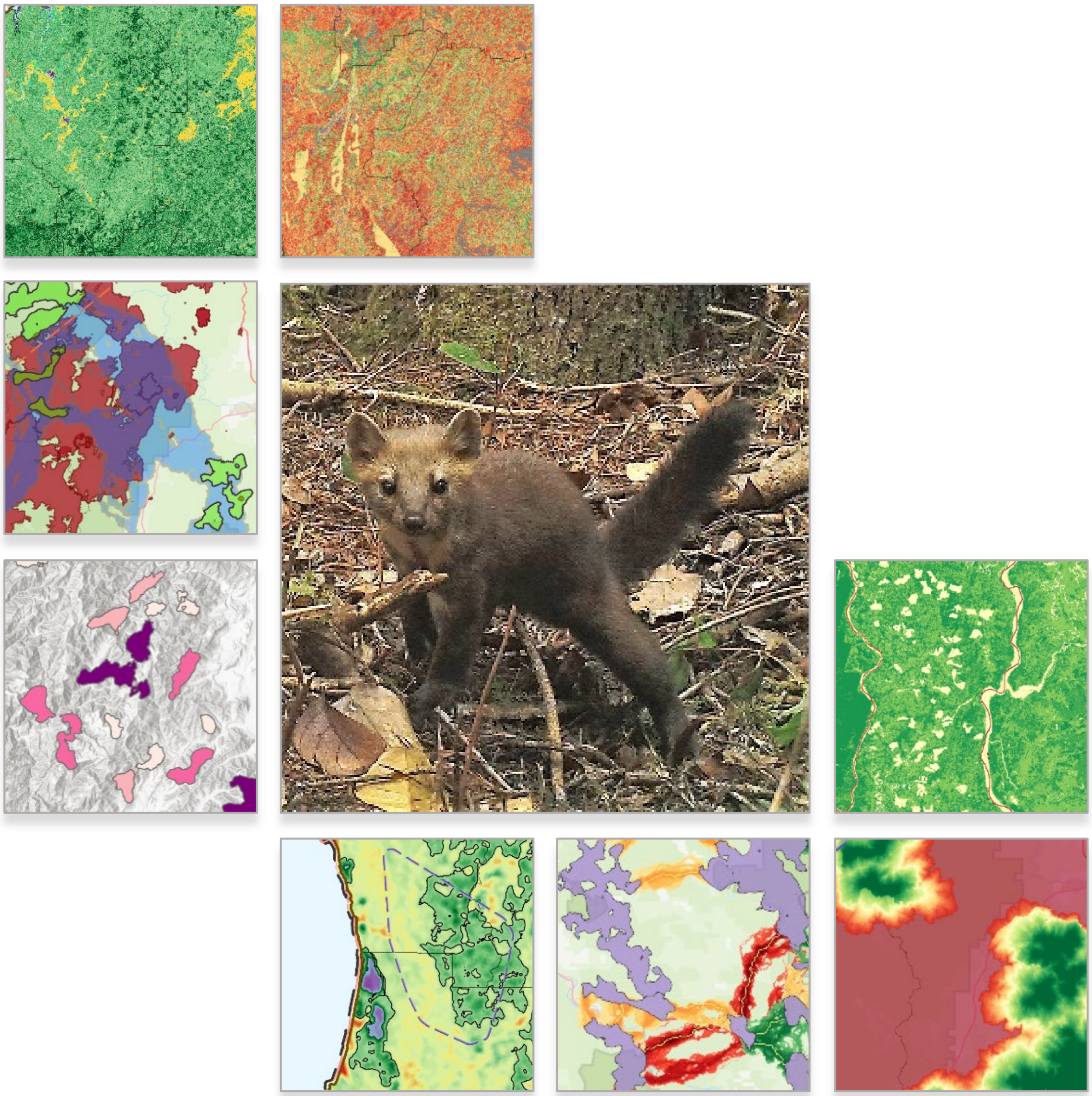

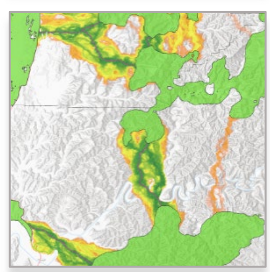

A LANDSCAPE CONNECTIVITY ANALYSIS FOR THE COASTAL MARTEN (*Martes caurina humboldtensis*)

Gregory Schrott & Joel Shinn
USFWS Strategic Habitat Conservation Team
Arcata Fish & Wildlife Office, Arcata, CA

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Author Contact: Gregory_Schrott@fws.gov

Document Availability

This report and associated spatial data are available online at:

https://www.fws.gov/arcata/shc/marten

Cover

Cover photo of coastal marten cropped from original by U.S. Forest Pacific Northwest Research Station and Oregon State University, .

Executive Summary

The coastal marten (also known as Humboldt marten) is a medium-sized carnivore that is endemic to northwestern California and western Oregon. In 2018, it was proposed for listing as threatened under the Federal Endangered Species Act and was listed as endangered under the California Endangered Species Act. The species primarily inhabits mature coastal forests in this region, but can also be found in dune forest habitat and certain areas with dense shrub cover on serpentine soils. Coastal marten populations declined from a combination of heavy trapping pressure in the late 19th and early 20th Centuries and the loss and fragmentation of mature forests. It is currently known to exist in four isolated populations, two in California and two in Oregon. A conservation strategy document for the species was recently produced by the Humboldt Marten Conservation Working Group that incorporated a landscape-scale habitat model. However, this model had limitations in terms of its ability to assess habitat connectivity at scales that would facilitate conservation planning efforts. It was also based on locations of the remnant marten populations at high elevations and therefore did not depict suitable habitat in lower elevation coastal areas where the species is known to occur. Therefore, we developed a landscape-scale habitat connectivity model for the coastal marten across the extent of its historical range, with the goals of better understanding the distribution of habitat, the likely degree of isolation of the existing populations, and the potential for the species to recolonize areas of suitable but unoccupied habitat. We hope that this model would be able to inform ongoing Species Status Assessment (SSA) and Endangered Species Act listing processes, as well as ongoing conservation planning efforts related to the coastal marten.

We developed our coastal marten connectivity model using a spatial analysis tool called Linkage Mapper, along with information about the species' biology to assess its ability to move through or occupy habitat. We first identified "habitat cores", which are relatively large patches (>1500ha) that are likely to contain sufficient high quality habitat to support long-term occupancy by coastal martens, and thus represent important areas for the species' conservation. They are not intended to represent all potentially suitable habitat on the landscape. These habitat cores are then connected via "least-cost corridors", which are estimated to be the easiest routes for dispersing martens to move through based on land cover types and the presence of features such as rivers and roads. The habitat cores were primarily identified using an old-growth structure index (OGSI), a fine-scaled spatial data layer that has been developed by researchers with the U.S. Forest Service and Oregon State University as a means of evaluating the effectiveness of the Northwest Forest Plan. We also incorporated data on the location of serpentine soils that likely support habitat suitable for coastal martens. The corridors were mapped by dividing the landscape into a raster (gridded) surface with 30m X 30m cells, and assigning each cell a "resistance value" based on land cover type, with higher values indicating the cover type as more difficult or hazardous for martens to move through. Based on a set of guiding assumptions about marten biology and habitat use, we assigned higher resistance values to younger forests, non-forested areas, and large roads and rivers, and lower resistance values to areas with mature forest or suitable serpentine habitat. The least-cost corridors were mapped onto swaths of land that collectively had the lowest resistance between habitat cores. Corridor width varied considerably depending local conditions.

Using our input parameters, our model identified 51 habitat cores linked by 97 least-cost corridors (Fig. 10). Habitat cores ranged in size from 1,624 – 178,091ha, with the total area of all habitat cores being 788,290ha (3,043.6 miles^2). Over 82% of this area was on lands managed by the U.S. Forest Service. Only 29% of the total area of the habitat cores was on lands managed with the strictest protections for biodiversity (USGS GAP status categories 1 and 2). There are several possible ways of assessing the degree of isolation of habitat cores or populations from one another. We used a metric called "cost-weighted distance", which takes the resistance values within the least cost corridor into account so that the difficulty of the terrain and barriers crossed was factored in as well as the physical distance. This cost-weighted distance was then normalized for comparison

to km. We used a standard of ≤15 cost-weighted km for a corridor to be considered "well connected" (based on published marten dispersal distances), with corridors ≤45 cost-weighted km considered "moderately connected" and longer corridors "poorly connected". Based on this standard, 36.1% of the corridors were considered well connected, 28.9% moderately connected, and 35% poorly connected. The more traditional method of assessing connectivity by the Euclidean ("as-the-crow-flies") distance between habitat cores without taking the nature of the intervening landscape into account would have classified 64.9% of the corridors as well connected (Table 4).

The two Oregon populations were indicated to be poorly connected to one another, and to the populations in California no matter what metric was applied. The two California populations were mapped as connected by a large habitat core rather than a corridor (although the intervening area is not known to be currently occupied). We also ran a separate Linkage Mapper trial that treated the known population boundaries as habitat cores, and this classified these two populations as moderately connected by cost-weighted distance and well connected by Euclidean distance (Fig. 12). We identified five "habitat core clusters" that could be linked by least-cost corridors of ≤45 cost-weighted km (Figs. 13 and 14). Three of these were in Oregon, two of which included extant coastal marten populations. Another cluster included most of the habitat cores in California along with some adjacent ones in Oregon, and the fifth cluster linked two relatively small, isolated, unoccupied cores in California. Habitat cores within the same cluster can be considered "functionally connected" for coastal martens to some degree, with long-term potential for dispersal, gene flow, and recolonization between them.

We also ran Linkage Mapper trials on two landscape scenarios that we developed to illustrate ways in which this model might be used as a conservation planning tool. The first trial explored how timber harvest might affect habitat connectivity, using an example from the area between the Six Rivers National Forest and Prairie Creek Redwoods State Park. Timber harvests here since 2012 (subsequent to the collection of the data used to produce the OGSI) shifted the least-cost path somewhat and increased the cost-weighted distance between two habitat cores (Fig. 15). The second trial identified a discrete area of the landscape in the Rogue River-Siskiyou National Forest where a modest improvement in habitat quality (for example, through allowing the forest to mature) had the potential to significantly improve connectivity between the habitat core clusters in Oregon and California (Fig. 16). Linkage Mapper's output includes several metrics that can be used to assess the connectivity value of individual linkages between habitat cores. These metrics can then be used to help evaluate the potential impacts of changes on the landscape, including comparing among alternative proposed management actions.

There are some important caveats in considering the results of our habitat connectivity analyses. First, there are a number of aspects of coastal marten dispersal behavior that are not well understood; in particular how dispersing animals respond when encountering sub-optimal habitat, non-habitat, and barriers such as rivers and roads. There is also more to be learned about what constitutes high and low quality habitat for the species, and how this might vary over the breadth of its range. We incorporated a number of simplifying assumptions into our model to account for data gaps. Second, the OGSI and most other habitat data we used were based on surveys and analyses conducted in 2012 and therefore do not reflect changes from more recent disturbance events such as timber harvest and fires. They also have a modest rate of misclassification errors, and while the data are very useful for describing patterns of forest structure at the landscape scale, the potential for such errors to give an inaccurate view of forest structure increases at finer scales. Third, it needs to be understood that the habitat cores do not represent all coastal marten habitat on the landscape, nor should all of the area within these cores be considered suitable habitat for the species. The final set of habitat cores we used in the model had a minimum size of 1500ha, but many smaller areas of "habitat core" could be identified using a smaller minimum size threshold; indeed, many of these smaller cores form important anchors of the corridors identified in the model.

Given that the SSA identified small and isolated populations as one of the major threats to the coastal marten, maintaining habitat connectivity between populations where it exists and improving it where it is poor should be high conservation priorities. Connectivity between existing populations and large patches of suitable but unoccupied habitat will be important in allowing the species to expand its distribution and increase its numbers. The clearest examples of such areas identified by our model include (1) a set of habitat cores in and around the Siuslaw National Forest to the east of the Central Coastal Oregon population, and (2) a number of habitat cores adjacent to the two California populations that are primarily located on the Six Rivers National Forest and Redwood State and National Parks. If translocations are considered as a conservation tool for the coastal marten, the chances of successfully establishing and maintaining new populations using this method will be higher if they can have a degree of connectivity to one or more established populations. Therefore, patterns of landscape connectivity should be taken into account in selecting potential release sites.

There are several areas of research that could provide important new data to improve our understanding of habitat connectivity for coastal martens. These include: (1) additional surveys aimed at better understanding the current distribution of the coastal marten (especially in California), (2) habitat use studies of radio-collared animals that could improve our understanding of preferred habitat types and dispersal behavior, and (3) analyses of genetic structure among and within coastal marten populations that could provide insights into gene flow patterns over time.

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Introduction

The coastal marten (*Martes caurina humboldtensis*, also referred to as Humboldt marten) is a medium-sized carnivore in the mustelid family that is endemic to northwestern California and western Oregon. A subspecies of the more widely distributed Pacific marten (*M. caurina*) (USFWS 2018), it has been proposed for listing under the Federal Endangered Species Act (83 FR 50574) and was listed as endangered under the California Endangered Species Act in 2018 (14 CCR 670.5). The coastal marten has experienced a long-term population decline and range contraction from a combination of heavy trapping pressure in the late 19th and early 20th Centuries and the historical and ongoing loss and fragmentation of the mature forests that are their primary habitat (USFWS 2018, Slauson *et al.* 2019a). In addition to late seral coastal forests, coastal martens have also been found inhabiting habitat on serpentine soils that may have sparse canopy cover (Zielinski *et al.* 2001, Slauson *et al.* 2007) and in relatively young coastal dune forests in Oregon (Zielinski *et al.* 2001, Moriarty *et al.* 2016, Linnell *et al.* 2018). All of these habitat types have a dense layer of ericaceous shrubs (i.e. members of the family Ericaceae, such as salal (*Gaultheria shallon*), huckleberries (*Vaccinium* sp.), and Pacific rhododendron (*Rhododendron macrophyllum*)) (Slauson and Zielinski 2007, Slauson *et al.* 2007, 2019a, USFWS 2018). The presence of structures usable for denning and resting, such as large tree cavities or hollow logs, are also an important habitat component (Slauson *et al.* 2007, 2019a, Slauson and Zielinksi 2009) (large boulder piles may sometimes be used in serpentine-dominated sites). Various rodents are usually the coastal marten's primary prey, with birds and berries also comprising a significant portion of their diet (Slauson and Zielinski 2017, Eriksson *et al.* 2019). The relative importance of different food types varies with season and site. For more detailed treatments of the taxonomy, life history, and conservation status of the coastal marten, see USFWS (2018) and Slauson *et al.* (2019a).

A conservation strategy for the species has been developed by the multi-agency/organization Humboldt Marten Conservation Working Group (Slauson *et al.* 2019a). The conservation strategy incorporated a landscape scale habitat model (based on Slauson *et al.* 2019b) that was used to designate "Population Reestablishment Areas" (PRAs) where enough suitable but unoccupied habitat appeared to exist to support five or more female marten home ranges (≥1500ha) (see Fig. 25 in Slauson *et al.* 2019a). It also identified a Landscape Connectivity Area (LCA) that lies between a PRA and an existing population, but does not appear to hold much suitable habitat. While this model was useful in many ways, it also had drawbacks that limited its utility for some aspects of conservation planning. First, because the habitat model characterized conditions at locations where martens have been detected within existing remnant populations, it included terms for elevation and precipitation. It therefore did not explicitly map suitable habitat in redwood-dominated forests close to the coast, in spite of the species having been recorded from there historically and there being recent records of animals dispersing to Prairie Creek Redwoods State Park and settling on territories there (USFWS 2018). The conservation strategy suggests that Prairie Creek and other units of the Redwood National and State Parks nearby be considered to represent PRAs, but the model this recommendation is based on does not show these areas as suitable habitat. Indeed, it does not map suitable habitat in other low elevation coastal areas in the historical distribution of the coastal marten. Second, the LCA is coarsely mapped over a relatively large area of 33,977.1ha (131.2 miles^2) and is mostly composed of private industrial forest land and tribal lands. As such, it is unlikely that conservation measures aimed at improving habitat connectivity for coastal marten could be applied over the entire area, or that the habitat over all of this broad area has similar potential for connectivity management. Third, the strategy did not identify any PRAs or LCAs in Oregon in spite of there being two extant populations there. With several marten-related conservation initiatives in progress, there seemed to be a need for a model that explicitly mapped remaining large blocks of potentially suitable habitat for the species throughout its historical range combined with an assessment of connectivity between these blocks at relatively fine spatial scales.

"Connectivity" is a concept in landscape ecology and conservation planning that can be applied to habitats, ecological processes, and both gene flow and the movement of individual organisms within a species (see Aune *et al.* 2011, Rudnick *et al.* 2012, and Kool *et al.* 2013 for general reviews). This report describes a model of *functional connectivity* (Rudnick *et al.* 2012) for the coastal marten, in that it uses data on the distribution of habitat and important features on the landscape as well as information about the species' biology to assess its ability to move through or occupy areas in its historical range. Such biological information could come from empirical data on coastal martens, expert opinions, or inferences from studies of related marten subspecies or species. Having a better understanding of habitat connectivity for the coastal marten could improve conservation outcomes for this species in several ways. The small size and isolation of existing coastal marten populations was identified as a major threat in the Species Status Assessment (SSA) (USFWS 2018) and was a primary reason for the proposal listing the species as threatened. Management that maintains or improves connectivity can reduce some of the negative effects of habitat fragmentation, such as inbreeding. It can facilitate the recolonization of unoccupied habitat, thereby increasing the overall population size, and help "rescue" populations that have experienced sudden declines from events like disease outbreaks or wildfires (Crooks and Sanjayan 2006). Incorporating habitat connectivity into conservation planning could also help in selecting optimal locations for potential reintroduction efforts (Dunham *et al.* 2016), by providing a better understanding of the degree to which they might be able to interact with other populations (i.e. potential gene flow) and how much unoccupied habitat the new populations might be able to recolonize on their own.

We developed a model of habitat connectivity for the coastal marten throughout its historical range. This model was created using a spatial analysis tool called Linkage Mapper (along with a set of related tools), and its primary purpose was to identify large blocks of potentially suitable habitat and the most likely potential corridors connecting them. We also used it to assess the relative quality of these linkages. Our goal was for this model to be used to inform ongoing and future coastal marten conservation efforts, including the SSA (USFWS 2018) and listing decisions under the Endangered Species Act. We do not consider this to be a comprehensive model of all potential habitat on the landscape or to realistically simulate marten behavior in all circumstances as there are significant data and knowledge gaps in what is known about the species, as well as some limitations to the spatial data available. Rather, the model should be viewed as a best effort to summarize the available information into a useful tool for informing decision makers. We describe a number of ways the model might be improved or expanded upon in the future in the Discussion section.

Methods

Study Area

Coastal marten distribution

Our modeled landscape encompassed the entire historical distribution of the coastal marten, along with an extension to the east (Fig. 1). The historical range of the coastal marten is thought to extend from the Columbia River in northern Oregon to central Sonoma County, California along a coastal strip <100km wide, an area of about 57,293km^2 (22,121 miles^2). The California portion was described by Grinnell and Dixon (1926) based on museum specimens and trapping records, and this area largely correlates with the area over which coastal fog influences forest communities (Grinnell *et al.* 1937, Slauson *et al.* 2019a). More recent research indicates that marten populations in coastal Oregon should also be considered to be part of the same evolutionary lineage as *M. c. humboldtensis* rather than *M. c. caurina*, a different subspecies of Pacific marten into which they had originally been classified (Slauson *et al.* 2009a, Schwartz and Pilgrim 2016). There is some uncertainty as to whether the potential distribution of coastal marten should be considered to extend further east in some areas than is shown in Fig. 1 (Zielinski *et al.* 2001, Moriarty *et al.* 2018a). This uncertainty was one of our reasons for including the extension to delineate the modeled landscape, although buffering the primary area of interest is also considered a best practice when developing a landscape connectivity model (Beier *et al.* 2011, McRae and Kavanagh 2016). The total area of the modeling extent was 108,845km^2. The boundaries were determined in part by the extent to which we were able to achieve complete overlap between our chosen spatial datasets, as some of these had more limited coverage than others.

There are currently four known spatially distinct populations of coastal marten, referred to as the Northern Coastal California, California-Oregon Border, Southern Coastal Oregon, and Central Coastal Oregon populations (USFWS 2018) (Fig. 1). The Northern Coastal California population is thought to be the largest of the four, although it likely contains <100 animals (USFWS 2018). Much of the information on habitat use by the coastal marten derives from studies conducted in this population. The California-Oregon Border population was discovered in 2011, and while its size and distribution are still being studied, it was presumed by USFWS (2018) to be small. The Southern Coastal Oregon population also appears to be relatively small, but seems to be distributed in pockets over the largest area of the four (USFWS 2018). Research into habitat use in this area is ongoing, but may differ somewhat from what is typically seen in the California populations (Moriarty *et al.* 2019). The Central Coastal Oregon population numbers about 70 adults (Linnell *et al.* 2018), and inhabits a unique habitat situation in a narrow strip of coastal dune forest between the Coos and Siuslaw Rivers. These patches are dominated by shore pine (*Pinus contorta contorta*) and Sitka spruce (*Picea sitchensis*), and have extensive areas of relatively young forest (<70 years old) (Eriksson *et al.* 2019).

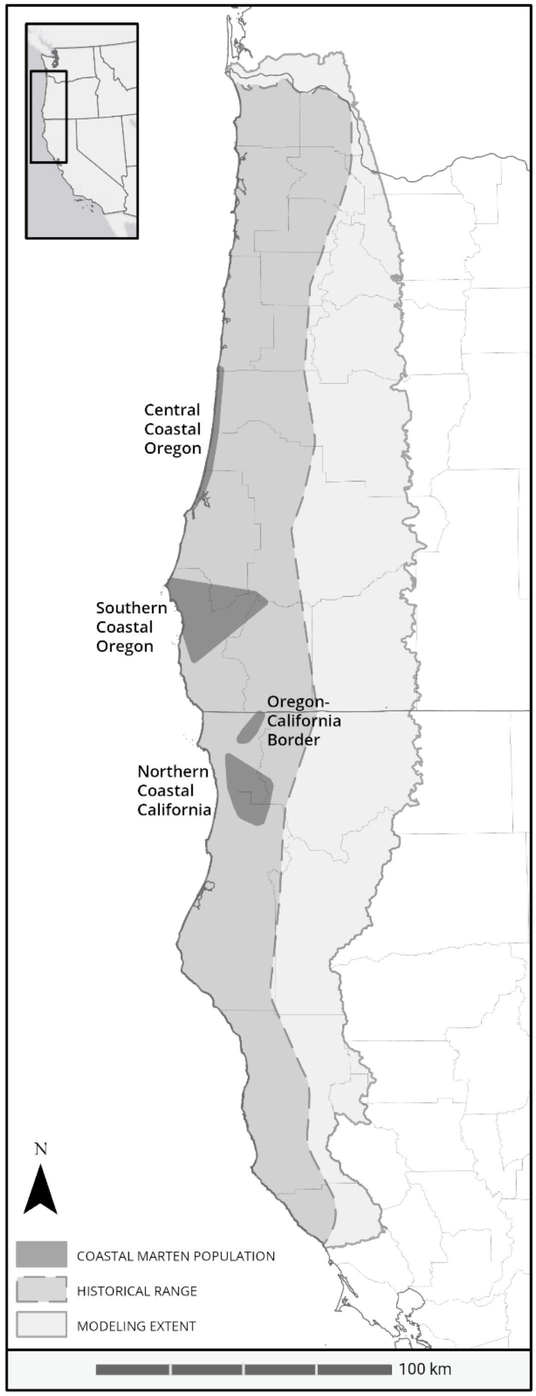


Figure 1. Historical range of the coastal marten (*Martes caurina humboldtensis*), with the four known extant populations described in USFWS (2018). The "modeling extent" shows the additional buffer we added around the historical range to create the entire modeled landscape.

Landscape description

Most of the coastal marten's historical range is included in the Coast Range and the Klamath Mountains/California High North Coast Range Ecoregions under the U.S. Environmental Protection Agency's

Level III Ecoregion classification system (Omernik and Griffith 2014), with small areas included in the Willamette Valley and the Central California Foothills and Coastal Mountains Ecoregions (Fig. 2a). Vegetation patterns vary considerably with latitude, distance from the coast, slope, and aspect, and species richness is very high (Ohmann and Spies 1998, Sawyer 2006, Sawyer 2007). Much of the historical range remains undeveloped and 85.6% can be classified as forested (≥10% tree cover) (Fig. 2b), although <30% of the landscape is classified as mature or old growth forest*. Human population density is relatively low, with few major population centers. Significant rivers running through this landscape include the Eel, Klamath, Trinity, Smith, Rogue, Coquille, Coos, Umpqua, Siuslaw, Alsea, and Tillamook, with the Columbia River delineating its northern boundary.

Nearly half of the landscape (46.8%) is managed by the Federal government†, 6.9% by state governments, 1.3% by local governments, and 1.2% by tribal governments‡, with the remaining 43.8% privately owned§ (Fig. 2c). Much of the private land in this landscape is managed for commercial timber production (Christensen *et al.* 2016). Over half of the land within the coastal marten's historical range is identified as having some sort of conservation status in the Protected Areas Database of the United States (USGS 2018) (Table 1, Fig. 2d), although not all of this area is potential marten habitat and only 10.4% of the area is primarily managed for biodiversity (GAP status 1 and 2).

Table 1. Percentage of the coastal marten's historical range in some sort of conservation status according to the USGS' Protected Areas Database of the United States. GAP status is a measure of the management intent to permanently protect biodiversity as defined by the USGS' Gap Analysis Program (GAP).

GAP status	Definition	Percentage
1	Managed for biodiversity, disturbance events proceed or are mimicked	5.0
2	Managed for biodiversity, disturbance events suppressed	5.4
3	Managed for multiple uses, subject to extractive use	44.5
4	Public lands with no known mandate for protection	2.4
Sum		*57.3*

* Based on the variables "OGSI 80" and "OGSI 200" in the GNN model output (LEMMA 2014a).
† Primarily the U.S. Forest Service (USFS), Bureau of Land Management (BLM), and National Park Service (NPS).
‡ Including but not restricted to tribal trust lands administered by the Bureau of Indian Affairs (BIA).
§ Includes some conservation areas managed by non-governmental organizations (NGOs) shown in Fig. 2c.

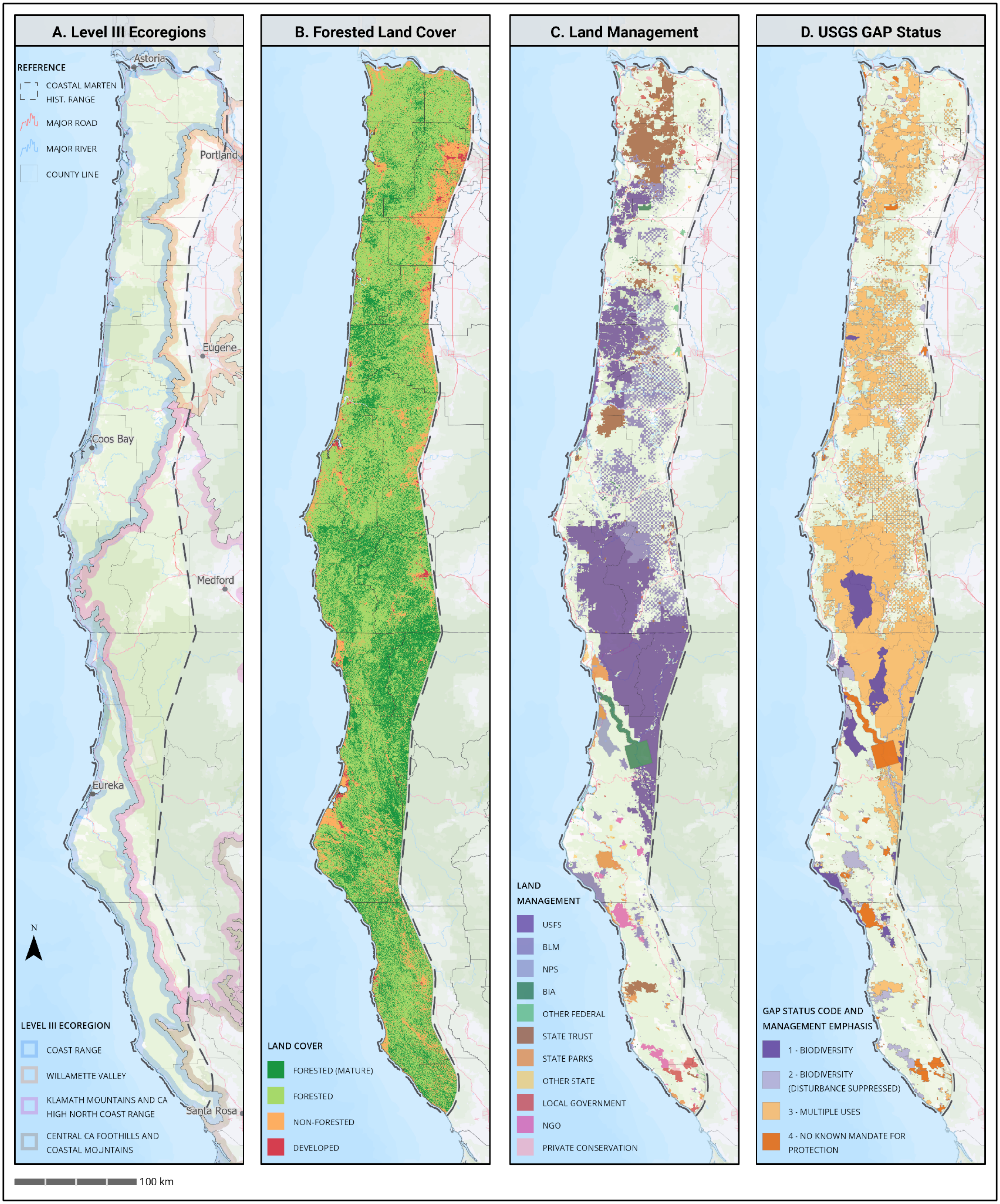


Figure 2. Descriptive maps showing ecological conditions and management status of the coastal marten's historical range.

Introduction to Linkage Mapper

Our assessment of coastal marten connectivity utilized a set of spatial analysis tools called Linkage Mapper (McRae and Kavanagh 2011), which was originally produced in conjunction with a state-wide landscape-scale habitat connectivity analysis done in Washington State (WHCWG 2010) and is now generally available online (http://www.circuitscape.org/linkagemapper). Linkage Mapper is comprised of a set of Python scripts shared in a toolbox for ArcMap (ESRI 2016). We opted for this approach to connectivity modeling primarily because of its relative ease of use and its excellent abilities to map corridors that were straightforward to interpret and could be readily incorporated into conservation planning.

Linkage Mapper identifies corridors on the landscape connecting habitat "cores", which are meant to represent the most important areas for conservation of the species or ecosystem of interest (Shirk and McRae 2015). These cores and the corridors connecting them are designated based on criteria provided by the user. The corridors estimate how well the species of interest could potentially move through the different land cover types and other features (roads, rivers, etc.) on the intervening landscape. This is done by using information about the species' biology to assign a "resistance value" to each land cover type or feature of interest: the higher the resistance value, the more difficult it is for the species to move through that part of the landscape. The landscape is then divided into a raster grid, with each grid cell being assigned a resistance value based on the dominant cover type it contains, with more resistance being added from the presence of other barriers such as major roads or rivers. Linkage Mapper identifies the "least-cost pathways" that incorporate the lowest additive value of cells between habitat cores, thus taking into account both the physical distance between the cores and the easiest terrain for the species to traverse (Adriaensen *et al.* 2003, Verbeylen *et al.* 2003, Beier *et al.* 2008, Sawyer *et al.* 2011). In addition, Linkage Mapper evaluates all possible pathways between neighboring habitat cores and compares the "cost-weighted" distance of each one in relation to the least-cost pathway. This allows the identification of wider corridors across the most traversable portions of the landscape that are of a more realistic size for use by wildlife. In general, we assumed that least-cost pathways reflect what would likely be the route of choice if a dispersing marten had complete knowledge of the landscape (McRae *et al.* 2008). We did not necessarily assume that all or even most dispersing martens would follow the least-cost corridors produced by our model, rather that those which did so would maximize their chances of survival and successful dispersal (LaRue and Nielsen 2008). (To be clear, the "cost" in the least-cost pathways refers to the cost of the animal in effort and risk from moving across that portion of the landscape; it does not refer to any financial costs related to land use or planning.)

Our overall approach to modeling habitat connectivity is summarized in Fig. 3, and described in detail in the Methods section.

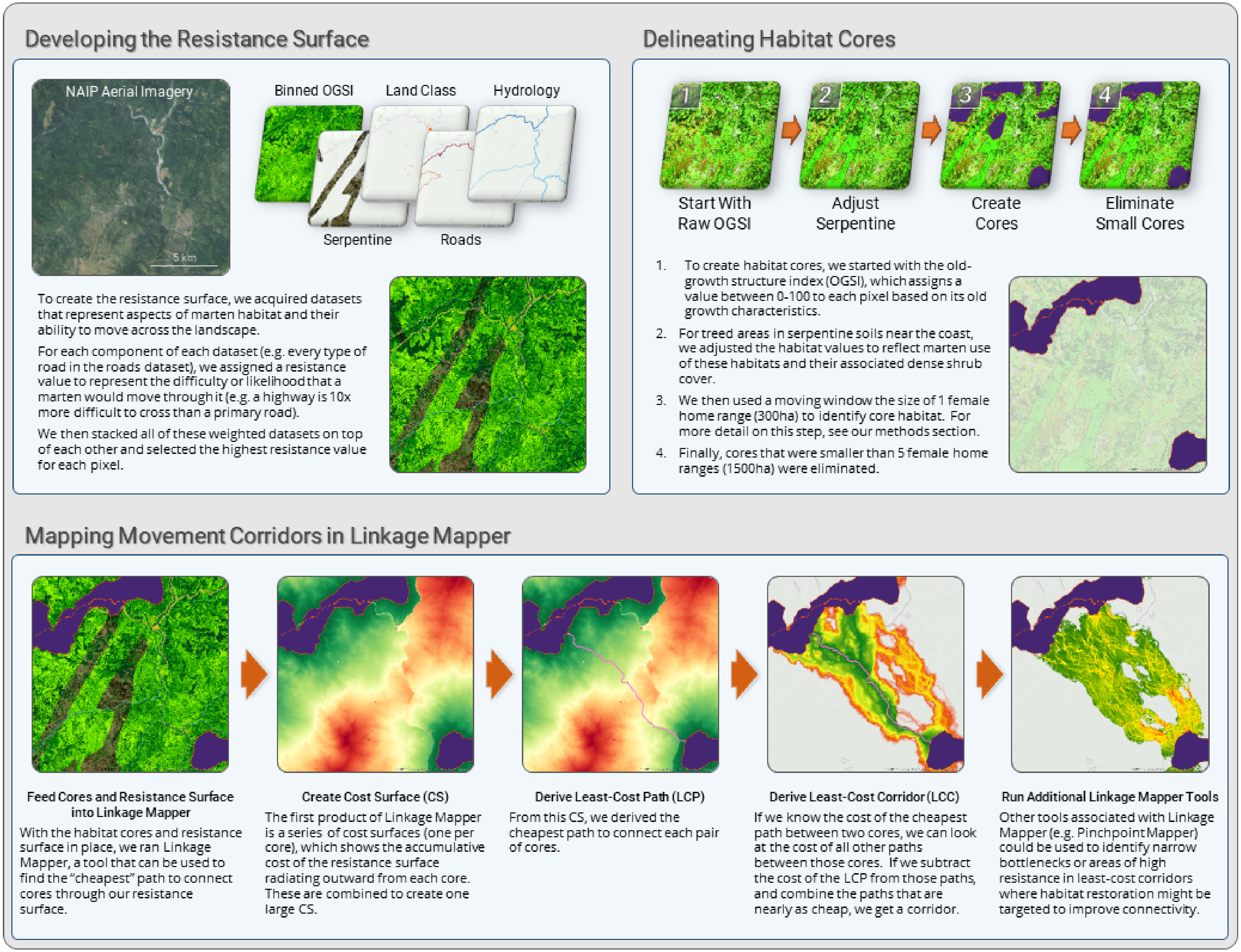


Figure 3. Summary of our approach to modeling habitat connectivity.

Key Assumptions

We based our model on a set of key assumptions derived from previous research on the coastal marten (or related marten species), and on landscape connectivity for wildlife more generally. These can be briefly summarized as:

1. **Mature forest is the best predictor of coastal marten habitat quality across most of its historical range.** Slauson *et al.* (2019a, b) tested an array of variables describing habitat, climate, topography, landscape pattern, ecological disturbance, and linear features on coastal marten occupancy and found that mature forest, as measured by an old-growth structure index at a scale of 1km, was one of two habitat parameters included in all of the best performing data models.

2. **Large blocks of mature forest (≥1500ha) within the coastal marten's historical range have the potential to support populations.** The Humboldt Marten Conservation Strategy (Slauson *et al.* 2019a) defined PRAs as blocks of suitable but unoccupied habitat ≥1500ha. While we used a somewhat different approach to modeling and mapping habitat, we adapted this standard as a minimum size in developing habitat cores to be connected by Linkage Mapper. We inferred that habitat at lower elevations near the coast is suitable based on the existence of the coastal populations in Oregon, recent detections in California, and historical trapping records.

3. **Mature forest is easier for dispersing coastal martens to move through (i.e. has lower resistance) than early seral forest.** Based on our assumption that mature forest represents the best quality home range habitat, we further assumed that dispersing martens would find these areas the easiest and least risky to move through and assigned them the lowest resistance. One of the primary drivers of risk in early seral forest may be predation by bobcats (*Lynx rufus*), which appear to be most abundant in parts of the landscape with higher concentrations of open or brushy areas or forests <30 years old (Slauson *et al.* 2019a). We modeled resistance to movement as increasing with decreasing forest maturity, reaching the highest resistance in the earliest seral stages corresponding with recent disturbance.

4. **Areas with serpentine soils within 30km of the coast that likely support ericaceous shrubs are usable (but often sub-optimal) habitat by coastal martens regardless of the presence of characteristics associated with mature forests.** Serpentine habitat at a 3km scale was the other best performing habitat parameter identified by Slauson *et al.* (2019a, b). While such sites often have sparse tree cover, they frequently have a dense layer of the types of shrubs favored by coastal martens.

5. **While the martens are also capable of using shore pine-dominated habitat along the coastal strand, this habitat type is very restricted in distribution and occupies a very small fraction (<2%) of the historical range.** While there are other areas with similar habitat, none appeared large or contiguous enough to support an independent population. Therefore, we modeled the occupied dune areas as suitable habitat but did not otherwise attempt to incorporate other patches of dune forest.

6. **While shrub cover appears to be an important component of all habitat types occupied by coastal marten, we did not incorporate it explicitly into the model.** Shrub cover was not a strong predictor of marten occupancy in the Slauson *et al.* (2019b) model. Furthermore, we expected that maturity of shrub cover and forest conditions would largely covary, with mature forest in most ecotypes within the coastal marten's historical range also having a well-developed ericaceous shrub layer or the capacity to develop one in the right conditions (Mayer 1988, Raphael 1988). We also were unable to locate a landscape-scale shrub data layer that we felt was suitable for modeling coastal marten habitat needs.

7. **Major roads and rivers are significant but not impermeable barriers to coastal marten dispersal.** Based on records of marked coastal martens dispersing across the lower Klamath River and U.S. Hwy. 101, we did not model resistance of any of these linear features as being so high that least-cost corridors would not be able to cross them. However, major rivers and roads have among the highest resistance values we gave to any landscape feature.

These assumptions guided the acquisition and parameterization of the spatial data layers we used for the model, as described below.

Data Layers

The connectivity model incorporates four spatial data layers: (1) a habitat layer based on a model of old-growth structure index (OGSI) for forests in the region and USGS GAP groundcover data for non-forested habitats; (2) a serpentine soils layer; (3) a rivers layer; and (4) a roads layer.

Habitat

OGSI is the primary estimator of habitat quality and cost-weighted distance in the connectivity model. It is a parameter derived by the Gradient Nearest Neighbor (GNN) model produced by the Landscape Ecology, Modeling, Mapping & Analysis laboratory in Corvallis, OR (LEMMA 2014a). The GNN model provides fine-scale spatially explicit data on forest structure across a vast area of California, Oregon, and Washington, and is one of the very few datasets available that provides such habitat information in a consistent manner across the CA/OR state border. GNN summarizes detailed data from thousands of forest survey points. It then uses a multi-step process to interpolate them to the unsurveyed areas on the landscape based on several explanatory datasets such as Landsat remote sensing imagery, elevation, climate, and geology (more information about this process can be found at https://lemma.forestry.oregonstate.edu/methods/methods, see also Ohmann & Gregory 2002). Like Landsat imagery, GNN has a spatial grain of 30mX30m (900 m^2). OGSI is used to characterize the suitability of forest habitat conditions for old-growth obligate species and processes. It is scaled to specific regions and ecotypes, and is derived from a conceptual model that incorporates: (1) the density of large trees (2) and snags, (3) the size class diversity of live trees, and (4) the amount of down woody material (Davis *et al.* 2015). These seem well aligned with critically important features in forests inhabited by Pacific martens generally and Humboldt martens specifically, and a range of literature describes the use of habitat types that are consistent with the presence of these features (for examples, see Ruggiero *et al.* 1998, Hargis *et al.* 1999, Slauson 2003, Bull *et al.* 2005, Kirk and Zielinski 2009, Wasserman *et al.* 2012, Zielinski *et al.* 2015, Moriarty *et al.* 2016, Tweedy 2018, Delheimer *et al.* 2019, Gamblin 2019, Spencer *et al.* 2019, and Tweedy *et al.* 2019; additional review of numerous references describing favorable habitat conditions as well as potentially higher prey availability and lower predation risk in late seral forests can be found in Slauson *et al.* (2019a, PP. 36, 50-51) and USFWS (2018, PP. 24-33)). Spencer *et al.* (2019) found OGSI at a 5km scale to be one of the most important variables in their habitat model for the Sierra subspecies of the Pacific marten (*M.c. sierrae*).

Non-forested landcover types (those with <10% tree cover) are not classified by the GNN model. These portions of the landscape are filled in with data from the U.S. Geological Survey's (USGS) Gap Analysis Program (GAP) data on Ecological System Life Form (ESLF) (USGS 2011, Comer *et al.* 2003). We would generally expect these areas to have higher resistance to movement by coastal martens than forested habitats, but it is important not to treat them uniformly as some cover types may be passable in certain conditions (e.g. relatively small areas of native shrub-dominated habitats), whereas others are probably complete barriers (e.g. high density urban development). We have identified 43 such non-forested cover types within the historic range of the coastal marten. Based on their description in the NatureServe web application (NatureServe 2017), we grouped these into six functional classes that we generally treated as increasing in resistance from first to last: shrub, grassland, dune, open, wet, and developed (see Appendix 1).

While the GNN model output does a good job of describing patterns in forest structure across large landscape scales, it performs only "moderately well to poorly" for predicting conditions at specific sites (Ohmann and Gregory 2002); thus, the smaller an area the GNN model is applied to, the less accurate it will likely be at assessing conditions on the ground. The scale at which we are modeling connectivity (i.e. the historic range of the coastal marten) makes GNN an appropriate dataset for assessing habitat conditions, however, applying the results at very fine scales should be done with caution and we recommend obtaining supplemental information on habitat conditions to further support such applications. Furthermore, the most recent GNN model available to us was based on data collected in 2012 so there will be some localized inaccuracies from disturbance events that occurred more recently (timber harvests, fires, etc.)

Serpentine soils

In addition to late seral forests, coastal martens have at times been found utilizing shrub-dominated habitats on serpentine soils in areas influenced by coastal fog (within 30 km of the coast) (Slauson 2003, Slauson *et al.* 2019a). These sites often have sparse tree cover (Jimerson *et al.* 1995), but support a dense layer of the ericaceous shrubs preferred by the species. Many of these areas would be mapped as having relatively low OGSI values in the GNN model and thus as having lower habitat value and connectivity potential for coastal martens in an OGSI-based model. While these habitats make up a relatively small percentage of the overall landscape, some occur on ridgetops that could potentially act as movement pathways for martens, and including these soils may help define habitat cores more realistically. While there is only a limited amount of data available on coastal marten use of serpentine habitats, several lines of evidence in the led us to treat these areas as sub-optimal but usable habitat in our model. Home range sizes appear to be larger in areas with a higher proportion of serpentine habitat based on the larger scale at which it influenced occupancy in Slauson *et al.*'s (2019a, b) habitat model relative to late seral forest. Furthermore, Slauson *et al.* (2009b) reported that populations appeared to be male-biased and had more transitory occupancy than in habitat dominated by late successional forest.

We obtained a dataset of serpentine soils in the Klamath-Siskiyou Ecoregion that was assembled by Noss *et al.* (1999) for an earlier conservation planning effort by merging data from USGS geology maps of the region and STATSGO soils data from the U.S. Natural Resource Conservation Service. We clipped these serpentine soil polygons to a distance of 30 km from the coast to obtain a map of serpentine polygons that could potentially support coastal marten habitat. We then used the ESLF categories to identify and remove habitat types that did not seem likely to support the type of shrub-dominated habitats the martens use in serpentine sites based on their lack of suitable ericaceous shrub species in the community descriptions (NatureServe 2017). The remaining 32 ESLF categories (see Appendix 1) were included in the final modified serpentine soils polygons.

Rivers

The rivers resistance layer was developed from the USGS' National Hydrography Dataset (NHD) (USGS 2013), which maps the surface waters of the United States. The NHD within our modeling extent included a derived attribute called Strahler Stream Order (Strahler 1952)[**], which is a hierarchical classification of streams moving from the headwaters to the ocean. Using these stream orders allows us to identify and apply relative weights to the major rivers and their corresponding reaches across the landscape (e.g. Klamath, Rogue, Coquille, and Umpqua Rivers as 9th order streams, and the Eel, Smith, Siuslaw, and Alsea Rivers as 8th order streams). Based on marten dispersal observations and our estimation of how challenging these rivers and creeks may be for

[**] For a succinct definition and explanatory figure of Strahler stream order, see: https://usgs-mrs.cr.usgs.gov/NHDHelp/WebHelp/NHD_Help/Introduction_to_the_NHD/Feature_Attribution/Stream_Order.htm.

martens to cross, we created a resistance layer incorporating all stream lines between stream order 4 (e.g. Redwood Creek) and 10 (the Columbia River), with increasing resistance as the order increases.

Roads

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). Roadkill is a known source of mortality for coastal martens (USFWS 2018), especially affecting the Central Coastal Oregon population along US Hwy. 101. Furthermore, studies in populations of American marten (*Martes americana*) have found that roads can reduce habitat quality, as evinced by reduced activity near logging roads in Ontario (Robitaille and Aubry 2000) and reduced detection rates with increasing road density in Idaho (Wasserman *et al.* 2012). Alexander and Waters (2000) found that a high-traffic multi-lane highway (in this case the Trans-Canada Highway) is a significant barrier to movement of American martens, although Clevenger *et al.* (2001) showed that they can use certain types of culverts to pass under such roads.

To build the roads resistance layer, we used the OpenStreetMap dataset (OpenStreetMap contributors 2018), which is a global, collaborative project to create a free, editable map of the world's roads. We opted to use this dataset for two key reasons: (1) to gain the benefits of having single unified dataset rather than attempting to merge data from the California and Oregon state transportation departments and account for inconsistencies between their classification systems; (2) U.S. Federal Highway Administration (FHA) data only included major highways, and usually failed to account for variation within these routes. For example, US Hwy. 101 varies significantly in width, number of lanes, and the speed and intensity of traffic throughout this landscape. This variation is captured very well in OpenStreetMap, while the highway is assigned a single classification for its entire length in the FHA data. OpenStreetMap contains 33 different types of roadway in the modeled landscape that are classified based on type, size, usage, etc. (https://wiki.openstreetmap.org/wiki/Map_Features#Highway). We selected 14 of these that seemed likely to have the potential to impact coastal marten habitat and movements, modeling them as having higher resistance with increasing width and the amount and speed of vehicle traffic, as implied in the classification descriptions.

Mapping Landscape Resistance to Movement

Land cover

Once the spatial data layers were assembled, we used them to create a single "resistance surface" covering the entire modeled landscape (Zeller *et al.* 2012). We assigned numerical resistance values to each distinct land cover type in the five spatial data layers based on our assessment of how easily and safely coastal martens could pass through them, which in turn were derived from the key assumptions we used in constructing the overall model. Because we lacked empirical data on how likely coastal martens are to move through various habitat types and other features on the landscape (especially while dispersing), we had to extrapolate from what is known about the species' home range habitat selection and dispersal patterns. This required a certain amount of estimation, and the exact resistance values and resulting cost-weighted distance estimates should be viewed with caution. However, we are reasonably confident in the general patterns that we modeled for the resistance values and generally strove to keep them relatively simple in the absence of data on coastal marten movement pathways demonstrating more complex relationships. Such extrapolation from incomplete data is a common practice in constructing resistance surfaces for habitat connectivity models needed to inform urgent conservation decisions (Aune *et al.* 2011, Zeller *et al.* 2012, Cushman *et al.* 2013).

In deciding how best to assign numerical resistance values to our estimates of the relative difficulty of martens moving through different land cover types, we followed guidance from the authors of Linkage Mapper in setting the resistance of the idealized dispersal habitat (in this case, late-seral forests with high OGSI) equal to 1 per pixel (McRae *et al.* 2015). This means that in calculating least-cost path (LCP) lengths, the cost of crossing best quality dispersal habitat would only result from the distance involved (one 30m pixel = 1 cost-weighted distance unit), with less desirable habitat being assigned higher resistance per pixel because of the presumed greater difficulty or risk for martens to move through it. Conceptually, we followed the authors' guidance to estimate the resistance of a landscape feature as the number of additional cells of best quality habitat a coastal marten would travel through to avoid one pixel of the feature being considered (WHCWG 2010, McRae *et al.* 2015). For example, a 30m pixel of early seral forest assigned a resistance value of 10 would have the same modeled movement cost as would accrue from moving through 10 pixels (300m traveled distance) of best quality mature forest habitat (i.e. one pixel with a resistance of 10 has the same cost as 10 pixels with a resistance of 1).

In assigning these resistance values, we also kept in mind the published estimates of mean and maximum dispersal distances for martens. Several studies of North American marten species (i.e. not necessarily *M. caurina* or *M.c. humboldtensis*) have found that most dispersing juveniles traveled <15km from their natal territory to their eventual home range (Phillips 1994, Broquet *et al.* 2006, Pauli *et al.* 2012, Slauson 2017). Johnson *et al.* (2009) documented dispersals of American marten in Ontario of up to 181km in females and 214km in males, while Fecske and Jenks (2002) documented an adult male Pacific marten that dispersed over 80km. However, dispersals of such long distances seem to be rare, and Johnson *et al.* (2009) found that mortality risk increased with dispersal distance in the Ontario study. An idealized 15km marten dispersal travelling exclusively through the best quality habitat in our model would cross 500 pixels of 30m, each with a resistance value of 1. We kept this figure in mind in assigning resistance values as well, especially for the higher resistance cover types (such as for large rivers). A dispersal that passed through unfavorable habitat could cover 15km of cost-weighted distance in a much shorter amount of physically traveled distance. Because we would ultimately use the cost-weighted lengths of the LCPs connecting pairs of habitat cores as a means of assessing their relative connectedness for coastal martens, we generally tried to set higher values that would act as significant filters without necessarily creating impassable barriers.

The OGSI layer determined the resistance value for the vast majority of 30m pixels (86.8% within the coastal marten's historical range) (Fig. 4B). We assumed that coastal martens would move more easily through areas of high OGSI than areas of low OGSI, based on Slauson et al.'s (2019a, b) habitat model. The risk of mortality to dispersing martens is also likely to be lower in areas of the landscape where there are higher concentrations of late seral forest with high OGSI because bobcats, their primary predator (USFWS 2018), select against mature forest in mixed conifer dominated areas (Wengert 2013). We further assumed that dispersing martens would be willing to move through habitat that might be considered sub-optimal for longer-term territorial occupancy (Wasserman *et al.* 2010, 2012, Moriarty *et al.* 2015). We assigned resistance values based on the likelihood of pixels with a given raw OGSI value being classified as OGSI 80 or OGSI 200 (Fig. 5), which are binary classifications within the GNN that indicate whether a given pixel is modeled as having similar characteristics to forest stands that are about 80 years old or 200 years old (Davis *et al.* 2015). These classifications represent the ages at which forests in the region usually begin to exhibit traits associated with old growth forests (OGSI 80) and at which they functionally act as old growth forest (OGSI 200). They are not directly linked to the raw OGSI value, and classification depends on the ecotype and local conditions (Davis *et al.* 2015). We created six "bins" of raw OGSI ranges and assigned resistance values to these based on inspection of the frequency of OGSI 80 and OGSI 200 pixels within them (Fig. 5, Table 2). We did this for simplicity's sake, and because we lacked empirical data on habitat use by dispersing martens (as opposed to those residing in a territory). The OGSI resistance bins reflected our assumptions that landscape permeability was not finely correlated with OGSI in the sense that resistance should measurably decrease across, for example, OGSI 35→36→37 (i.e. we opted not to model OGSI-based resistance as a continuous variable). We assumed that raw OGSI values likely to be classified as OGSI 80 or OGSI 200 would be easiest and safest to move through, while raw OGSI with few or no OGSI 80 pixels were assigned increasingly higher resistance values, with the highest resistance value related to a pixel's OGSI set at 10 (Table 2). Thus, in the absence of more precise data on marten dispersal behavior, we modeled pixels that were more likely to have been disturbed relatively recently to be more difficult or dangerous to traverse than pixels of increasing maturity.

In the areas with serpentine soils that potentially supported favorable habitat (as described above on P. 21), we halved the normal resistance values for pixels with OGSI of 0 – 30 (Table 2). These serpentine pixels represented about 1.2% of the coastal marten's historical range (Fig. 4C). Pixels with OGSI ≥31 retained their normal resistance value. This breakpoint corresponds with the significant drop in resistance values we assigned to OGSI pixels that are likely beginning to exhibit old growth characteristics (Fig. 5).

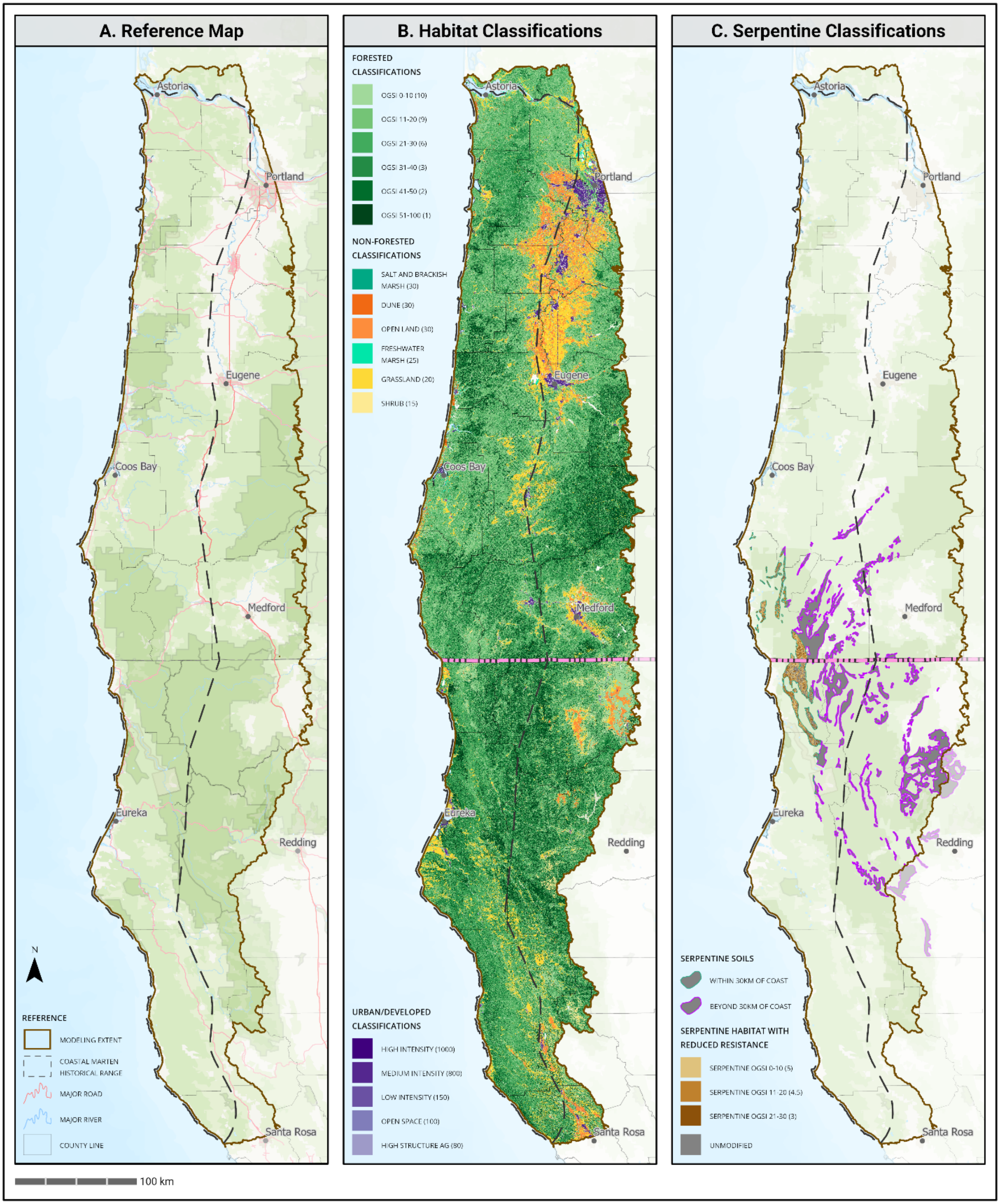


Figure 4, Part 1. The habitat and serpentine layers used to produce the resistance surface, along with a reference map. More detailed maps of the resistance surface components are shown in Appendix 2.

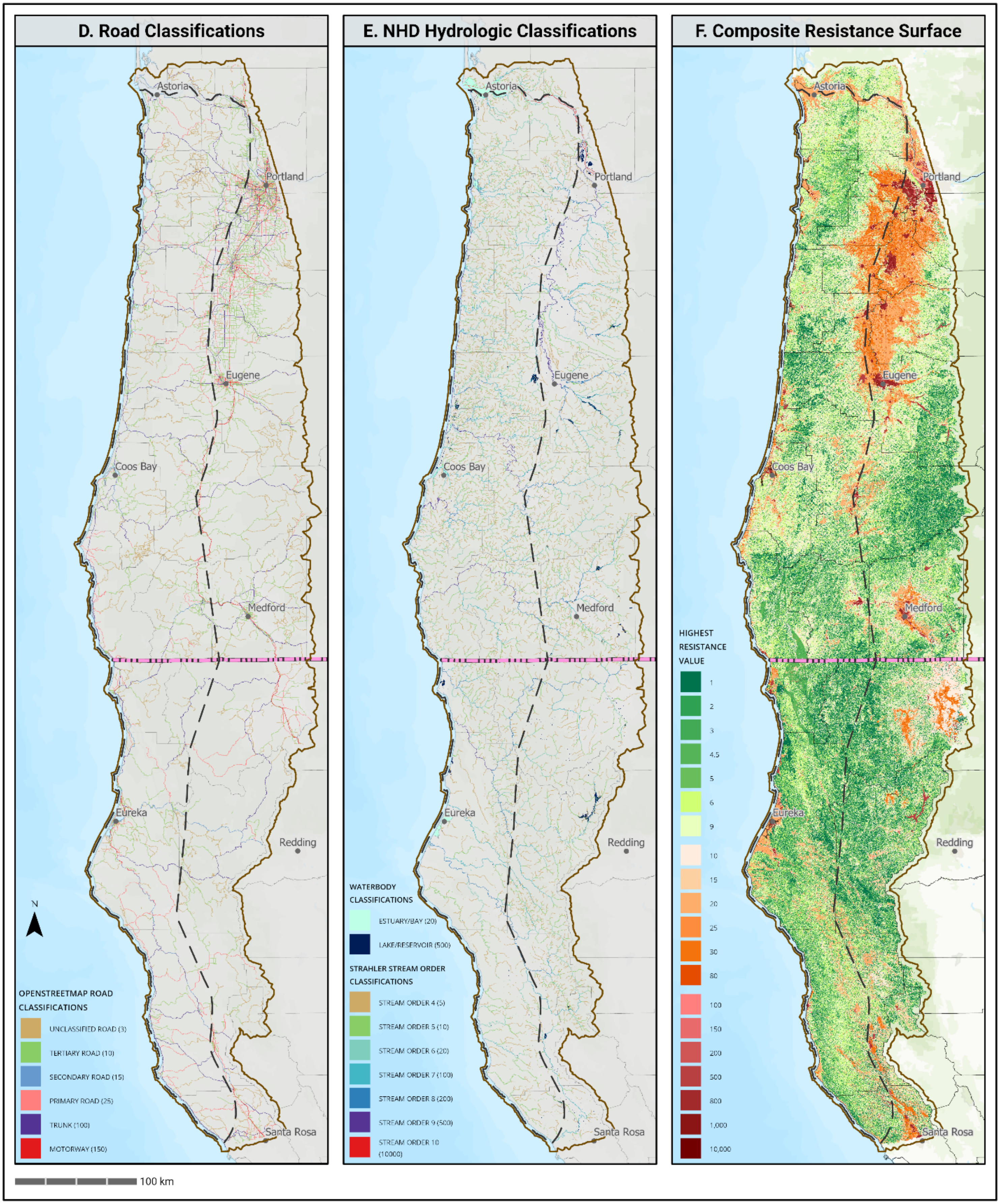


Figure 4, Part 2. The road and river layers used to produce the resistance surface, and the final composite resistance surface used in the connectivity model. This resistance surface was assembled by assigning the highest resistance value from the five base raster layers to each 30m X 30m pixel. More detailed maps of the resistance surface components are shown in Appendix 2.

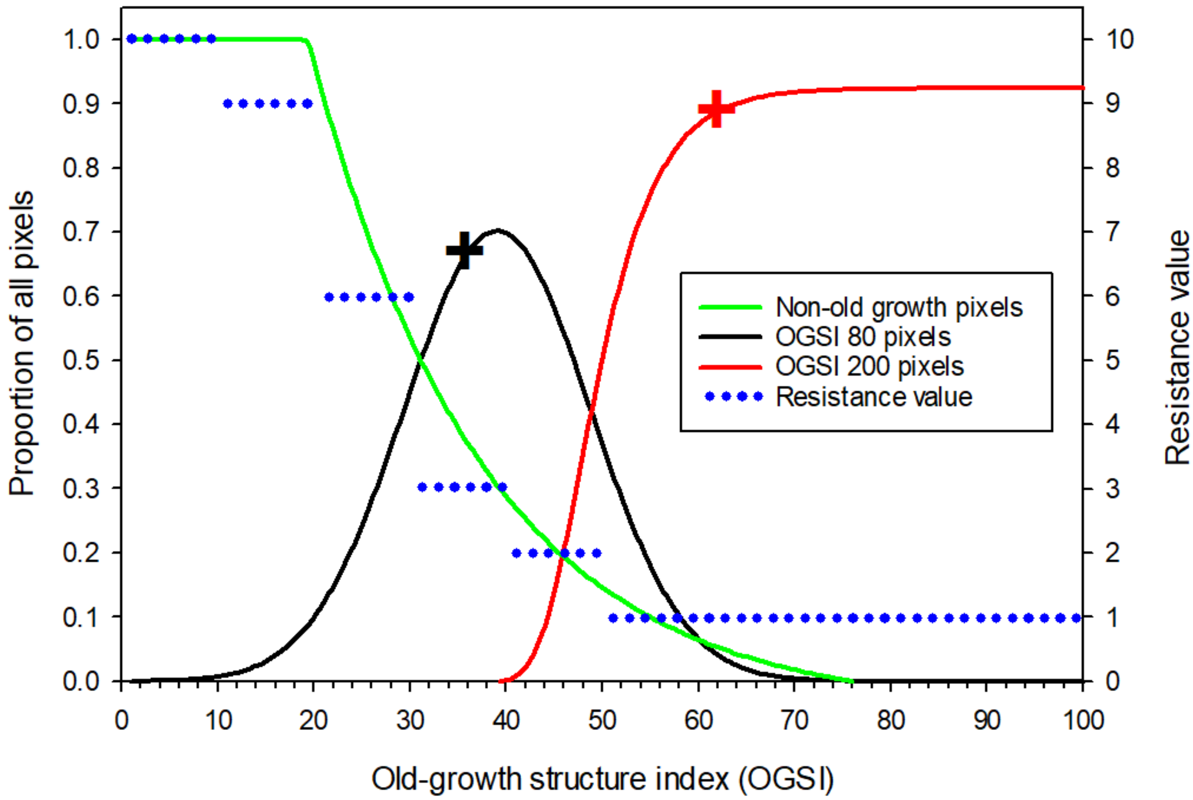


Figure 5. Relationship between pixels within the coastal marten's historical range classified as old growth forest and resistance value "bins" of OGSI. Solid curves from nonlinear regressions illustrate the frequency by raw OGSI value of 30m pixels classified by the GNN model as OGSI 80 (i.e. similar to forest stands that are beginning to exhibit typical old growth characteristics), OGSI 200 (i.e. similar to forest stands classified as mature old growth), and forested pixels not classified as old growth. Plus signs show the median value for pixels classified as OGSI 80 (=36) and OGSI 200 (=62) in this landscape. The blue dotted lines show the OGSI value "bins" we used for assigning per-pixel resistance values in the connectivity model, with lower resistance indicating pixels were modeled as easier for coastal martens to move through.

Pixels with <10% tree cover at not classified by the GNN model because they are not considered "forested" (LEMMA 2014a) and therefore have no OGSI (note that forested areas with low tree cover resulting from recent disturbance (e.g. from timber harvest or fire) were classified by the GNN and assigned a very low OGSI). Non-forested pixels were classified with GAP ESLF data (see above), and we assumed that none of them constituted coastal marten habitat. We also examined descriptions of the 43 non-forested land cover classifications (NatureServe 2017) that we identified within the historical range of the coastal marten to confirm that none seemed to have characteristics that might make them suitable for use by the species based on what is known of its habitat needs (e.g. presence of a dense ericaceous shrub layer (Slauson and Zielinski 2007, Slauson *et al.* 2007)). While we assigned all of these land cover types higher resistance values than any forested land cover (i.e. any pixels assigned an OGSI value in the GNN), we did not treat them uniformly. We assumed that some land cover types would be reasonably passable to dispersing coastal martens in some conditions (e.g. relatively small patches of shrub-dominated chaparral), whereas others are likely very strong barriers to marten movement (e.g. developed areas). To simplify the task of assigning resistance values to so many distinct non-forested habitat types that mostly comprised very small proportions of the landscape, we used their descriptions (NatureServe 2017) to group them into six functional classes that generally increase in resistance from first to last: shrub-dominated, grassland, dune, open/barren, wetland, and developed (Fig. 4B). Habitat types within most functional classes generally shared the same resistance value (Table 2). Some exceptions were made if a particular habitat type seemed different enough from others in the functional group that it warranted a unique resistance value (e.g. we assigned North American Alpine Ice Field a higher resistance score than other habitat types in the "Open/Barren" group). The cover types in the "Developed" functional group were all assigned different resistance values (Table 2). For a complete list of non-forested habitats and their assigned resistance values, see Appendix 1.

Linear features

Rivers and roads were represented as linear features a single pixel 30m wide. Because many of these features were modeled as representing significant barriers to movement by martens, we took care to ensure that they were mapped without breaks that would encourage LCPs to pass through them. In many instances, roads and rivers are in fact wider than 30m, and in these cases the pixels were classified by the relevant surrounding land cover type (OGSI or ESLF).

While coastal martens probably make regular crossings of smaller streams, wider rivers with high flow volumes and swift currents probably act as significant barriers to movement. However, the detection of an ear-tagged coastal marten in Prairie Creek Redwoods State Park in 2017 that had dispersed from the Northern Coastal California population across the Klamath River to the east (CA Dept. of Parks & Recreation *unpublished data*) indicates that even the largest rivers in this region are not impermeable barriers. It is somewhat difficult to set a single resistance value for rivers because the flow volume varies considerably both within years (highest in spring, lowest in early autumn) and between years (i.e. lower in drought years). Natal dispersal in coastal martens can take place over much of the year (starting as early as August and continuing into the following summer) (Johnson 2008, USFWS 2018), meaning that the degree of difficulty in crossing large rivers could vary considerably between dispersal events. Delheimer *et al.* (2015) modeled potential LCPs of coastal martens hypothetically dispersing from 29 known female territories within the Northern Coastal California population to Redwood State and National Parks, and simulated variation in river volume with a sensitivity analysis that varied the resistance values of the Klamath and Smith Rivers by a factor of 500. While it may be of interest to include such a sensitivity analysis in a future version of our model, the current version incorporated the assumption that, on average, large rivers represent significant but not impassable barriers to dispersing coastal martens and we therefore attempted to set a "median" resistance value for them. We based resistance values for the stream lines in the NHD with Strahler Stream Order 4 - 10 (see above) on our estimation of how challenging these barriers would be for coastal martens to cross (Fig. 4E, Table 2). At the lower end, the 4th order streams would act as only a slight barrier in otherwise high quality habitat (i.e. the assigned resistance value is lower than that for pixels with very low OGSI). At the higher end, 10th order streams were intended to be essentially impassable, although the only such river in this landscape is the lower Columbia, which represents the northern edge of the coastal marten's historical range (Fig. 1).

In areas where larger rivers entered estuaries near the ocean, we created a bay/estuary data layer and assigned each pixel a resistance value of 20. This layer was created because the GNN, our primary land cover dataset, terminated at the coastline and explicitly excluded estuaries as null values. As many estuaries (especially in Oregon) are narrow, sinuous, and traverse inland for substantial distances, it was necessary to fill in these null values with supplemental resistance values that would need to be crossed in addition to the resistance of our single pixel river stream lines. The cumulative resistance of the river and the estuary pixels represented significant barriers to modeled movement by martens.

We examined descriptions of the 33 roadway classifications in OpenStreetMap that occurred in our modeled landscape. Because of the origins of this global dataset, most of these road types have been given names more typically used in Great Britain than in the USA (e.g. "Motorway" instead of "Freeway"). However, good descriptions of all classifications were available in supporting documentation online (https://wiki.openstreetmap.org/wiki/Map_Features#Highway), allowing us to assess which categories were likely to impact coastal marten habitat and movements based on road width and the amount and speed of vehicle traffic implied in the classifications (we lacked data on actual traffic levels or speed limits). We assumed that freeways and major divided highways would act as strong "psychological" deterrents to martens crossing them because of the noise and disturbance from traffic and the likelihood of having to cross a wide area with little or no cover (Forman and Alexander 1998, Alexander and Waters 2000). These roads also probably pose the greatest risk of mortality from being struck by a vehicle; we assigned them a resistance value of 150. We then assigned decreasing resistance values through smaller highways and roads ("Trunk", "Primary", "Secondary", and "Tertiary" roads and their associated "link roads", which are generally very short offshoots connecting to other roads) (Fig. 4D, Table 2). Many of the smallest roads (such as residential roads and logging roads) overlapped with data represented in the OGSI or ESLF layers, which could have led to "double counting" the resistance of these features and modeling them as more significant barriers than we intended them to be. Therefore, we ended up not assigning resistance values to anything smaller than "Unclassified" roads, which are usually two-lane roads connecting to small rural communities. Ultimately, 11 road classifications were assigned resistance values >0 (see Appendix 1 for full list).

All unique resistance values along with the general categories that they represent are listed in Table 2. The complete list of resistance values for each specific land cover type can be found in Appendix 1.

Table 2. Per pixel resistance value categories for land cover types. Higher resistance values reflect estimates of greater difficulty for coastal martens to move through the cover types. For a complete list of all resistance values, see Appendix 1.

Cover class	Resistance value
OGSI 51 - 100	1
OGSI 41 - 50	2
OGSI 31 - 40; Serpentine w. OGSI 21 - 30; Unclassified roads††	3
Serpentine w. OGSI 11 - 20	4.5
Serpentine w. OGSI 0 - 10; Stream order 4	5
OGSI 21 - 30	6
OGSI 11 - 20	9
OGSI 0 - 10; Stream order 5; Tertiary roads	10
Shrub-dominated ESLF habitats; Secondary roads	15
Grass-dominated ESLF habitats; Estuary‡‡; Stream order 6	20
Most ESLF wetland habitats; Primary roads	25

†† These roads, which are mostly rural with relatively low traffic levels, were assigned a low resistance value so that they would influence connectivity in areas with relatively undisturbed habitat but not unduly increase resistance in more developed places or areas dominated by low-OGSI pixels.
‡‡ Per pixel, supplementing the resistance of the river stream line.

Cover class	Resistance value
Most ESLF open/barren and dune habitats; Tidal salt marshes	30
Developed, high structure agriculture (e.g. orchards, vineyards)	80
Developed, open space; Ice fields; Stream order 7; Trunk roads	100
Developed, low intensity; Motorway	150
Stream order 8	200
Stream order 9; Lake, pond, or reservoir	500
Developed, medium intensity	800
Developed, high intensity	1000
Stream order 10 (Columbia River)	10000

Final resistance surface

We created a raster file for the resistance surface covering the entire coastal marten historical range as well as the buffer to east (Fig. 4F) using the Resistance and Habitat Calculator component of Gnarly Landscape Utilities (McRae *et al*. 2015), which is a set of spatial analysis tools associated with Linkage Mapper. This required creating separate raster files for each of the spatial data layers (OGSI, serpentine, ESLF, NHD rivers and water bodies, bay/estuary, and roads). We then used the tool to compare the five layers and assign the highest resistance value possible for a given pixel to the final combined resistance surface raster. For example, if a given pixel had an OGSI of 5.0 (resistance value = 10) but was also included in the roads raster as including a "primary road" (resistance value = 25), that pixel would receive the higher resistance value (25) in the final resistance surface. See Appendix 2 for finer scale maps of the final resistance surface and the layers used to create it. See Appendix 3 for discussion of sensitivity analyses we performed related to various aspects of developing the resistance surface.

Mapping Habitat Cores

We derived the habitat cores using another component of Gnarly Landscape Utilities called Core Mapper (Shirk and McRae 2015). We assigned each 30m pixel on the modeled landscape a habitat value equal to its GNN OGSI (range = 0-100) (Fig. 6). We then used a moving window to calculate the average habitat value within a 977m radius around each pixel (derived from the estimated average size of a female marten's home range of 300 ha given in Slauson *et al*. 2019a based on studies of Pacific martens in the Sierras and a small number of coastal marten territories examined) (Fig. 7A). Pixels with an average habitat value ≥36.0 were then incorporated into habitat cores (Fig. 7B). After conducting a sensitivity analysis by running a set of Core Mapper trials using a broad range of habitat values, we chose ≥36.0 as the average habitat value because it is the median OGSI of pixels within the marten's historical range classified by the GNN as "OGSI 80" (Davis *et al*. 2015) (Fig. 5). It generated a set of habitat cores that were not overly generous (depicting most of the landscape as habitat core) or strict (only mapping cores in a few locations with very high OGSI such as Redwood State and National Parks) (see Appendix 3 for more details, including example maps from our sensitivity analysis). In areas with serpentine soils that support habitat potentially suitable for coastal marten (as described above), we assigned a minimum habitat value of 31, which is equivalent to the 33rd percentile of OGSI 80 pixels in the marten's historical range. Pixels with higher OGSI retained their normal habitat value. Our intention was to allow the modified serpentine pixels to be more easily incorporated into habitat cores if there were higher value OGSI pixels in the vicinity, but not to have them form the entire basis of a core. We also excluded pixels with a habitat value <1.0 from inclusion in habitat cores. We then set Core Mapper to expand the habitat cores by 977 cost-weighted meters, a step intended to consolidate smaller cores that were probably relatively close together from a marten's perspective. This was followed by a "trimming" step that removed pixels from the expansion that did not meet the moving window average so the net result was rather small changes in the size of the habitat cores, but filling in many individual isolated pixels with a habitat value of 0.

Once the initial set of pixels that met the minimum habitat value for cores was identified, we removed core areas that would likely be too small to support enough martens to viably sustain long-term occupancy (Fig. 7C). We set a minimum habitat core size of 1500ha, which was identified in the "Humboldt Marten Conservation Strategy" (Slauson *et al.* 2019a) as the amount of suitable habitat needed for a "Population Reestablishment Area" (equivalent to approximately five female home ranges). As was discussed above (see P. 13), there is some uncertainty about the exact boundary of the species' historical range (Zielinski *et al.* 2001, Moriarty *et al.* 2018a); therefore, we included all cores located within or intersecting the boundary of the historical range in our final model.

We made two significant "manual" modifications to the habitat core set. First, we added two habitat cores in the area occupied by the Central Coastal Oregon population, which is found in a unique dune forest setting that mostly has low OGSI values. We based the locations of these cores on marten detection data and the distribution of forested areas. They were bounded on the west side by the Pacific Ocean and on the east side by US Highway 101, and totaled nearly 15,065ha, which is about 0.26% of the historical range. Second, the initial set of habitat cores produced by the model included a very large and sprawling core that crossed the Klamath and Salmon Rivers with narrow "bridges" of just a few pixels at several spots. We broke this large core into four smaller cores (although three of these are still well above average in size) at these river crossing points to force Linkage Mapper to locate more least-cost corridors in the area. We also deleted some small patches associated with these habitat cores that were identified by Core Mapper as having a sufficient moving window average habitat value, but were "stranded" across a major river from other core habitat. This modification eliminated a potential bias for LCPs to be directed through spots where these habitat cores "leaked" across major river barriers. These patches were all <300ha, the average size of a female coastal marten home range.

It is important to note that the habitat cores do not represent all suitable coastal marten habitat on the landscape. Indeed, a number of recent marten detections fall outside of these core areas (Slauson *et al.* 2019a, Moriarty *et al.* 2019, M. Gunther *unpublished data*). Rather, the habitat cores are meant to map relatively large patches that are likely to contain sufficient high quality habitat to support long-term occupancy by coastal martens, and thus likely represent the most important areas for the species for conservation purposes. It is also worth specifying that not all of the habitat within the cores is highly suitable for coastal martens; rather, they are comprised of pixels that met the minimum habitat value threshold within the surrounding 977m radius moving window. Many areas that could have been classified as habitat cores at a smaller minimum size than five female home ranges were incorporated into least-cost corridors by Linkage Mapper. See Appendix 3 for discussion of sensitivity analyses we performed related to various aspects of developing the habitat cores.

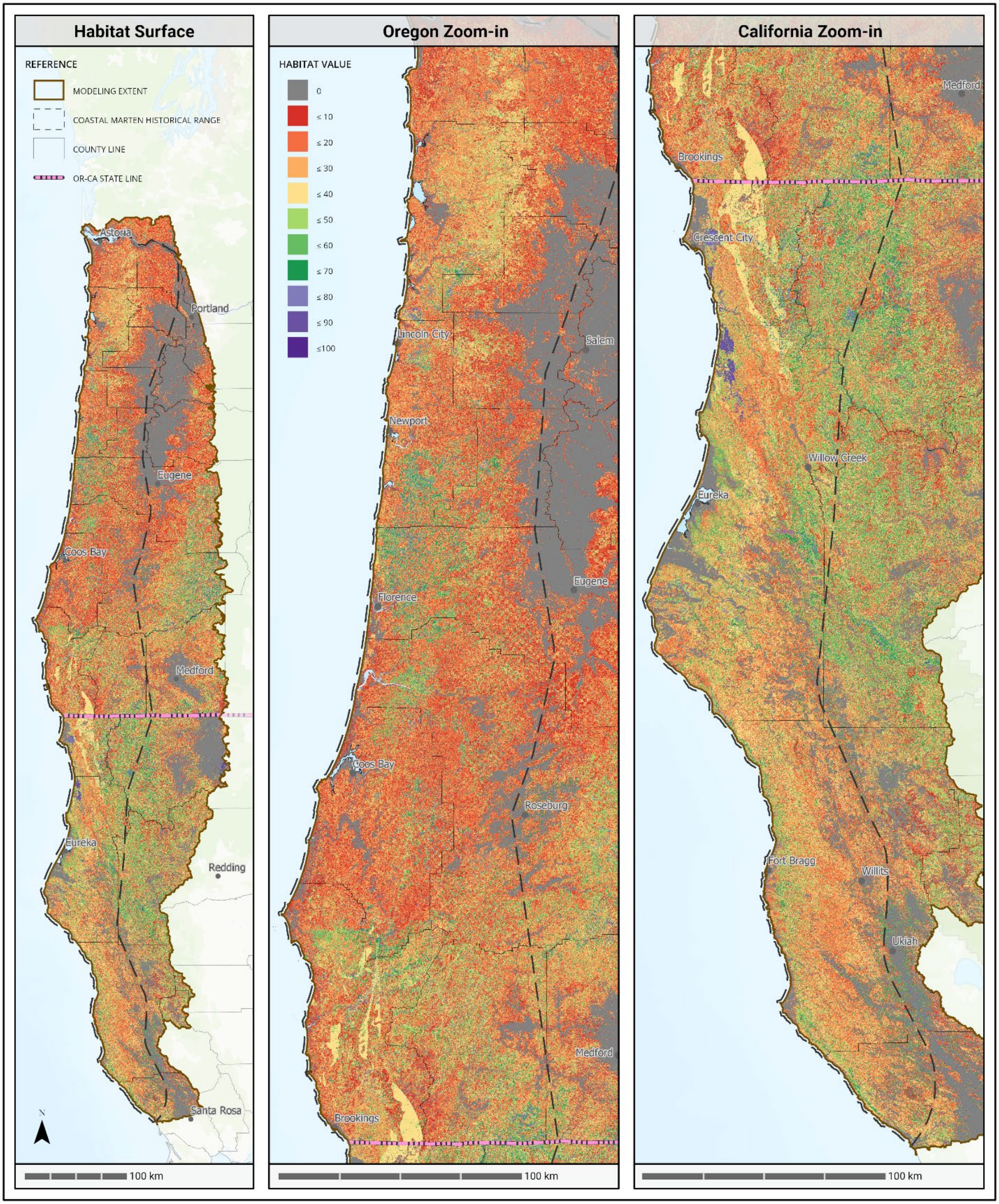


Figure 6. The habitat value raster surface used to delineate the habitat cores for the coastal marten connectivity model.

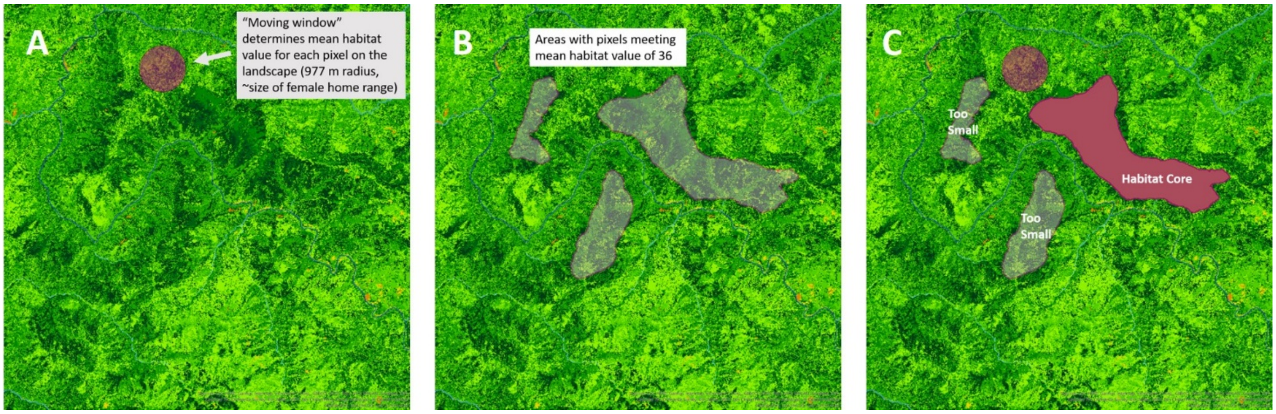


Figure 7. Illustration of the process used to delineate habitat cores. Each 30m pixel on the landscape was assigned a habitat value from 0-100 based on its old-growth structure index (OGSI). Darker pixels in this example have higher OGSI scores. The mean habitat value was calculated for each pixel in turn within a circle around it with a radius of 977m that moves across the landscape (referred to as a "moving window") (**A**). If the mean habitat value of all pixels within the circle was ≥36.0, the central pixel was incorporated into a habitat core (**B**). The set of habitat cores was then refined by removing those <1500ha (**C**), which is approximately the size of five female coastal marten home ranges (the moving window is the mean size of one female home range, and included in this panel for comparison).

Running Linkage Mapper

Mapping Least-cost Paths and Corridors

Once we developed our coastal marten landscape resistance model and mapped the habitat cores, we used Linkage Mapper to identify LCPs between cores and to map broader mosaicked corridors around these single-pixel width paths. Linkage Mapper proceeds through several steps to complete these tasks (McRae and Kavanagh 2016). First, it identifies and lists which habitat cores are nearest neighbors using both the Euclidean distance (the "as-the-crow-flies" straight-line distance between nearest points on the edges of a pair of cores regardless of the character of the intervening landscape) and the cost-weighted distance. Second, it creates a "stick map" using straight-line linkages to connect core area pairs that are candidates for corridor mapping. Third, it locates the LCPs through the resistance surface between these pairs of cores and calculates their cost-weighted distance. This LCP is a single 30m pixel wide. A single pixel's cost-weighted distance is the cell's resistance value multiplied by the size of the cell, and the cost-weighted distance of an LCP is the sum of the cost-weighted distances of the pixels it runs through. Thus, the cost-weighted distance for a linkage between two habitat cores could be summarized as:

$$CWD = \sum_{i=1}^{n_i} (r_i * 30)$$

Where CWD is the cost-weighted distance, *r* is the resistance of a given pixel *i*, *n* is the total number of pixels in the LCP, and 30 is the width of a pixel in meters. This allows CWD to be reported in units that are directly comparable to the LCP length and the Euclidean distance between habitat cores (i.e. normalized to meters or kilometers), and all three of these metrics are included in the Linkage Mapper output for each linked pair of cores. Finally, Linkage Mapper creates least-cost corridors, which are wider swathes surrounding the LCPs that have only slightly higher movement costs and are more biologically realistic for conservation planning (Theobald 2006, Rudnick *et al.* 2012). It does this by calculating for each pixel on the landscape how much more costly a pathway passing through it between two cores would be than the LCP. Pixels closest to the LCP tend to be relatively close to it in CWD value, with the potential contribution of pixels to connectivity tending to decrease further away from the LCP. Linkage Mapper then creates a composite linkage map by assigning each pixel its minimum value relative to the nearest LCP (WHCWG 2010). Thus, the final map is a mosaic of normalized least-cost corridors around the LCPs. These corridors will vary in width depending on the resistance values surrounding the LCP. The creation of mosaicked corridors is the most fundamentally important function of Linkage Mapper in providing an informative depiction of habitat connectivity on the landscape, and is a significant advance over the simple LCP estimation that is a basic function available in ArcMap.

The output of this least-cost corridor mosaic is a continuous surface that can be interpreted in various ways depending on how the relative values of pixels on the landscape are rendered in the final mapping process (WHCWG 2010). Setting the color ramp to depict only pixels that have relatively similar connectivity values to the LCPs will usually result in narrower, more strictly defined corridors (Fig. 8a). A broader setting of the color ramp will provide wider corridors (Fig. 8b and c), and may show alternative pathways that have a significantly higher CWD than the LCP, but may still provide useful habitat connectivity for the species of interest and give insights for conservation planning about potential redundancy in connections between habitat cores (Pinto and Keitt 2009, WHCWG 2010, Jones 2015). After some experimentation, we chose to depict corridors using a color ramp scaled from 0 – 4885 cost-weighted meters (i.e. the sum of the resistance values of the cells multiplied by their size to normalize them for comparison to meters). The specific value of the upper end of this scale was based on five times the radius of an average female coastal marten home range (977m X 5 = 4885m). Although this figure was somewhat arbitrary, it did provide a corridor map that we felt was not overly broad (showing vast areas of the landscape as being important for connectivity) or too strict (showing corridors as unrealistically narrow).

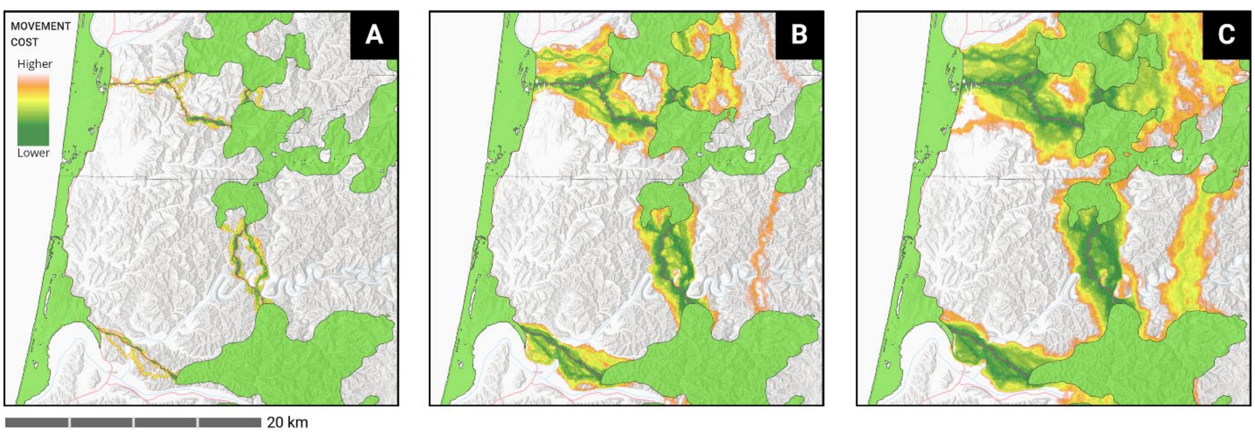


Figure 8. Examples of how least-cost corridor width can be varied based on how the Linkage Mapper mosaic output is visualized. In panel A, the corridors around the least cost paths (the purple lines) are rendered to 977 cost-weighted meters, the radius of one female marten home range. Those in panel B are rendered to 4885 cost-weighted meters (five home ranges), which is what we used for our model. Those in panel C are rendered to 9770 cost-weighted meters. Note the increasing proportion of the landscape that is identified as being part of a least-cost corridor as more liberal visualization values are applied, and that additional alternative corridors not directly adjacent to the least-cost paths are mapped as well.

Linkage Mapper includes several optional functions that we did not use for our model. First, we opted to drop LCPs between pairs of habitat cores that would pass through a third core. Second, we opted not to have Linkage Mapper group the habitat cores into disjunct "constellation" clusters that are then connected together by a single corridor. We felt this option could unnecessarily limit our picture of habitat connectivity for the coastal marten, when showing potential redundancy in the core and linkage network might ultimately increase the possibilities for conservation planning around the species. For the same reasons, we opted not to set a maximum number of linkages that could connect to any single habitat core. Finally, we opted not to set a maximum Euclidean or cost-weighted distance for least-cost corridors between habitat cores in our primary model. While we did have some information from the literature on mean and maximum dispersal distances (described and cited above), we felt there was enough uncertainty surrounding the dispersal abilities and potentially suitable habitat of the species that we decided to model all least-cost corridors between habitat cores regardless of length. Furthermore, least-cost corridors that may currently be longer than is desirable could present opportunities for future management aimed at improving connectivity. However, we present supplemental results below depicting connectivity using maximum dispersal values, and used a set of descriptive metrics produced as part of the Linkage Mapper output to assess the relative quality of all linkages (see Appendix 6).

We classified the relative connectedness of habitat core pairs by comparing the CWD units of the LCPs connecting them to the standard for Euclidean distances between populations used in the coastal marten SSA (USFWS 2018). We defined core pairs as "well connected" if they were connected by <15,000 cost-weighted meters (i.e. modeled equivalent movement difficulty of traveling 15km through best quality habitat), "moderately connected" if they were connected by 15,000 – 45,000 cost-weighted meters, and "poorly connected" if they were connected by >45,000 cost-weighted meters. Once again however, a good deal of estimation went into setting the exact resistance values for the various land cover types, so the exact categorization of connectedness of specific habitat core pairs should be treated with caution, especially in borderline cases. We mapped habitat cores into "functionally connected clusters" based on whether they met the standard of "well" or "moderately connected" as assessed using CWD. The cost surface produced by Linkage Mapper during the corridor building process was used to map outwards from each core to a distance of 22,500 cost-weighted meters, thus including all habitat cores within 45,000 cost-weighted meters in the same cluster.

We consider the output of the Linkage Mapper trial on the set of habitat cores we produced using Core Mapper to be our "primary model". In addition, we also produced three secondary models. One used the same resistance surface as the primary model, but used only the polygons delineating the boundaries of the four existing populations (USFWS 2018) (Fig. 1) as habitat cores. This allowed us to examine whether there were additional areas that may represent potentially significant corridors that were not identified in the primary model. The other two models were scenarios aimed at exploring approaches to conservation planning that used modified versions of the resistance or habitat value surfaces, and these are described below.

Scenarios as examples of conservation planning

One of our primary motivations for developing a habitat connectivity model for coastal marten was to help inform conservation planning decisions throughout their historical range. The quantitative output provided by Linkage Mapper describing LCP lengths and CWDs between habitat cores allows us to compare how alternative configurations of habitat on the landscape might affect connectivity for the coastal marten. By modeling potential management scenarios or disturbance events, we can get a better understanding of the potential implications of habitat restoration, timber harvest, wildfire risk, etc. To illustrate some potential ways in which the model might be used in a practical sense for decision support or conservation planning, we developed two scenarios, one describing timber harvest and one describing habitat restoration.

For the timber harvest scenario, we examined an area of industrial forest in northeastern Humboldt County, CA to the east of Prairie Creek Redwoods State Park. A visual comparison of satellite imagery using Google Earth Engine (Gorelick *et al.* 2017) indicated that a considerable amount of timber harvest had occurred in this area since the data used in the most recent GNN model were collected in 2012 (LEMMA 2014a), and therefore these disturbances were not included in the resistance surface of our modeled landscape. We explored how this might affect our assessment of habitat connectivity for coastal marten by creating a revised resistance surface incorporating data on post-2012 timber harvest. We used Google Earth Engine to locate areas where timber harvest had occurred in this part of the landscape between 2012 and 2016, which was the year of the most recent high resolution NAIP aerial imagery (USDA 2013) we had available at the time to digitize the harvest boundaries. We then created a spatial data layer of the perimeters of these harvest areas, and added these polygons to the original resistance surface classified as the lowest OGSI bin (0 – 10) with a resistance value of 10. We ran Linkage Mapper on this revised resistance surface with the same habitat cores used in our primary model, and compared the lengths and CWDs of new LCP and the original LCP to obtain numerical estimates of how much habitat connectivity might have been affected by the post-2012 timber harvest in this part of the landscape.

For the habitat restoration scenario, we examined a portion of the landscape in Curry and Josephine Counties, OR (mostly in the Rogue River – Siskiyou National Forest) where we identified a significant gap between two habitat core clusters in our primary model. We ran a Core Mapper trial using a minimum habitat core size of 300ha (one female marten home range) which revealed several smaller cores in this gap that likely had potential to act as the nucleus of a new core that could greatly enhance connectivity between the populations in Oregon and California. We modeled hypothetical habitat restoration scenarios by creating new versions of the habitat value raster file where we systematically increased the OGSI of every pixel by 1. We then re-ran Core Mapper on these raster files to see how much larger the habitat cores were with each increase in OGSI, and at what point a new core emerged that was above our 1500ha minimum size threshold. We then re-ran Linkage Mapper on this new landscape to quantify the new LCP lengths and CWDs. These restoration scenarios were not intended to mimic any specific habitat management actions beyond modeling plausible rates of forest growth. Rather, they were meant to illustrate ways in which we might use the model to estimate the effects of proposed restoration actions.

Results

Habitat Cores and Least-cost Corridors

General patterns

Using the input parameters selected, we identified 51 habitat cores located throughout the historical range of the coastal marten (Fig. 9), and 97 LCPs linking them (Fig. 10). Twenty-four of these cores were located entirely in Oregon, 26 occurred entirely in California, and one core (the largest) crossed the state boundary although most (85.3%) of it was located in California. These numbers include the two "hand drawn" dune forest habitat cores in Oregon (Core ID #s 10 and 15) and four cores in California that we manually split from a very large single core using major rivers as breaks (Core ID #s 25, 27, 30, and 34) (see Methods and Fig. 9 for details). Large fractions (>20%) of six habitat cores extended to the east of the boundary of the historical range of the coastal marten (Appendix 4). The total area of the habitat cores was 788,290.3ha (3043.6 miles^2). However, 232,385.1ha (29.5%) of this total occurred east of the historical range polygon. Most of the habitat core area (70.9%) occurred in California, although this included almost all (98.1%) of the area occurring east of the historical range polygon. Considering only the 555,905.2ha of habitat core within the historical range polygon, 330,814.6ha occurred in California (59.5%) and 225,090.6ha (40.5%) occurred in Oregon. The habitat cores ranged in size from 1624.8ha to 178,091.1ha, with an average size of 15,456.7ha (std. dev. = 30,478.7) and a median size of 4439.5ha. Over half (52.9%) of the cores were <5000ha in size (Table 3), while four very large cores (>50,000ha) accounted for over half (52.4%) of the total core habitat area. See Appendix 4 for summary statistics describing each habitat core.

Table 3. Size distribution of modeled coastal marten habitat cores.

Habitat core size (ha)	*n*
1500-3000	18
3000-4500	8
4500-6000	4
6000-7500	3
7500-10000	2
10000-20000	7
20000-30000	3
30000-40000	1
40000-50000	1
>50000	4

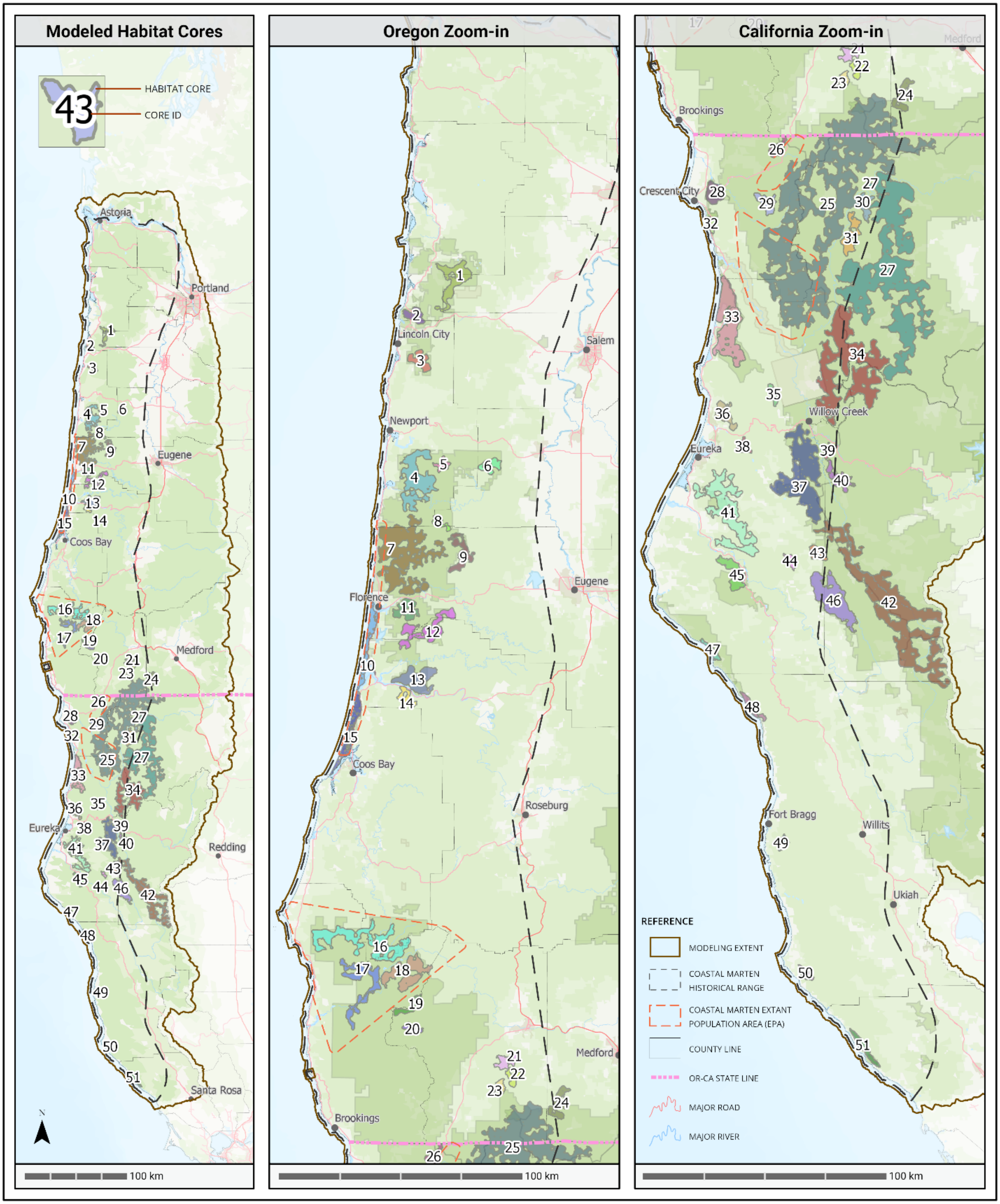


Figure 9. Coastal marten habitat cores identified using our model parameters. See Appendix 4 for summary data about individual habitat cores listed by Core ID number.

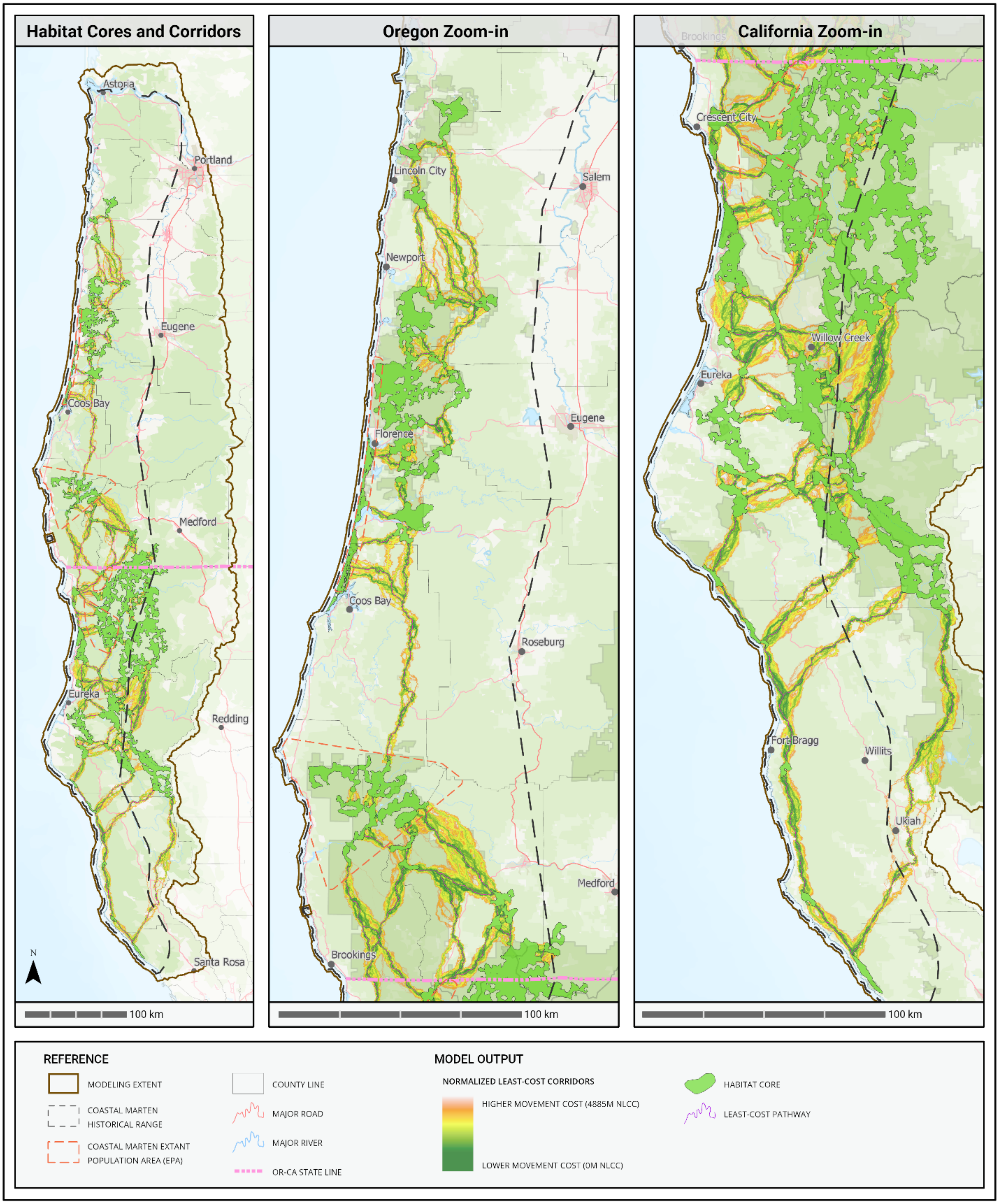


Figure 10. Modeled habitat cores and least-cost corridors for the coastal marten.

Most of the habitat cores occurred on federal lands (704,111.3ha, or 89.3%), primarily managed by the U.S. Forest Service (648,785.3ha, 82.3%), U.S. Bureau of Land Management (25,491.1ha, 3.2%), and the National Park Service (19,066.6ha, 2.4%). Smaller areas (3.0%) occurred on lands managed by the states of Oregon (4,622.9ha) and California (19,411.9ha), with over half of the latter (52.6%) occurring in the state-owned portions of Redwood State and National Parks. Tribal lands accounted for just 1,773.8ha (0.2%) of the habitat cores, mostly on lands of the Yurok (253.2ha) and Hoopa Valley (1497.2ha) Tribes. We estimated that 59,691.4ha (7.6%) occurred on private lands, mostly in California. Over a quarter of the habitat core acreage (228,530.5ha, 29.0%) occurred on lands primarily managed for biodiversity protection (GAP status 1 and 2). Additional supplemental maps illustrating the habitat cores and corridors through the lenses of various land management boundaries are provided in Appendix 5, including agency or entity responsible for management, land use allocations under the Northwest Forest Plan (Tuchmann *et al.* 1996), and USGS GAP status categories, as well as how they relate to serpentine habitats.

Euclidean distances between core pairs varied considerably (range 0.067[§§] – 146.328km), with the mean being 18.300km (st. dev. = 24.012) and the median being 10.033km. The LCPs ranged in length from 0.114 – 204.628km, with a mean of 25.130km (st. dev. = 34.053) and a median of 13.419km. CWD between habitat core pairs ranged between 0.115 and 383.395 cost-weighted kilometers, with a mean of 50.657 (st. dev. = 66.385) and a median of 27.301. Linkage Mapper also provides the ratio of CWD:LCP, which can be an informative metric of how resistant the landscape the LCP traverses is to movement. This ratio ranged between 1.0 and 47.1, with a mean of 2.7 (st. dev. = 4.8) and a median of 1.9. See Appendix 4 for summary statistics describing each habitat core pair linkage.

Whether habitat core pairs were classified as "well connected", "moderately connected", or "poorly connected" varied considerably depending on the metric used. Nearly twice as many core pair linkages would be considered "well connected" based on the Euclidean distance separating them compared to the CWD (Table 4, Fig. 11), while the number of linkages considered "poorly connected" was 2.6 times higher using CWD rather than Euclidean distance.

Table 4. Percentages of habitat core pair linkages classified as "well", "moderately", or "poorly connected" based on three distance metrics. Euclidean distance and least-cost path length are measured in km, while cost-weighted distance is measured in "cost-weighted units" normalized to km.

	Euclidean distance	Least-cost path length	Cost-weighted distance
Well connected (0-15km)	64.9%	54.7%	36.1%
Moderately connected (15-45km)	21.7%	27.8%	28.9%
Poorly connected (>45km)	13.4%	17.5%	35.0%

[§§] This is the shortest Euclidean distance produced by the Core Mapper output. Linkage ID #39 has a Euclidean distance of 30m, which was artificially created when we broke up the very large core (see Methods).

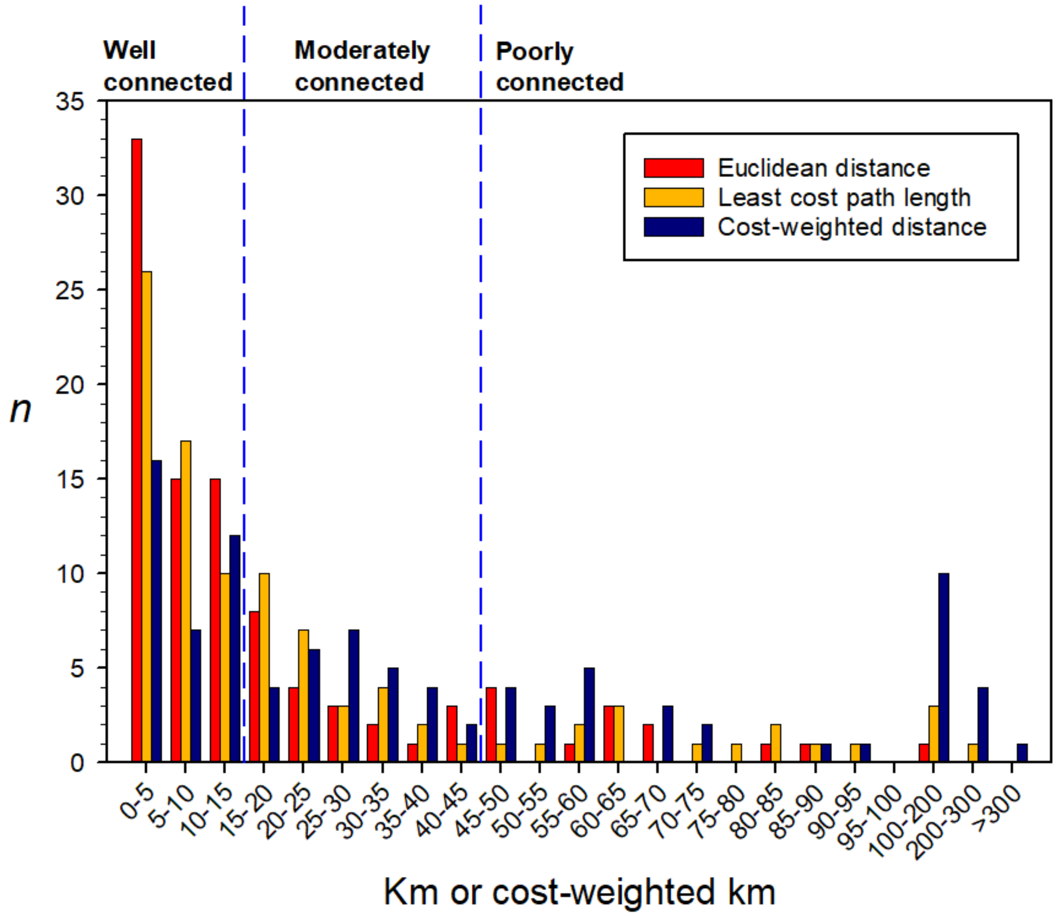


Figure 11. Distance equivalents of habitat core pair linkages assessed with three different metrics. Euclidean distance and least-cost path length are measured in km, while cost-weighted distance is measured in "cost-weighted units" normalized to km. Blue dashed lines show divisions between linkage quality categories based on the coastal marten's Species Status Assessment (USFWS 2018): "well connected" = 0-15km, "moderately connected" = 15-45km, and "poorly connected" = >45km.

Functional connectivity of existing populations and habitat cores

Based on all three of the metrics we used to assess connectivity of habitat cores, the two known populations of coastal marten in Oregon appear to be functionally isolated both from one another, and from the populations in California. Both are connected to the nearest existing population by least-cost corridors that are longer than 80km (Fig. 10), farther than the longest dispersal event recorded for a Pacific marten (Fecske and Jenks 2002, USFWS 2018). The LCP between the two Oregon populations is particularly long, at 125.9km (Link ID #27), while that connecting the Southern Coastal Oregon and California-Oregon Border populations is 82.5km (Link ID #31). The two California populations were both primarily associated with and connected by the same large habitat core (Core ID #25, Figs. 9 and 10), rather than a corridor between cores. This single core accounts for 30.8% of the core habitat within the historical range polygon. The Euclidean distance between the two California populations described in the SSA (USFWS 2018) is 11.2km, and the travel distance between them measured within their shared habitat core is about 30km. The secondary model that treated only the four existing population polygons as habitat cores mostly identified least-cost corridors in the same areas as the primary model (Fig. 12), although with two corridors significantly widening in spots. The main difference was the presence of a new least-cost corridor between the Oregon-California Border and Northern Coastal

California populations that passes through habitat core ID #29. It would be classified as "well connected" based on Euclidean distance and LCP length (11.2km and 14.3km, respectively) and "moderately connected" based on CWD (21.2 cost-weighted km).

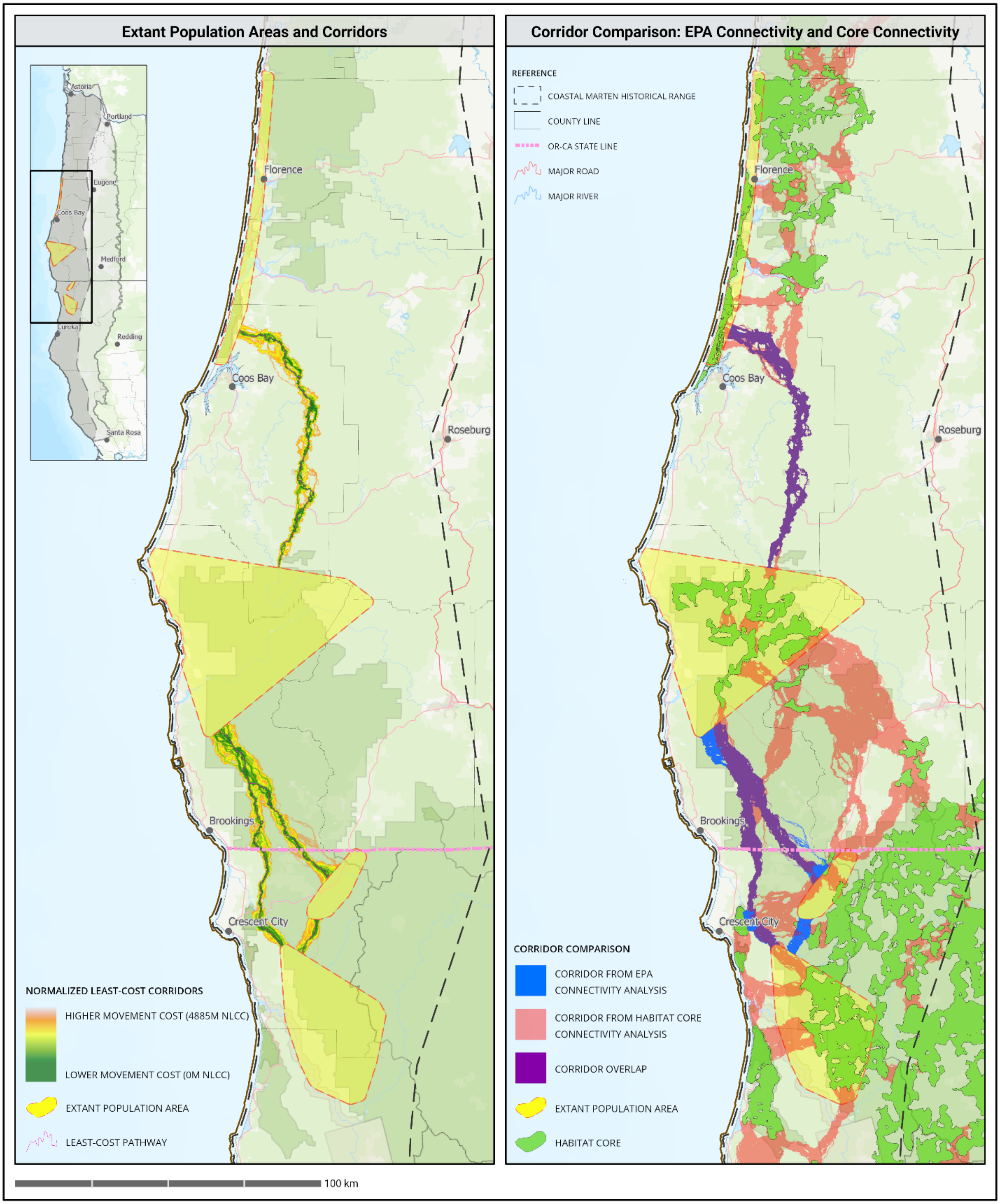


Figure 12. Results of the secondary model treating the four known existing coastal marten populations as habitat cores. Least-cost corridors largely overlapped those derived from the primary model results shown in Fig. 10, except for a novel corridor linking the Oregon-California Border and Northern Coastal California populations.

We identified five "functionally connected clusters" of habitat cores separated by ≤45 cost-weighted km (Fig. 13, Appendix 6). We re-emphasize here that these estimates are very sensitive to the exact numbers used to parameterize the resistance values of the various modeled land cover types, and should be taken more as a relative estimate (e.g., one set of habitat cores is probably better connected than another) than a rigorous prediction of coastal marten movements through the landscape. The habitat core clusters illustrated with this analysis included:

A. A cluster of three habitat cores (Core ID #s 1, 2, and 3) on the Siuslaw National Forest in northwestern Oregon (Tillamook, Yamhill, and Lincoln Counties). To our knowledge, there have been no recent marten detections in this area.
B. A cluster of 12 cores associated with the Central Coastal Oregon population, mostly on the Siuslaw National Forest, but also some on land administered by the U.S. Bureau of Land Management and the State of Oregon (Core ID #s 4 - 15). While most of the unoccupied inland habitat cores (Core ID #s 4-9 and 11-14) appear to be reasonably well connected to each other, connectivity between these areas and the occupied habitat in the coastal dune forests (Core ID #s 10 and 15) appeared to be rather marginal considering the relatively low Euclidean distances separating these cores (6.2 – 14.8km). Only the bifurcated linkage (Linkage ID#s 18 and 20) leading inland from one of the occupied habitat cores (Core ID# 10) was <45 cost-weighted km. Significant barriers between the coastal and inland habitat included U.S. Highway 101, several developed areas, rivers and estuaries in the Umpqua and Siuslaw River watersheds, and Siltcoos and Woahink Lakes.
C. The five habitat cores associated with the Southern Coastal Oregon population (Core ID #s 16 - 20) constituted a single cluster. All linkages between these cores except one were well connected (with the fifth being moderately connected), in spite of the Rogue and Coquille Rivers flowing through this area. Most of these cores are within the Rogue-Siskiyou National Forest.
D. The California-Oregon Border and Northwestern Coastal California populations occur amidst the greatest extent of coastal marten core habitat and the largest number of cores. This cluster of 26 cores, most of which are not known to be occupied, is centered in Humboldt, Del Norte, and Siskiyou Counties in CA, but extends into Trinity County, CA and Josephine and Jackson Counties, OR (Habitat Core ID #s 21 - 46). Most of this habitat occurs within the Six Rivers and Klamath National Forests, although there are a number of other land management entities (federal, state, tribal, and private) with noteworthy holdings. Significant barriers to movement separate many of these habitat cores, particularly the Klamath River, Mad River, and Redwood Creek drainages, U.S. Highway 101, and some large expanses of industrial forest land.
E. Two small habitat cores (both <3000ha) centered on the King Range National Conservation Area in southern Humboldt County and the Sinkyone Wilderness State Park in Mendocino County form a cluster together (Habitat Core ID #s 47 and 48).

In addition, three smaller habitat cores (two <3000ha, one <4500ha) towards the southern end of the coastal marten's historical range (Habitat Core ID #s 49 - 51) were too isolated to form clusters with any other cores. These were centered on the Jackson Demonstration State Forest and a block of private lands east of Point Arena in Mendocino County, and Salt Point State Park in Sonoma County.

Re-examining the habitat core and least-cost corridor map shows the specific linkages that form these clusters. Those shown in green or orange in Fig. 14 were classified as "well connected" or "moderately connected", respectively. While two of the clusters (B and D) have internal linkages that were classified as "poorly connected" (shown in red), all of their cores can all be linked to the broader cluster by at least one green or orange least-cost corridor. Additional supplemental maps illustrating coastal marten habitat connectivity using cost-weighted and Euclidean distances can be found in Appendix 6.

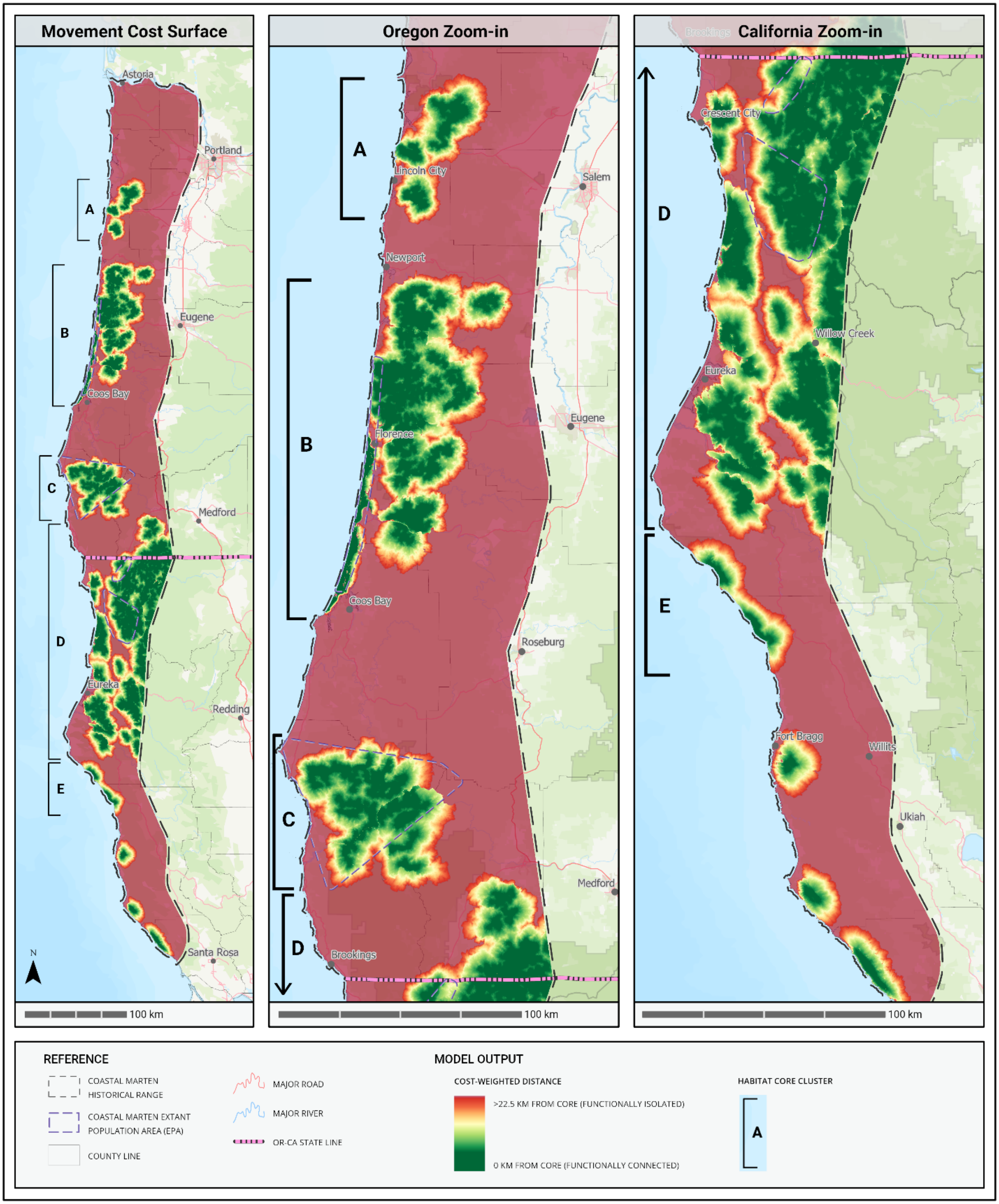


Figure 13. Clustering of coastal marten habitat cores connected by ≤45 cost-weighted km.

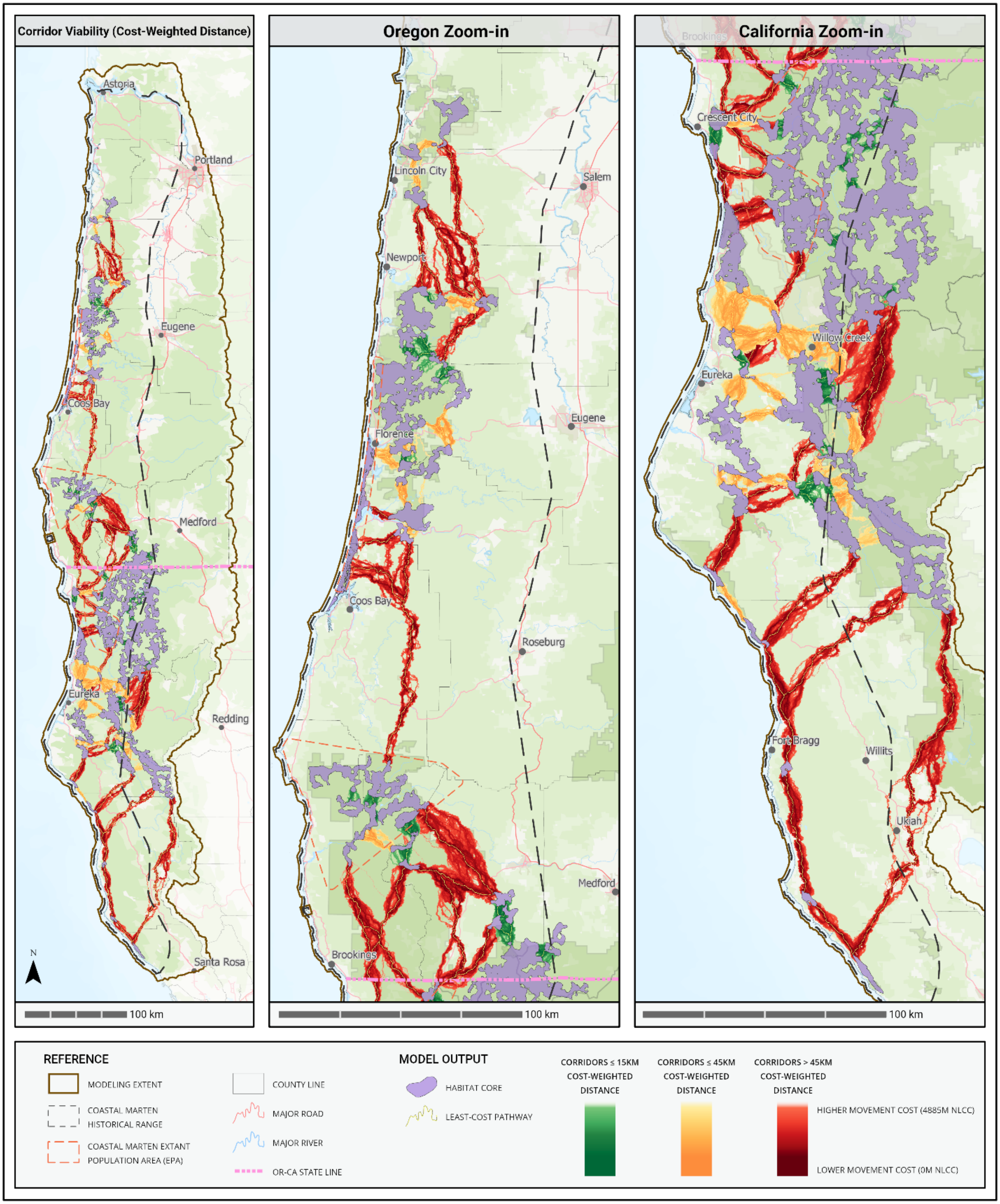


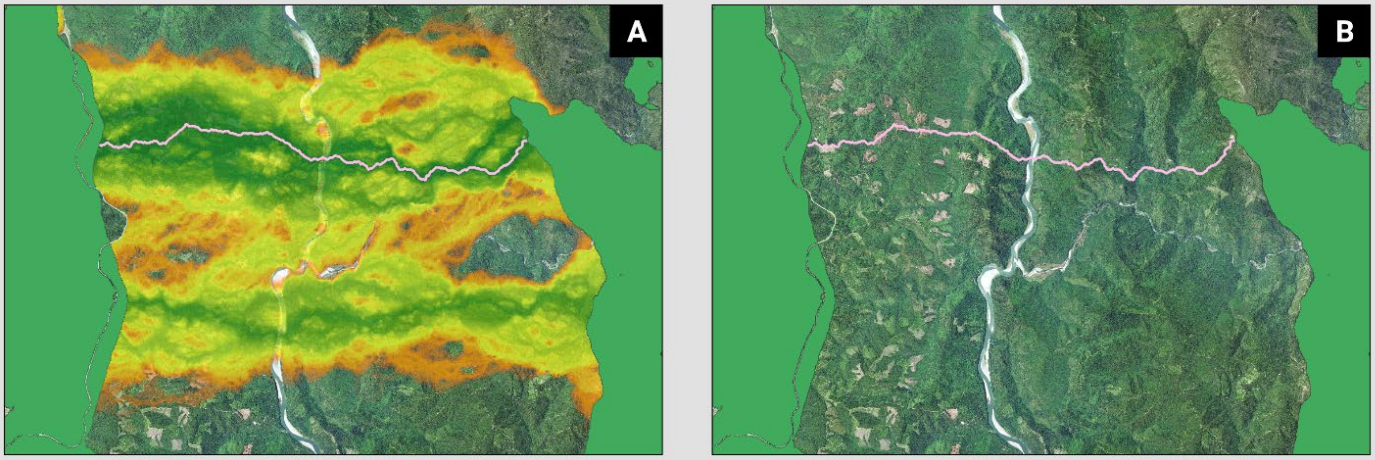
Figure 14. Coastal marten least-cost corridors classified as well connected (≤15km), moderately connected (≤45km), or poorly connected (>45km) based on cost-weighted km. The habitat cores within the clusters depicted in Fig. 13 can all be linked by either a well or moderately connected corridor (i.e. green or orange, respectively).

Management Planning Scenarios

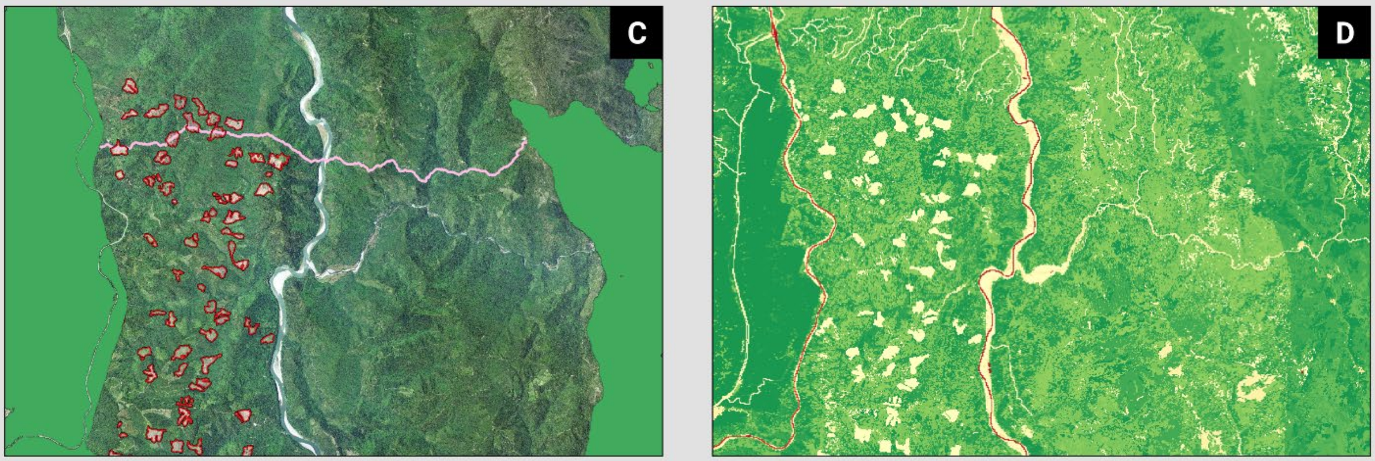
Timber harvest scenario

Our primary model showed the least-cost corridor connecting habitat cores 25 and 33 as having a somewhat unusual "parallel" structure (Fig. 15). The LCP passed through the North Fork Ah Pah Creek drainage on the west side of the Klamath River and the northern side of the Blue Creek drainage on the east side, while a few km to the south was another discrete area of relatively high connective value running through the South Fork Ah Pah Creek drainage and south side of the Blue Creek drainage. The LCP length was 16.69km, with a CWD equivalent of 47.71km (Link ID #50, see Appendix 4), with both corridors crossing the Klamath River. This area is of conservation significance because it represents the least-cost linkage between the area occupied by the bulk of the Northern Coastal California population and the Redwood State and National Parks, which have been identified as a potential Population Reestablishment Area (Slauson *et al.* 2019a).

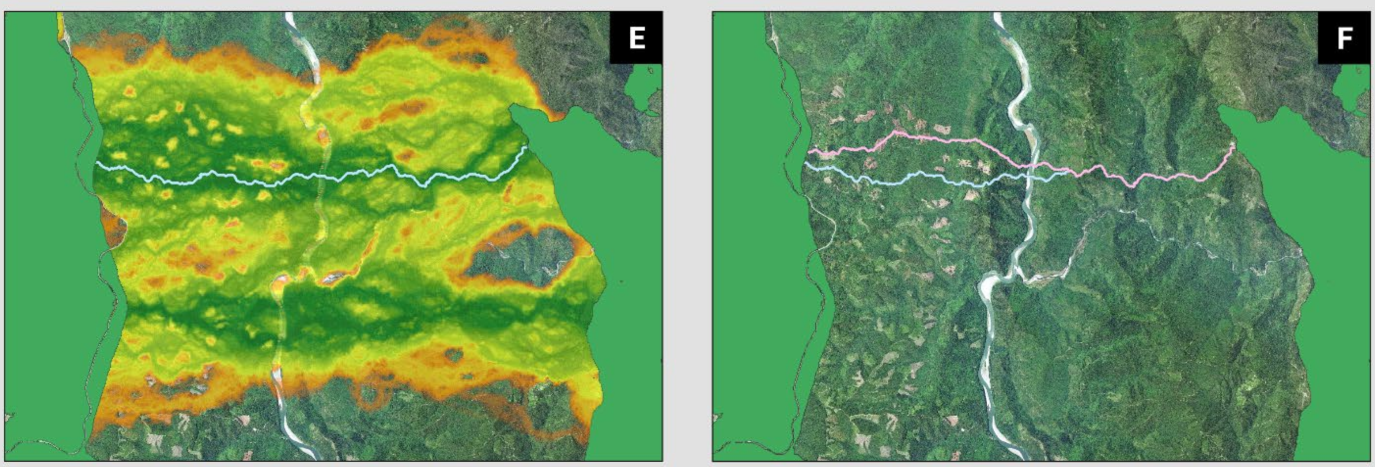
Running Linkage Mapper on the revised resistance surface incorporating post-2012 timber harvests showed the same general pattern of parallel corridors running through the same areas of the landscape. The new LCP length was 16.96km, with a CWD equivalent of 49.46km. Thus, in the post-2012 harvest landscape, the modeled connectivity between these two habitat cores was slightly diminished, with the LCP length increasing 1.6% and CWD increasing 3.7%. Another noteworthy change in the post-harvest landscape is the seeming relative improvement of the connective value of the swath of land between the northern and southern areas of high connectivity, with it becoming more similar to that of the LCP than was modeled in the pre-harvest landscape (i.e. the increase in green pixels in this area visible in Fig. 15E). This is not so much a result of a genuine improvement of connectivity in this area as the fact that it saw relatively less harvest during these years and the connectivity value of the linkage area as a whole became more uniform.



The primary coastal marten connectivity model identified a least-cost corridor (pink line) linking two habitat cores (in green) with considerable conservation significance (A); the Northern Coastal California population occupies the eastern core, and Slauson et al. (2019a) identified the western core (Redwood State and National Parks) as a potential Population Reestablishment Area. However, examination of aerial imagery showed that several patches of mature forest traversed by the least-cost path have been harvested since the data used to derive the old-growth structure index (OGSI) were produced in 2012 (B).



We used Google Earth Engine to delineate areas where timber had been harvested on this part of the landscape between 2013-2018 (in red) (C), and then created an updated resistance surface where all of the pixels in these harvest areas were assigned to the highest OGSI-based resistance category (10/pixel) (D). See Fig. 4, Part 2 for map legend.



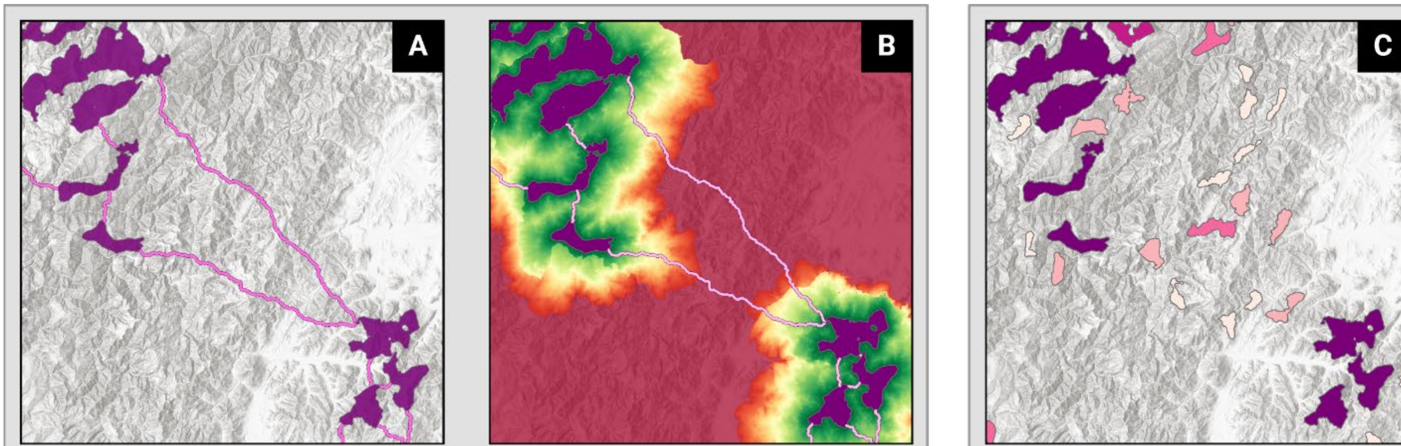
We reran Linkage Mapper using the updated resistance surface (E). The least-cost path shifted slightly to the south (F), and the least-cost corridor became somewhat more diffuse as the amount of pixels with higher OGSI values has declined, although the general "parallel" corridor structure still exists. The post-harvest least-cost path was 1.58% longer than in the main model, and the cost-weighted distance was 3.67% longer. Note that this methodology could also be applied to examining the effects of proposed harvests or other habitat alterations.

Figure 15. Development and outcome of the timber harvest scenario. After an examination of the landscape revealed significant changes since the GNN data were collected in 2012, we modified the primary model to explore the potential effects of recent timber harvests on habitat connectivity for coastal marten. A similar methodology could also be used to estimate the potential impact of proposed timber harvests.

Habitat restoration scenario

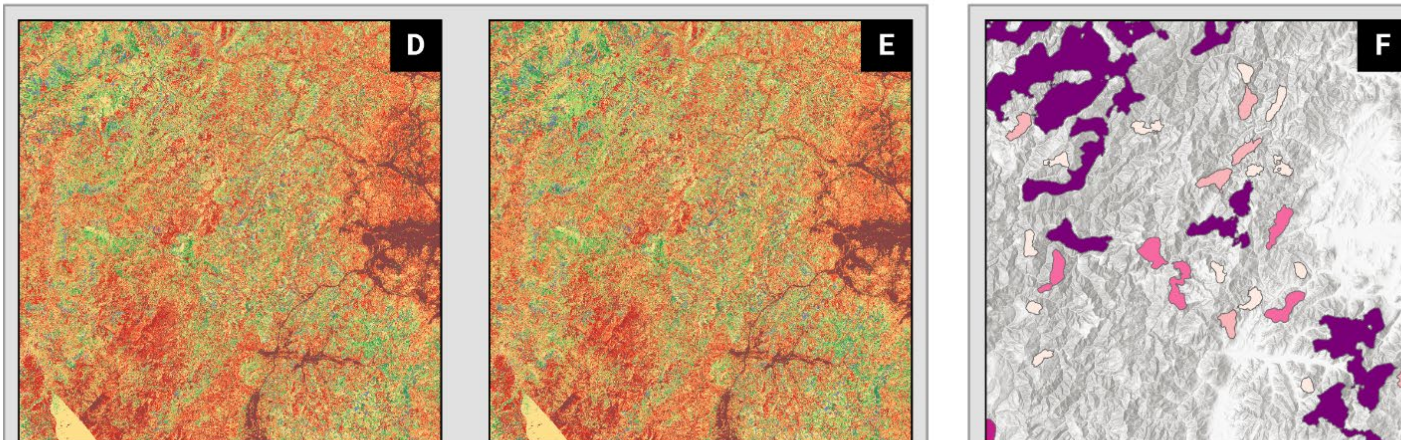
Our primary model found the shortest LCP between habitat core clusters C and D in southwestern Oregon to be Link ID #36 connecting cores 20 and 21 (Fig. 16, see also Appendix 4). The Euclidean distance between these cores was 31.8km, while the LCP length was 39.9km and the cost-weighted equivalent distance was 59.8km. Thus, under our connectivity metric categories these cores would be considered “moderately connected” based on Euclidean distance and LCP length and “poorly connected” based on CWD. This area is of broader conservation significance because it represents the best quality linkage between habitat core cluster C supporting the Southern Coastal Oregon population with habitat core cluster D (Fig. 13).

Increasing the OGSI of each pixel on the landscape by 1.0 turned out to be sufficient to form a new core within the linkage between cores 20 and 21. While we had planned to create a factorial set of modified landscapes in which we increased OGSI by 1.0 in each until we achieved this result, we were successful with the first slight modification. Two small habitat cores (1155.5ha and 760.9ha in size) present in the original modeled landscape that were below our 1500ha threshold to be included in the final model (Fig. 16C) combined to form a new core of 2861.9ha in the modified landscape (Fig. 16F). In this modified landscape, habitat cores 20 and 21 are now bridged by the new core, which is located in the Rogue River-Siskiyou National Forest in Josephine County about 20km west of the town of Grants Pass. The LCP connections from the existing cores to this new core are 15.0km and 18.4km, respectively, and the cost-weighted equivalent distances are 20.5km and 32.8km. Both of these linkages would be considered “moderately connected” using these two connectivity metrics, and “well connected” using Euclidean distance (both <15km). Buffering this new core by a distance of 977m (the size of the moving window we used in our Core Mapper trials) illustrates the discrete portion of the landscape where management for connectivity would have the best impact in this area (Fig. 16H). The total amount of land where a modest improvement in habitat quality would have to take place to achieve these gains in connectivity would be 7268.3ha (the area of the new core plus the buffer).



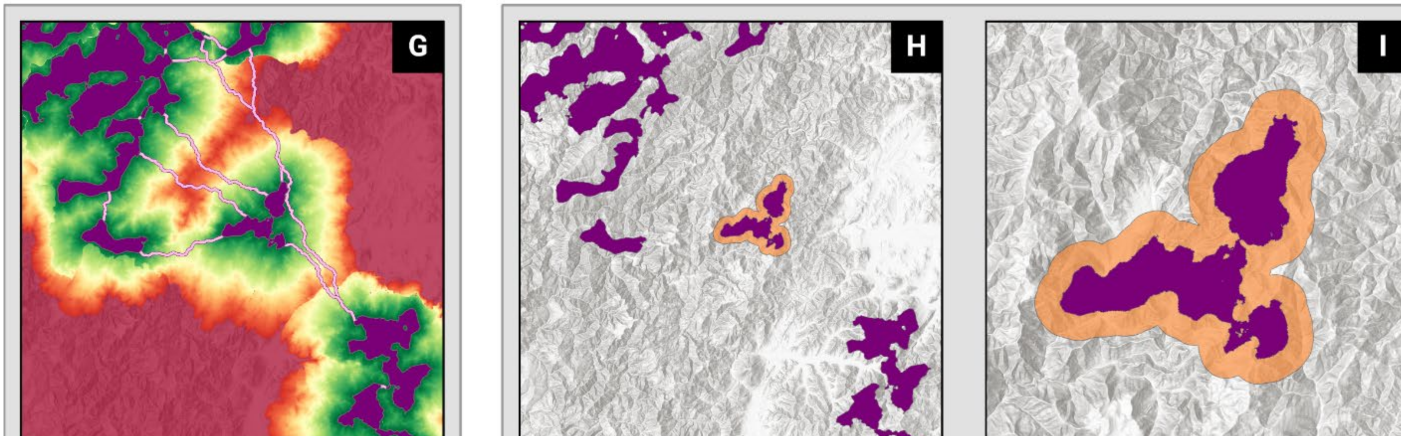
Identify Functional Isolation: While the results of our primary model run identified a least-cost path in this part of the landscape as the best quality linkage between Oregon and California (A), analysis of the cost-weighted distance indicated that these habitat cores (shown in purple) were poorly connected (B). See Figs. 8 and 11 for map legends. Least-cost corridor delineation has been omitted here for clarity.

Identify Smaller Habitat Cores: Cores that are too small to be included in our final model but large enough to include one female home range (i.e. 300-1499ha; shown in shades of pink) can identify areas where new full-sized cores might form with small improvements in habitat values.



Increase Habitat Value by 1: The habitat cores in our primary model were derived from a surface with pixel values ranging from 0-100 (D). These pixel values were mostly based on the old-growth structure index (OGSI), a model of forest maturity incorporating data on tree size diversity and the abundance of large trees, snags, and downed wood. By growing all non-zero values on the habitat surface by 1.0, we simulated a modest amount of forest growth (E). See Fig. 5 for map legend. While this modest increase is barely discernable by visual inspection, it can have a noticeable impact on moving window averages in Core Mapper.

Rerun Core Mapper: In this example, running Core Mapper on the revised habitat surface showed that even the slight increase in pixel values (1.0 each) was sufficient for a new habitat core meeting our 1500ha minimum size threshold to form from the nucleus of two smaller "mini-cores".



Verify Functional Connectivity: We then combined this new habitat core with those used in the primary model and re-ran them in Linkage Mapper with the original resistance surface. Functional habitat connectivity for coastal martens improved greatly (see text for details).

Delineate Target for Restoration: This scenario revealed that modest changes in habitat quality on a relatively small portion of the landscape (7268.3ha) could yield significant improvements in connectivity for the coastal marten (H). This area encompasses the new habitat core (2861.9ha) plus a buffer the radius of the moving window we used in Core Mapper (977m) (I). This methodology appears to be a relatively simple and efficient way to identify and prioritize areas where management could be targeted to improve connectivity.

Figure 16. Development and output of the habitat restoration scenario. This secondary model explored a method for identifying discrete areas on the landscape where modest improvements to habitat quality might result in significant gains in habitat connectivity for coastal martens.

Discussion and Conclusions

Habitat Connectivity Patterns

Our model output indicates that habitat connectivity for the coastal marten varies across its current distribution, with the two populations in California showing greater connectivity to other populations and to suitable but unoccupied habitat than the two populations in Oregon.

The coastal marten populations within California (known as the Northern Coastal California and California-Oregon Border populations (USFWS 2018)) likely have reasonably good connectivity, as they were connected by a very large habitat core area rather than a corridor between two cores, indicating that a large amount of potentially suitable habitat exists between them (Fig. 10). The Euclidean distance between the two populations is 11.2km, and the travel distance within the core connecting them is <30km. Furthermore, the secondary model treating the polygons of the four existing populations described in the SSA (USFWS 2018) as the only habitat cores on the landscape produced a least-cost corridor between these populations that was "moderately connected" by CWD (21.2 cost-weighted km) (Fig. 12). It is likely that animals move between these populations at least occasionally. A generally applied "rule of thumb" in conservation biology states that isolated populations need a minimum of one immigrant individual per generation to recruit successfully as a breeder if they are to avoid inbreeding depression (Franklin, 1980, Mills and Allendorf 1996, Wang 2004, Grauer *et al.* 2019). The estimated generation time for coastal martens is about five years (Slauson *et al.* 2019a), and while we have no current estimates of actual movement rates between these populations, we believe that the degree of habitat connectivity present (both Euclidean and cost-based) and the known dispersal abilities of the species make it likely that there is adequate gene flow between them. Management that maintains or improves connectivity between these populations should be a priority for conservation planning related to this species.

Recent detections in Prairie Creek Redwoods State Park, including evidence of successful reproduction (CA Dept. of Parks & Recreation *unpublished data*), show that coastal martens may be recolonizing this area on their own from the Northern Coastal California population (it is possible there was a small remnant population residing there all along, although in our opinion the evidence suggests this to be unlikely). The least-cost corridor connecting these areas (Link ID #50) has a Euclidean distance of only 12.3km and a LCP length of 16.7km, but a CWD of 47.7 cost-weighted km because it crosses the Klamath River and large areas of industrial forest land. While the modest amount of dispersal that appears to occur in this area of marginal connectivity may in time lead to the development of a self-sustaining population west of the Klamath River, this process would likely be facilitated by assisted dispersal (Seddon 2010) as has been proposed by the Humboldt Marten Conservation Working Group (Slauson *et al.* 2019a). The long-term persistence of a marten population at Prairie Creek (either reintroduced or naturally established) should be greatly enhanced by maintaining a high degree of connectivity to the Northern Coastal California population (i.e. between Core ID #s 25 and 33), similar to the situation documented in American marten populations in Wisconsin by Grauer *et al.* (2019). Slauson *et al.* (2019a) identified this area as a "Landscape Connectivity Area" (LCA), which was defined as low suitability habitat located between two areas of higher quality habitat where restoration to improve connectivity should be a priority. This LCA was mapped as a broad swath of land nearly 40 km long and 5-15 km wide (339.8 km^2 or 131.2 miles^2 total) between Six Rivers National Forest and Redwood National and State Parks. The swath of land delineated as a least-cost corridor by our primary model (Link ID #50) was 42% of this size at 143.0 km^2 (55.2 miles^2), and only about half of this corridor had the highest values for connectivity (the green areas in the corridor mapped in Figs. 10 and 15). About 45% of this corridor was also included in the Conservation Strategy as part of an Existing Population Area (i.e. occupied habitat) rather than the LCA. Both our primary model and our secondary timber harvest scenario model allow for a more targeted focus for conservation efforts aimed at maintaining or improving habitat connectivity for coastal martens that is more practical from a planning perspective. Furthermore, the Linkage Mapper output allows for corridors to be visualized even more narrowly than we chose to in our primary and secondary models and applied to an even smaller area if desired (see Fig. 8 for an example).

The Southern Coastal Oregon population delineated in the SSA (USFWS 2018) contains three habitat cores, and two more cores occur in relatively close proximity to the southeast of the area considered to be occupied by martens. Four of the five linkages between these cores (Link ID #s 28, 29, 30, 33, and 35) were classified as "well connected" by CWD, with the fifth being "moderately connected" (Fig. 14). Marten records within this population are scattered, occurring in several discrete pockets with large gaps between them (Zielinski *et al.* 2001, Moriarty *et al.* 2016), indicating that what is referred to as the Southern Coastal Oregon population may functionally act as multiple smaller populations rather than one larger one. Management aimed at maintaining or improving habitat connectivity between areas known to be occupied by martens should be a management priority. Our model further indicated that this population is likely functionally isolated from the nearest populations in Oregon and California by long distances and a relative paucity of mature forest habitat in the intervening areas, meaning it may be susceptible to inbreeding depression in the long-term. The 2018 Klondike Fire may have further degraded connectivity between these populations (see Fig. 19 below), and this disturbance would not have been incorporated into the habitat data we used in the model.

The two coastal marten populations in Oregon are likely isolated both by long distances to the nearest known population and the fragmented nature of the forest on much of the intervening landscape (Figs. 13 and 14). Much of the forestland in southwestern Oregon occurs in a "checkerboard" pattern of public and private ownership with different harvest patterns that fragment habitat at a landscape scale (Meyer *et al.* 1998, Stanfield *et al.* 2002) (the effects are visible in Figs. 2, 4, and 6). Other areas are dominated by large industrial tree plantations that are usually harvested on a rotation too short for late seral characteristics to develop (Schillinger *et al.* 2003). Several large wildfires in the area over the past two decades have probably further degraded habitat quality between the Southern Coastal Oregon and California-Oregon Border populations (Short 2017). Hargis *et al.* (1999) found that Pacific martens in Utah were sensitive to forest fragmentation at a landscape scale, with capture rates declining as the proportion of natural openings and clearcuts increased, and that martens were nearly absent in areas with >25% non-forest cover. Several studies have analyzed movement pathways of Pacific martens at more localized scales and demonstrated that martens strongly select paths that avoid crossing unforested areas (Cushman *et al.* 2011, Moriarty *et al.* 2015, 2016, Martin *et al.* 2019). These findings fit a more generalized pattern reported by a number of authors indicating that marten species respond negatively to loss and fragmentation of forest cover (e.g. Chapin *et al.* 1998, Potvin *et al.* 2001, Virgós and García 2002, Bull *et al.* 2005, Moriarty *et al.* 2011). Management aimed at creating usable corridors reducing the isolation of the Oregon populations will likely present considerable challenges, although there are likely opportunities to create "stepping stones" of habitat over the long term that could improve connectivity (Baum *et al.* 2004, Saura *et al.* 2014) (for example, see Fig. 16).

It is noteworthy that the model indicated high CWDs between the two habitat cores supporting the Central Coastal Oregon population (Core ID #s 10 and 15) and the nearest habitat cores to the east (Figs. 13 and 14). The ratio of cost-weighted:Euclidean distances for these five linkages were all in the top 20th percentile for this metric, ranging from 3.45 – 6.58 (Appendix 4, Link ID #s 18, 20, 22, 24, and 27). This indicates that all of these corridors cross areas of high resistance in spite of four of the five having Euclidean distances of <15km. One reason is that our "manually added" cores in the Oregon coastal dune forests were restricted to the west side of U.S. Highway 101, necessitating that LCPs connecting to these habitat cores cross this barrier. However, our analysis also pointed to the importance of waterways and developed areas in isolating this population by creating barriers to dispersal to the inland blocks of potential habitat. There have been very few marten detections in these inland cores (Zielinski *et al.* 2001, Moriarty *et al.* 2016), and Moriarty *et al.* (2019) suggested that there is little suitable habitat in the area based on a lack of similarity to the occupied coastal dune forests. However, we believe it is possible that these barriers prevent dispersal out of the coastal dune forest areas at rates that would allow sustainable colonization of the unoccupied habitat inland. The Central Coastal Oregon population has the highest home range density ever recorded for any North American marten species or subspecies (Linnell *et al.* 2018). While it is likely that the high prey availability and low predation rates evinced by Eriksson et al. (2019) are the primary drivers, it is worth speculating that the highly resistant landscape features bordering the coastal dune forests may also be contributing by discouraging potential dispersers from making forays out of the established population area.

The unoccupied habitat cores with the greatest connectivity to known populations appear to be within habitat core cluster D located in Del Norte and Northern Humboldt Counties in California (Fig. 13). These are largely congruous with the four "Population Re-establishment Areas" identified in the Humboldt Marten Conservation Strategy (see Fig. 25 in Slauson *et al.* 2019a). This is not surprising given that both of these classifications were derived by identifying large areas with high OGSI values. A high proportion of habitat core ID #25 is not currently known to be occupied by martens, even though both of the known California populations are supported by it. It should be a priority to conduct future survey and monitoring efforts for martens in these areas given that it is likely that animals could disperse into them and there is good potential for natural recolonization.

A large amount of habitat core area exists east of the Klamath River (Core ID #s 27, 30, 31, and 34), and also to the south along the edge of the coastal marten's historical range as it is currently defined (Core ID #s 39, 40, 42, 43, and 46). These habitat cores total 273,815ha (34.7% of the total area of habitat cores in the model), and none are known to be occupied. While these appear to represent large areas that may potentially be suitable for management actions aimed at restoring coastal marten occupancy, their location along the edge of the historical range may make them a lower priority in this regard than habitat closer to the coast. There are indications that habitat quality may diminish towards the edge of the historical range, even in areas with high OGSI, because drier summer conditions and higher fire frequency result in areas with shrub cover suitable for martens being more localized than closer to the coast (Fitzhugh 1988, Skinner *et al.* 2006, Stuart and Stephens 2006, Halofsky *et al.* 2011, K. Slauson *pers. comm.*). Our model would not include this potential spatial variation in habitat quality because of its reliance on OGSI. A more sophisticated understanding of home range and dispersal habitat across the coastal marten's distribution, as well as availability of improved landscape scale data on suitable shrub cover, would allow us to improve our ability to model high quality habitat for the species.

To the south of the North Coastal California population, there is a group of habitat cores in central and southern Humboldt County that are either well or moderately connected to each other (Core ID #s 35, 36, 37, 38, 41, 44 and 45). While none of these cores is within an easy dispersal distance from the existing population, they likely support suitable habitat that could potentially be recolonized by martens in the long term. This process would be facilitated by the re-establishment of marten populations in Prairie Creek Redwoods State Park and Redwood National Park. Core ID #33 is largely comprised of old growth redwood habitat in these Parks, and it represents the most direct linkage from the existing population to the unoccupied cores to the south. Because the least-cost corridors connecting these habitat cores cross significant barriers such as the Klamath River, Mad River, and Redwood Creek, assisted dispersal could be explored as a means to accelerate recolonization.

Connectivity metrics

Linkage Mapper provided several metrics for assessing connectivity between coastal marten habitat cores: Euclidean distance, LCP, and CWD, as well as the CWD:Euclidean distance and CWD:LCP ratios (see Appendix 4). In the absence of empirical data on coastal marten movements through the landscape, it is not clear which of these would be best suited for estimating habitat connectivity. Most previous literature on marten dispersal distances is based on measuring the Euclidean distances between observations (e.g. Phillips 1994, Fecske and Jenks 2002, Broquet *et al.* 2006, Pauli *et al.* 2012, Slauson 2017). Furthermore, past studies of population isolation or metapopulation structure have mostly been based on Euclidean distance between habitat patches (Doak *et al.* 1992, Wiens 1996, Rickets 2001), and this includes a number of studies on martens or closely related species (e.g. Virgós and García 2002, Lewis *et al.* 2012), including the coastal marten (Slauson *et al.* 2019a, USFWS 2018). However, Euclidean distance ignores the character of the intervening landscape between habitat patches (referred to by landscape ecologists as the "matrix"), and treats all land cover types as equally crossable. This results in inter-patch distance being the only determinant of isolation, a concept that has its roots in Island Biogeography Theory (MacArthur and Wilson 1967).

LCP and CWD are both derived from the resistance surface, and thus are based on the estimated difficulty and risk of movement by the focal species through the matrix. LCP is perhaps more straightforward to interpret. However, it is based on the "best" pathway available through the landscape between habitat cores, even if this pathway may in fact be of little use by the species because it is too long, crosses significant barriers, etc. While Linkage Mapper includes an option for dropping LCPs above a specified length, we opted not to incorporate it into our model because of the uncertainties around the dispersal abilities of the coastal marten and the relative costs that we used to create the resistance surface. CWD provides a better estimate of functional isolation by using estimates of movement difficulty into an isolation estimate that is directly comparable to the Euclidean distance and LCP. For our model, we did this by converting estimates of the distance through best quality habitat that a dispersing coastal marten might be expected to travel to avoid one pixel of a sub-optimal land cover type. CWD provides insights into the challenges of moving across certain areas of the landscape, and examining linkages with high CWD allows the identification of significant barriers to movement. The exact value CWD is very sensitive to how the resistance values are estimated, and in most cases (including for our model) empirical data to support these estimates are lacking (Sawyer *et al.* 2011). However, as long as the assumptions that were used to produce the model are consistent (i.e. resistance increases as OGSI decreases, rivers and major roads are significant barriers, etc.), the locations of LCPs and the relative functional isolation of habitat core pairs should be similar even if the precise resistance values are adjusted.

Habitat Cores, Marten Occupancy, and Habitat Quality

The extent of the habitat cores identified by our model indicates that a good deal of potentially suitable coastal marten habitat still seems to exist on the landscape. However, most habitat cores are not known to be currently occupied. This is probably the result of several factors: (1) the legacy of historic trapping, (2) the effects of recent and ongoing fragmentation of mature forests, and (3) the species having relatively low capacity for range expansion stemming from its low birth rate and limited dispersal capability relative to the scale at which its habitat is now fragmented (USFWS 2018).

A good deal of potentially suitable but unoccupied habitat connects the two described coastal populations in California. There are several possible explanations for the gaps in marten occupancy within the very large habitat core (Core ID #25) shared by these populations. One is that some of the habitat within this core is of lower quality than is indicated by our model, as was discussed above (see P. 53). Another is that these marten populations may be demographically limited by low birth and/or survival rates and have not been able to colonize new habitat, or have only done so very slowly (Schrott *et al.* 2005). Finally, it is possible that martens do in fact inhabit some of this intervening area, but have not yet been detected. Much of habitat core ID #25 has not been systematically surveyed, and other parts of it were only surveyed once sometime between 1990 and 2010, although published descriptions of prior survey efforts make it difficult to parse out sampling dates and frequency and the exact methods used at each site (see Zielinski *et al.* 2001, Slauson *et al.* 2019b). The California-Oregon Border population was only discovered in 2011 (U.S. Forest Service *unpublished data*, USFWS

2018), and its status and extent are still not completely understood. Survey efforts aimed at delineating the area of occupancy and size of this population are ongoing. Furthermore, recent research on coastal martens suggests that camera traps have a false negative detection rate of about 36% (Moriarty *et al.* 2018b), and a study in fishers (*Pekania pennanti*), a closely related species, found that the bait lures used in live traps and camera traps have an effective sampling diameter of 70m – 140m (K. Moriarty and M. Ellis *unpublished data*). This indicates that marten occupancy may have been missed in lightly surveyed areas or areas with low population density and/or transient occupancy. It is likely that continuing surveys for coastal martens will result in refined or expanded boundaries of some of the existing population polygons, and may reveal occupancy in habitat cores or other areas not currently known to be occupied. While the rugged wilderness character of much of this area makes conducting systematic survey efforts a challenge, these could potentially fill a significant and critical gap in our understanding of the conservation status of the coastal marten.

It is important to bear in mind that our habitat cores were not intended to be synonymous with coastal marten habitat or to comprehensively map it on the landscape; rather, they were intended to represent large blocks containing enough suitable habitat to potentially support a population. Clearly, much suitable habitat is found outside of these larger cores, as there are a number of recent records of marten detections that fall outside of habitat cores within the California-Oregon Border population, on the western edge of the Northern Coastal California population, and within the Southern coastal Oregon population (USFWS *unpublished data*). Likewise, not all of the area inside of a core will be good quality habitat; the boundaries reflect the mean habitat values within the surrounding portion of the landscape measured by the "moving window". The Core Mapper output is very sensitive to the mean habitat values and size of the moving window used in delineating habitat cores (Appendix 3). Habitat quality often varies within individual cores, with discrete concentrations of higher and lower quality habitat evident when the habitat cores are overlain on the habitat value data layer from which they were derived (Fig. 17). The 1500ha minimum core size that we used also resulted in 118 smaller habitat cores being dropped from the final model (Fig. 18). Many of these smaller cores form important parts of the least-cost corridors mapped by Linkage Mapper, and many could also be capable of supporting some marten territories. While the corridors show the lowest cost linkages between habitat cores, many may function more like "stepping stones" of small remnant patches of higher quality habitat embedded in a matrix of lower quality habitat. Habitat stepping stones have often been identified as important determinants of habitat connectivity across landscapes (Schultz 1998, Hale *et al.* 2001, Baum *et al.* 2004, Kramer-Schadt *et al.* 2011, Saura *et al.* 2014).

Our habitat model is primarily based on a modeled old-growth structure index meant to map seral stage and forest structure across a broad landscape scale. While Slauson *et al.* (2019b) found OGSI to be the best predictor of coastal marten habitat based on occurrence, it is an imperfect one and there is still more to be learned about which habitats the species will use, and which ones may be of higher or lower quality for its various needs (breeding, foraging, dispersal, etc.) We also did not explicitly include data on the shrub layer in the model (in part because we lacked a suitable landscape-scale data layer) even though there are known instances of habitats occupied by the species that have low OGSI but a highly developed ericaceous shrub layer. Some instances where martens have repeatedly been detected outside of our habitat cores may represent areas where the shrub layer is a more important driver of habitat quality than the presence of mature forest conditions.

Finally, it is likely that, as with other marten subspecies that have been studied, fragmentation of mature forest is associated with negative effects such as higher predation rates and lower survival (Drew 1995, Hargis *et al.* 1999, Cushman *et al.* 2011, Moriarty *et al.* 2011) and areas of high forest fragmentation may represent sink habitat (Pulliam 1988, Pulliam and Danielson 1991). Our habitat core mapping did not take patterns of fragmentation into account beyond their effects of the moving window and core expansion processes. It is possible that some habitat cores or portions of large cores are embedded in or adjacent to highly fragmented areas where marten populations would not be able to achieve demographic rates needed to sustain themselves in the long term, even if some level of occupancy or even reproduction was possible. However, such areas would likely still play an important role in maintaining stability and resiliency in the broader population (Howe and Davis 1991, Falcy and Danielson 2011, Heinrich *et al*. 2015).

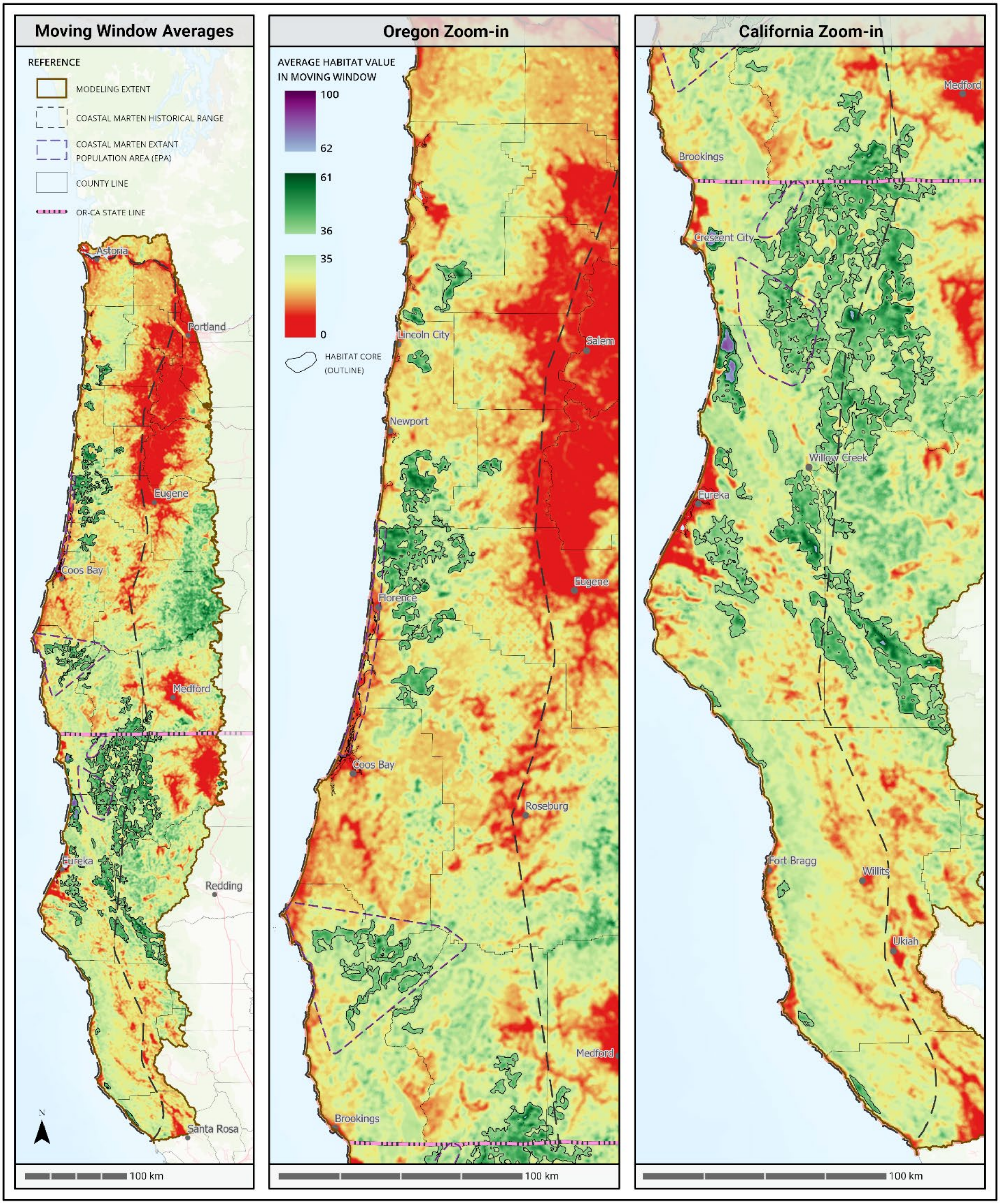


Figure 17. Distribution of per pixel moving window average habitat values reveals variation in habitat quality within cores and across the landscape. Habitat values are primarily based on OGSI; 36 is the median value of pixels estimated as beginning to exhibit old growth characteristics, while 62 is the median value of pixels estimated to be mature old growth forest.

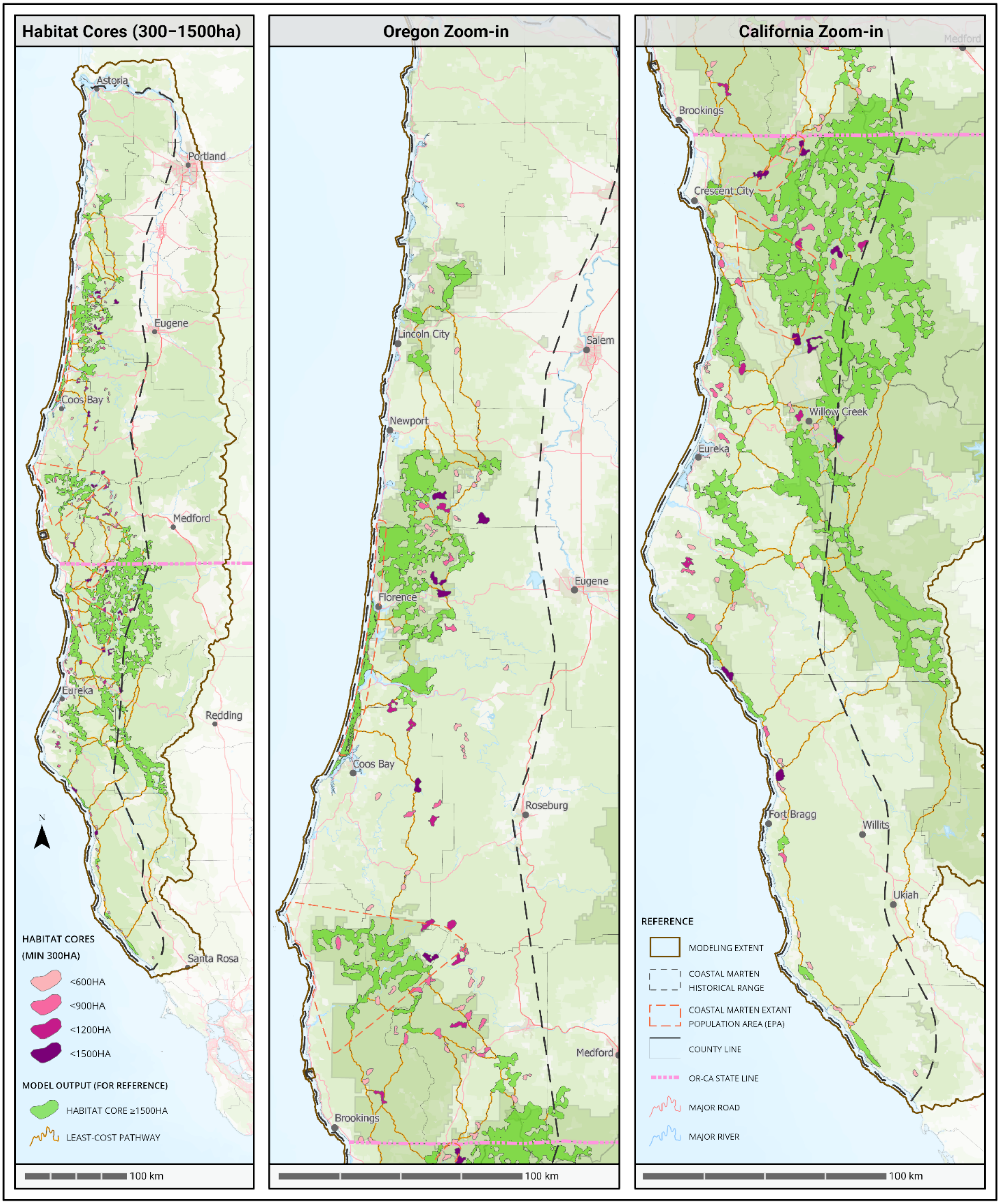


Figure 18. Locations of habitat cores smaller than our 1500ha threshold for inclusion in the final model, but large enough to potentially support at least one coastal marten territory (>300ha). Many of these smaller cores form important components of least-cost corridors (note their presence along many of the least-cost pathways), and may represent important "stepping stones" that could facilitate dispersal across the landscape.

Implications of Management Boundaries

Publically managed lands

Most of the habitat cores mapped by our model occur on lands managed by the U.S. Forest Service (82.3% of the total area), as do most of the areas currently known to be occupied by coastal marten. Management policies on USFS lands vary across sites, ranging from largely unmanaged wilderness areas to areas managed for multiple uses where timber harvesting and other resource extraction activities are allowed. This is also the case on lands of the U.S. Bureau of Land Management, which is responsible for the second largest amount of habitat core area managed by a single public entity (3.2%). While much of the habitat in public ownership has some degree of protection from modification or conversion, less than a third (29.3%) of the habitat core area occurs on lands managed most strictly for biodiversity conservation (GAP Status 1 and 2). All of the habitat core area managed by the National Park Service and most managed by the state of California fall into this category. Because of the various mandates and statutes (e.g. the Northwest Forest Plan, Endangered Species Act, etc.) that public land management agencies are beholden to, lands managed for multiple uses will play a vital role in maintaining suitable habitat conditions for coastal martens over much of the area where they currently exist.

Other lands

There are some areas on non-publically owned lands of potential importance for coastal marten habitat connectivity. The corridor connecting the Northwestern Coastal California population to Prairie Creek Redwoods State Park that we explored in more detail in our timber harvest scenario (Fig. 16) crosses both privately owned and tribal lands. There are also noteworthy habitat cores that are entirely comprised of private timberlands in central Humboldt County, CA (Core ID #s 36 and 38) that link the Redwood National and State Parks (which have potential for natural recolonization or human-assisted dispersal of martens) with suitable habitat to the south and east. These cores may act as future "stepping stones" that facilitate the expansion of the species into more of its former range (Kramer-Schadt *et al.* 2011, Saura *et al.* 2014), such as habitat core ID #s 37, 41, and 45, which are mixtures of private and public lands (see Appendix 5). Most of the land traversed by the linkages between all of these habitat cores is privately owned as well, meaning that the importance of conservation efforts on private lands to the long-term potential for the recolonization of central and southern Humboldt County by the coastal marten cannot be overstated.

Unlike California, Oregon had no habitat cores that were predominately privately owned, and most corridors in the state primarily crossed public lands as well. This difference likely results from regulations governing timber management on private lands in Oregon contributing to a predominance of younger, even-age stands with relatively little older forest (Kennedy *et al.* 2012). However, a number of the habitat cores in the state have small areas of private lands within them or around the edges. Furthermore, all of the corridors linking the habitat occupied by the Central Coastal Oregon population (Core ID #s 10 and 15) cross significant amounts of private land.

Wildfire Risk and Coastal Marten Landscape Connectivity

Wildfire is a natural disturbance in coastal marten habitat, and natural fire regimes are vital over the long-term for creating features such as denning and resting sites (USFWS 2018, and references therein). However, in the aftermath of a fire, marten habitat quality can be greatly degraded for decades or even centuries. Fire regimes vary considerably over the historical range of the species, with lower severity, higher frequency regimes predominating in most of California and southwestern Oregon, and higher severity, lower frequency regimes in most of Oregon north of Cape Blanco (USFWS 2018). Habitat recovery from a lower severity fire may take between one and three decades (depending on the ecological function of the habitat for the marten – i.e. breeding, resting, foraging, or dispersal), while recovery from high severity "stand replacing" fires may take a century or more (Naney *et al.* 2012), and may ultimately even result in "type conversion" from forest to shrub-dominated chaparral-type habitat when combined with the effects of climate change (Perry *et al.* 2011, Collins and Roller 2013).

Several large fires have affected marten populations or areas of potential importance for habitat connectivity in recent years. These include the Biscuit Fire in Oregon in 2002 and the Klamath Complex Fire in California in 2008, the effects of which were included in the 2012 GNN data that we used to develop our habitat cores and resistance surface. The effects of the more recent fires were not incorporated into the data, and therefore our model. This includes the large Chetco Bar and Klondike Fires (in 2017 and 2018, respectively) and several smaller fires that occurred to the south and east of the Southern Coastal Oregon population (Fig. 19) (Walters *et al.* 2011). It also includes fires of the Gasquet Complex (2015) and Eclipse Complex (2017) that affected the area connecting the two California populations. It is important to keep in mind that fire perimeter maps do not necessarily indicate that all marten habitat within has been "destroyed" or severely degraded as fire intensity can vary considerably across the landscape, and in fact over time fires may create important habitat features for the marten. However, more recent habitat data should be consulted when assessing the status of habitat cores or corridors in areas that have been impacted by wildfires.

Wildfire is probably less likely to disrupt or severely degrade coastal marten habitat cores and corridors located closer to the coast than those further inland because of the lower severity and frequency of the fire regime (Miller *et al.* 2012). This lower risk makes these cores more attractive for conservation efforts aimed at encouraging recolonization by the species. Translocations of coastal martens into suitable habitat in units of the Redwood State and National Parks has been suggested by the Humboldt Marten Conservation Working Group (Slauson *et al.* 2019a). Most of the known extant populations of coastal marten are located in more fire-prone inland areas, and prioritizing the maintenance or restoration of connectivity from these areas to suitable habitat closer to the coast should also be explored. Climate models project increased warming in inland areas in northwest California relative to the coast (Lenihan *et al.* 2008, Rapacciuolo *et al.* 2014), which in turn will likely cause wildfires in these areas to grow in severity and extent (Westerling *et al.* 2011, Mann *et al.* 2016, Westerling 2016). Therefore, habitat cores near the coast and viable linkages connected to these areas may take on a particularly important role in the long-term conservation of the coastal marten. As an added benefit, conserving these areas may allow the coastal marten to act as a "surrogate species" that aids in increasing the climate resilience of other late seral forest specialist species in the region that are not legally protected or actively managed for (Heller and Zavaleta 2009, Algador *et al.* 2012, Breckheimer *et al.* 2014), thus contributing to the resilience of the ecosystem as a whole.

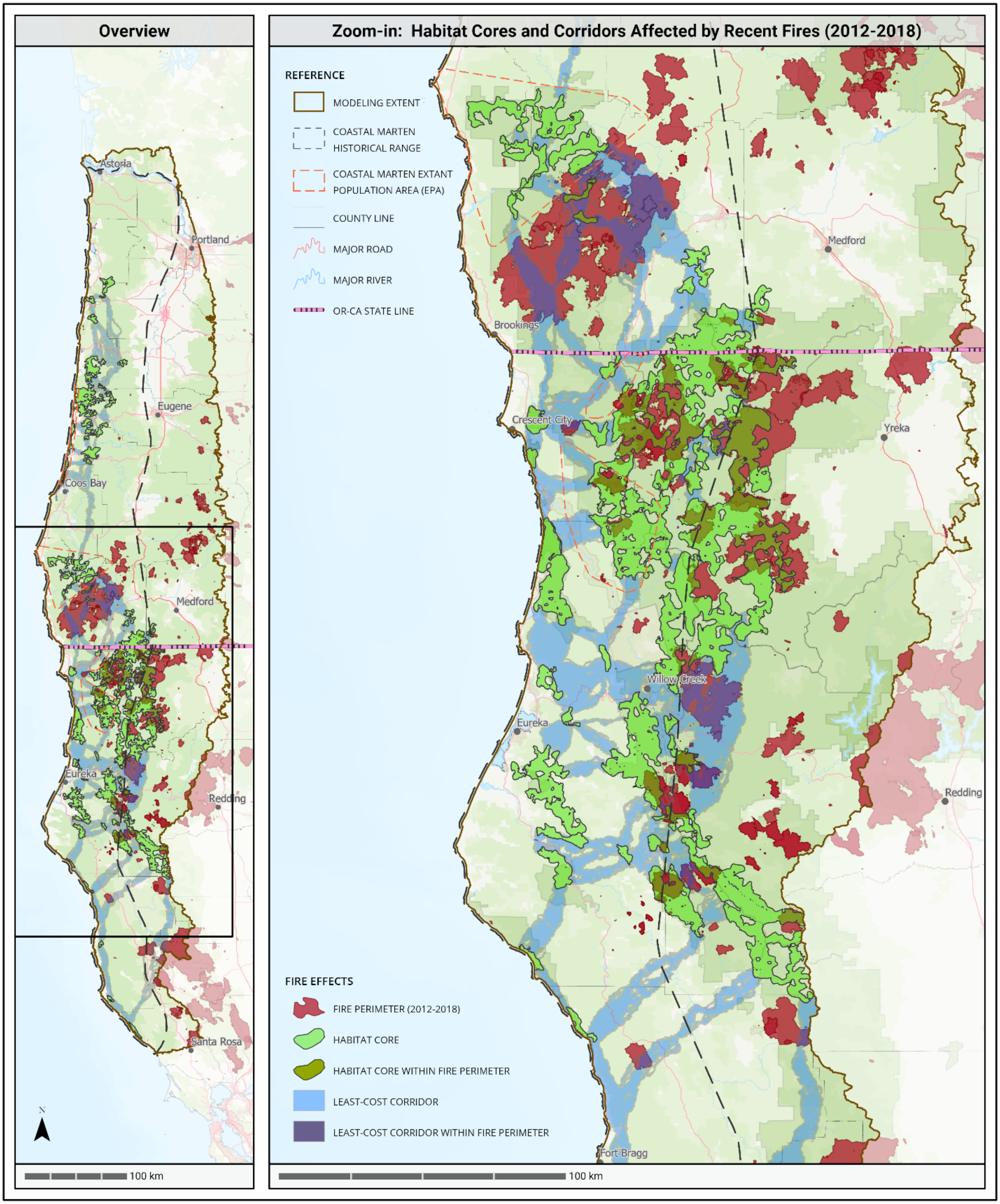


Figure 19. Perimeters of wildfires that affected coastal marten habitat cores and least-cost corridors from 2012-2018. The effects of these fires on habitat conditions were not factored into the OGSI used to model marten habitat quality and resistance to movement across the landscape. Data on fire intensity are not included with these perimeters, and marten habitat within them should not be considered "destroyed".

Implications of the Data Sources Used in the Model

Old-growth structure index (OGSI)

While several factors made OGSI attractive to use as the primary estimator of habitat quality in our model, it also had some potential drawbacks.

The GNN model from which the OGSI is derived provides spatially explicit estimates of forest structure at relatively fine spatial scales across most of the Pacific Northwest (Ohmann and Gregory 2002). It is important to attempt to assess a species' habitat use at a spatial grain similar to that at which the organism responds to landscape pattern (Wiens 1976, Turner and Gardner 2015). The fine-grained structure of the GNN model (30m X 30m resolution) is likely reasonably similar to the "stand scale" of habitat selection described in Slauson *et al.* (2019a). Of the four scales of habitat selection Slauson *et al.* described (microsite, stand, home range, and landscape), stand scale is probably the closest match to that at which dispersing martens make decisions about which pathways to follow (Johnson 2008, Slauson *et al.* 2019a), although landscape scale factors such as predation risk likely play a role as well (Slauson *et al.* 2019a). In addition to its spatial grain, the broad extent of the GNN model was also appealing as it allowed us to compare habitat across the coastal marten's entire historical range in a uniform manner without having to rely on locating and then combining smaller scale habitat datasets that often end at the California-Oregon border.

Using the OGSI also allowed us to produce a relatively simple model that is probably rather conservative in its assumptions. Namely, we assumed that most late seral forest within the species' historical range is likely suitable as habitat to some degree based on studies done in coastal martens (Slauson *et al.* 2007, 2019a) and other populations of Pacific marten (summarized in Slauson *et al.* 2019a). We further assumed that dispersing animals would be willing to use a broader range of conditions than might be acceptable for a permanent home range (Wasserman *et al.* 2010, 2012, Moriarty *et al.* 2015). Finally, the landscape-scale habitat model incorporated into the Humboldt Marten Conservation Strategy (Slauson *et al.* 2019a, b) found that OGSI, along with suitable serpentine habitat where it occurs, did the best job of the variables examined of predicting coastal marten occupancy, although it was largely based on detections from California. Moriarty *et al.* (2019) modeled coastal marten habitat in Oregon and found late seral forest structure to be less associated with occupancy there than understory conditions, although their results were strongly influenced by habitat conditions in the occupied coastal dune forests which have few of the large trees that are an important component of marten habitat in most other areas (Bull and Heater 2005, USFWS 2018, Slauson *et al.* 2019a). Furthermore, Moriarty *et al.* did find that large snags and logs were an important habitat component, and these are two of the four variables used to derive OGSI.

A key limitation of both the Slauson *et al.* (2019a, b) and Moriarty *et al.* (2019) coastal marten habitat models is that they are based strictly on recent occupancy and do not incorporate information about historical distribution or habitat quality. As our connectivity model's use of OGSI was largely based on the Slauson *et al.* habitat model, it also does not directly address habitat quality. In fact, there has been very little, if any, research done on how habitat conditions affect demographic rates in any North American marten species, although Thompson *et al.*'s (2012a) review found a broad preference for selecting mature forest habitats when available. Slauson *et al.* (2009) found that relatively large patches of old growth forest in California had more consistent occupancy by coastal martens over time than smaller patches or serpentine sites, and were more likely to be occupied by females and therefore support breeding activity, although this is based on a limited amount of data. Furthermore, large areas of the coastal marten's historical range would historically have been dominated by old growth forests, much of which would be classified as OGSI 200 (Davis *et al.* 2015), even though habitat cores predominately composed of timber this old are rare today (Fig. 17, Appendix 3). We believe that, at the range-wide landscape-scale of our model, OGSI is a reasonable estimator of coastal marten habitat quality in the absence of more rigorous empirical studies.

Two final drawbacks of using OGSI are the age of the data and the potential for the index to misclassify local site conditions. The most recent version of the GNN model is based on sampling plots and imagery from 2012 and is seven years old at the time of this writing (LEMMA 2014a). Therefore, it does not reflect disturbance events such as fires and timber harvest that have occurred since then, and some early to mid-seral forest may have matured into higher quality coastal marten habitat in the ensuing years. GNN also has a modest rate of misclassification of pixels (LEMMA 2014b-e), and while it does a good job of describing patterns of forest structure at a landscape scale (i.e. several km^2), it is more likely to contain errors at finer scales (Ohmann and Gregory 2002, Ohmann *et al.* 2012). For both of these reasons, caution should be used when scaling down the results of our model to evaluating local "site level" activities, although we do not expect either factor would dramatically alter our assessment of habitat core and linkage locations in the absence of a significant post-2012 landscape-scale disturbance event.

Serpentine soils

Our treatment of coastal marten habitat on serpentine soils probably involved even more simplifying assumptions than our treatment of OGSI. Based on published descriptions, we treated serpentine areas within 30km of the coast that hosted ecological communities prominently featuring ericaceous shrubs as usable but sub-optimal habitat for coastal martens (see Methods for details). While our estimated area of potentially suitable habitat on serpentine soils only extends over about 1.2% of the coastal marten's historical range, there are large areas of it affecting existing populations as well as habitat cores and corridors linked to these populations. Therefore, we felt it was important to incorporate serpentine soils into our model. We suspect that our approach likely overestimated how much "good" serpentine habitat was on the landscape because it assumed that all sites with potentially suitable community composition were equally usable by martens, which is likely not the case. We did not directly assess shrub cover in part because we lacked suitable data. However, high shrub cover is a vital habitat component at serpentine sites that have few large trees and snags (Slauson *et al.* 2019a, b). Therefore, we clipped the serpentine soils layer to within 30km of the coast to simulate the association of marine fog and dense shrub cover in these areas (Slauson 2003, Slauson *et al.* 2019a). Also, some areas are currently recovering from fires (such as the 2002 Biscuit Fire and the 2018 Klondike Fire in Oregon), which can be a prolonged process on these low productivity sites (Safford and Harrison 2008). There is still more to be learned about how coastal martens use serpentine habitat, in terms of within established home ranges as well as dispersal.

Rivers

In general, we modeled rivers as single-pixel wide linear barriers that increased in resistance as they gathered tributaries and descended through the watershed. (The exception to this width occurred in certain estuary areas, see Methods for details.) Obviously, most rivers have significant variation at fine scales in characteristics such as width, depth, flow rate, water temperature, the presence of sandbars and large woody debris, and other things that could affect the ability of a coastal marten to cross them successfully. Furthermore, the presence or magnitude of such features can vary considerably through time (both within and between years). We incorporated a number of simplifying assumptions into our connectivity model because we lacked data with sufficient resolution to map such variability at the scale of the coastal marten's historical range, as well as a rigorous understanding of their potential effects on the species' movements. It is likely that many rivers have viable crossing points in relatively consistent locations at least in certain times of the year where it would be appropriate to model lower resistance. It would be very powerful to have a good understanding of these potential crossing points and incorporate them into a habitat connectivity model for the coastal marten (or any other terrestrial species). Conversely, the lack of such information in a connectivity model means that the places where least-cost corridors cross rivers is based primarily on the resistance of the habitat on either side, whether the crossing point is one that is realistically usable by martens or not. In some cases, this could result in the legitimacy of the entire corridor being compromised. Attempting to more realistically model fine-scale variability in rivers would be very useful in landscape-scale conservation planning exercises such as this one, but the data acquisition and model parameterization challenges are such that the problem is probably best approached at a local scale (for example, a set of reaches on the lower Klamath River.)

Roads

As with rivers, we modeled roads as single-pixel wide barriers. Resistance increased with the width and amount of traffic implied in the road category descriptions. This simplifying assumption likely obscured areas where crossing a road may be easier or more difficult than was reflected in our modeled resistance values. For example, Clevenger *et al.* (2001) found that American martens in Alberta used culverts to cross the busy Trans-Canada highway, preferentially using culverts with low clearance and good through-visibility. Conversely, it has often been suggested that concrete median barriers (such as Jersey barriers) may impede animals from crossing otherwise passable roadways, or serve to trap them in the roadway and increase their risk of being struck by a vehicle (Clevenger and Kociolek 2006, 2013). We are aware of such barriers being present on sections of US Hwy. 101 where the corridor linking the Six Rivers National Forest with Prairie Creek Redwoods State Park crosses it (Linkage ID #50), and they are likely distributed along other roadways within the coastal marten's range. Clevenger and Kociolek (2013) reviewed the risk of different types of median barriers to a variety of wildlife species, and found that concrete barriers represented a moderate to high risk of mortality and of blocking the movement of mammals the size of a marten (the variation depending on whether or not gaps were provided between sections of barrier). As with sites where crossing rivers is potentially easier than usual, identifying sites where martens could more easily cross major roads would also improve our ability to model and plan for their habitat connectivity. Likewise, identifying roads that have features making them more difficult to cross could alert us to corridors that are less valuable than they otherwise appear or that might funnel animals into dangerous road crossings. Obtaining and analyzing data from the California and Oregon Departments of Transportation on the locations of suitable culverts, median barriers, and other fine-scale road features that may influence marten movements within habitat cores and linkages could be a useful future project. Such an analysis could also serve to guide the placement of suitable wildlife crossing structures in areas where there is a significant risk of road mortality and few safe crossings exist (Kintsch and Cramer 2011).

We modeled roads as simple linear features in the landscape. However, Pacific martens have been found to be less likely to use habitat with a high density of roads, including abandoned and unmaintained "ghost roads" (Wasserman *et al.* 2012). Road density has often been used as a surrogate for human activity, can be a powerful predictor of habitat fragmentation impacts on wildlife (Watts *et al.* 2007, Girvetz *et al.* 2008, Ahmed *et al.* 2014, D'Amico *et al.* 2015). Furthermore, the coastal marten's risk of falling prey to bobcats, one of the species' most significant predators, appears greatest in areas of early seral forest near roads (Slauson *et al.* 2019a), indicating that a higher density of roads may be associated with higher mortality. A more complex approach to modeling the effects of roads on coastal marten habitat use could utilize a moving window to calculate a road density around each pixel (for example, linear km of roads within 1km^2), which could then be factored into the estimates of habitat value and resistance. This might be especially useful if the goal is to model areas with a high density of narrow, low traffic logging roads (which were not generally differentiated from adjacent timber harvest areas in our modeling approach) as having a higher risk to movement and lower habitat value to coastal martens. Further empirical research would be helpful in parameterizing such a model and justifying the added complexity.

Coastal Marten Habitat Knowledge Gaps

While we have already pointed out a number of cases where improved data or further empirical studies would improve our ability to understand and model the coastal marten's habitat connectivity needs, there are several aspects of the species' habitat use and dispersal behavior where more research would be particularly valuable. These include having a better understanding of how dispersal habitat might differ from habitat used within a home range, as well as how dispersing animals use or avoid different land cover types and how they react to barriers such as roads and rivers.

Telemetry studies would probably provide the data most directly relevant to filling some of these knowledge gaps. Tracking martens and related species such as fishers by attaching or implanting animals with VHF radio or Global Positioning System (GPS) antennae is a commonly used research technique that has been employed to study a number of aspects of their biology (Thompson *et al.* 2012b). While studies of coastal marten home range habitat use have been conducted with this technique (Linnell *et al.* 2018, Slauson *et al.* 2019a), research specifically designed to quantify movement pathways through different land cover types (especially of dispersing individuals) could greatly improve the parameterization of the resistance surface used to estimate habitat connectivity. In the absence of such studies, we are forced to rely on inferences gathered from home range studies of coastal martens, or on landscape-use studies on related species that may be found in environments that are quite different from the coastal region of Oregon and northern California. Conducting successful telemetry studies of animals dispersing over large areas can be challenging, but there are a number of examples such work being used to inform the development of modeled resistance surfaces (e.g. Driezen *et al.* 2007, Cushman and Lewis 2010, Richard and Armstrong 2010, Trainor *et al.* 2013, Zeller *et al.* 2014 and 2018, Krishnamurthy *et al.* 2016).

Genetic analyses can be used as a tool for evaluating landscape connectivity patterns in a species over relatively long time periods (Kool *et al.* 2013, Wade *et al.* 2015). Several studies have examined landscape-scale genetic structure of populations of various marten species to infer how various landscape features facilitate or hinder gene flow (Broquet *et al.* 2006, Wasserman *et al.* 2010, Koen *et al.* 2012, Ruiz-Gonzalez *et al.* 2014), which by extension can be used to derive or evaluate a resistance surface. Genetic analyses have some potential advantages over telemetry-based studies for resistance estimation: (1) they may be less intrusive and not require capturing animals if the genetic material is obtained from sources such as scat samples, hair snares, museum specimens, or road killed individuals (Wade *et al.* 2015); (2) they may avoid the logistic and financial costs associated with tracking animals (Rudnick *et al.* 2012); and (3) they may capture more information about long-term patterns of movement across the landscape than direct observation of a relatively small number of marked animals (Rudnick *et al.* 2012). However, there will be a lag time of several generations from the reduction or loss of connectivity between populations and the resulting loss of gene flow being detectable by genetic analysis (Landguth *et al.* 2010). Given that the generation time for the marten is about 5 years (Slauson *et al.* 2019a), that timeframe for this species could be expected to be 15-25 years. Furthermore, species that have undergone past "population bottlenecks" where they were greatly reduced in size may show genetic similarity regardless of the amount of gene flow between existing populations (Rudnick *et al.* 2012).

Management Planning Scenarios

The quantitative output provided by Linkage Mapper has great potential for giving relatively simple, easily understood estimates of how land management actions may affect habitat connectivity. We compared LCP lengths or CWDs between the pre-harvest/post-harvest and unrestored/restored scenarios. Other metrics provided by Linkage Mapper that could also be used in such analyses include Euclidean distance between habitat cores, and the CWD:Euclidean distance and CWD:LCP ratios. Analyses could also be done on changes to the habitat cores themselves, examining estimated changes either to core size or to a variety of other patch-based descriptive metrics that have been developed by landscape ecologists (Wang *et al.* 2014, Turner and Gardner 2015). These could be used for a variety of pre- or post-hoc analyses in regards to landscape scale changes to marten habitat, such as timber harvest, wildfire, habitat restoration, or development projects.

The timber harvest scenario served to further improve our understanding of an important least-cost corridor by updating the OGSI data for this portion of the landscape to better reflect conditions in 2016 (as opposed to

2012, the year the most recent GNN data used in the primary model were obtained). It showed that the general location of the least-cost corridor did not change significantly even after a number of areas of mature forest that the original LCP (based on data from 2012) passed through were harvested. However, the location of the LCP within the corridor shifted to the south and overall connectivity was somewhat degraded.

The habitat restoration scenario illustrated a potential method for examining a large area of landscape and identifying discrete areas that might be priorities for habitat restoration aimed at improving connectivity. By simply improving the habitat value of all pixels by 1.0, a new habitat core was formed that significantly improved connectivity in a part of the landscape that holds an important linkage between the existing populations in Oregon and California. The habitat restoration scenario was not necessarily intended to simulate any specific management actions beyond modeling plausible rates of forest growth, and in many cases, "habitat restoration" might largely entail allowing time for younger forest stands to grow into age classes where they provide high quality habitat for coastal martens. Another approach could be working with restoration practitioners and experts on the marten to assess how specific management actions that are being contemplated might affect the habitat and resistance values. Different alternatives could be considered (both in terms of the management actions themselves and the estimates of the magnitude of their effects), and quantitative connectivity metrics used to assess the implications of the proposed actions.

Potential Future Improvements to the Coastal Marten Habitat Connectivity Model

Any ecological model can be improved and refined, and during the course of developing the coastal marten habitat connectivity model we have identified a number of areas where future efforts could potentially be made to update it, improve the input data, explore additional methods of assessing connectivity, refine the assumptions, or enhance its utility as a planning tool.

Input data

The most important data layer in our model is the OGSI. This is an output component of the GNN model (LEMMA 2014a), and as was mentioned above, the most recent available version is based on field data collected in 2012. An updated version of GNN based on data collected in 2016 is scheduled to be released in mid- to late 2019 (R. Davis *pers. comm.*) Once these data are available to us, it should be relatively straightforward to create an updated habitat value map and resistance surface and re-run Core Mapper and Linkage Mapper to generate new habitat cores and least-cost corridors. This version of the model would include the effects of changes on the landscape such as from wildfires, timber harvest, or forest regrowth through 2016.

A dense layer of ericaceous shrubs is an important habitat component for the coastal marten at all locations where they are known to occur (USFWS 2018, Slauson *et al.* 2019a). We did not directly incorporate shrub cover into our model, partly because we assumed it would covary to some extent with OGSI, and partly because we lacked a landscape-scale data layer for shrub cover of the specific species preferred by the marten that covered its entire historical distribution. GNN does include a shrub cover data layer, but it does not distinguish suitable shrub communities from unsuitable types such as chaparral. It is likely that finding an effective way to incorporate such shrub cover data as exist into our estimation of per pixel habitat value and resistance to marten movement could improve the predictive value of the model beyond basing these largely on OGSI and coastal serpentine soils. One approach could be to first map out vegetation communities likely to include a significant component of ericaceous shrubs using GAP ESLF or another landscape scale vegetation mapping data set, and then somehow combining the shrub cover value within the suitable communities with OGSI or other forest structure data to derive a single per-pixel habitat value usable by Linkage Mapper. Given that there is still a paucity of empirical data on coastal marten habitat quality and on how the species disperses through different habitat types, it would be preferable to solicit expert opinion on how best to parameterize such a model (Drew and Collazo 2012, Aylward *et al.* 2018).

There are a number of potential ways to incorporate into the model more of the complexity of rivers and roads as linear obstacles to marten movement. Several of these are discussed in more detail above (see PP. 63-64), such as adding details on the locations of culverts and Jersey barriers for roads or incorporating data on river depth, flow rate, temperature, etc. More explicit data on traffic levels and speed limits may be available from state or federal highway agencies for at least some roads, and these could be used to improve the realism of the resistance surface, which incorporated these factors only implicitly based on the road category descriptions in the OpenStreetMap dataset (OpenStreetMap contributors 2018). Buffering larger roads to increase the resistance of pixels a certain distance away from the road itself may also add realism to the model. Forman and Alexander (1998) found that many wildlife species avoided even approaching major highways, likely because of the noise and visual disturbance from traffic. This effect of the roadway extended between 100 and 1000m depending on the species and habitat (Forman and Alexander 1998, Forman and Deblinger 2000). American martens have been found to avoid major highways (Alexander and Waters 2000) and areas with high densities of smaller roads (Robitaille and Aubry 2000, Wasserman *et al.* 2012). Broadening the area of the landscape impacted by the presence of roads away from the actual roadbed itself may produce a more realistic depiction of the location and quality of least-cost corridors, and some other landscape connectivity studies have taken a similar approach (Li *et al.* 2004, Liu *et al.* 2014). However, our assessment indicated that the availability of data for smaller roads (logging roads, etc. that we did not include in our roads layer) was inconsistent, potentially making such an analysis somewhat challenging. Alexander and Waters (2000) also found that American martens in Alberta were more likely to cross the Trans-Canada highway at flatter sites and avoided crossing at sites with >5° of topographic slope, and this could also potentially be incorporated into the treatment of roads in the resistance surface of a marten connectivity model.

Other Linkage Mapper tools

In addition to the Core Mapper (Shirk and McRae 2015) and Resistance and Habitat Calculator (McRae *et al.* 2015) spatial analysis tools that we used in developing our model, Linkage Mapper also includes additional spatial analysis tools that could be used in assessing aspects of coastal marten habitat connectivity. These include Pinchpoint Mapper (McRae 2012a), Barrier Mapper (McRae 2012b), and Centrality Mapper (McRae 2012c). Pinchpoint Mapper identifies constrictions in least-cost corridors where the loss of a small amount of habitat could disproportionately affect connectivity. Barrier Mapper identifies discrete areas of high resistance that limit the connectivity value of a corridor. Centrality Mapper analyzes the entire linkage network and identifies the corridors and habitat cores that are most important for maintaining its overall connectivity. We found that many of these outputs could be readily surmised from visually inspecting the Linkage Mapper output of LCPs and corridors in combination with the resistance surface. However, attempting to produce these results in this manner over the entire landscape or a large fraction of it would be quite tedious, and would probably also be less complete than if these tools were used for the task.

Examining alternative assumptions

The assumptions used in developing a connectivity model based on least-cost corridors can have significant implications for the output and final map, with different cost-weighting schemes or habitat models potentially leading to very different corridor maps (Beier *et al.* 2009, Sawyer *et al.* 2011). As was discussed above (see P. 65), amending the resistance surface in our model by incorporating data from new studies on population genetic structure or habitat use in coastal martens would ultimately yield the most significant improvements. Improved habitat-specific data on mortality rates would also help in this regard, especially further insights into the relationship between bobcat predation and the amount and pattern of younger forest habitat on the landscape (Wengert 2013, Slauson *et al.* 2019a). However, amendments could be made to our current connectivity model based on existing hypotheses, and doing so could produce new versions of the modeled corridors and habitat cores that may be of interest to planners in comparing to those we derived largely from OGSI and coastal serpentine habitat. Moriarty *et al.* (2019) studied coastal marten habitat at cites associated with the Southern

Coastal Oregon and Central Coastal Oregon populations and found that detections in those areas were better modeled by vegetation cover (shrub cover or shrub + canopy cover combined) than by forest characteristics more specifically associated with old growth forest such as presence of large trees. These findings conflict somewhat with those of Slauson *et al.* (2019a, b) who found marten detections were best modeled by the OGSI, serpentine soils, precipitation, and elevation. This model was the basis for the habitat value parameterization we used to assess connectivity, although as mentioned in the Methods, we did not incorporate precipitation or elevation in order to capture potential habitat cores in areas where Slauson *et al.* showed little suitable habitat present, particularly in coastal redwood forests. It should be possible to produce alternative habitat core and least-cost corridor maps based on the Moriarty *et al.* or other habitat models for the marten, or to add other "hand drawn" habitat cores around areas where isolated marten detections have been made outside of the known populations. The results for the different models could be compared, although in the absence of new empirical data on marten dispersal it would probably not currently be possible to rigorously evaluate the performance of these models relative to one another.

Conservation planning

While we have provided examples of some ways in which the coastal marten habitat connectivity model could be used for conservation planning purposes, there are a number of other applications to which it could be put, either with its existing structure and data inputs or with amendments.

While the GNN model approximates vegetation conditions across the Pacific Northwest, it is intended to be used at a landscape scale and has the potential for classification errors at finer scales (Ohmann and Gregory 2002). Comparing this model output to additional finer-scale data on habitat quality and conditions within individual corridors or habitat cores may be desirable for some conservation planning purposes. LiDAR data have been used for assessing habitat conditions for American martens in studies of habitat connectivity (Guo *et al.* 2018) and mortality risk (Joyce *et al.* 2018). More traditional ground-based vegetation sampling could also be used for habitat assessment at the stand or project scale. Such data could be used in several contexts, such as identifying particularly important areas of habitat within cores or corridors, providing a more up-to-date depiction of habitat conditions than the output from the GNN model, or assessing the effects of a disturbance event such as a wildfire on a corridor or habitat core that has already been incorporated into a conservation plan. Ground truthing, practitioner expertise, and the identification of features that may affect marten habitat use that were not included in this model (e.g. sources of high levels of artificial noise or lighting) could also be integrated into finer-scale planning efforts.

A potential extension of our coastal marten habitat connectivity model would be to simulate the landscape as dynamic rather than as a static snapshot in time. Planning decisions regarding the conservation of habitat patches or corridors can have implications for the landscape over a period of many years, and it may be useful to try to estimate how a planned network of habitat linkages may function well into the future. It should be possible to model not just the current landscape, but also make estimates about habitat conditions some years in the future incorporating the effects of forest regrowth or planned timber harvests. This could be done by modifying OGSI values to create alternative habitat value and resistance surface rasters for a set of time steps into the future (for example, one every decade for 50 years). If more detailed data on the relationship between the amount and configuration of early successional habitat and the risk of bobcat predation in coastal martens becomes available, this sort of dynamic landscape modeling could be particularly useful in the long-term planning of timber harvest activities.

It will also be desirable to examine a potential resilience of a habitat linkage network or set of alternative networks to wildfire and climate change. Overlaying a landscape-scale data model of fuels and habitat conditions such as are available in LANDFIRE (USDA 2014) on a set of habitat cores and corridors could be used to assess the relative risk of large-scale high intensity wildfires in the network. It would also allow for incorporating redundancy of habitat and populations into land management and species recovery planning. In the longer term, there is the potential for considerable changes in vegetation communities and forest conditions as the climate continues to warm (Halofsky *et al.* 2013, Thorne *et al.* 2016). However these effects will probably be more pronounced in some parts of the landscape than others, with the potential for localized "climate refugia" in areas where conditions may change more slowly or remain stable such as mature forests on north facing slopes or in canyon bottoms (Olson *et al.* 2012). Assessing a coastal marten habitat linkage network through the lens of areas that are likely to be climate resilient would also be a prudent planning measure to help maintain connectivity for the species over the long term. There are several spatially explicit climate models currently available that could be used in this regard (Olson *et al.* 2012, Flint *et al.* 2013, Buttrick *et al.* 2015, McRae *et al.* 2016), with others likely to be developed in the future.

Our model was intended to facilitate conservation planning aimed at maintaining habitat connectivity for the coastal marten, but there are many other species in the region that also depend on a connected landscape for population resilience. While the marten may serve as a connectivity "umbrella species" (Heller and Zavaleta 2009, Breckheimer *et al.* 2014) for organisms with similar habitat needs and dispersal capabilities, there are certainly others for which our general modeling approach would have produced a very different map of habitat cores and corridors. One means of enhancing the conservation value of the coastal marten model would be to use it as a building block in a broader effort to assess landscape connectivity for a suite of focal species. These should be carefully chosen to meet a variety of habitat use and dispersal ability characteristics, with the goal of identifying a linkage network that could function to support a large fraction of the biodiversity in the region. This approach was encouraged for developing regional landscape connectivity maps to complement the statewide California Essential Habitat Connectivity Project (Spencer *et al.* 2010), and it has been used in a number of conservation planning efforts (e.g. WHCWG 2010, Penrod *et al.* 2013, Koen *et al.* 2014, Leonard *et al.* 2017) including recent projects using Linkage Mapper (Liu *et al.* 2018, Gallo *et al.* 2019). Expanding the scope of habitat connectivity modeling efforts in the region to include other species could provide insights into areas on the landscape that are broadly important for connectivity for a range of species, and further identifying such areas as broadly resilient to climate change and/or wildfire risk would provide even more value.

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Appendix 1

Complete list of land cover types incorporated into the modeled landscape for the coastal marten connectivity model. Table A1.1 groups cover types by category and provides the assigned resistance value for each. When a single pixel had a non-zero resistance value in more than one data layer, we applied the highest value to the final resistance surface. Table A1.2 lists the land cover types incorporated in the serpentine data layer as potentially including suitable coastal marten habitat.

Table A1.1. Resistance values assigned to all land cover types.

Land cover type	Resistance value
GNN OGSI value	
0 – 10	10
11 - 20	9
21 – 30	6
31 – 40	3
41 – 50	2
51 – 100	1
GNN OGSI value in selected habitat types in coastal serpentine areas (see Table A1.2 for habitat types)	
0 – 10	5
11 - 20	4.5
21 - 30	3
31 – 40	3
41 - 50	2
51 - 100	1
GAP ESLF non-forested habitats: 'Developed' functional group	
Developed, high intensity	1000
Developed, medium intensity	800
Developed, low intensity	150
Developed, open space	100
Orchards, Vineyards, and Other High Structure Agriculture	80
GAP ESLF non-forested habitats: 'Dune' functional group	
Mediterranean California northern coastal dune	30
North Pacific maritime coastal sand dune and strand	30
GAP ESLF non-forested habitats: 'Grassland' functional group	
California northern coastal grasslands	20
California Mesic Serpentine Grassland	20
Introduced upland vegetation - annual grassland	20
Mediterranean California Alpine Dry Tundra	20
Mediterranean California Serpentine Fen	20

Land cover type	Resistance value
North Pacific herbaceous bald and bluff	20
North Pacific montane grassland	20
Pasture/hay	20
Recently burned grassland	20
Temperate Pacific Montane Wet Meadow	20
Willamette Valley Upland Prairie and Savanna	20
Willamette Valley wet prairie	20
GAP ESLF non-forested habitats: 'Open' functional group	
Cultivated cropland	30
Central California Coast Ranges cliff and canyon	30
Klamath-Siskiyou cliff and outcrop	30
Mediterranean California Serpentine Barrens	30
North American Alpine Ice Field	100
North Pacific coastal cliff and bluff	30
North Pacific Montane Massive Bedrock, Cliff and Talus	30
GAP ESLF non-forested habitats: 'Shrub' functional group	
California xeric serpentine chaparral	15
North Pacific Bog and Fen	15
North Pacific hypermaritime shrub and herbaceous headland	15
Northern and central California dry-mesic chaparral	15
Northern California coastal scrub	15
North Pacific Montane Shrubland	15
Southern California coastal scrub	15
GAP ESLF non-forested habitats: 'Wet' functional group	
North Pacific Intertidal Freshwater Wetland	25
North Pacific Shrub Swamp	25
Temperate Pacific Freshwater Aquatic Bed	25
Temperate Pacific freshwater emergent marsh	25
Temperate Pacific tidal salt and brackish marsh	30
Estuary or Bay	20
Rivers -- Strahler stream order	
4	5
5	10
6	20
7	100
8	200
9	500
10	10000***

*** Only applied to the Columbia River, which also represented the northern boundary of the model landscape.

Land cover type	Resistance value
Lake, pond, or reservoir	500
OpenStreetMap major road classifications	
Motorway and Motorway link	150
Trunk and Trunk link	100
Primary and Primary link	25
Secondary and Secondary link	15
Tertiary and Tertiary link	10
Unclassified	3†††
Residential, Service, Track, and all others	0‡‡‡

††† Rural, low-traffic roads assigned a low resistance value to influence connectivity in areas with relatively undisturbed habitat but not unduly increase resistance in developed areas or areas dominated by low OGSI pixels.

‡‡‡ While we felt that these road categories likely had the potential to affect landscape connectivity for coastal martens, we ultimately opted not to assign them a specific resistance value because their effects seemed largely to be already captured and more simply dealt with by the OGSI and ESLF layers. Some other categories were not primarily intended to carry motor vehicle traffic such as bike paths or driveways.

Table A1.2. Gap ESLF habitat types (NatureServe 2017) that were used to modify the resistance and habitat value surfaces in areas of serpentine soils within 30km of the coast. Most of these types were represented by a relatively small number of pixels within the serpentine soil polygons (Fig. 4C, Appendix 2). See Methods for more details.

GAP ESLF Habitat Type
California Central Valley Mixed Oak Savanna
California Coastal Closed-Cone Conifer Forest and Woodland
California Coastal Live Oak Woodland and Savanna
California Coastal Redwood Forest
California Lower Montane Blue Oak-Foothill Pine Woodland and Savanna
California Mesic Chaparral
California Montane Jeffrey Pine-(Ponderosa Pine) Woodland
California Montane Woodland and Chaparral
Great Basin Foothill and Lower Montane Riparian Woodland and Shrubland
Harvested Forest - Northwestern Conifer Regeneration
Harvested Forest-Shrub Regeneration
Klamath-Siskiyou Lower Montane Serpentine Mixed Conifer Woodland
Klamath-Siskiyou Upper Montane Serpentine Mixed Conifer Woodland
Klamath-Siskiyou Xeromorphic Serpentine Savanna and Chaparral
Mediterranean California Dry-Mesic Mixed Conifer Forest and Woodland
Mediterranean California Foothill and Lower Montane Riparian Woodland
Mediterranean California Lower Montane Black Oak-Conifer Forest and Woodland
Mediterranean California Mesic Mixed Conifer Forest and Woodland
Mediterranean California Mesic Serpentine Woodland and Chaparral
Mediterranean California Mixed Evergreen Forest
Mediterranean California Mixed Oak Woodland
Mediterranean California Serpentine Foothill and Lower Montane Riparian Woodland and Seep
North Pacific Broadleaf Landslide Forest and Shrubland
North Pacific Dry Douglas-fir-(Madrone) Forest and Woodland
North Pacific Hypermaritime Sitka Spruce Forest
North Pacific Lowland Mixed Hardwood-Conifer Forest and Woodland
North Pacific Lowland Riparian Forest and Shrubland
North Pacific Maritime Dry-Mesic Douglas-fir-Western Hemlock Forest
North Pacific Maritime Mesic-Wet Douglas-fir-Western Hemlock Forest
North Pacific Oak Woodland
Recently Burned
Recently burned forest

Appendix 2

Supplemental maps showing the various components of the resistance surface. These are essentially the same as those shown in Fig. 4, but with zoomed-in panels to allow them to be viewed in greater detail.

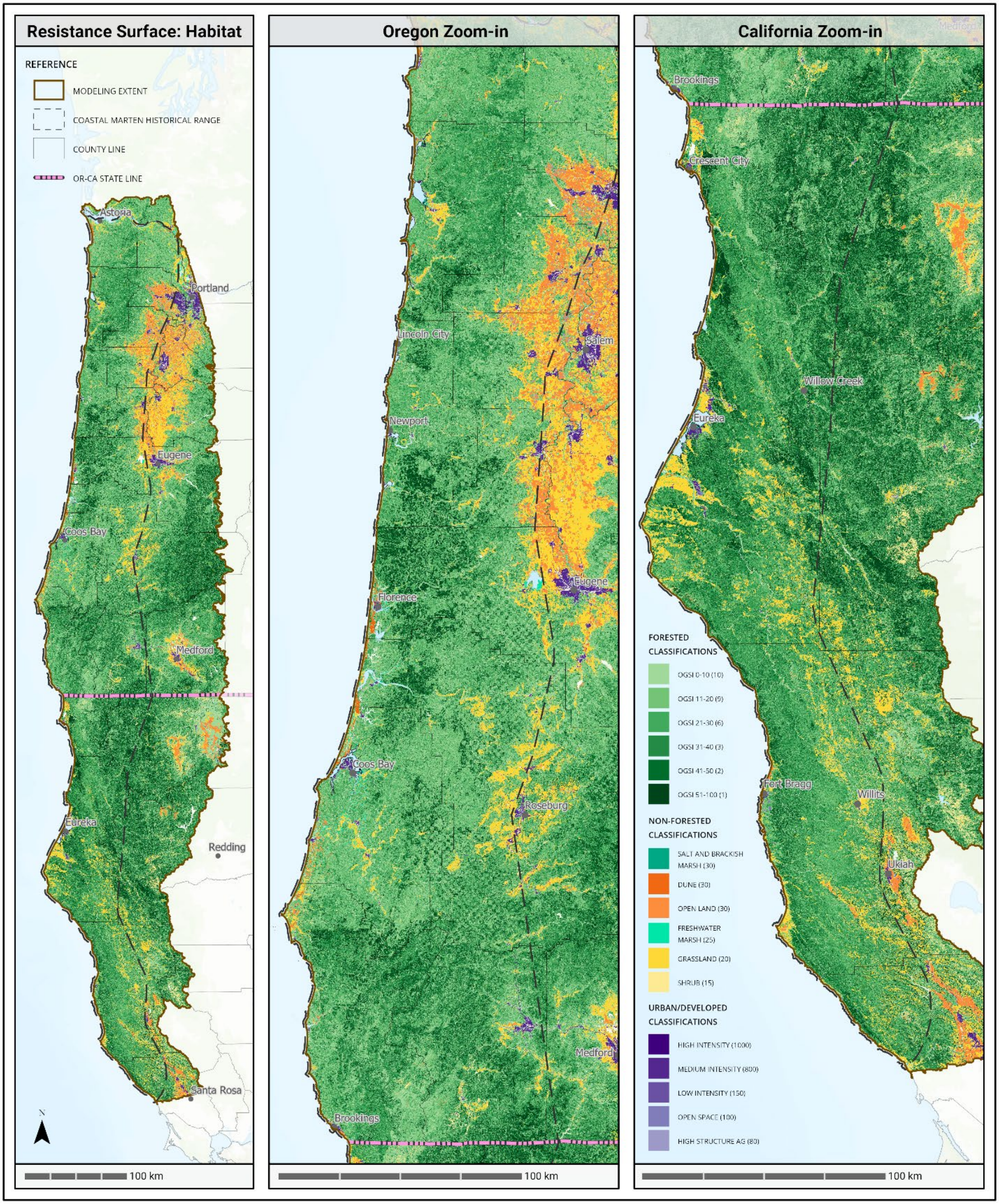

Figure A2.1. Resistance values based on habitat types derived from the GNN.

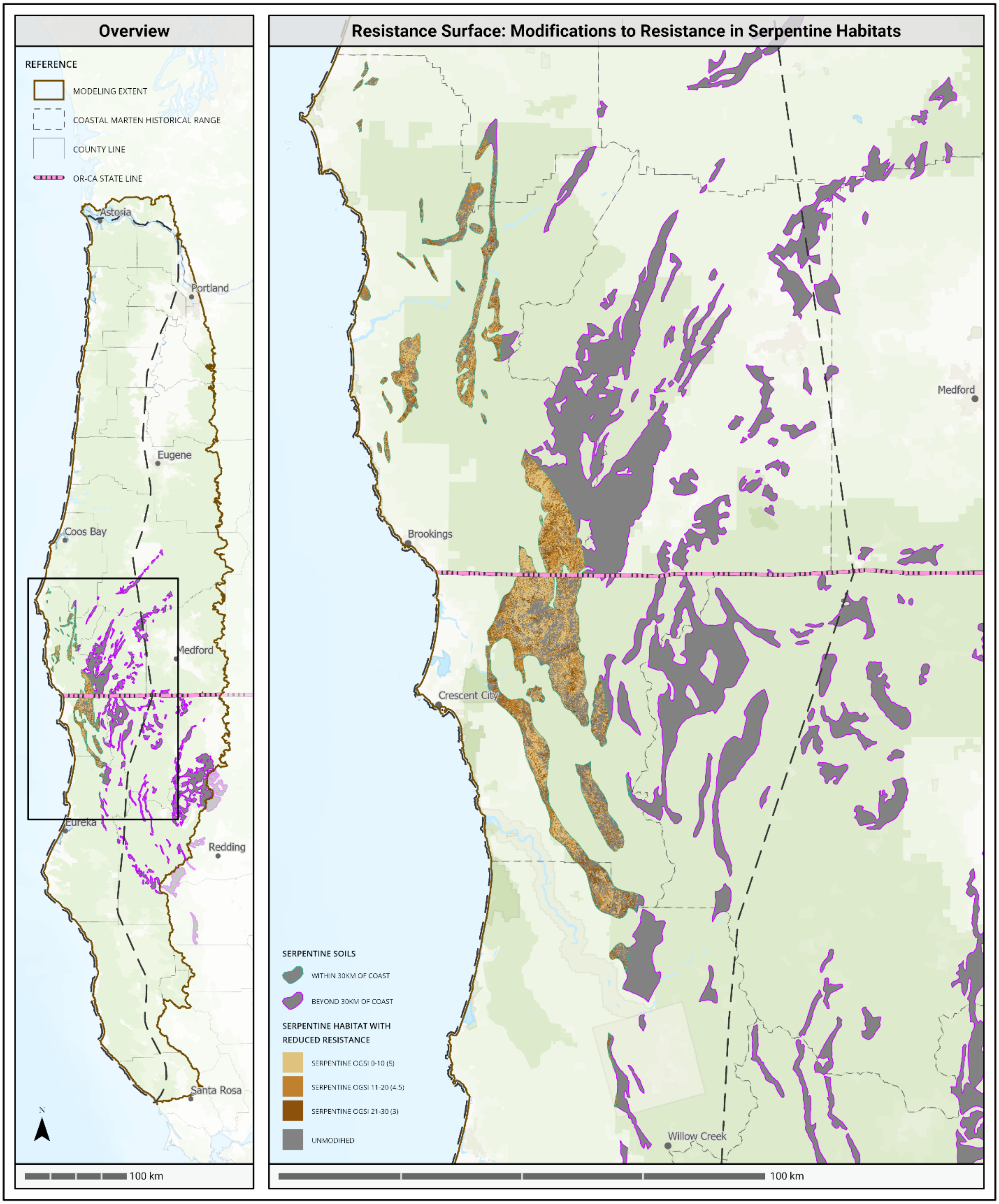

Figure A2.2. Resistance modifications based on potentially suitable habitat on serpentine soils within 30km of the coast.

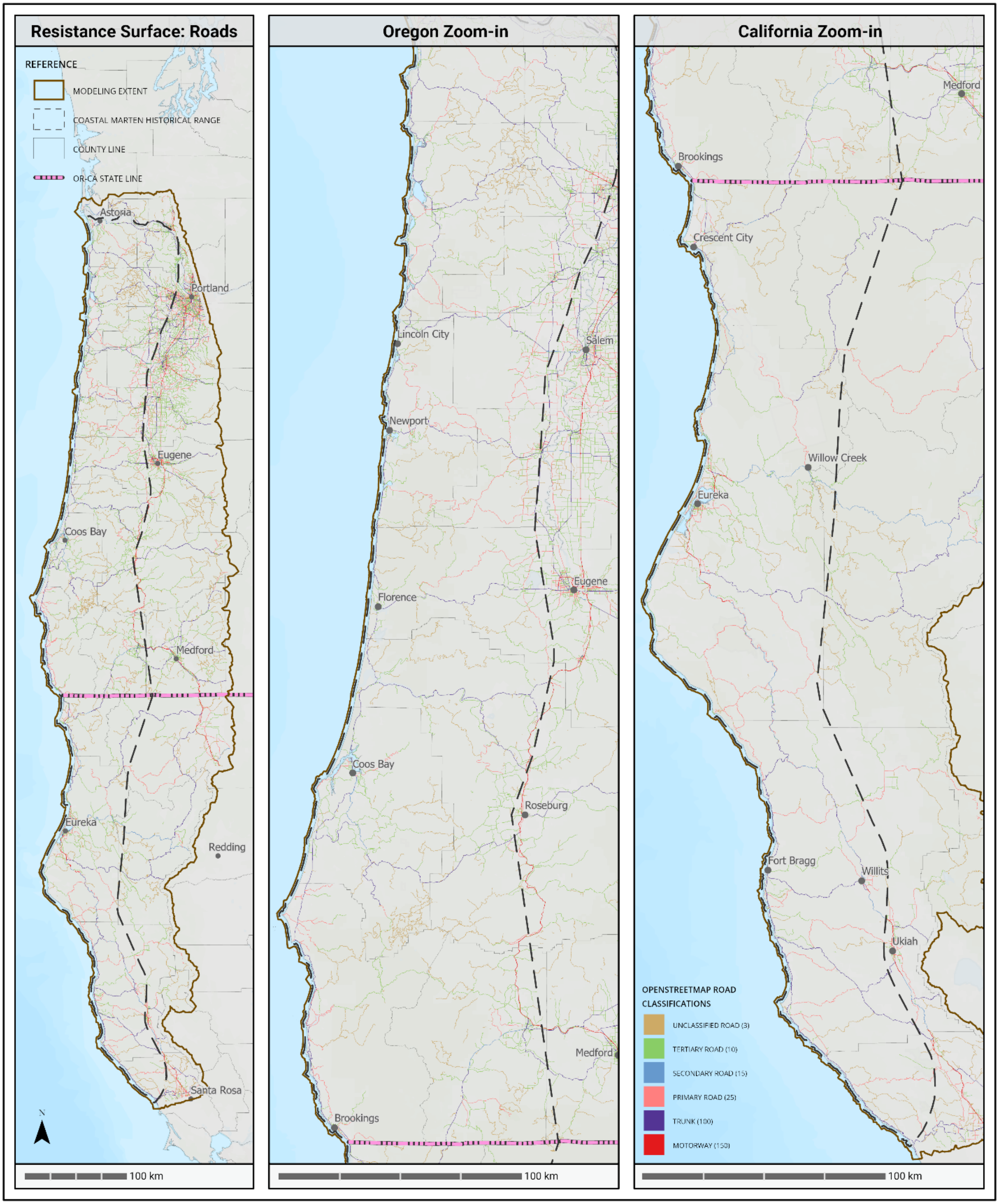

Figure A2.3. Resistance values based on roads.

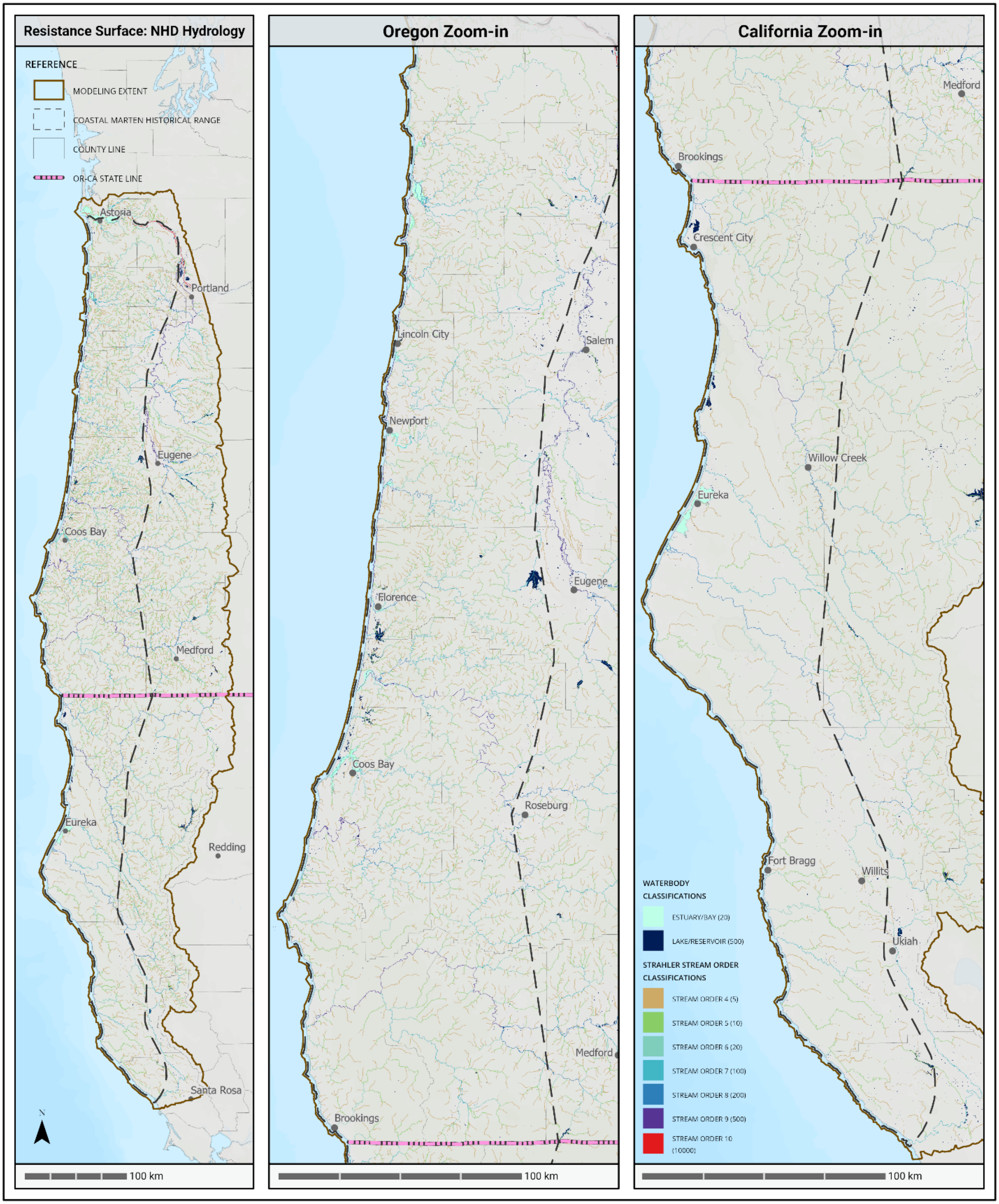

Figure A2.4. Resistance values based on rivers, lakes, reservoirs, and estuaries.

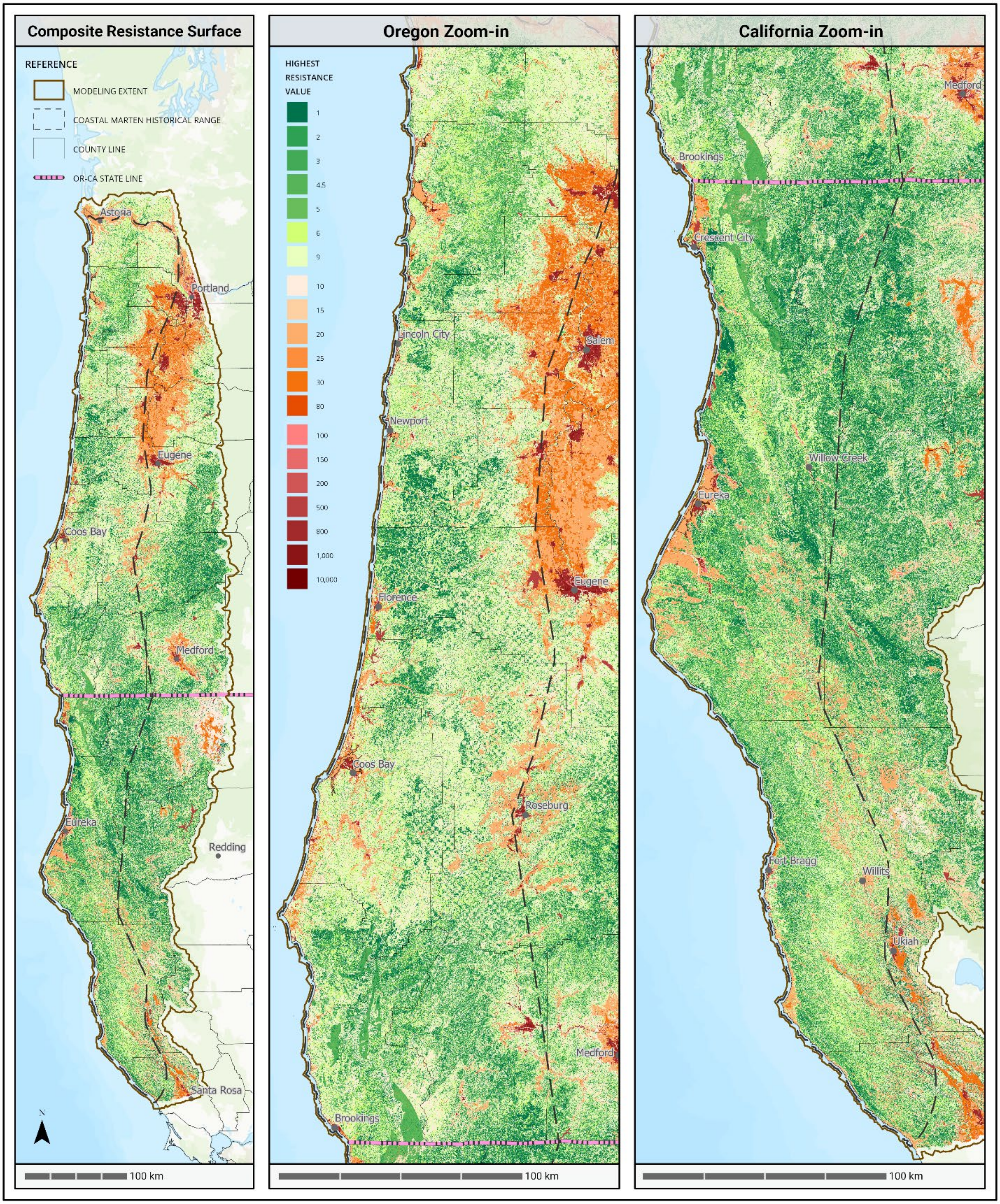

Figure A2.5. The final resistance surface. Pixels that had values assigned to them in more than one of the individual component layers were classified by the highest resistance value in the final resistance surface.

Appendix 3

Descriptive summary of sensitivity analyses. In the course of developing the coastal marten connectivity model, we explored the effects of a range of values for most of the input parameters used to delineate the habitat cores and least-cost corridors. Below we provide the values we examined for each parameter along with a qualitative summary of the results of varying each one. For each given sensitivity analysis trial, all other parameter values were kept the same as in our primary model unless otherwise specified. In some cases, we also provide additional tables, graphs, or maps to illustrate how varying the parameter changed the model output. We did not include maps here for every parameter variation we examined, both for space reasons and because the effects of many trials were fairly negligible. However, the authors invite interested parties to contact them with requests for maps of the outcome of specific sensitivity analyses not shown here.

In exploring how varying the inputs used by Core Mapper to delineate habitat cores could affect the output of the model, we conducted trials with alternate values for the moving window radius, minimum mean habitat value within the moving window, minimum habitat value for inclusion of a pixel in a core, and the degree of cost-weighted expansion of habitat cores after they are mapped. The results are summarized in the tables and figures below.

Table A3.1. Summary of results of sensitivity analyses conducted by adjusting the input parameters used to develop habitat cores for the coastal marten connectivity model. Values shown in brackets were those used in our primary model.

Parameter adjusted	Values explored	Qualitative description of results
Moving window radius (m)	564†, 736†, [977*], 1382*, 1693*, 1954*, 2111†, 2185*, and 2932† †Correspond to low end and high end mean home range size estimates for female and male Pacific martens, respectively (Moriarty *et al.* 2016 and references therein). *Correspond to the diameter of 1-5 average female home ranges (300ha), respectively, based on Slauson *et al.* 2019b.	Incrementally increasing the moving window size mostly resulted in modest changes to the boundaries of existing habitat cores. Smaller cores disappeared as the larger moving window captured too much adjacent low OGSI habitat for enough pixels to meet the mean habitat value to form a 1500ha core. The number of small cores lost increased as the window size increased, although the effect is already evident with a 1382m radius. Decreasing window size has the opposite effect, with most core boundaries changing a modest amount and some new smaller cores appearing. There were changes in the total area of habitat cores concordant with the changes in core numbers resulting from altering the moving window radius. **See Table A3.2 for specific results.**
Minimum mean habitat value within moving window	10, 11, 18-35, [36], 37-62	• Trials at 10 & 11 failed to run to completion, presumably because too much of the landscape mapped as habitat core. Based on the trend line in Fig. A3.2, we expect that >90% of the historical range would be included in habitat cores at these levels. • At 18 (the lowest value with any pixels classified as OGSI 80) 69.6% of the marten's historical range mapped as habitat core, mostly in one enormous core (Figs. A3.1, A3.3) • There was a notable decrease in habitat connectivity in the area between the South Coastal Oregon population and the habitat cores to the south when the mean habitat value shifts from 34 to 35 • Trials ≥57 only mapped three habitat cores, all in Redwood National and State Parks. For reference, the median habitat value of pixels within the historical range classified as OGSI 200 was 62 (Fig. A3.3). • **See Figs. A3.1, A3.2, and A3.3 for illustrations of these results.**
Minimum habitat value for a pixel to be included in a core	0, [1], 36	Minimum habitat value of 0 results in filling in small "holes" within habitat cores but leaves their outer boundaries almost unaffected. A minimum habitat value of 36 resulted in the disappearance of 15 smaller habitat cores and the appearance of a few new small "holes" in those remaining. The outer boundaries of the remaining cores were largely unchanged.

Parameter adjusted	Values explored	Qualitative description of results
Habitat core expansion (cost-weighted m)	0, [977], 1954, 4885	• Removing the expansion removes small “lobes” from some habitat cores, and in a few cases the shape of the core changed significantly. Numerous small “holes” appear in most cores, often concentrated along roads or rivers. • Increasing the expansion fills in additional small holes in the habitat cores but otherwise has little effect, presumably because of the “trimming back” step in Core Mapper.

Table A3.2. Results of sensitivity analysis of the size of the moving window used in mapping habitat cores. Our primary model used a radius of 977m, which approximates the size of an average female marten home range. Net changes in the number of habitat cores and the percent total area of all habitat core are reported relative to the output of the primary model.

Moving window radius (m)	Net change in number of habitat cores	Change in total area of all habitat cores (%)
564	+10	+1.3
736	+5	+0.6
[977]	0	0
1382	-4	-1.1
1693	-6	-2.8
1954	-14	-4.6
2111	-15	-5.6
2185	-17	-6.7
2932	-25	-10.2

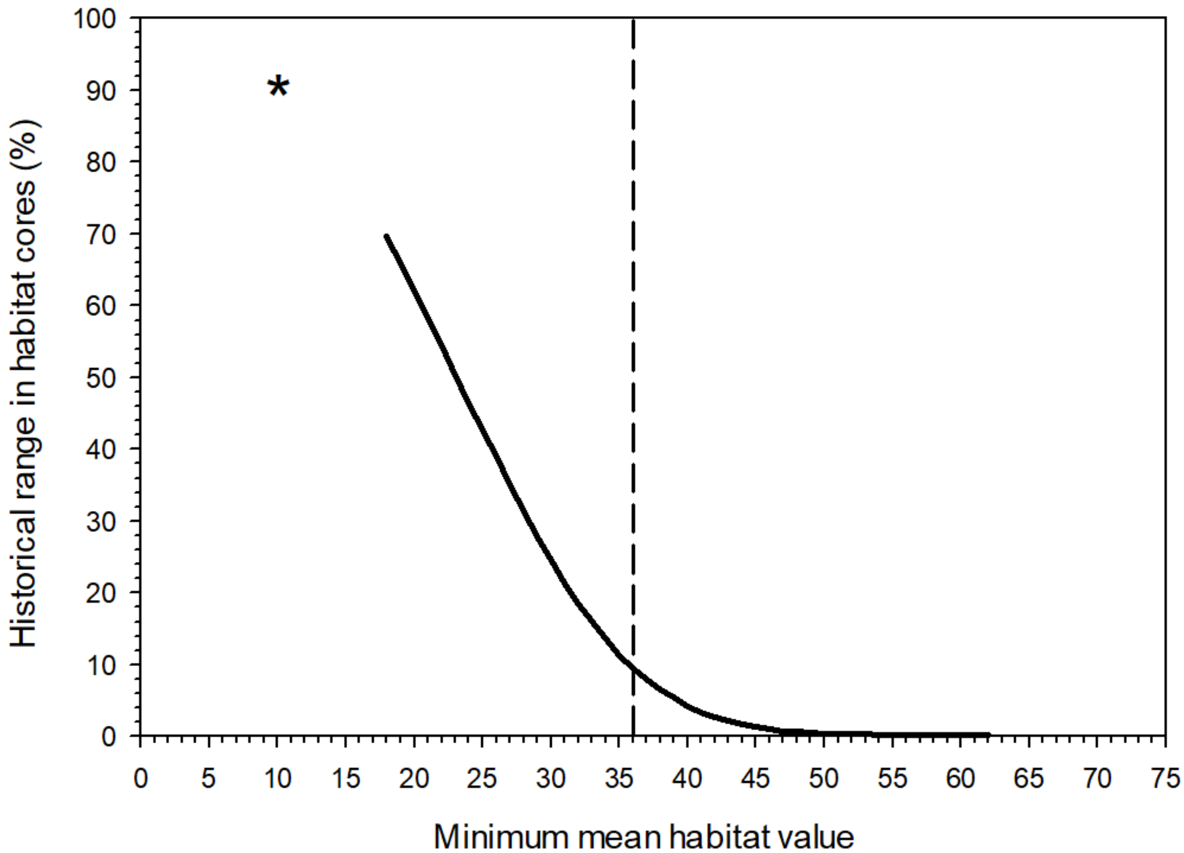

Figure A3.1. Effects of varying the minimum mean habitat value used in mapping habitat cores on the percentage of the coastal marten's historical range that was mapped as being within a habitat core. The vertical dashed line shows the habitat value used in our primary model (36.0). We attempted to run trials with the minimum mean value set to 10 and 11, but these failed to complete, probably because too much of the total landscape was mapping as habitat core. Based on the trend line, we expect these trials would have resulted in >90% of the historical range mapping as habitat core (approximated by the asterisk).

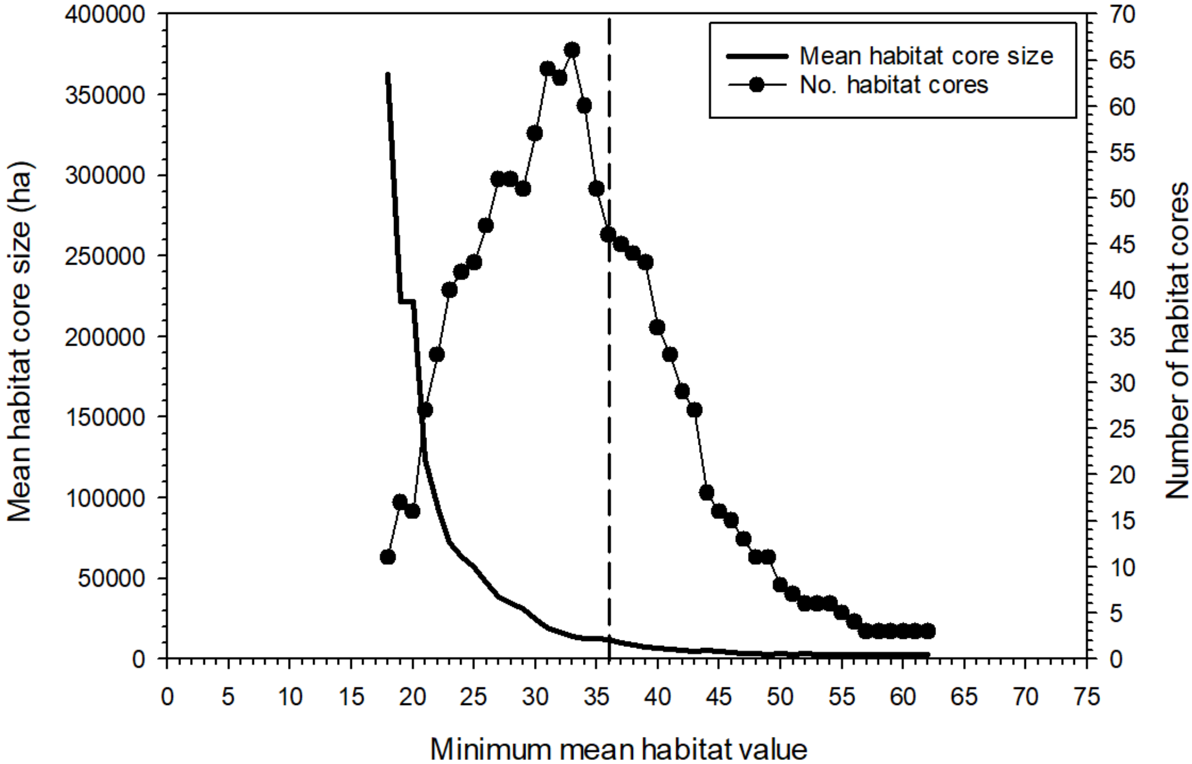

Figure A3.2. Effects of varying the minimum mean habitat value used in mapping habitat cores on mean habitat core size and the total number of habitat cores. The vertical dashed line shows the habitat value used in our primary model (36.0). At lower minimum mean habitat values, much of the landscape is classified as being within a habitat core, resulting in a smaller number of large cores and a greater mean core size. Increasing the minimum mean habitat value from this lower end results in the fragmentation of these very large cores, with an increasing number of cores and decreasing mean core size. Above a habitat value of 33.0, the number of habitat cores begins to decline again while the mean core size continues to decrease. This trend results from essentially defining cores with increasingly higher value habitat (based on OGSI). Fewer and fewer concentrations of suitably high value pixels exist as the minimum mean habitat value is increased, until at values >57.0 there are only 3 modest size habitat cores remaining, all in Redwood National and State Parks.

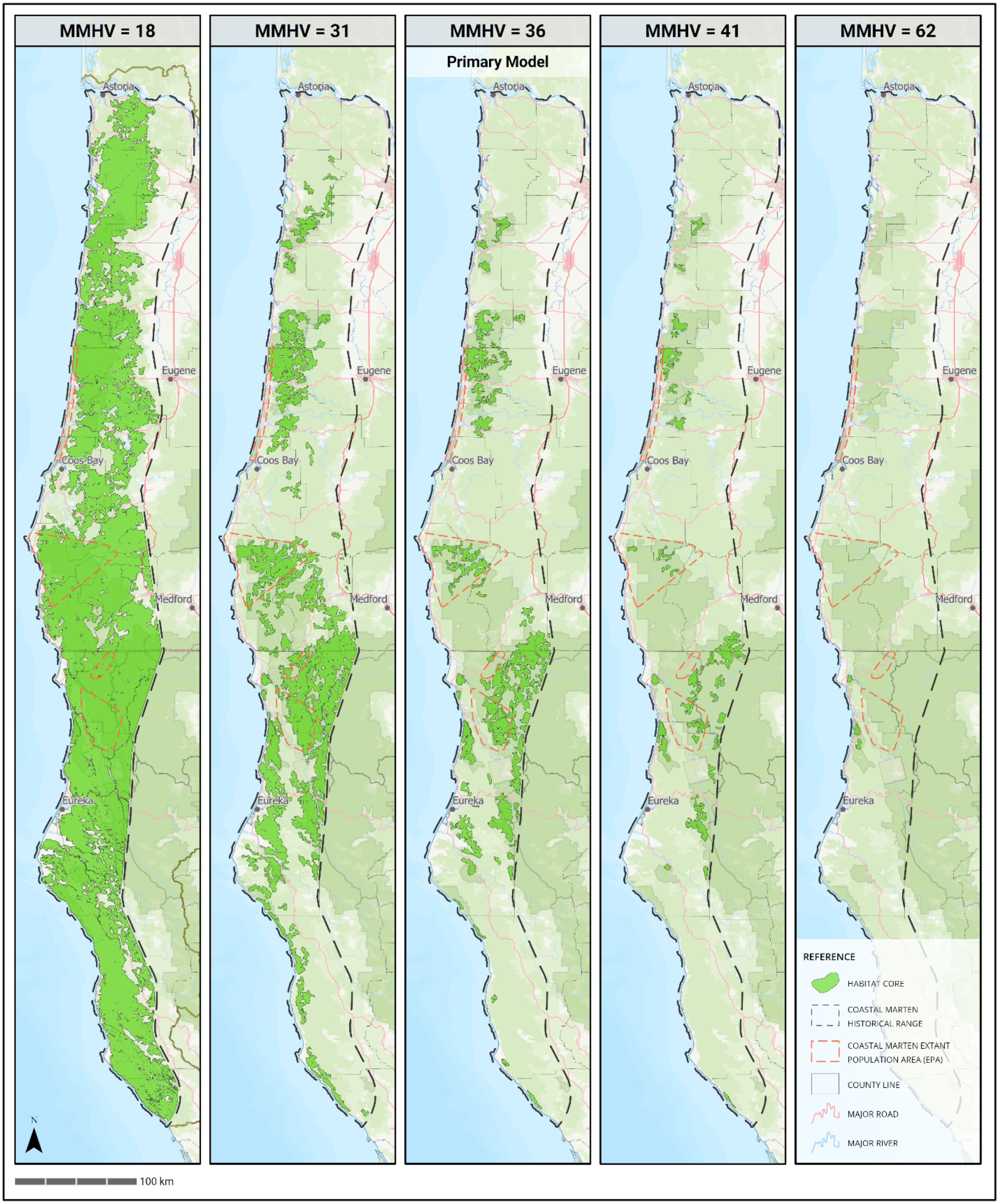

Figure A3.3. Examples of habitat cores produced in the sensitivity analysis for minimum mean habitat value (MMHV) used in the moving window. These five maps illustrate the range of results from the values that we tested. At lower MMHV, most of the landscape is dominated by a few very large habitat cores. As MMHV increases, these large cores fragment into smaller cores; the number of cores increases and the total area of cores decreases. The number of cores peaks at MMHV=33 (see Fig. A3.2) and then begins to decline as less and less area is suitable for delineating 1500ha cores. The median habitat value of pixels in the coastal marten's historical range classified as OGSI200 is 62, but at this MMHV only three small habitat cores are mapped, all in Redwood National and State Parks. In historical times such habitat would have been much more extensive on the landscape.

In exploring how varying the inputs used by Linkage Mapper to create least-cost corridors could affect the output of the model, we conducted trials where we removed some elements of the resistance surface in different combinations (roads, rivers, and serpentine habitat), and some where we reduced the resistance of certain habitat types (serpentine or lower values of OGSI). The results are summarized in the tables and figures below.

Table A3.3. Summary of results of sensitivity analyses conducted by running Linkage Mapper on amended resistance surfaces. All of these models used the same set of habitat cores as the primary model. Here we report linkages that were lost or added compared to the primary model because the changes to the resistance surface affected the calculation of nearest neighbor habitat pairs. Assessments of "connectedness" of new linkages are based on CWD. We also report or summarize where the LCP of a linkage deviated from that of the primary model by >1km.

Change to resistance surface used in primary model	Qualitative description of results
No roads	• Link ID #s 40 and 46 dropped from the model • "Moderately connected" linkage added between habitat core ID #s 36 and 41 • Link ID #44 shifts about 2km to the north • Western third of Link ID #54 shifts several km to the north • Southern half of Link ID #57 shifts <2km to the west • Western quarter of Link ID #93 shifts several km to the north

No rivers	• Link ID #s 15, 46, 74, and 79 dropped from the model
	• Four additional linkages mapped compared to the primary model:
	    o "Well connected" linkage added between core ID #s 16 and 18
	    o "Moderately connected" linkage added between core ID #s 1 and 3
	    o "Poorly connected" linkage added between core ID #s 42→48 and 49→51
	• Number of "well connected" linkages increases relative to the primary model as some relatively short LCPs no longer have to cross narrow but highly resistant rivers (Table A3.4)
	• Link ID #14 shifts <2km to the west
	• Link ID #17 shifts several km to the east
	• Link ID #30 shifts several km to the east
	• Link ID #50 shifts up to 1km to the south
	• Western third of Link ID #54 shifts several km to the north
	• Location of Link ID #63 shifts approximately 27km to the northwest, to the northern juncture between habitat core ID #s 27 and 34
	• Link ID #65 shifts up to 7km to the south
	• Link ID #66 shifts up to 5.5km to the east
	• Central half of Link ID #93 shifts up to 5km to the south

Change to resistance surface used in primary model	Qualitative description of results

No roads or rivers	• Link ID #s 15, 46, 74, and 79 dropped from the model
	• Five additional linkages mapped compared to the primary model:
	  ○ "Well connected" linkage added between core ID #s 16→18
	  ○ "Moderately connected" linkages added between core ID #s 1→3, 36→41
	  ○ "Poorly connected" linkages added between core ID #s 42→48, 49→51
	• Number of "well connected" linkages increases relative to the primary model as some relatively short LCPs no longer have to cross narrow but highly resistant rivers (Table A3.4)
	• Link ID #14 shifts <2km to the west
	• Link ID #17 shifts several km to the east
	• Link ID #30 shifts several km to the east
	• Link ID #44 shifts about 2km to the north
	• Link ID #50 shifts up to 1km to the south
	• Most of Link ID #54 shifts 1-3km to the north
	• Southern half of Link ID #57 shifts <2km to the west
	• Location of Link ID #63 shifts approximately 27km to the northwest, to the northern juncture between habitat core ID #s 27 and 34
	• Link ID #65 shifts up to 7km to the south
	• Link ID #66 shifts up to 5.5km to the east
	• Link ID #93 deviates up to 5km from the LCP in the primary model

No serpentine	• Northern third of Link ID #30 shifts up to 3km to the east • Link ID #32 shifts to the west to follow the path of Link ID #38 as it moves south into California (in the primary model it traveled through serpentine habitat several km to the east) • Link ID #46 dropped from the model

Change to resistance surface used in primary model	Qualitative description of results

No roads, rivers, or serpentine
 	
See Figs. A3.4 and A3.5
	• Link ID #s 15, 46, 74, and 79 dropped from the model
	• Four additional linkages mapped compared to the primary model:
	  o "Well connected" linkage added between core ID #s 16→18
	  o "Moderately connected" linkages added between core ID #s 1→3, 36→41
	  o "Poorly connected" linkages added between core ID #s 42→48, 49→51
	• Number of "well connected" linkages increases relative to the primary model as some relatively short LCPs no longer have to cross narrow but highly resistant rivers (Table A3.4)
	• Link ID #14 shifts <2km to the west
	• Link ID #17 shifts several km to the east
	• Link ID #30 shifts several km to the east
	• Link ID #32 shifts to the west as was seen in the "No serpentine" trial
	• Link ID #44 shifts about 2km to the north
	• Link ID #50 shifts up to 1km to the south
	• Western third of Link ID #54 shifts several km to the north
	• Southern half of Link ID #57 shifts <2km to the west
	• Location of Link ID #63 shifts approximately 27km to the northwest, to the northern juncture between habitat core ID #s 27 and 34
	• Link ID #65 shifts up to 7km to the south
	• Link ID #66 shifts up to 5.5km to the east
	• Link ID #93 deviates up to 5km from the LCP in the primary model

Resistance value of all "good" serpentine habitat set to 1.0	• Link ID #49 dropped from the model • "Moderately connected" linkage added between habitat core ID #s 26 and 29 • Link ID #30 deviates from LCP in primary model by up to 2km • Link ID #s 31 and 32 pulled substantially to the east into a large area of serpentine soils for most of their routes, with deviations from the LCPs followed in the primary model of up to 17km • Link ID #37 moves substantially to the west, traveling through a large area of serpentine and entering Core ID #26 on the west side rather than the east side. LCP has shifted up to 27km in places • Link ID #46 deviates from LCP in primary model by up to 2.5km • Southern quarter of Link ID #52 shifts up to 3.5 km to the east • Link ID #54 shifts up to 10km to the north

Change to resistance surface used in primary model	Qualitative description of results

Resistance for OGSI >20 =1.0
 	
See Figs. A3.4 and A3.5
	• Link ID #s 38, 79, and 91 were dropped from the model
	• Thirteen additional linkages mapped compared to the primary model:
	  ○ A "well connected" linkage between core ID #s 16→18
	  ○ "Moderately connected" linkages between core ID #s 1→3, 6→9, 17→20, 26→29, 29→32, and 42→44
	  ○ "Poorly connected" linkages between core ID #s 10→14, 19→21, 20→23, 42→48, 42→50, and 46→47
	• Over half of the LCPs deviated significantly from those in the primary model, shifting by at least 1km, and in many cases much further. In several cases, the least-cost corridor shifted to a completely different location
	• Most least-cost corridors generally broader than in the primary model, with a number of additional secondary corridors appearing that are not directly adjacent to LCPs
	• Percentage of linkages classified as "poorly connected" by CWD is much lower than in the primary model (see Table A3.4)

Resistance for OGSI >30 =1.0
 	
See Figs. A3.4 and A3.5
	• Link ID #s 46, 79, and 87 were dropped from the model
	• Five additional linkages mapped compared to the primary model:
	  ○ "Moderately connected" linkages were added between core ID #s 1→3 and 29→32
	  ○ "Poorly connected" linkages were added between core ID #s 6→9, 42→48, and 49→51
	• About a third of the LCPs deviated significantly from those in the primary model, shifting by at least 1km, and in many cases much further. In several cases, the least-cost corridor shifted to a completely different location
	• Percentage of linkages classified as "poorly connected" by CWD is much lower than in the primary model (see Table A3.4)

Table A3.4. Summary of percentages of habitat core pair linkages classified as "well", "moderately", or "poorly" connected based on three distance metrics for each Linkage Mapper sensitivity analysis trial. Numbers show the percentage of linkages in each category for each trial as "well connected / moderately connected / poorly connected". Euclidean distance and least-cost path length are measured in km, while cost-weighted distance is measured in "cost-weighted units" normalized to km (well connected = 0-15km, moderately connected = 15-45km, poorly connected >45km). The values for the primary model are the same as those shown in Table 4 in the Results. The number of linkages varies somewhat across these trials because the changes in the resistance surface affect the calculations of nearest neighbor habitat core pairs that occurs during the first step of running Linkage Mapper (this is also responsible for the modest changes in Euclidean distance connectedness categories across trials conducted on the same set of cores). The most significant differences in these trials were in the estimates of connectedness using cost-weighted distance. Removing rivers from the model had the primary effect of increasing the number of well connected linkages using this metric as relatively short least-cost paths no longer had to cross these narrow but highly resistant features. Reducing the resistance of all OGSI pixels >20 to 1.0 (the lowest resistance value possible) dramatically decreased the number of poorly connected linkages compared to the primary model, as this variable was the primary contributor to resistance on the landscape.

Model	Euclidean distance	Least-cost path length	Cost-weighted distance	Linkages (*n*)
Primary model	64.9 / 21.7 / 13.4	54.7 / 27.8 / 17.5	36.1 / 28.9 / 35.0	97
No roads	64.6 / 21.9 / 13.5	54.2 / 28.1 / 17.7	35.4 / 30.2 / 34.4	96
No rivers	64.9 / 19.6 / 15.5	55.6 / 25.8 / 18.6	42.3 / 24.7 / 33.0	97
No roads or rivers	64.3 / 20.4 / 15.3	55.1 / 26.5 / 18.4	41.8 / 25.5 / 32.7	98
No serpentine	65.6 / 20.8 / 13.6	55.2 / 27.1 / 17.7	36.5 / 29.2 / 34.3	96
No roads, rivers, or serpentine	64.3 / 20.4 / 15.3	55.1 / 26.5 / 18.4	41.8 / 25.5 / 32.7	98
"Good serpentine" resistance = 1.0	66.0 / 21.6 / 12.4	55.7 / 27.8 / 16.5	36.1 / 33.0 / 30.9	97
OGSI >20 resistance = 1.0	62.6 / 23.4 / 14.0	55.2 / 28.0 / 16.8	41.1 / 38.3 / 20.6	107
OGSI >30 resistance = 1.0	63.7 / 22.2 / 14.1	52.5 / 30.3 / 17.2	38.4 / 38.4 / 23.2	99

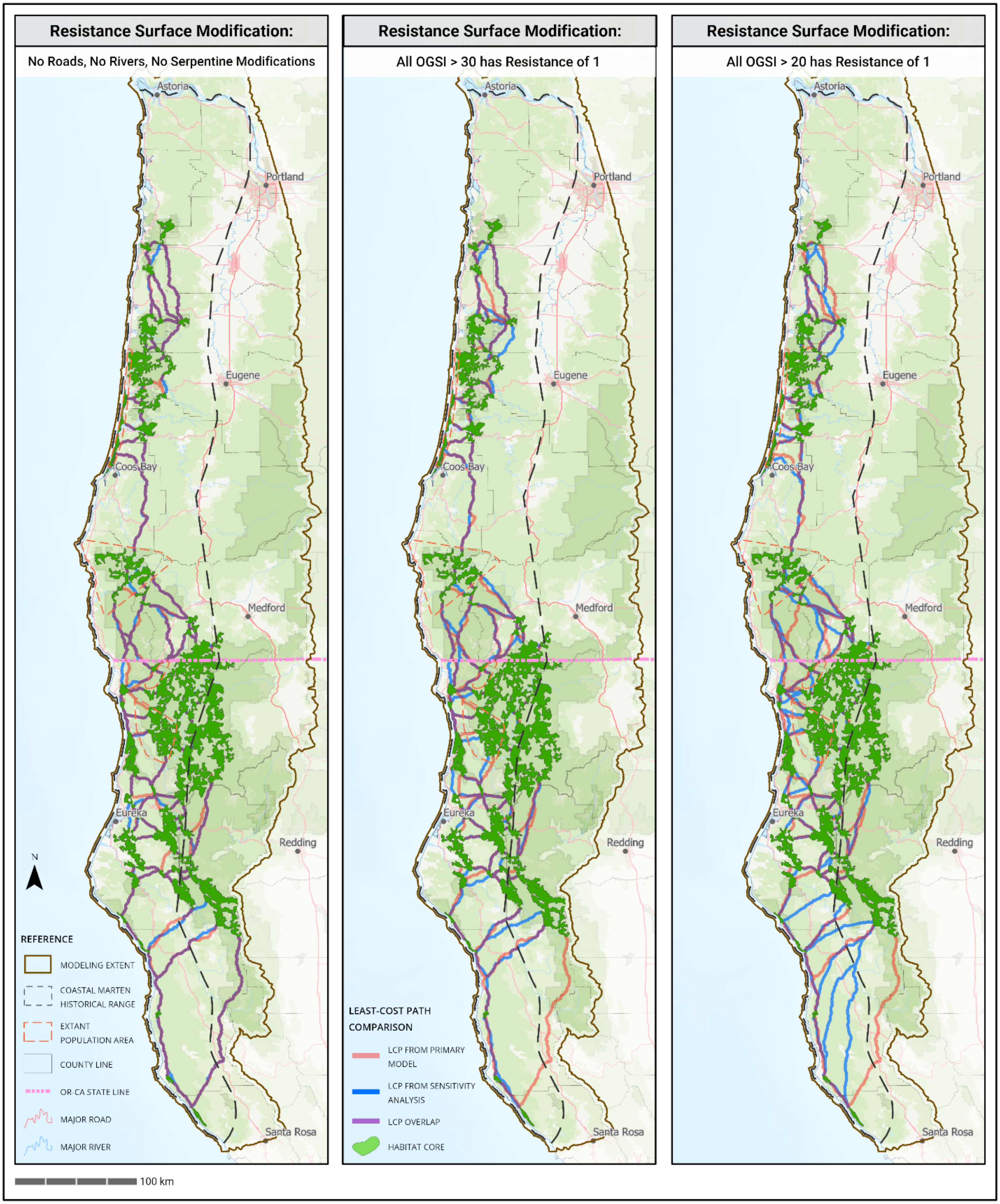

Figure A3.4. Examples of differences in the least-cost paths between the primary model and some of the sensitivity analyses conducted by modifying the resistance surface. The “no roads, rivers, or serpentine” modification replicated many of the effects of trials we conducted that only removed one or two of those variables. Altering the resistance related to OGSI resulted in the most significant differences from the primary model. See Table A3.3 for more details.

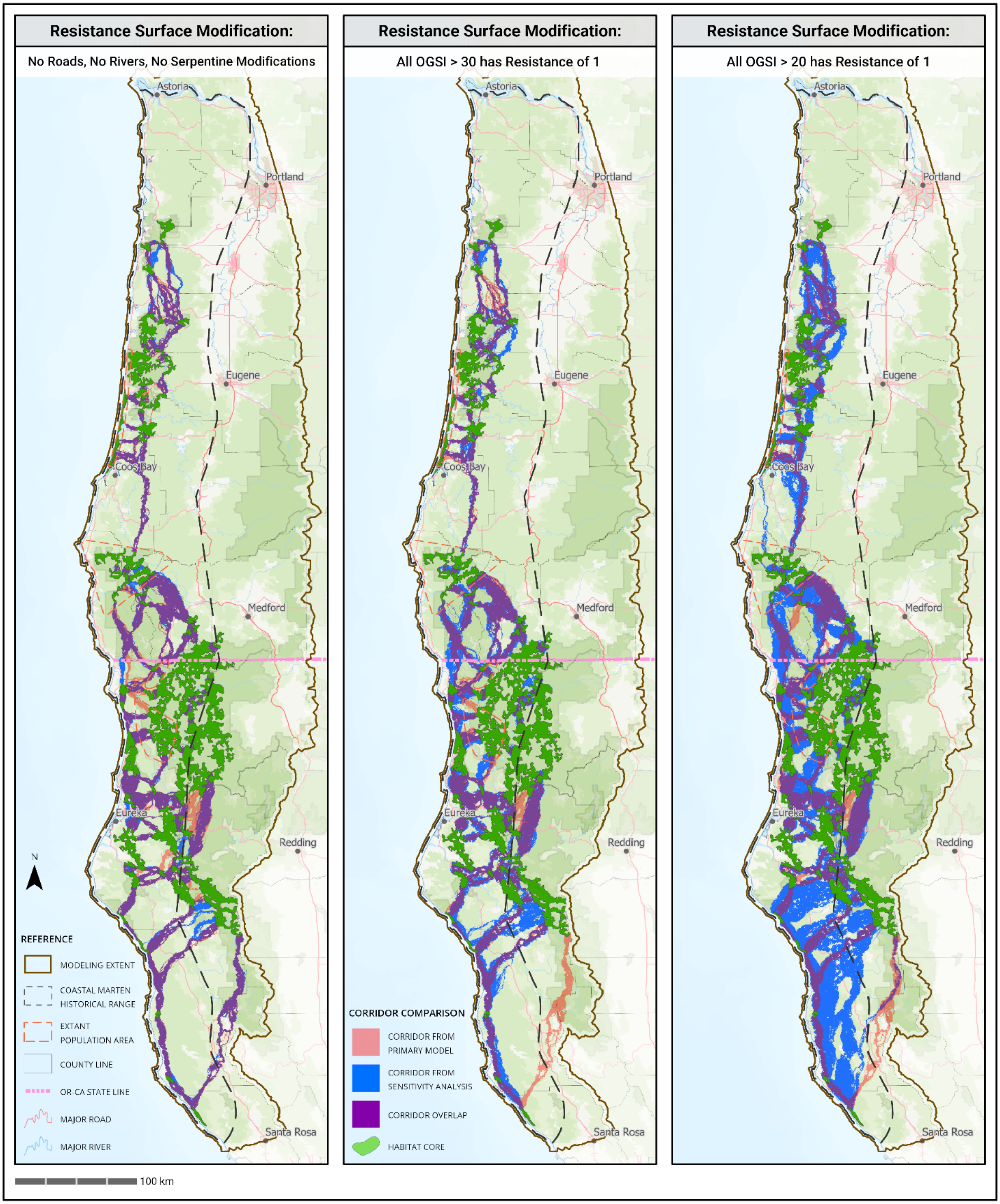

Figure A3.5. Examples of differences in the least-cost corridors between the primary model and some of the sensitivity analyses conducted by modifying the resistance surface. The "no roads, rivers, or serpentine" modification replicated many of the effects of trials we conducted that only removed one or two of those variables. Altering the resistance related to OGSI resulted in the most significant differences from the primary model. See Table A3.3 for more details.

Appendix 4

Summary statistics for habitat cores and least-cost corridor linkages.

Table A4.1. Summary statistics for all coastal marten habitat cores included in primary model. Core ID numbers were assigned by the Core Mapper spatial analysis tool and approximately order the habitat cores from north to south. Locations of habitat cores identified by Core ID# are shown in Fig. 9, and again in Fig. A4.1 below. We included in our final model any habitat core that had any portion of its extent inside of the historical range of the coastal marten, and for those that extend beyond the historical range we provide the area and percentage outside of this polygon. We defined a habitat core as supporting a known marten population if it overlapped with one of the four existing population polygons shown in Fig. 1.

Core ID#	Total area (ha)	Intersects historical range?	Area (ha) and % outside historical range	Supports known marten population?
1	10631.79	N	0	N
2	3546.27	N	0	N
3	4439.52	N	0	N
4	19198.53	N	0	N
5	2106.72	N	0	N
6	3497.31	N	0	N
7	40658.76	N	0	Y[§§§]
8	1717.56	N	0	N
9	6426.72	N	0	N
10[****]	7573.23	N	0	Y
11	5202.45	N	0	N
12	10990.98	N	0	N
13	11242.98	N	0	N
14	2990.34	N	0	N
15[****]	7542.72	N	0	Y
16	23572.71	N	0	Y
17	12147.93	N	0	Y
18	12928.95	N	0	Y
19	2467.71	N	0	Y
20	1677.24	N	0	N
21	3595.68	N	0	N
22	2300.22	N	0	N
23	1968.57	N	0	N

§§§ The northern boundary of the Central Coastal Oregon population mapped in the SSA (USFWS 2018) overlaps slightly with this habitat core, although this is based on buffering several marten detections in a small area of dune forest just outside of the core to the west.

**** Habitat cores created by "hand drawing" around coastal dune forest rather than through model output; see Methods for more details.

Core ID#	Total area (ha)	Intersects historical range?	Area (ha) and % outside historical range	Supports known marten population?
24	5082.39	Y	4519.68 (88.93%)	N
25††††	178091.10	Y	6906.66 (3.88%)	Y
26	2649.96	N	0	Y
27††††	108818.37	Y	98081.88 (90.13%)	N
28	4938.66	N	0	N
29	3979.53	N	0	N
30††††	3305.97	N	0	N
31	6101.82	N	0	N
32	1768.59	N	0	N
33	24334.92	N	0	N‡‡‡‡
34††††	57667.41	Y	34638.08 (59.60%)	N
35	1823.13	N	0	N
36	4365.90	N	0	N
37	37443.24	N	0	N
38	1624.77	N	0	N
39	1753.47	Y	59.05 (3.37%)	N
40	5091.12	Y	3906.36 (76.73%)	N
41	28842.57	N	0	N
42	68858.82	Y	67950.55 (98.68%)	N
43	2729.70	Y	404.94 (14.83%)	N
44	1718.73	N	0	N
45	7329.06	N	0	N
46	19488.69	Y	15917.89 (81.68%)	N
47	1934.64	N	0	N
48	2642.49	N	0	N
49	1750.86	N	0	N
50	1965.78	N	0	N
51	3763.71	N	0	N
Sum	788290.29		232385.09 (29.48%)	

†††† Habitat cores created by "manually" splitting a very large and sprawling core along natural river barriers. This was done to better illustrate potential corridors on this part of the landscape. See Methods for more details.

‡‡‡‡ Includes Prairie Creek Redwoods State Park, which has had resident coastal martens in recent years, but not yet in such numbers or consistency that is considered to have an established population.

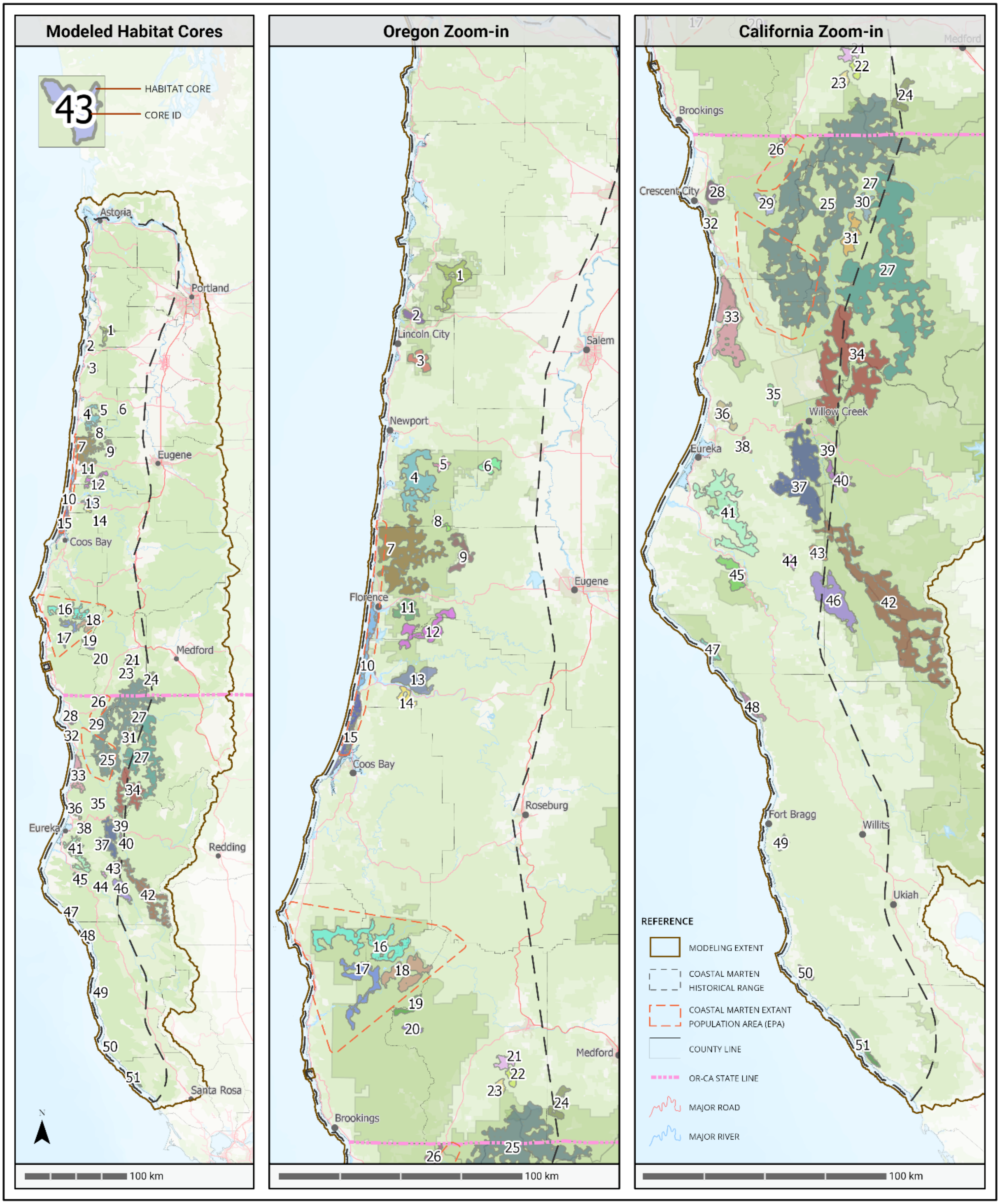

Figure A4.1. Coastal marten habitat cores identified using our primary model parameters. This map is identical to Fig. 9, and is provided here for the reader's convenience in referencing Tables A4.1 and A4.2.

Table A4.2. Summary statistics for all habitat core pair linkages included in the primary model. See Fig. 10 for mapped locations of habitat cores. "Euclidean" is the distance in km between the nearest points on the edges of two habitat core pairs regardless of the nature of the intervening landscape. "LCP" is the least-cost path length, which is also measured in km. "CWD" is the cost-weighted distance of the LCP, which is the sum of the resistance values multiplied by the raster grid cell size for all cells the path passes through, thus normalizing the resistance values to the distance units. We have normalized them here to km for direct comparison to the Euclidean distance and LCP length. The ratios of CWD:Euclidean distance and CWD:LCP length provide an indication of how resistant the habitat between the core pairs was to marten movement in the model. A ratio of 1.0 would indicate a connection that completely passes through the highest quality modeled habitat; the higher the ratio, the more difficult the modeled habitat between the core pairs to traverse.

Link ID#	From Core ID#	To Core ID#	Euclidean distance (km)	LCP length (km)	CWD (km equivalent)	CWD:Euclidean	CWD:LCP
1	1	2	5.41	7.8	17.22425195	3.18378	2.208237
2	1	6	63.72	84.901	209.9450781	3.294807	2.472822
3	2	3	10.401	14.517	29.19504492	2.806946	2.011094
4	3	4	31.747	39.886	136.9546094	4.313939	3.433651
5	3	5	35.288	46.377	142.215	4.030124	3.066499
6	3	6	43.475	58.889	166.1099844	3.820816	2.82073
7	4	5	1.527	2.285	2.976137451	1.949009	1.302467
8	4	7	1.358	2.582	3.532827881	2.601493	1.368253
9	4	8	6.307	7.64	12.28105762	1.947211	1.607468
10	5	6	11.276	14.768	26.75222266	2.372492	1.811499
11	6	8	25.599	32.442	56.14634766	2.193302	1.730669
12	7	8	4.118	6.209	8.568670898	2.080785	1.38004
13	7	9	2.266	2.976	3.925584473	1.732385	1.319081
14	7	11	1.465	2.278	12.03356348	8.214037	5.282513
15	7	12	10.416	16.488	27.00119141	2.59228	1.637627
16	8	9	1.698	2.333	3.732533447	2.198194	1.599886
17	9	12	14.494	19.859	38.26045313	2.639744	1.926605
18	11	10	6.262	8.128	30.24446484	4.829841	3.721022
19	11	12	0.768	1.55	1.867020508	2.431016	1.204529
20	12	10	10.627	14.668	38.75235938	3.646595	2.641966
21	12	13	6.976	9.729	23.27601367	3.336585	2.392437
22	13	10	7.046	9.344	46.39922656	6.585187	4.965671
23	13	14	0.432	1.353	20.68804688	47.889	15.2905
24	14	15	14.762	20.529	50.94625	3.451175	2.481672
25	14	16	89.004	119.613	240.1008438	2.697641	2.007314
26	15	10	0.45	0.51	24[§§§§]	53.33333	47.05882
27	16	15	61.86	125.877	270.4861563	4.372553	2.148813

[§§§§] This relatively short linkage between two habitat cores supporting the Central Coastal Oregon population has a very high CWD because it directly crosses the estuary of the Umpqua River.

Link ID#	From Core ID#	To Core ID#	Euclidean distance (km)	LCP length (km)	CWD (km equivalent)	CWD:Euclidean	CWD:LCP
28	16	17	1.894	2.79	4.245548828	2.241578	1.521702
29	17	18	0.53	0.766	2.400807373	4.529825	3.134213
30	17	19	10.033	14.853	27.9560293	2.786408	1.882181
31	17	26	58.832	82.464	148.920875	2.53129	1.805889
32	17	28	65.83	89.598	157.2551563	2.388807	1.755119
33	18	19	4.111	5.257	7.016574707	1.706781	1.334711
34	18	21	44.005	56.713	92.35739063	2.098793	1.628505
35	19	20	3.63	5.899	10.04054395	2.76599	1.702076
36	20	21	31.801	39.931	59.82020703	1.881079	1.498089
37	20	26	46.693	79.11	131.7394844	2.821397	1.66527
38	20	28	67.034	90.305	151.2112969	2.25574	1.674451
39	21	22	0.067	0.114	0.114852814	1.714221	1.007481
40	21	23	3.603	4.72	8.152051758	2.262573	1.72713
41	22	23	1.59	2.162	3.201320313	2.013409	1.480722
42	22	25	7.343	8.562	11.83877832	1.612254	1.382712
43	23	25	6.149	8.967	12.96094629	2.107814	1.445405
44	24	25	2.303	3.791	6.423670898	2.789262	1.694453
45	25	26	4.582	5.96	9.573336914	2.089336	1.606265
46	25	28	20.997	25.644	57.10325781	2.719591	2.226769
47	25	29	1.103	1.379	2.425035645	2.198582	1.758546
48	25	31	1.044	1.388	8.655807617	8.291003	6.236173
49	25	32	19.011	25.307	55.75819141	2.932944	2.203271
50	25	33	12.344	16.694	47.70894531	3.86495	2.85785
51	25	34	1.705	2.374	12.15234668	7.127476	5.118933
52	25	35	24.168	31.907	73.97478125	3.060857	2.31845
53	23	26	29.614	40.67	68.23700781	2.304214	1.677822
54	26	28	23.307	29.79	68.56908594	2.941995	2.301749
55	27	30	0.3	0.499	0.514705627	1.715685	1.031474
56	28	29	11.81	15.633	36.90626172	3.125001	2.360792
57	28	32	5.058	7.829	12.33884961	2.439472	1.576044
58	31	27	1.361	3.001	3.155254395	2.318335	1.051401
59	31	30	1.395	1.755	2.171726074	1.556793	1.237451
60	32	33	17.907	21.908	67.76683594	3.784377	3.093246
61	33	35	13.778	20.163	42.87856641	3.112104	2.126596
62	33	36	14.72	19.962	30.42769531	2.067099	1.524281
63	34	27	0.03	0.906	1.03882251	34.62742	1.146603
64	35	34	15.222	18.188	35.38754688	2.324763	1.945654

Link ID#	From Core ID#	To Core ID#	Euclidean distance (km)	LCP length (km)	CWD (km equivalent)	CWD:Euclidean	CWD:LCP
65	35	36	16.258	21.829	43.72110547	2.689206	2.002891
66	35	37	11.782	15.654	26.45788672	2.245619	1.690168
67	35	38	17.323	22.212	52.48337891	3.029693	2.362839
68	36	38	4.436	5.488	11.02477246	2.485296	2.008887
69	37	34	7.465	10.944	27.30079492	3.657173	2.49459
70	37	39	2.059	3.292	12.25529004	5.952059	3.722749
71	37	40	2.28	2.73	10.25025391	4.495726	3.754672
72	37	41	11.287	13.419	34.07843359	3.019264	2.539566
73	37	42	1.509	1.92	2.594482666	1.719339	1.351293
74	37	44	16.07	22.961	46.50765234	2.894067	2.025506
75	38	37	13.463	16.534	31.26824414	2.322532	1.891148
76	38	41	9.023	12.305	18.23645508	2.021108	1.482036
77	39	34	7.122	10.643	22.35024609	3.138198	2.099995
78	39	40	0.484	0.779	0.945182922	1.952857	1.213328
79	40	27	40.447	52.629	70.50107813	1.743048	1.339586
80	40	42	12.718	16.17	30.54633789	2.401819	1.889075
81	41	44	9.982	13.162	29.50567383	2.955888	2.241732
82	41	45	3.807	5.227	20.08124023	5.27482	3.841829
83	42	27	61.652	74.912	106.9687734	1.735041	1.427926
84	42	43	4.924	8.022	19.16793164	3.892756	2.389421
85	42	46	6.138	7.586	17.9879707	2.930592	2.371206
86	42	49	84.135	122.71	232.0903281	2.758547	1.891373
87	42	51	146.328	204.628	383.3952188	2.620108	1.873621
88	43	46	4.11	4.973	7.566244629	1.840935	1.521465
89	44	43	6.401	8.416	13.16686426	2.057001	1.564504
90	44	46	7.183	10.116	13.17495117	1.834185	1.302387
91	45	44	17.073	21.063	55.23408594	3.235172	2.622328
92	45	47	23.017	30.162	53.35341016	2.318	1.768895
93	46	48	47.367	64.628	140.0086406	2.955827	2.166378
94	47	48	15.281	17.718	22.78491211	1.491062	1.285975
95	48	49	47.494	61.356	86.58261719	1.823022	1.411152
96	49	50	47.741	61.94	107.3857578	2.24934	1.733706
97	50	51	25.073	30.967	48.72040234	1.943142	1.573301

Appendix 5

Supplemental maps illustrating the location of coastal marten habitat cores and corridors through the lenses of various land management boundaries and the serpentine soils layer we incorporated into our habitat model and resistance surface. These are intended to provide interested readers with a more detailed picture of the location of the habitat cores and least-cost corridors produced by our primary model in relation to certain boundaries and landscape features they might be particularly interested in from a management perspective.

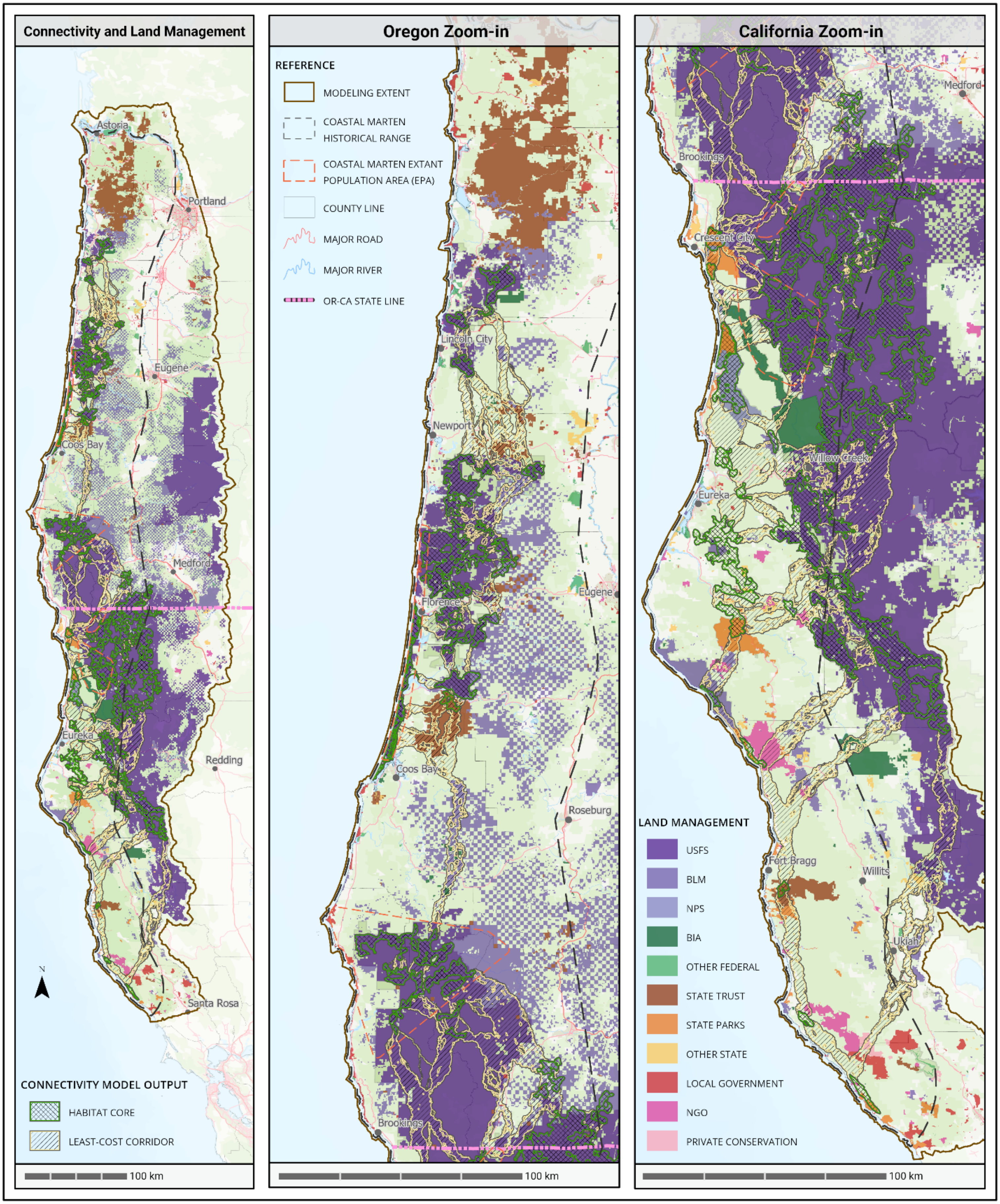

Figure A5.1. Coastal marten habitat cores and least-cost corridors overlaid with land management boundaries derived from the USGS' Protected Areas Database of the United States (PADUS) (USGS 2016).

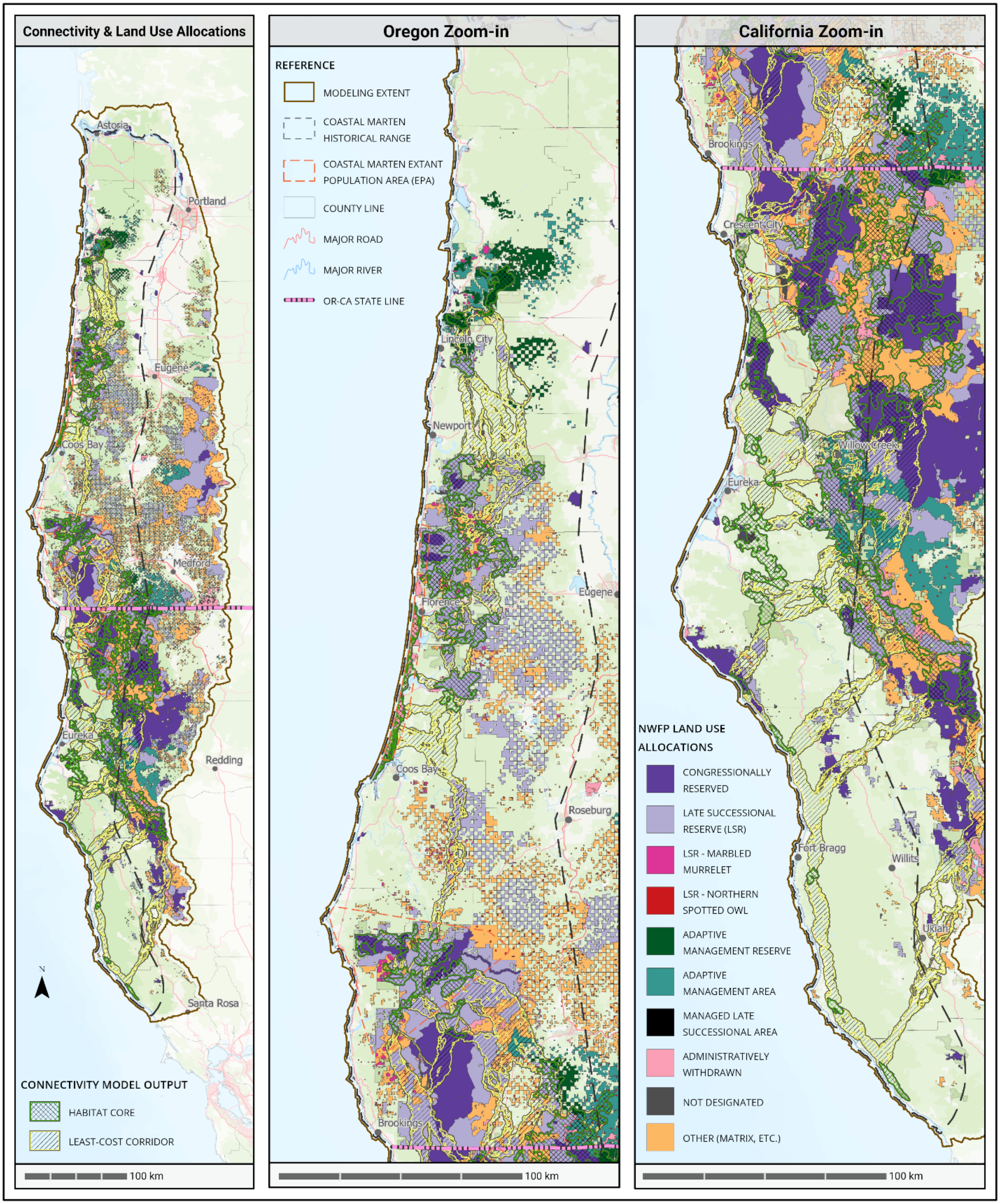

Figure A5.2. Coastal marten habitat cores and least-cost corridors overlaid with land use allocation boundaries derived from the Northwest Forest Plan (Tuchmann *et al*. 1996).

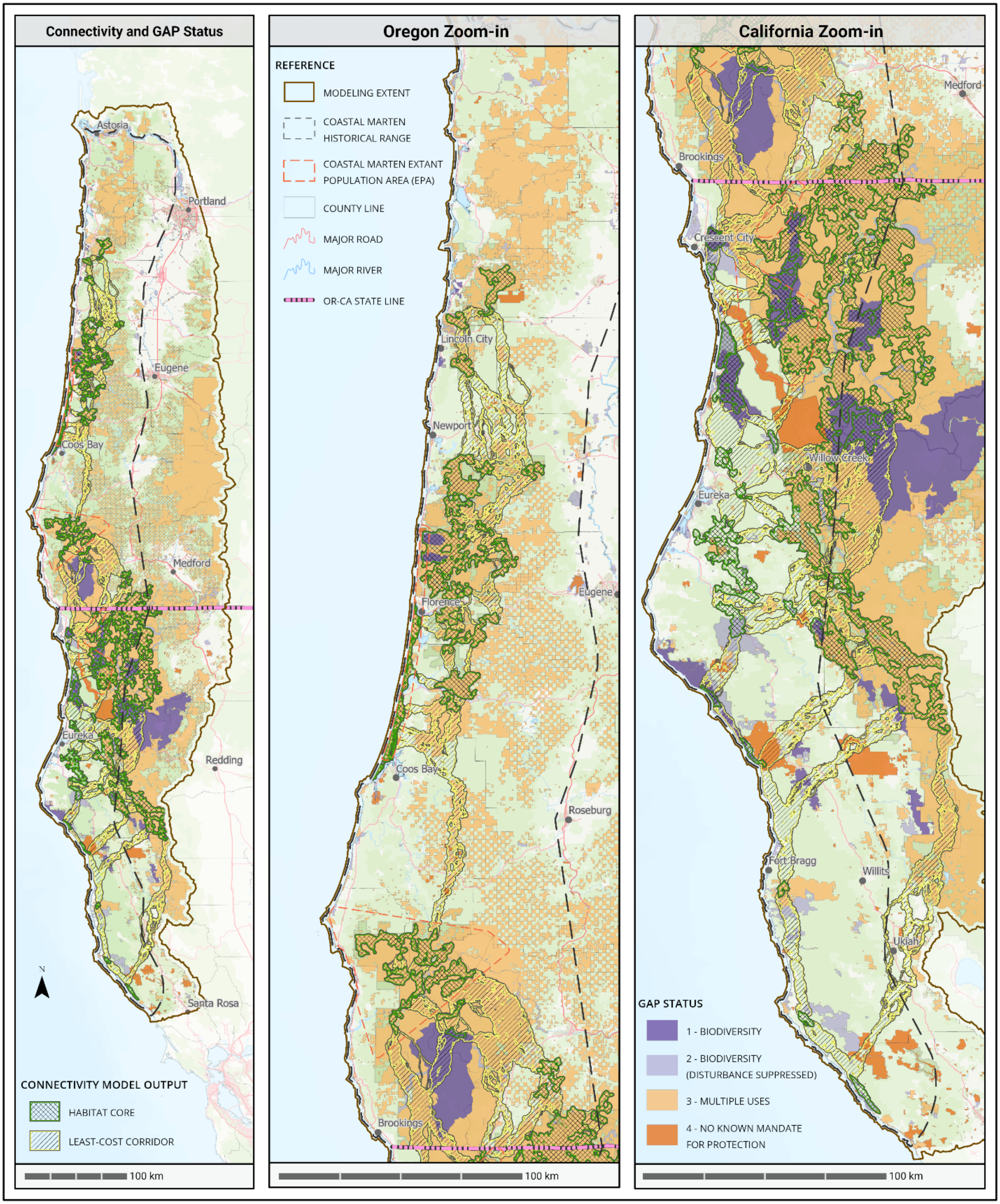

Figure A5.3. Coastal marten habitat cores and least-cost corridors overlaid with USGS GAP status categories of conservation management (USGS 2016).

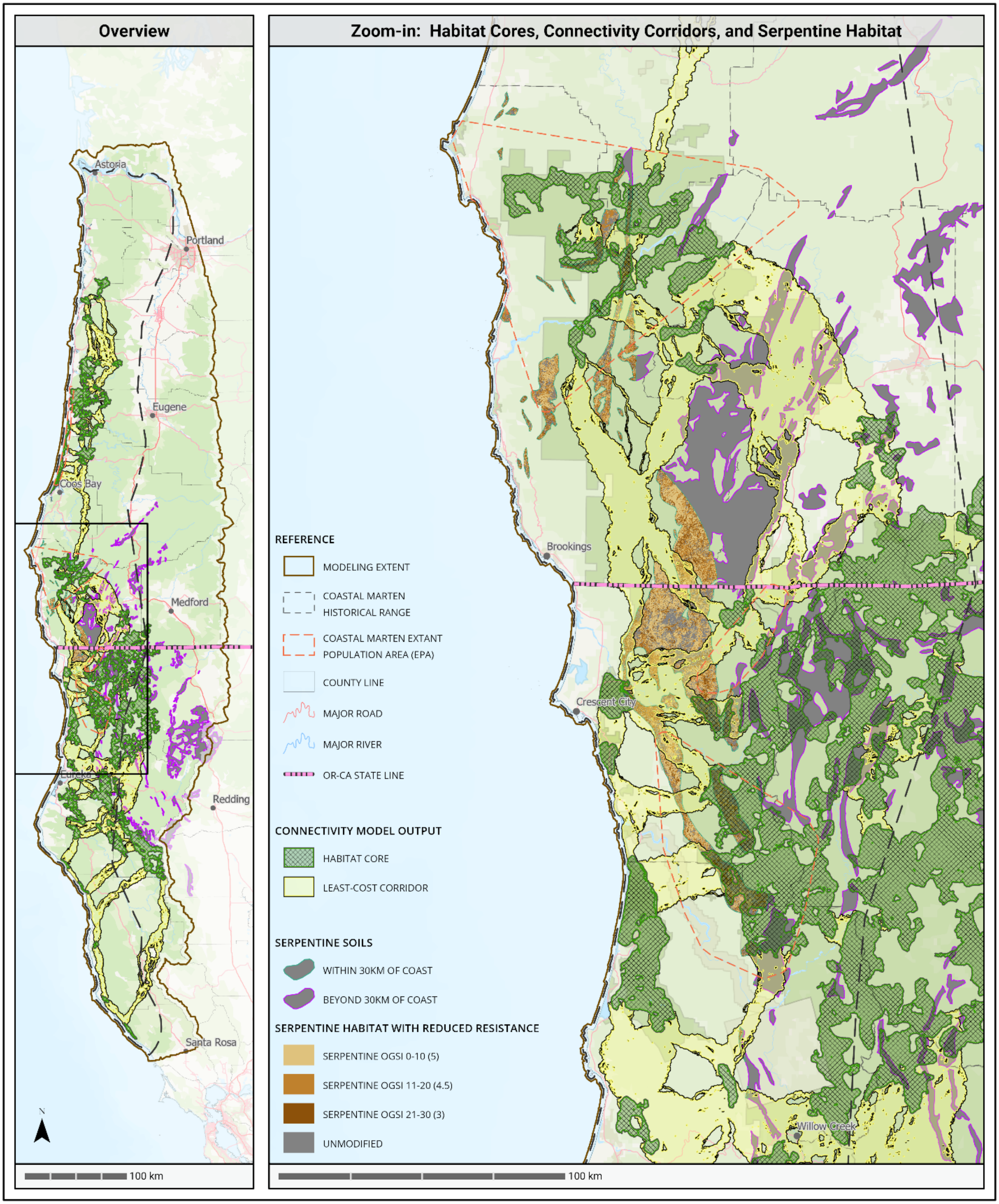

Figure A.5.4. Coastal marten habitat cores and least-cost corridors overlaid with the serpentine soils layer used in the model.

Appendix 6

Supplemental maps illustrating coastal marten least-cost corridor quality based on the Euclidean or CWD metrics. These are intended as a supplement to the maps included in the Results section as a way to assist readers in visualizing how habitat cores are linked into clusters and the relative quality of different least-cost corridors. Two maps are provided for each metric. The first depicts the habitat cores and least-cost corridors in the same manner as they are in Fig. 10, but omits those linkages classified as "poorly connected" (i.e. those >45km or cost-weighted km). The second shows all least cost corridors, but color-codes these by the three quality categories (i.e. ≤15km or cost-weighted km are "well connected", ≤45km or cost-weighted km are "moderately connected", and <45km or cost-weighted km are "poorly connected").

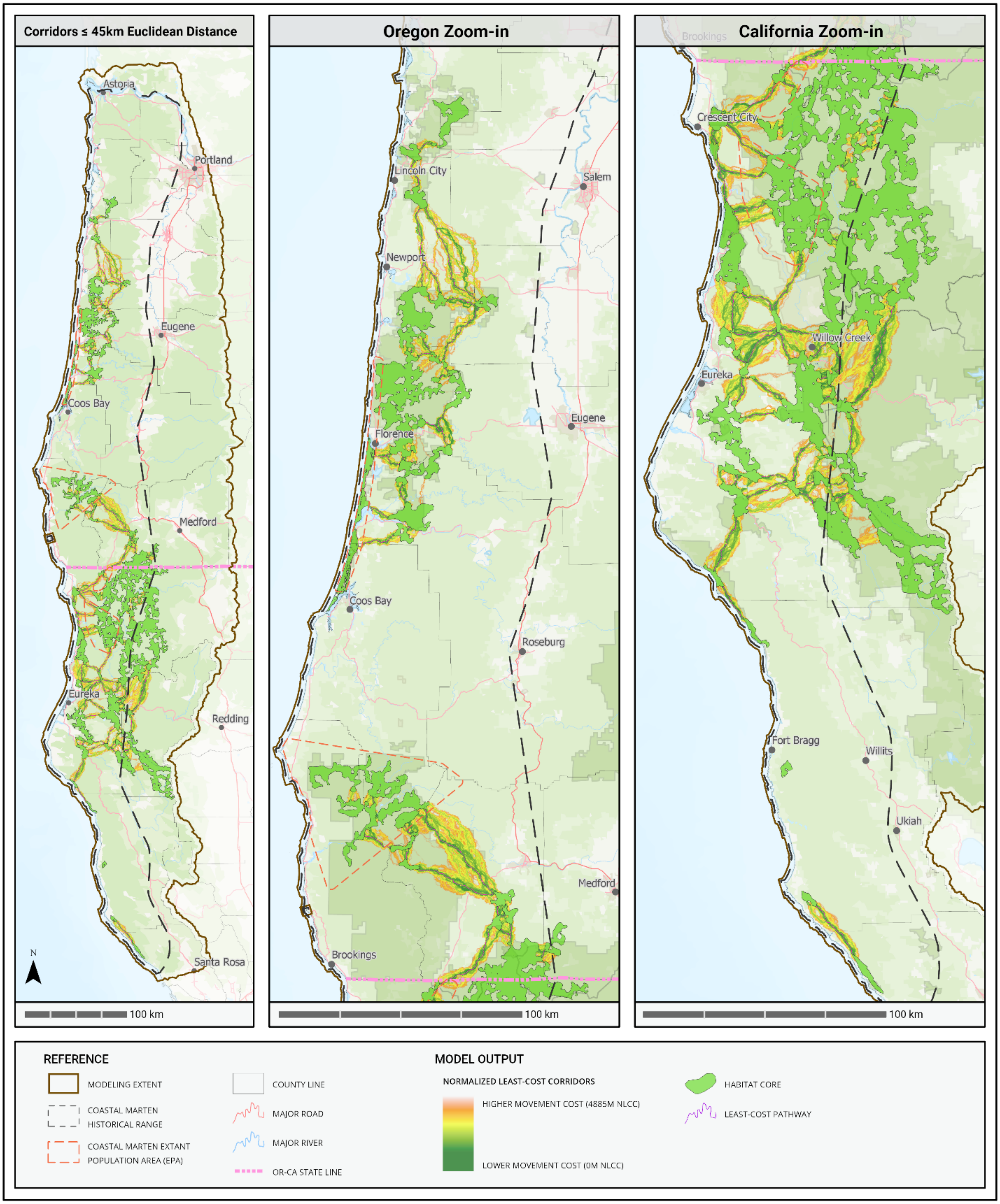

Figure A6.1. Coastal marten habitat cores connected by least-cost corridors classified as “well connected” (≤15km) or moderately connected (≤45km) based on the Euclidean distance metric, with longer “poorly connected” linkages omitted.

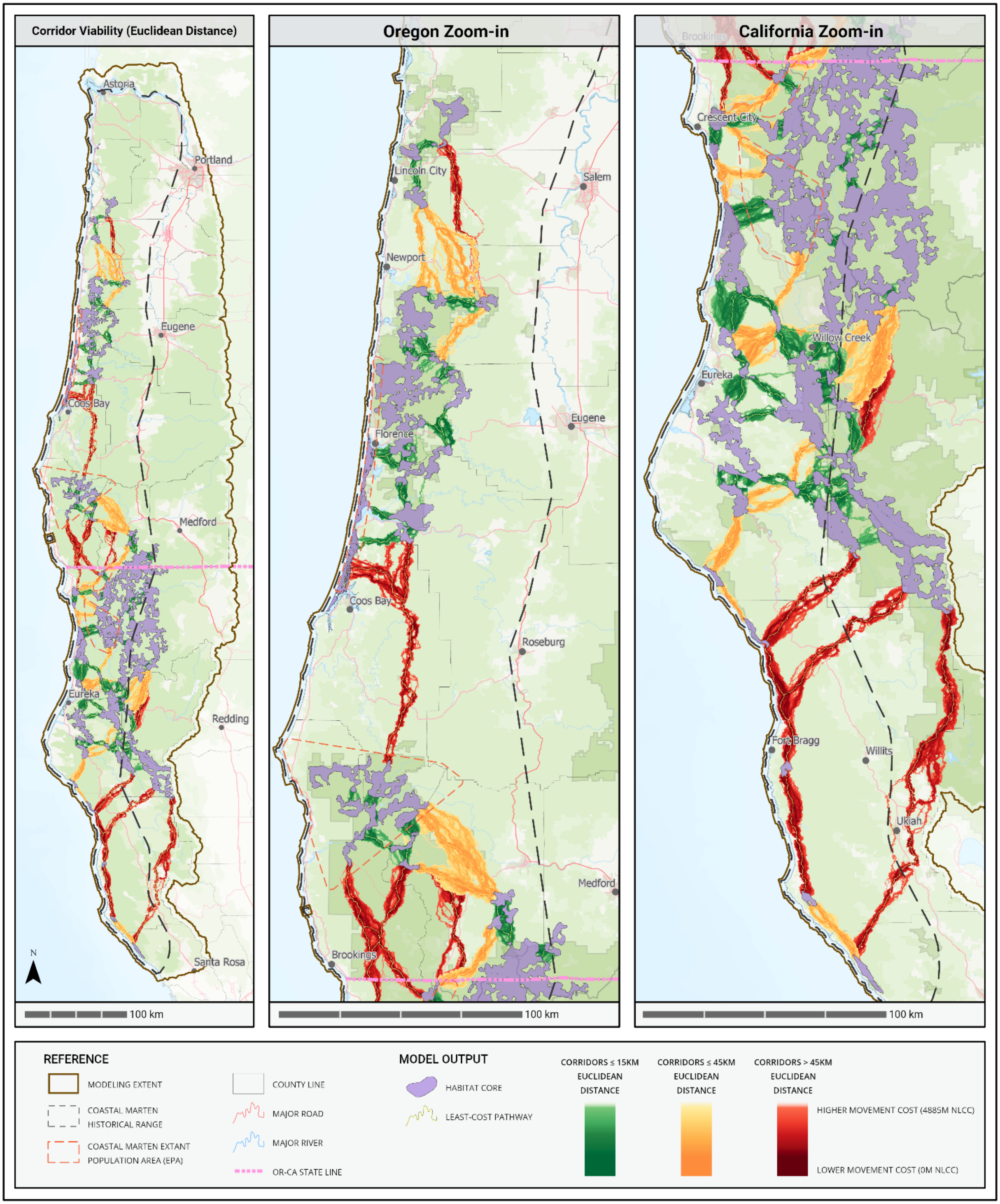

Figure A6.2. Coastal marten least-cost corridors classified as well connected (≤15km), moderately connected (≤45km), or poorly connected (>45km) based on Euclidean distance.

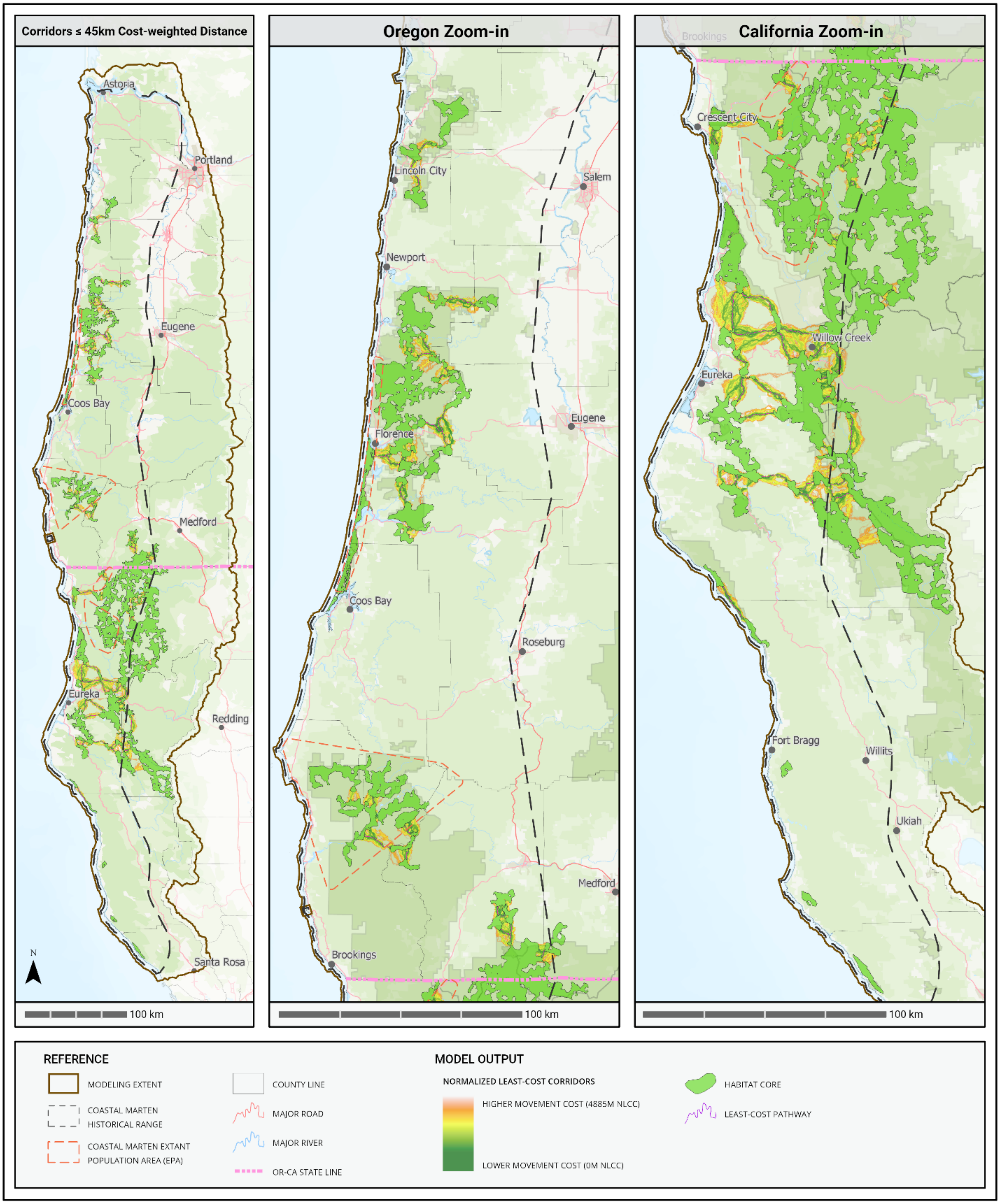

Figure A6.3. Coastal marten habitat cores connected by least-cost corridors classified as "well connected" (≤15km) or moderately connected (≤45km) based on the cost-weighted distance metric, with longer "poorly connected" linkages omitted.

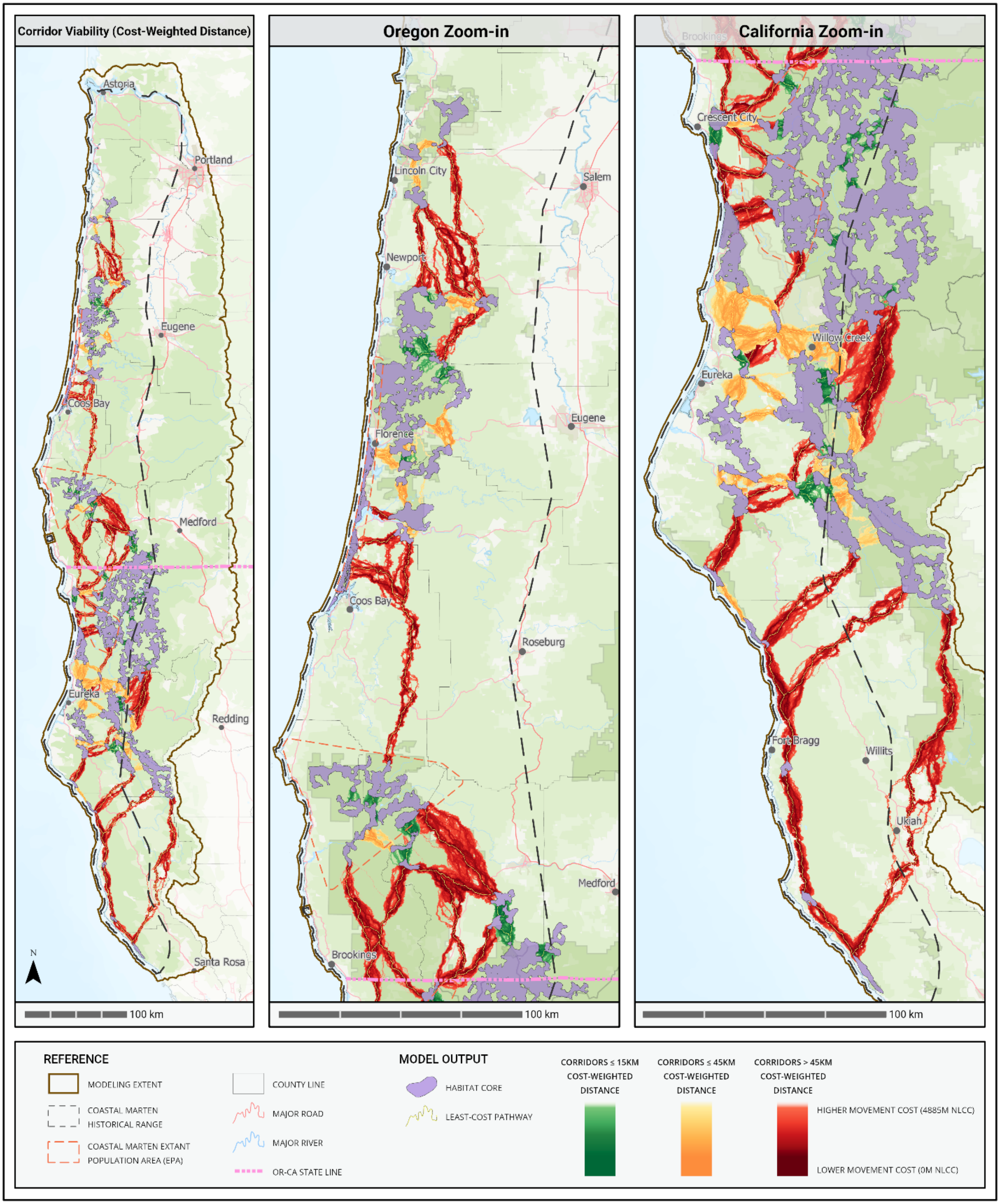

Figure A6.4. Coastal marten least-cost corridors classified as well connected (≤15km), moderately connected (≤45km), or poorly connected (>45km) based on cost-weighted km. This map is identical to Fig. 14 and shown again here for comparison to the other maps in this appendix.