 2.10. Use and availability of the largest contiguous patch of old–growth, old–growth plus late–mature, or serpentine habitat within a 1–km (0.62 mi) radius that encompassed locations occupied by the coastal marten in northern coastal California. Source: Slauson *et al.* 2007, p. 464.

2.5.3.4 *Landscape Scale Habitat Use*

The pattern and composition of habitat at the landscape scale affects the ability of martens to successfully disperse and find suitable home ranges, affects marten survival and spatial occupancy dynamics, and ultimately affects population size and persistence. The amount and spatial arrangement of suitable habitat capable of supporting one to multiple home ranges, their distances from adjacent suitable habitat patches, and the quality of the intervening habitat are key factors at the landscape scale (Chapin *et al.* 1998, pp. 1334–1336; Hargis *et al.* 1999, pp. 162–169; Potvin et al. 2000, pp. 851–854; Kirk and Zielinski 2009, pp. 765–771).

Dispersal Habitat: Dispersal is the means by which marten populations maintain and expand their distribution and dispersal habitat refers to habitat that provides a sufficient prey base and cover from predators. Successful dispersal requires the existence of functional habitat connectivity between patches of habitat suitable for reproduction to maintain or expand population size and distribution. Dispersal is also essential to maintain viable metapopulations. Studies for American martens in Ontario, Canada showed that dispersal distances are reduced by 50 percent and the success rate of juvenile dispersal is reduced in intensively logged landscapes (Johnson *et al.* 2009, p. 59). The combined effects of reduced foraging efficiency in logged stands (Andruskiw *et al.* 2008, pp. 2275–2277) and likely increased predation risk in post–logging early seral habitat may pose significant impediments for dispersing coastal martens, as demonstrated for American martens (Slauson et al., 2019a, p. 58).

During dispersal, martens use a search strategy that is not random or linear, suggesting they are responding to habitat cues and that landscape pattern likely influences movement trajectories (Johnson 2008, pp. 27–29, 36–39). Given the energetic demands of dispersal, habitats with high or low quality foraging opportunities are likely to influence movement. For example, a dispersing juvenile female coastal marten moved from her natal area along a peninsula of unmanaged stands of various seral stages, but showed repeated use of a small (less than 0.5 km^2 (0.2 mi^2)) older forest prior to returning to her natal area (Slauson and Zielinski, unpublished data). Marten search behavior during dispersal often involves sharp turning angles to avoid barriers or low quality habitat and reversing direction to return to familiar areas (Johnson 2008, pp. 33–38; Slauson *et al.* 2014, entire).

Table 2.1 Life history needs of coastal marten summarized.

Life Stage	Resources and/or circumstances needed for INDIVIDUALS to complete each life stage
Kit (birth to dispersal, ~6 months)	• **Female** provides food, thermal source, and protection from predators. (Markley and Bassett 1942, pp. 606–607) • **Den sites** are enclosed areas to shelter from weather and predators and are most often large diameter trees (live or dead) with cavities, but also include hollow logs, crevices under rocks, log piles, and squirrel nests. (Slauson and Zielinski 2009, p. 40; Thompson *et al.* 2012, pp. 223–224; Moriarty 2017a, pp. 82–88)
Juvenile *and* Adults 2+ years	• **Dispersal habitat** is an area that supports movement from natal area to a location where home range can be established. (Chapin *et al.* 1998, pp. 1334–1336; Johnson *et al.* 2009, p. 3365) • **Resting sites** include cavities, brooms, hollow logs, large limbs, rock crevices, debris piles are used to conserve energy and avoid predators. Thompson *et al.* 2012, pp. (Taylor and Buskirk 1994, pp. 253–255; Shumacher 1999, pp. 26–58; Slauson and Zielinski 2009, pp. 39–40; 223–224; Early *et al.* 2017, entire) • **Food** consists primarily of squirrels and chipmunks, birds, berries and insects seasonally. (Slauson and Zielinski 2017, entire; Slauson and Zielinski, 2019, entire; Eriksson *et al.,* 2019, entire) • **Understory** consists of dense shrub layer and decayed wood structures providing prey habitat. Shrub layer also provides protection from predators. (Slauson and Zielinski 2009, pp. 39–42; Andruskiw *et al.* 2008, pp. 2275–2277; Eriksson 2016, 19–23) • **Forest canopy cover** provides protection from aerial and terrestrial predators. Unfragmented habitat excludes bobcats, the primary predator of coastal marten, which are found in more fragmented landscapes (Slauson and Zielinski 2001; Powell et al 2003; Linnell *et al.* 2018, p. 10; Slauson et al., *In prep*) • **Home range** is habitat that provides an adequate mix of resting and foraging habitat and overlap with opposite sex individuals to provide breeding season encounters. (Ellis 1998, pp. 35–41; Bull and Heater 2001, p. 1; Self and Kerns 2001, p. 5; Slauson 2003, p. 49–54; Slauson 2007, pp. 462–463; Moriarty et al. 2017b, pp. 684–686; Linnell *et al.* 2018, p. 10)

Chapter 3 Influences on Viability

In this chapter, we evaluate the past, current, and future influences that are affecting what the coastal marten needs for long term viability. We analyzed these factors in detail in terms of causes and effects to the species by analyzing the pathways by which each influence affects the species, and each of the causes is examined for its historical, current, and potential future effects on the species' status. Current and potential future effects, along with current expected distribution and abundance, determine present viability and, therefore, vulnerability to extinction. We organized these influences around the stressors (i.e., changes in the resources needed by the coastal marten) and discuss the sources of those stressors.

3.1 Stressors with primarily direct effects on martens

3.1.1 Trapping

Historically, coastal marten were first trapped by Native Americans in California and Oregon for use as tribal ceremonial regalia and later by European settlers for fur (USFWS 2015b, p. 45). By the late 1800s and early 1900s, European settlers began trapping for the fur trade industry, with accounts of individual trappers taking 35 and 50 martens in single winters within the California portion of the historical range (Anonymous 1914, p. 20; Grinnell *et al.* 1937, pp. 5–10, 205).

By the 1920s, annual harvest totals of martens in the historical range were already in decline (Dixon 1925, pp. 23–25; Zielinski *et al.* 2001, pp. 482–483), prompting a call for closing the marten trapping season in California for fear of their extirpation (Dixon 1925, p. 25). Ultimately, marten trapping was banned in northwestern California in 1946, and was prohibited statewide in 1953 (Biberdorf 1982, p. 1). However, it is legal to trap other furbearers (e.g., bobcat, gray fox), which could result in incidental capture of martens; however, records from 2010 to 2016 show no licensed fur trappers operating in Del Norte County and less than 2 trappers per year in Humboldt County, suggesting a low likelihood of non–target capture of coastal martens (CDFW 2018, p. 23). California trapping regulations ban the use of body–gripping traps, including snares and leghold traps designed to restrain the animal as well as traps designed to kill the animal, such as conibear traps (California Code of Regulations Title 14, §3003.1(a)). Instead, cage or box traps are allowed, improving the potential to release non–target animals with less trauma than if captured with limb–holding traps (e.g., White *et al.* 1991, entire; Powell and Proulx 2003, p. 264).

Marten trapping is legal throughout Oregon (Oregon Administrative Rules 635–050–0110; ODFW 2016, p. 4). Both killing and restraining traps are legal in Oregon (Oregon Administrative Rules 635–050–0045; ODFW 2016, p. 7); restraining traps are designed to capture but not kill animals, and can include cage or box traps, as well as snares or leghold traps. The number of martens harvested in coastal Oregon counties has declined since the 1940s (Zielinski *et al.* 2001, p. 482), and by the 1970s martens were considered very rare along the

Oregon coast (Maser *et al.* 1981, pp. 293–294; Zielinski *et al.* 2001, p. 483). Today only a few trappers (generally 4 to 8 per year) pursue martens statewide in Oregon, with most of the harvest occurring in the Cascade Range. In contrast, harvest of martens in the Coast Range is infrequent, and considered extremely rare by Hiller (2011, p. 17). A total of 35 martens were harvested in coastal Oregon counties between 1969–1995, ranging from 0–4 per year and averaging 1.3 martens harvested annually (USFWS 2015b, p. 46); this excludes harvests in Lane and Douglas counties because these counties extend to the Cascades, where most of the marten harvest occurs. More recently, based on harvest reports from fur trappers and including harvests in the coastal portions of Douglas and Lane counties, 13 martens were harvested in coastal Oregon over an 28–year period from 1989 to 2016 (Broman 2018, pers. comm.). An average of 0.46 martens were trapped per year, with a maximum of three trapped in a given year. All but three of the reported martens in this latest analysis were trapped in Coos County, which encompasses the northern and southern portions of the southern coastal Oregon population, and the central coastal Oregon population, respectively. However, no coastal martens have been legally trapped or harvested in Oregon since 2014.

Best management practices have been established to minimize the likelihood of trapping of non–target organisms and reduce their chance of injury if captured in restraint traps, allowing safe release (AFWA 2006, entire). Although we are unaware of marten–specific information, injuries of varying severity have been documented in other mid–sized carnivores captured by restraint traps, including cage or box traps (see Iossa *et al.* 340–341 for summary). We assume martens would also be at risk to injury if caught incidentally in restraint traps, although we cannot predict the frequency or magnitude of injury. Furthermore, martens captured in restraint traps may be subject to capture myopathy (mortality due to the stress caused from being trapped). We are unaware of numbers of incidental capture rates or mortalities due to capture myopathy as a result of legal fur trapping. However, three martens live–trapped by researchers in the central coastal Oregon population in 2015–2016 either died in the trap or shortly after removal; cause of death was unconfirmed, but was suspected to be a result of stress–induced shock (OSU 2016, p. 1).

Historical trapping of coastal martens for fur is considered by researchers as the likely cause of the marked contraction in coastal marten distribution and reduction in population size observed in the early 20th century (Zielinski *et al.* 2001, pp. 486–487; Slauson 2003, pp. 1–2). Strickland (1994, p. 151) concluded that marten populations reduced to low levels by excessive harvest will take many years to recover, with loss of genetic variation as a potential consequence. However, decades of protection from trapping have not resulted in the recovery of coastal marten populations in northern coastal California (Slauson and Zielinski 2004, p. 61), suggesting that other factors may be limiting martens.

Few martens are harvested annually in coastal Oregon, averaging less than 1/year over the past 28 years. Recently ODFW was petitioned to ban marten trapping west of Interstate 5, which includes all of the historical range of coastal marten in Oregon. In addition, the petition seeks a ban on trapping all mammals in the Oregon Dunes National Recreation Area, where the central

coastal Oregon population occurs, to minimize incidental capture of martens in that population (Cascadia Wildlands *et al.* 2018, p. 4). Such a ban would eliminate the risk of central Oregon coast martens to be legally harvested, either intentionally or incidentally. It would also eliminate the risk of southern Oregon coast martens to be targeted by trappers, but they would still be vulnerable to incidental capture in traps targeting other furbearers.

3.1.2 Roadkill

Since 1980, 19 mortalities of coastal martens caused by vehicles have been documented in the historical range, and all were in Oregon (USFWS 2015b, p. 52; Williams 2017, unpublished data; Broman 2018, pers. comm.; Moriarty 2018, pers. comm.). Almost all were in or adjacent to the central coastal Oregon population, or in the general central Oregon coast area. Most of the reported roadkills were along U.S. Highway 101, which runs north to south the length of the historical range, and intersects the two Oregon marten populations. It runs along the eastern edge of the central coastal Oregon population, and is within 1–2 km (0.6–1.2 mi) of most known detections in that population (Figure 4.4).

Mortality rates for martens based on reported road kills is less than 1/year (roughly 0.5/year), but some unknown amount of martens killed by vehicles are likely not detected for a multitude of reasons, including scavenging of carcasses by wildlife, highway clearing or disposal by road crews, or injured martens that immediately survive a collision but move away from the roadway into cover and ultimately die from their injury. For example, persistence times of roadkill mammals similar in size to martens were 1–2 days (Santos *et al.* 2011, p. 2). Consequently, 0.5 roadkills annually is likely an underestimate of true marten roadkill mortality (Teixeira *et al.* 2013, p. 322).

Of the four marten population areas, the central coastal Oregon population appears substantially more vulnerable to roadkill mortalities than the other three. A number of factors influence the relatively high number of roadkill detections in or near this population area. First is the relatively high traffic volume on Highway 101. It is the major north–south route along the Oregon coast, and is a popular and highly promoted tourist destination. U.S. Highway 199 is the only other large highway to cross a marten population area, the California–Oregon border population. Yet daily average traffic volumes in peak months on Highway 199 are less than half that of Highway 101 near Lakeside, OR (ODOT 2016, p. 240; CalTrans undated, p. 190). Forest roads in the remainder of the population areas are mostly dirt or gravel roads with less traffic and slower vehicle speeds, reducing the risk of martens being hit by vehicles. The higher traffic levels, combined with high speeds, associated with Highway 101 increase the chances of a vehicle collision with a marten. Also, due to high levels of human activity throughout the central coastal Oregon population area, road kill martens are more likely to be seen and reported.

Second, martens in the central coastal Oregon population are closer to a busy highway compared to other population areas. The central coastal Oregon population is distributed in a long and extremely narrow configuration, and Highway 101 runs the entire length of this population.

Some marten detections are nearly adjacent to the highway, and almost all are within 1 or 2 km (Figure 4.4).

While roadkills certainly influence marten survival, we don't know whether specific segments of the population (e.g., male vs. female, juveniles vs. adults) are more vulnerable. Studies of other species indicate that dispersing animals may be more vulnerable (Conard and Gipson 2006, p. 256; Glista *et al.* 2007, p. 84; Smith–Patten and Patten 2008, pp. 849–850). Of five road kill martens of known sex reported by the Siuslaw National Forest in or near the central coastal Oregon population, four were females, but sample size is too small to conclude a sex predisposition to highway mortality (Williams, unpublished data 2017). Data compiled on nine marten road kill mortalities on the Siuslaw National Forest (in or closest to the central coastal Oregon population) showed mortalities happening in all seasons. Mortalities skewed towards a particular sex, age class, or season (e.g. denning season when a denning female mortality could consequently result in the death of attended kits) could exacerbate impacts to the population beyond reduced survival rates. However, more information is needed to determine this.

Barring any physical or regulatory modifications along Highway 101 (e.g. underpasses or speed restrictions), exposure rates of martens to vehicles is not expected to reduce in the future. Travelers to Oregon have increased 2.6 percent annually since 2010 (Dean Runyan Associates 2017, executive summary) and visitation to the Oregon coast has also been increasing, especially the central and southern coastal areas (Dean Runyan Associates 2017, pp. 28, 32, 36, and 40). Furthermore, Oregon's population is projected to grow 1.3 percent annually through 2025 (OEA 2017, unnumbered pages). Increased human population and activity along the coast will increase exposure risk to martens and likely increase roadkill mortality.

3.1.3 Rodenticides

Rodenticides are pesticides registered for the specific purpose of killing rodents. While rodenticides can be legally used in a wide array of situations (e.g., urban landscapes or growing food crops), we limit our discussion to rodenticides primarily used on forest lands, which is where martens would have their greatest likelihood of exposure to these agents. Strychnine and zinc phosphide are rodenticides often used on forest lands. In addition, the anticoagulant chlorophacinone (the commercial formulation is known as Rozol®) is registered for control of voles; in Oregon, a special use label is in place to also use Rozol to control mountain beavers (*Aplodontia rufa*). Both Federal and non–Federal forest managers use strychnine, although Federal managers in Oregon (BLM and the Forest Service) have not used any rodenticides in recent years (Bautista 2013, pers. comm.; Standley 2013, pers. comm.; Weikel 2016, pers. comm.). In addition, private forest landowners have reported use of Rozol on their lands; these reports are in compliance with the Oregon Forest Practices Act requirement to notify the Oregon Department of Forestry if pesticides are to be used, but reporting does not necessarily indicate that the pesticide was ultimately used, or how much of and where specific pesticides were

applied (Weikel 2016, pers. comm.). Thus, there is not good data on actual application of rodenticides on non–Federal lands.

Martens can be exposed to rodenticides through direct ingestion of toxic bait, many of which include flavorizers to make them more palatable to rodent pests. Martens can also be exposed indirectly by capturing prey poisoned with anticoagulant rodenticides (ARs). After rodents receive a lethal dose, it can be several days before clinical signs of contamination are evident and mortality occurs, increasing the risk of exposure in rodent predators (Gabriel *et al.* 2012b, p. 8). Predators may preferentially select substandard prey, such as those which are compromised by an additional stressor, physiologically impaired, or exhibit increased activity (Galindo *et al.* 1985, entire; Temple 1987, entire; Hunt *et al.* 1992, entire; Taylor 2009, p. 642).

Consumption of ARs through the direct ingestion of the poison or indirectly through contaminated prey species can ultimately result in death due to internal bleeding. ARs can have other effects that predispose animals to other forms of mortality (sub–lethal effects). Because the lethal mechanism of anticoagulants is to impair the normal clotting ability of blood, sub–lethal exposure levels have resulted in wildlife dying from minor wounds that otherwise may have healed with no further consequences (Gabriel *et al.* 2012b, p. 9). For martens that pursue and capture live prey, minor injuries caused by the prey animal or as a result of pursuing the prey could become lethal in AR–exposed martens.

Fishers are close relatives of martens and occupy similar niches (e.g., similar distributions, habitat use patterns, reproductive biology, prey species, and physiology). Because there has been limited testing for rodenticides in marten carcasses, we use the fisher as a surrogate for describing the prevalence of rodenticide exposure that also likely occur for martens. Over the past decade, anticoagulant rodenticides have been detected in a majority of fishers (*Pekania pennanti*) tested in California (85 percent of 101 carcasses tested), and for 13 of these fishers, poisoning by anticoagulant rodenticides was determined to have been the cause of death (Gabriel *et al.* 2015, pp. 10, 14).

Fishers that were exposed to ARs were found throughout their range in California, with no significant concentration of individuals that might implicate a specific point source. Most registered (legal) use of ARs is for non–forestry uses outside of fisher habitat. Furthermore, fishers that were radio–tagged and monitored their entire lives and known to occupy home ranges that did not encompass agriculture or developed areas where AR application is legal were exposed to ARs (Gabriel *et al.* 2012b, p. 12).

Illegal marijuana cultivation sites on public, tribal, and private forest lands are implicated as the likely source for most of the detected ARs (Gabriel *et al.* 2012b, pp. 11–13; Thompson *et al.* 2014, pp. 97–98). Known cultivation sites are widely distributed throughout the fisher's range, as well as the historical marten range in California and into southern Oregon (USFWS 2015b, pp. 58–59; USFWS 2016, pp. 146–147). Significant amounts of anticoagulant rodenticides are

scattered throughout these cultivation sites to protect plants and infrastructure (e.g., miles of plastic irrigation piping) from rodents who will destroy young plants or gnaw on plastic lines. As an example of the amount of pesticides used at these sites, 68 kg (150 lb.) were removed from select illegal cultivation sites on the Mendocino National Forest during a three–week period (Gabriel *et al.* 2012b, pp. 12–13).

The broad distribution of illegal marijuana cultivation sites, combined with the large amounts of pesticides occurring at these sites has substantial documented effects on fishers and other predators. In California's Sierra Nevada Range, female fishers with AR exposure had more marijuana cultivation sites within their home range than females with no exposure, and the number of cultivation sites also influenced female survival rates (Thompson *et al.* 2014, p. 96). Other non–target predators that have tested positive for AR exposure include northern spotted owls (*Strix occidentalis caurina*) (70 percent of 10 owls tested) and barred owls (*Strix varia*) (40 percent of 84 owls tested) in northwestern California (Humboldt and Del Norte Counties) (Gabriel *et al.* 2018, pp. 4–5).

In addition, other pesticides documented at cultivation sites (e.g., organophosphates, carbamates, and organochlorines) present an exposure risk to fishers and other wildlife, with one documented and another suspected fisher mortality associated with pesticides other than ARs (Gabriel *et al.* 2013, entire; Gabriel 2013, p. 127; USFWS 2016, p. 150). Some of these pesticides documented at grow sites have not been tested for in fisher carcasses, or are short–lived in fisher tissue, making their detection in fisher carcasses less likely (USFWS 2016, p. 150). Consequently, the magnitude of exposure to and effects from the other pesticides used in marijuana cultivation sites on fishers is not yet quantified, but could also be problematic.

Known illegal marijuana cultivation sites have occurred within marten population areas in California (USFWS 2015b, p. 58) and have been found near the southern coastal Oregon population area. The central coastal Oregon population is less likely to overlap illegal sites because the Oregon Dunes have high levels of recreation and regular visitation. Illegal marijuana cultivators favor less populated areas; the vast majority of cultivation sites in Humboldt County are located more than 0.5 km (0.3 mi) from developed roads (Butsic and Brenner 2016, p. 5). Even though the central coastal Oregon population area is unlikely to be exposed to illegal marijuana cultivation sites and associated pesticides, because of its proximity to human developments where ARs can be legally used, exposure of martens in this population may be similar to the more remote populations.

Compared to fishers, very few martens have been tested for rodenticide exposure. To date, nine martens in the historical range have been tested for ARs, six from California and three from the central coastal Oregon population. Three of those tested positive for ARs, two of which were from the central coastal Oregon population (Slauson *et al.* 2014, p. 2; OSU 2016, p. 1). Sources of ARs for the central coastal Oregon population are unknown, but the population is surrounded

by areas of concentrated human activity (e.g. recreation, campgrounds, day–use areas, communities) that may be a likely source.

We don't know specific sublethal effects of ARs on martens, but effects of ARs and other pesticides on other species is summarized in USFWS 2016 (pp. 151–157) and include: difficulty thermoregulating and increased susceptibility to hypothermia; impaired locomotion and changes in behavior that may predispose exposed individuals to other mortality sources (e.g. vehicle strikes, predation, adverse weather); reduced immune system function; and higher prevalence of infections and disease. Exposure to ARs can result in minor wounds that would normally heal to become life–threatening due to the impairment of blood clotting ability (Erickson and Urban 2004, pp. 100, Attachment D). Transfer of AR compounds from female to offspring has been documented in fishers (Gabriel *et al.* 2012b, p. 7), and parental exposure to ARs is known to cause fetal abnormalities, miscarriages, and newborn mortality in mammals (Pauli *et al.* 1987, entire; Mackintosh *et al.* 1988, p. 87; Munday and Thompson 2003, entire; Rady *et al.* 2013, entire).

Exposure to ARs may influence marten viability in other indirect ways. The reduction in prey numbers caused by ARs in localized areas may limit marten foraging success in those areas, affecting survival, reproduction, and recruitment. In addition, the timing of AR application in marijuana grow sites coincides with denning season of martens and increased energetic requirements of pregnant or lactating females, thus increasing exposure at an energetically stressful time of year for females, and at a time when effects to the population can be most influential in terms of survival and recruitment of young into the breeding population. Anticoagulant rodenticides have been documented to affect kits in utero or actively nursing, increasing likelihood of miscarriages of fetuses or starvation of kits (USFWS 2015b, p. 60).

Trends in both marijuana cultivation sites as well as wildlife exposure to ARs are on the rise. Since the first documentation of fisher exposure to ARs in 2012, there has been an increase in exposure rates and mortalities directly attributed to AR exposure (Gabriel *et al.* 2015, p. 14). Marijuana grow sites have increased substantially in California since the mid–1990s, comprising a mix of illegal operations as well as cultivation for medical purposes and, most recently, for recreational use (Franklin *et al.* 2018, p. 1). Located within the coastal marten historical range and just south of the northern coastal California population, Humboldt, Trinity, and Mendocino Counties may be the top cannabis–producing region in the world (Butsic and Brenner 2016, p. 2), with 49 of 60 sampled watersheds in or near Humboldt County containing marijuana grow operations (Butsic and Brenner 2016, p. 5).

Legal marijuana is the fastest growing industry in the United States, and legal sales in California are projected to more than triple by 2021 (Varghese 2017, entire). Consequently, an increase in cultivation sites is expected, although not all of these are near marten habitat. However, in urban–wildland interfaces where private lands abut public forest lands or occur as inholdings, grow sites may be more likely to occur within marten home ranges and those of other wildlife

species (e.g., Franklin *et al.* 2018, p. 3). In addition, the illegal grow sites implicated as the principle source of ARs are still expected to occur, although trends on how legalization will affect the distribution, abundance, and size of those sites is uncertain.

3.1.4 Carnivore community interactions

Although the historical assemblage of marten predators and intra–guild competitors were likely similar to the current assemblage, human–caused changes in vegetation composition and distribution and extensive road building have altered the landscape and likely changed distribution and densities of these species within the analysis area. For example, higher levels of predation and other interspecific killing of fishers, compared to earlier research, are thought to be associated with human–induced changes in forest landscapes that increase the likelihood of contact between fishers and predators such as bobcats (*Lynx rufus*) and mountain lions (*Puma concolor*) (Wengert et al. 2014, p. 609; Gabriel et al. 2015, pp. 12–14; Sweitzer et al. 2016, p. 446). Similarly for coastal martens, bobcats are their predominant predator, with predation accounting for 41 percent of marten mortalities in one study, and the sources of all those predations being bobcat (Slauson, *In prep*). Bobcats are a generalist species that select for regenerating harvested stands less than 30 years old (Slauson, *In prep*; Wengert 2013, pp. 93, 95). Conversely, bobcats are nearly absent from landscapes comprising a mix of 40–60–year old regenerating stands and older forests, or in primarily unmanaged mid– and late–successional stands (Slauson, *In prep*). Recent research focusing on bobcat occupancy has shown roughly over 40 percent occupancy in forested areas where less than 10 percent is older forest and more than 25 percent is younger than 40 years old (Slauson, unpublished data). Elsewhere in North America, martens generally exhibit higher predation rates and lower survival rates where landscapes are fragmented compared to landscapes that are more contiguous and unlogged (Potvin and Breton 1997, p. 467; Thompson 1994, pp. 276, 278–279).

Competition between species can affect a species distribution, abundance, and resource use. For example, the recovery of American martens in their former range in Wisconsin is thought to be limited by the presence of fishers, which exhibit similarity in diet and habitat associations (Manlick *et al.* 2017, p. 697). In the California Sierras, fisher competition is affecting marten activity and access to portions of their home ranges, with martens avoiding areas of their home range that overlapped a fisher home range (Zielinski *et al.* 2017, p. 543). Such avoidance likely has to do with minimizing marten interaction with a larger predator that could increase the chance of mortality or predation (Donadio and Buskirk 2006, entire; Zielinski *et al.* 2017, p. 543). The ability of martens to avoid fishers in areas of overlapping home ranges depends on having a heterogeneous landscape whereby a diversity of resources are available to martens (sensu Amaresekare 2003; Manlick *et al.* 2017 p. 698; Zielinski *et al.* 2017, p. 543). In Wisconsin, extensive homogenization of forest landscapes through timber harvest is thought to

have increased competition between martens and fishers, limiting marten distribution (Manlick *et al.* 2017, pp. 698–699).

In northern coastal California over the past 80 years, fishers and gray foxes (*Urocyon cinereoargenteus*) have not only maintained their interior distributions but also appear to have expanded their distributions in coastal redwood forest habitat concurrently with the decline in the distribution of coastal martens (Slauson and Zielinski 2007, p. 242). These distribution changes have occurred in areas where past logging of coastal forest has reduced the complexity of the shrub and herb layers due to various factors such as mechanical treatments, prescribed burning, herbicides, and densely stocked conifer stands (Slauson and Zielinski 2007, p. 241). In northern California, fishers and gray foxes were infrequently detected in the dense shrub layers that martens used, despite their relative abundance in other parts of the study area and in adjacent forest areas with sparser shrub layers (Slauson 2008, pp. 65–66); conversely, these areas are typically preferred by small–bodied coastal martens where they may have an advantage over other larger–bodied carnivores (Slauson *et al.* 2007, p. 466). Within and adjacent to the two Oregon populations, multiple detections of bobcats, gray foxes, and coyotes (*Canis latrans*) have been documented (Ellison *et al.* 2017, entire). Although historical data were not obtained consistently, it appears that the ranges of coyotes, gray foxes, and mountain lions (*Puma concolor*) in western Oregon have increased compared to historical records (Ellison *et al.* 2018, entire), although the authors did not quantify any population changes.

Martens are vulnerable to predation and increased competition in habitats that have been subject to either high–moderate severity fires or intensive logging in the last 40 years because both of these events remove the structural characteristics of the landscape that provide escape cover and are important to marten viability (canopy cover, shrub cover, etc.). While we raise the possible effect of these landscape changes on vulnerability to predation and competition here, we describe these effects to other specific habitat structural features used by martens under the stressors that cause this removal, specifically wildfire and vegetation disturbances other than fire (Sections 3.2.1 and 3.2.2).

Although the literature suggests that marten predator prevalence increases with increased habitat fragmentation, at this time it is difficult to determine if the rate of predation on marten has increased compared to historical levels. Further information is needed to determine if predation is increasing and how predation rates correspond to habitat fragmentation.

3.1.5 Disease

The pathogens of most concern to martens and other carnivore populations include canine distemper viruses (CDV), rabies viruses, parvoviruses, and the protozoan (single–celled organism) *Toxoplasma gondii* (Gabriel et al 2012a, p. 140). These pathogens occur naturally in the environment and can be transmitted via conspecifics or among other carnivore species. Some can also be transmitted through human actions, such as exposure to traps used by fur trappers or

researchers that have come in contact with infected animals. There has been limited testing of coastal martens for the presence of pathogens or exposure to pathogens in the historical range; the following summarizes pathogen analyses and results for captures in the central coastal Oregon population (K. Moriarty 2017, unpublished data). Nineteen blood samples from live–captured martens were tested for exposure to canine parvovirus, CDV, and *T. gondii*. Of the 19 samples, none were positive for CDV, five (26%) samples had antibodies for canine parvovirus, and 14 (74%) had antibodies for *T. gondii*. One marten showed clinical symptoms of CDV (blackened foot pads with skin lesions) but the virus was not detected, although attempts to confirm infection in such instances can be unsuccessful. We also report diseases documented in fishers tested on the Hoopa reservation, which is 5–10 km to the south of the northern coastal California population and could be a transmission vector to coastal martens. Of fishers tested from 2004 to 2007, 5 of 98 fishers tested were exposed to CDV, 28 of 90 were exposed to canine parvovirus, and 45 of 77 were exposed to *T. gondii* (Brown *et al.* 2008, pp. 13–14)

Canine distemper virus is highly contagious, and many carnivores, including all canids and mustelids, are susceptible (Thorne *et al.* 1982, pp. 10–11). Transmission occurs primarily through aerosol or direct contact with individuals, and often affects juveniles and immunosuppressed individuals more than healthy adults (Thorne *et al.* 1982, pp. 10–11; Gabriel *et al.* 2012a, p. 141–142). Animals infected with CDV may exhibit disorientation, aggressiveness, lack of fear of humans, or aimless wandering (Gabriel *et al.* 2012a, p. 142; Thorne *et al.* 1982, p. 11), likely increasing their risk to injury or mortality from other agents such as predation, accident, vehicle collision, or human interactions. Canine distemper virus can have strong immunosuppressive effects, which can enhance the severity of other infections and increase the likelihood of death (Gabriel *et al.* 2012a, p. 142).

Several parvovirus strains occur that infect or are likely to infect martens; newborn kits and juveniles are most likely to be affected. Parvoviruses are highly resistant to degradation in the environment, so can remain viable and persist in the environment under suitable conditions for months or years, increasing the infection likelihood. Feces from infected animals are sources of exposure, as well as transmission from parent to offspring either in the womb or during birth. Parvovirus infections are unlikely to limit wild carnivore populations, unless the population is small or otherwise vulnerable (Gabriel *et al.* 2012a, pp. 144–145). In these cases, because young animals are more vulnerable, the reduced recruitment associated with this disease could increase extinction risk in small populations or prevent small populations from expanding (Woodroffe 1999, p. 186).

Toxoplasma gondii is a one–celled parasite that rarely causes obvious harm in healthy individuals that are not immunosuppressed, but outbreaks resulting in significant mortalities have been documented in captive and free–ranging mustelid populations. In some of these cases, immunosuppression as a result of a previous CDV infection may have increased their vulnerability to *T. gondii* (Gabriel *et al.* 2012a, p. 145). Infection occurs when animals ingest oocysts (a life cycle stage of the parasite that is shed in the feces of the infected individual) in the

environment or consume infected hosts. Infected animals can exhibit lethargy, blindness, lameness, abortion, and loss of fear to novel stimuli, meaning they are less likely to avoid new and potentially dangerous situations, increasing their susceptibility to predation (Gabriel *et al.* 2012a, pp. 145–146). An increased susceptibility to predation in rats infected with toxoplasmosis has been observed and similarly, high predation rates currently observed by fisher populations in California could be a result of *T. gondii* infections (Wengert, unpublished data, as cited in Gabriel *et al.* 2012, p. 146).

Rabies is a virus that infects the central nervous system. It is nearly always fatal and has been implicated in several cases of extirpation of small (~50 or less) African wild canid populations (Woodroffe 1999, pp. 185–186). Transmission occurs through a bite or other direct contact with infected saliva (Gabriel *et al.* 2012a, p. 141; OHA 2018, p. 93). In the northwest, bats are the primary reservoir for rabies and represent 86 percent of the 219 rabies cases documented statewide in Oregon since 2000, and 70 percent of the 64 rabies cases documented since 2010 in the counties that occur in the California portion of the coastal marten historical range (CDPH 2010, 2011, 2012, 2013, 2014, 2015, 2016, 2017, entire; OHA 2018, pp. 93–94). Given that martens do not actively prey on bats and there is limited or no documentation of bats in marten diets (e.g., Martin 1994, entire; Slauson 2017, p. 6), the most likely encounter a marten would have with a rabid animal would be another carnivore within a marten's range. In both states, foxes had the highest prevalence of rabies cases behind bats, (CDPH 2010 through 2017, entire; OHA 2018, p. 94). Coyotes and skunks have also exhibited rabies and they commonly occur, along with gray foxes, within the population areas and historical range (CDPH 2010 through 2017; OHA 2018, p. 94; Ellison *et al.* 2017, entire). Since 2010, no cases of rabies have been documented in Del Norte County, in which the California–Oregon border and much of the northern coastal California populations occur. During the same period, 17 cases were reported in Humboldt County, where much of the rest of the northern coastal California population occurs. Hence, exposure opportunities where known coastal martens occur are extremely limited. There is currently no data that suggests that rabies is having a population level impact on their viability.

World–wide there have been several documented extinctions and near–extinctions of at–risk mammal populations as a result of disease, the closest geographically and taxonomically to the coastal marten being the black–footed ferret (*Mustela nigripes*) in North America. A few general patterns have been observed in these instances, the first being that affected populations were small. In the case of the black–footed ferret, the population was at 58 animals before a CDV outbreak occurred; similarly, social species (bighorn sheep (*Ovis canadensis*) and African wild canids) had less than 75 animals documented in their population at the time of the disease outbreak (Woodroffe 1999, p. 186). While larger populations might display a mass mortality as a result of disease infections, extinction or extirpation is rare. The other observed pattern is that most extinctions or near–extinctions are caused by generalist pathogens with a wide host range. In many of these instances, infections are often contracted from domestic species (Woodroffe 1999, p. 186). Instances where diseases are implicated in limiting populations are usually

associated with small, insular populations, or when diseases act synergistically with other population–limiting factors (e.g., habitat loss or degradation, predation, competition, nutritional stress) (Daszak *et al.* 2001, entire; Fenton and Pedersen 2005,pp. 1818–1820; Lafferty 2008, as cited in Gabriel *et al.* 2012, p. 139).

Each of these diseases presents a risk to marten populations, but exposure levels and ultimate effect on populations are difficult to document until an outbreak is actually observed. The disease with the best data on exposure opportunities by martens is rabies. Though the disease is often fatal, carnivore vector sources are infrequent (2.4 cases per year in the 5–county area of California, and 1.7 cases per year statewide in Oregon) (CDPH 2010 through 2017, entire; OHA 2018, pp. 93–94), though irruptions can occur, and all marten populations are within the range of multiple vector sources. The limited number of documented cases in potential disease vectors (e.g., other carnivores) limits the risk of martens contracting rabies and presents a limited risk for an entire marten population succumbing to an outbreak.

We suspect martens may have greater vulnerability to the other diseases described because there are more opportunities for transmission of most of these pathogens compared to rabies. That is, rabies requires encountering a rabid animal and getting bitten, whereas the other pathogens can be transmitted through feces or, in the case of CDV, through the air when in proximity of an infected individual. Furthermore, in the case of parvovirus, the lengthy duration in which it can remain viable in the environment increases exposure opportunity to this pathogen. In addition, we know that the central Oregon coast martens have been exposed to canine parvovirus and *T. gondii*, and that exposure to these diseases, in addition to CDV, has occurred in fishers near the northern coastal California population. Additionally, exposure rates of sympatric carnivores were similar to fishers on the Hoopa, providing more vectors for disease transmission to nearby coastal martens in the northern coastal California population (Brown 2008, pp. 57–59). Fortunately, all but the central coastal Oregon population is relatively removed from human developments and concentrated activities, limiting marten exposure to domestic animals that are often disease vectors (Woodroffe 1999, pp. 186,191). Campgrounds, day use areas, and development around the Oregon Dunes increase opportunities for disease transmission to central coastal Oregon martens as a result of the human–associated activity and increased exposure to domestic animals. Furthermore, given this population exhibits the highest marten densities observed in North America (Linnell *et al.* 2018, p. 1), con–specific transmission of diseases could be more widespread in this population than in other coastal marten populations. With population sizes estimated at less than 100 each for all four coastal marten populations, an outbreak in an individual population puts it at a higher risk for extirpation compared to larger populations. Disease transmission between populations, however, may be limited given the distance between populations, limiting the chance that martens throughout the historical range succumb.

3.2 Stressors with primarily indirect effects on martens (e.g. on marten habitat)

3.2.1 Wildfire

Fires have been a historical disturbance throughout the marten's historical range, and are critical to the creation of structural features and conditions conducive to marten habitat, including snags, down logs, hollow trees, resting platforms, as well as increased horizontal and vertical diversity in the tree canopy. Fires that burn small patches in a forest stand increase structural complexity by opening a gap in the canopy that allows regeneration of different tree species, resulting in multiple age and size classes and canopy layers (Lofroth *et al.* 2010, pp. 123–124). Standing snags and down woody debris are created by fire (and other natural disturbance events) and important for martens and their prey. Fire can also injure trees to the degree that they become infected with heart–rot fungi, resulting in the hollow trees and logs used by martens for denning and resting (Lofroth *et al.* 2010, pp. 124–127; Weir *et al.* 2012, pp. 237–238). Consequently, fires are a necessary disturbance feature for maintaining and developing habitat structural features used by martens and their prey.

The effects of fire on plants and vegetation communities, composition, and structure is a complex interaction between fire components such as intensity, frequency, duration, and seasonality, combined with vegetation characteristics such as defense strategies/adaptations or developmental stage. A fire regime describes the nature of fires that occur over long periods in a particular ecosystem type, with a generalized pattern of effects of the fire that characterize the ecosystem (Brown 2000, p. 3).. For this analysis, we categorize fire regimes based on fire severity, which relates to the effect of the fire on the dominant vegetation. The three regimes are low, moderate, and high severity (Agee 1993, pp. 22–24), which correspond to understory, mixed severity, and stand–replacement regimes, respectively (Brown 2000, p. 5; Arno 2000, pp. 97–98, 105–109, 113–115). While an individual fire regime can exhibit fires of each of the three severity classes, it is the predominant fire severity that determines the fire regime (Agee 1993, p. 23).

Several fire regimes occur within the area. Low severity regimes are most prevalent in southwest Oregon and the California portion of the coastal marten historical range (Figure 3.1). Fires are most frequent in this regime, generally occurring every 5–25 years (Arno 2000, pp. 97–98, 105). When applied to a forest stand, low severity fires are generally non–lethal to the overstory trees and result in little structural change to the dominant vegetation. Approximately 80 percent or more of the dominant overstory vegetation survives (Brown 2000, p. 3). The large retention of overstory canopy results in little effect of the fire on landscape fragmentation, but it can reduce shrub cover, which can affect marten dispersal and population expansion (see below).
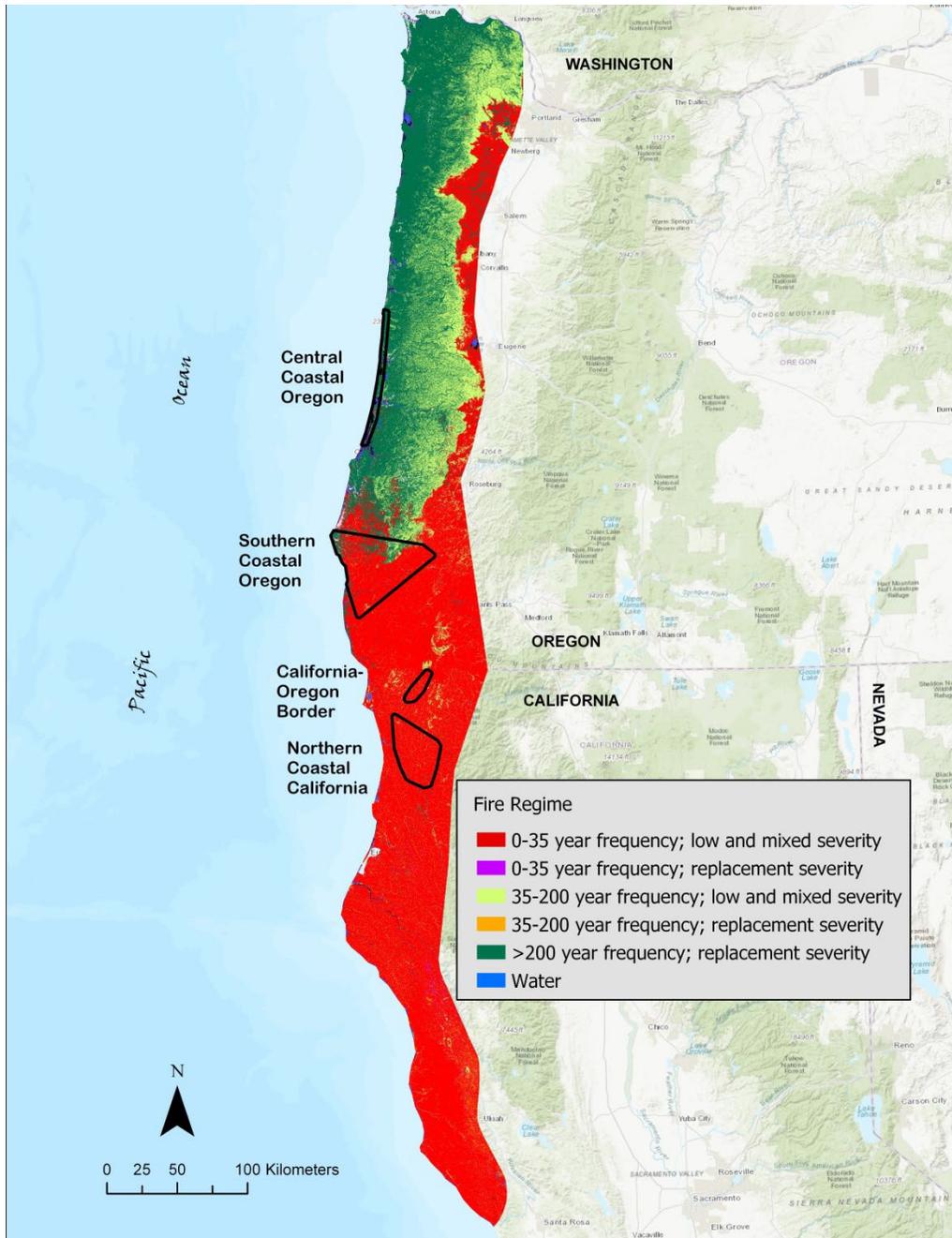
Figure 3.1. Fire regime map for the historical range of the coastal marten. Mixed severity and replacement severity is equivalent to moderate and high severity in the text. Black polygons are boundaries of coastal marten populations. Derivation of population boundaries are described in Section 4.2 Current Range and Distribution (1980–current).

Understory vegetation (e.g., the shrub layer) generally succumbs to low–severity fires, decreasing the density, diversity, and abundance of understory vegetation, at least in the short term. These understory reductions may diminish marten prey habitat quality and quantity, decrease prey abundance and availability, or remove cover for effective foraging, although

abundance of some prey species may increase (Lyon *et al.* 2000a, p. 26; Lehmkuhl *et al.* 2006, pp. 596–597; Monroe and Converse 2006, pp. 237–238; Fontaine and Kennedy 2012, p. 1553). Removal of the understory by fires also affects cover for martens, likely reducing the suitability of the stand as marten habitat and affecting population trends (Slauson *et al.* 2009b, p. 11). Areas devoid of understory expose martens to predators; these areas require from one to two decades, and as much as 30 years, for the understory to return to conditions suitable for martens (Slauson et al. 2010, pp. 8–9, 12; Slauson 2014, pers. comm.).

In the high severity fire regime, substantial changes to the forest structure occur, with approximately 80 percent of the dominant vegetation being consumed or killed by the fire (Brown 2000, p. 3). Wetter, lower elevation, coastal forests dominated by Sitka spruce (*Picea sitchensis*), Douglas–fir (*Pseudotsuga menziesii*), and western hemlock (*Tsuga heterophylla*) typify this regime. High severity regimes typically exhibit long fire return intervals, from 200 to over 500 years in the wetter forests (Arno 2000, p. 113); mean fire return intervals in the Oregon Coast Range over the past 2,700 years are 230–240 years (Long *et al.* 1998, p. 786, Long and Whitlock 2002, p. 223). Major fires are infrequent occurrences, often requiring specific conditions such as uncommon and extreme drought accompanied by a higher than normal amount of lightning ignitions (Arno 2000, p. 113). This regime dominates the northern and much of the central coastal Oregon portion of the coastal marten historical range, with scattered inclusions to the south (Figure 3.1).

High–severity, as well as moderate–severity fires, may eliminate structural features used by marten prey and used by martens for denning and resting (Innes *et al.* 2006, p. 3184). Such features include large downed wood, snags, hollow trees, and trees with mistletoe brooms or other platforms. Generally, wildlife response to wildfire varies by species and by fire severities, with different avian and small mammals exhibiting both positive and negative population effects from different fire severities (Fontaine and Kennedy 2012, pp. 1554–1556). Avian and small mammal prey species may avoid stands where structures and canopy cover are lost (Lyon *et al.* 2000a, pp. 26, 28). In some cases, severely burned stands may have increased insect abundance or seed availability, increasing the abundance of some small mammal and bird species (Lyon *et al.* 2000a, pp. 26–29); however, as noted above, martens typically avoid these stands if there is not sufficient overhead cover or mature shrub cover.

Beyond the effects of direct habitat loss to martens, the fragmentation may also affect martens by increasing the abundance of habitat generalist carnivores such as bobcats, which prey on martens and are associated with young regenerating stands (Slauson *In prep*). The reluctance of martens to enter recently created openings in an effort to avoid predation raises the energy costs of traveling through their home range as they avoid these openings (e.g. Moriarty *et al.* 2015, pp. 1871–1873). Once overstory is removed, it may take many decades to reestablish the canopy cover that existed prior to the fire (Naney *et al.* 2012, p. 2). Substantial amounts of snags and down wood exist after high severity fires that may be available for marten denning and resting once suitable canopy cover is developed. However, it may take a century or more for the stand to

develop the live–tree structural features used by martens, such as tree platforms and cavities, as well as recruiting replacement snags and down logs (Bull *et al.* 1997, p. 10; Lofroth *et al.* 2012, pp. 123–130; Naney *et al.* 2012, p. 7).

The moderate–severity fire regime is the most complex of the three regimes, encompassing a mix of fire severities. Even individual fires can exhibit a mix of severities, resulting in a mosaic of residual vegetation composition and abundance within a fire perimeter. Fire return intervals generally range between the low and high severity regimes, but also overlap those regimes (Arno 2000, p. 107). The inland Douglas–fir/hardwood zone found at low and mid–elevations in the Oregon Coast Range, down through the Siskiyou and Klamath mountains and into the California Coast range typifies this regime (Figure 3.1). Compared to the other vegetation types within the marten historical range, this zone might have undergone the greatest structural changes because of fire suppression efforts over the past century (Agee 1993, p. 317). For instance, suppression of more frequent fires that would reduce fuel loads and fuel continuity across the landscape has resulted in larger amounts of more homogeneous fuels, putting the landscape at risk for larger, more severe wildfires than historically occurred in this regime. This is of particular consequence given the projected increase in fire season and fire events predicted due to climate change (see Section 3.2.3, Climate Change), which may result in substantial and abrupt changes to the vegetation communities in this portion of the historical range.

The marked reduction of overstory canopy that occurs in high– and moderate–severity regimes affects important microclimate characteristics, such as increasing temperature or reducing shelter from wind and precipitation. This can affect not only the quality of the area as marten habitat, but also the composition and density of vegetation that ultimately revegetates the area. The future mature forest may differ substantially from the pre–burn forest (Huff and Smith 2000, p. 40). In some cases, fires may lead to vegetation type conversion from forest to shrub land, which may permanently change landscape permeability for martens, reducing connectivity within and between populations (Perry *et al.* 2011, pp. 707–709; Naney *et al.* 2012, p. 7; Collins and Roller 2013, p. 1801). Such changes are expected to be more likely with projected climate change, substantially altering marten habitat composition over the next century.

Fire trends in the coastal marten historical range

Fires are a regular occurrence outside of the high severity fire regime and in southwest Oregon and northwest California portion of the historical range, where the southern three marten population areas occur. We used Gradient Nearest Neighbor (GNN) data (Davis *et al.* 2015, pp.9–26; Cohen *et al.* 2018, entire; LEMMA 2018, entire) to quantify fire and other disturbance events in all forest age classes in each of the population areas as well as the coastal marten historical range between 1987 and 2016 (Table 3.1). While the Central Coastal Oregon population area has not experienced any fires over this period, the northern Coastal California population has experienced the largest fires over the most years, affecting over a third of the population area (Table 3.1). Not included in Table 3.1 are fires outside of the population areas;

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of particular interest is the 2017 Chetco Bar fire, which burned 769 km^2 (297 mi^2) just south and outside of the southern coastal Oregon population. Given the location of this fire between the southern Oregon and the two California populations, it is likely a further impediment to population connectivity by the southern coastal Oregon population.

These data overestimate the amount of fires in marten habitat because they represent all fire severities in all forest habitat types. Consequently, some of these areas may not have burned in marten habitat, or some areas may have burned in lower severities that retained forest overstory canopy with little contribution to landscape fragmentation, although understory may still have been removed, affecting marten cover and ability to avoid predators. Conversely, these data do not capture smaller fires ac, so do not reflect all the fires that have occurred in marten population areas. Nevertheless, they indicate substantial areas within marten population areas, particularly the Northern Coastal California population, that have been affected by fires over the past three decades. With climate change projections, this is expected to increase (see climate change section below).

Table 3.1. Number and sizes of large fires (>100 ac) within coastal marten population areas from 1987 to 2016. Fires are of all severities and affect all forest age classes. Data derived from Gradient Nearest Neighbor (GNN) (LEMMA 2018, entire). Areas are km^2 (mi^2)

Coastal marten population	N fire years[a]	Forest Area affected by fire	Range of forest area affected by fire per fire year	Percent of Population Area	Proportion of Federal area affected	Proportion non–Federal area affected
Central Coastal Oregon	0	0	0	0	0	0
Southern Coastal Oregon	3	221 (85)	35–125 (14–48)	10	14	<1
California–Oregon Border	3	19 (7)	3–10 (1–4)	10	10	7
Northern Coastal California	6	391 (151)	1–199 (<1–77)	35	42	17

[a] A "fire year" is the number of years with fire events. A single year could have multiple fire events.

Specific to marten habitat known to be affected by wildfires, between 2000 and 2014, approximately 17 percent of the moderate and high suitability habitat in the north coastal California population was burned; a single fire, the 2008 Blue Fire, was responsible for a large percentage. In the California–Oregon border population area, roughly 12 percent of the moderate–high suitable habitat was burned in the Longwood Fire of 1987. This indicates that substantial amounts of marten habitat in a population area can be burned in single fire events or over a few years at varying severities, affecting marten habitat suitability to different degrees.

The large fire sizes that can occur throughout the historical range will continue to affect marten population areas. Though infrequent, larger fires in the high severity fire regimes of the northern

and central Oregon Coast Range during the 1800s to the mid–1900s have ranged from 1,200 to 3,200 km^2 (460–1,240 mi^2) (Morris 1934, pp. 317–322; Pyne 1982, pp. 336–337; Agee 1993, p. 212; Wimberly *et al.* 2000, p. 172). In the moderate severity regimes of southwest Oregon and northwest California, fires are more frequent, but some have been especially large, including: 1) the Biscuit Fire of 2002, which burned almost 2,000 km^2 (780 mi^2) ac on the Rogue River–Siskiyou National Forest, including over 120 km^2 (46 mi^2) in the southern coastal Oregon population area; 2) the Klamath Complex fire in 2008, which burned 780 km^2 (300 mi^2) in Siskiyou County, California, including 200 km^2 (77 mi^2) of the northern coastal California population area; and most recently, 3) the Chetco Bar fire in 2017, burning over 770 km^2 (297 mi^2) on the Rogue River–Siskiyou National Forest just south of the southern coastal Oregon population. Climate change is projected to result in longer fire seasons, producing more and larger fires (see climate change section). Fires large enough to encompass all or most of all four individual population areas are already occurring and are expected to increase, raising concern over the resiliency of at least the three southern marten population areas, which have been most affected by recent fires and are in a fire regime particularly vulnerable to future fires.

3.2.2 Vegetation disturbances other than fire

Forest vegetation composition, age, and structure in the analysis area have changed substantially since European colonization. Older forests, which generally represent suitable habitat for martens in much of the analysis area, have declined substantially from historical amounts. Temperate coniferous forests specifically are considered one of the most heavily impacted terrestrial habitats in western North America (Ricketts *et al.* 1999, pp. 83–87, 93–98).Within the analysis area, older forests historically encompassed >75 percent of the coastal California area, 50 percent of the Klamath and Siskiyou region in northern California and southwest Oregon (Strittholt *et al.* 2006, p. 367), and 25 to 85 percent of the Oregon Coast Range (Ripple 1994, pp. 46–47; Strittholt *et al.* 2006, pp. 367–368; Teensma *et al.* 1991, pp. 2–4, 8–9; Wimberly *et al.* 2000, p. 167). Remaining older forests in the redwood region, Oregon Coast Range, and Klamath–Siskiyou region is estimated at 5, 18–24, and 38 percent, respectively, of what occurred historically (Bolsinger and Waddell 1993, p. 3; USFWS 1997, p. 4; Wimberly *et al.* 2000, p. 176; Strittholt *et al.* 2006, p. 367; Mooney and Dawson 2016, p. 548).

Not only has the amount of older forest substantially declined, but the composition and spatial distribution has changed as well. Historically, old forest patches were large (2,100 to 8,500 km^2, 810 to 3,280 mi^2), in near proximity to younger stands, and ranged from 25 to as much as 75 percent of the Coast Range Province (Wimberly *et al.* 2000, p. 167; Wimberly 2002, p. 1322; Wimberly *et al.* 2004, p. 152). Conversely, forest patches less than 80–years old were generally less than 2,000 km^2 (772 mi^2). Today, the composition has reversed, with the largest old–growth forest patch (>200 yrs) in the province being 6.5 km^2 (2.5 mi^2) ha, while the largest patch of <80–year forests is larger than 3,000 km^2 (1,160 mi^2) (Wimberly *et al.* 2004, p. 152). Historically, forests greater than 200 years old were well dispersed in large patches across the Oregon Coast Range Province (Wimberly *et al.* 2004, p. 152). After large–scale disturbance

events, large tracts of old–forest habitat were available and could serve as refugia for associated species while adjacent disturbed areas grew into habitat (Wimberly *et al.* 2000, p. 177). These historically large patches no longer occur, limiting martens to a more fragmented landscape with smaller refugia patches.

Though fires are a natural occurrence in these systems, most of the older forests in the historical range have been lost to timber harvest (Bolsinger and Waddell 1993, pp. 1–4; Wimberly and Ohmann 2004, pp. 643–644; Strittholt *et al.* 2006, p. 367). Historically, forests lost to natural disturbances such as fire would regenerate and eventually advance towards and often return to a climax condition until another disturbance occurred. Today, disturbances such as timber harvests designed to maximize timber volume production does not allow for the return of the forests to their climax condition; instead, forests harvested in such a manner are regenerated and then harvested on a timeline much shorter than that needed to reach climax condition. Hence, while large areas of older forests might have been lost to fires historically, much of it would eventually return.

Forest harvest trends in the Pacific Northwest have generally been shaped by land ownership. Timber harvest rates have historically been higher on private industrial lands than on private non–industrial or public lands (Cohen *et al.* 2002, pp. 122, 128–129; Easterday *et al.* 2018, p. 144). With the implementation of the Northwest Forest Plan (NWFP) in 1994, the magnitude of disturbance on Federal lands has substantially declined (Kennedy *et al.* 2012, p. 128). As an example, in the Oregon Coast Range Province, timber harvest has substantially shifted even further from Federal to State and private lands, with an 80 to 90 percent reduction in timber harvest rates on Federal lands (Azuma *et al.* 2004, p. 1; Spies *et al.* 2007, p. 50). Federal lands managed under the NWFP includes all of the coastal marten historical range.

Understanding the general timber management regimes of different land ownerships informs our understanding of the potential to retain or develop marten habitat. As an example from western Oregon, private industrial timberlands exhibit the most intensive forest management, having the greatest harvest rates and largest harvest unit sizes, followed by private non–industrial timberlands, then Federal lands (Cohen *et al.* 2002, pp 128–130). Industrial timber management generally emphasizes clear–cut logging, 40–60 year stand rotations, removal of late seral structure and hardwoods, and road building, producing forests with highly altered structure and composition. Managed regenerating stands often lack key habitat elements used by martens, especially suitable resting and denning structures in large standing live and dead trees, and dense, ericaceous shrub cover (Slauson and Zielinski 2007, entire, Slauson *et al.* 2010, pp. 8–9, 11–12). On industrial timberlands, older regenerating stands that begin to develop such conditions are rarely retained for any duration, unless they are part of other conservation agreements for other fish or wildlife species, or retention requirements for unstable slopes or riparian and wildlife retention buffers (e.g. Oregon Administrative Rules Chapter 629; ODF 2018, entire). Barring any changes in forest practice regulations, or timber markets, this trend is likely to continue on these lands.

Vegetation management effects on marten habitat

Marten habitat loss due to logging of older forests is considered a likely cause of population declines and low population levels in northwest California over the past half century since trapping has been banned since 1946 (Buskirk and Ruggiero 1994, p. 11; Zielinski *et al.* 2001, p. 487; Proulx *et al.* 2004, p. 51). Vegetation management, such as timber harvest, thinning, fuels reduction, and habitat restoration, can result in temporary or permanent loss, degradation, or fragmentation of suitable coastal marten habitat.

Fragmentation occurs when otherwise contiguous habitat is segmented into smaller pieces. This can happen through natural processes, such as fire, or human activities such as development or timber harvest. Fragmentation not only results in a decrease in available habitat, but also reduces available habitat patch sizes and increases isolation of habitat patches, contributing to a decline in biological diversity within the original habitat (Andrén 1994, pp. 355–356). Martens are sensitive to fragmentation (Chapin *et al.* 1998, pp. 1327, 1335; Hargis *et al.* 1999, pp. 157, 167–168; Thompson *et al.* 2012, pp. 212–221). Fragmentation creates large expanses of unsuitable habitat with little cover, which martens are reluctant to cross or venture into for risk of exposure to predators (Spencer *et al.* 1983, p. 1185; Hargis and McCullough, 1984, pp. 142–143, 145; Thompson 1994, pp. 276, 279; Thompson and Harestad 1994, p. 357; Drew 1995, pp. 40–41, 58–66; Potvin *et al.* 2000, pp. 849–850; Payer and Harrison 2003, pp. 151–152 2004; Moriarty *et al.* 2015, pp. 1871–1873). Multiple studies of martens across North America indicate that when the availability of mature and old forests in a landscape was reduced to less than 70 percent, from either wildfire or forest management, the population density of marten declines (Thompson *et al.* 2012, p. 228). With less than 30 percent of a landscape in unsuitable habitat (openings or young stands), martens may traverse the landscape without crossing large gaps, thus perceiving the landscape as connected (Andrén 1994, pp. 357, 359; Hargis *et al.* 1999, pp. 169–170; Chapin *et al.* 1998, pp. 1334–1336). An exception to this 70 percent threshold is martens in Newfoundland, where martens are found using regenerating forests and pre–commercially thinned stands (Hearn *et al.* 2010, p. 724). However, a notable difference in Newfoundland is that martens have few predators and competitors (Thompson *et al.* 2012, p. 228), thus resulting in less risk of predation or injury when they venture into these more open stands.

Landscape fragmentation can also influence marten body conditions. The combination of a marten's small body size, high metabolic rate, and spatial requirements 3–4 times larger than similar–sized carnivores (Sirén *et al.* 2016, p. 170) creates a high energy demand that makes them particularly sensitive to habitat loss and degradation. Body condition scores of marten have been negatively associated with increased amounts of regenerating forests (Johnson *et al.* 2009, p. 3364; Cheveau *et al.* 2013, p. 755), where increased landscape fragmentation may increase travel distances to avoid openings and get to other suitable habitats (Andrén 1994, pp. 357, 359). Lowered body condition can consequently affect predator elusion and avoidance, foraging, reproduction, and ability to fight off disease and infection.

As noted in Section 2.5.3 (Habitat Use), coastal martens are closely associated with dense shrub layers, whether it is in the temperate conifer forests, serpentine sites with a sparser forest overstory, or shore pine–dominated coastal dune forests (Slauson *et al.* 2007, p. 464; Eriksson 2016, pp. 14–23). It has been postulated that because coastal martens are so closely associated with dense shrub communities, forest fragmentation effects may be ameliorated with natural shrub development occurring in post–harvest stands that may continue to provide suitable cover and prey for martens. However, the shrub communities that coastal martens are associated with are a complex and dense combination of shade–tolerant, long–lived, mast–producing species, and do not include the shade–intolerant and short–lived *Ceanothus* species that dominate recently disturbed sites (Slauson *et al.* 2007, pp. 465–466). Hence, until more is understood about coastal marten use of shrub communities in recently disturbed areas, we consider the information about effects of forest canopy fragmentation on North American martens to be applicable to coastal martens. Many timber harvest activities also reduce the patch size of older forest stands and eliminate or substantially reduce the amount of forest structural elements required by coastal martens, such as large trees, snags, and logs. The continued simplification of the structure of forests combined with their fragmentation into smaller, more isolated, patches is a potential threat facing the coastal marten within redwood forests (Cooperrider *et al.* 2000, p. 163). The probability of detecting the coastal marten increases as patch size of late–successional forest increases, indicating the importance of larger stands of late–successional forests to coastal martens (Slauson 2003, p. 67). Simplification of stands by removing overhead cover, large diameter trees and logs, the shrub layer, and resting and denning structures reduces marten habitat quality. Among the various timber harvest practices, clearcutting and other openings such as heavily thinned areas are especially detrimental to marten suitable habitat and populations, with martens selecting against these openings (Steventon and Major 1982, pp. 177–178; Snyder and Bissonette 1987 pp. 173–174; Fredrickson 1990, pp. 28–29; Katnik 1992, p. 97; Chapin *et al.* 1998, p. 1328; Cushman et al. 2011, pp. 1142–1146; Moriarty et al. 2016a, p. 628).

Some vegetation management activities, such as thinning, fuels reduction projects, and forest habitat restoration have the potential to improve habitat suitability for the coastal marten in the long term by minimizing loss of late–successional stands due to wildfires and accelerating the development of late seral characteristics. This is especially applicable in the mixed conifer, low–moderate fire regimes in southern Oregon and inland California, where the southern three marten populations occur, and where the influences of fire suppression and exclusion over the past century have had the greatest effects on forest composition. These types of projects could ultimately increase the overall amount, distribution, and patch size of suitable coastal marten habitat, especially with the projected effects of climate change increasing fire sizes and longevity in the future (see climate change section). However, these same activities would likely result in a short–term degradation, loss, or fragmentation of suitable coastal marten habitat because many of these treatments focus on removal of surface fuels, often made up of the shrub layers that are important components of marten habitat. Thinning, fuels reduction, and restoration projects would need to be carefully planned to minimize the simplification of the stands and degradation

of the shrub layer that provides marten cover, while restoring fuel loadings and stand composition to a condition more resilient to hotter, drier summers and longer fire seasons (Credo 2017, pp. 65–71, 89–91; Moriarty *et al*. 2017a, p. 51).

Other non–forest restoration projects can result in removal of suitable marten habitat. The Forest Service is planning a restoration project to return sand dunes to their natural movements that occurred before non–native beach grass was introduced and stabilized the dunes (USFS 2017, entire). This stabilization facilitated the growth of the ericaceous shrubs and conifers that are used by martens today. Though alternatives are not yet developed, restoration may entail removing this shrub and conifer habitat in specific areas to facilitate sand movement and dune formation processes. Vegetation removal in marten habitat or near occupied marten areas may increase the areas of open sand that would further fragment an already small marten population footprint and increase distances between shrub and conifer patches that expose martens to predators.

Timber harvest trends in the coastal marten historical range over the past 3 decades

To describe trends in old forest vegetation in the coastal marten historical range, we looked at vegetation types that provide a rough approximation of marten habitat. Changes in amounts of older forest on all ownerships throughout the Northwest Forest Plan (NWFP) area, which includes the historical range, have been monitored since the NWFP's implementation (Davis *et al*. 2015, p. 1). Monitored older forests were defined using an "old–growth structure index" (OGSI) that consisted of measurable forest structure elements, such as density of large live and dead trees, diversity of tree size classes, and percent cover of down woody material (Davis *et al.* 2015, pp. 5–7). These elements are commonly considered key ecological and structural attributes of old–growth forests within the NWFP area, and are valuable forest structures for martens. Similarly, OGSI was used in a landscape–scale habitat suitability model for coastal martens and consistently occurred in the top–ranked models, indicating its importance in habitat prediction under this model (Slauson *et al.* 2019(b), pp. 38–41). We specifically used the OGSI–80 values (Davis *et al.* 2015, pp. 16–18) to represent forests that exhibit the structural characteristics used by martens because of the consistency in comparison with available NWFP data.

The index does not track some stands that martens are known to occupy, such as the shore pine community used by the central coastal Oregon population. Conversely, it can also represent some stands that may have large tree structure but are too open (as low as 10 percent canopy cover (Davis *et al.* 2015, pp. 13)) for martens to use. Thus, we acknowledge some unknown level of over–representation and under–representation of marten habitat using the OGSI–80 category of forests. In addition potential sources of error in the satellite imagery used to monitor OGSI–80 forests can result in erroneous changes in mapped OGSI–80 pixels, particularly in highly fragmented landscapes or landscapes with dramatic stand edges (e.g. clearcuts or recent plantations next to a forest stand several decades or more older) that create shadows that can be attributed incorrectly (Davis *et al.* 2015, pp. 51–52; Davis, 2018, pers. comm.). Finally, OGSI–

80 forests were mapped at a scale of approximately 2.5 ac, and may represent many small fragments and strands that meet the stand definition of OGSI–80, but are too small, isolated, or discontiguous to function as marten resting, denning, or foraging habitat. Compared to Federal lands, non–Federal lands comprised a larger proportion of these "scatter" pixels, and very little "Core" pixels (at least 98 ft. from a non–OGSI–80 pixel); conversely, Core pixels and their associated edge accounted for half or more of the total amount of older forest on Federal lands (Davis *et al.* 2015, pp. 24–25, 27–29, see Figure 7). Consequently, OGSI–80 values for non–Federal lands in Tables 3.2–3.4 are less likely to represent suitable conditions for martens than values on Federal lands. OGSI–80 amounts should not be considered an absolute representation of available habitat, but rather an approximation of distribution and trends in forest conditions conducive to marten habitat.

Trends in OGSI–80 values since 1993, as well as amount of specific disturbances affecting those values, are available from Davis *et al.* (2015, pp. 30–33) at the physiographic province scale (see Davis *et al.* 2015, p. 10 for map of physiographic provinces). These data also allowed us to calculate the amount of OGSI–80 that was a result of ingrowth over the 20–year monitoring period. We used existing GNN data (LEMMA 2018, entire) to scale the OGSI–80 data to the individual marten population areas as well as the entire historical range. While we were also able to track specific disturbance types (e.g. fire vs. a combination of timber harvest and other disturbances such as landslides, windthrow, forest insects and forest diseases) at the marten population and range scale, it was not tied to OGSI–80 forests, but rather to all forested landscapes. Thus, at the marten population area and marten historical range scale, we were unable to track what proportion of OGSI–80 vegetation was lost to disturbance. Because we did not know what proportion of OGSI–80 forests were lost to disturbance at this scale, we also could not determine proportion of OGSI–80 forests that were a result of ingrowth.

Harvest rates since 1985 has been substantially greater on non–Federal lands than Federal lands throughout the NWFP area (Davis *et al.* 2015, pp. 26–38). This mostly holds true within the marten population areas as well, where timber harvest rates on non–Federal lands are substantially greater than on Federal lands, with the exception of the California–Oregon border population (Table 3.3; it should be noted that this table represents disturbances in forests of all age classes, and the harvest disturbance category also includes other non–fire disturbances). Because this population, as well as the Central Coastal Oregon population, exist in relatively small areas, harvest or fire events can appear as disproportionately large effects and likely do not approximate general trends in the area. Indeed, at the province scale, the trend of substantially greater harvest on non–Federal land holds true (Table 3.4). Also similar to the trend across the NWFP, fires are the predominant disturbance source on Federal lands, especially compared with non–Federal lands (Tables 3.3 and 3.4). Particularly striking is the northern coastal California population area, where 42 percent of the Federal lands have experienced fire over the past 30 years (Table 3.3). This raises concerns about future habitat trends in this population area with the projected increase in fires and area burned as the climate warms and dries.

Looking at trends in OGSI–80 at the larger scale of NWFP physiographic provinces that overly the historical range reflects conditions outside of the population areas, which is valuable as connections between populations as well as establishment of future populations. Similar to the NWFP area and most of the marten population areas, within individual provinces across the marten historic range, percent of OGSI–80 stands harvested for timber range from less than 1 percent to 2 percent on Federal lands, and 7.2 to 35 percent on non–Federal lands (Table 3.4). Percentage of loss of OGSI forest across all four provinces because of timber harvest is 1 percent on Federal lands and 17 percent on non–Federal lands (Table 3.4). Total loss of OGSI–80 forest on non–Federal lands (18.5 percent), regardless of disturbance type, was more than double that on Federal lands (8.3 percent) (Table 3.4). However, ingrowth of younger stands into the OGSI–80 condition replaced much of this loss on both Federal and non–Federal lands (Table 3.4), although replacement of an OGSI–80 value is not necessarily by a forest of equivalent OGSI–80 value. That is, ingrowth reflects forests that have, through succession, recently attained the characteristics sufficient to meet the OGSI–80 thresholds, but may not be as structurally complex as those stands that were removed via disturbance during the analysis period. Thus, this analysis does not reflect potential degradation and loss of structural complexity of older forest habitats that may be suitable for martens. Nevertheless, the substantially greater removal of older forests on non–Federal lands compared to Federal lands suggests that current and future marten habitat is most likely to be retained on Federal lands, though retention in the smaller population areas may be more unpredictable due to random distribution of disturbances.

Monitoring of older forests within the NWFP area indicate that total losses on Federal lands to disturbance is within the range originally expected when the NWFP was developed. Loss to wildfire on federal lands was similar to that expected with the NWFP, but losses from timber harvest are approximately one quarter of what was projected. The NWFP anticipated a five percent per decade loss of older forests due to timber harvesting and wildfires, combined with recruitment eventually expecting to exceed those losses; the NWFP further projected that 50 to 100 years after implementation began, older forests on Federal lands would return to within the range that occurred prior to logging and extensive fire suppression (Davis *et al.* 2015, pp. 7–9). Thus, net loss of older–forests that could provide marten habitat are not occurring at a rapid rate on Federal lands, and are in line with projections made 20 years ago in the NWFP. If NWFP projections continue to hold, older–forests are expected to increase throughout the coastal marten historical range. However, projected increases in the number and sizes of fires because of climate change may exceed NWFP predictions on loss due to disturbance and the location and severity of fires may have unknown negative consequences for coastal marten.

Table. 3.2 Older forest amounts (OGSI–80) by Federal and non–Federal ownership within physiographic provinces and associated marten populations within the coastal marten historic range from 1993 to 2012. Physiographic province data from Davis *et al.* (2015, Tables 6 and 7, pp. 30–31). Coastal marten population data from Gradient Nearest Neighbor (GNN) (LEMMA 2018, entire).

Physiographic Province[a] and Associated Marten Population Areas	Area of OGSI–80 in km^2 (mi^2)								All lands
	Federal lands				Non–federal lands				
	1993	2012	Net change	Net % change	1993	2012	Net change	Net % change	Net % change
Oregon Coast Range Province	2,564 (990)	2,589 (1,000)	25 (10)	1.0	3,299 (1,274)	2.414 (932)	–885 (–342)	–26.8	–14.7
Central Coastal Oregon Population	39 (15)	40 (16)	1 (<1)	2.9	17(6)	13 (5)	–4 (–1)	–21.9	–4.47
Oregon Klamath Province	4,584 (1,770)	4,130 (1,595)	–454 (–175)	–9.9	1,580 (610)	1,420 (548)	–160 (–62)	–10.1	–10
Southern Coastal Oregon Population	902 (348)	863 (333)	–38 (–15)	–4.3	146 (56)	146 (57)	<1 (<1)	0.4	–3.62
California Klamath Province	10,859 (4,193)	10,413 (4,020)	–446 (–172)	–4.1	2,036 (786)	2,000 (772)	–36 (–14)	–1.8	–3.7
California – Oregon Border Population	78 (30)	79 (31)	1 (1)	1.9	13.7 (5.3)	14.5 (6.0)	<1 (<1)	5.8	2.5
Northern Coastal California Population	516 (199)	476 (184)	–40 (–15)	–7.8	91 (35)	87 (33)	–4 (–2)	–4.9	–8.15
California Coast Range Province	639 (247)	660 (255)	21 (8)	3.4	5,127 (1,980)	5,498 (2,123)	371 (143)	7.2	6.8
Coastal marten historical range (not a summation of columns)	9,376 (3,620)	8,761 (3,383)	–615 (–237)	–6.6	8,609 (3,324)	7,822 (3,020)	–787 (–304)	–9.1	–7.8

[a]Physiographic province data includes province areas outside of the coastal marten historical range.

Table 3.3. Area of all forest age classes subjected to disturbance within the coastal marten historical range, by marten population area, from 1993 to 2012. Data derived from Gradient Nearest Neighbor (GNN) (LEMMA 2018, entire).

	Forested area subject to disturbance, 1993–2012. Areas in km² (mi²)								
	Disturbance amounts on Federal land			Disturbance amounts on non–Federal land			Disturbance amounts all ownerships		
Marten population area	Total forest area	Fire Loss Percent area	Harvest and Other Loss[a] Percent area	Total forest area	Fire Loss Percent area	Harvest and other Loss[a] Percent area	Fire Loss Percent area	Harvest and other Loss[a] Percent area	Total Disturbance Percent area
Central coastal Oregon	132 (51)	0	14	106 (41)	0	51	0	31	31
Southern coastal Oregon	1,616 (624)	11	5	674 (260)	<1	41	8	15	23
California–Oregon Border	156 (60)	2	3	39 (15)	<1	4	2	3	5
Northern coastal California	799 (309)	32	3	327 (126)	14	17	27	7	34
Coastal Marten Historical Range	17,202 (6,642)	19	7	31,598 (12,200)	1	37	8	26	34
		Fire Loss	Harvest and other disturbance		Fire Loss	Harvest and other disturbance	Fire Loss	Harvest and other disturbance	
Percent of total explained disturbance in marten historical range		74	26		3	97	22	78	

[a]This disturbance primarily comprises timber harvest, but can include other non–fire disturbances such as landslides, blowdown, forest insects, and forest pathogens.

Table 3.4. Loss of OGSI–80 forest types to different disturbance categories on Federal and non–Federal lands within the Physiographic provinces that overlay the coastal marten historical range from 1993 to 2012. Data from Davis *et al.* (2015, Tables 6 and 7, pp. 30–31).

	Federal Lands					Non–Federal lands				
	Percent of Federal lands subject to specific disturbance			Percent All Disturbance loss from 1993	Percent ingrowth	Percent of non–Federal lands subject to specific disturbance			Percent all disturbance loss from 1993	Percent ingrowth
Physiographic Province (associated marten population area)	Timber harvest	wildfire	Other			Timber harvest	wildfire	Other		
Oregon Coast Range (CCO)	1.9	0.0	0.1	2.0	3.0	35.0	0.0	0.6	35.6	8.8
Oregon Klamath (SCO)	1.6	12.2	0.1	13.9	4.0	23.2	0.9	0.5	24.6	14.5
California Klamath (CAOR, NCC)	0.5	7.0	0.2	7.7	3.6	10.1	2.0	0.8	12.8	11.0
California Coast Range	0.3	2.5	0.5	3.2	6.5	7.2	0.5	0.3	8.0	15.2
Total	1.0	7.1	0.2	8.3	3.7	17.4	0.7	0.5	18.5	12.6
Percent of total explained loss	11.5	86.0	2.4			93.7	3.7	2.6		

3.2.3 Climate Change

Observed changes in the global climate is unequivocal, with many changes observed since 1950 being unprecedented over the span of tens to thousands of years. Specifically, the atmosphere and oceans have warmed, amounts of snow and ice have decreased, and sea levels have risen. Human influence on the changes in global climate have been extensively documented and human–related emissions of greenhouse gases are the highest in history (IPCC 2014, p. 2). Multiple lines of evidence indicate a strong and consistent relationship between cumulative carbon dioxide emissions and projected global temperature change to the year 2100 (IPCC 2014, p. 8). Changes in precipitation, combined with melting snow and ice are affecting the quality and quantity of water resources, while many animal species have shifted their geographic ranges, seasonal activities, migration patterns, abundances, and interactions with other species in response to ongoing changes (IPCC 2014, p. 6).

Many species are subject to an increased risk of extinction because of climate change projected through this century and beyond, particularly as climate change interacts with other stressors. Most plant species cannot shift their geographical range fast enough to respond to current projected rates of climate change predicted in most landscapes. It is also predicted that most small mammal species will not be able shift their ranges fast enough to keep up with changes in climate and vegetation predicted under moderate and high greenhouse gas emissions scenarios. In particular, species with low rates of dispersal, especially in landscapes with no elevation refugia, and species in isolated habitats such as mountaintops, islands or small, protected areas are especially at risk (IPCC 2014, p. 70). The high future risk to plant and animal species is corroborated by observations of historical climate change rates, which are less than the current human–associated rates, causing significant ecosystem shifts and species extinctions over the past several millennia (IPCC 2014, pp. 13, 67).

Global mean surface temperature is projected to increase from 0.3 to 0.7 ºC (0.5 to 1.3 ºF) over the period 2016–2035. Beyond 2035, the projections diverge substantially depending on which emissions scenario is modeled (a range of greenhouse gas emissions are represented in the literature ranging from a stringent emissions mitigation scenario to very high emissions amounts) (IPCC 2014, pp. 58–59). Consequently, the level of confidence in the model projections drops substantially after 2035.

Climate change is projected to result in warmer temperatures for Oregon, California, and the entire Pacific Northwest. Increasing temperatures are projected across the historical range during the 21st century, ranging from an increase of 1.6 to 3.9 ºC (3 to 7 ºF) by the mid–21st century and from 2.8 to 6.1 ºC (5 to 11 ºF) by the late 21st century (Hayhoe *et al.* 2004, p. 12423; Mote and Salathé 2010, p. 41; Halofsky *et al.* 2011, p. 14; Cayan *et al.* 2012, p. 4; Pierce *et al.* 2013a, p. 844; Dalton *et al.* 2017, p. 4). Summer temperatures are projected to increase more than winter temperatures (Mote and Salathé 2010, pp. 41–42; Salathé *et al.* 2010, pp. 65–66; Cayan *et al.* 2012, p. 8; Pierce *et al.* 2013a, p. 845). In addition, heat waves are projected to increase in

frequency, intensity and duration, especially under higher–emissions scenarios (Hayhoe *et al.* 2004, p. 12423; Tebaldi *et al.* 2006, pp. 191–200; Salathé *et al.* 2010, p. 69; Cayan *et al.* 2012, p. 10; Pierce *et al.* 2013a, p. 848; Dalton *et al.* 2017, p. 8). However, atmospheric high–pressure ridges over the Pacific Ocean are projected to weaken, which would result in reduced heat extremes for coastal Oregon compared to inland areas (Dalton *et al.* 2017, p. 8), diminishing, though not abating, the effects of climate change in the Oregon portion of the historical range.

Future precipitation trends vary considerably (Pierce *et al.* 2013b, entire), but most simulations project a north-to-south gradient for total precipitation (with precipitation decreasing from north to south) across the region, with slight increases projected for Oregon and an overall drying trend for California (Hayhoe *et al.* 2004, p. 1242; Christensen *et al.* 2007, p. 890; Littell *et al.* 2011, p. 74; Dalton *et al.* 2017, p. 9). Nearly all simulations show a strong decrease in summer precipitation across the entire region, and many show an increase in winter precipitation, especially in Oregon (Mote and Salathé 2010, pp. 42– 43; Halofsky *et al.* 2011, p. 15; Cayan *et al.* 2012, pp. 13–20; Pierce *et al.* 2013a, p. 849; Dalton *et al.* 2017, p. 9). Annual precipitation is projected to decrease in California and increase in Oregon (Cayan *et al.* 2012, pp. 14–17; Dalton et al 2017, p. 9), however, model projections are variable and there is less confidence in projecting precipitation amounts than in projecting temperature (Dalton *et al.* 2017, p. 9). Overall, summers are expected to be warmer and drier, and extreme heat and precipitation events are projected to be more frequent.

Warmer and drier conditions are expected to affect vegetation, and ultimately marten habitat, in several ways. First, the distribution and range of vegetation communities are predicted to shift in response to changing temperatures and precipitation and increased growing season. In Oregon and California, the range and area of temperate conifer forest (e.g. redwood and coastal western hemlock forests associated with northern coastal California and the northern and central Oregon Coast Range) is projected to contract and be replaced by mixed evergreen forests (e.g. the vegetation type currently occurring throughout much of the Siskiyou and Klamath mountains), which is projected to increase (Lenihan *et al.* 2008, p. S221; Shafer *et al.* 2010, pp. 180–181; Dalton *et al.* 2017, p. 54). Because coastal martens currently occupy both vegetation types, this simple shift in vegetation types would be expected to function as suitable habitat for martens, assuming structural characteristics were retained (e.g. dense understories and large trees, snags, and logs).

Change in vegetation distribution, however, is more complex than vegetation communities merely shifting with suitable or unsuitable climate condition. Changing disturbance regimes, such as fires and insects, will likely have the largest effects on vegetation composition, structure, and distribution. Warmer and drier summers over the past decades have created drier fuel conditions in forests in the western U.S., resulting in more frequent and larger fires, increased area burned, and longer fire seasons. Longer fire seasons are a result of declining snowpack, earlier snowmelt, patterns of monsoonal moisture, or reduced high pressure ridging in the Pacific Ocean. The temperatures, precipitation, and snowpack exhibited in Oregon in 2015 may

represent conditions that will be normal by mid–century based on climate models; 2015 was the warmest in Oregon's recorded history, and while winter precipitation was near normal, the 2.8–3.3 ºC (5–6 ºF) increase in winter temperature resulted in much of the precipitation falling as rain rather than snow, resulting in record low snowpack across the state. Consequently, the 2015 fire season was the most severe in the recorded history of the Pacific Northwest. Near normal precipitation with low snowpack can be expected more often with continued warming (Dalton *et al.* 2017, pp. 12–13).

With more winter precipitation falling as rain rather than snow, the Klamath Mountains, currently classed in the rain–snow transition zone, are projected to become mainly rain–dominated by mid–century (Dalton *et al.* 2017, p. 17). Loss of snowpack will influence the hydrologic regime, affecting water availability to the local vegetation and resiliency to fire, insects, and diseases (Dalton *et al.* 2017, p. 17). Consequently, fire return intervals in the Coast Range and Klamath Mountains are expected to decrease by half, which would result in a near tripling of the annual area burned in this century compared to last (Sheehan *et al.* 2015, pp. 20–22; Dalton *et al.* 2017, p. 46). Specifically, the annual percent area burned in the Coast Range and Klamath Mountains are expected to increase from 0.53 percent in the 20th century to 1.12 percent (111 percent increase) and 1.27 percent (140 percent increase) under moderate representative concentration pathway (RCP 4.5) and high (RCP 8.5) emission concentrations, respectively throughout the 21st century. Correspondingly, mean fire return intervals are expected to decrease from 81 years to 47 (42 percent decrease) and 37 (54 percent decrease) under moderate and high emission concentrations, respectively (Sheehan *et al.* 2015, pp. 20–22).

Along with increased disturbance associated with wildfires, warming and more frequent drought is expected to increase tree susceptibility to insects and pathogens, resulting in more frequent and severe insect outbreaks and increased damage to trees. The range of Swiss needle cast in the Coast Range of Oregon has substantially increased over the past two decades; originally limited to the northern coast, it now extends south through Coos County and its distribution and severity is expected to increase with warmer winters at higher elevation sites. Swiss needle cast affects Douglas–fir, and can result in forest species composition change when severe outbreaks affect Douglas–fir growth (Dalton *et al.* 2017, p. 51). Warmer temperatures also cause trees to become more susceptible to the fungal disease sudden oak death (*Phytophthora ramorum*), which is expected to continue to spread northward in the Oregon Coast Range (Shafer *et al.* 2010, p. 185).

Such abrupt changes in vegetation communities because of fire, insects, or diseases, and synergistic interactions among these disturbances, will have an immediate effect on marten through direct habitat loss or modification. Partial– to stand–replacing disturbances, primarily caused by wildfires but also insects and diseases, open the canopy and facilitate a shift in vegetation. With changing climates, the new vegetation that becomes established under these scenarios may differ from those that were in the original stand. For instance, conifer forests in the Klamath–Siskiyou region that are disturbed by fires under warmer and drier climate scenarios are expected to be replaced not by mixed evergreen conifer forests, which may

continue to be suitable marten habitat (see above), but rather by woodlands, shrub lands, or even grasslands (Lenihan *et al.* 2008, p. S221), resulting in a permanent loss of suitable habitat for coastal marten. Already fires are becoming more frequent and larger as fire season lengths have increased in the Pacific Northwest from 23 days in the 1970s to 116 days in the 2000s (Dalton *et al.* 2017, p. 46). Consequently, loss of local species populations associated with late–successional forests are expected to increase (Lawler *et al.* 2012, p. 384; McKenzie *et al.* 2004, p. 899).

In northern coastal California and southwestern Oregon, climate projections within 50 years suggest that drier conditions will result in a narrower extent, greater fragmentation, and increasingly limited inland distribution of coastal forests (DellaSala 2013, entire). Refugial bioclimatic conditions, including year–round cool, moist conditions, and summer fog, for the redwood region were projected to only occur reliably in the current northern extent of the redwood's range in northern Humboldt and Del Norte counties of California under moderate emissions scenarios within 50 years (DellaSala 2013, entire). A decline in the frequency of summer fog has been inferred based on trends of maximum land temperature and its association with fog (Johnstone and Dawson 2010, entire). Projecting change in number of fog hours as a result of climate change is difficult with many uncertainties, but research along the central California coast indicates that, while fog hours in the southern part of the study area (e.g. Monterey Bay) are projected to increase, hours in the northern part of the study area (Point Reyes) are expected to decrease (Rogers 2016, pp. 44–50). It is unknown if this trend is consistent into northern coastal California. If so, fog reduction is expected to reduce the inland and southern extent of coastal forest conditions, potentially reducing the distribution of dense shrub layers that are an important component of coastal marten habitat.

Marten species throughout North America are expected to exhibit a northward range shift over the next century in response to climate change, with the greatest effect expected to occur at the southern latitudes and lower elevations (Lawler *et al.* 2012, pp. 371–372). Even moderate warming scenarios could result in large reductions in population connectivity and ultimately, genetic diversity (Wasserman *et al.* 2012b, p. 211). This is because much of the projected change in the distribution of martens is driven, in part, by change in persistent snowpack with the assumption that martens associated with montane environments will shift to follow climates where snowpack is retained. However, because coastal marten do not occur in areas of persistent snowpack and are not as tied to elevational gradients as marten in montane environments, any potential shift in the range of coastal marten may be comparatively less (Lawler *et al.* 2012, pp. 394). Indeed, 84 to 99 percent of current known coastal marten locations in California are predicted to remain in climate conditions suitable for marten at the end of this century (Stewart *et al.* 2016, pp. 75–76). These numbers represented three climate scenarios, a warm and wet scenario, each with a low and high emission scenario, and a hot and dry low emission scenario. Only the hot and dry scenario with high emissions (i.e., the most extreme climate projection) resulted in a substantial decline (77 percent) in currently occupied sites that would be outside of

a suitable climate envelope for coastal marten. Furthermore, the warm and wet climate scenario, under both emission levels, is projected to increase the area where climate will be suitable for marten. However, although suitable climate may exist for marten, as noted above, this does not mean that habitat will remain suitable, especially if it is substantially altered by fires or other disturbances.

Changes in prey populations will also likely drive marten distribution as it is expected to drive their close relative, the fisher (Lawler *et al.* 2012, p. 395). Indeed, small mammals in Yosemite National Park have exhibited range shifts into higher elevations compared to surveys a century ago, while the average minimum monthly temperature has increased 3.7 ºC (6.7 ºF) in that time frame (Moritz *et al.* 2008, pp. 261–262). This is consistent with predicted effects of climate change and a plausible scenario to expect with increasing temperatures over the next century. With the exception of the central coastal Oregon population, marten populations in the historical range already occupy sites near the elevational limits of the area, providing little if any climate refugia if lower elevations warm enough to affect martens or their habitat.

A final projected effect of climate change is sea level rise, which would have the most effect on the Central Coastal Oregon population. Median sea levels are projected to increase from 5 cm (2 in) in Crescent City, CA up to 13 cm (5.1 in) in Newport, Oregon by 2030, with no difference in projection between low and high emission pathways. By 2050, median seal level in Crescent City is projected to be 11 and 13 cm (4.3 and 5.1 in) using the low and high emissions pathway, respectively, and 25 and 26 cm (9.8 and 10.2 in) in Newport (Dalton *et al.* 2017, p. 33). In addition to sea level rise, coastal flooding is expected to increase because of storm surges, sea level anomalies, and intense rainfall, substantially increasing the height above sea level that can be affected by sea level rise (Dalton *et al.* 2017, p. 34). Furthermore, saline groundwater intrusion due to flooding can affect existing vegetation in areas not directly flooded (Stewart *et al.* 2016, p. 18). Hence, even if martens in the Central Coastal Oregon population occupy ground that is at an elevation higher than projected sea level rise, saltwater intrusions into the groundwater may affect the existing shore pine and ericaceous shrub vegetation that martens are occupying, potentially degrading or removing habitat.

3.2.4 Cascadia Subduction Zone Earthquake and Associated Tsunami

A coastal hazard for which the Central Coastal Oregon population is particularly vulnerable is the possibility of a large magnitude earthquake and subsequent tsunami. The Cascadia Subduction Zone runs offshore along the Pacific Coast of North America from northern California to British Columbia, Canada, where the Juan de Fuca tectonic plate descends (subducts) below the North American plate. Marine core samples along the subduction zone indicate a recurring frequency of earthquakes over the past 10,000 years, documenting 43 major earthquakes over this time period, averaging an earthquake every 220 to 430 years, depending on which of the four different segments of the subduction zone was sampled (PhysOrg 2016, p. 1). When the entire zone is involved, a magnitude 9.1 earthquake is believed possible. At least 19 of

the 43 documented earthquakes were believed to involve the entire subduction zone and likely a magnitude 8.9 to 9.2, with the most recent event occurring in January of 1700 (ODGMI 2012, entire).

Such undersea quakes cause a vertical displacement of water that creates tsunamis, as exhibited in similar offshore quakes in recent history near Chile (1960 and 2010), Alaska (1964 Good Friday), Sumatra (2004), and Japan (2011) (ODGMI 2012, entire). The closer the undersea earthquake is to shore, the larger the onshore wave and the more destruction that occurs. Sand deposits carried onshore by the tsunami associated with the 1700 earthquake have been found 1.2 miles inland, while older tsunami sand deposits have been found in estuaries 6 miles inland (ODGMI 2012, entire). Yet wave effects can reach lands on the opposite side of the ocean, as was seen in North America after the 2011 quake in Japan. Conversely, the 1700 Cascadia quake created a tsunami that was recorded in Japan (Phys.Org 2016, p. 1). Consequently, large magnitude quakes elsewhere in the Pacific also present a tsunami risk to the Oregon coast, though of smaller size.

There is a 16–22 percent chance of a magnitude 8 or greater earthquake off the central and northern Oregon portion of the Cascadia subduction zone in the next 50 years (Dalton *et al.* 2017, p. 34; Goldfinger *et al.* 2017, p. 42). Modeling of magnitude 8.7 to 9.2 Cascadia subduction earthquakes yielded tsunamis with shoreline wave heights ranging from 4 to 25 m (Witter *et al.* 2013, p. 1783). The subsequent tsunami from the quake would flood significant acreage of low–lying areas along the coast and damage human developments and infrastructure as well as alter ecological communities (Figures 3.1, 3.2, and 3.3; Dalton *et al.* 2017, p. 34). The largest of waves would inundate almost all of the area occupied by martens in the central coastal Oregon population. Few martens would remain, if any, and substantial areas of habitat would be degraded or destroyed through physical damage from the wave and associated debris. Even smaller waves associated with lower magnitude earthquakes from the Cascadia subduction zone, or from more powerful distant earthquakes such as off the Alaskan coast (ODGMI 2012, entire) (e.g., small and medium tsunamis, as labeled in Figures 3.1, 3.2, and 3.3), could inundate some of the low–lying marten home ranges found in the shore pine forest population, reducing the population size and potentially further fragmenting habitat.

Figure 3.2. Tsunami inundation map representing five tsunami scenarios (Priest *et al.* 2013, entire) within the northern portion of the central coastal Oregon marten population. Color bands represent different inundation levels of different tsunamis, whose size is based on multiple subduction and earthquake metrics, including earthquake magnitude.

Figure 3.3. Tsunami inundation map representing five tsunami scenarios (Priest *et al.* 2013, entire) within the central portion of the central coastal Oregon marten population. Color bands represent different inundation levels based on multiple subduction and earthquake metrics, including earthquake magnitude.
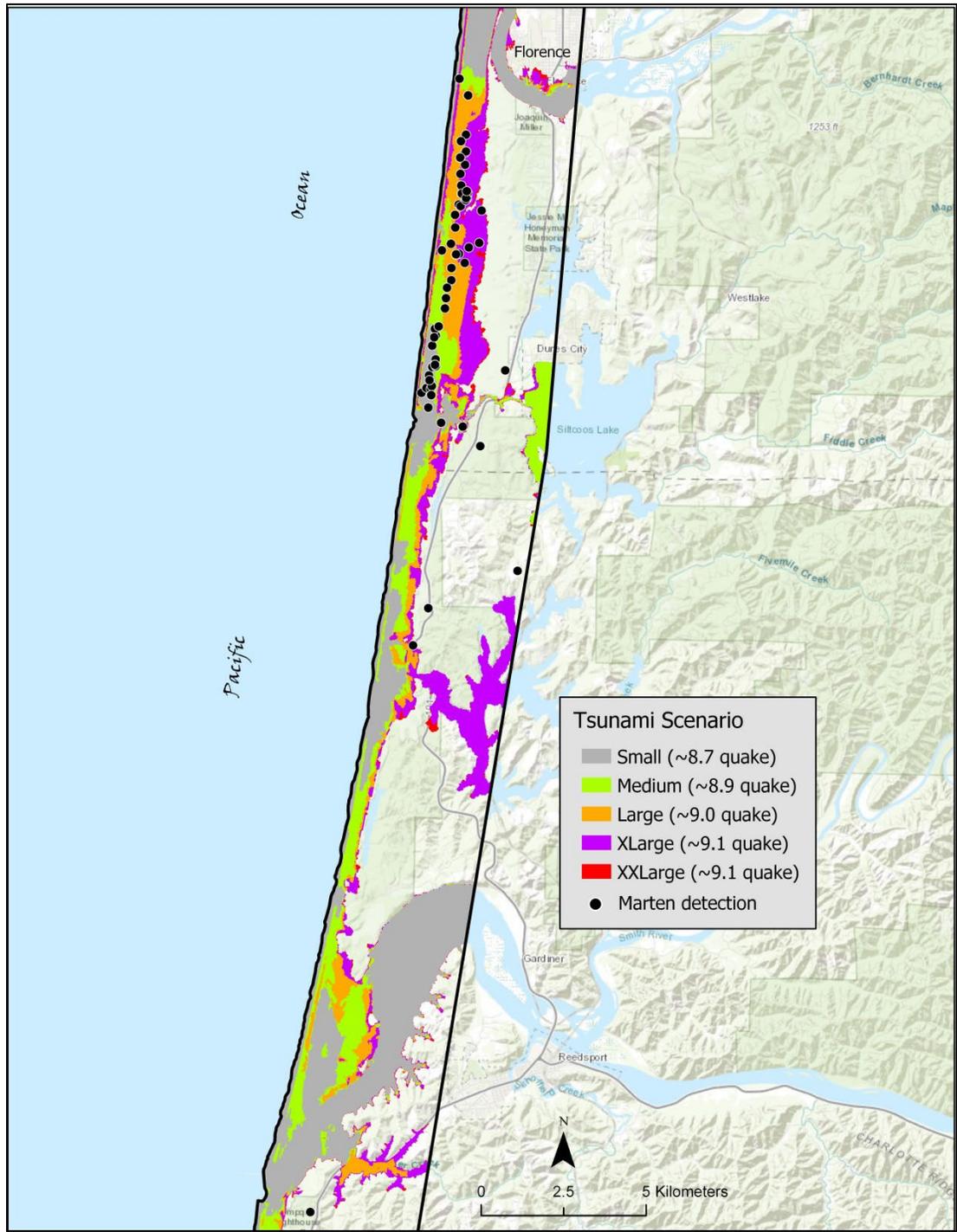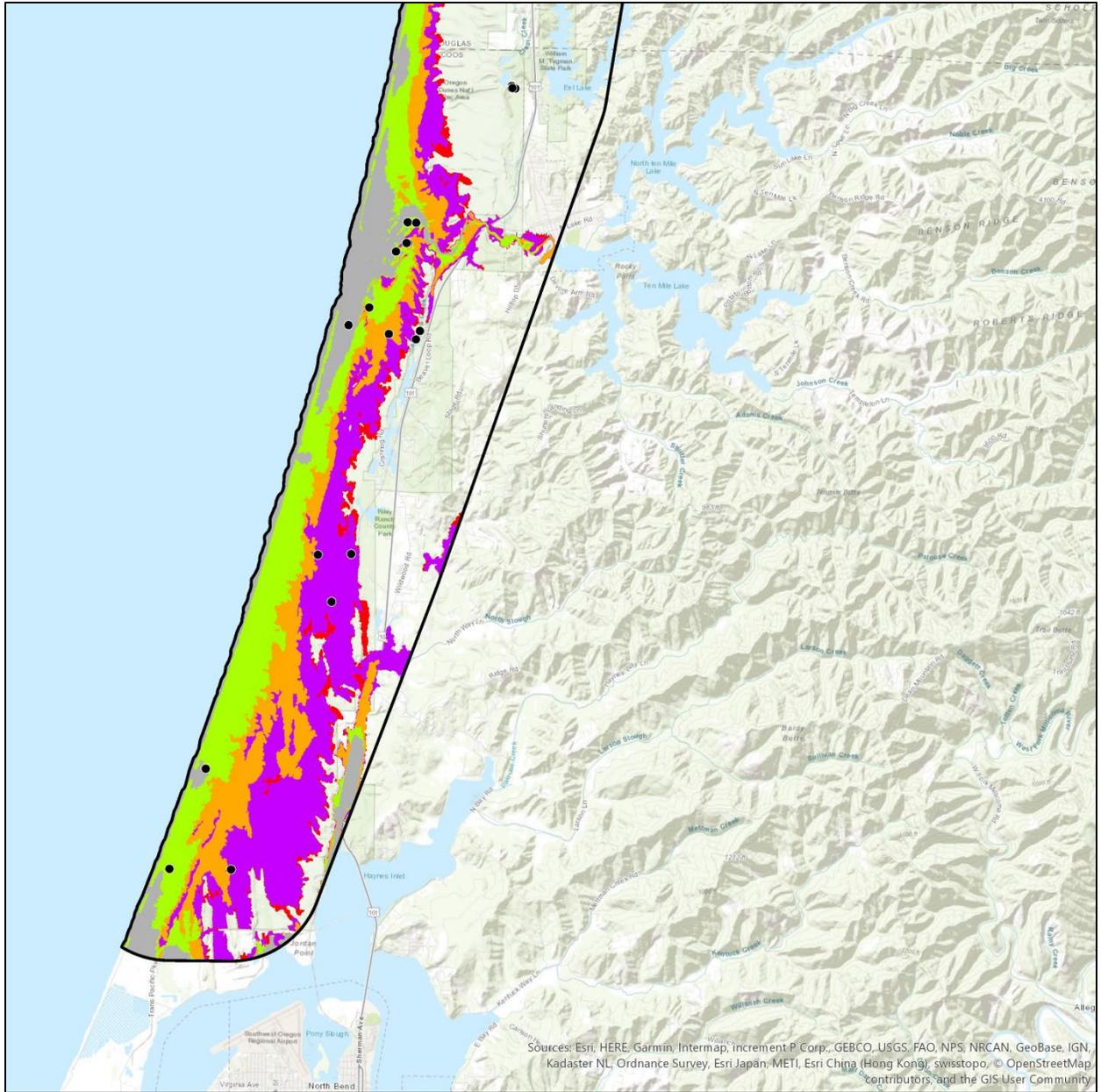
Figure 3.4. Tsunami inundation map representing five tsunami scenarios (Priest *et al.* 2013, entire) within the southern portion of the central coastal Oregon marten population. Color bands represent different inundation levels based on multiple subduction and earthquake metrics, including earthquake magnitude.

Chapter 4 Current Conditions

In this chapter, we consider the coastal marten's historical distribution, its current distribution, and what the species needs for viability. We first review the historical information on the range and distribution of the species. We next review the ecological needs of the species at the individual, population, and species levels. We analyzed the best available information in order to describe population resiliency, redundancy, and representation (3Rs). Finally, we use the concept of the 3Rs to describe the current conditions of coastal marten populations.

4.1 Historical Range and Distribution (pre–1980)

All available verifiable marten detections in coastal Oregon and California through 1998 were compiled and described by Zielinski and others (2001, entire). Verifiable detections include records with physical evidence such as tracks, photographs, and carcasses (mainly roadkill mortalities and martens trapped for fur), and reliable reports of martens captured and released by trappers. The dividing year between historical and current marten records we chose is 1980, primarily because all verifiable detections at that time in central coastal Oregon (in this case in the form of road kills) were from 1980 to 1998. Earlier verifiable detections exist from the 1940s (trapping records), but there are no verifiable detections from 1946 through 1996. Declining numbers led to the closure of the trapping season in California in 1946 and it was presumed that coastal martens were either very rare or extinct until 1996. Unverifiable marten detections, mainly visual observations from inexperienced observers, have occurred within the analysis area from the 1950s to 1980, but were excluded from Zielinski et al.'s (2001, entire) compilation and this report.

To delineate the historical distribution of martens in coastal Oregon and northern coastal California we used locations of verifiable detections (museum specimens, trapping records) and pre–1950s published interviews with trappers, that were collected during the late 19th century and the early and mid–20th century (Zielinski *et al.* 2001, p. 480). We modified the historical range boundary in northern coastal California to exclude several historical trapping records in eastern Mendocino County and northern Lake County because of imprecise locational data or information indicating currently unsuitable habitat conditions (i.e., dry, interior forest with inadequate shrub cover) for the coastal marten based on recent field site visits (Zielinski, unpublished data). In addition, trapping records in northern coastal Oregon counties were used to delineate the historical range boundary in coastal Oregon, but were excluded from verifiable detection maps since they did not contain precise locational data (i.e., only to the nearest county) (Figure 8.1; Zielinski *et al.* 2001, p. 480; Slauson *et al.* 2019(a), entire).

Within northern coastal California, the currently described coastal marten subspecies was historically known to occur within 80 km (50 mi) of the coast, "north through the humid coast belt from the vicinity of old Fort Ross, Sonoma County, into Oregon" (Grinnell and Dixon 1926, p. 413). The historical range of the coastal marten subspecies in California was described on the

basis of 43 martens trapped at 14 locations from 1919 to 1924 (Figure 8.1; Grinnell *et al.* 1937, p. 210). Twining and Hensley (1947, pp. 133, 136) described the range of the coastal marten subspecies as the northern coast ranges of California, noting that coastal martens had not been trapped in Lake or Sonoma counties for many years and records were scarce for Mendocino County. Hemphill (1952, p. 146) stated that martens still occurred on the Mendocino National Forest (which stretches north–south along the eastern boundary of Mendocino County, the northern portion of Lake County, as well as other inland counties) as of 1948, but provided no data to support his assertion.

Of the 24 historical verifiable detections (trapping records and museum specimens) of coastal marten in northern coastal California that contained precise locational data, 20 (83 percent) occurred in redwood and Douglas–fir forests less than 25 km (15 mi) from the coast and no records occurred greater than 35 km (22 mi) from the coast (Figure 4.1; Slauson and Zielinski 2007, p. 241).

The historical distribution of martens in coastal Oregon includes "…the humid mixed zones of the coast and Coast Ranges" (Bailey 1936, p. 296), extending from the Columbia River south through the coastal portions of the Klamath–Siskiyou mountains to the California border (Figure 3.1; Zielinski *et al.* 2001, p. 480). Museum or trapping records for martens exist for every county within their historical range in coastal Oregon (Marshall 1994, p. 1; Zielinski *et al.* 2001, p. 483). Some counties do not have detection locations assigned on the map in Figure 4.1 because the location information was unavailable. These county names are in bold. In northern coastal Oregon, historical records are sparse, but early trapping records verify occurrence of martens in Benton, Clatsop, Columbia, Lincoln, Polk, Tillamook, Washington, and Yamhill counties (Zielinski *et al.* 2001, p. 483). Similar to the pattern of distribution of northern coastal California historical records, historical records of marten in coastal Oregon predominantly (>90 percent) occur closer to the coast than in the interior portions of the historical range (Figures 2.2 and 4.1).
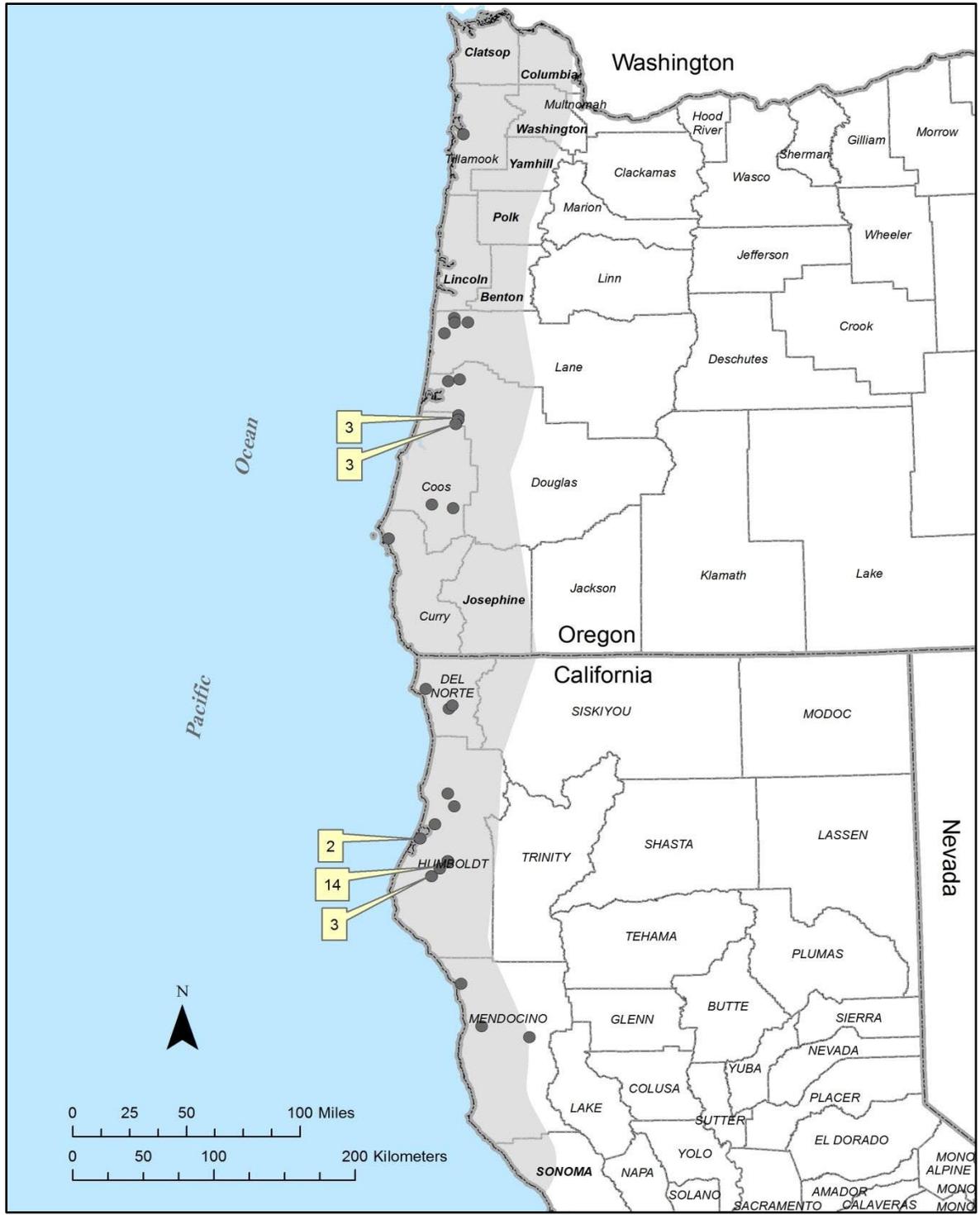
Figure 4.1. Historical (pre–1980) verifiable marten detections (dark gray circles) within the historical range (light gray shading) of the coastal marten. Each circle represents a single detection unless otherwise noted (number of detections in boxes). Counties with bold labels have one or more historical trapping and\or museum records, but lacked precise locational data for mapping.

4.2 Current Range and Distribution (1980–current)

All current (since 1980) verifiable marten detections were used to delineate extant population areas (EPAs) within the historical home range. The number of detections available to guide the delineation of the boundaries of the EPAs varied across the analysis area (Figure 4.2). In addition, sampling techniques varied across the range. Marten detections were buffered by 2 km and connected using a minimum convex polygon tool. Similar to methods used in the Humboldt Marten Conservation Strategy and Assessment, a 2 km buffer distance was used because most coastal marten survey and monitoring grids use a 2–km grid spacing, thus to feel confident about where animals do not occur, one would need to survey the next grid point without detections. If the total number of detections in an area was less than five or they were separated by greater than 5 km from other verifiable detections, the combined detections were not designated as an EPA due to the insufficient level of information to suggest a likely self–sustaining population (Slauson et al*.,* 2019, entire). Because some detections did not meet this definition of a population, they appear on Figure 4.3 as points but are not included in the population areas.

Based on the distributions of current verifiable marten detections and adjacent suitable habitat, we identified four EPAs within coastal Oregon and northern coastal California (Figures 4.3):

1) Central Coastal Oregon Extant Population Area (CCO_EPA)
2) Southern Coastal Oregon Extant Population Area (SCO_EPA)
3) Oregon–California Border Extant Population Area (CAOR_EPA)
4) Northern Coastal California Extant Population Area (NCC_EPA)
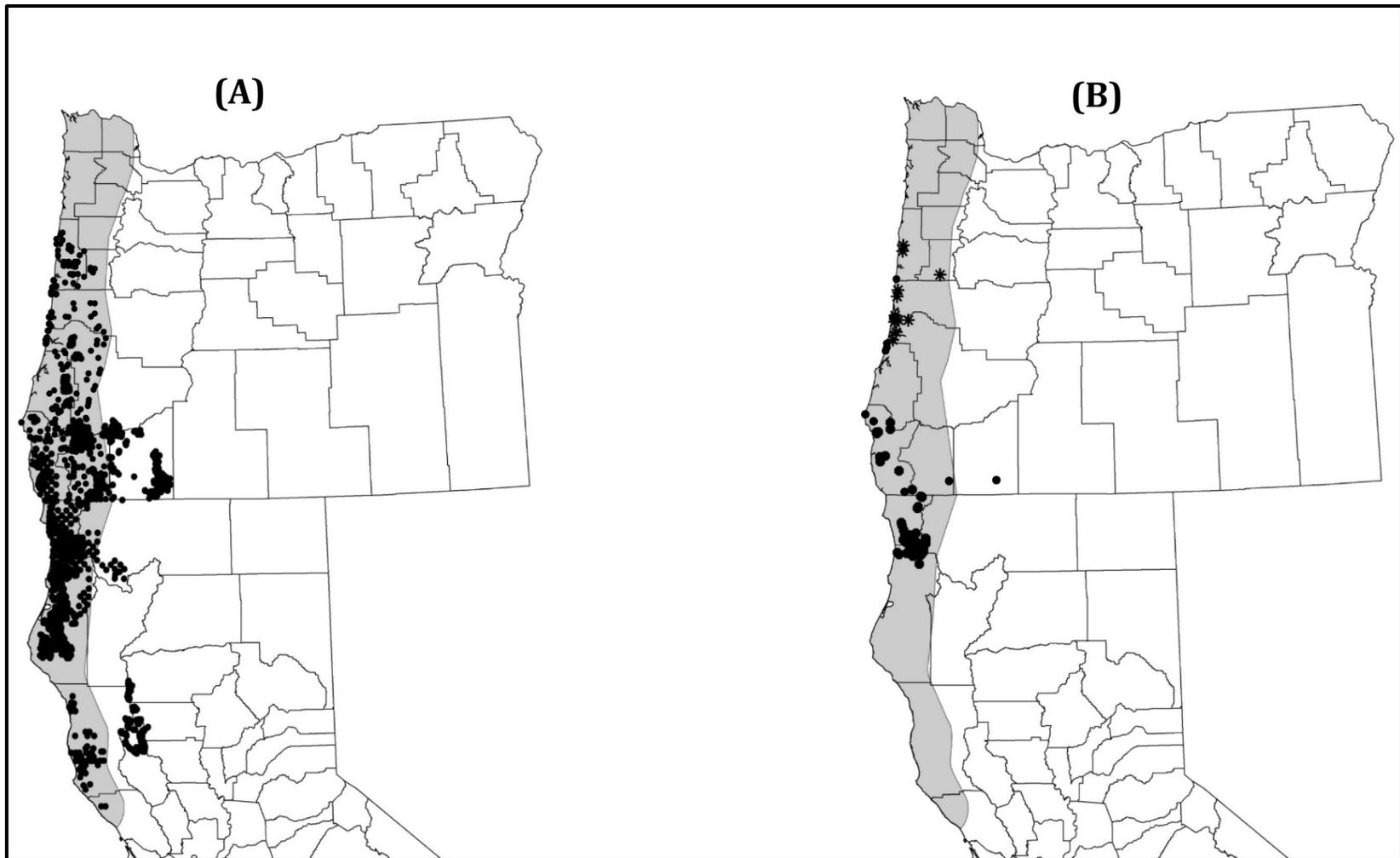
Figure 4.2 Distribution of contemporary survey effort (A), and contemporary verifiable coastal marten (*Martes caurina humboldtensis*) detections (B) from surveys (circles), and roadkill mortalities (n = 14) or trapped (n = 3) individuals (asterisks; central coastal Oregon only) in the assessment area. The map does not depict more recent surveys or detections in Oregon, but delineation of population areas considered these records. Source: Slauson et al.. 2019.
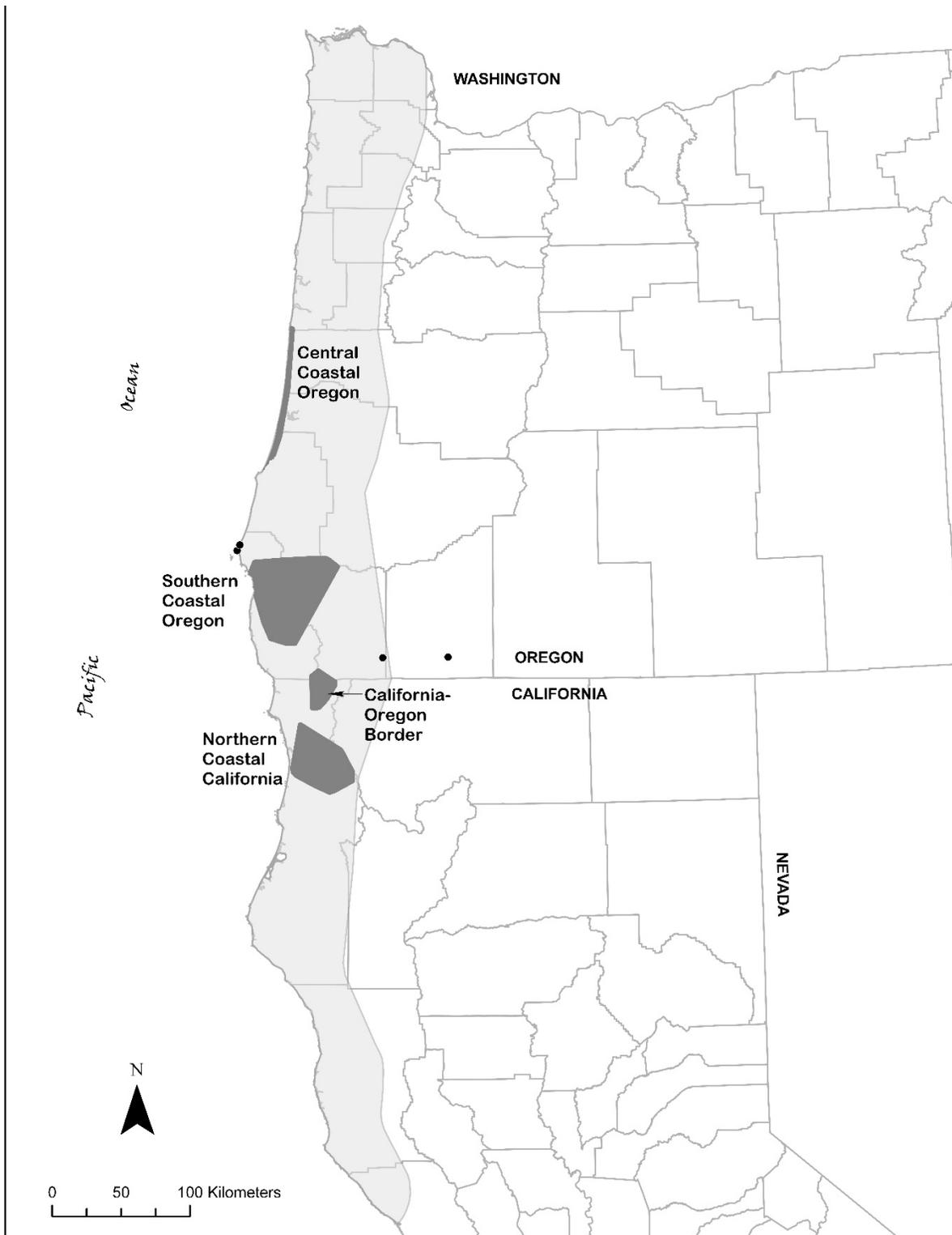
Figure 4.3 Dark shaded areas are the four current populations based on survey effort and verifiable marten detections from Figure 4.2 and updated information. Points indicate detections that do not constitute a population. Light gray is the historical range of coastal marten.

Figure 4.4 Central coastal Oregon population of coastal marten.
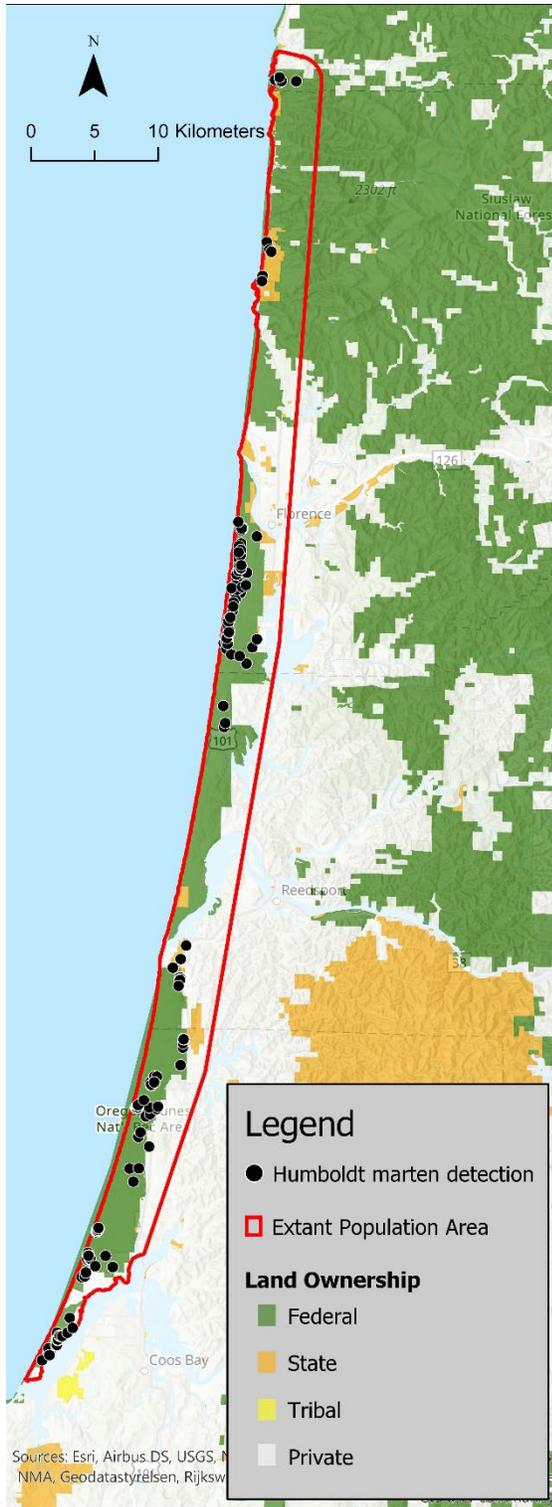
Moving north to south, these four populations are described in more detail below.

4.2.1 Central Coastal Oregon population

The Central Coastal Oregon population occupies 403 km^2 in shore–pine dominated forest in the coastal dunes ecosystem in the Oregon Dunes Recreational Area, and is managed by the Siuslaw National Forest (Figure 4.4). The area occupied is <500 meters wide in coastal forest that is <70 years old. Until recent survey efforts increased it was assumed that animals were abundant because of a relatively high incidence of road kills in the past 30 years (*n* = 17, Zielinski *et al.* 2001, p. 487; Moriarty *et al.* 2016b, p. 78). Recent survey efforts have refined the extent of this population and estimated the population size at approximately 71 adults (Linnell *et al.* 2018, p. 1). There is no information at this time on long–term trends in population size. This population is divided into two subpopulations of ~30 adults, separated by the Umpqua River, a relatively large barrier to movement and dispersal. Martens in this population occur in the highest densities reported for any North American marten subspecies (1.13 per km^2; Linnell *et al.* 2018, p. 10).

4.2.2 Southern Coastal Oregon population

The Southern Coastal Oregon population occupies 2420 km^2 in Coos and Curry counties (Figure 4.5). Land management is a mix of Cape Blanco State Park, Siskiyou National Forest, Coos Bay BLM, Medford BLM, and private lands, including some private industrial timberlands. Of all four populations, this one has had the least amount of survey or research effort. However, given recent survey results it is unlikely there are >100 animals in this population (Slauson 2018, personal communication; Moriarty 2018, pers. comm.) and there were fewer individuals documented than expected given survey efforts (Moriarty *et al.* 2016b, pg. 77). The majority of detections occurred on the Rogue River–Siskiyou National Forest with a single detection in Cape Blanco State Park (Moriarty *et al.* 2016b, p. 73). The occupied area is bisected by the Rogue River, which likely acts as a strong barrier to movement within the population, potentially creating two subpopulations similar to the Central Coastal Oregon EPA. There are no documented cases of animals crossing the Rogue River, although martens have been documented crossing the Klamath River, which is similar in size to the Rogue (see Section 4.4.1 Population Resiliency).
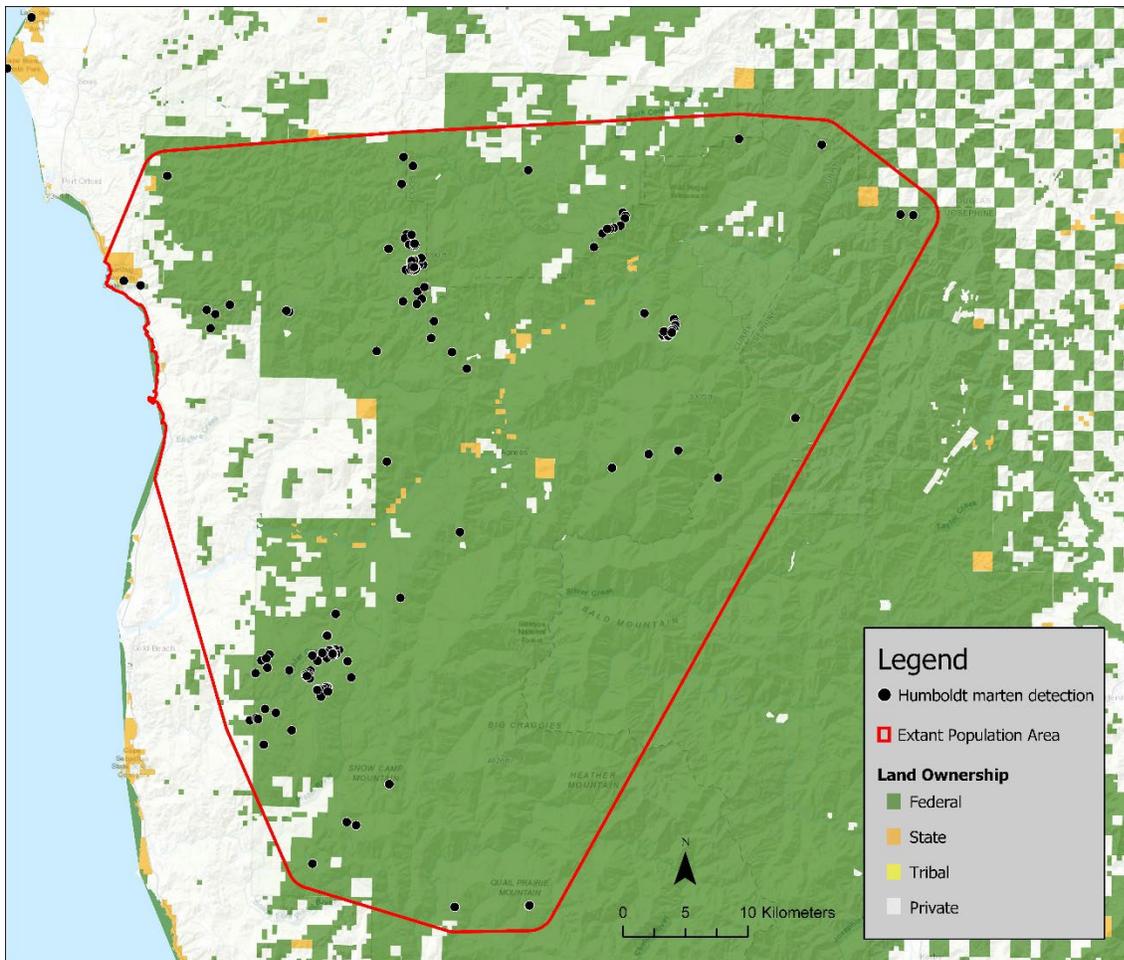
Figure 4.5 Southern coastal Oregon population of coastal marten.

4.2.3 California–Oregon Border Population

The California–Oregon Border population occupies 206 km^2 in northeastern Del Norte County (Figure 4.6). Land management is a combination of Six Rivers National Forest in the Smith River National Recreation Area and private land, including private timber operations at the northern edge of the currently occupied boundary (Table 4.1). The first detection in this population was in 2011 and there are ongoing surveys in an effort to refine the population boundaries. While current surveys have detected at least 12 individuals there is no current population estimate. The occupied area is small and supports <100 animals (Slauson 2018, personal communication).
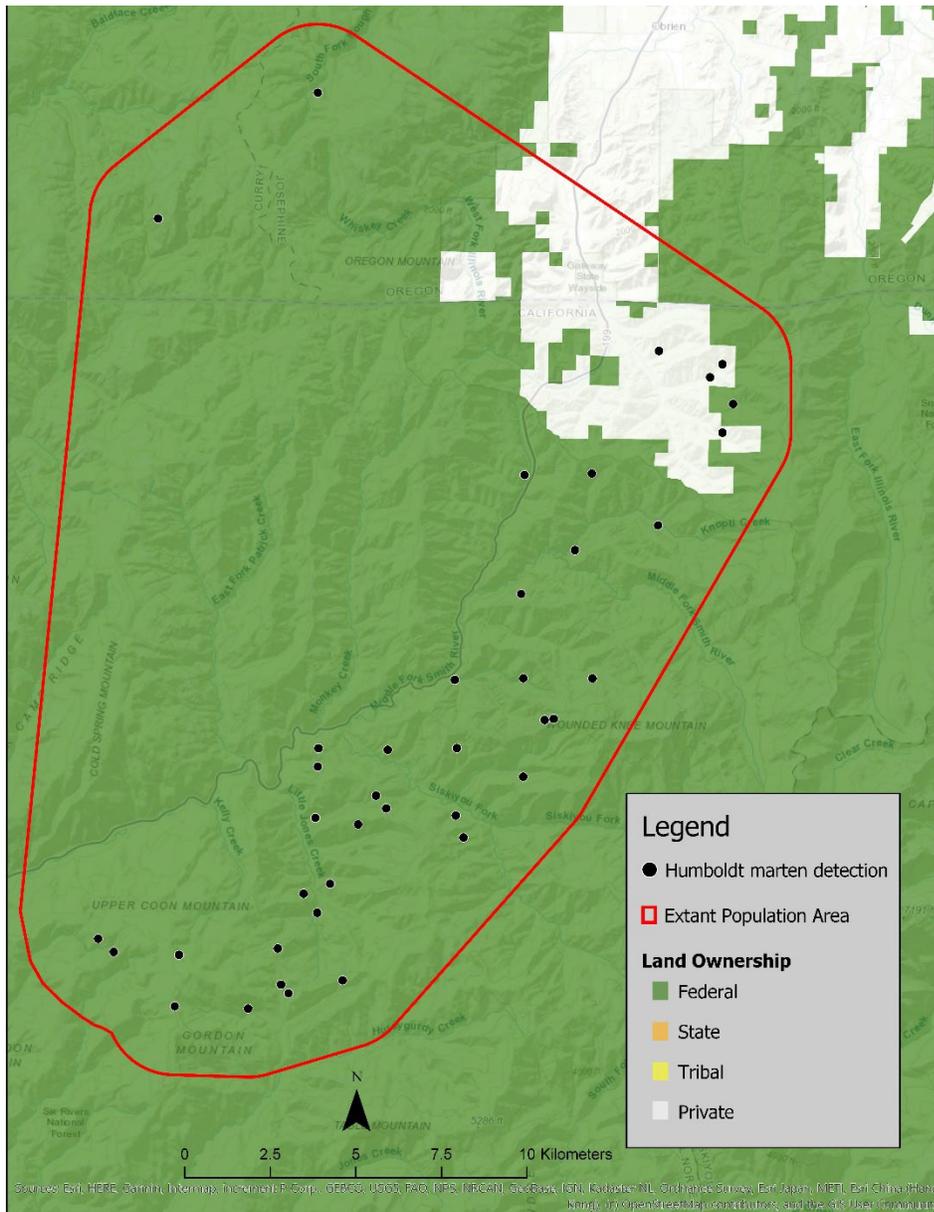
Figure 4.6 California–Oregon border population of coastal marten.

4.2.4 Northern Coastal California population

The Northern Coastal California population occupies 1170 km^2 (Figure 4.7). This population has been surveyed more than any other extant population and much of what is known about coastal martens has been gathered from this population. Coastal martens were considered extremely rare or extinct in this portion of their range until 1996 when a single population was rediscovered in this EPA. The most recent population estimate was conducted in 2012 and it was estimated that there were between 60–80 individuals. This was a decrease from a 2008 population estimate that calculated >100 animals in this area. Land management in this population is a combination of Green Diamond Resource Company (a private timber operation), Klamath National Forest, Six Rivers National Forest, and the Yurok Tribe (Table 4.1).
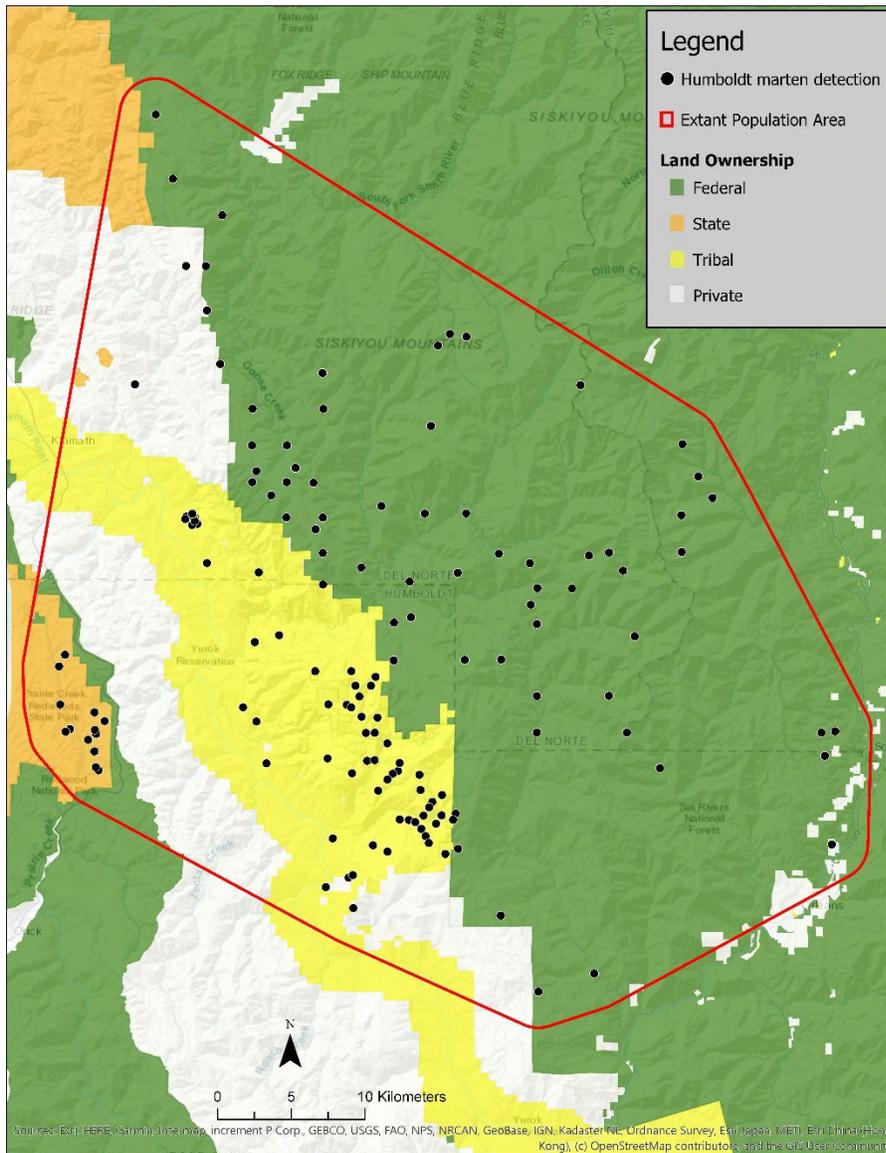
Figure 4.7 Northern coastal California population of coastal marten.

4.2.5 Additional detections and areas of presumed former populations

Additional detections have been made outside of the current EPA's but were excluded from our current population boundaries because they did not meet the criteria of a population described above in Section 4.2 (see Figure 4.3). There have been 11 detections of marten in Prairie Creek Redwoods State Park from 2009 to 2017. However, these detections are thought to represent only two individuals. In addition, they were approximately 10 km (6.2 mi) from the nearest population to the east (NCC_EPA) and were insufficient in number to be considered a population. One of these individuals was marked and is known to have come from the NCC_EPA, and since it was a juvenile male, it was presumed to be dispersing from its natal range. In addition, little suitable marten habitat exists between these marten records and the NCC_EPA to the east. In Jackson County Oregon, two marten locations did not meet our distance criteria for inclusion into the southern coastal Oregon population (see Section 4.2 Current Range and Distribution (1980–current)). One location in the Applegate Valley recorded three marten detections over multiple years (Clayton 2018, pers. comm.); another detection occurred on Mt. Ashland, in the eastern part of Jackson County and well outside the historical range. Because of increased genetic samples, this marten is considered most closely related to the coastal marten clade (K. Pilgrim 2018, pers. comm.). Extensive camera surveys as well as live trapping for fishers in these areas over the past 4–8 years has yielded no other marten detections. There are also several areas of suitable habitat within the historical range that, based on extensive surveys, are not currently occupied (Figures 4.8).

Table 4.1. Amount (km^2) and percent of total area of coastal marten habitat within each extant population area by suitability class and land ownership. Values for the entire analysis area include all four extant population areas and all areas within the historical range of the coastal marten outside the four EPAs. All values rounded to the nearest whole number.

| Extant population area | Land owner | | | | | | | | Total area (km^2) |
| | Federal | | State | | Private | | Tribal | | |
	Area (km^2)	Percent of total area	Area (km^2)	Percent of total area	Area (km^2)	Percent of total area	Area (km^2)	Percent of total area	
CCO	243	60	23	6	141	34	–	–	407
SCO	1,689	69	33	1	705	29	<1	<1	2,428
CAOR	186	90	–	–	20	10	–	–	206
NCC	824	70	9	<1	105	9	232	20	1,170
Entire historical range	18,579	32	5,010	9	33,077	58	822	1	57,488

4.3 Landscape Habitat Suitability

To assist in our analysis of habitat suitability for the coastal marten, we used a landscape habitat suitability model developed using data from detections of coastal marten in California (Slauson *et al.* 2019, entire) (Figure 4.8). The model was developed by identifying the combination of environmental, topographic, disturbance history, and vegetation variables that best described the distribution of marten detection and non–detection survey data. The top models (i.e., those candidate models that best "fit" the data set) contained a combination of four base variables: (1) old–growth structural index, (2) serpentine habitat, (3) precipitation, and (4) adjusted elevation (Slauson *et al.* 2019(b), pp. 38–41). Less important variables included recent timber harvest and stream density. The model did not identify the coastal shore–pine and ericaceous shrub habitat that the central coastal Oregon population currently occupies and though that habitat type occurs in less than three percent of the historical range, caution should be exercised when applying this model to the entire range. Furthermore, an Oregon–specific marten habitat analysis indicates that while coastal martens in Oregon used older forests similar to coastal martens in California, they are also often found in forests with fewer and smaller diameter trees as long as combined overstory and understory cover remain high (Moriarty et al., 2019, entire). Further work needs to be done to understand habitat requirements for coastal martens and the degree to which the California coastal marten model (Slauson *et al.* 2019(b), entire) represents suitable marten habitat in coastal Oregon. In the interim, we have chosen to use the model developed by Slauson and others because it is seamless across the coastal marten historical range and because of available documentation.

The final averaged model reveals three significant regional habitat areas: two in Oregon, one on the central coast and one on the southern coast, and one in northwestern California that supports the only known populations for the state (Figure 3.3). The model predicts some potential habitat connectivity between the northern coastal California and southern coastal Oregon populations, but suggests there is little habitat connectivity between the central and southern coastal Oregon populations.

The old–growth structural index (OGSI) is a composite index based on stand age, number of large trees, diversity of tree sizes, density of large snags, and volume of large snags (Spies *et al.* 2007, p. 10). The OGSI was one of the four variables included in all top landscape habitat suitability models (Slauson *et al.* 2019(b), pp. 39–41). The OGSI helps to indicate areas that have strong potential for future marten reproduction.

The habitat suitability map (Figure 4.8) derived from model results is based on a suitability index scale from 0 = "not habitat" to 1 = "habitat of high suitability", based on the probability of coastal marten occurrence. The habitat suitability index was split into three suitability classes or bins (unsuitable, suitable OGSI, suitable serpentine) representing the relative probability of

marten occurrence at the landscape scale. The suitability classes are based on prevalence (proportion of detections relative to total survey locations).
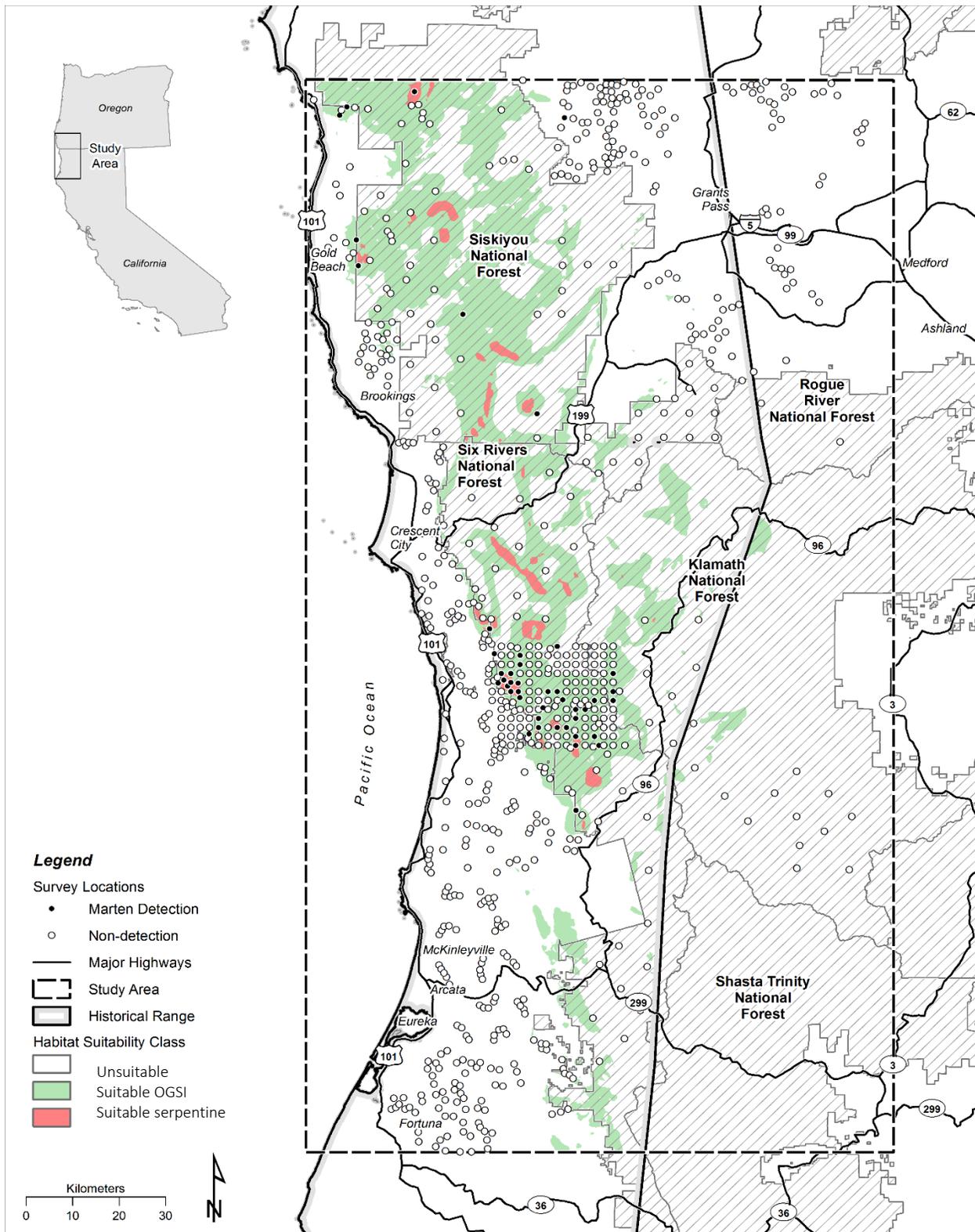
Figure 4.8 Current landscape habitat suitability for coastal martens throughout their historical range (Slauson *et al*. 2019(b); p. 44). Coastal dune forest habitat in northern Coos County, Douglas County, and Lane County, Oregon are considered suitable, but were excluded by the model.

4.4 Needs of Coastal Marten

As discussed in Chapter 1, for the purpose of this assessment, we define **viability** as the ability of the species to sustain populations in the wild over time. Using the SSA framework, we describe the species' viability by characterizing the status of the species in terms of its **resiliency**, **redundancy**, and **representation** (the 3 Rs). Using various time frames and the current and projected levels of the 3 Rs, we thereby describe the species' level of viability currently and into the future.

4.4.1 Population Resiliency

For the coastal marten to maintain viability, its populations or some portion thereof must be resilient to the ongoing stressors and stochastic events that the species faces in the wild. Stressors affecting coastal marten populations are described in Chapter 3 and include habitat loss from fire and vegetation management, disease, predation, trapping, rodenticide exposure, and mortality from vehicular collisions. Examples of stochastic events, or those that cannot be reliably predicted, that can occur in the historical range based on the stressors that were evaluated, include uncharacteristically large fire events, disease outbreaks, and tsunamis.

A number of factors influence the resiliency of populations, including habitat quality (habitat that allows for increased predation) and quantity (number of potential home ranges) and connectivity between populations. These include between–population and within–population factors. We quantified these factors for each population in an attempt to characterize their resiliency. Factors are discussed below and summarized in Tables 4.2 and 4.3.

Between Population Factors

Distance between populations – Dispersal is the means by which marten populations maintain and expand their distribution. Successful dispersal requires the existence of functional habitat connectivity between patches of habitat suitable for reproduction to maintain or expand population size and distribution. Dispersal is also essential to maintain viable meta–populations. Because a Euclidean (straight–line) distance does not account for habitat suitability (or functional connectivity), we calculated a least–cost path distance between populations using patches of suitable habitat from the landscape suitability model described in Section 4.3 (Slauson *et al.* 2019(b), entire). For more information on dispersal needs for individuals, refer back to Section 2.4.3 **Dispersal**.

A resilient coastal marten population would have suitable habitat between populations that provides important habitat for key prey, abundant daily resting sites, and a maximum distance within the range of average dispersal distance. Based on studies of dispersal distances described in Section 2.4.3 of coastal martens and other marten species, and using the least–cost path distance described above, we assigned the following categories to describe population resiliency in terms of dispersal ability. Highly resilient populations would be within 15 km of the closest population, which is within the range of dispersal events described for martens. Moderately

resilient populations would be between 16–45 km of the next closest population, equivalent to up to 3 times the average dispersal distance described for North American martens. Any populations that were >45 km from another population would be considered low resiliency.

Filters between populations – Filters are barriers to movement and can be either natural or manmade. The following two paragraphs describing filters and their effect on movement are excerpted from Schrott and Shinn 2018 (unpublished, pp. 3–4). Major roads can act as barriers to movement for many wildlife species through the disruption of habitat, risk of mortality from being struck by vehicles, and a reluctance to approach an area with the noise and disturbance of traffic (Forman and Alexander 1998, pp. 212–216). While there is little direct data concerning the impacts of roads on coastal marten movement, studies of populations of American marten have found that roads can reduce habitat quality, as evinced by reduced activity near logged roads in Ontario (Robitaille and Aubry 2000, pp. 140–141) and reduced detection rates with increasing road density in Idaho (Wasserman et al, 2012a, pp. 5–9). High–traffic, multi–lane highways have been shown to be significant barriers to movement of American martens (Alexander and Waters 2000, pp. 312–314), although some types of culverts can be used by martens to pass under such highways (Clevenger *et al.* 2001, pp. 1343–1345).

While martens probably make regular crossing of smaller streams, wider rivers with high flow volumes and swift currents almost certainly act as significant barriers to movement. However, the detection of an ear–tagged coastal marten in Prairie Creek Redwoods State Park (PCRSP) in 2017 that had dispersed from the Northern Coastal California population across the Klamath River to the east indicates that even the largest rivers in this region are permeable barriers. Based on ongoing surveys in PCRSP one marten per generation (5 years) travels across the Klamath River. Natal dispersal in coastal martens can take place over much of the year (beginning in early August and continuing for up to four months) (Johnson 2008, pp. 32–33; Slauson 2017, p. 112), meaning that the degree of difficulty in crossing large rivers would also vary considerably between dispersal events based on flow volume of the river (highest in spring, lowest in early autumn, with more variability in drought years).

Ideally, resilient marten populations would not be separated by more than one strong filter between populations as these filters limit dispersal ability, which ultimately affects recolonization, and gene flow. For our analysis, we assumed that highly resilient marten populations would not be separated by any strong filters (defined as large rivers or multi–lane highways), moderately resilient populations would only have one filter present, and low resiliency populations would have two or more filters.

Number of populations in proximity – The number of populations in proximity to one another (measured using the least–cost path distance) is important when considering resiliency to stochastic and catastrophic events. If a wildfire, disease outbreak, or other catastrophic event were to significantly decrease or functionally eliminate one of the EPA's, there would be fewer individuals to serve as a source population or fewer areas to serve as a refuge for subsequent

stochastic events. If there were more than one population nearby there would be a higher chance for recolonization after a catastrophic event.

For our analysis, we assumed that highly resilient populations would have at least two populations in proximity, moderately resilient populations would have one population in proximity, and low resiliency populations would have zero populations in proximity. Populations 6–45 km away would be considered as "in proximity."

Within Population Factors

Population Size – Population size (abundance) is a metric commonly used to assess population viability and is determined by changes in birth and death rates. Population size is influenced by biotic factors (such as food, disease, competitors, and predators) and abiotic factors (such as rainfall, fire, and temperature). Factors that affect population dynamics, including demographic shifts, environmental uncertainty, natural catastrophes, and random genetic changes, have a larger effect with decreasing population size (Shaffer 1981, p. 131). Minimum viable population (MVP) size, the smallest number of individuals in a species capable of persisting for a predetermined amount of time, is a concept that has received a lot of attention by researchers, remains controversial, and has had several different thresholds proposed. For example, in reviews of many vertebrate taxa, the range of minimum population sizes varies from the many hundreds to a few thousand (Boitani and Powell 2012, p. 365, Traill *et al.* 2007, p. 159).

While no studies have analyzed coastal marten MVP specifically, several studies have looked at other marten species to determine MVP under various criteria and with varying recommendations. One study suggested >6,000 American martens range wide would be needed to ensure a 99 percent probability of persistence for 100 years (Reed *et al.* 2003, p. 32). Another study of American martens found an initial population size (range–wide) of 1,200 was required for a 90 percent probability of persistence over 100 years (O'Grady *et al.* 2008, p. 446). The most robust assessment of marten population viability analysis (PVA) to date was completed by Lacy and Clark in 1993 (entire) and is summarized here. This PVA of American martens analyzed various combinations of timber harvest and commercial trapping, initial population sizes of 50 or 100, and immigrant rates of 0, 2, or 20. Without immigration, only the most optimistic management scenario (no timber harvest or trapping and an initial population size of 100) had any probability of persistence (66 percent) over 100 years. Additionally, for populations in landscapes subject to timber harvest but no trapping, with an initial size of 100, receiving 2 immigrants per year, and no stochastic events, the probability of persistence over 100 years was 100 percent; however, the population had declined by over 50 percent after 100 years (see Figure 1 in Lacy and Clark 1993, p. 286). The authors concluded that well over 100 individuals were needed to maintain marten populations, especially in populations that had relatively low rates of immigration and emigration. It is important to note that when this model was created it was difficult to accurately model demographic and environmental stochastic events and catastrophic events and application of these conclusions should reflect that. The only

PVA to analyze coastal marten was conducted on the Central Coastal Oregon population. The results of the study suggests that even a small amount of human–caused mortalities (road–kills, trapping) will greatly increase the probability of extirpation over the next 30 years. This is despite the researchers' acknowledgement of assuming relatively high survival and fecundity rates and not fully considering environmental stochasticity or potential catastrophic events in their analysis (Linnell *et al.* 2018, pp. 14–15).

Comparing these studies and their recommendations for minimum viable population sizes to what we know about current coastal marten populations, connectivity between those populations, and the stressors currently acting on coastal martens, we established what we think are conservative estimates for minimum viable population sizes. Considering that 100 individuals without stochastic events or any vegetation management is not secure, and using the lower estimates from minimum viable population analyses, we determined that populations with >300 animals would be considered highly resilient. Furthermore, populations ranging from 100–300 individuals are considered moderately resilient and populations with <100 individuals demonstrate low resiliency.

Number of available male home ranges

Male and female martens have different home range requirements that vary in size and number in each population. The number of male home ranges or territories available is a function of the size of suitable habitat patches. This metric, combined with potential female territories (described below), is an indication of the carrying capacity of the available habitat. The number of male territories was calculated using the amount of suitable habitat from the habitat suitability model described in Section 3.3 and dividing by the average size of a male marten home range (3–4 km^2) for the three southern populations. While there have been coastal marten detections outside of the suitable habitat, the detections were always within 2 km of suitable habitat, therefore for this analysis we assume that the unsuitable category generally cannot support entire home ranges (Slauson, pers comm). In addition, the habitat suitability model performed poorly for the CCO EPA, based on the range of training data used to create the model and the lack of older forest in the shore-pine dominated forest, so the number of potential male home ranges was based on the territory mapping analysis in Linnell *et al.* (2018, entire). The categories of high, moderate, and low resiliency are a function of the overall population size categories described in the "Population size" section above.

Number of available female home ranges

Female martens select home ranges that have a higher proportion of higher quality habitat based on the energetic demands of reproduction and lactation. In the three southern populations, 38 percent of serpentine and 68 percent of forest habitats are considered to contain the higher quality habitat associated with female home ranges and as a result are suitable for reproduction (Slauson pers. comm.). The number of potential female territories was calculated for the three

southern populations in a similar manner to male home ranges with the corrections for what proportions of serpentine and late–seral forests are considered suitable for reproduction. Similar to our calculation of male home ranges, we did not use the habitat suitability model for the CCO EPA and instead relied on the territory mapping analysis in Linnell *et al.* (2018, entire). The categories of high, moderate, and low resiliency are a function of the overall population size categories described in the "Population size" section above.

Human–influenced predation risk

As noted in the discussion on carnivore community distribution in Section 3.1.4, coastal martens are vulnerable to predation in habitats that have been subject to either high–moderate severity fires or logging in the last 30 years since both of these events increase the abundance of predators and remove the late–seral characteristics of the landscape that allow marten to avoid predators (canopy cover, shrub cover, etc.). In a review of multiple habitat fragmentation studies, a threshold of 30 percent habitat loss was a good predictor of effects to species richness and population size (Andren 1994, p. 359). To represent the effect of increased fragmentation and increased predation risk on marten resiliency, we characterized high resiliency populations as those with >70 percent of the landscape that was unaffected by fire or vegetation management in the past 30 years since martens do not generally exhibit higher predation or lower survival rates until landscapes are fragmented below 70 percent (Potvin and Breton 1997, p. 467; Thompson 1994, pp. 276, 278–279). Furthermore, we considered populations moderately resilient when 30–70 percent of the forest was older than 30 years and because North American marten species tend to avoid open areas (see 2.5.3 Habitat Use). We characterized low resiliency populations as containing <30 percent forest older than 30 years (Spencer *et al.* 1983, pp. 1183).

The areas within each EPA subject to fire or vegetation management was calculated using a disturbance tool based on data from the GNN Lemna lab. Trends in OGSI–80 values since 1993, as well as amount of specific disturbances affecting those values, are available from Davis *et al.* (2015, pp. 30–33) at the physiographic province scale (see Davis *et al.* 2015, p. 10 for map of physiographic provinces). These data also allowed us to calculate the amount of OGSI–80 that was a result of ingrowth over the 20–year monitoring period. We used existing GNN data (LEMMA 2018, entire) to scale the OGSI–80 data to the individual marten population areas as well as the entire historical range to analyze the areas within each EPA subject to fire or vegetation management to assess the human–influenced predation risk.

4.4.2 Species Representation

Maintaining representation in the form of genetic or ecological diversity is important to maintain the coastal marten's capacity to adapt to future environmental changes. Carnivorous mammals, like coastal martens, need to retain populations throughout their range to maintain the overall potential genetic and life history attributes that can buffer the species' response to environmental changes over time (Frankham and Ralls 1998, p. 441–442). Genetic variability is lost from small populations due to drift and inbreeding which can lead to declines in fecundity and survival

(Lacy and Clark 1993, p. 283). It is unknown how much genetic variation was historically present among the coastal marten populations, but we expect additional variation was present in populations and the coastal marten has likely lost genetic diversity as populations have been extirpated or become more isolated. At a minimum, maintaining the remaining representation in the form of genetic diversity will be important to the capacity of the coastal marten to adapt to future environmental change.

Historically, the coastal marten predominantly occupied late–seral forests with interspersed areas of serpentine habitat. Currently, the coastal marten occupies three distinct habitat types – older forests, areas characterized by serpentine soils, and shore pine forests found in coastal margins and dune ecosystems. It is unknown how long coastal martens have been occupying shore pine forests. While most of the shore pine stands currently occupied by martens are <70 years old and are a consequence of sand dune stabilization through introduction of *Ammophila arenaria* (European beachgrass), historical vegetation maps from the 1930s show that shore pine forests stretched along much of the inland edge of the dune and beach areas in central Oregon (Williams 2017, *pers. comm.*). Our understanding of successional patterns in coastal shore pine forests indicates that the dense shrub communities favored by martens is a consistent feature across different moisture conditions (Franklin and Dyrness 1988, pp. 291–294) and likely provided historical marten habitat. However, much of these areas appear to have been fragmented or lost because of development of coastal communities, perhaps pushing martens further seaward to more recently established shore pine forests. This shore pine habitat is less susceptible to fire and does not support stands that would be subject to management for timber. The remaining three populations face these two stressors. Maintaining populations in all three habitat types will contribute to increased species representation.

4.4.3 Species Redundancy

The coastal marten needs to have multiple resilient populations distributed throughout its range to provide for redundancy. The more populations, and the wider the distribution of those populations, the more redundancy the species exhibits. Redundancy reduces the risk that a large portion of the species' range will be negatively affected by a catastrophic natural or anthropogenic event at a given point in time. Species that are well distributed across their range are considered less susceptible to the impacts from threats and more likely to withstand stochastic and catastrophic events than species confined to a small portion of their range (Carroll *et al.* 2010, entire; Redford et al 2011, entire).

4.5 Current Conditions

The available information indicates that the coastal marten is currently restricted to approximately 7.3 percent of its known historical range, which includes two populations in Oregon and two populations in California. The species has been extirpated from Sonoma and Mendocino Counties in CA, and largely from Humboldt County, Del Norte, and Siskiyou Counties in CA. In Oregon, coastal martens have been largely extirpated from much of the

inland counties within the historical range, and known to currently occur in Coos, Curry, Josephine, Douglas, Lane, and Lincoln Counties.

4.5.1 Current Population Resiliency

Methodology

To summarize the overall current conditions of coastal marten populations, we ranked the four populations into three categories (high, moderate, and low) based on the population factors and habitat elements discussed above (Tables 4.2 and 4.3) (see section 4.4.1 **Population Resiliency).** In addition to assigning each population factor and habitat elements a category the table contains the data we used to determine categories. We did not include in the table areas that are (or are presumed to be) extirpated. The current condition category is a qualitative estimate based on the analysis of the three between population factors (least–cost path distance, filters, and number of populations in proximity) and four within population factors (population size, available male home ranges, available female home ranges, and proportion of habitat suitable to predation) for each population.

	Between Population Factors			Within Population Factors			
	Distance between populations	Number of dispersal filters between populations	Number of populations in proximity (6–45 km)	Population Size	Number of available male home ranges	Number of available female home ranges	Proportion suitable habitat resilient to human–influenced predation
High	0–15 km	0	2	>300	>150	>150	>70%
Moderate	16–45 km	1	1	100–300	50–150	50–150	30–70%
Low	>45 km	>1	0	<100	<50	<50	<30%

Table 4.2 Description of factors used to create condition categories describing population resiliency in Table 4.3.

Population (Quantity of suitable habitat out of minimum convex polygon)	Between Population Factors			Within Population Factors				Overall Current Condition
	Least–cost path distance through suitable habitat	Number of filters	Number of populations in proximity (6–45 km)	Population Size	Number of available male home ranges	Number of available female home ranges	Proportion of suitable habitat that allows for predator avoidance	
Central Coastal Oregon 62 km²/ 403 km²	**Low** 201 km	**Low** >1	**Low** 0	**Low** 71	**Low** 30	**Low** 44	**Low** 15%	**Low**
Southern Coastal Oregon 1103 km²/ 2420 km²	**Low** 65 km	**Low** >1	**Low** 0	**Low** 12–<100	**High** 276–368	**High** 173–230	**Moderate** 65%	**Low**
CA–OR Border 56 km²/ 206 km²	**High** 14 km	**Moderate** 1	**Moderate** 1	**Low** 12–<100	**Low** 14–19	**Low** 7–9	**High** 82%	**Low–Moderate**
Northern Coastal CA 704 km²/ 1170 km²	**High** 14 km	**Moderate** 1	**Moderate** 1	**Low** 80–100	**High** 176–235	**Moderate** 96–128	**Moderate** 52%	**Moderate**

Table 4.3 Resiliency of coastal marten populations. Data used to assign categories are included for each population and each factor.

Overall, the extant populations occur in areas of relatively suitable habitat, but they vary in size and connectivity. The overall current condition is an average of each of the population factors with all factors being considered equal. The CCO EPA is considered to be in overall low condition based on its small size and extremely limited connectivity to any other EPA. While the SCO EPA has the potential to support a population large enough to improve resiliency, it is also considered to be in overall low condition based on a current small population size, limited connectivity to the CA–OR EPA, and limited habitat that allows for predator avoidance. The CA–OR Border EPA has better connectivity to the south than either of the Oregon populations but is in low to moderate current condition based largely on the small population size (<100 individuals) and its limited potential carrying capacity (58–78 home ranges available). The NCC EPA is in overall moderate condition due to relatively good connectivity to the CA–OR Border EPA and the relatively large number of available home ranges.

4.5.2 Current Species Representation

We consider the coastal marten to have representation in the form of two different ecological settings. There are animals that are adapted to the shore pine forest and others that are adapted to late–seral forest containing interspersed serpentine ridges. There is one population representing the shore pine ecological setting and three representing the forest and serpentine ecological settings.

Genetic variation between populations is unknown at this time, as no robust studies have been conducted to determine the degree of genetic variation between the four populations.

4.5.3 Current Species Redundancy

Within the identified areas of representation, the CCO EPA exhibits no redundancy. There are plans to survey similar shore pine habitats along the northern coast of Oregon in Tillamook County in summer 2018, but for now, the CCO EPA is the only known population supporting several animals in the unique shore pine forest ecosystem. Furthermore, this population does not have functional connectivity to any other population and if a stochastic or catastrophic event eliminated it, natural recolonization would not be feasible. The late–seral and serpentine ecosystem has redundancy among the three populations; however, they are relatively concentrated in the center of the historical range.

Historically, most coastal marten populations were likely connected through corridors of suitable habitat, or through refugia in historically large habitat patches that sustained populations while disturbed areas returned to suitable habitat. However, due to extensive and prolonged habitat fragmentation, coastal marten populations are currently relatively isolated from one another and repopulation of extirpated locations is unlikely to occur without human assistance. Based on historical trapping information it can be assumed there was significant redundancy since trapping records indicate the species occurred along the length of the Oregon and northern California coast.

Chapter 5 Future Condition

We have considered what the coastal marten needs for viability and the current condition of those needs (Chapters 2 and 4), and we reviewed the risk factors that are driving the historical, current, and future conditions of the species (Chapter 3). We now consider what the species' future conditions are reasonably likely to be. We apply our future forecasts to the concepts of resiliency, redundancy, and representation to describe the future viability of the coastal marten.

5.1 Introduction

The coastal marten has declined significantly in overall distribution and abundance, with the species currently occupying 7.5 percent of its historical range. The resulting remnant populations occupy smaller tracts of forest compared to presumed historical populations, and two of the four are functionally isolated from any other populations. The primary historical reason for this reduction in range was overtrapping for the fur trade in the early 1900s and since then vegetation management of large tracts of forest have affected the remaining habitat and reduced the likelihood of natural reestablishment of historical populations.

The effects of this habitat alteration extend beyond fragmentation of populations: the remaining habitat does not always support the resource needs discussed in Section 2.5 and there is a marked increase in risk of predation and individual survival. As discussed in Section 3.2.1, exacerbating these effects is an increased risk of fire through changes in precipitation and temperature due to climate change. The two populations that are isolated (Central Coastal Oregon and Southern Coastal Oregon EPAs), or occupying small areas (Central Coastal Oregon and the California–Oregon Border EPAs), are particularly vulnerable since a single stochastic event such as a large fire or disease outbreak could eliminate an entire population of coastal marten. Ultimately, the three southern populations are vulnerable to large fires affecting a significant portion of the occupied area with those that are isolated having a lower chance of natural recolonization. These effects are heightened at the species level because the isolation of the populations prohibits natural recolonization, which historically likely allowed the species to occupy suitable areas and recolonize after stochastic and catastrophic events.

All four populations face risks from natural and anthropogenic sources. As discussed in Section 3.2.3, changes in climate patterns have already begun to affect the regions of California and Oregon where coastal martens occur, resulting in a decreased fire return interval and an increase in sizes of wildfires and changing precipitation patterns. Changes in precipitation and temperature will likely further alter the vegetation structure and habitat of coastal martens and put individuals and populations at an elevated risk of predation and disease, reduce suitable habitat availability, and decrease the quality of available habitat.

These risks, alone or in combination, could result in extirpation of one or more remaining populations, further reducing the overall redundancy and representation of the coastal marten. Historically, coastal martens, with a larger range of interconnected populations, would have been

more resilient to stochastic events such as fire and habitat loss because even if some populations were extirpated by such events, they could be recolonized over time by dispersal from nearby surviving populations. This connectivity would have made for a highly resilient species overall. However, under current conditions, restoring that connectivity on a scale similar to the historical range is not feasible in the short term due to large-scale habitat alterations and development between populations.

As a consequence of these current conditions, the viability of the coastal marten now primarily depends on maintaining the remaining isolated populations and potentially establishing new populations in formerly occupied habitat and in strategic locations to restore connectivity between populations (Slauson *et al.* 2019(a), entire). Further analysis could identify corridor locations and characteristics that would facilitate connectivity and researchers have begun to identify opportunities to manage forests in ways that minimize impacts to martens and their habitats.

5.2 Scenarios

Because we have significant uncertainty regarding the impacts of stressors affecting individuals and populations, we have forecast what coastal marten may have in the future in terms of resiliency, redundancy, and representation under three plausible scenarios. These future scenarios forecast a range of likely coastal marten viability over the next 15, 30, and 60 years. We chose 15 years as a temporal extant for assessing the impact of stressors to marten populations in the near term because it is roughly the length of three marten generations and is a recommended timeframe established by the International Union for Conservation of Nature (IUCN 2017, p. 35). We chose the two longer periods of 30 and 60 years as multiples of generation length (6 and 12 marten generations, respectively) and to provide a longer temporal extant to assess the stressors of wildfire and climate change based on availability of wildfire data and climate models (IPCC 2014, pp. 58–59).

A range of time frames with a multitude of possible scenarios allows us to create a "risk profile" for the coastal marten and its viability into the future. Scenario 1 evaluates the future condition of the coastal marten if there is no change in trends in threats to the populations from what exists today, while the other two scenarios evaluate the response of the species to increases or decreases in the major factors that are influencing marten viability. While we do not expect every condition for each scenario to be fully realized, we are using these scenarios as examples for the range of possibilities. Scenarios 2 and 3 can be considered the outside bounds for the range of potential future conditions. For each scenario, we describe the stressors that would occur in each population. We use the best available science to predict trends in future stressors (timber harvest, wildfire, etc.). Data availability varies across states and populations. Where data on future trends is not available, we look to past trends and evaluate if it is reasonable to assume these trends will continue.

We examine the resiliency, representation, and redundancy of coastal marten under each of these three plausible scenarios. Resiliency depends on presence of suitable habitat for reproduction and dispersal, connectivity between populations, and a sufficient number of individuals and populations to withstand stochastic events. We expect the four extant coastal marten populations to experience changes to these aspects of their habitat in different ways under the different scenarios. We projected the expected future resiliency of each population based on the events that would occur under each scenario. We then projected an overall condition for each population. For these projections, populations in high condition are expected to have high resiliency at that time; i.e., they occupy habitat of sufficient size and in sufficient numbers to survive stochastic events. Populations in high condition are expected to persist into the future and have the ability to withstand stochastic events that may occur. Populations in moderate condition are less resilient than those in high condition, but the majority of these populations are expected to persist into the future. Populations in moderation condition are smaller and less dense than those in high condition. Finally, the populations in low condition have low resiliency and are not necessarily able to withstand stochastic events. As a result, they would not be likely to persist in that condition into the future.

5.3 Scenario 1 (Continuation)

5.3.1 Resiliency

Central Oregon Coast – In this future scenario, trapping in Oregon is legal but there are no registered trappers targeting coastal martens in Oregon. The only anticipated impact from trapping would be incidental trapping. There is no change in connectivity across Highway 101. The population remains relatively stable with no area for expansion. There is a continued loss of older forest habitat on both Federal and private land projected at a combined rate of 15 percent lost every 30 years within the surrounding province, but 4.5 percent within the population area (Table 3.2). The restoration plan on the Oregon Dunes National Recreation Area continues to be developed, which, if implemented, would eliminate a currently unknown amount of presently occupied marten habitat. This reduction in habitat and increased fragmentation would reduce the number of available home ranges in this population. All of the existing between population factors (distance to closest population, number of filters, and number of populations in proximity) would remain low due to the isolated nature of this population as discussed above. Overall, because the population is currently small and would be unlikely to grow due to limits in habitat availability, this population would remain in low condition and continue to be vulnerable to extirpation from environmental and demographic stochastic events, assuming no immigration.

Southern Oregon Coast – In this future scenario, trapping in Oregon is legal but there are no registered trappers targeting coastal martens in Oregon. The only anticipated impact from trapping would be incidental trapping. For this population, we would expect there to be a continued loss of older forest habitat on both Federal and private land projected at a combined rate of about 10 percent loss every 30 years at the province scale and 4 percent at the population

area scale (Table 3.2). The driver of the loss on Federal land at both scales is due to wildfire events, driven by changing climate from current emissions rates, and occurs at random intervals while the loss on private land is due primarily to timber harvest (Tables 3.3 and 3.4). This habitat reduction and increased fragmentation would reduce the amount of suitable habitat, ultimately reducing the number of available home ranges and potential carrying capacity. Increased fragmentation would also increase the risk of predation (as discussed in Section 3.1.4). Connectivity does not improve to any adjacent populations and as a result, the overall condition of this population remains low.

California–Oregon Border – We expect there to be very little change in the amount of older forest habitat on either Federal or private land for this population. From 1993–2012, there was an overall increase of older forest on both Federal and private lands due to ingrowth (2 percent and 6 percent, respectively, but see caveats for this data in section 3.2.2, Vegetation Disturbance) (Table 3.2). However, because private lands make up a small percentage of the entire EPA and timber harvest would continue (Figure 5.1), we expect the increases in older forest ingrowth would not significantly increase overall habitat conditions and be relatively stable. As a result, due to the amount of habitat available, this population would remain at overall low–moderate resiliency.

Northern Coastal California – For this EPA, we would expect a continued loss of older forest habitat at the current rate of 8 percent every 30 years; this is equivalent to the overall loss within the surrounding province (Table 3.2). The driver of loss on Federal land is largely due to wildfire events which occur at random intervals and are driven by changing climate from current emissions rates, while the loss on private land is largely driven by timber harvest (Tables 3.3 and 3.4). This continued habitat reduction and fragmentation would reduce the amount of suitable habitat, ultimately reducing the number of available home ranges and potential carrying capacity. The fragmentation also increases the risk of predation (see Section 3.1.4). Connectivity does not improve to any adjacent populations due to continued timber harvest surrounding the currently occupied habitat. Specifically, the habitat to the west of the occupied area becomes less suitable to dispersing individuals after recently approved timber harvest plans are harvested (see Figure 5.1) (CalFire, unpublished report). The area west of the population boundary is particularly critical to connectivity to the older forests within and around Prairie Creek Redwoods State Park where detections of two confirmed dispersing individuals have occurred during 2009–2017. We expect this population to continues to exhibit overall moderate resiliency for the first 30 years, however both the reduction in available home ranges and an increase of the proportion of habitat subject to predation would reduce the resiliency to below moderately resilient levels after 60 years leading to an overall low–moderate population condition.

Summary for Scenario 1 — Using data from the past regarding forestry, wildfire, and climate and the impact they have on coastal martens, we were able to forecast a steady trend into the future without any increase or decrease in current trends in threats and conservation efforts. Overall, the two Oregon populations would be in low condition with the Central Coastal Oregon

vulnerable to extirpation. The two California populations would be in low–moderate condition with the Northern Coastal California population declining in size (Table 5.1).

5.3.2 Representation

As identified above, we consider the coastal marten to have representation in the form of ecological diversity in two habitat types: (1) the shore pine forest habitat represented by the Central Coastal Oregon populations and (2) the older forest/serpentine ridge habitat represented by the Southern Coastal Oregon, California–Oregon border, and Northern Coastal California populations. In Scenario 1, the current level of representation would be maintained, although the Central Coastal Oregon population would remain small and relatively vulnerable to extirpation and therefore there is a possibility of loss of that area of representation. The remaining three populations would be in low or low–moderate condition and representation would be similar to current levels. Overall, across all populations, representation could be similar to current levels or it could be reduced if the Central Coastal Oregon population is extirpated.

5.3.3 Redundancy

Within these representation areas, we then examined what redundancy would exist under the various scenarios. Under Scenario 1, redundancy would remain the same. The Central Coastal Oregon population that is adapted to living in the shore pine ecosystem would remain small with no redundancy, and according to recent studies relatively vulnerable to extirpation (Linnell *et al.* 2018, entire). The remaining three populations in older forest/serpentine habitat would provide some redundancy, but with one in low condition and two in low–moderate condition.
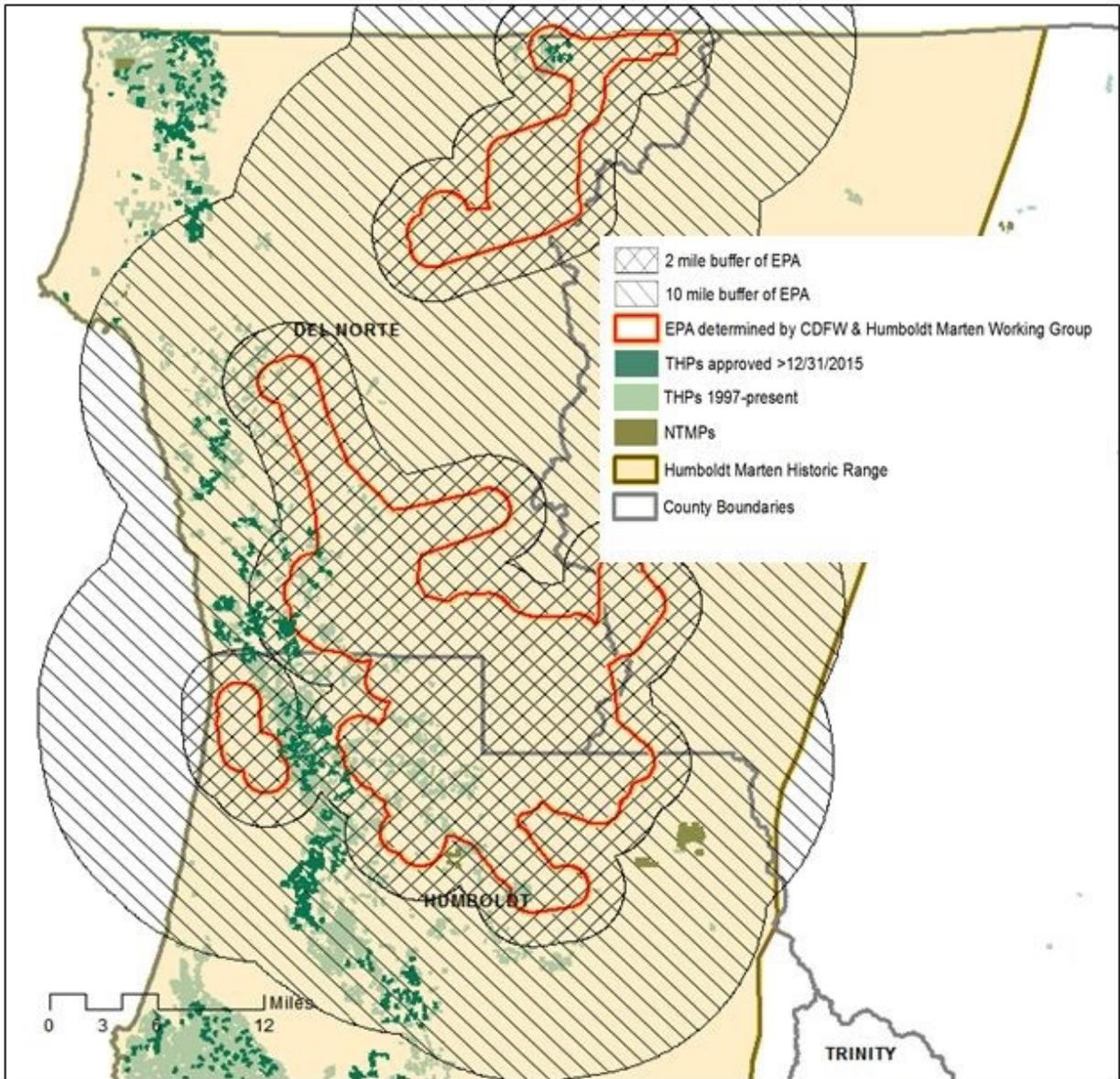
Figure 5.1. Approved timber harvest plans within, within 2 mi, and within 10 miles of extant population areas as determined by the California Department of Fish and Wildlife and the Humboldt Marten Working Group. Source: CalFire, unpublished report.

Table 5.1 Coastal marten population resiliency under Scenario 1 projected 15, 30, and 60 years into the future.

Population	Years into the future	Between Population Factors			Within Population Factors				Overall Future Condition
		Distance between populations	Number of filters	Number of populations in proximity (6–45 km)	Population Size	Number of available male home ranges	Number of available female home ranges	Proportion of suitable habitat that allows for predator avoidance	
Central Oregon	15 30 60	Low Low Low	Low Low Low	Low Low Low	Low Low Low – 0	Low Low Low	Low Low Low	Low Low Low	**Low Low Low – 0**
Southern Oregon	15 30 60	Low Low Low	Low Low Low	Low Low Low	Low Low Low	High High High	High High High	Moderate Moderate Low	**Low Low Low**
CA–OR Border	15 30 60	High High High	Moderate Moderate Moderate	Moderate Moderate Moderate	Low Low Low	Low Low Low	Low Low Low	High High High	**Low–Mod Low–Mod Low–Mod**
Northern Coastal California	15 30 60	High High High	Moderate Moderate Moderate	Moderate Moderate Moderate	Low Low Low	High High Mod–High	Moderate Moderate Low–Mod	Moderate Moderate Low	**Moderate Moderate Low–Mod**

5.4 Scenario 2

5.4.1 Resiliency

*Central Oregon Coast –*In this future scenario, the threat of trapping is eliminated with the banning of trapping in Oregon. This would reduce the number of martens lost by 0.46/year (see Section 3.1.1), however it would not result in a significant improvement in population resiliency. Habitat restoration in Oregon Dunes National Recreation Area avoids removal and fragmentation of suitable and occupied habitat. Connectivity to the interior forest east of the currently occupied population is improved through design and implementation of functional corridors across Highway 101 (see Figure 4.4), resulting in fewer roadkill mortalities. The rate of loss of older forest due to timber harvest does not increase over the levels outlined in Scenario 1. All of the between population factors (distance to closest population, number of filters, and number of populations in proximity) would remain low. Overall, because the population is currently small and would be unlikely to expand, this population would remain in low overall condition.

Southern Oregon Coast – The threat of trapping is eliminated with the banning of trapping in Oregon. Based on the moderate emissions scenario (RCP 4.5 in Sheehan *et al.* 2015, pp. 19–22) outlined in 3.2.3, there is a further decrease in suitable habitat due to a shorter fire interval and increased impacts to forest from drought. For our analysis, we assume a 25% increase over the rate of loss outlined in Scenario 1 over the next 30 years (based on Sheehan *et al.* 2015, pp. 20–22). The rate of loss of older forest due to timber harvest does not increase over the levels outlined in Scenario 1. This habitat reduction and increased fragmentation would reduce the amount of suitable habitat, ultimately reducing the number of available male and female home ranges to reflect moderate resiliency. Increased fragmentation also increases the percentage of habitat susceptible to predation as discussed in Section 3.1.4. In this scenario, models are developed to target important connectivity corridors to the south through the Rogue–Siskiyou National Forest and forestry prescriptions are evaluated to limit impacts to marten dispersal, informing adaptive management practices. Connectivity may improve but not to levels that would affect population resiliency. Overall condition of this population remains low based on a declining amount of suitable habitat and continued low connectivity.

California–Oregon Border – There is no increase in the rate of loss of older forest due to timber harvest over the levels outlined in Scenario 1. Based on the moderate emissions scenario (RCP 4.5 in Sheehan *et al.* 2015, pp. 19–22) outlined in 3.2.3, there is a further decrease in suitable habitat due to a shorter fire interval and impacts to forest from drought. For our analysis, we assume a 25% increase over the rate of loss outlined in Scenario 1 over the next 30 years. Although the fire risk increases, suitable habitat between this population and the Northern Coastal California population is not lost due to wildfire and connectivity remains high between these two populations. With the relatively stable amount of older forest, this population's condition continues to exhibit low to moderate resiliency.

Northern Coastal California – The Humboldt Marten Conservation Group has suggested the development of a Habitat Management Guide that is "capable of identifying the habitat characteristics most important for supporting marten occupancy, reproduction, and survival at the stand and home range scales" (Slauson et al., 2019, entire). While this strategy is not currently being implemented, all landowners within this EPA are co–authors on the publication that outlines this conservation strategy so we can assume they are supportive of this type of adaptive management and that development and implementation would increase the amount of suitable habitat available to coastal martens. In this scenario, we assume this guide is developed and implementation begins which increases both the amount of suitable habitat present and connectivity between populations over time. Assisted dispersal is found to be feasible and is implemented creating an additional population with connectivity to this population increasing the number of populations in proximity to reflect high resiliency. Based on the moderate emissions scenario (RCP 4.5 in Sheehan *et al.* 2015, pp. 19–22) outlined in 4.2.3, there is a decrease in suitable habitat due to a shorter fire interval and impacts to forest from drought. For our analysis, we assume a relatively modest 25% increase over the rate of loss outlined in Scenario 1

over the next 30 years. Even though habitat continues to be lost due to wildfire, with the increase in number of populations, increased connectivity, and implemented adaptive management strategies this population increases to moderate–high resiliency.

Summary for Scenario 2 — Assuming a moderate emissions scenario and implementation of proposed conservation measures combined with what we know about current threats to coastal marten we were able to project trends into the future for a plausible optimistic scenario. Overall, the two Oregon populations would be in low condition. The two California populations would be in low–moderate to moderate–high condition (Table 5.2).

Table 5.2 Coastal marten population resiliency under Scenario 2 projected 15, 30, and 60 years into the future.

Population	Years into the future	Between Population Factors			Within Population Factors				Overall Future Condition
		Distance through suitable habitat	Number of filters	Number of populations in proximity (6–45 km)	Population Size	Number of available male home ranges	Number of available female home ranges	Proportion of suitable habitat that allows for predator avoidance	
Central Oregon	15	Low	Low	Low	Low	Low	Low	Low	**Low**
	30	Low	Low	Low	Low	Low	Low	Low	**Low**
	60	Low	Low	Low	Low	Low	Low	Low	**Low**
Southern Oregon	15	Low	Low	Low	Low	High	High	Moderate	**Low**
	30	Low	Low	Low	Low	High	High	Moderate	**Low**
	60	Low	Low	Low	Low	High	High–Mod	Low	**Low**
CA–OR Border	15	High	Moderate	Moderate	Low	Low	Low	High	**Low –Mod**
	30	High	Moderate	Moderate	Low	Low	Low	High	**Low –Mod**
	60	High	Moderate	Moderate	Low	Low	Low	High	**Low –Mod**
Northern Coastal California	15	High	Moderate	Moderate	Low	High	Moderate	Moderate	**Moderate**
	30	High	Moderate	High	Low	Mod – High	Moderate	Moderate	**Mod–High**
	60	High	Moderate	High	Moderate	Mod – High	Moderate	Moderate	**Mod–High**

5.4.2 Representation

As identified above, we consider the coastal marten to have representation in the form of ecological diversity in two habitat types: (1) the shore pine forest habitat represented by the Central Coastal Oregon populations and (2) the older forest/serpentine ridge habitat represented

by the Southern Coastal Oregon, California–Oregon border, and Northern Coastal California populations. In Scenario 2, the current level of representation would be maintained, although the Central Coastal Oregon population would remain small. The remaining three populations would range from low to moderate–high condition and representation would be similar to current levels. Overall, across all populations, representation could be similar to current levels.

5.4.3 Redundancy

Within these representation areas, we then examined what redundancy would exist under the various scenarios. Under Scenario 2, redundancy would remain the same as Scenario 1. The Central Coastal Oregon population that is adapted to living in the shore pine ecosystem would remain small with no redundancy. There would be an additional three populations providing redundancy ranging in condition from low to moderate–high.

5.5 Scenario 3
5.5.1 Resiliency

Central Oregon Coast – Trapping in Oregon remains legal. There is no change in connectivity across Highway 101. While we do not currently have data on what proportion of the occupied population will be impacted, we assume that dune restoration at the Oregon Dunes National Recreation Area negatively impacts currently occupied habitat, threatening the future viability of this population. All of the between population factors (distance to closest population, number of filters, and number of populations in proximity) would remain low. Overall, because the population is currently small and would be unlikely to expand, this population would remain in low overall condition and as recent studies have shown remain vulnerable to extirpation with only modest increases in mortality (Linnell *et al.* 2018, entire).

Southern Oregon Coast – Trapping in Oregon remains legal. Based on the higher emissions scenario (RCP 8.5 in Sheehan *et al.* 2015, pp. 19–22) outlined in 3.2.3, there is a further decrease in suitable habitat due to a 50% shorter fire interval and impacts to forest from drought (Sheehan *et al.* 2015, pp. 20–22). For our analysis, we assume a 50% increase over the rate of loss outlined in Scenario 1 over the next 30 years (see Section 3.2). With this increase in loss of habitat due to fire the number of available home ranges and the amount of habitat that allows for predator avoidance would decline over time reducing the population resiliency from moderate to a low–moderate condition. We would expect this decline to continue and further reduce the population's resiliency below the low–moderate condition.

California–Oregon Border – Based on the higher emissions scenario (RCP 8.5 in Sheehan *et al.* 2015, pp. 19–22) outlined in 3.2.3, there is a further decrease in suitable habitat due to a 50% shorter fire interval and increased impacts to forest health from drought. For our analysis, we assume a 50% increase over the rate of loss outlined in Scenario 1 over the next 30 years based on these increases. This increase in fire, and subsequent loss of dispersal habitat, decreases

connectivity between this population and the Northern Coastal California population. With an increase in loss of habitat due to increased fire the number of available home ranges would decline to low condition and the amount of habitat that allows for predator avoidance would decline over time to moderate conditions over time. Overall the population would remain in low–moderate condition.

Northern Coastal California – Based on the higher emissions scenario (RCP 8.5 in Sheehan *et al.* 2015, pp. 19–22) outlined in 3.2.3, there is a further decrease in suitable habitat due to a 50% shorter fire interval and impacts to forest from drought. For our analysis we assume a 50% increase over the rate of loss outlined in Scenario 1 over the next 30 years. This increase in fire, and subsequent loss of dispersal habitat, decreases connectivity between this population and the California–Oregon Border population. With this increase in loss of habitat due to fire the number of available home ranges would decrease and the amount of habitat subject to predation would increase causing a decline over time of overall population resiliency to low–moderate and eventually low condition.

Summary for Scenario 3 — Assuming a high emissions scenario combined with what we know about current threats to coastal marten we were able to project trends into the future for a plausible pessimistic scenario. Overall, the two Oregon populations would be in low condition, with one potentially extirpated. The two California populations would be in low to low–moderate condition (Table 5.3).

5.5.2 Representation

As identified above, we consider the coastal marten to have representation in the form of ecological diversity in two habitat types: (1) the shore pine forest habitat represented by the Central Coastal Oregon populations and (2) the older forest/serpentine ridge habitat represented by the Southern Coastal Oregon, California–Oregon border, and Northern Coastal California populations. In Scenario 3, the current level of representation would be maintained, although the Central Coastal Oregon population would remain small and relative vulnerable to extirpation and therefore there is a possibility of loss of that area of representation. The remaining three populations would be in low or low–moderate condition and representation would be similar to current levels. Overall, across all populations, representation could be similar to current levels or it could be reduced if the Central Coastal Oregon population is extirpated.

5.5.3 Redundancy

Within these representation areas, we then examined what redundancy would exist under the various scenarios. Under Scenario 3, redundancy would remain the same. The Central Coastal Oregon population that is adapted to living in the shore pine ecosystem would remain small with no redundancy, and according to recent studies relatively vulnerable to extirpation (Linnell *et al.*

2018, entire). The remaining three populations would provide some redundancy, but with two in low condition and one in low–moderate condition.

Table 5.3 Coastal marten population resiliency under Scenario 3 projected 15, 30, and 60 years into the future.

Population	Years into the future	**Between Population Factors**			**Within Population Factors**				**Overall Future Condition**
		Distance through suitable habitat	Number of filters	Number of populations in proximity (6–45 km)	Population Size	Number of available male home ranges	Number of available female home ranges	Proportion of suitable habitat that allows for predator avoidance	
Central Oregon	15	Low	Low	Low	Low	Low	Low	Low	**Low**
	30	Low	Low	Low	Low – 0	Low	Low	Low	**Low – 0**
	60	Low	Low	Low	Low – 0	Low	Low	Low	**Low – 0**
Southern Oregon	15	Low	Low	Low	Low	High	High	Moderate	**Low**
	30	Low	Low	Low	Low	High	High	Moderate	**Low**
	60	Low	Low	Low	Low	High	High–Mod	Low	**Low**
CA–OR Border	15	High	Moderate	Moderate	Low	Low	Low	High	**Low–Mod**
	30	High	Moderate	Moderate	Low	Low	Low	Moderate	**Low–Mod**
	60	High	Moderate	Moderate	Low	Low	Low	Moderate	**Low–Mod**
Northern Coastal California	15	High	Moderate	Moderate	Low	High	Moderate	Moderate	**Moderate**
	30	High	Moderate	Moderate	Low	High	Moderate	Moderate	**Moderate**
	60	High	Moderate	Moderate	Low	Mod–High	Low–Mod	Low	**Low–Mod**

5.6 Summary of Future Conditions

We used the best available information to forecast plausible future conditions of the populations of coastal marten. Our goal was to describe the future viability of the species in a manner that will address the needs of the species in terms or resiliency, representation, and redundancy. We considered a range of potential scenarios involving plausible factors that are important influences on the viability of the species. Our results describe a range of possible conditions in terms of how many and where coastal marten populations are likely to persist into the future (Table 5.4).

Coastal marten face a variety of risks from small populations, loss of habitat, fragmentation that affects predation risk, and decreased connectivity between populations. These risks play a large role in the future viability of the coastal marten. If populations lose resiliency, they are more vulnerable to extirpation, with resulting loses in representation and redundancy. Underlying all of these scenarios is the threat to the Central Coastal Oregon population of a large tsunami. This

would have impacts to resiliency of that population and the redundancy and representation of the coastal marten. We did not include this in our future conditions scenarios due to the uncertainty of when this could occur (see Section 3.2.4).

Under Scenario 1 – We would expect the coastal marten's viability to be characterized by a loss of resiliency and potentially a loss of representation and redundancy if the Central Coastal Oregon population were extirpated. No populations would be in high condition, two would be in low–moderate condition, one would be in low condition, and the Central Coastal Oregon population would be at a high risk of being lost.

Under Scenario 2 – We would expect the coastal marten's viability to be characterized by higher levels of resiliency, representation, and redundancy than it exhibits under the current condition. One of the current populations would improve to moderate condition, another would improve to moderate–high condition, while two would remain in low condition. We would anticipate all of the current populations to persist into the future.

Under Scenario 3 – We would expect the coastal marten's viability to be characterized by additional losses of resiliency, representation, and redundancy. The Central Coastal Oregon population would be at a high risk of being extirpated and the Northern Coastal California population would decline to low condition.

Table 5.4. Coastal marten population conditions under each scenario.

Population	Current Condition	Years into the future	Scenario 1	Scenario 2	Scenario 3
Central Oregon	Low	15 30 60	Low Low Low – 0	Low Low Low	Low Low – 0 Low – 0
Southern Oregon	Low	15 30 60	Low Low Low	Low Low Low	Low Low Low
CA–OR Border	Low–Moderate	15 30 60	Low–Mod Low–Mod Low–Mod	Low–Mod Low–Mod Low–Mod	Low–Mod Low–Mod Low–Mod
Northern Coastal California	Moderate	15 30 60	Moderate Moderate Low–Mod	Moderate Mod–High Mod–High	Moderate Moderate Low–Mod

Chapter 6 References Cited

AFWA. (Association of Fish and Wildlife Agencies) 2006. Furbearer management and best management practices for trapping. Chapters for marten and for introduction downloaded June 4, 2018 from: https://www.fishwildlife.org/afwa–inspires/furbearer–management

Agee, J.K. 1993. Fire Ecology of Pacific Northwest Forests. Island Press. Washington, D.C. 493 pp.

Alexander, S.M. and N.M. Waters. 2007. The effects of highway transportation corridors on wildlife: a case study of Banff National Park. Transportation Research Part C 8. 307–320.

Amaresekare, P. 2003. Competitive coexistence in spatially structured environments: a synthesis. Ecology Letters 6:1109–1122.

Andrén, Henrik. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. Oikos 71:355–366.

Andruskiw, M., J. Fryxell, I.D. Thompson, and J.A. Baker. 2008. Habitat–mediated variation in predation risk by the American marten. Ecology 89:2273–2280.

Anonymous. 1914. Report of fur–bearing animals. The Oregon Sportsman II (6):20.

Arno, S.F. 2000. Fire in western forest ecosystems. Pp. 97–120 in, J.K. Brown and J.K. Smith (eds.), Wildland fire in ecosystems: effects of fire on flora. General Technical Report RMRS–GRT–42–volume 2. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT. 257 pp.

Azuma, D.L., L.F. Bednar, B.A. Hiserote, and C.F. Veneklase. 2004. Timber resource statistics for western Oregon, 1997. Resource Bulletin PNW–RB–237 (revised). USDA Forest Service, Pacific Northwest Research Station, Portland, OR. 120 pp.

Bailey, V. 1936. The mammals and life zones of Oregon. North American Fauna 55:1–117 and 261–404.

Biberdorf, A. 1982. Status of the marten in California. State of California, The Resources Agency, Department of Fish and Game, Nongame Wildlife Investigations. Unpublished report.

Boitani L, and R.A. Powell. 2012. Carnivore Ecology and Conservation: A Handbook of Techniques.

Bolsinger, C.L. and K.L. Waddell. 1993. Area of old–growth forests in California, Oregon, and Washington. Resource Bulletin PNW–RB–197. USDA Forest Service, Pacific Northwest Research Station, Portland, OR. 26 pp.

Bowler, D.E., and T.G. Benton. 2005. Causes and consequences of animal dispersal strategies: relating individual behavior to spatial dynamics. Biological Review 80:205–225.

Broquet, T., C.A. Johnson, E. Petit, I.D. Thompson, F. Burel, J.M. Fryxell. 2006. Dispersal and genetic structure in the American marten, *Martes americana*. Molecular Ecology. 15:1689–97.

Brown, J.K. 2000. Introduction and fire regimes. Pp. 1–7 in, J.K. Brown and J.K. Smith (eds.), Wildland fire in ecosystems: effects of fire on flora. General Technical Report RMRS–GRT–42–volume 2. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT. 257 pp.

Brown, R.N., M.W. Gabriel, G.M. Wengert, S. Matthews, J.M. Higley, and J.E. Foley. 2008. Pathogens associated with fishers. Pages 3–47 *in* Pathogens associated with fishers (*Martes pennanti*) and sympatric mesocarnivores in California: final draft report to the U.S. Fish and Wildlife Service for Grant #813335G021. U.S. Fish and Wildlife Service, Yreka, CA, USA.

Bull, E.L., and T.W. Heater. 2001a. Survival, causes of mortality, and reproduction in the American marten in northeastern Oregon. Northwestern Naturalist 82:1–6.

Bull, E.L., and T.W. Heater. 2001b. Home range and dispersal of the American marten in northeastern Oregon. Northwestern Naturalist 82:7–11.

Bull, E.L., C.G. Parks, and T.R. Torgersen. 1997. Trees and logs important to wildlife in the interior Columbia River basin. General Technical Report PNW–GTR–391. USDA Forest Service, Pacific Northwest Research Station, Portland, OR. 55 pp.

Buskirk, S.W., and L.R. Ruggiero. 1994. American marten. Pages 7–37 *in* L.F. Ruggiero, K.B. Aubry, S.W. Buskirk, L.J. Lyon, and W.J. Zielinski (editors), American marten, fisher, lynx, and wolverine in the western United States. General Technical Report RM–254. U.S. Department of Agriculture, Forest Service. Rocky Mountain Research Station. Fort Collins, CO, USA.

Butsic, V., and J.C. Brenner. 2016. Cannabis (*Cannabis sativa* or *C. indica*) agriculture and the environment: a systematic, spatially–explicit survey and potential impacts. Environmental Research Letters. 11:044023

CalTrans. Undated. 2014 traffic volumes on the California State highway system. State of California, California State Transportation Agency, Department of Transportation, Division of Traffic Operations. Sacramento, CA.

Carey, A.B. 1991. The biology of arboreal rodents in Douglas–fir forests. Pages 1–46 in M.A. Huff, R.S. Holthausen, and K.B. Aubry (technical coordinators), Biology and management of old–growth forests. General Technical Report. PNW–GTR–276. U.S. Department of Agriculture, Forest Service. Pacific Northwest Research Station. Portland, OR, USA.

Carey, A.B., and M.L. Johnson. 1995. Small mammals in managed, naturally young, and old growth forests. Ecological Applications 5(2):336–352.

Carroll, C., J.A. Vucetich, M.P. Nelson, D.J. Rohlf, and M.K. Phillips. 2010. Geography and recovery under the U.S. Endangered Species Act. Conservation Biology 24:395–403.

Cascadia Wildlands. 2018. Petition to initiate rulemaking to amend OAR 635–050–0110 to prohibit marten trapping in portions of western Oregon. Downloaded April 11, 2018 from: https://www.cascwild.org/wp–content/uploads/2018/04/Humboldt–Marten–Oregon–Trapping–Petition–April–4–Final.pdf.

Cayan, D., M. Tyree, D. Pierce, and T. Das. 2012. Climate Change and Sea Level Rise Scenarios for California Vulnerability and Adaptation Assessment. California Energy Commission. Publication number: CEC-500-2012-008.

CDFW (California Department of Fish and Wildlife). 2018. Report to the Fish and Game Commission: A status review of the Humboldt marten (*Martes caurina humboldtensis*) in California. State of California, Natural Resources Agency, Department of Fish and Wildlife, Sacramento, California, USA.

CDPH (California Department of Public Health). 2010. Reported animal rabies by county and species California, January 1 – December 31, 2010: final. California Department of Public Health, Veterinary Public Health Section. Downloaded April 26, 2018 from: https://www.cdph.ca.gov/Programs/CID/DCDC/CDPH%20Document%20Library/2010RARTable.pdf

CDPH (California Department of Public Health). 2011. Reported animal rabies by county and species California, January 1 – December 31, 2011: final. California Department of Public Health, Veterinary Public Health Section. Downloaded April 26, 2018 from: https://www.cdph.ca.gov/Programs/CID/DCDC/CDPH%20Document%20Library/2011RARTable.pdf

CDPH (California Department of Public Health). 2012. Reported animal rabies by county and species California, January – December 31, 2012: final. California Department of Public Health, Veterinary Public Health Section. Downloaded April 26, 2018 from: https://www.cdph.ca.gov/Programs/CID/DCDC/CDPH%20Document%20Library/2012RARTable.pdf

CDPH (California Department of Public Health). 2013. Reported animal rabies by county and species California, January 1 – December 31, 2013. California Department of Public Health, Veterinary Public Health Section. Downloaded April 26, 2018 from: https://www.cdph.ca.gov/Programs/CID/DCDC/CDPH%20Document%20Library/2013RARTable.pdf

CDPH (California Department of Public Health). 2014. Reported animal rabies by county and species California, January 1 – December 31, 2014: final. California Department of Public Health, Veterinary Public Health Section. Downloaded April 26, 2018 from: https://www.cdph.ca.gov/Programs/CID/DCDC/CDPH%20Document%20Library/2014RARTable.pdf

CDPH (California Department of Public Health). 2015. Reported animal rabies by county and species California, January 1 – December 31, 2015: final. California Department of Public Health, Veterinary Public Health Section. Downloaded April 26, 2018 from: https://www.cdph.ca.gov/Programs/CID/DCDC/CDPH%20Document%20Library/2015RARTable.pdf

CDPH (California Department of Public Health). 2016. Reported animal rabies by county and species California, January 1 – December 31, 2016: final. California Department of Public Health, Veterinary Public Health Section. Downloaded April 26, 2018 from: https://www.cdph.ca.gov/Programs/CID/DCDC/CDPH%20Document%20Library/2016ReportedAnimalRabiesCalifornia.pdf

CDPH (California Department of Public Health). 2017. Reported animal rabies by county and species California, January 1 – November 30, 2017: provisional. California Department of Public Health, Veterinary Public Health Section. Downloaded April 26, 2018 from: https://www.cdph.ca.gov/Programs/CID/DCDC/CDPH%20Document%20Library/RARTable2017.pdf

Chapin, T.G., D.J. Harrison, and D.D. Katnik. 1998. Influence of landscape pattern on habitat use by American marten in an industrial forest. Conservation Biology 12:1327–1337.

Chappell, C. B., and J. Kagan. 2001. Westside grasslands. Pages 41–43 *in* D. H. Johnson and T. A. O'Neil, managing directors. Wildlife–habitat relationships in Oregon and Washington. Oregon State University Press, Corvallis, Oregon.

Cheveau, M., L. Imbeau, P. Drapeau, and L. Belanger. 2013. Marten space use and habitat selection in managed coniferous boreal forests of eastern Canada. Journal of Wildlife Management 77:749–760.

Christensen, J.H., B. Hewitson, A. Busuioc, A. Chen, X. Gao, R. Held, R. Jones, R.K. Kolli, W.K. Kwon, R. Laprise, V.M. Rueda, L. Mearns, C.G. Menéndez, J. Räisänen, A. Rinke, A. Sarr, and P. Whetton. 2007. Regional climate projections. Pages 847–940 *in* S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor & H.L. Miller (editors), Climate Change 2007: The Physical Science Basis. Contribution of Working group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Clark, T.W., E. Anderson, C. Douglas, and M. Strickland. 1987. *Martes americana*. Mammalian Species 289:1–8.

Clark, T.W., and T.M. Campbell. 1976. Population organization and regulatory mechanisms of pine martens in Grand Teton National Park, Wyoming. Pages 293–295 *in* R.M. Linn (editor), Proceedings of the first conference on scientific research in National Parks. New Orleans, LA, USA.

Clevenger, A.P., B. Chruszcz, and K. Gunson. 2001. Drainage culverts as habitat linkages and factors affecting passage by mammals. Journal of Applied Ecology 38:1340–1349.

Cohen, W.B., T.A. Spies, R.J. Alig, D.R. Oetter, T.K. Maiersperger, and M. Fiorella. 2002. Characterizing 23 years (1972–95) of stand replacement disturbance in western Oregon forests with Landsat imagery. Ecosystems 5:122–137.

Cohen, W.B., Z. Yang, S.P. Healey, R.E. Kennedy, N. Gorelick. 2018. A LandTrendr multispectral ensemble for forest disturbance detection. Remote Sensing of Environment 205:131–140.

Colella, J.P., T. Lan, S.L. Talbot, C. Lindqvist, and J.A. Cook. 2021. Whole-genome resequencing reveals persistence of forest-associated mammals in Late Pleistocene refugia along North America's North Pacific Coast. Journal of Biogeography 48: 1153-1169.

Collins, B.M. and G.B. Roller. 2013. Early forest dynamics in stand–replacing fire patches in the northern Sierra Nevada, California, USA. Landscape Ecology 28:1801–1813.

Conard, J.M., and P.S. Gipson. 2006. Spatial and seasonal variation in wildlife–vehicle collisions. The Prairie Naturalist 38:251–260.

Cooperrider, A., R.F. Noss, H.H. Welsh, Jr., C. Carroll, W. Zielinski, D. Olson, S.K. Nelson, and B.G. Marcot. 2000. Terrestrial fauna of redwood forests. Pages 119–164 *in* R.L. Noss (editor), The Redwood Forest. Island Press, Washington, D.C and Covelo, CA, USA.

Credo, K. 2017. Assessing alternatives for fuel reduction treatment and Pacific marten conservation in the southern Cascades and northern Sierra Nevada. M.S. Thesis, Oregon State University, Corvallis, Oregon, USA.

Cushman, S.A., M.G. Raphael, L.F. Ruggiero, A.S. Shirk, T.N. Wasserman, E.C. O'Doherty. 2011. Limiting factors and landscape connectivity: the American marten in the Rocky Mountains. Landscape Ecology 26:1137–1149.

Dalton, M.M., K.D. Dello, L. Hawkins, P.W. Mote, and D.E. Rupp. 2017. The Third Oregon Climate Assessment Report. Oregon Climate Change Research Institute, College of Earth, Ocean and Atmospheric Sciences, Oregon State University, Corvallis, OR.

Daszak, P., A.A. Cunningham, and A.D. Hyatt. 2001. Anthropogenic environmental change and the emergence of infectious diseases in wildlife. Acta Tropica. 78:103–116.

Davis, R.J., J.L. Ohmann, R.E. Kennedy, W.B. Cohen, M.J. Gregory, Z.Yang, H.M. Roberts, A.N. Gray, and T.A. Spies. 2015. Northwest Forest Plan—the first 20 years (1994–2013): status and trends of late–successional and old–growth forests. General Technical Report PNW–GTR–911. USDA Forest Service, Pacific Northwest Research Station, Portland, OR. 112 pp.

Dawson, T.E. 1998. Fog in the California redwood forest: ecosystem inputs and use by plants. Oecologia 117:476–485.

Dawson, N.G., and J. A. Cook. 2012. Behind the genes: diversification of North American martens (*Martes americana* and *M. caurina*). Pages 23–38 in K. Aubry, W. Zielinski, M. Raphael, G. Proulx, and S. Buskirk (editors). Biology and conservation of martens, sables, and fishers: a new synthesis. Cornell University Press, Ithaca, NY, USA.

Dawson, N.G., J.P. Colella, M.P. Small, K.D. Stone, S.L. Talbot, and J.A. Cook. 2017. Historical biogeography sets the foundation for contemporary conservation of martens (genus *Martes*) in northwestern North America. Journal of Mammalogy 98:715-730.

Day, C.C., N.P. McCann, P.A. Zollner, J.H. Gilbert, and D.M. McFarland. 2019. Temporal plasticity in habitat selection criteria explains patterns of animal dispersal. Behavioral Ecology 30:528-540.

Day, C.C., P.A. Zollner, J.H. Gilbert, and N.P. McCann. 2020. Individual-based modeling highlights the importance of mortality and landscape structure in measures of functional connectivity. Landscape Ecology 35:2191-2208.

Dean Runyan Associates. 2017. Oregon travel impacts 1992–2016p: statewide preliminary estimates, detailed county estimates, overnight visitor volume. Prepared for Oregon Tourism Commission, Salem, OR by Dean Runyan Associates, Portland, Oregon.

Delheimer, M.S., K.M. Moriarty, K.M. Slauson, A.M. Roddy, D.A. Early, and K.A. Hamm. 2021. Comparative reproductive ecology of two subspecies of Pacific marten (*Martes caurina*) in California. Northwest Science 94: 271-285.

DellaSala, D.A. 2013. Rapid Assessment of the Yale Framework and Adaptation Blueprint for the North America Pacific Coastal Rainforest. *in* Data Basin. [File initial publication in Data Basin on January 31, 2013; File last modified on March 15, 2013; File last accessed on August 27, 2014: http://databasin.org/articles/172d089c062b4fb686cf18565df7dc57

Dixon, J. 1925. A closed season needed for fisher, marten, and wolverine. California Fish and Game 11:23–25.

Drew, G.S. 1995. Winter habitat selection by American marten (*Martes americana*) in Newfoundland: why old growth? Ph.D. Dissertation. Utah State University, Logan, Utah.

Donadio, E. and S.W. Buskirk. 2006. Diet, morphology, and interspecific killing in carnivore. The American Naturalist 167:524–536.

Early, D.A., K.A. Hamm, L.V. Diller, K.M. Slauson, and W.J. Zielinski. 2017. ABSTRACT: A landowner case study: Humboldt marten on a managed forest landscape. Forest carnivores and their habitats: a focus on fisher, marten and fox. Wildlife in Managed Forests Symposium, Albany, Oregon. Oregon Forest Resources Institute.

Easterday, K., P. McIntyre, and M. Kelly. 2018. Land ownership and 20th century changes to forest structure in California. Forest Ecology and Management 422:137–146.

Ellis, L.M. 1998. Habitat–use patterns of the American marten in the southern Cascade Mountains of California, 1992–1994. M.S. thesis, Humboldt State University. Arcata, CA, USA.

Ellison, J., K.M. Moriarty, and B.R. Barry. 2017. Preliminary detection data from remote camera surveys for carnivores in western Oregon. Poster presentation. Oregon Chapter of The Wildlife Society.

Ellison, J.L., K. Moriarty, B. Barry. 2018. Changes in carnivore distribution in western Oregon. Abstract of poster presented at the 2018 joint annual meeting of the Oregon and Washington chapters of The Wildlife Society, the Society of Northwest Vertebrate Biology, and the Northwest Partners in Amphibian and Reptile Conservation.

Erickson, W., and D. Urban. 2004. Potential risks of nine rodenticides to birds and nontarget mammals: a comparative approach. United States Environmental Protection agency, Office of Prevention, Pesticides and Toxic Substance. Washington, D.C.

Eriksson, C. 2016. Martens in a novel habitat – the importance of prey and habitat structure. M.S. thesis, Lund University, Lund, Sweden.

Eriksson, C.E., K.M. Moriarty, M.A. Linnell, and T. Levi. 2019. Biotic factors influencing the unexpected distribution of Humboldt marten (Martes caurina humboldtensis) population in a young coastal forest. PLoS ONE 14(5): e021 4653. https://doi.org/10.1371/journal.pone.0214653.

Fecske, D.M., and J.A. Jenks. 2002. Dispersal by a male American marten, *Martes americana*. The Canadian Field Naturalist 116:309–311.

Fenton, A., and A.B. Pedersen. 2005. Community epidemiology framework for classifying disease threats. Emerging Infectious Diseases 11:1815–1821.

Fontaine, J.B., and P.L. Kennedy. 2012. Meta–analysis of avian and small–mammal response to fire severity and fire surrogate treatments in US fire–prone forests. Ecological Applications 22:1547–1561.

Forman, R.T.T. and L.E. Alexander. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics 29:207–231.

Fortin, C., and M. Cantin. 2004. Harvest status, reproduction and mortality in a population of American martens in Quebec, Canada. Pages 221–234 in D.J. Harrison, A.K. Fuller, and G. Proulx, editors. Martens and fishers (Martes) in human–altered environments: an international perspective. Springer, New York, NY, USA.

Frankham, R. and K. Ralls. 1998. Inbreeding leads to extinction. Nature 392:441–442.

Franklin, A.B., P.C. Carlson, A. Rex, J.T. Rockweit, D. Garza, E. Culhane, S.F. Volker, R.J. Dusek, V.I. Shearn–Bochsler, M.W. Gabriel, and K.E. Horak. 2018. Grass is not always greener: rodenticide exposure of a threatened species near marijuana growing operations. BMC Research Notes. 11:94 https://doi.org/10.1186/s13104–018–3206–z

Franklin, J.F., and C.T. Dyrness. 1988. Natural vegetation of Oregon and Washington. Oregon State University Press, Corvallis, Oregon, USA.

Fredrickson, R.J. 1990. Effects of disease, prey fluctuation, and clear–cutting on American marten in Newfoundland, Canada. M.S. thesis, Utah State University. Logan, UT, USA.

Fuller, A.K., and D.J. Harrison. 2005. Influence of partial timber harvesting on American martens in north–central Maine. Journal of Wildlife Management 69(2):710–722.

Gabriel, M.W. 2013. Population health of California fishers [dissertation]. Davis: University of California. 157 pp.

Gabriel, M.W., G.M. Wengert, and R.N. Brown. 2012a. Pathogens and parasites of *Martes* species. Pages 138–162 in Aubry, K.B., W.J. Zielinski, M.G. Raphael, G. Proulx, and S.W. Buskirk (editors). Biology and conservation of martens, sables, and fishers: a new synthesis. Cornell University Press, Ithaca, NY, USA.

Gabriel, M.W., L.W. Woods, R. Poppenga, R.A. Sweitzer, C. Thompson, S.M. Matthews, J.M. Higley, S.M. Keller, K. Purcell, R.H. Barrett, G.M. Wengert, B.N. Sacks, and D.L. Clifford. 2012b. Anticoagulant rodenticides on our public and community lands: spatial distribution of exposure and poisoning of a rare forest carnivore. PLoS ONE 7(7):e40163. doi:10.1371/journal.pone.0040163

Gabriel, M.W., J.M. Higley, G. Wengert, L. Woods, and R. Poppenga. 2013. Intentional poisoning of a fisher (*Martes pennanti*) with a carbamate insecticide laced bait at an illegal marijuana trespass cultivation site, unpublished data.

Gabriel, M.W., L.W. Woods, G.M. Wengert, N. Stephenson, J.M. Higley, C. Thompson, S.M. Matthews, R.A. Sweitzer, K. Purcell, R.H. Barrett, S.M. Keller, P. Gaffney, M. Jones, R. Poppenga, J.E. Foley, R.N. Brown, D.L. Clifford, and B.N. Sacks. 2015. Patterns of natural and human–caused mortality factors of a rare forest carnivore, the fisher (*Pekania pennanti*) in California. PLoS ONE 10(11): e0140640. doi:10.1371/journal.pone.0140640

Gabriel, M.W., L.V. Diller, J.P. Dumbacher, G.M. Wengert, J.M. Higley, R.H. Poppenga, and S. Mendia. 2018. Exposure to rodenticides in northern spotted and barred owls on remote forest lands in northwestern California: evidence of food web contamination. Avian Conservation and Ecology. 13(1):2 https://doi.org/10.5751/ACE–01134–130102

Galindo, J.C., R.J. Kendall, C.J. Driver, and T.E. Lacher. 1985. The effect of methyl parathion on susceptibility of bobwhite quail (*Colinus virginianus*) to domestic cat predation. Behavioral and Neural Biology 43(1):21–36.

Gibilisco, C.J. 1994. Distributional dynamics of modern Martes in North America. Pages 59–71 in Buskirk, S.W., A.S. Harestad, M.G. Raphael, and R.A. Powell (editors), Martens, sables, and fishers: biology and conservation. Cornell University Press, Ithaca, NY, USA.

Gilbert, J.H., J.L. Wright, D.J. Lauten, and J.R. Probst. 1997. Den and rest–site characteristics of American marten and fisher in northern Wisconsin. Pages 135–145 *in* G. Proulx, H. Bryant, and P. Woodard (editors), *Martes*: taxonomy, ecology, techniques and management. Provincial Museum of Alberta. Edmonton, Alberta, Canada.

Gilbert, J.H., P.A. Zollner, A.K. Green, J.L. Wright, and W.H. Karasov. 2009. Seasonal field metabolic rates of American martens in Wisconsin. The American Midland Naturalist. 162:327–334.

Glista, D.J., T.L. DeVault, and J.A. DeWoody. 2007. Vertebrate road mortality predominantly impacts amphibians. Herpetological Conservation and Biology 3:77–87.

Goldfinger, C., S. Galer, J. Beeson, T. Hamilton, B. Black, C. Romsos, J. Patton, C.H. Nelson, R. Hausmann, and A. Morey. 2017. The importance of site selection, sediment supply, and hydrodynamics: a case study of submarine paleoseismology on the northern Cascadia margin, Washington USA. Marine Geology 384:4–46.

Grinnell, J., and J.S. Dixon. 1926. Two new races of the pine marten from the Pacific Coast of North America. Zoology 21:411–417.

Grinnell, J., J.S. Dixon, and J.M. Linsdale. 1937. Fur–bearing mammals of California. Vol. 1. University of California Press, Berkeley, CA, USA

Hagmeier, E.M. 1956 Distribution of marten and fisher in North America. Canadian Field-Naturalist 70:149-168.

Hall, E.R. 1981. The mammals of North America, 2nd edition, Volume II. John Wiley & Sons, New York, NY, USA.

Halofsky, J.E., D.L. Peterson, K.A. O'Halloran, and C.H. Hoffman. 2011. Adapting to climate change at Olympic National Forest and Olympic National Park. General Technical Report PNW–GTR–844. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. Portland, OR.

Hargis, C.D., J.A. Bissonette, and D.L. Turner. 1999. The influence of forest fragmentation and landscape pattern on American martens. Journal of Applied Ecology 36:157–172.

Hargis, C.D. and D.R. McCullough. 1984. Winter diet and habitat selection of marten in Yosemite National Park. Journal of Wildlife Management 48:140–146.

Hayes, J.P., and S.P. Cross. 1987. Characteristics of logs used by western red–backed voles, *Clethrionomys californicus*, and deer mice, *Peromyscus maniculatus*. Canadian Field–Naturalist 101:543–546.

Hayes, J.P., E.G. Horvath, and P. Hounihan. 1995. Townsend's chipmunk populations in Douglas–fir plantations and mature forests in the Oregon Coast Range. Canadian Journal of Zoology 73:67–73.

Hayhoe, K., D. Cayan, C.B. Field, P.C. Frumhoff, E.P. Maurer, N.L. Miller, S.C. Moser, S.H. Schneider, K.N. Cahill, E.E. Cleland, L. Dale, R. Drapek, R.M. Hanemann, L.S. Kalkstein, J. Lenihan, C.K. Lunch, R.P. Neilson, S.C. Sheridan, and J.H. Verville. 2004. Emissions pathways, climate change, and impacts on California. Proceedings of the National Academy of Sciences of the United States of America 101:12422–12427.

Hearn, B.J., D.J. Harrison, A.K. Fuller, C.G. Lundrigan, and W.J. Curran. 2010. Paradigm shifts in habitat ecology of threatened Newfoundland martens. Journal of Wildlife Management 74:719–728.

Hemphill, D.V. 1952. The vertebrate fauna of the boreal areas of the southern Yolla Bolly Mountains, California. M.S. thesis. Oregon State College, Corvallis, OR, USA.

Hiller, T.L. 2011. Oregon furbearer program report, 2010–2011. Oregon Department of Fish and Wildlife. Salem, OR, USA.

Huff, M.H., and J.K. Smith. 2000. Chapter 5: fire effects on animal communities. Pages 35–42 *in* J.K. Smith (editor), Wildland fire in ecosystems: Effects of fire on fauna. General Technical Report. RMRS–GTR–42–vol.1. U.S. Department of Agriculture, Forest Service. Rocky Mountain Research Station, Ogden, UT, USA.

Hunt, K.A., D.M. Bird, P. Mineau, and L. Shutt. 1992. Selective predation of organophosphate–exposed prey by American kestrels. Animal Behaviour 43(6):971–976.

Innes, J.C., M.P. North, and N. Williamson. 2006. Effect of thinning and prescribed fire restoration treatments on woody debris and snag dynamics in a Sierran old–growth, mixed–conifer forest. Canadian Journal of Forest Research 36:3183–3193.

International Union for Conservation of Nature (IUCN). 2017. IUCN Guidelines for Using the IUCN Red List Categories and Criteria. Version 13. Prepared by the Standards and Petitions Subcommittee. Downloadable from http://www.iucnredlist.org/technical–documents/red–list–documents

Iossa, G., C.C. Soulsbury, and S. Harris. 2007. Mammal trapping: a review of animal welfare standards of killing and restraining traps. Animal Welfare 16:335–352

IPCC (Intergovernmental Panel on Climate Change). 2014. Climate change 2014: synthesis report. Contribution of working groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate change (Core writing team, R.K. Pachauri and L.A. Meyer (editors)). IPCC, Geneva, Switzerland. 151 pp.

Jimerson, T.M., L.D. Hoover, E.A. McGee, G. DeNitto, and R.M. Creasy. 1995. A field guide to serpentine plant associations and sensitive plants in northwestern California. U.S. Forest Service, Pacific Southwest Region.

Johnson, C.A. 2008. Mammalian dispersal behavior and its fitness correlates. Ph.D. dissertation. University of Guelph. Guelph, ON, Canada.

Johnson, C.A., J. M. Fryxell, I.D. Thompson, and J.A. Baker. 2009. Mortality risk increases with natal dispersal in American martens. Proceedings of the Royal Society of Biological Sciences 276:3361–3367.

Johnstone, J.A., and T.E. Dawson. 2010. Climatic context and ecological implications of summer fog decline in the coast redwood region. Proceedings of the National Academy of Sciences of the United States of America 107:4533–4538.

Katnik, D.D. 1992. Spatial use, territoriality, and summer–autumn selection of habitat in an intensively harvested population of martens on commercial forestland in Maine. M.S. thesis. University of Maine. Orono, ME, USA.

Katnik, D.D., D.J. Harrison and T.P. Hodgman. 1994. Spatial relations in a harvested population of martens in Maine. Journal of Wildlife Management 58:600–607.

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

Kirk, T.A., and W.J. Zielinski. 2009. Developing and testing a landscape habitat suitability model for the American marten (*Martes americana*) in the Cascades mountains of California. Landscape Ecology 24:759–773.

Lacy, R.C. and T.W. Clark. 1993. Simulation modeling of American marten (*Martes americana*) populations: vulnerability to extinction. Great Basin Naturalist. 53: 283–293.

Lafferty, K.D. 2008. Effects of disease on community interactions and food web structure. Pp. 205–222 in Infectious disease ecology: effects of ecosystems on disease and of diseases on ecosystems. R.S. Ostfeld, F. Keesing, and V.T. Eviner, editors. Princeton University Press, New Jersey.

Lawler, J.J., H.D. Safford, and E.H. Girvetz. 2012. Martens and fishers in a changing climate. Pages 371–397 *in* K.B. Aubry, W.J. Zielinski, M.G. Raphael, G. Proulx, and S.W. Buskirk (editors), Martens, sables, and fishers: a new synthesis. Cornell University Press. Ithaca, NY.

Lehmkuhl, J.F., K.D. Kistler, J.S> Begley, and J. Boulanger. 2006. Demography of northern flying squirrels informs ecosystem management of western interior forests. Ecological Applications 16:584–600.

LEMMA (Landscape Ecology, Modeling, Mapping, and Analysis). 2018. Data available for download at https://lemma.forestry.oregonstate.edu/data. Data downloaded March 16, 2018.

Lenihan, J.M., D. Bachelet, R. Neilson, and R. Drapek. 2008. Simulated response of conterminous United States ecosystems to climate change at different levels of fire suppression, CO_2 emission rate, and growth response to CO_2. Global Planetary Change 64:16–25.

Linnell, M.A., K. Moriarty, D.S. Green, and T. Levi. 2018. Density and population viability of coastal marten: a rare and geographically isolated small carnivore. PeerJ 6:4530; DOI 10.7717/peerj.4530.

Littell, J.S., M.M. Elsner, G.S. Mauger, E. Lutz, A.F. Hamlet, and E. Salathé. 2011. Regional Climate and Hydrologic Change in the Northern US Rockies and Pacific Northwest: Internally Consistent Projections of Future Climate for Resource Management. Climate Impacts Group, College of the Environment. University of Washington, Seattle, WA.

Lofroth, E.C., C.M. Raley, J.M. Higley, R.L. Truex, J.S. Yaeger, J.C. Lewis, P.J. Happe, L.L. Finley, R.H. Naney, L.J. Hale, A.L. Krause, S.A. Livingston, A.M. Myers, and R. N. Brown. 2010. Conservation of fishers (*Martes pennanti*) in south–central British Columbia, western Washington, western Oregon, and California—volume I: Conservation assessment. USDI Bureau of Land Management, Denver, Colorado.

Long, C.J. and C. Whitlock. 2002. Fire and vegetation history from the coastal rain forest of the western Oregon Coast Range. Quaternary Research 58:215–225.

Long, C.J., C. Whitlock, P.J. Bartlein, and S.H. Millspaugh. 1998. A 9000–year fire history from the Oregon Coast Range, based on a high–resolution charcoal study. Canadian Journal of Forest Research 28:774–787.

Lucid, M., S. Cushman, L. Robinson, A. Kortello, D. Hausleitner, G. Mowat, S. Ehlers, S. Gillespie, L.K. Sullivan, A. Rankin, and D. Paetkau. 2020. Carnivore Contact: a species fracture zone delineated amongst genetically structured North American marten populations (*Martes americana* and *Martes caurina*). Frontiers in Genetics 11.

Luoma, D.L., J.M. Trappe, A.W. Claridge, K.M. Jacobs and E. Cazares. 2003. Relationships among fungi and small mammals in forested ecosystems. Pages 343–373 *in* C.J. Zabel and R.G. Anthony (editors), Mammal community dynamics: management and conservation in the coniferous forests of western North America. Cambridge University Press. Cambridge, UK.

Lyon, L.J., M.H. Huff, E.S. Telfer, D.S. Schreiner, and J.K. Smith. 2000a. Chapter 4: fire effects on animal populations. Pages 25–34 *in* J.K. Smith (editor), Wildland fire in ecosystems: Effects of fire on fauna. General Technical Report. RMRS–GTR–42–vol.1. U.S. Department of Agriculture, Forest Service. Rocky Mountain Research Station, Ogden, UT, USA.

Manlick, P.J., J.E. Woodford, B. Zuckerberg, and J.N. Pauli. 2017. Niche compression intensifies competition between reintroduced American martens (*Martes Americana*) and fishers (*Pekania pennanti*). Journal of Mammalogy 98:690–702.

Mackintosh, C.G., F.J. Laas, M.E.R. Godfrey, and K. Turner. 1988. Vitamin K1 treatment of brodifacoum poisoning in dogs. Pp. 86–90 *In:* Proceedings of the Thirteenth Vertebrate Pest Conference.

Marshall, D.B. 1994. Status of the American marten in Oregon and Washington. Audubon Society. Portland, OR, USA.

Markley, M.H., and C.F. Bassett. 1942. Habits of captive marten. American Midland Naturalist 28(3):604–616.

Martin, M.E., K.M. Moriarty, and J.N. Pauli. 2020. Forest structure and snow depth alter the movement patterns and subsequent expenditures of a forest carnivore, the Pacific marten. Oikos 129:356-366.

Martin, S.K. 1994. Feeding ecology of American martens and fishers. Pages 297–315 in S. Buskirk, A. Harestad, M. Raphael and R. Powell, editors, Martens, sables, and fishers. Comstock Publishing Associates, Cornell University Press, Ithaca, NY, USA.

Maser, C., B.R. Mate, J.E. Franklin, and C.T. Dyrness. 1981. Natural history of Oregon coast mammals. General Technical Report PNW–133. U.S. Department of Agriculture, Forest Service. Pacific Northwest Forest and Range Experiment Station, Portland, OR, USA.

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

McNicol, C.M., D. Bavin, S. Bearhop, J. Bridges, E. Croose, R. Gill, C.E. Goodwin, J. Lewis, J. MacPherson, and D. Padfield. 2020. Postrelease movement and habitat selection of translocated pine martens *Martes martes*. Ecology and Evolution 10:5106-5118.

Minta, S.C., P.M. Kareiva, and A.P. Curlee. 1999. Carnivore research and conservation: learning from history and theory. Pages 323-404 *in* T.W. Clark, A.P. Curlee, S.C. Minta and P.M. Kareiva, editors. Carnivores in ecosystems: the Yellowstone experience. Yale University Press, London, United Kingdom.

Monroe, M.E., and S.J. Converse. 2006. The effects of early season and late season prescribed fires on small mammals in a Sierra Nevada mixed conifer forest. Forest Ecology and Management. 236:229–240.

Mooney, H. and T.E. Dawson. 2016. Coast Redwood Forests. Pp. 535–552 in, H. Mooney and E. Zavaleta (editors), Ecosystems of California. University of California Press. Oakland, California. 984 pp.

Moriarty, K.M., C.W. Epps, M.G. Betts, D.J. Hance, J.D. Bailey, and W.J. Zielinski. 2015. Experimental evidence that simplified forest structure interacts with snow cover to influence functional connectivity for Pacific martens. Landscape Ecology 30:1865–1877.

Moriarty, K.M., C.W. Epps, and W.J. Zielinski. 2016a. Forest thinning changes movement patterns and habitat use by Pacific marten. The Journal of Wildlife Management 80:621–633

Moriarty, K.M., J.D. Bailey, S.E. Smythe, and J. Verschuyl. 2016b. Distribution of Pacific marten in coastal Oregon. Northwestern Naturalist 97:71–81.

Moriarty, K.M., M.S. Delheimer, P.J. Tweedy, K. Credo, J.D. Bailey, M.E. Martin, A.M. Roddy, and B.V. Woodruff. 2017a. Identifying opportunities to increase forest resilience, decrease fire risk, and manage for Pacific martens (*Martes caurina*) population persistence within the Lassen National Forest, California. Draft Report: USDA Forest Service, Pacific Northwest Research Station.

Moriarty, K.M., M.A. Linnell, B.E. Chasco, C.W. Epps, and W.J. Zielinski. 2017b. Using high–resolution short–term location data to describe territoriality in Pacific martens. Journal of Mammalogy 98:679–689.

Moriarty, K.M, J. Verschuyl, A.J. Kroll, R. Davis, J. Chapman, and B. Hollen. 2019. Describing vegetation characteristics used by two rare, forest–dwelling species: Will established reserves provide for coastal marten in Oregon? PLoS ONE 14(1): e0210865. https://doi.org/10.1371/journal.pone0210865.

Moriarty, K.M., B.R. Barry, D.S. Green, and T. Levi. 2021a. Assessing evidence of risk: using DNA metabarcoding (diet) and predicted occupancy overlap to evaluate competition and predation risk of fishers and martens, *abstract within*: Conference Western Section of the Wildlife Society Annual Meeting. Virtual, recorded presentation available on Whova.

Moriarty, K. M., J. Thompson, M. Delheimer, B. R. Barry, M. Linnell, T. Levi, K. Hamm, D. Early, H. Gamblin, M. Szykman Gunther, J. Ellison, J. S. Prevéy, J. Hartman, and R. Davis. 2021b. Predicted distribution of a rare and understudied forest carnivore: Humboldt marten (*Martes caurina humboldtensis*). PeerJ 9:e11670.

Moritz, C., J.L. Patton, C.J. Conroy, J.L. Parra, G.C. White, and S.R. Beissinger. 2008. Impact of a century of climate change on small–mammal communities in Yosemite National Park, USA. Science 322:261–264.

Morris, W.G. 1934. Forest fires in western Oregon and western Washington. Oregon Historical Quarterly 35:313–339.

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

Munday, J.S., and L.J. Thompson. 2003. Brodifacoum toxicosis in two neonatal puppies. Veterinary Pathology Online 40(2):216–219.

Naney, R.H., L.L. Finley, E.C. Lofroth, P.J. Happe, A.L. Krause, C.M. Raley, R.L. Truex, L.J. Hale, J.M Higley, A.D. Kosic, J.C. Lewis, S.A. Livingston, D.C. Macfarlane, A.M. Meyers, and J.S. Yaeger. 2012. Conservation of fishers (*Martes pennant*) in south–central British Columbia, western Washington, western Oregon, and California—volume III: threat assessment. USDI Bureau of Land Management, Denver, Colorado.55 pp.

ODGMI (Oregon Department of Geology and Mineral Industries) 2012. Local source (Cascadia subduction zone) tsunami inundation map: Lakeside West, Oregon. Oregon Department of Geology and Mineral Industries, Portland, Oregon.

ODOT (Oregon Department of Transportation). 2016. Summary of Trends at automatic traffic recorder stations. Downloaded 26 February 2018 from: http://www.oregon.gov/ODOT/Data/Documents/Summary_2016.pdf

OEA (Office of Economic Analysis). 2017. Oregon's demographic trends. Office of Economic Analysis, Department of Administrative Services, State of Oregon, Salem, Oregon.

OHA (Oregon Health Authority). 2018. Selected reportable communicable disease summary. Oregon Health Authority, Public Health Division, Center for Public Health Practice. Portland, OR.

O'Grady, J.J., D.H. Reed, B.W. Brook, and R. Frankham. 2008. Extinction risk scales better to generations than to years. Animal Conservation. 11: 442–451.

OSU (Oregon State University). 2016. Unpublished OSU Veterinary Diagnostic Laboratory necropsy report performed on three martens. Corvallis, Oregon.

Pauli, J.N., W.P. Smith and M. Ben–David. 2012. Quantifying dispersal rates and distances in North American martens: a test of enriched isotope labeling. Journal of Mammalogy 93:390–398.

Pauli, R.M., J.B. Lian, D.F. Mosher, and J.W. Suttie. 1987. Association of congenital deficiency of multiple vitamin K–dependent coagulation factors and the phenotype of the warfarin embryopathy: clues to the mechanism of teratogenicity of coumarin derivatives. American Journal of Human Genetics 41(4):566–583.

Payer, D.C. and D.J. Harrison. 2003. Influence of forest structure on habitat use by American marten in an industrial forest. Forest Ecology and Management 179:145–156.

Perry, D.A., P.F. Hessburg, C.N. Skinner, T.A. Spies, S.L. Stephens, A.H. Taylor, J.F. Franklin, B. McComb, and G. Riegel. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and northern California. Forest Ecology and Management 262:703–717.

Phillips, D.M. 1994. Social and spatial characteristics, and dispersal of marten in a forest preserve and industrial forest. M.S. thesis. University of Maine, Orono, ME, USA.

Phys.Org. 2016. Subduction zone earthquakes off Oregon, Washington more frequent than previous estimates. Downloaded April 9, 2018 from: https://phys.org/news/2016–08–subduction–zone–earthquakes–oregon–washington.html

Pierce, D.W., D.R. Cayan, T. Das, E.P. Maurer, N.L. Miller, Y. Bao, M. Kanamitsu, K. Yoshimura, M.A. Snyder, L.C. Sloan, G. Franco, and M. Tyree. 2013a. Probabilistic estimates of future changes in California temperature and precipitation using statistical and dynamical downscaling. Climate Dynamics 40**:**839–856.

Pierce, D.W., D.R. Cayan, T. Das, E.P. Maurer, N.L. Miller, Y. Bao, M. Kanamitsu, K. Yoshimura, M.A. Snyder, L.C. Sloan, G. Franco, and M. Tyree. 2013b. The key role of heavy precipitation events in climate model disagreements of future annual precipitation changes in California. Journal of Climate 26(6):5879–5896.

Potvin, F., and L. Breton. 1997. Short–term effects of clearcutting on martens and their prey in the boreal forest of western Quebec. Pages 452–474 *in* G. Proulx, H. Bryant, and P. Woodard (editors), *Martes*: taxonomy, ecology, techniques and management. Provincial Museum of Alberta. Edmonton, Alberta, Canada.

Potvin, F., L. Belanger, and K. Lowell. 2000. Marten habitat selection in a clearcut boreal landscape. Conservation Biology 14:844–857.

Powell, R.A. 1994. Structure and spacing of *Martes* populations. In martens, sables and fishers: biology and conservation: Pages 101–121 *in* Buskirk, S.W., Harestad, A.S., Raphael, M.G., and Powell, R.A. (Eds.). Cornell University Press, Ithaca, NY.

Powell, R.A., S.W. Buskirk, and W.J. Zielinski. 2003. Fisher and Marten (*Martes pennanti* and *Martes americana*). Pages 635–649 in G. Feldhamer, B. Thompson, and J. Chapman, editors. Wild mammals of North America, 2nd Ed. Johns Hopkins University Press. Baltimore, MD, USA

Powell, R.A., and G. Proulx. 2003. Trapping and marking terrestrial mammals for research: integrating ethics, performance criteria, techniques, and common sense. ILAR Journal 44:259–276.

Priest, G.R., R.C. Witter, Y.J. Zhang, K. Wang, C. Goldfinger, L.L. Stimely, J.T. English, S.G. Pickner, K.L.B. Hughes, T.E. Wille, and R.L. Smith. Open–file report O–13–19, tsunami inundation scenarios for Oregon. Oregon Department of Geology and Mineral Industries, Salem, Oregon. GIS layers and metadata downloaded April 9, 2018 from: http://www.oregongeology.org/pubs/ofr/p–O–13–19.htm

Proulx, G., K. Aubry, J. Birks, S. Buskirk, C. Fortin, H. Frost, W. Krohn, L. Mayo, V. Monakhov, D. Payer, M. Saeki, M. Santos–Reis, R. Weir, and W. Zielinski. 2004. World distribution and status of the genus Martes in 2000. Pages 21–76 in D.J. Harrison, A.K. Fuller, and G. Proulx (Editors), Martens and fishers (Martes) in human–altered environments: an international perspective. Springer Science+Business Media. New York, NY, USA.

Pyne, S.J. 1982. Fire in America: a cultural history of wildland and rural fire. Princeton University Press, Princeton, New Jersey. 654 pp.

Rady, G.H., F.S. El–Mahrouky, H.M. Abdelnabby, and H.A. Ahmed. 2013. Sub–Lethal and Teratogenicity Action of Bromadiolone and Chlorophacinone Anticoagulant Rodenticides on Albino Rats. American–Eurasian Journal of Toxicological Sciences 5(1):7–14.

Raphael, M.G. 2004. Ecology of the American marten in the Oregon Cascade Range (Presentation Abstract). Page 27 *in* Programme and Abstracts of the Fourth International *Martes* Symposium. Faculty of Sciences, University of Lisbon, Portugal. [Available online]: http://www.carnivoreconservation.org/files/meetings/*Martes*_2004_lisbon.pdf.

Redford, K.H., G. Amoto, J. Baillie, P. Beldomenico, E.L. Bennett, B. Clum, R. Cook, G. Fonseca, S. Hedges, F. Launay, S. Lieberman, G.M. Mace, A. Murayama, A. Putnam, J.G. Robinson, H. Rosenbaum, E.W. Sanderson, S.N. Stuart, P. Thomas, and J. Thorbjarnarson. 2011. What does it mean to successfully conserve a (vertebrate) species? Bioscience 61:39–48.

Reed, D.H., J.J. O'Grady, B.W. Brook, J.D. Ballou, and R. Frankham. 2003. Estimates of minimum viable population sizes for vertebrates and factors influencing those estimates. Biological Conservation. 113: 23–34.

Ricketts, T.H., E. Dinerstein, D.M. Olson, C.J. Loucks, W. Eichenbaum, D. DellaSala, K. Kavanaugh, P. Hedao, P. Hurley, K.M. Carney, R. Abell, and S. Walters. 1999. Terrestrial ecoregions of North America: a conservation assessment. Island Press, Washington DC and Covelo, CA, USA.

Ripple, W.J. 1994. Historic spatial patterns of old forests in western Oregon. Journal of Forestry 92:45–49.

Robitaille, J. and K. Aubry. 2000. Occurrence and activity of American martens *Martes americana* in relation to roads and other routes. Acta Theriologica 45:137–143.

Rogers, C. 2016. The impact of climate change on coastal fog hours of California's central coast. M.S. Thesis, San José State University, San José, California, USA.

Rosenberg, D.K., K.A. Swindle and R.G. Anthony. 1994. Habitat associations of California redbacked voles in young and old–growth forests in western Oregon. Northwest Science 68(4):266–272.

Ruggerio, L.F., D.E. Pearson, and S.E. Henry. 1998. Characteristics of American marten dens in Wyoming. Journal of Wildlife Management 62(2): 663–673.

Salathé, E.P., Jr., L.R. Leung, Y. Qian, and Y. Zhang. 2010. Regional climate model projections for the State of Washington. Climatic Change 102**:**51–75.

Santos, S.M., F. Carvalho, and A. Mira. 2011. How long do the dead survive on the road? Carcass persistence probability and implications for road–kill monitoring surveys. PLoS ONE 6(9): e25383. doi:10.1371/journal.pone.0025383

Schrott, G.R. and J.M. Shinn. 2018. Coastal marten connectivity analysis summary. 7 p. Unpublished report. On file with: U.S. Fish and Wildlife Service, Arcata Field Office, 1655 Heindon Road, Arcata, CA 95521.

Schumacher, T.V. 1999. A multi–scale analysis of habitat selection at den and resting sites of American martens in southeast Alaska. M.S. thesis. University of Wyoming. Laramie, WY, USA.

Schwartz, M.K.; Slauson, K.M.; Pilgrim, K.; Zielinski, W.J. [et al.]. 2016. Genomic evidence showing the California Coast/Oregon Coast Populations of Pacific Marten Representing a Single Conservation Unit. 38 p. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, 800 E. Beckwith Avenue, Missoula, MT 59802.

Schwartz, M.K., A.D. Walters, K.L. Pilgrim, K.M. Moriarty, K.M. Slauson, W.J. Zielinski, K.B. Aubry, B.N. Sacks, K.E. Zarn, and C.B. Quinn. 2020. Pliocene-early Pleistocene geological events structure Pacific martens (*Martes caurina*). The Journal of Hereditary 111: 169-181.

Self, S., and S. Kerns. 2001. Pine marten use of a managed forest landscape in Northern California. Wildland Research Paper No. 4. Sierra Pacific Industries, Redding, CA, USA.

Shafer, S.L., M.E. Harmon, R.P. Neilson, R. Seidl, B. St. Clair, and A. Yost. 2010. The potential effects of climate change on Oregon's vegetation. Pages 173–208 *in* K.D. Dello and P.W. Mote (editors), Oregon Climate Assessment Report. Oregon Climate Change Research Institute. College of Oceanic and Atmospheric Sciences, Oregon State University, Corvallis, OR.

Shaffer, M.L. 1981. Minimum population sizes for species conservation. Bioscience. 31:131–134.

Sheehan, T., D. Bachelet, and K. Ferschweiler. 2015. Projected major fire and vegetation changes in the Pacific Northwest of the conterminous United States under selected CMIP5 climate futures. Ecological Modeling 317:16–29.

Simon, T.L. 1980. An ecological study of the pine marten in the Tahoe National Forest. M.S. thesis. California State University, Sacramento, CA, USA.

Sirén, A.P.K., P.J. Pekins, M.J. Ducey, and J.R. Kilborn. 2016. Spatial ecology and resource selection of a high–elevation American marten (*Martes Americana*) population in the northeastern United States. Canadian Journal of Zoology 94:169–180.

Slauson, K.M. 2003. Habitat selection by American martens (*Martes americana*) in coastal northwestern California. M.S. thesis. Oregon State University, Corvallis, OR, USA.

Slauson, K.M. 2017. Linking landscape pattern to population processes in a carnivorous mammal. Ph.D. dissertation. University of Montana, Missoula, MT. 173 pp.

Slauson, K.M., W.J. Zielinski, and J.P. Hayes. 2007. Habitat selection by American martens in coastal California. Journal of Wildlife Management 71:458–468.

Slauson, K.M., G.A. Schmidt, W.J. Zielinski, P.J. Detrich, R.L. Callas, J. Thrailkill, B. Devlin–Craig, D.A. Early, K.A. Hamm, K.N. Schmidt, A. Transou, and C.J. West. 2019(a). A conservation assessment and strategy for the Humboldt marten (*Martes caurina humboldtensis*) in California and Oregon. General Technical Report PSW–GTR–260. Arcata, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 124 p.

Slauson, K.M., W.J. Zielinski, T.A. Kirk, and D.W. LaPlante. 2019(b). A landscape habitat suitability model for the Humboldt marten (*Martes caurina humboldtensis)* in coastal California and coastal Oregon. Northwest Science 93: 30–51.

Slauson, K. M., and W. J. Zielinski. 2004. Conservation status of American martens and fishers in the Klamath–Siskiyou bioregion. Pages 60–70 in K. Merganther, J. Williams, and E. Jules (editors), Proceedings of the 2nd conference on Klamath–Siskiyou ecology, Cave Junction, OR, USA. May 29–31, 2003. Siskiyou Field Institute, Cave Junction, Oregon.

Slauson, K.M., and W.J. Zielinski. 2007. The relationship between the understory shrub component of coastal forests and the conservation of forest carnivores. Pages 241–243 *in* R.B. Standiford, G.A. Giusti, Y. Valachovic, W.J. Zielinski, W.J. Furniss, Technical Editors. Proceedings of the redwood region forest science symposium: what does the future hold? General Technical Report PSW–GTR–194. U.S. Department of Agriculture, Forest Service. Pacific Southwest Research Station. Albany, CA, USA.

Slauson, K.M., and W.J. Zielinski. 2009. Characteristics of summer/fall resting structures used by American martens in coastal northwestern California. Northwest Science 83:35–45.

Slauson, K.M.; Zielinski, W.J.; and Kirk, T.A. 2010. Effects of forest restoration on mesocarnivores in the northern Redwood Region of northwestern California. Unpublished final report. On file with: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Redwood Sciences Laboratory, 1700 Bayview Drive, Arcata, CA 95521. 29 pp.

Slauson, K.M., W.J. Zielinski, and K.D. Stone. 2009a. Characterizing the molecular variation among American marten (*Martes americana*) subspecies from Oregon and California. Conservation Genetics 10:1337–1341.

Slauson, K.M., J.A. Baldwin, W.J. Zielinski, and T.A. Kirk. 2009b. Status and estimated size of the only remnant population of the Humboldt subspecies of the American marten (*Martes Americana humboldtensis*) in northwestern California: final report. U.S.D.A. Forest Service, Pacific Southwest Research Station. Arcata, CA. 28 pp.

Slauson, K.M.; Zielinski, W.J.; Early, D.A. [et al.]. 2014. Humboldt marten dispersal and movement ecology study: Progress Report. 6 p. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Redwood Sciences Laboratory, 1700 Bayview Drive, Arcata, CA 95521.

Slauson, K.M. and W.J. Zielinski. 2017. Seasonal specialization in diet of the Humboldt marten (*Martes caurina humboldtensis*) in California and the importance of prey size. Journal of Mammalogy 98(6):1697-1708.

Slauson, K.M. *In prep*. Habitat mediates interactions between Humboldt martens and bobcats.

Slough, B.G. 1989. Movements and habitat use by transplanted marten in the Yukon Territory. Journal of Wildlife Management 53:991–997.

Smith, D.R., N.L. Allan, C.P. McGowan, J. Szymanski, S.R. Oetker, and H.M. Bell. 2018. Development of a Species Status Assessment Process for Decisions under the U.S. Endangered Species Act. Journal of Fish and Wildlife Management In–Press.

Smith, J.E., R. Molina, M.M.P. Huso, D.L. Luoma, D. McKay, M.A. Castellano, T. Lebel and Y. Valachovic. 2002. Species richness, abundance, and composition of hypogenous and epigeous ectomycorrhizal fungal sporocarps in young, rotation–age, and old–growth stands of Douglas–fir (*Pseudotsuga menziesii*) in the Cascade Range of Oregon, USA. Canadian Journal of Botany 80:186–204.

Smith–Patten, B.D., and M.A. Patten. 2008. Diversity, seasonality, and context of mammalian roadkills in the southern great plains. Environmental Management 41:844–852.

Snyder, J., and J.A. Bissonette. 1987. Marten use of clear–cuttings and residual forest stands in western Newfoundland. Canadian Journal of Zoology 65:169–174.

Spencer, W.D. 1987. Seasonal rest–site preferences of pine martens in the northern Sierra Nevada. Journal Wildlife Management 51:616–621.

Spencer, W.D., R.H. Barrett, and W.J. Zielinski. 1983. Marten habitat preferences in the northern Sierra Nevada. Journal Wildlife Management 47:1181–1186.

Spencer, W., H. Rustigian–Romsos, J. Strittholt, R. Scheller, W.J. Zielinski, and R. Truex. 2011. Using occupancy and population models to assess habitat conservation opportunities for an isolated carnivore population. Biological Conservation 144:788–803.

Spies, T.A., B.C. McComb, R.S.H. Kennedy, M.T. McGrath, K. Olsen, and R.J. Pabst. 2007. Potential effects of forest policies on terrestrial biodiversity in a multi–ownership province. Ecological Applications 17:48–65.

Stamps, J.A. 2001. Habitat selection by dispersers: integrating proximate and ultimate approaches. Pages 230–242 *in* J. Clobert, E. Danchin, A.A. Dhondt and J.D. Nichols (editors), Dispersal. Oxford University Press. New York, NY, USA. Read–only file available at:
http://www.researchgate.net/publication/264623013_Habitat_selection_by_dispersers_int egrating_proximate_and_ultimate_approaches

Steventon, J.D., and J.T. Major. 1982. Marten use of habitat in a commercially clear–cut forest. Journal of Wildlife Management 46(1):175–182.

Stewart, J.A.E., J.H. Thorne, M. Gogol–Prokurat, and S.D. Osborn. 2016. A climate change vulnerability assessment for twenty California mammal taxa. Information Center for the Environment, University of California, Davis, CA.

Strickland, M.A. 1994. Harvest management of fishers and American martens. Pages 149–164 in S. Buskirk, A. Harestad, M. Raphael and R. Powell (editors), Martens, sables, and fishers. Biology and Conservation. Comstock Publishing Associates, Cornell University Press, Ithaca, NY, USA.

Strickland, M.A., C.W. Douglas, M. Novak, and N.P. Hunziger. 1982. Marten (*Martes americana*). Pages 599–612 in J.A. Chapman and G.A. Feldhamer (editors), Wild mammals of North America: biology, management and economics. Johns Hopkins University Press. Baltimore, MD, USA.

Strickland, M.A., and C.W. Douglas. 1987. Marten. Pages 531–546 in M. Novak, J.A. Baker, M.E. Obbard and B. Malloch (editors), Wild furbearer management and conservation in North America. Ontario Trappers Association, North Bay, Ontario, CN.

Strittholt, J.R., D.A. DellaSala, and H. Jiang. 2006. Status of mature and old–growth forests in the Pacific Northwest. Conservation Biology 20:363–374.

Sweitzer, R.A., V.D. Popescu, C.M. Thompson, K.L. Purcell, R.H. Barrett, G.M. Wengert, M.W. Gabriel, and L.W. Woods. 2016. Mortality risks and limits to population growth of fishers. The Journal of Wildlife Management 80:438–451.

Taylor, I. 2009. How owls select their prey: a study of barn owls *Tyto alba* and their small mammal prey. Ardea 97:635–644.

Taylor, S.L. 1993. Thermodynamics and energetics of resting site use by the American marten (*Martes americana*). M.S. thesis. University of Wyoming. Laramie, WY, USA.

Taylor, S.L., and S.W. Buskirk. 1994. Forest microenvironments and resting energetics of the American marten *Martes americana*. Ecography 17:249–256.

Tebaldi, C., K. Hayhoe, J.M. Arblaster, and G.A. Meehl. 2006. Going to the Extremes. Climatic Change 79:185–211.

Teensma, P.D.A., J.T. Rienstra, and M.A. Yeiter. 1991. Preliminary reconstruction and analysis of change in forest stand age classes of the Oregon Coast Range from 1850 to 1940. Technical Note T/N OR–9. US Department of the Interior Bureau of Land Management, Oregon State Office, Portland, OR.

Teixeira, F.Z., A.V.P. Coelho, I.B. Esperandio, and A. Kindel. 2013. Vertebrate road mortality estimates: effects of sampling methods and carcass removal. Biological Conservation. 157: 317–323

Temple, S.A. 1987. Do predators always capture substandard individuals disproportionately from prey populations? Ecology 68:669–674.

Thompson, C., Sweitzer, R., Gabriel, M., Purcell, K., Barrett, R. and Poppenga, R. 2014. Impacts of rodenticide and insecticide toxicants from marijuana cultivation sites on fisher survival rates in the Sierra National Forest, California. Conservation Letters 7: 91–102.

Thompson, I.D., and P.W. Colgan. 1987. Numerical responses of martens to a food shortage in north–central Ontario. Journal of Wildlife Management 51:824–835.

Thompson, I.D., J. Fryxell, and D. J. Harrison. 2012. Improved insights into use of habitat by American martens. Pages 209–230 *in* K. Aubry, W. Zielinski, M. Raphael, G. Proulx, and S. Buskirk (editors), Biology and conservation of martens, sables, and fishers: a new synthesis. Cornell University Press. Ithaca, NY, USA.

Thompson, I.D. and A.S. Harestad. 1994. Effects of logging on American martens, and models for habitat management. Pp. 355–367 in, S.W. Buskirk, A.S. Harestad, M.G. Raphael, and R.A. Powell, Martens, sables, and fishers: biology and conservation. Cornell University Press, Ithaca, New York. 484 pp.

Thompson, I.D. 1994. Marten populations in uncut and logged boreal forests in Ontario. Journal of Wildlife Management 58:272–280.

Thorne, E.T., N. Kingston, W.R. Jolley, and R.C. Berbstrom. 1982. Diseases of wildlife in Wyoming. Wyoming Game and Fish Department, Cheyenne, WY.

Traill, L.W., C.J.A. Bradshaw, and B.W. Brook. 2007. Minimum viable population size: a meta–analysis of thirty years of published estimates. Biological Conservation. 139:159–166.

Tweedy, P.J. 2018. Diel rest structure selection and multiscale analysis of Pacific marten resting habitat in Lassen National Forest, California. M.S. thesis, Oregon State University. Corvallis, OR, USA.

Twining, H., and A. Hensley. 1947. The status of pine martens in California. California Fish and Game 33:133–137.

U.S. Fish and Wildlife Service (USFWS). 1997. Recovery plan for the threatened marbled murrelet (*Brachyramphus marmoratus*) in Washington, Oregon, and California. Portland, Oregon. 203 pp.

U.S. Fish and Wildlife Service (USFWS). 2015a. USFWS species status assessment framework: an integrated analytical framework for conservation. Version 3.3, dated August 2015.

U.S. Fish and Wildlife Service (USFWS). 2015b. Coastal Oregon and northern coastal California populations of the Pacific Marten (*Martes caurina*) species report. U.S. Fish and Wildlife Service, Region 8, Arcata Fish and Wildlife Office, Arcata, CA.

U.S. Fish and Wildlife Service (USFWS). 2016. Final species report: fisher (*Pekania pennant*), west coast population. U.S. Fish and Wildlife Service, Yreka Fish and Wildlife Office, Yreka, CA.

USFS 2017. Letter from U.S. Forest Service, Oregon Dunes National Recreation Area to Oregon Dunes stakeholders, dated December 14, 2017, subject, "Oregon Dunes Restoration Project Scoping Letter."

Varghese, R. 2017. California localities primed for legal recreational cannabis use. Bloomberg. Downloaded May 1, 2018 from: https://www.bloomberg.com/news/articles/2017–06–05/california–localities–primed–for–legal–recreational–cannabis–use

Verts, B.J., and L.N. Carraway. 1998. Land Mammals of Oregon. University of California Press, Berkeley, California.

Wasserman, T.N., S.A. Cushman, D.O. Wallin, and J. Hayden. 2012a. Multi scale habitat relationships of *Martes americana* in northern Idaho, U.S.A. Research Paper RMRS–RP–94. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 21 pp.

Wasserman, T.N., S.A. Cushman, A.S. Shirk, E.L. Landguth, and J.S. Littell. 2012b. Simulating the effects of climate change on population connectivity of American marten (*Martes americana*) in the northern Rocky Mountains, USA. Landscape Ecology 27:211–225.

Waters, J.R., and C.J. Zabel. 1995. Northern flying squirrel densities in fir forests of northeastern California. Journal of Wildlife Management 59:858–866.

Weir, R.D., M. Phinney, and E.C. Lofroth. 2012. Big, sick, and rotting: why tree size, damage, and decay are important to fisher reproductive habitat. Forest Ecology and Management. 265:230–240.

Wengert, G.M. 2013. Ecology of intraguild predation on fishers (*Martes pennant*) in California. Ph.D. dissertation, University of California, Davis. Davis, CA.

Wengert, G.M., M.W. Gabriel, S.M. Matthews, J.M. Higley, R.A. Sweitzer, C.M. Thompson, K.L. Purcell, R.H. Barrett, L.W. Woods, R.E. Green, S.M. Keller, P.M. Gaffney, M. Jones, and B.N. Sacks. 2014. Using DNA to describe and quantify interspecific killing of fishers in California. The Journal of Wildlife Management 78:603–611.

White, P.J., T.J. Kreeger, U.S. Seal, and J.R. Tester. 1991. Pathological responses of red foxes to capture in box traps. Journal of Wildlife Management. 55:75–80

Wimberly, M.C. 2002. Spatial simulation of historical landscape patterns in coastal forests of the Pacific Northwest. Canadian Journal of Forest Research 32:1316–1328.

Wimberly, M.C. and J.L. Ohmann. 2004. A multi–scale assessment of human and environmental constraints on forest land cover change in the Oregon (USA) coast range. Landscape Ecology 19:631–646.

Wimberly, M.C., T.A. Spies, C.J. Long, and C. Whitlock. 2000. Simulating historical variability in the amount of old forests in the Oregon Coast Range. Conservation Biology 14:167–180.

Wimberly, M.C., T.A. Spies, and E. Nonaka. 2004. Using criteria based on the natural fire regime to evaluate forest management in the Oregon Coast Range of the United States. Pp. 146–157 in, A.H. Perera, L.J. Buse, and M.G. Weber (editors) Emulating Natural Forest Landscape Disturbances: Concepts and Applications. Columbia University Press, New York. 352 pp.

Witter, R.C., Y.J. Zhang, K. Wang, G.R. Priest, C. Goldfinger, L. Stimely, J.T. English, and P.A. Ferro. 2013. Simulated tsunami inundation for a range of Cascadia megathrust earthquake scenarios at Bandon, Oregon, USA. Geosphere 9:1783–1803.

Wolf, S., B. Hartl, C. Carroll, M.C. Neel, and D.N. Greenwald. 2015. Beyond PVA: Why recovery under the Endangered Species Act is more than population viability. Bioscience doi: 10.1093/biosci/biu218 1–8.

Woodroffe, R. 1999. Managing disease threats to wild mammals. Animal Conservation. 2:185–193.

Zielinski, W.J., W.D. Spencer, and R.H. Barrett. 1983. Relationship between food habits and activity patterns of pine martens. American Society of Mammalogists 64(3): 387–396.

Zielinski, W.J., K.M. Slauson, C.R. Carroll, C.J. Kent, and D.K. Kudrna. 2001. Status of American marten populations in the coastal forests of the Pacific States. Journal of Mammalogy 82:478–490.

Zielinski, W.J. and R.T. Golightly. 1996. The status of marten in redwoods: is the Humboldt marten extinct? Pages 115–119 *in* J. LeBlanc (editor), Conference on coast redwood forest ecology & management. June 18–20, 1996. Humboldt State University, Arcata, CA. University of California Cooperative Extension, Forestry. Berkeley, CA, USA.

Zielinski, W.J., J.M Tucker, and K.M. Rennie. 2017. Niche overlap of competing carnivores across climatic gradients and the conservation implications of climate change at geographic range margins. Biological Conservation 209:533–545.

Personal Communications

Bautista, S. 2013. RE: pesticide application question. Electronic mail correspondence, dated December 17, 2013 and December 18, 2013, sent to Sue Livingston, U.S. Fish and Wildlife Service, Portland, Oregon.

Broman, D. 2018. Carnivore–Furbearer Coordinator for Oregon Department of Fish and Wildlife. Letter to Sue Livingston dated January 3, 2018 quantifying marten mortality sources in coastal Oregon counties.

Clayton, D. 2018. Email exchange between Sue Livingston, U.S. Fish and Wildlife Service, and Dave Clayton, Forest Wildlife Biologist, Rogue–Siskiyou National Forest, May 23, 2018.

Davis, R. 2018. Phone conversation between Sue Livingston, U.S. Fish and Wildlife Service, and Ray Davis, Forest Service Monitoring lead for older forests, April 27, 2018.

Moriarty, K. 2018. Conversation between Jenny Hutchinson, U.S. Fish and Wildlife Service, Sue Livingston, U.S. Fish and Wildlife Service, and Katie Moriarty, Pacific Northwest Research Station, during a phone call, March, 8, 2018.

Pilgrim, K. 2018. Electronic mail correspondence dated January 25, 2018, sent to Sue Livingston, U.S. Fish and Wildlife Service, Portland, Oregon. Subject: Coastal Marten SSA comments from Forest Service.

Slauson, K.M. 2014c. Conversation between Gregory Schmidt, U.S. Fish and Wildlife Service, and Keith Slauson, USDA Forest Service, Pacific Southwest Research Station, during an executive meeting of the Humboldt Marten Conservation Group, September 26, 2014.

Slauson, K. 2018. Conversation between Jenny Hutchinson, U.S. Fish and Wildlife Service and Keith Slauson, Pacific Southwest Research Station, during a meeting, February, 15, 2018.

Standley, J. 2013. Electronic mail correspondence dated December 9, 2013, and December 11, 2013, sent to Sue Livingston, U.S. Fish and Wildlife Service, Portland, Oregon. Subject: Use of anticoagulant rodenticides on BLM lands in Oregon and Washington.

Weikel, Jennifer. 2016. Electronic mail correspondence dated June 6, 2016, sent to Sue Livingston, U.S. Fish and Wildlife Service, Portland, Oregon. Subject: OR FPA chemical application reporting.

Unpublished data

Broman, D. 2018. Carnivore Furbearer Coordinator for Oregon Department of Fish and Wildlife. Information provided on ages of marten carcasses collected Oregon–wide, May 29, 2018.

Moriarty, K. 2017. Wildlife researcher with Forest Service Pacific Northwest Research Station. Letter written to Deanna Williams, Siuslaw National Forest wildlife biologist, December 28, 2017, providing information on disease testing of coastal martens.

Williams, Deanna. 2017. Forest wildlife biologist, Siuslaw National Forest. Email to Fish and Wildlife Service dated December 29, 2017 summarizing road killed martens on the Siuslaw National Forest in the vicinity of the central coastal Oregon marten population.