

Pacific Marten Occupancy Study on the Olympic Peninsula 2022–2023

Final Report Submitted to Olympic National Park and Olympic National Forest

July 18, 2024

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Summary

Pacific marten populations began to decline along the US west coast at the turn of the twentieth century due to deforestation and increased mortality rates linked to unregulated trapping for their pelts and the use of poisons targeting larger carnivores. The population of martens on Washington's Olympic Peninsula once occupied forests from the coast to the high elevation tree line. However, owing to anthropogenic pressures, the population has suffered an extreme contraction in its range and is thought to only occupy high elevation forests. Multiple surveys targeting forest carnivores, and specifically martens, since 1991 have yielded very few marten detections. Indeed, between 1968 and 2022, there had been only 25 verified marten detections on the Olympic Peninsula. Furthermore, researchers have a poor understanding of the population's genetic health as only a single recent (2008) sample was available for recent genetic analyses. Here, we implemented a marten occupancy study using a novel technology—automated scent dispensers—which allowed us to conduct surveys through the winter, when mustelids are most likely to be detected, in high elevation forests that would otherwise be inaccessible due to snow and rugged terrain. These automated scent dispensers have proven effective at detecting martens in the Olympics and elsewhere but had not been deployed on a large scale in this region. We also paired passive hair snare devices with the automated scent dispensers, opportunistically collected scat samples, and deployed transects of hair snare devices in sample units in which martens were recently detected. We used a stratified random sampling design and a geographically extensive sampling frame to survey 67 sites in Olympic National Park and Olympic National Forest. We detected martens in only five of the surveyed sample units, further suggesting the limited presence of martens on the Peninsula and precluding our ability to conduct a full occupancy analysis. We collected 41 hair samples and 24 scat samples

which we have sent to the US Forest Service National Genomics Center for Wildlife and Fish Conservation for species identification and microsatellite analysis. Finally, we recommend that 1) future surveys focus on high-elevation locations and on the collection of genetic samples to help guide future management actions, 2) managers begin the petitioning process to have the Olympic marten population listed as threatened or endangered at the federal and state level, and 3) direct management actions should be considered, such as a population augmentation, to increase the population abundance and distribution within the national park and national forest.

Acknowledgements

We are grateful for the support of the many project partners that form the Olympic Marten Working Group: National Park Service and particularly Olympic National Park, USDA Forest Service and particularly Olympic National Forest, Woodland Park Zoo, and University of Washington. This work would not have been possible without the hard work and dedication of field crew members: Calypso Gagorik, Ivy Arden, Cameron Green, Michael O'Neill, and Ben Wymer. We also thank Olympic Park Advocates for their generous support, which allowed us to hire additional technicians and made our workload much more manageable. Finally, a huge thank you to all the volunteers who helped carry gear through steep and rough terrain in their free time: John Bridge, Kyle Glick, Sara Hamilton, Michael Stoneback, Mason White, Cynthia Sheller, and Ryan St. Martin.

Introduction

Understanding the biotic and abiotic drivers of species distribution has become increasingly important as climate change and the direct effects of human activity have accelerated rates of landscape change (Parmesan 2006, Chaudhary and Mooers 2018, IPCC 2019, Jia et al. 2019). Species considered to be habitat specialists are especially vulnerable to

rapidly changing landscapes because habitats that meet their requirements are unlikely to develop at the same pace that they are lost or become inaccessible due to lack of connectivity (Warren et al. 2001, Clavel et al. 2011, Hof et al. 2012, Freeman et al. 2018). Species unable to meet their needs with the resources available in these changing landscapes will experience range contractions that, in extreme cases, can ultimately lead to extirpation (Parmesan 1996, Warren et al. 2001, Bellard et al. 2012, Freeman et al. 2018). Studies of occupancy and resource use at the edges of species distributions have the potential to identify causes of range contractions and quantify thresholds of tolerance for changing landscape variables (Laurance 1997, Pounds et al. 2006, Behnke et al. 2016, Freeman et al. 2018, Mangan et al. 2019, Malhotra et al. 2021). Such studies will be useful for efficiently allocating management resources to conserve species at high risk of local extirpation.

Two marten species—mesocarnivores in the weasel family (Mustelidae)—occur in North America: the American marten (*Martes americana*) and the Pacific marten (*M. caurina*). The range of *M. americana* historically spanned the boreal forests from Alaska, USA, to Nova Scotia, Canada, and included temperate and montane forests in Minnesota, Wisconsin, and Michigan, USA as well as portions of the US east coast as far south as West Virginia while *M. caurina* historically inhabited the coastal and montane forests of British Columbia, Canada, California, Oregon, and Washington, USA, with interior populations extending into Idaho, Montana, Wyoming, Utah, and New Mexico (Stone et al. 2002, Powell et al. 2012, Fig. 6.1, Colella et al. 2019, Fig. 1). These two species are extremely similar in life history and morphology and are known to hybridize in areas of sympatry (Colella et al. 2019). Indeed, due to these similarities, taxonomists previously considered the species to be conspecifics with *M. caurina* classified as a subspecies of *M. americana* (Wright 1953). Genetic analyses resulted in the elevation of *M.*

caurina to its current full-species status (Carr and Hicks 1997, Stone et al. 2002, Small et al. 2003, reviewed by Dawson and Cook 2012).

Both species of marten are considered forest specialists (Spencer et al. 1983, Thompson et al. 1989, 2012, Bull and Heater 2000, Bull et al. 2005, Slauson et al. 2007, Godbout and Ouellet 2008, Lavoie et al. 2019, Delheimer et al. 2023) and are generally associated with habitats that receive high amounts of snowfall (Raine 1983, Krohn et al. 1995, Carroll 2007, Krohn 2012, Zielinski et al. 2017, Martin et al. 2021, Pauli et al. 2022). Owing to the collectively wide distribution of these two species, researchers have found that martens inhabit a variety of forested ecosystems that differ in composition (i.e., deciduous, mixed woods, or coniferous) and seral stage (Chapin et al. 1997, Payer and Harrison 2003, Poole et al. 2004, Hearn et al. 2010, Delheimer et al. 2023, Viau et al. 2024). Regardless of forest composition or seral stage, however, researchers have found that martens require complex and heterogeneous habitats—characterized by closed canopies, dense understories often with ground cover in the form of coarse woody debris (i.e., fallen trees, tree limbs, and stumps), and trees that have developed accessible cavities—and that these habitat requirements are driven by their physiological requirements and trophic position (reviewed by Thompson et al. 2012). The long, slender body shape of martens reduces their ability to store fat and increases the rate at which body heat is lost to the environment, resulting in a high daily caloric need and increased reliance on landscape features to behaviorally thermoregulate (i.e., rest and den in sheltered locations) (Bull and Heater 2000, Slauson and Zielinski 2019, Martin et al. 2021, Delheimer et al. 2023). Furthermore, complex and heterogeneous habitats facilitate increased foraging efficiency (Andruskiw et al. 2008) and provide escape cover to evade larger-bodied predators. Multiple species have been documented as predators of martens, including: coyotes (*Canis latrans*), bobcats (*Lynx rufus*),

fishers (*Pekania pennanti*), and various raptors (e.g., *Bubo* spp. and *Accipiter* spp.) (Thompson 1994, Bull and Heater 2001, Weir et al. 2005, Mccann et al. 2010, Martin et al. 2022). Due to their affinity for complex habitats, Hearn et al. (2010), in their work with *M. americana*, described martens as “structural obligates.” Indeed, martens have been shown to avoid open habitats and to be sensitive to relatively small-scale land-use practices that result in simplified forests. For example, occupancy has been found to decline in landscapes composed of >25% regenerating habitat following timber harvest (Hargis et al. 1999, Potvin et al. 2000, Thompson et al. 2012).

The fisher, also a mesocarnivore in the weasel family, is associated with habitat requirements that mirror those of martens, as fishers are also considered structural obligates that rely on complex and heterogenous understories for foraging and predator evasion, and landscape features such as tree cavities for thermoregulation and protection (Raley et al. 2012). Due to high overlap in their habitat requirements and diet, martens and fishers are expected to exhibit pronounced interference competition, with the larger-bodied fisher being the dominant species capable of suppressing marten abundance (Donadio et al. 2006, Prugh and Sivy 2020, Pauli et al. 2022). Both species exhibit extreme sexual dimorphism, in regard to body size, with males being larger than females, and only exceptionally large male martens are similar in size to female fishers (male martens: 0.6 – 1.4 kg, female fishers: 2.0 – 2.5 kg, Powell et al. 2003). Marten and fisher competition and the factors that facilitate their co-occurrence—or lack thereof—have received much attention in the literature (Raine 1983, Krohn et al. 1995, Zielinski et al. 2017, Jensen and Humphries 2019, Manlick et al. 2020, Suffice et al. 2020, Evans and Mortelliti 2022, Pauli et al. 2022), with numerous studies having found inverse relationships between marten and fisher occurrence and abundance (Krohn et al. 1995, Krohn 2012, Zielinski et al. 2017, Jensen

and Humphries 2019, Suffice et al. 2020). In regions where these species' distributions overlap, zones of sympatry and parapatry have been found to be defined by gradients of precipitation, elevation, temperature, and prey abundance and availability (Zielinski et al. 2017, Jensen and Humphries 2019, Manlick et al. 2020). These studies suggest that martens are able to utilize high-elevation areas that receive large accumulations of snow, and support the hypothesis that deep snow offers martens a release from predation and competitive pressure, as larger-bodied carnivores avoid the high energy expenditure of moving through deep snow (Raine 1983, Krohn et al. 1995). By contrast, the marten's smaller and lighter body incurs a lower energetic cost while traversing on top of snow, and allows exclusive access to subnivean areas to forage on small mammals and rest in thermally insulated areas (Raine 1983, Krohn et al. 1995, Bull and Heater 2000, Pauli et al. 2013). However, Jensen and Humphries (2019) found that fishers were also able to persist in some places with deep snow, as long as they could access sufficient food resources to overcome the high energy expenditure required for movement in those areas. In zones of sympatry at lower elevations, fishers have been found to be more abundant than martens, and martens that inhabit these zones of sympatry are more likely to be juveniles—observations that reflect the expected outcomes of interference competition between these species (Krohn et al. 1995, Jensen and Humphries 2019).

At the turn of the twentieth century, *M. caurina* populations inhabiting the coastal forests of western North America were subjected to rapidly shifting anthropogenic pressures. The influx of European settlers, in the late 1800s and early 1900s, spurred the onset of commercial timber harvest in the highly valuable old-growth forests of the region (United States Department of Agriculture 1993). Martens (hereafter usage of “marten(s)” will refer to *M. caurina* unless otherwise specified), inhabiting these coastal regions, are highly reliant upon old-growth,

coniferous forests (Spencer et al. 1983, Bull and Heater 2000, Zielinski et al. 2001, 2005, Bull et al. 2005, Slauson et al. 2007, Slauson and Zielinski 2009, Thompson et al. 2012, Moriarty et al. 2016b:201, Delheimer et al. 2023) and thus experienced rapid and sustained losses of critical habitat. Meanwhile, predator control campaigns, funded by the US government in the early 1900s and aimed at eradicating wolves (*Canis lupus*) and mountain lions (*Puma concolor*), employed the extensive use of poisons, which likely also killed non-target species such as martens (Bailey 1907, Aubry and Lewis 2003, Robinson 2016, Schmidly 2016). Additionally, the lucrative fur market during this time (e.g., up to \$350 per fisher pelt) drove intense and unregulated trapping targeting martens and other furbearers (Lewis et al. 2012). The loss and fragmentation of old-growth forests, coupled with this high mortality from trapping and incidental poisoning, caused dramatic declines in coastal marten populations across western North America (Zielinski et al. 2001). Northern California's population of a regional subspecies of coastal martens, the Humboldt marten (*M. c. humboldtensis*), was assumed to have been extirpated for over 50 years before being rediscovered in 1996 (Zielinski et al. 2001). Currently, martens are widely considered extirpated from their historical ranges in southern coastal Washington and northern Oregon (Zielinski et al. 2001, Moriarty et al. 2016a, 2019).

Martens on Washington's Olympic Peninsula were historically documented from sea level to the subalpine tree line (Dalquest 1948, Scheffer 1995). Limited records detail the decline of the Olympic Peninsula's marten population, but the extirpation of martens from southern coastal Washington suggests that Washington's coastal martens faced the same pressures of habitat loss and over-harvest that contributed to the species' decline in Oregon and California (Dalquest 1948). McNulty (2018) describes extensive timber harvesting operations on the Peninsula in the early to mid-1900s. Scheffer (1995) notes that, in 1948, there was an estimated

population of 2,575 martens on the Olympic Peninsula, although it's unclear how this estimate was determined, or how accurate it was. Martens may have persisted in the Olympics because of their ability to use high-elevation forests, where they were relatively safe from anthropogenic pressures. In contrast, the closely related fisher, which exhibits similar habitat needs to the marten but may be excluded from high elevations in the winter due to deep snow (Raine 1983, Krohn et al. 1995, Zielinski et al. 2017), was extirpated from the Peninsula (Lewis and Stinson 1998).

Most of the Olympic Peninsula's high-elevation forests are contained within the contemporary boundaries of the Olympic National Park (ONP) and Olympic National Forest (ONF). Timber harvest has been prohibited on the ~405,000 ha of land comprising the ONP and designated Wilderness Areas in the ONF. Despite these habitat protections, the current marten population appears to be limited in extent and size (Moriarty et al. 2019), and from 1968–2016 there had been only nine verified observations of martens on the Peninsula (Moriarty et al. 2019). It is unclear if current conditions within the ONP and ONF continue to limit marten populations, or if past pressures continue to affect the population.

Since 1991, biologists have conducted numerous large-scale surveys targeting martens and other forest carnivores on the Olympic Peninsula (Jones and Raphael 1991, Sheets 1993, Happe et al. 2005, Jenkins et al. 2015, Howell et al. 2019, 2020, 2021, 2023, Moriarty et al. 2019). The cumulative results of these studies indicate that, although martens are still present within the boundaries of the ONP and ONF, the population has suffered an extreme reduction in range and may be at critically low numbers. From 1968–2022, researchers documented only 26 verified detections of martens on the Olympic Peninsula. These studies also indicate that the remaining marten population is very likely relegated to isolated sub-populations inhabiting high-

elevation forests. The average elevation of marten detections from 2008–2022 was 1,281 m ($n = 21$), with a range of 563–1,783 m. Researchers have found that martens are commonly detected where they occur in other areas of their range (Slauson et al. 2007, Zielinski et al. 2015) and were the most commonly detected carnivore in surveys conducted in Mount Rainier National Park and North Cascades National Park in Washington’s Cascade Range (Christophersen et al. 2005, Reid et al. 2010). In contrast, contemporaneous surveys conducted in ONP, using the same methods, failed to detect any martens (Happe et al. 2005), suggesting that martens on the Olympic Peninsula are very rare.

The Olympic Peninsula population has been in decline since at least the late 1960s (Moriarty et al. 2019), and likely since the turn of the 20th century (Zielinski et al. 2001). Martens on the Olympic Peninsula are isolated from neighboring marten populations owing to the geography of the Peninsula and anthropogenic land use making natural immigration highly unlikely (Moriarty et al. 2019). It is unknown if the population has been segregated into smaller, disjunct sub-populations due to a lack of connectivity within their current distribution. This protracted period of decline and isolation makes the loss of genetic diversity, through inbreeding and genetic drift, highly likely, and thus places the population at increased risk of extirpation. However, only a single genetic sample is available to evaluate the population’s current genetic health.

Schwartz et al. (2020) conducted phylogenetic and population-based analyses to describe the relatedness of marten populations in Washington, Oregon, and California, as well as the genetic diversity of those populations. Their study used six historical specimens—collected from 1886–1902—and one modern specimen that was collected in 2008 to represent the Olympic Peninsula’s population (Moriarty et al. 2019, Schwartz et al. 2020). Only the 2008 sample could

be amplified for microsatellite analyses. The study found that of all populations sampled ($n = 14$), the Olympic Peninsula population had the lowest mean allelic diversity as well as the lowest level of observed heterozygosity (Schwartz et al. 2020). These results suggest that the Olympic marten population is indeed at increased risk of extirpation because of low genetic diversity, but a sample size of one limits our ability to make inferences to the larger population. Clearly, more data are required to better evaluate the population's genetic health and to make informed decisions concerning its management.

In 2017, researchers began implementing novel survey protocols utilizing automated scent dispensers paired with camera traps (Long et al. 2024), and recorded marten at nine scent dispenser sites in two overwinter survey periods (Howell et al. 2019, 2020, 2021). Prior to the use of scent dispensers, survey methods required researchers to regularly return to survey sites to replenish bait and lure that would degrade over time, thus compromising effectiveness. The need to replenish bait or lure precluded high-elevation areas from being extensively surveyed in the winter due to the snowpack and rough terrain limiting access to those areas (e.g., Happe et al. 2005). Baits and lures could also be completely consumed by the first animals to encounter the survey station, resulting in reduced likelihood of attracting a target species. Scent dispensers address such problems with an internal reservoir of liquid scent lure and a pump programmed to release a small quantity of lure at programmed intervals (Long et al. 2024). This tool allows field crews to deploy survey equipment during the summer to conduct year-long surveys in areas that would be inaccessible to researchers during the winter months. The ability to survey overwinter is important, as martens may be more detectable in winter than summer, possibly because populations are generally larger and juveniles begin to disperse from natal ranges in the winter (Zielinski et al. 2015). This novel approach has proven effective at detecting martens (Long et al.

2024), but has yet to be deployed at scale across ONP and ONF as the basis for drawing large-scale inferences about drivers of marten distribution in the region. We conducted overwinter surveys using automated scent dispensers paired with camera traps, and a geographically extensive sampling frame, with the goal of both estimating occupancy and assessing marten distribution across high elevation forests of the Olympic Peninsula, and to explore factors that affect the population's distribution.

Study Area

All field work was conducted on public lands within the boundaries of ONP and ONF on Washington's Olympic Peninsula. Elevations on the Olympic Peninsula range from sea level to 2,427 m. The rapid transition from sea level to the tallest peaks of the Olympic Range results in a strong rain shadow and thus a marked precipitation gradient from west to east. The west side of the Peninsula receives 300–500 cm of precipitation per year while the eastern portion receives 40 cm of precipitation per year (Nielsen et al. 2021). Forests within ONP and ONF are predominantly coniferous, with broadleaf forests located in low elevation riparian zones (Nielsen et al. 2021). The alpine zone ranges from approximately 1,600 m upward, and tree cover is generally reduced to stunted krummholz in this zone (Nielsen et al. 2021).

Methods

Study Design

We defined high elevation forests as occurring at and above 1,066 m. While we aimed to concentrate survey sites at or above this elevation, we also sought to ensure that a wide gradient of elevation was surveyed. The extent of the study area was delineated, using ArcGIS Pro (ESRI, Redlands, California), by selecting all lands within the ONP and ONF boundaries that occurred

between elevations of 609 m and 1,982 m. The elevation band was capped, as our goal was to deploy survey sites at, or as close as possible to, each sample unit's centroid and wanted to prevent the inclusion of sample units that had centroids well above the tree line, where landscapes would be dominated by barren rock and ice at mountain peaks.

These elevation parameters resulted in a large, contiguous area of the targeted elevation, along with multiple small "islands." These non-contiguous "islands" were removed from the study area to avoid sample units primarily composed of non-target elevations. The resulting study area covered 374,932 ha.

To create a sample frame from which to select sample units, we projected a tessellation of 10 km² hexagons over the extent of the study area, from which we selected all hexagons that were intersected by a road or trail (a necessity given terrain and vegetation considerations), and whose centroid fell within the targeted elevation band of the study area. A total of 258 hexagons met those parameters, from which we removed one because it was deemed inaccessible, and another because the majority of its area was composed of barren rock and ice. The final sample frame consisted of 256 hexagons (Figure 1).

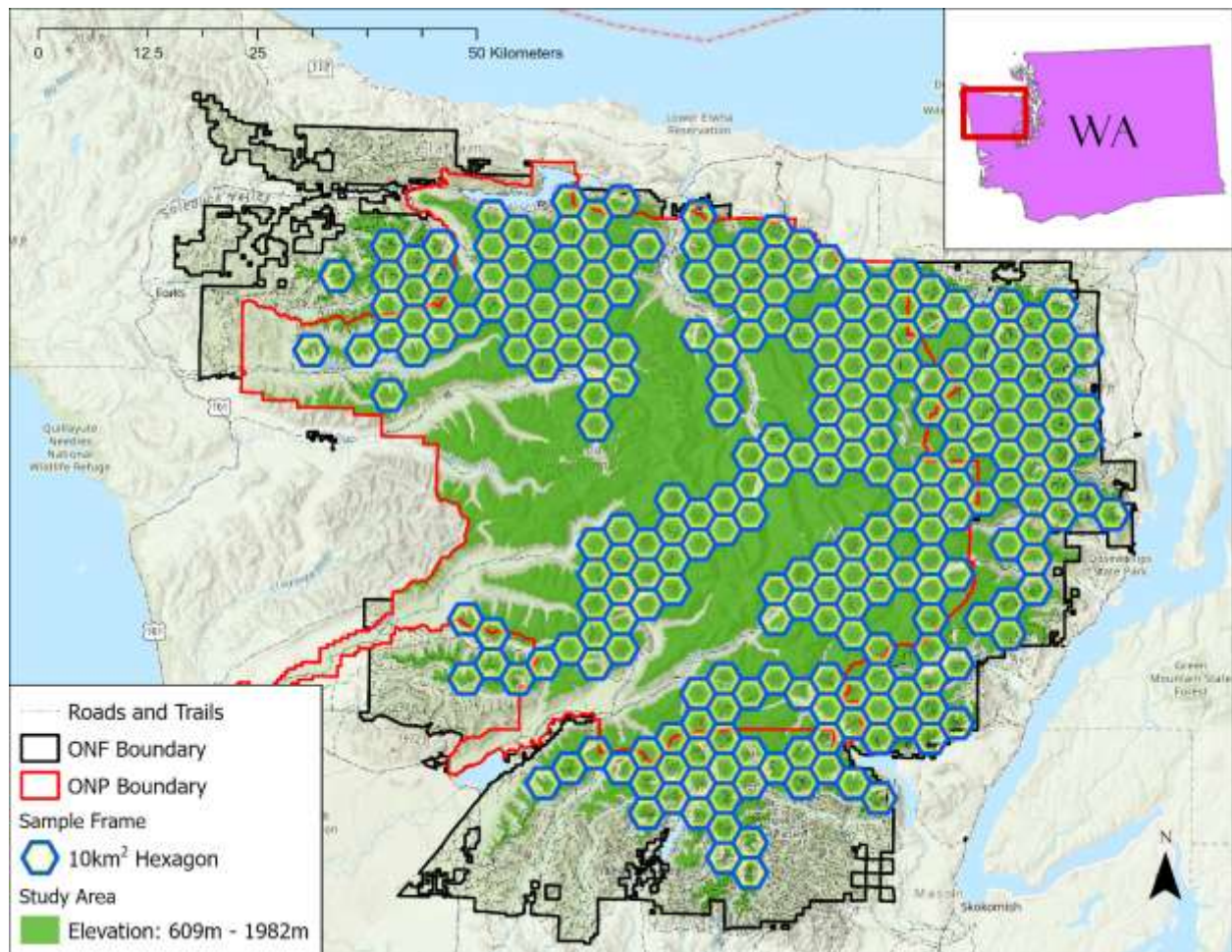


Figure 1. Study area delineated (in green) within the boundaries of the Olympic National Park and Olympic National Forest, Washington, USA, with sample frame consisting of 256, 10 km² hexagonal sample units. Each sample unit is intersected by a road or trail and has its centroid within the study area.

We used a random sampling design, stratified by both elevation and distance to a past marten detection, because the sample frame covered a wide range of elevations and historic marten detections that were not evenly distributed. Given that marten detections were rare in this system, we aimed to survey near past detections to 1) increase the odds of detecting a marten and, 2) determine if martens have persisted in areas where they were previously detected. To stratify based on elevation, we calculated the percentage of each sample unit's area comprising elevations $\geq 1,066$ m. We then assigned sample units to 1 of 4 elevation categories based on that percentage: 1 = $< 25\%$, 2 = $25\% - 50\%$, 3 = $50\% - 75\%$, 4 = $\geq 75\%$. To stratify based on distance

to past marten detections, we identified sample units where one or more marten detections occurred between 1968 and 2019 (i.e., all known marten detections at the time of study design) and categorized those units and all sample units forming three concentric rings around them as “close,” and all other sample units as “far”. These 2 stratification levels resulted in 8 stratification categories (4 categories of elevation by 2 categories of distance to a past detection).

Initially, we designed this study with the intent of sampling 140 sample units over the course of two years. We originally planned to deploy survey equipment in 70 sample units during the summer of 2022. A year later, those 70 sites were to be removed and reset in 70 new sample units, which were to be collected at the conclusion of the study in the summer of 2024. However, at the end of the 2022 field season, we realized that removing and resetting 70 sites would not be feasible given the funding and personnel that would be available to us in 2023. Thus, we did not re-deploy survey equipment in 2023.

To select sample units to be surveyed each season, we first determined how sample units were distributed across the stratification categories (Table 1). Rather than selecting an even number of sample units from each stratification category, we chose to prioritize surveying sample units representing high elevation and “close” categories to maximize our chances of detecting martens. With that motivation, we chose the proportion of each stratification category that would be randomly selected (Table 2). We randomly selected 140 sample units following the predetermined proportion from each stratification category. We then split the 140 sample units into 2 groups of 70 by randomly selecting 50% of the sample units in each of the stratification categories (Table 3). In cases where a stratification category featured an odd number of sample units, and thus could not be divided evenly, we used a coin toss to determine which survey year

would receive the extra sample unit. From the remaining sample units in the sampling frame, we randomly selected three from each stratification category to act as replacements if necessary.

Table 1. The distribution of the 256 sample units in the full sample frame across the eight stratification categories. Elevation categories indicate the percentage of a sample unit's area composed of elevations greater than 1,066 m: 1 = <25%, 2 = 25–50%, 3 = 50–75%, 4 = >75%. We split the distance to detection categories into “close” (1) and “far” (2). We categorized any sample units which contained a past marten detection and all of the sample units which formed three concentric rings around them as “close” and all other sample units as “far”.

Distance to Detection	Elevation			
	1	2	3	4
1	23	27	28	28
2	71	28	19	32

Table 2. We prioritized surveying in sample units at high elevation and close to locations of past marten detections by unevenly sampling from the available sample units in each stratification category to select 140 sample units.

Distance to Detection	Elevation			
	1	2	3	4
1	50% = 11	50% = 13	90% = 25	90% = 25
2	20% = 14	40% = 11	70% = 13	90% = 28

Table 3. The number of sample units to be selected from each stratification category for Season 1 and Season 2.

Distance to Detection	Season 1				Season 2			
	Elevation				Elevation			
	1	2	3	4	1	2	3	4
1	6	6	12	13	5	7	13	12
2	7	5	7	14	7	6	6	14

Survey Station Deployment

We intended to establish survey sites at locations where historical marten detections had occurred within a given sample unit, or at the centroid of the sample unit if there were no previous detections. In instances where we were unable to reach a historical detection location or the centroid, due to inaccessible topography or dangerous river crossings, or where the centroid lacked suitable trees for deployment (i.e., trees were too small), we deployed equipment as close to those locations as possible. We maintained a minimum distance of 1,900 m between survey sites in adjacent sample units. We deployed survey stations at a minimum distance of 100 m from a designated trail or road and 200 m from any designated campsites.

Each survey site consisted of a primary survey station (Figure 2) and a secondary survey station. We selected survey sites and installed equipment following protocols recently developed for overwinter deployment of scent dispensers with camera traps (Long et al. 2024). At the primary survey station location, we selected a pair of trees spaced approximately 4.8–6.7 m apart oriented to one another in a north-south direction. Trees selected for survey equipment installation ranged from approximately 45–110 cm in diameter at breast height. We installed all survey equipment on trees approximately 4.5 m above ground level to stay above the potential snowline.

On the northern tree, we installed an automated scent dispenser with a 12 cm section of a cow (*Bos taurus*) femur 12 cm below the dispenser, to act as a visual attractant as well as to help concentrate and disperse the scent lure. Approximately 15 cm below the bone, we installed a passive hair snare device. We constructed the hair snare devices using a 7.5 cm x 73.5 cm strip of corrugated plastic with four gun brushes attached and spaced 7.5 cm apart (Figura and Knox 2008, California Department Fish and Game, unpublished report). We pre-mixed scent lure

consisting of pure skunk “quill” (i.e., spray), beaver castor, fish oil, anise oil, water, and propylene glycol (a non-toxic, anti-freezing agent) following the same recipe that had been used by previous marten and fisher surveys in the Olympics and Cascades (R. Long, pers. comm.), and filled each scent dispenser with approximately 621 ml of liquid lure. We programmed the scent dispensers to release 3 ml of scent lure every 48 hours.

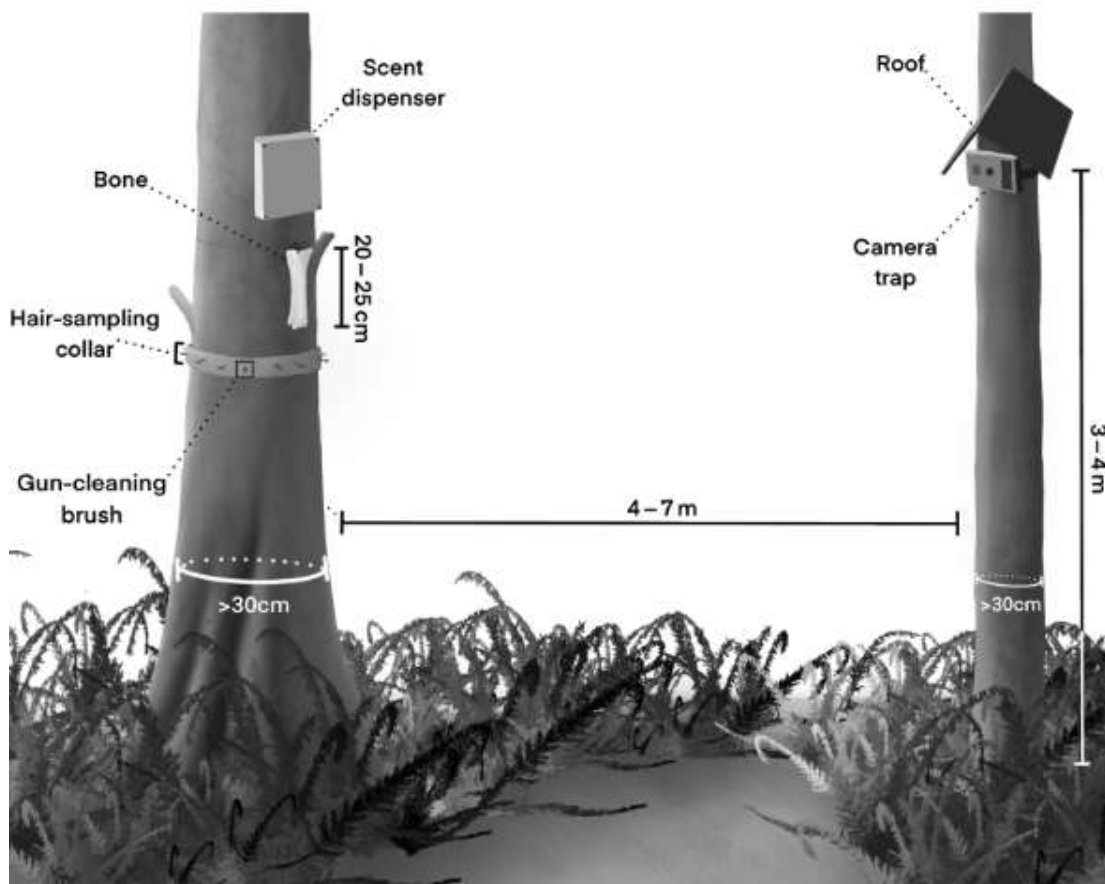


Figure 2. Illustration of the primary survey station. Trees bearing survey equipment were oriented in a north-south direction with the camera trap facing north to avoid sun glare. Illustration by Samantha Kreling (from Long et al. 2024).

On the southern tree, we installed a trail camera (Reconyx Hyperfire 2 [n = 45], Reconyx Hyperfire 2 Professional [n = 14], or Reconyx Ultrafire [n = 8] [Reconyx Inc. Holmen, WI]), oriented horizontally such that the detection zone included the length of the scent dispenser tree as well the area around the base of the tree. We adjusted the tilt and height of the

camera to orient its field of view to capture the bone at the top of the frame and the ground surrounding the base of the tree at the bottom. The tree's trunk was positioned in the middle to right third of the frame to align with the camera's detection zone. We conducted a crawl test to ensure that the camera could be triggered by a small, quadrupedal animal on each side of the tree trunk. We equipped camera traps with a 32 GB memory card and lithium batteries. Not all cameras were available for deployment at the beginning of the study and thus camera models were not randomly assigned to sample units. We standardized the camera settings as follows: photos during both day and night, high sensitivity, five photos taken per trigger event and the minimal delay possible between trigger events (i.e., 0 or 1 second depending on camera model). We also programmed the camera to take a timelapse photo every 24 hours to monitor the camera's functionality. We standardized survey sites to the greatest extent possible by cutting branches and trimming low shrubs and vegetation that could obscure a marten and/or trigger the camera in windy conditions. We selected sites that minimized the amount of trimming necessary.

After deploying the primary survey station, we then searched in a pre-determined, random direction for a natural game trail located 100 m to 200 m from the primary station. In cases where the random direction was inaccessible due to terrain features, or a game trail could not be located within the distance range, we searched as close to that direction as possible. We selected the most heavily used game trail within the distance and direction parameters. We did not use any lures or baits at the secondary survey station to allow the comparison of detection rates of carnivores at baited and unbaited sites. We installed a trail camera (Moultrie M-880 [n = 22], Moultrie M-880 Gen2 [n = 11] [PRADCO Outdoor Brands, Calera, AL], or Bushnell Trophy Cam HD [n = 28] [Bushnell Outdoor Products, Overland Park, KS]), on the south side of the game trail, approximately 4.8–6.7 m from the trail. The camera model was randomly assigned to

each secondary survey station. We standardized the secondary cameras' settings as follows: photos during both day and night, 8 MP photo quality, high sensitivity, three photos taken per trigger event, and a five second delay between trigger events. We programmed the Moultrie M-880 Gen2 and Bushnell cameras to take a timelapse photo once per day to monitor camera functionality (the Moultrie M-880 did not offer this option). Again, we standardized survey sites to the greatest extent possible by trimming branches and low vegetation that could obscure animal movement and/or trigger the camera in windy conditions and we selected sites that minimized the amount of trimming necessary.

In instances where we were unable to locate a game trail in suitable habitat (i.e., trees large enough to support a climber deploying a camera trap) that also met distance parameters from the primary survey station, trails, or campsites we did not deploy a secondary survey station. In the latter half of the season, we deployed the camera trap that was intended for a secondary station as a supplemental camera at the primary survey station. We oriented supplemental cameras in the traditional "landscape" view rather than rotating them 90 degrees to match the primary cameras.

As we removed survey stations from the field at the conclusion of the study, we also recorded the distance between the scent dispenser tree and the primary camera tree, the condition of the trail cameras (i.e., operating or non-operational), the number of photos taken, the condition of the scent dispenser (i.e., operating or non-operational), if the scent dispenser lure bag had leaked, and how much scent lure remained in the bag.

Genetic Sample Collection and Analyses

We began collections of genetic samples during the survey station removal phase of the study. While hiking to survey stations we opportunistically collected any scats that morphologically resembled mustelid scats. Once at survey sites, we searched a 15 m radius around the primary survey station for approximately 10 minutes to collect any scats that could have been deposited by a mustelid. We placed each scat sample into a labeled, paper coin envelope and sealed it in a plastic bag with a desiccant packet. We also checked each of the gun brushes on the hair snare for hairs, and deposited brushes that held hair into individually labeled, paper coin envelopes—sealing these in a plastic bag with a desiccant packet. Scat and hair samples were stored in a freezer upon return from the field.

We examined the camera photos from each station, either in the field with an iPhone and an SD card reader or at the office on a laptop, to determine if a marten had been detected—and thus if we should deploy a transect of hair snare stations at the site. We returned to sample units, in which martens were detected, to deploy a hair snare transect as time availability, personnel logistics, and trail access allowed.

Each hair snare transect consisted of five stations. We attempted to keep the transect as linear as possible, but topography prevented us from accessing all areas and we specifically avoided open habitat types to set stations in forested habitats that we thought martens were more likely to use. In cases where the topography was inaccessible and precluded our ability to deploy all five stations in a single line, we deployed the stations in two parallel lines. We attempted to center the transect on the most recent marten detection in the sample unit and maintained a minimum distance of approximately 100 m between stations.

Each station consisted of two hair snare devices—a tree cubby and a ground cubby—and a camera trap (Reconyx Hyperfire 2). We constructed each tree cubby from a 16 cm x 27 cm sheet of corrugated plastic by cutting the top layer of the sheet from the mid-point of the longer edge to the midpoint of its opposite edge and then making a cut on the back layer of the sheet from the mid-point of the longer edge to each of the opposite corners. Cutting through only a single layer of the sheet allowed it to be folded at those cuts, resulting in a triangular cubby with two “wings” (Figure 3). We secured two gun brushes to the interior of the cubby to passively collect hair from martens entering the cubby (Kendall and McKelvy 2008). We constructed each ground cubby from the same corrugated plastic sheeting material, which we cut to fold into a triangular tube that was 20 cm wide on all sides and 80 cm long (modified from Schwartz et al. 2006; Figure 4). We inserted a 2.54 cm x 0.95 cm x 20 cm wooden slat through the walls of the cubby, perpendicular to its long axis, approximately 3 cm from each end and 6 cm above the floor of the cubby. We then stapled a 2.54 cm x 15 cm strip of a non-poisonous glue-impregnated sheet (Catchmaster 72MB, Atlantic Paste and Glue Co., Brooklyn, NY) to the bottom of the wooden slat to catch and hold hairs from martens entering and exiting the cubby (Zielinski et al. 2006). We inserted two wooden slats, above the slat bearing the glue strip, to force martens to go under the glue strip to enter or exit the cubby. We also secured two gun brushes on the interior walls of the cubby, set back 6 cm from the wooden slats, as a secondary method of collecting hair samples. Both collection methods (glue strips and gun brushes) have been shown to be effective in collecting hair samples from martens (Foran et al. 1997, Zielinski et al. 2006, Kendall and McKelvy 2008, Williams et al. 2009). We used the methods concurrently to maximize the probability of collecting a hair sample if a marten visited the cubby.

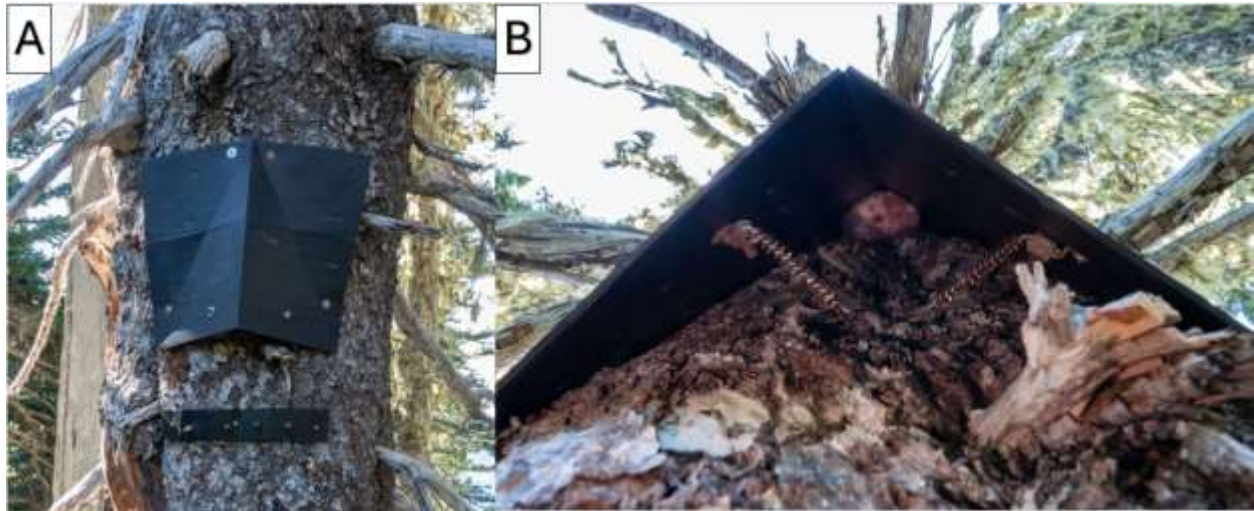


Figure 3. A) Tree cubby deployed on the bole of a tree. The strip of corrugated plastic below the tree cubby included additional gun brushes to trap hair. B) View looking up the bole of the tree into the tree cubby from below. We secured a piece of chicken to the tree at the top of the cubby, and gun brushes to each side of the cubby's opening. Photos by B. Wymer.



Figure 4. We used a ground cubby at each hair snare transect station. Each side of the cubby measured 20 cm with a length of 80cm. The cubby was held together with a bolt and nut at each end. A) We tied a chicken leg to the center of the cubby, and secured gun brushes and a slat with a glue strip in place to trap hair. The glue strip was held in place with strips of duct tape here, but we later found stapling to be the best method. B) We inserted wooden slats at the end of the cubby to force a marten to go under the glue strip to enter and exit the cubby. Photos by B. Wymer.

At each station, we located a 60–110 cm diameter tree with enough flat ground at its base to accommodate the ground cubby. We secured the tree cubby to the tree bole with nails, approximately 1.75 m above the ground, and sized the opening to leave approximately 15 cm of space from the bole of the tree to the peak of the cubby. We nailed a 7.5 cm x 73.5 cm strip of corrugated plastic, with four gun brushes spaced 7.5 cm apart, approximately 15 cm below the opening of the tree cubby. We deployed the ground cubby at the base of the tree and pinned it in place with rocks or logs to prevent animals from removing it. We secured a single piece of raw chicken inside each of the cubbies either with a fencing staple (tree cubby) or by tying the chicken inside the cubby with cotton string (ground cubby). Finally, we smeared Gusto (Minnesota Trapline Products, Pennock, MN) scent lure on the bole of the tree. We positioned the trail camera 5–6 m from the cubbies and oriented it so that both cubbies were visible in the photo frame (Figure 5).



Figure 5. A full hair snare transect station. The tree cubby is secured to a tree bole with nails and the ground cubby is pinned in place with branches and stones. Each cubby is baited with a piece of raw chicken and Gusto scent lure is smeared on the tree below the opening of the tree cubby.

We revisited the transects after 7–14 days, as logistics allowed, to collect any hair samples and rebait the cubbies. We concluded the survey after another 7–14 days, at which point we removed the survey equipment from the field. We processed and stored hair samples collected via gun brushes using the methods described above. We wrapped glue strips bearing hair samples in wax paper and sealed them in a plastic bag with a label and a desiccant packet then stored them in the freezer. Photos from the accompanying trail cameras were used to help determine what species may have deposited hair samples but were not included in any photo analyses.

We sent all hair samples which consisted of more than a single hair and all scat samples to the US Forest Service National Genomics Center for Wildlife and Fish Conservation and analyses are on-going. Samples that are determined to be from martens or fishers will be genotyped using methods used in previous genetics studies on the Olympic Peninsula (Happe et al. 2020 for fishers, Schwartz et al. 2020 for martens). Using the same genetic markers as previous studies will allow us to pool marten samples from this study with the single recent sample analyzed by Schwartz et al. (2020) to compare to other marten populations from the U.S. west coast and better assess the health of the population. Similarly, analyzing the same genetic markers as previous fisher studies will allow us to determine the relatedness of fishers to known, reintroduced individuals.

Photo Analysis

We uploaded all photo data collected during the occupancy study to Wildlife Insights (Ahumada et al. 2020) to process and code photos with the ID of detected species. We used a sequence-based photo analysis method that grouped photos into sequences, with a new sequence generated if 60 seconds or more elapsed between two subsequent photos. In other words, a single sequence could be composed of any number of photos spanning any amount of time, as long as there were never more than 60 seconds between any two subsequent photos. We tagged all species, and the number of individuals of each species, detected in each sequence. After all sequences were coded based on species and number of individuals, we downloaded the coded data from Wildlife Insights for further analysis.

Results

From 07/01/2022–10/15/2022, we deployed survey stations in 67 of 70 sample units (Figure 6). We were unable to install survey equipment in all 70 sample units given time constraints. We deployed survey equipment in three “replacement” sample units because the original sample units were either inaccessible due to steep terrain ($n = 2$) or because we were unable to meet permitting requirements to deploy equipment. In six instances, we were unable to locate a suitable area to deploy a secondary survey station due to proximity to roads or trails, inaccessible topography, or unsuitable vegetation. In these cases, the secondary camera was either installed to monitor the primary survey station as a supplemental camera ($n = 3$) or was not deployed at all ($n = 3$). We removed survey stations from a single sample unit on 05/19/2023 and from the remaining sample units between 06/12/2023 and 09/12/2023. We collected 529,174 images in total from all primary and secondary survey stations.

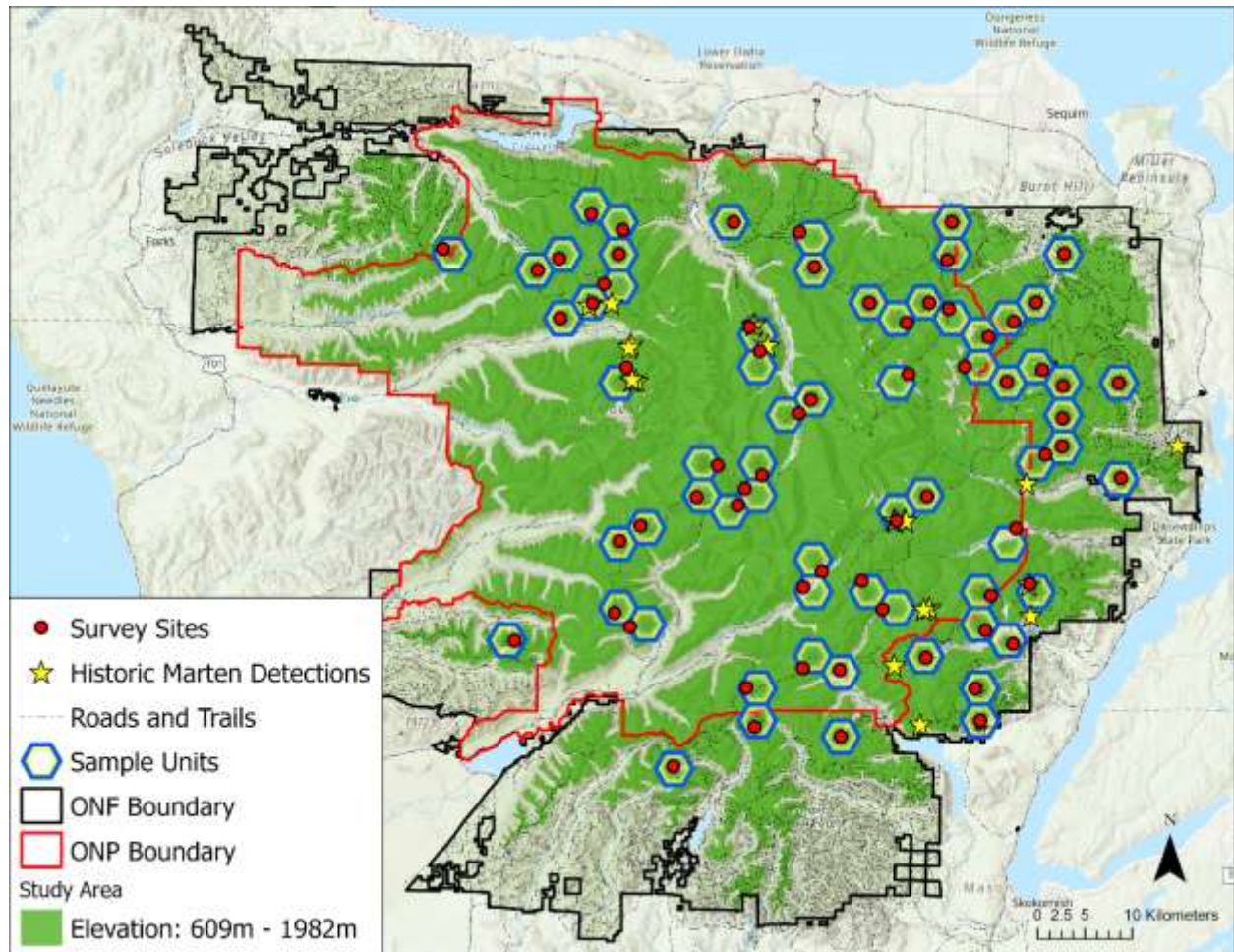


Figure 6. Locations of deployed survey sites within randomly selected sample units ($n = 67$) and the locations of marten detections from 1968–2022.

Sixty-two of 67 cameras (92.5%) at primary stations operated properly, as evidenced by a daily time lapse photo, through at least 05/01/2023. After reviewing photo data, we found that one primary camera did not have functional infrared lights, which prevented it from capturing night images. Thus, we determined that cameras operated without issue in 62 sample units. We found that seven cameras had inadvertently been set to take a timelapse photo every 15 minutes rather than once per day, but all cameras were deployed with consistent motion trigger settings. Of the 7 cameras set to take a timelapse photo every 15 minutes, only 2 depleted their batteries before 05/01/2023. Two additional primary cameras captured high numbers of false trigger events, resulting in battery depletion. The number of photos taken by primary cameras over the

survey period ranged from 242–54,290. After removing nonfunctional cameras and those with >10,000 false triggers, the remaining cameras (n = 56) captured an average of 595 photos (range: 331–1,918 photos) per deployment.

We deployed scent dispensers with approximately 621 ml of lure and set them to dispense 3 ml every 48 hours. Primary sites were deployed an average of 332 days (range: 281–390 days); thus, we expected there to be lure remaining in each dispenser (average = 123 ml, range: 36–201 ml) at the conclusion of the study. Across all sample units, nine scent dispensers malfunctioned at an unknown point in the survey and had 50–100% of their lure remaining at the end of the survey. Three scent dispensers had approximately 100% of their lure remaining upon collection, suggesting they became nonoperational shortly after being deployed. In the 62 sample units with fully operational cameras, 53 scent dispensers had 0–25% of the lure remaining. We found that 15 of those scent dispensers had developed an internal leak and that 15–120 ml of liquid had pooled in the lower pump chambers of those units. These internal leaks were attributed to incompatible lure reservoir bags and caps, resulting in a slow drip, and have since been addressed by the manufacturer (R. Long, pers. comm.). Of the 53 scent dispensers with 0–25% of lure remaining, 6 were nonoperational when revisited, with 3 of those having internal leaks. It is impossible to know at what point scent dispensers malfunctioned during the survey, or if internal leaks completely depleted lure and, if so, how quickly otherwise properly operating scent dispensers ran out of lure. In total, 47 primary survey stations had both cameras that were fully operational through at least 05/01/2023 and scent dispensers that were still operational when we revisited, with $\leq 25\%$ of their lure remaining. The other 20 primary survey stations still collected camera data for variable amounts of time, but with ambiguity as to the duration the scent dispenser functioned.

The cameras at secondary survey stations—and those paired at primary stations as supplemental cameras—experienced higher rates of failure, with 42 of 64 (65.6%) total cameras remaining operational through 05/01/2023. Secondary and supplemental cameras, on average, took 2,431 photos (range: 66–9831 photos). These non-Reconyx cameras were much older than the primary survey station cameras, were more prone to false triggers, and at times failed to take daily time lapse photos as programmed.

Martens

We detected martens in only five sample units (7.46%, Table 4, Figure 7). We are unable to conduct the planned occupancy modeling with so few detections, and instead will report descriptive statistics here. Four of the five sample units in which we detected martens overlapped the locations of past survey stations, however only one of them also contained a historic marten detection. We recorded seven independent marten detections in 5 sample units that occurred at both primary and secondary survey stations. In 4 of the 5 sample units, martens were detected only once. The average latency to the first marten detection at primary and secondary stations was 169 days ($n = 4$, range: 77–241 days) and 154 days ($n = 2$, range: 100–207 days), respectively. All marten detections occurred between November 2022 and April 2023 and while snow was present in the sample unit (Figure 8). Marten detections occurred at survey stations with an average elevation of 1,240 m (range: 903–1,523 m).

Table 4. Number (and percentage) of sample units, primary survey stations, and secondary survey stations in which each species was detected.

* includes detections at any station deployed in sample unit.

** includes detections from supplemental camera paired at scent dispenser.

	Sample Units (n=67)	Primary Stations (n=67)	Secondary Stations (n=61)
Species	# Units in Which Detected*	# Primary Stations at Which Detected **	# Secondary Stations at Which Detected
Pacific Marten (<i>Martes caurina</i>)	5 (7.46%)	4 (5.97%)	2 (3.28%)
Fisher (<i>Pekania pennanti</i>)	17 (25.37%)	16 (23.88%)	1 (1.64%)
Bobcat (<i>Lynx rufus</i>)	34 (50.75%)	25 (37.31%)	15 (24.59%)
Coyote (<i>Canis latrans</i>)	15 (22.39%)	13 (19.40%)	3 (4.92%)
Mountain Lion (<i>Puma concolor</i>)	22 (32.84%)	22 (32.84%)	5 (8.20%)
American Black Bear (<i>Ursus americanus</i>)	58 (86.57%)	48 (71.64%)	36 (59.02%)
Roosevelt Elk (<i>Cervus elaphus roosevelti</i>)	20 (29.85%)	16 (23.88%)	15 (24.59%)
Blacktail Deer (<i>Odocoileus hemionus columbianus</i>)	54 (80.60%)	42 (62.69%)	45 (73.77%)
Snowshoe Hare (<i>Lepus americanus</i>)	21 (31.34%)	7 (10.45%)	17 (27.87%)
Douglas's Squirrel (<i>Tamiasciurus douglasii</i>)	35 (52.24%)	26 (38.81%)	17 (27.87%)
Northern Flying Squirrel (<i>Glaucomys sabrinus</i>)	23 (34.33%)	19 (28.36%)	6 (9.84%)
Sciuridae Family	4 (5.97%)	3 (4.48%)	1 (1.64%)
<i>Tamias spp.</i>	5 (7.46%)	3 (4.48%)	3 (4.92%)
<i>Muridae</i> Family	12 (17.91%)	10 (14.93%)	3 (4.92%)
Class Aves	28 (41.79%)	20 (29.85%)	11 (18.03%)

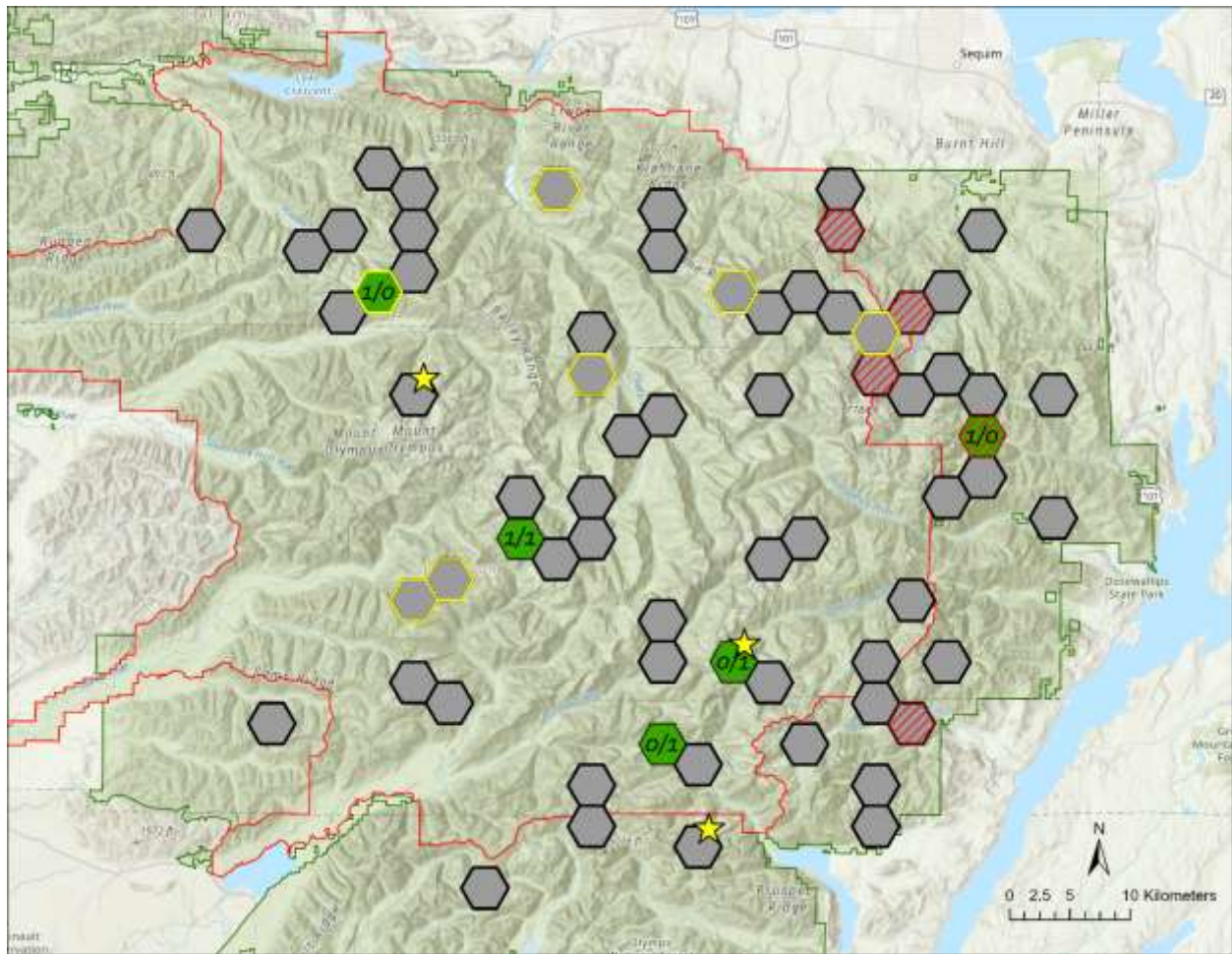


Figure 7. Martens were detected in 5 of 67 sample units (7.46%, green hexes). The 1s and 0s within those sample units indicate whether detections occurred at the primary survey station and/or the secondary survey station, with the value on the left of the “/” indicating detection or non-detection at the primary station and the value on the right indicating detection or non-detection at the secondary station. Sample units labelled with a star indicate sites where a supplemental camera was paired at the scent dispenser and thus the value on the right of the “/” represents the supplemental camera. Red hash marks within sample units indicate primary cameras that malfunctioned or stopped operating prior to 05/01/2023. Sample units highlighted in yellow indicate scent dispensers that still contained 50–100% of their lure volume when we revisited the site.

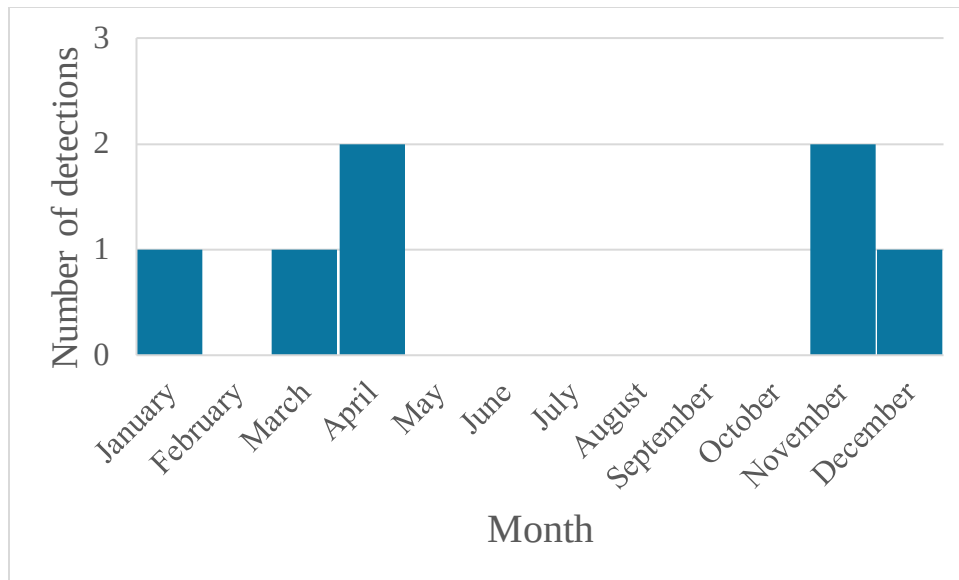


Figure 8. Number of independent marten detections per month from 07/01/2022–09/12/2023. Only one survey station detected marten on multiple occasions and those detections were separated by more than one month.

High Divide

A single marten was detected on 04/03/2023 (Figure 9). Only a single image of the marten was captured as it leapt from the scent dispenser tree and landed outside of the camera's field of view. The scent dispenser had malfunctioned at this station and there was approximately 90–100% of the lure still remaining when we revisited the site indicating the malfunction likely occurred soon after it was deployed on 08/05/2022. This detection occurred at an elevation of 1,523 m and was approximately 458 m from where a marten was detected in 2019 (Howell et al. 2020).



Figure 9. A marten leaping from the scent dispenser tree in the High Divide sample unit.

Low Divide

We detected martens on two occasions at the primary survey station, on 11/11/2022 and 01/21/2023, and once at the secondary station, on 03/21/2023. This was the only sample unit in which we detected martens on more than one occasion. This sample unit did not overlap any past survey stations and thus these marten detections are the first in the area, occurring approximately 3.5 km from the nearest past survey station. The elevation of the primary and secondary stations were 1,328 m and 1,324 m, respectively.

Six Ridge

We detected a marten at the secondary survey station on 12/21/2022 at an elevation of 1,170 m. We did not detect martens at the primary survey station. The secondary camera that detected the marten became nonoperational on 12/22/2022, the day after the marten was detected.

Marmot Lake

We paired a supplemental camera at the primary survey station rather than deploying a secondary station due to a lack of suitable habitat within the distance parameters set for secondary stations. A marten was detected on 04/14/2023, at an elevation of 1,144 m (Figure 10). The marten was detected by the supplemental camera but not the primary camera. The primary camera at this location was set to take a time lapse photo every 15 minutes (but was programmed with the other standardized motion detection settings) and took a photo approximately seven minutes after the marten was in the area and its tracks were clearly visible in the snow. The marten approached the scent dispenser tree and thus should have been within the primary camera's detection zone, but still failed to trigger the camera.



Figure 10. A marten detected by the supplemental camera in the Marmot Lake sample unit.

Tunnel Creek

We detected a marten at the primary station on 11/30/2022 at an elevation of 903 m. This was the only marten detection on ONF lands in this study, and represents only the second detection ever by a camera study on ONF lands (Howell et al. 2023). The primary camera was set to take a time lapse photo every 15 minutes (again with otherwise standardized motion detection settings) and the marten was photographed by one of the time lapse photos rather than by triggering the camera's motion sensor. This camera became nonoperational on 03/21/2023.

Mt. Olympus

Surveys conducted in 2015, 2016, 2017-2018 (over-winter), and 2018-2019 (over-winter) were successful in detecting martens in the upper Hoh River Valley and near Mt. Olympus, specifically east of the Glacier Meadows camp site (Moriarty et al. 2019, Howell et al. 2020). Due to the recent history of marten detections in this area, we were surprised to have none in the Mt. Olympus sample unit during our surveys. We paired a supplemental camera at the primary survey station. Both cameras and the scent dispenser were operational for the duration of the survey. We were unable to place the primary survey station within the cluster of past marten detections east of the Glacier Meadows camp site due to a large trail washout that crew members were uncomfortable crossing with large, cumbersome packs. The primary station was deployed approximately 1.4 km from the nearest, past marten detection in the sample unit.

Fishers

We detected fishers in 17 sample units (25.37%, Table 4, Figure 11). All but one fisher detection occurred at primary survey stations. In one sample unit, a fisher was detected at the secondary survey station only. The average latency to the first fisher detection at primary survey stations was 110 days (range: 12–281 days). The fisher detection at the secondary survey station occurred nine days after deployment. We detected fishers in two sample units in which we also detected martens: Low Divide and Six Ridge. Fisher detections occurred in every month of the year except June (Figure 12), and 53% of the survey stations that detected fishers did so on more than one occasion. The lack of detections in June, and only one detection in both July and August, may have resulted from a lower number of survey stations deployed in those months, as deployments in 2022 occurred from July–October, and retrievals in 2023 occurred from June–September.

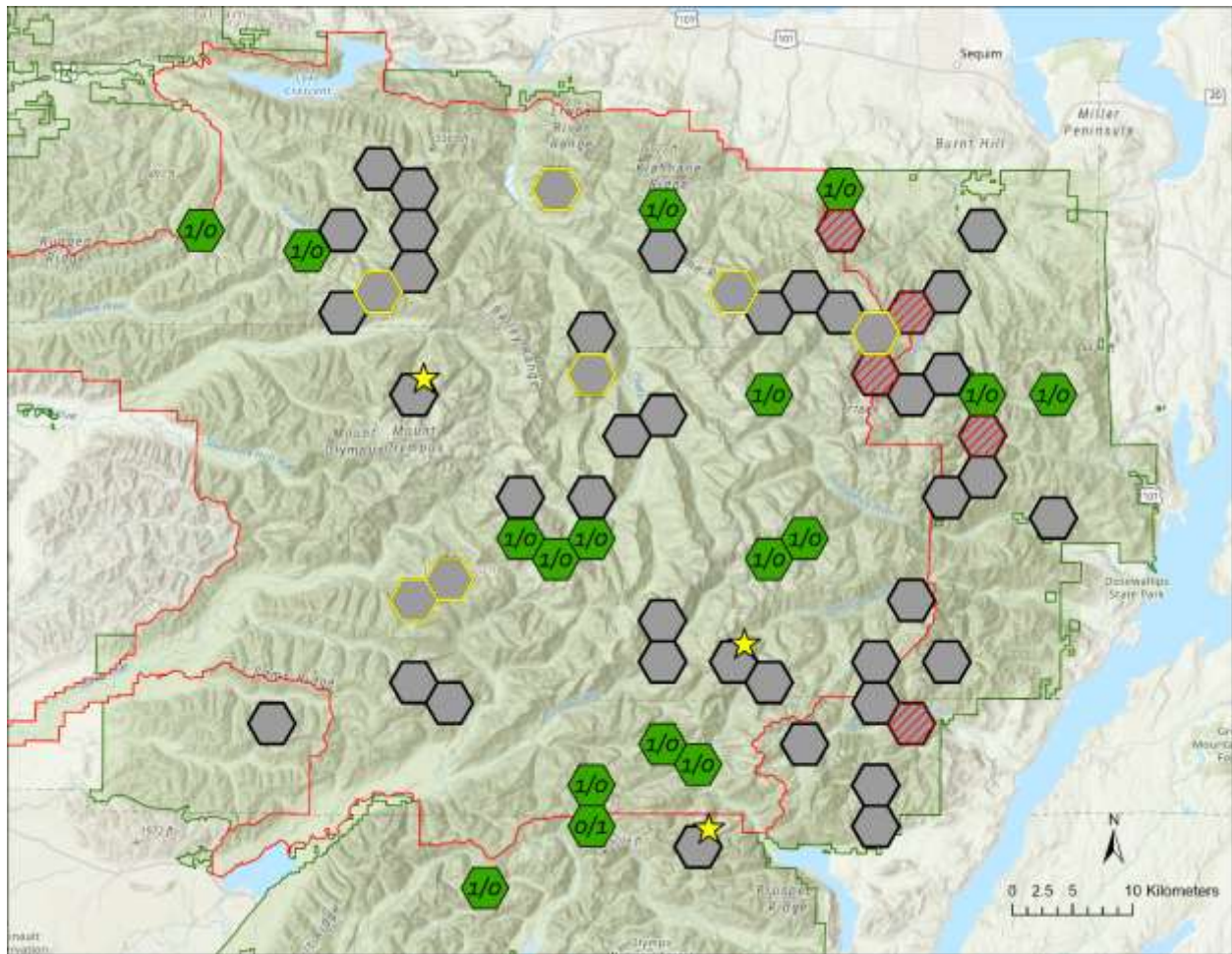


Figure 11. Fishers were detected at 17 of 67 sample units (25.37%, green hexes). The 1s and 0s within those sample units indicate whether detections occurred at the primary survey station and/or the secondary survey station, with the value on the left of the “/” indicating detection or non-detection at the primary station, and the value on the right indicating detection or non-detection at the secondary station. Sample units labelled with a star indicate sites where a supplemental camera was paired at the scent dispenser and thus the value on the right of the “/” represents the supplemental camera. Red hash marks within sample units indicate primary cameras that malfunctioned or stopped operating prior to 05/01/2023. Sample units highlighted in yellow indicate scent dispensers that still contained 50–100% of their lure volume when we revisited the site.

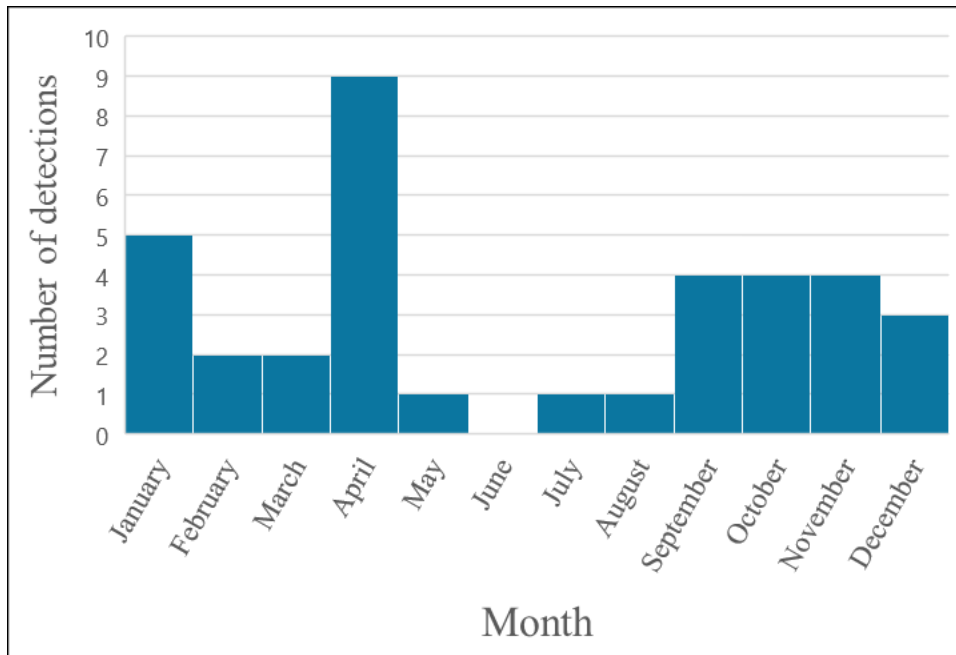


Figure 12. Number of independent fisher detections per month from July 2022 to September 2023. Detection events were considered independent if there was at least 12 hours between subsequent detections at the same survey station.

Fishers were detected at an average elevation of 1,046 m (range: 675–1,576 m). The subsets of fisher detections that occurred during the snow-on period (November–April) and snow-off period (May–October), had average elevations of 1,004 m (range: 675–1,328 m) and 1,076 m (range: 756–1,576 m), respectively, indicating a high overlap in elevational ranges throughout the year. However, the highest elevation fisher detection occurred during the snow-off period (July) at 1,576 m.

A fisher occupancy study conducted after their reintroduction in 2008 found that occupancy was associated with the boundary separating designated Wilderness areas from managed, regenerating forest types (Happe et al. 2020). Patterns of fisher detections from this study were similar to those identified in the previous occupancy study, with detections occurring on the north, northeastern, and southern boundaries of the ONP. However, this study also detected fishers in the interior of the ONP, at Low Divide, the upper section of the Elwha River,

and in two sample units along the West Fork of the Dosewallips River, where they had not previously been detected. The previous occupancy study (Happe et al. 2020) did deploy survey stations in the interior portions of the ONP, however our fisher detections at Low Divide and the upper section of the Elwha River occurred approximately 3.5 km and 1.2 km, respectively, from the nearest survey station used in their study and thus occurred outside of the area they sampled. In contrast, our fisher detections along the West Fork of the Dosewallips River occurred approximately 880 m and 380 m from the nearest survey stations used by the previous occupancy study.

Genetic Sample Collection

We deployed transects of hair-snare stations in three sample units. We selected High Divide because we detected martens there as had a previous study conducted in 2019 (Howell et al. 2020). We selected Mt. Olympus because martens had been detected there by multiple past surveys (Moriarty et al. 2019, Howell et al. 2020). Finally, we selected Harrison Lakes because a marten was detected there during a camera survey the year prior (Howell et al. 2023).

We collected 41 hair samples in total, with the majority coming from hair snare devices paired with scent dispensers. At a given site, each gun brush or glue strip that held hair was treated as a unique hair sample. Four hair samples were collected from the Low Divide site, where a marten was detected, and 16 hair samples were collected at sites where fishers were detected. Additionally, four samples were collected from the Skyline Trail, where putative mustelid tracks were identified in the snow during photo analysis, but positive species identification was not possible.

We did not detect any martens or fishers on-camera at any hair snare transect stations. However, we collected one hair sample via a glue strip (ground cubby) and one hair sample via a gun brush (tree cubby) in the High Divide and Mt. Olympus transects. We found that bears frequently destroyed ground cubbies by attempting to access the bait. However, no tree cubbies were disturbed, despite also being baited with chicken. All losses of ground cubbies to bears occurred after cubbies had been rebaited (i.e., 7–14 days post deployment). No ground cubbies were disturbed in the High Divide transect, but a single ground cubby at Harrison Lakes and all five ground cubbies at Mt. Olympus were disturbed by bears.

We also collected 24 scats via opportunistic detections while hiking to and from survey sites ($n = 20$), or while conducting scat searches at primary survey stations ($n = 4$). In April 2024, we secured funding to analyze hair and scat samples and these samples have been sent to the US Forest Service National Genomics Center for Wildlife and Fish Conservation, in Missoula, Montana, and analyses are on-going.

Discussion

We acquired sparse marten detections during this study, consistent with the results of past surveys in the Olympic Range. The exception to this general trend remains the 2018–2019 survey that detected martens at 11/31 sites (Howell et al. 2020). However, the survey sites in that study were selected to maximize the probability of detecting martens based on the recent history of detections and survey stations were deployed in clusters (i.e., 100–250 m between paired stations and 800 m–2.5 km between adjacent pairs in the cluster) in five areas across ONP (P. Happe pers. comm.). In contrast, we conducted a geographically extensive survey using a stratified random design. We included a wide band of elevation in our study area rather than focusing completely on high-elevation forests, even though past surveys have suggested martens

may be restricted to high elevation forests (Moriarty et al. 2019, Howell et al. 2020). We chose to cover a wider range of elevations because we were using novel automated scent dispensers that have proven effective in detecting martens (Long et al. 2024), and we wanted to make sure we weren't missing possible low-elevation remnants of the population. More specifically, we deployed 50.7% (34/67) of primary survey stations below 1000 m in elevation, while the 2018–2019 survey featured only 12.9% (4/31) of its deployments below 1000 m. Our intended study area ranged in elevation from 609–1,982 m, but on four occasions, we were unable to reach the target elevation due to steep, inaccessible terrain; stations in those sample units were deployed below 600 m (Appendix A). Although we did detect a single marten at 903 m in the Tunnel Creek area, the average elevation of survey stations where we detected martens was 1,240 m which is consistent with the 2008–2022 average marten detection elevation of 1,281 m. We also recorded a similar latency to the first marten detection at primary stations as a study conducted in Washington's Cascades Range where martens were the most commonly detected forest carnivore, 169 days and 134 days respectively (Long et al. 2024).

Our marten detections at Tunnel Creek and Marmot Lake demonstrated the imperfect detection process inherent to camera surveys. The Tunnel Creek detection was captured solely due to an error in programming the camera, which was set to take a timelapse photo every 15 minutes rather than once per day. The marten was not captured by the standardized motion trigger, but rather by happenstance in a timelapse photo. At Marmot Lake, we detected the marten via the supplemental camera and not the primary camera. The primary camera at Marmot Lake was also set to take a time lapse photo every 15 minutes and captured the marten's tracks in the snow seven minutes after it had passed through the area. Importantly, we confirmed that the motion settings were correct on all cameras as other species were detected via motion triggered

images. Interestingly, we also observed putative mustelid tracks in snow in images from a survey station along the Skyline Trail, as well as very close-up images of an animal investigating the camera. Unfortunately, the species could not be identified from these images.

Given the imperfect detection process and the rarity of martens on the Olympic Peninsula, we suggest that future surveys deploy two cameras at all scent dispenser stations to increase the detection probability for martens visiting the area. Sasse et al. (2023) showed that deploying two cameras at each survey station increased the detection rate of Eastern spotted skunks (*Spilogale putorius*)—another rare, small bodied predator. A second camera at each survey station could also act as a back-up in cases where one camera malfunctions. Future studies using two cameras at scent dispenser stations could compare camera orientations (traditional “landscape” view vs. cameras rotated 90 degrees) to determine the optimal detection probability for small and medium-sized animals such as martens and fishers.

The height at which we deployed our cameras may have also exacerbated low detection rates of martens. In other overwinter surveys for mustelids, cameras were generally located ~2.5–4.0 m above the ground (Krohner 2020, Lukacs et al. 2020, Long et al. 2024) and previous surveys in the Olympic Range deployed cameras ~2.5–3.5 m above the ground (Howell et al. 2019, 2020, 2021, 2023), whereas we deployed our cameras at ~4.5 m. This differential was caused by incorrectly measured cords we used to mark height in the field. Previous research has shown that detection rates can vary depending on camera height, angle, and tilt (P. Apps and McNutt 2018, P. J. Apps and McNutt 2018), with increased camera height resulting in lower detection rates in some cases (Meek et al. 2016).

We used three different Reconyx camera models at primary survey stations in this study: Hyperfire 2, Hyperfire 2 Professional, and Ultrafire. The Hyperfire 2 and Hyperfire 2

Professional cameras featured a passive infrared (PIR) motion detector that triggered the camera to begin taking photographs when any object whose temperature differed from the ambient temperature moved horizontally within its detection zone, approximately 1/8 of the way across the camera's field of view (Reconyx, Inc. 2023a, b). The Ultrafire camera featured a similar PIR motion detector, but the detection zone was subdivided into 12 subzones and required the object to cross a boundary between 2 subzones to trigger the camera (Reconyx, Inc. 2017). Given geometry, an increased camera height would require an object to travel farther along the ground or scent dispenser tree to pass through 1/8 of the camera's field of view or to cross a boundary between two detection zones, thus potentially reducing the probability that the camera would be triggered. Additionally, we presume that the body heat of a small animal like a marten may be less detectable as camera height increases. We detected Douglas's squirrels (*Tamiasciurus douglasii*) and/or northern flying squirrels (*Glaucomys sabrinus*), both of which are smaller than martens, at 49% (33/67) of primary stations (Table 4). In contrast, previous marten surveys in the Olympics, that used automatic scent dispensers, detected one or both squirrel species at 83–97% of survey stations (5/6 stations, Howell et al. 2019, 5/6 stations, Howell et al. 2023, 30/31 stations, B. Howell pers. comm.). While we were obviously still capable of detecting small-bodied mammals at our primary sites, comparative evidence suggests that we may have had lower detection rates than previous studies.

Notably, however, issues associated with camera height may have been ameliorated by the fact that snowfall accumulations would have decreased the distance between the camera and any animal traversing the surface of the snow during the portion of the year in which mustelids were most likely to be detected. The arboreal nature of martens should have also assisted in generating detections, as the distance between the camera's motion detector and the animal

would have decreased as it climbed the tree to investigate the dispenser and bone. Ultimately, without a second camera simultaneously deployed at a lower height, we cannot know what effect our camera height had on the marten detection rate. Regardless, we suggest that future researchers in the Olympics deploy cameras at 2.75–3.5 m above ground, as very few sites in our study accumulated more than 2.75 m of snow.

We detected fishers in a wider band of elevation than martens. The maximum detection elevations for both species were similar (martens: 1,523 m, fishers: 1,576 m), but fishers were detected at lower elevations than martens (675 m and 903 m, respectively). Elevation is a proxy for expected snow depth and other studies have found that, during winter, fishers are excluded from high-elevation sites by deep, soft snow, and martens are excluded from lower elevation sites by increased competition and interspecific killing by fishers (Jensen and Humphries 2019, Pauli et al. 2022). Our study supports this elevational relationship, although with a limited sample size, as the highest-elevation fisher detection (1,576 m) occurred in the summer, and winter detections occurred at a reduced range of elevations, between 675–1,328 m. Importantly, martens were historically sympatric with fishers at low elevations, including along the coast line on the Olympic Peninsula (Dalquest 1948). Our study supports the hypothesis that martens have suffered an extreme contraction in range on the Olympic Peninsula and may persist only in high-elevation forests. Moreover, we speculate that a growing fisher population could threaten continued marten persistence if a warming climate allows fishers to occupy the highest-elevation forests year-round and martens experience reduced access to subnivalium refugia due to reduced snowpack (Pauli et al. 2013, Fountain et al. 2022).

Recommendations

The consistently low number of marten detections here and in past surveys provides strong evidence that researchers have not simply missed a robust marten population remaining on the Peninsula. Although martens persist within ONP and ONF, our results support past findings that they are at critically low abundances and are very likely to be relegated to high elevation forests. Due to the rarity of this population—and despite extensive survey efforts since 1991—we continue to have a poor understanding of where population cores exist and uncertainty remains as to why the population has not recovered within the protected boundaries of ONP and designated Wilderness Areas of ONF. Below we make suggestions for the direction of future monitoring efforts and direct management actions.

Future Survey Efforts

We suggest the highest priority for future survey efforts is the collection of genetic samples. Understanding the genetic health of the population may provide insights as to why the population remains at low abundances and what management actions would be the most appropriate to address those causes. The collection of genetic samples will also provide opportunities to continue to collect occurrence data on the population. We suggest that these efforts should be intensive rather than extensive and focus on areas where martens have been detected the most recently and frequently. Furthermore, survey efforts exploring areas outside of known marten detections should be focused at elevations $\geq 1,000$ m.

Researchers have found that martens are most likely to be detected during the winter (Zielinski et al. 2015) and, indeed, we only detected martens during the snow-on period of the year. While winter presents the most likely time of the year to attract and detect martens at survey stations it is also the most challenging time of the year to conduct research in the Olympic Mountains due to large amounts of snow accumulation. The passive hair snaring techniques used

in this study leave genetic samples exposed to precipitation and UV light, which degrades DNA, until samples can be collected. During the winter, inaccessible survey stations could potentially leave genetic samples exposed to the elements for months at a time. Thus, future genetic collection efforts should also explore methods to protect collected samples from degradation. Alternatively, active scat searches during the summer months—and in accessible locations during the winter—could potentially collect fresh samples which may be more likely to be viable. Conservation detection dogs have been shown to be effective in detecting scat from rare and cryptic species (MacKay et al. 2008, Thomas et al. 2020, McKeague et al. 2024). In 2018, conservation detection dogs were used to survey for mustelids and felids in ONF and were successful in collecting two fisher scats (Howell et al. 2019).

Federal and State Listing

Currently, martens in Washington are not considered to be threatened or endangered at either the state or federal level. This lack of designation likely owes to a large and stable marten population in the Washington Cascades (Zielinski et al. 2001, Christophersen et al. 2005, Reid et al. 2010). However, NatureServe lists Washington's coastal marten population as critically imperiled (NatureServe 2024). Additionally, the state of Washington categorizes the coastal marten population as a Species of Greatest Conservation Need (SGCN). Whereas the SGCN designation signifies “a clear need in future research and monitoring efforts” (Washington Department of Fish and Wildlife 2015a pg. 3-7), it does not convey the same urgency as state or federal designations of “threatened” or “endangered.” For instance, of the 44 mammal species designated as SGCN, 7 are considered to be in critical condition, 22 are at “low” population sizes compared to their historical abundance, 4 are considered to have moderately sized populations, and 11 have unknown population sizes (Washington Department of Fish and Wildlife 2015a).

Furthermore, the population of coastal martens was designated as a SGCN in 2015, at which time it was considered to be in critical condition, as it “probably [has a] state [population] of fewer than 25 individuals at any one time” (Washington Department of Fish and Wildlife 2015*a*, pg. 3–7). But the population did not receive protections from harvest until 2021, when a moratorium was placed on trapping in Callam, Jefferson, Grays Harbor, and Mason counties (Washington Department of Fish and Wildlife 2009, 2012, 2015*b*, 2018, 2019, 2020, 2021, 2022, 2023). Despite this lack of legal protections on the Olympic Peninsula until recently, we are unaware of any martens being harvested by trappers since the 1970s (Moriarty et al. 2019), and, indeed, speculate that harvesting is unlikely given the rarity of martens and their probable restriction to isolated, high-elevation habitat.

The SGCN list comprises 268 species of mammals, birds, reptiles, amphibians, fish, and invertebrates that are in need of conservation actions and/or further research to understand population trends (Washington Department of Fish and Wildlife 2015*a*). However, SGCN status does not guarantee that Washington Department of Fish and Wildlife (WDFW) will initiate conservation actions or provide funding to support research or management actions (Washington Department of Fish and Wildlife 2015*a*). Indeed, WDFW has recognized that they do not have the funding capacity to implement all needed research and management actions for SGCN-designated species (Washington Department of Fish and Wildlife 2015*a*). Thus, efforts to recover this population would be facilitated by additional funding that may accompany an elevated threat status at the state and federal level.

The results of our study, in conjunction with those of previous marten and forest carnivore surveys conducted on the Olympic Peninsula, appear to support the federal and state listing of the Olympic marten population as either “threatened” or “endangered.” The

Endangered Species Act (ESA) allows the listing of a species, sub-species, or Distinct Population Segment (DPS) (Federal Register 1996). Genetic evidence suggests that the Olympic Peninsula was historically colonized by martens dispersing from the Washington Cascades, and the two populations, though now disjunct, remain in the same phylogenetic clade (Schwartz et al. 2020). With martens in the Cascades at high abundance, the Olympic marten population would need to be recognized as a DPS to be federally listed under the ESA. Washington state also allows population segments to be listed as threatened and endangered at the state level (e.g., southern resident killer whale [*Orcinus orca*]) and if a species or DPS is listed at the federal level Washington matches that listing at the state level (Washington Department of Fish and Wildlife 2015a). To be considered a DPS, the population must meet two requirements: discreteness and significance (Federal Register 1996)—both of which can be argued to apply here.

The Olympic Peninsula is functionally an island for martens. No dispersal data are available from the Olympic population, but juvenile marten dispersal distances are generally less than 15 km (reviewed by U.S. Fish and Wildlife Service 2019), and martens have been shown to avoid moving through simplified forests and open habitat types to reduce predation risks (Moriarty et al. 2015, 2016b). The Olympic Peninsula and the Cascades Range are separated by more than 100 km, and today's martens would be required to traverse the human-modified landscapes (e.g., urbanization, agricultural fields, and managed forests) of Puget Sound and Chehalis River Valley regions to move between these populations, making dispersal events extremely unlikely.

One metric the ESA uses to evaluate whether a population is significant to species conservation is if “its loss would result in a significant gap in the range of a taxon” (Federal Register 1996, pg. 4725). Historically, martens in Washington occupied coastal forests from

southern Washington north to the Olympic Peninsula as well as the Cascade and Blue Mountain ranges (Dalquest 1948, Zielinski et al. 2001). Martens are believed to be extirpated from all of Washington's coastal forests, with the only population west of the Cascade Range now residing in the high-elevation forests of the Olympic Peninsula. Thus, a loss of the Olympic population would represent a complete extirpation of marten populations from western Washington.

Because this marten population appears to meet the discreteness and significance requirements of a DPS, we recommend that a petitioning process be considered to evaluate endangered or threatened status. Marten populations in coastal northern California and Oregon were recently listed as a threatened DPS of the Pacific marten under the ESA, and exemplify the process and standards used by the listing agencies (U.S. Fish and Wildlife Service 2019, Federal Register 2020). Threatened or endangered status would clearly demonstrate the reality of the Olympic marten population's size, and the threats to its continued persistence, as well as the urgency for conservation action. Importantly, a federal listing could also increase access to funding sources and the likelihood that proposed conservation action would be funded.

Population Augmentation

Finally, we suggest that direct management actions, such as population augmentation through translocations, should be considered to increase the Olympic marten population and expand its range throughout the ONP and ONF. Taking direct management action may demonstrate that current policy and management is in place to secure this population and obviate the need for state and federal listing. Historical species accounts and trapping data indicate that the Olympic marten population's decline began to be detected in the late 1940s (Dalquest 1948, Scheffer 1995, Moriarty et al. 2019), and the results of surveys conducted since 1991 indicate that the population has not recovered. The ONP and ONF designated Wilderness areas provide

extensive tracts of old-growth forest and protection from the direct anthropogenic disturbances that are considered drivers of mustelid decline on the Olympic Peninsula (i.e., habitat destruction and direct mortality from harvest). The lack of a population rebound within these protected areas is a strong indication that this population is not capable of recovering without human intervention. While continued monitoring will be valuable to determine where martens are occurring on the Peninsula and should be pursued, monitoring without conservation action is unlikely to alter the trajectory of this population's trend.

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Appendix A

Primary Survey Station Locations

All location data provided use datum NAD 87.

Survey Station ID	UTM_X	UTM_Y	Elevation (m)
AA15	486475	5275264	897
AA16	485883	5271559	501
AA18	484809	5265412	1099
AA19	485321	5262041	672
AA7	486213	5302560	1202
AA8	483691	5299422	1439
AB14	489114	5282353	411
AB17	488814	5270130	1043
AB7	488873	5304198	1356
AB9	488168	5297810	956
AC11	492210	5290118	1327
AC15	490568	5276460	997
AC6	491246	5306176	759
AC8	491928	5299073	1403
AD10	494051	5293966	903
AD11	494018	5290995	1017
AD5	494196	5311337	690
AD9	494091	5297390	1180
AF12	500216	5287662	879
AF9	499990	5297702	942

H5	428467	5311884	1066
J17	436045	5270481	878
K5	438503	5309638	951
L5	440721	5310832	731
L7	440838	5304555	1265
M3	444146	5315576	1463
M6	444311	5306131	1523
N14	447153	5281015	1377
N16	446605	5273404	1003
N4	447515	5313879	763
N5	447082	5311311	1038
N6	445510	5308180	1177
N9	447846	5299333	1088
O13	449362	5282630	1449
O16	448257	5271942	432
P21	452836	5257198	675
Q11	457536	5289008	869
Q12	455293	5285639	1328
R13	459686	5284769	1242
R4	459236	5314691	1427
S11	462234	5287985	671
S12	460433	5286552	756
S18	460569	5265494	938
S19	461458	5261365	999
S7	460931	5303599	1443
S8	462009	5301086	1225
T10	466112	5294500	535
U14	468563	5277775	604
U15	466589	5276110	571
U17	466567	5267586	1170
U4	466194	5313639	1113
U5	467769	5309979	1437
U9	467381	5295915	711
V18	470471	5267388	974
V20	470573	5260325	620
W15	472784	5276784	1144
W6	473596	5306156	1402
X13	476496	5283109	1172
X16	475001	5273794	691
X7	477532	5304075	1049
X9	477760	5298623	1576
Y12	479699	5285716	901
Y17	479572	5268675	813

Y6	479950	5306153	908
Z4	482339	5314701	801
Z5	481815	5310691	1672
Z7	481996	5305509	785

Secondary Survey Station Locations

All location data provided use datum NAD 87.

Survey Station ID	UTM_X	UTM_Y	Elevation (m)
AA15X	486402	5275185	887
AA16X	485821	5271425	468
AA18X	484965	5265380	1176
AA19X	485422	5261937	654
AA7X	486313	5302491	1160
AA8X	483748	5299574	1426
AB14X	489229	5282348	415
AB17X	488944	5270174	1006
AB7X	488992	5304116	1369
AB9X	488134	5297714	960
AC11X	492350	5290129	1303
AC15X	490651	5276627	971
AC6X	491158	5306277	798
AC8X	492029	5298988	1401
AD10X	494177	5293850	828
AD11X	493966	5290880	1029
AD5X	494288	5311209	689
AD9X	493913	5297389	1190
AF12X	500192	5287484	964
AF9X	500048	5297588	926
H5X	428468	5312007	1065
J17X	436061	5270378	868
K5X	438416	5309557	966
L5X	440873	5310775	745
L7X	440785	5304668	1271
M3X	444176	5315679	1469
M6X	444148	5306130	1527
N16X	446590	5273518	991
N4X	447639	5313886	732
N5X	447119	5311422	1012
N6X	445546	5308288	1154
O13X	449238	5282613	1436

P21X	452668	5257195	708
Q11X	457629	5288883	822
Q12X	455356	5285532	1325
R13X	459653	5284908	1254
R4X	459117	5314581	1380
S11X	462133	5287819	660
S12X	460477	5286418	773
S18X	460547	5265601	948
S19X	461563	5261450	990
S7X	461020	5303651	1462
S8X	462167	5301072	1236
T10X	466189	5294594	532
U14X	468687	5277721	569
U15X	466691	5275992	499
U17X	466631	5267474	1219
U4X	466358	5313668	1117
U5X	467936	5310126	1479
U9X	467472	5295790	720
V18X	470430	5267521	991
W6X	473733	5306114	1387
X13X	476392	5283115	1165
X16X	475128	5273788	716
X7X	477429	5303947	1036
X9X	477679	5298750	1558
Y12X	479837	5285766	906
Y17X	479685	5268698	780
Y6X	479869	5306055	904
Z4X	482355	5314818	800
Z7X	482035	5305639	792