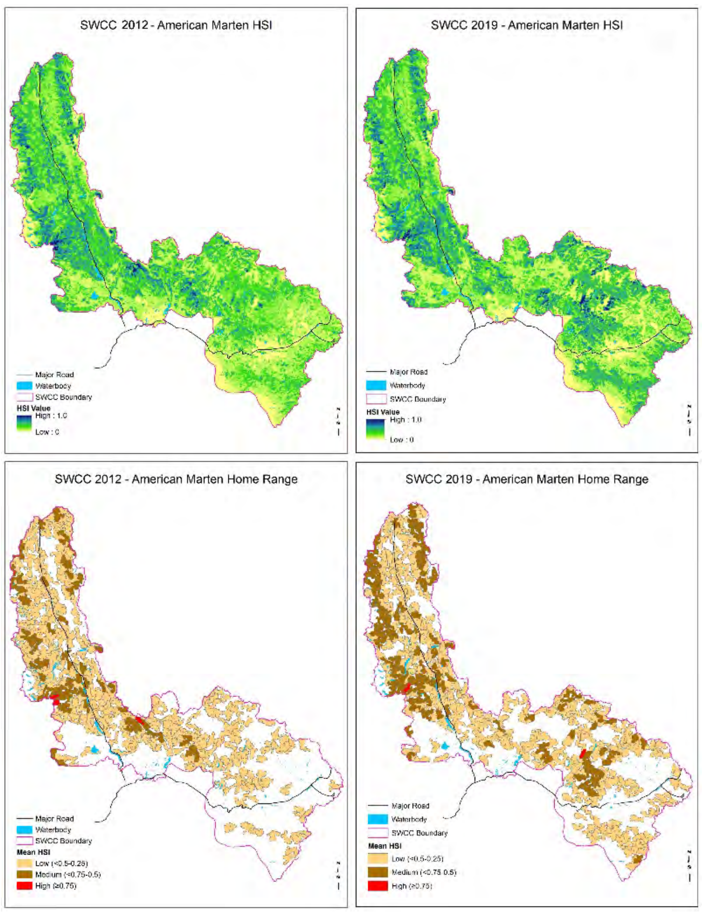
SECOND ASSESSMENT OF WILDIFE HABITAT FOR THE SOUTHWESTERN CROWN OF THE CONTINENT CFLR PROJECT

January 2020



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Introduction

The Southwestern Crown-of-the-Continent Collaborative (SWCC) Forest Landscape Restoration Project (FLRP) has been a 10-year project designed to plan and support collaborative opportunities for addressing fuel mitigation, forest restoration, watershed enhancements, and economic development through improvements to U.S. Forest Service lands within the 1.5 million acre project area. Treatments to be conducted over the duration of the project were designed to collectively address these project goals. One specific objective of the project was to improve fish and wildlife habitat, including for endangered, threatened, and sensitive species. Monitoring of accomplishments including both implementation and effectiveness monitoring were an important component of the overall project.

Threatened, endangered and sensitive species could potentially be affected by any of the terrestrial treatments including mechanical thinning and prescribed fire. Monitoring the effects of treatments on these species was identified as a desirable component of monitoring. This wildlife monitoring project was initiated by the SWCC Wildlife Monitoring Committee to monitor the effects of treatments on selected species that fit the above criteria in the SWCC: fisher (*Martes pennanti*), American marten (*Martes americana*), northern goshawk (*Accipiter gentilis*), flammulated owl (*Otus flammeolus*), hairy woodpecker (*Picoides villosus*), and pileated woodpecker (*Dryocopus pileatus*).

Most of the threatened, endangered and sensitive species in the SWCC occur in relatively low densities, have large home ranges, and have complex habitat requirements. Because of this, monitoring population responses of these species at the scale of specific projects is not meaningful in terms of gathering statistically relevant information. Consequently, we chose to monitor the effects of treatments on the habitat quality of the selected species through habitat modeling and collection of site specific vegetation parameters.

The SWCC Wildlife Monitoring Committee selected habitat suitability modeling as the underlying framework to monitor the response of the identified species to treatments. These models evaluate habitat suitability for each species at appropriate scales and compare habitat suitability before and after treatment.

The purpose of the habitat suitability modeling discussed in this document is for use by the multi-party monitoring efforts of the SWCC. It is not a new standard, guideline or protocol for species habitat analysis. The models were initially developed in 2012 with model outputs derived from VMAP data available at that time. This document represents a revision of the models using newer VMAP data available for the Lolo and Helena National Forests from 2018 that captures recent disturbances and incorporates growth projections from earlier data.

The new data from 2018 include changes occurring in the landscape from all sources including treatments conducted by the Forest Service, but also vegetation treatments on other land ownership, changes caused by wildfires, and successional change. The effects of the Forest Service treatments have not been singled out, as they are only one component of landscape-scale changes affecting species habitat.

Model validation is always desired, but often not feasible through available resources. This was the case with the SWCC species modelling. We evaluated model outputs qualitatively using expert opinion and known information on species distributions within the landscape. We consider our model outputs to generally reflect changes in species habitat quality over the duration of this project. However, the difficulty in generating accurate vegetation parameters from remotely sensed sources for most habitat attributes of the selected species makes model outputs for specific locations less reliable. This coupled with the generally large home ranges of the species evaluated and the various additional factors that could influence occurrence of these species means that while we would expect the overall habitat conditions and trends for these species to be captured by the models, we do not expect these species to necessarily be present where our models suggest high-quality habitat occurs.

Acknowledgements
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Methods
Wildlife models were developed using a HSI model framework that identifies the known habitat characteristics of each species, and the relationship between identified variables to the habitat suitability of the species. An example is shown in Figure 1.

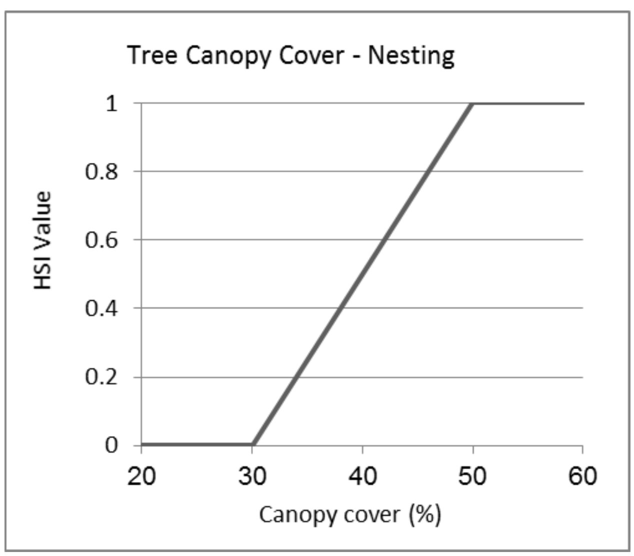


Figure 1. Example of the relationship between a vegetation parameter and habitat suitability of a modeled wildlife species. HSI value runs from 0, meaning no habitat quality, to 1, meaning optimum habitat conditions for a species in terms of this specific vegetation parameter.

HSI models evaluate the contributions of vegetation conditions measured either through remotely-sensed information or site-collected information. In an earlier report (Haufler et al. 2016), we described how site-collected information provided a means to determine changes in habitat quality at the scale of a specific project. In this current analysis, we compared habitat quality at landscape-scales based only on remotely-sensed information.

Each GIS pixel within the SWCC landscape received a rating for each habitat variable used in each of the HSI models, and these ratings were combined to develop a final rating for the overall habitat quality for each species. By conducting analyses based on conditions in 2012 compared to conditions mapped from the 2018 data, the changes to overall habitat quality of each species were calculated.

For the initial 2012 SWCC project area, the starting point for landscape scale assessment of existing vegetation conditions was the VMAP data and GIS layers provided by the Blackfoot Swan Landscape Restoration Project Team and specifically compiled by Chip Fisher and Eric Henderson. The map of existing conditions was classified based on tree size classes and canopy closure classes. This map was then combined with a map of ecological sites (Figure 2) produced for a landscape assessment of the SWCC project area (Haufler et al. 2016), such that each GIS pixel in the landscape was assigned an ecological site and also had VMAP data for tree size (DBH) and canopy cover (percent). The unique values that were created were called disturbance states. Disturbance states are described by each unique combination of ecological site, canopy closure and tree size.

Once each pixel was assigned a disturbance state then FIA plot data, and available TSMRS data, could be used to assign habitat variable values for each classification category. The mean value for each habitat variable was used. Two habitat variables were provided directly from the VMAP classification; tree canopy cover and tree size class.

Some classification categories included in the VMAP/ecological site combination lacked any plot sampling to provide an empirical data set. For these classification categories, we assigned values to habitat variables based on the general vegetative characteristics that defined each classification category. Whenever possible, plot data from adjacent ecoregions were used to help inform the habitat values chosen.

HSI maps for the SWCC landscape were created using the combined VMAP map and ecological site map (disturbance states), and assigned habitat values using the associated data generated from the available plot data. The appendices of this report provide more details on the specific HSI model created for each wildlife species.

For the 2019 modeling update new disturbance states were calculated using 2018 VMAP data from the Lolo and Helena National Forests (new data were not available for the Flathead NF). The updated VMAP data included disturbances through 2018 along with successional changes to vegetation to account for forest stand changes since the source imagery was captured. Values derived from FIA and TSMRS sample data in the previous version of this analysis were still used to generate HSI values for each disturbance state.

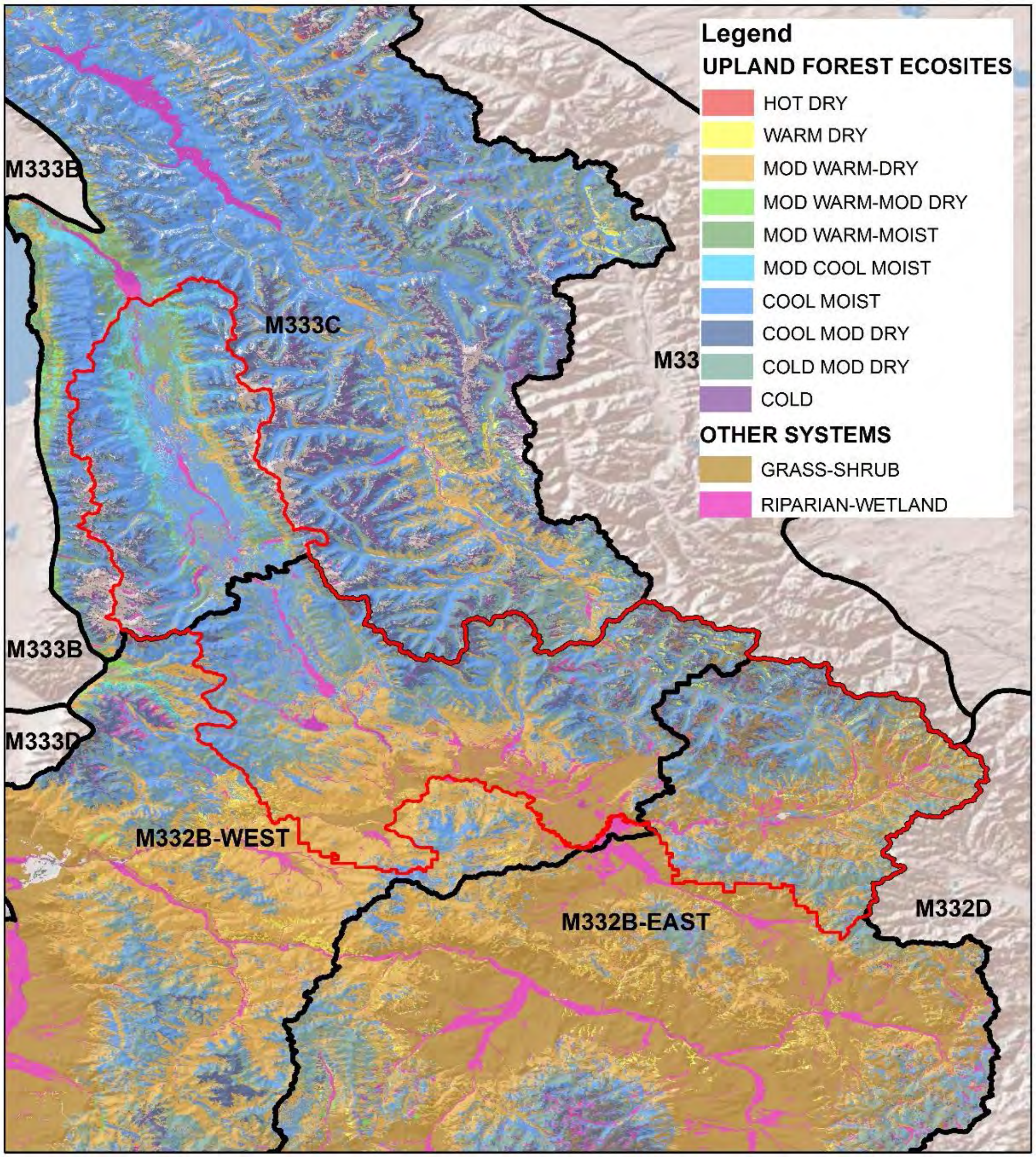


Figure 2. The distribution of 10 upland forest ecological sites as well as a single grouping of grass-shrub and riparian-wetland-aquatic ecological site within the SWCC project area (From Haufler et al. 2016).

Assessing Spatial Distribution of Habitat Suitability for Use by Each Species

Each wildlife model combines the various key habitat variables used to characterize the habitat quality of a species into an overall habitat suitability rating for each classification category of mapped vegetation. The resulting habitat suitability map reveals a range of expected high to low quality habitat based on the input vegetation data sources. The habitat suitability maps were then used to estimate the number and quality of potential home ranges using the home range assessment approach. HOMEGROWER is a program developed to automate the home range assessment approach by aggregating the required elements into appropriate sized home ranges for each species within the planning landscape. Each species has minimum and maximum home range sizes that it will utilize. This process has been described by Roloff and Haufler (1997, 2002).

HOMEGROWER builds home ranges by evaluating the cells around a starting point of the highest quality habitat not already contained in a home range, and growing a new home range using the neighboring cells of highest quality. Cells are accumulated until the growth target, expressed as total HSI scores for that species has been met. HSI scores are tallied based on area multiplied by the habitat quality for each pixel that is added to the home range. The target for each species is based on a multiplier of its allometric home range. Allometric home ranges are the estimated minimum area that a species could occur in based on its estimated metabolic requirements. For consistency a rate of 5x the allometric home range was used to calculate the target home range size. This resulted in the following targeted minimum possible home range sizes in HOMEGROWER:

- fisher - 3039 acres
- flammulated owl - 42 acres
- northern goshawk – 825 acres
- hairy woodpecker – 56 acres
- pileated woodpecker – 305 acres
- American marten – 1149 acres

For example, if a bird has an allometric home range of 1 acre, its targeted home range requirements would be 5 acres or 5 HSI units. This could be met with a home range of 5 acres if all acres in that home range contributed 1.0 in HSI value, and would receive an overall home range quality of 1.0, and then be designated a high-quality home range. However, this rarely occurs in the real world. Home ranges are typically comprised of patches of habitat for the species of varying quality. HOMEGROWER builds home ranges for a species by starting with a single cell of the highest quality in the landscape that has not already been included in another home range. It then grows by aggregating cells of the next highest quality until it has acquired the HSI units desired for the species, in this case, 5 HSI units. An upper threshold of size is set at 10 times the target size, or in this example 50 acres, beyond which HOMEGROWER ceases attempting to build a home range if the area becomes too large to provide the necessary density of habitat required by the species. If in this example, HOMEGROWER identified a potential home range that took 8 acres to reach its target of 5 HSI units, it would be mapped as a home range, assigned an HSI value of 0.63 as calculated from the size of the mapped home range relative to the optimum sized home range, and would be designated a medium-quality home range.

For this assessment, home ranges with total HSI values >0.75 were considered high-quality home ranges, HSI values of 0.5-0.74 were considered medium-quality home ranges, and HSI values of 0.25-0.49 were considered low quality. Roloff and Haufler (2002) discussed the implications of these ratings to their support of a species population. High-quality home ranges are assumed to have high rates of occupancy, support high reproductive rates, and have high survival rates, thus providing good demographic support of the population of the species (Roloff and Haufler 2002). Kroll and Haufler (2010) tested this relationship for dusky flycatchers in Idaho and reported empirical support for this relationship. Medium-quality home ranges would have lower occupancy rates, being occupied when population sizes were high following good years in terms of weather or other conditions, would have intermediate reproductive and survival rates, and would provide some demographic support to the species. Low-quality home ranges would have low occupancy rates, being occupied only when populations were relatively large in the best of years, have low reproductive and survival rates, and would generally not contribute significantly to population demographics. Non-habitat is assumed to not be occupied and provides no demographic support to the population.

This analysis produces a map of home ranges of varying quality distributed across the landscape for each species. The landscape is then evaluated based on the number of potential high, medium, and low-quality home ranges mapped through this process. Comparisons of numbers of high, medium, and low-quality home ranges can be done between various landscapes and a determination of the likely response to changes, in terms of general population potential of the species, can be estimated. The modeled number of high, medium, and low-quality home ranges are estimates of habitat conditions and actual population sizes of modeled species may be quite different. However, the overall habitat quality and trends in conditions are evaluated in a consistent manner in this modeling approach, and should represent actual changes in overall habitat suitability for each species within the landscape. HOMEGROWER is a stochastic model and there can be variations between each model run based on where the starting seeds are located. This is particularly evident when comparing mapped locations of home ranges. The total number of home ranges and the quality of each are the best measure of habitat for each species and provide the best means of comparison versus visually inspecting home range maps and comparing the spatial extent of each home range.

Results and Discussion

The analysis of habitat at the landscape scale generated HSI maps for each species using the same methodology for comparing 2012 conditions to those found occurring from the 2018 imagery, with the only difference being the new disturbance states calculated from the 2018 VMAP data. Figures 3-10 compare habitat suitability grids for each species for the 2012 and 2019 landscapes.

As discussed, habitat suitability maps were then evaluated and converted to potential home ranges of varying quality for each of the species using HOWEGROWER. Table 3 lists the mean number of varying quality home ranges for each species and the confidence intervals around these estimates for both 2012 and 2019 landscapes. Figures 11-16 compare home range quality maps for each species for 2012 and 2019 landscapes.

One thing to consider in the assessment of existing conditions for flammulated owls and pileated woodpeckers is that VMAP is known to underestimate the occurrence of very large trees. While efforts have been made to correct this, there is still concern that current mapping underestimates the area containing very large trees within the landscape. Similarly, this same concern exists for underestimating the area containing very large snags. With this key habitat variable being a major driver of both flammulated owl and pileated woodpecker habitat quality, the estimates of high and medium-quality home ranges may be underestimated as well. However, until additional empirical data can be generated to better map the numbers and areas containing very large snags, this evaluation represents the best available information at this time.

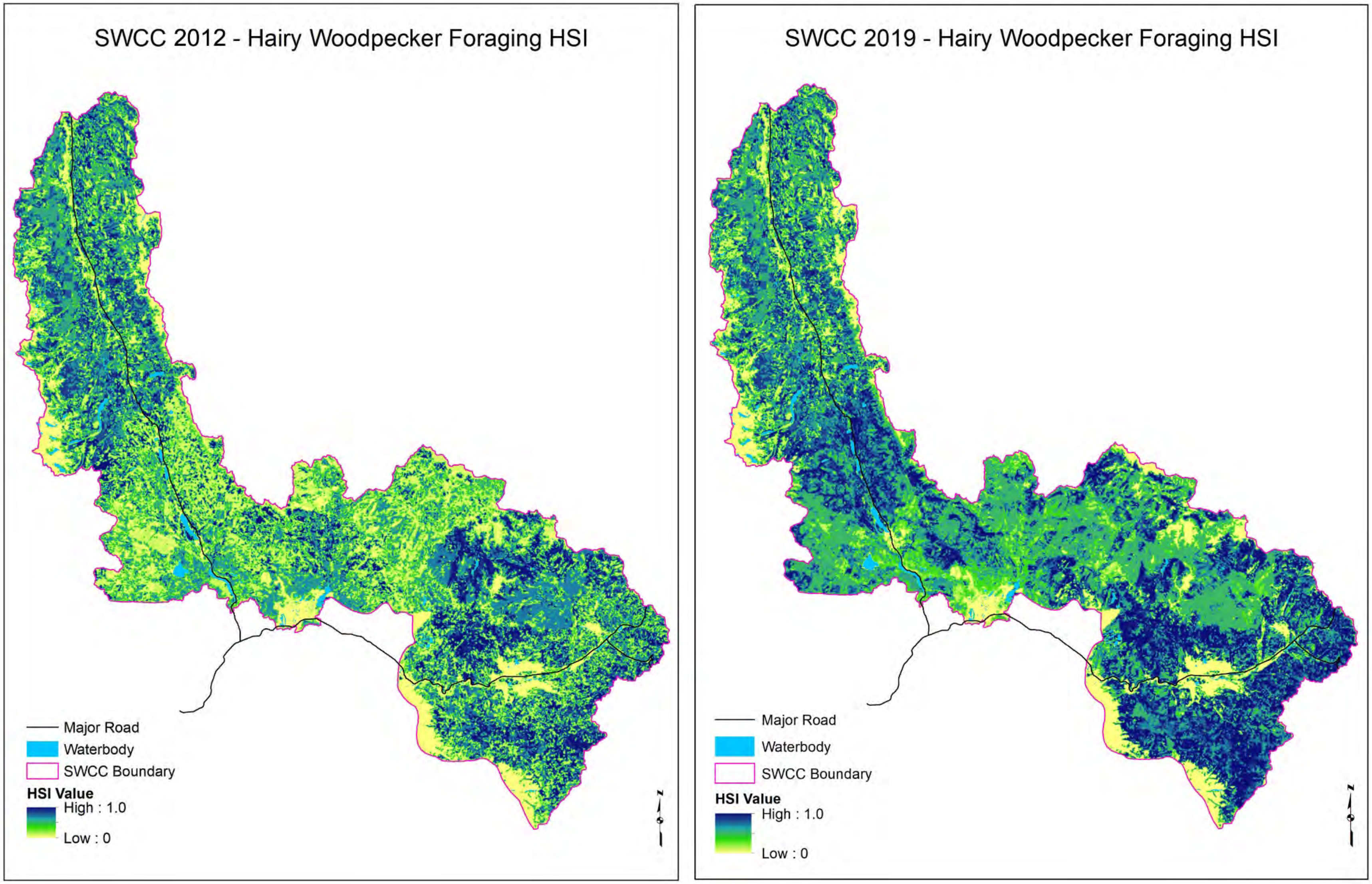


Figure 3. Comparison of foraging quality for hairy woodpecker between 2012 and 2019 in SWCC project area.

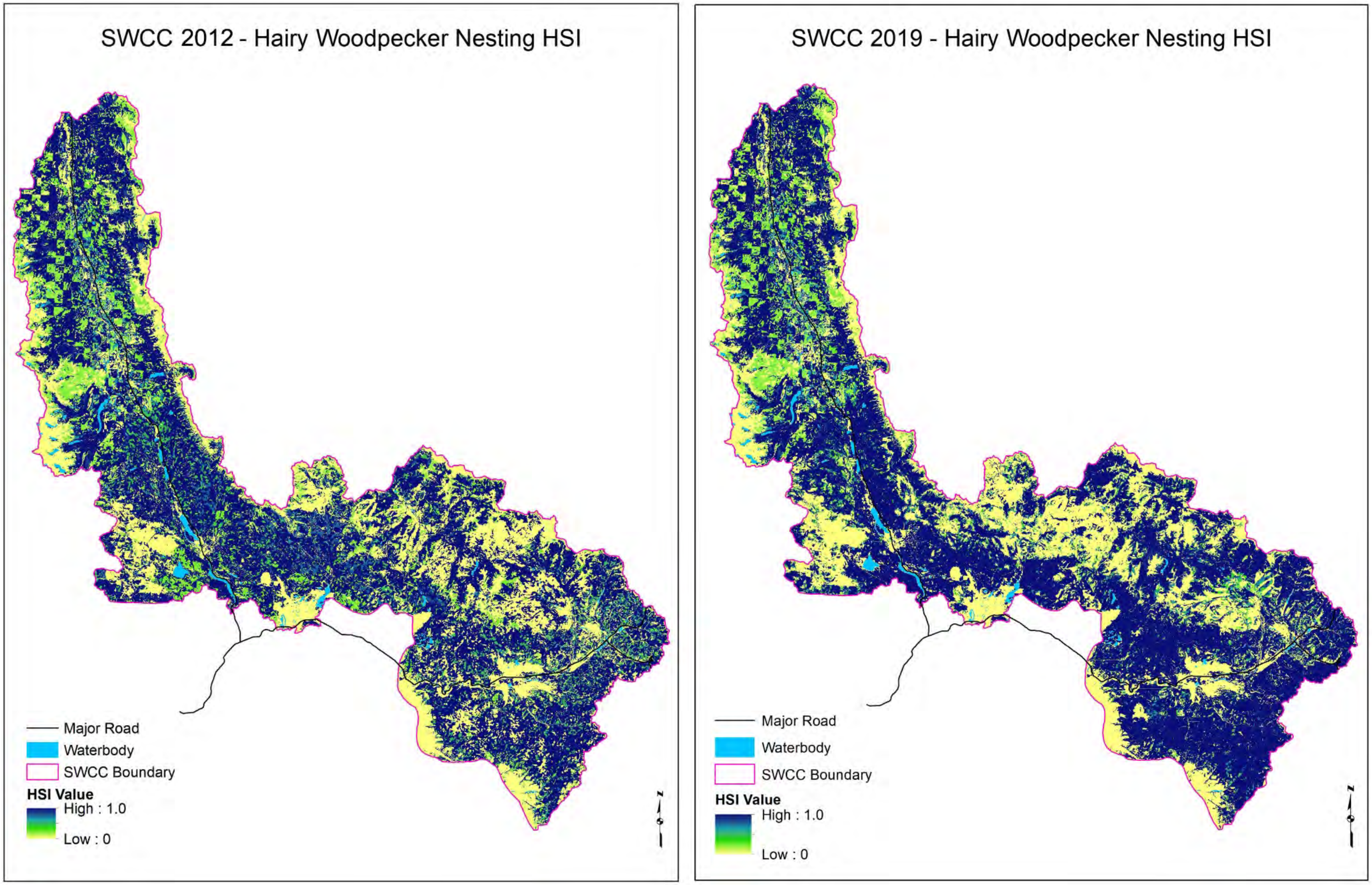


Figure 4. Comparison of nesting quality for hairy woodpecker between 2012 and 2019 in SWCC project area.

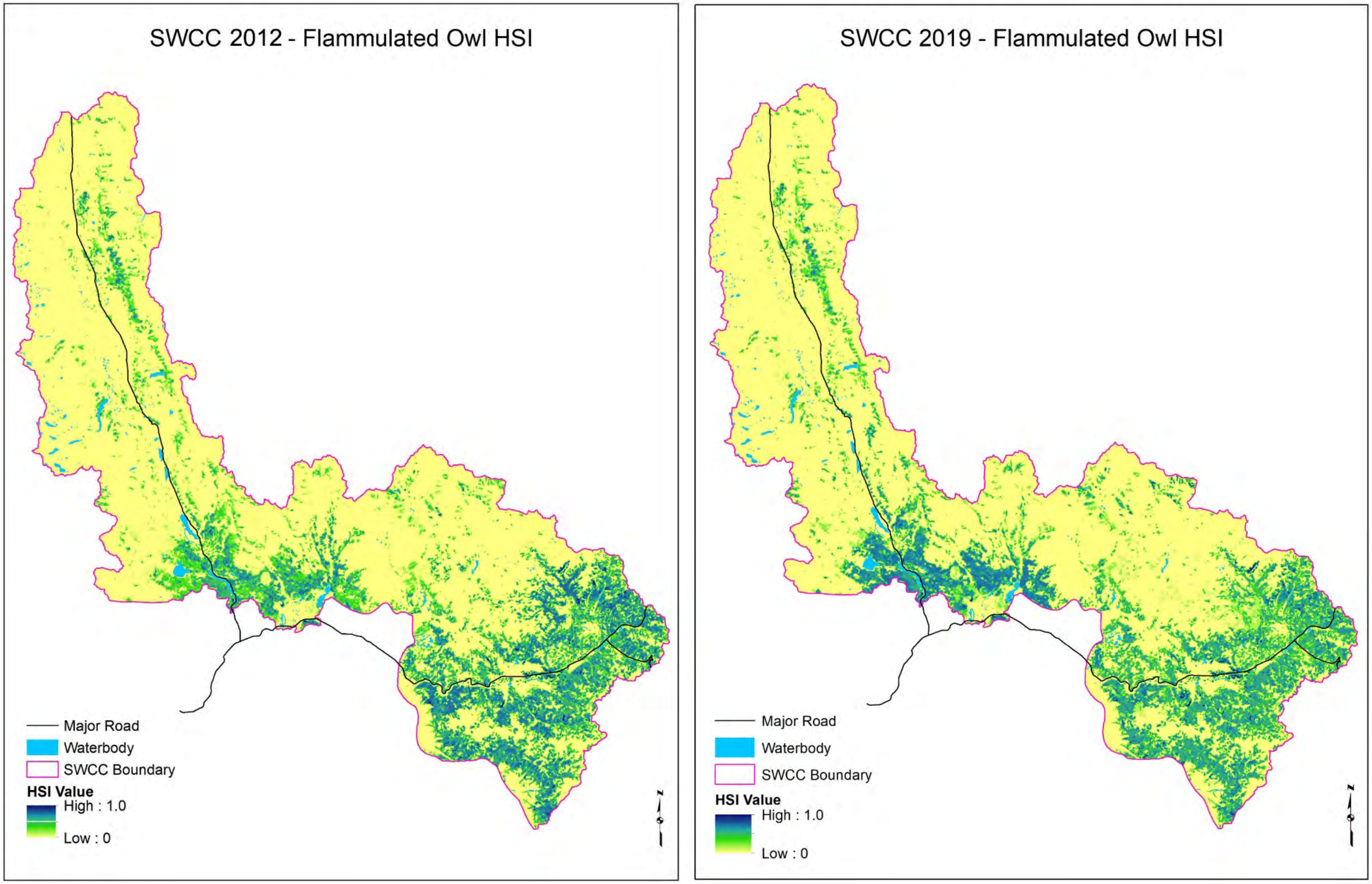
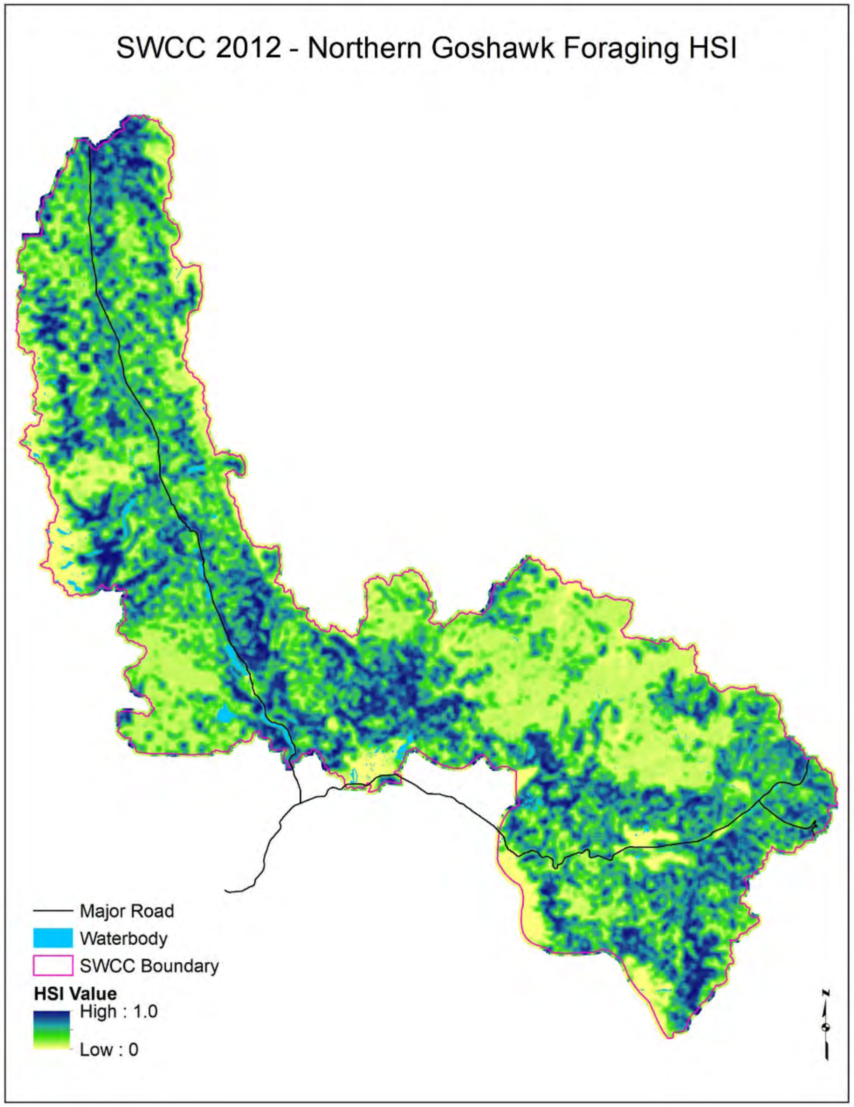


Figure 5. Comparison of habitat quality for flammulated owl between 2012 and 2019 in SWCC project area.



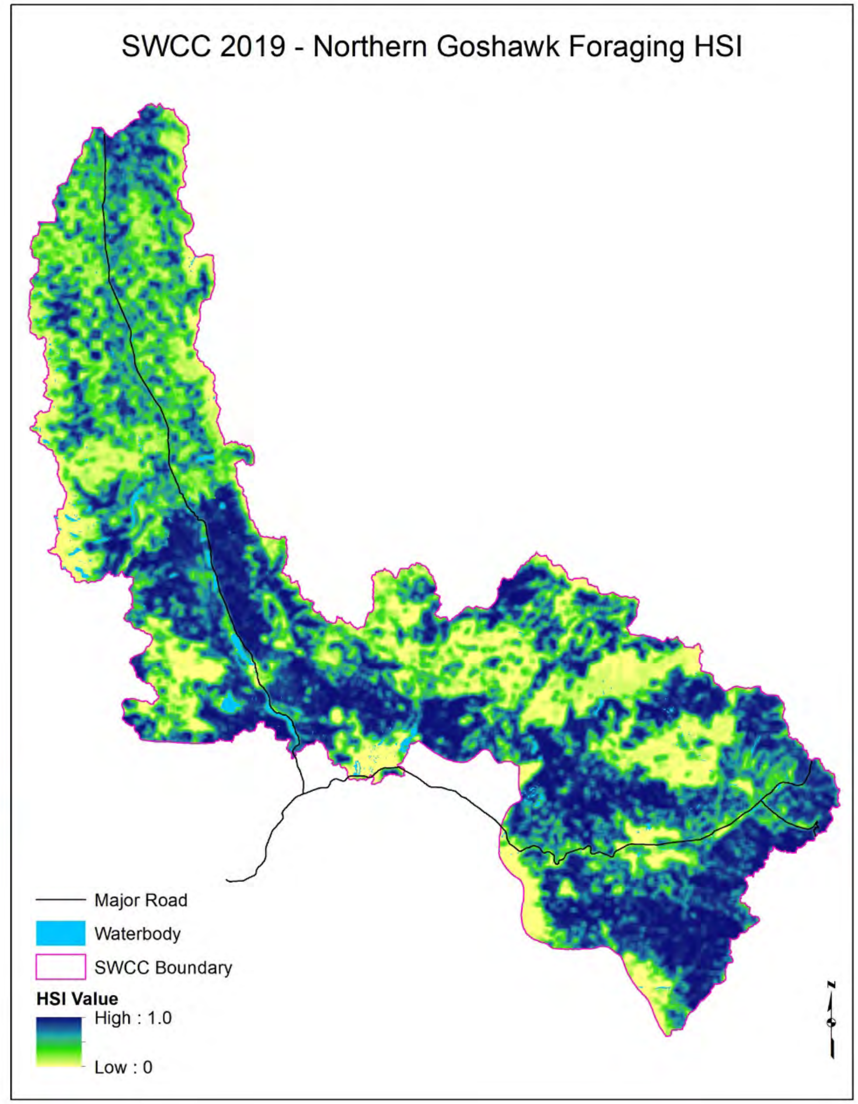


Figure 6. Comparison of foraging habitat quality for northern goshawk between 2012 and 2019 in SWCC project area.

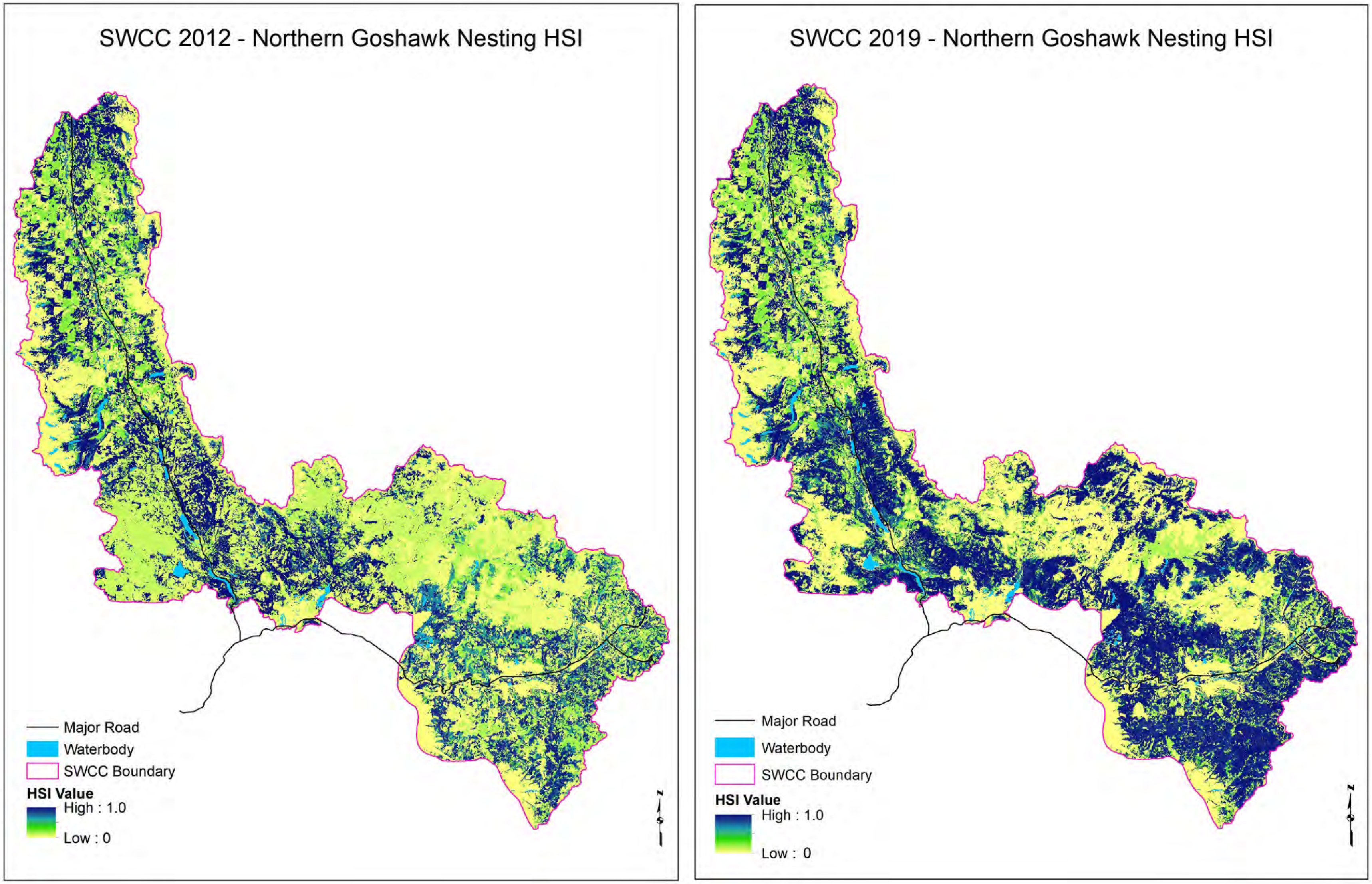


Figure 7. Comparison of nesting habitat quality for northern goshawk between 2012 and 2019 in SWCC project area.

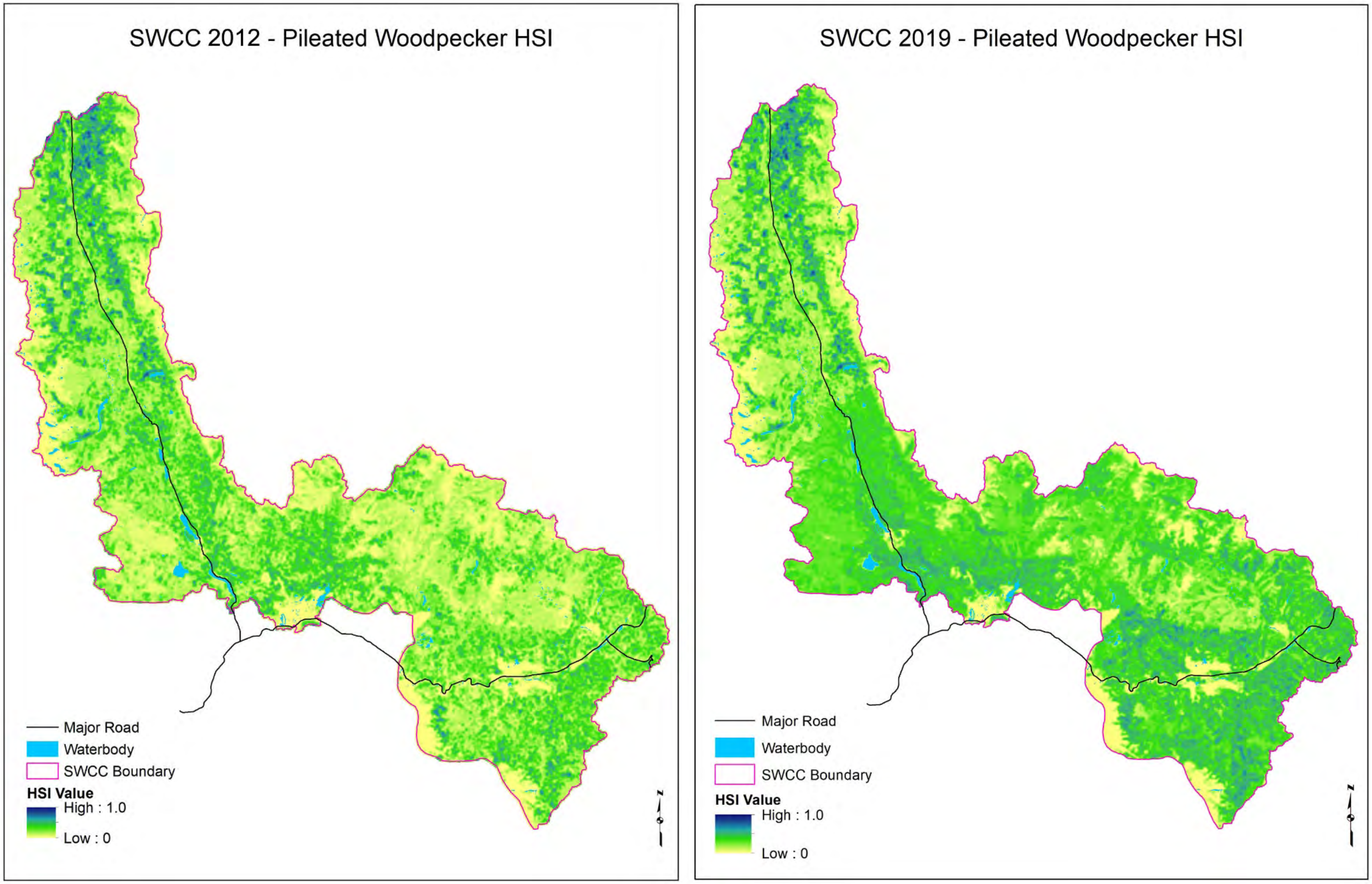


Figure 8. Comparison of habitat quality for pileated woodpecker between 2012 and 2019 in SWCC project area.

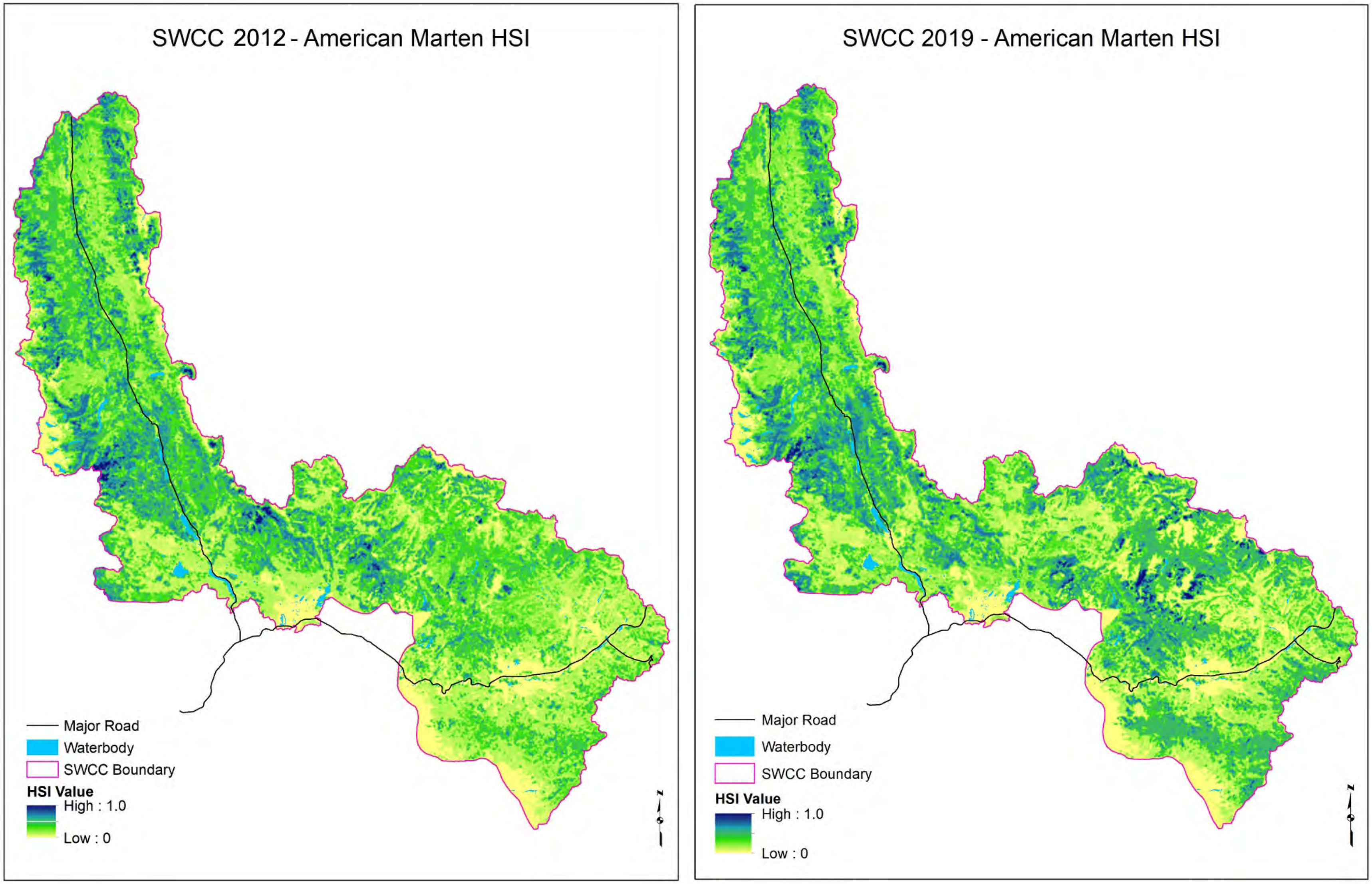


Figure 9. Comparison of habitat quality for American marten between 2012 and 2019 in SWCC project area.

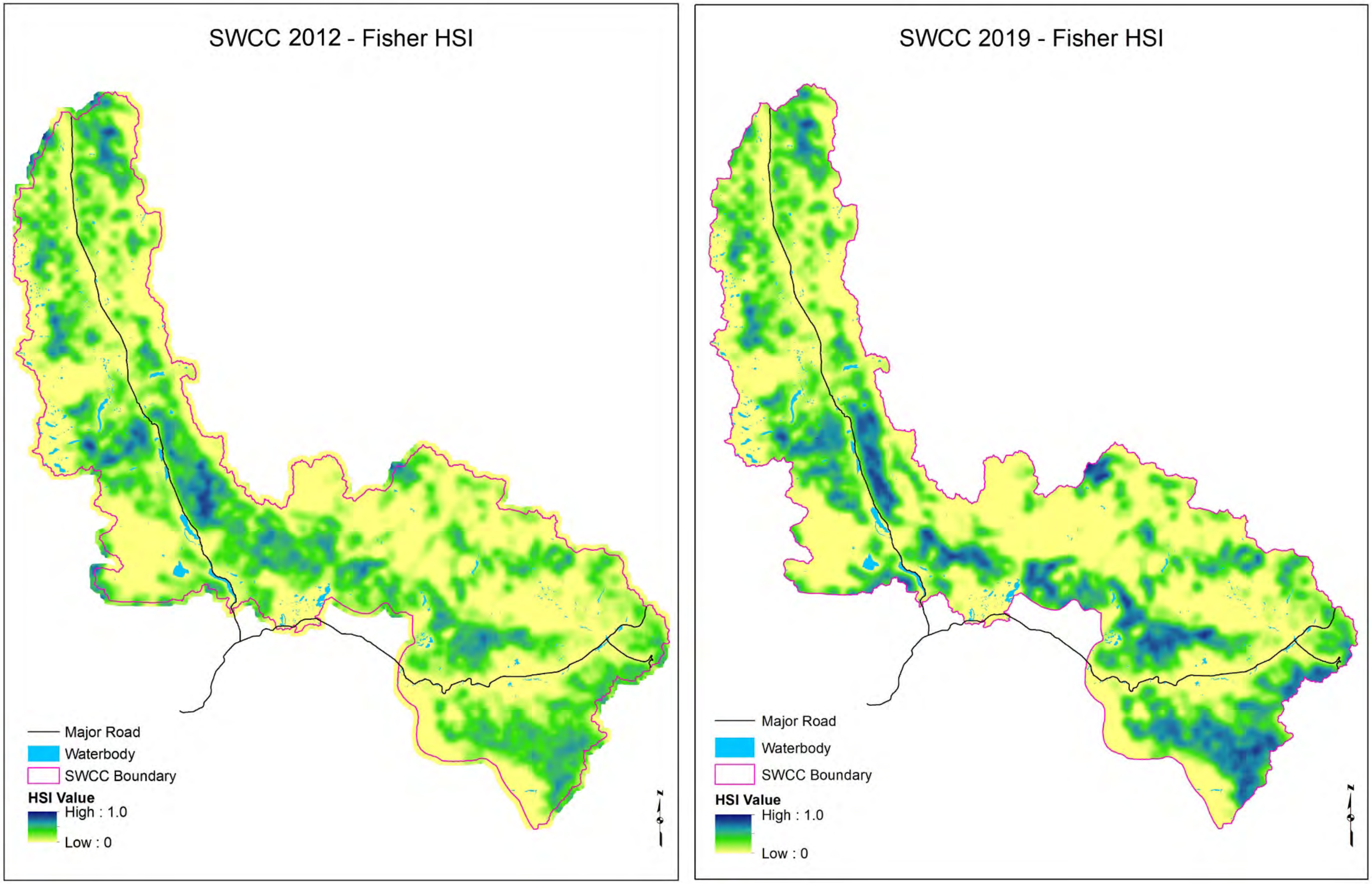


Figure 10. Comparison of habitat quality for fisher between 2012 and 2019 in SWCC project area.

Table 3. Comparison of total number of potential home ranges of varying quality for 6 modeled species in the SWCC project area, and the calculated 95% confidence intervals around the mean values for 2012 and 2019 modeling efforts.

	Habitat Quality					
	High		Medium		Low	
	2012	2018	2012	2018	2012	2018
Species	**Mean (CI[2])**	**Mean (CI)**	**Mean (CI)**	**Mean (CI)**	**Mean (CI)**	**Mean (CI)**
Hairy Woodpecker	3451.0 (232.2)	5612.7 (8.4)	1665.4 (262.3)	333.0 (6.7)	135.6 (37.6)	13.0 (1.4)
Flammulated Owl	38.6 (8.0)	48.3 (0.9)	1593.4 (272.5)	2297.37 (4.17)	1660.4 (119.3)	1384.7 (3.3)
Northern Goshawk	106.2 (11.3)	446.3 (0.9)	264.6 (9.0)	74.0 (2.8)	38.6 (4.9)	3.3 (0.5)
Pileated Woodpecker	0.4 (0.7)	1.0 (0)	52.0 (13.4)	168.0 (0)	503.6 (20.9)	623.3 (2.1)
Fisher	0.6 (0.4)	3.0 (0)	15.4 (4.7)	33.7 (3.1)	33.2 (2.7)	14.0 (3.6)
American Marten	2.4 (0.4)	2.0 (0)	59.0 (8.0)	124.7 (0.5)	239.6 (8.2)	343.3 (1.7)

[2]CI - Confidence Interval

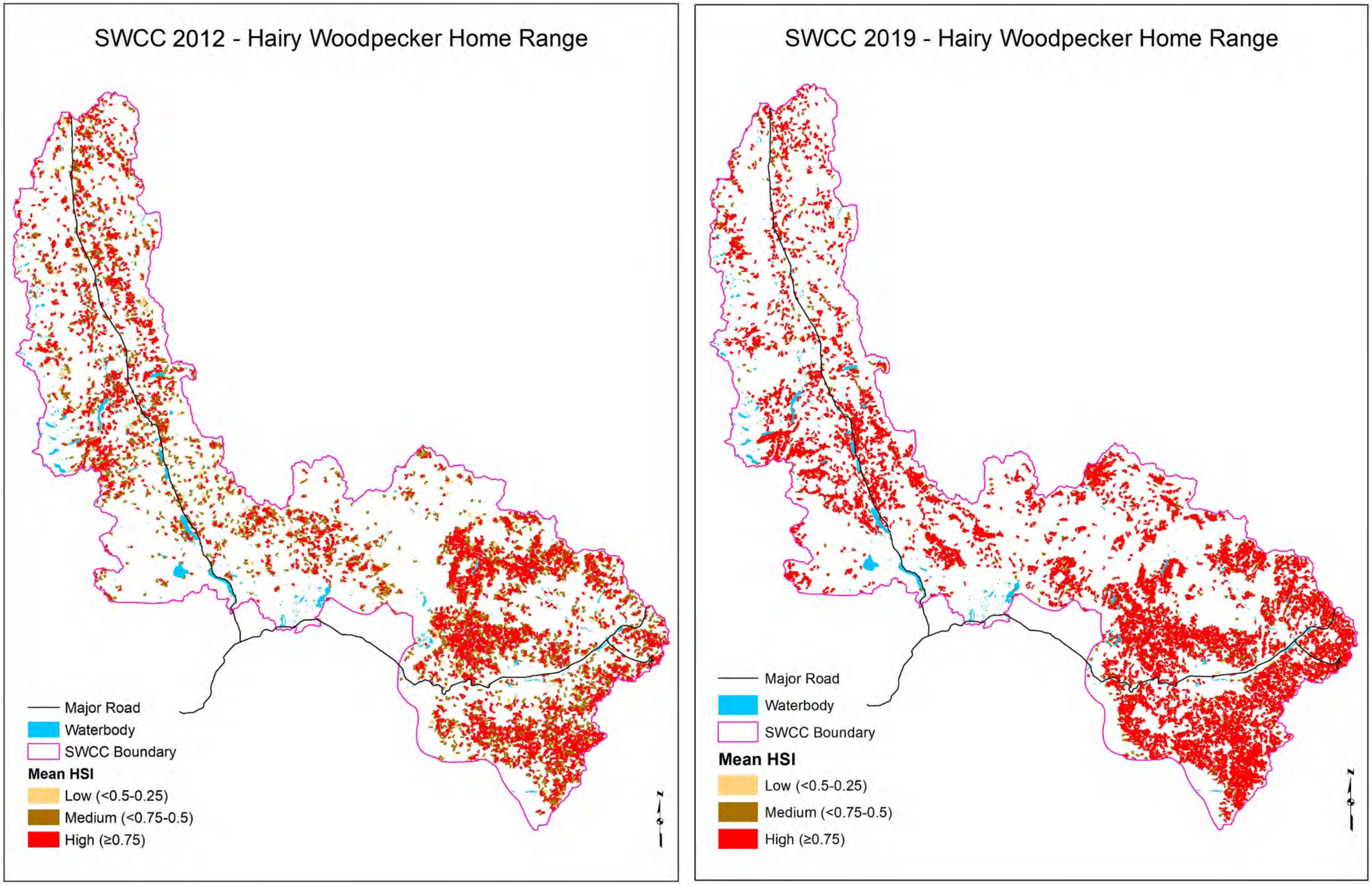
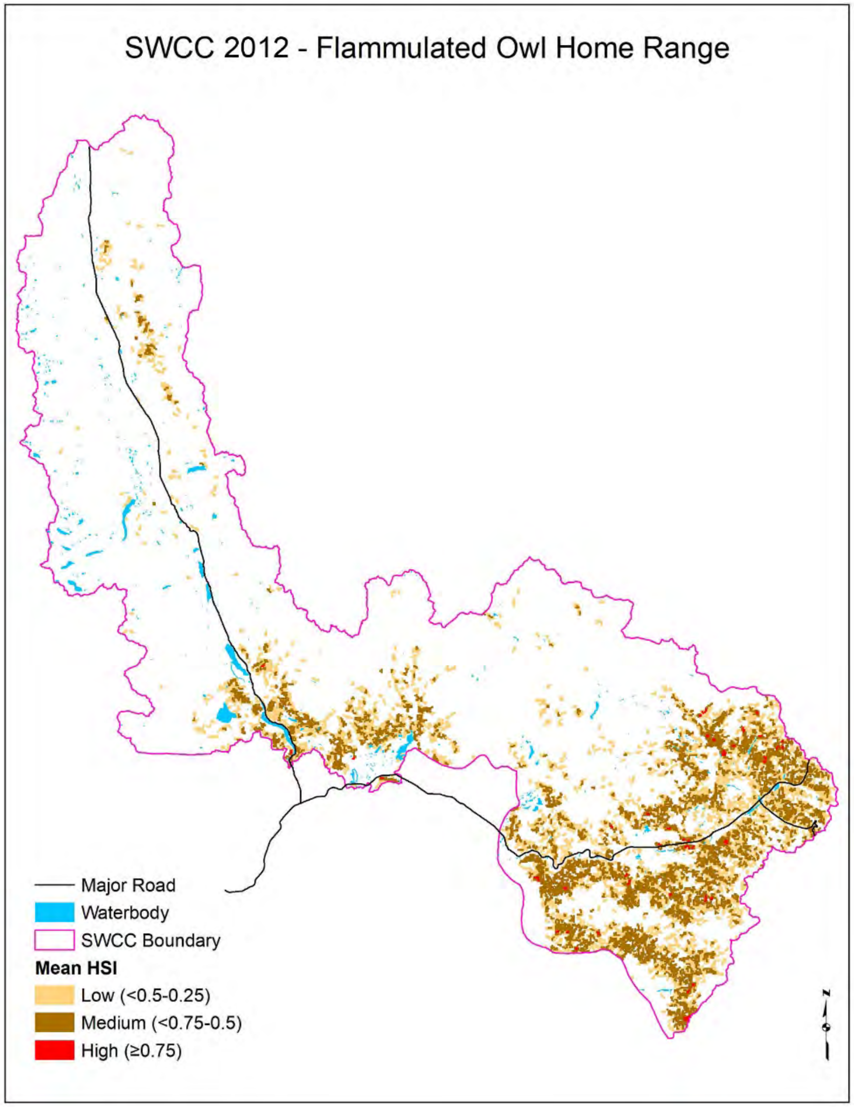


Figure 11. Comparison of potential home ranges of low, medium, or high quality for hairy woodpecker between 2012 and 2019 in SWCC project area.



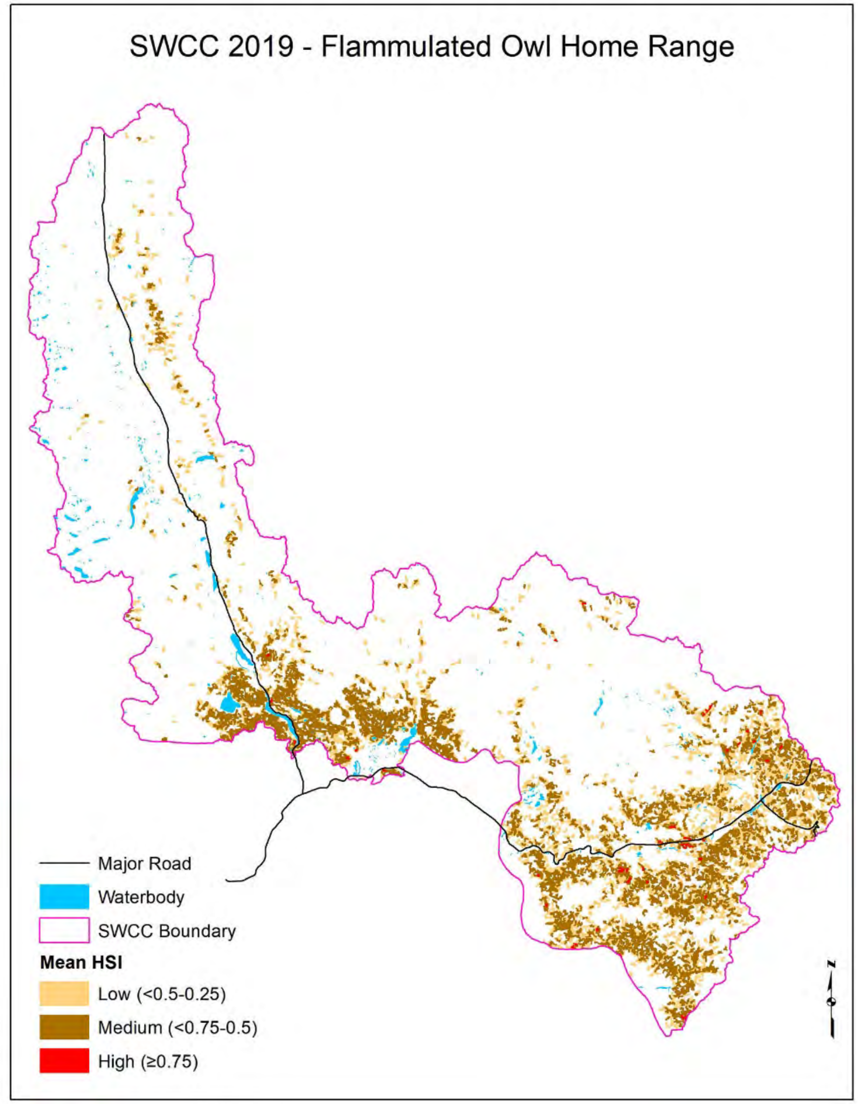


Figure 12. Comparison of potential home ranges of low, medium, or high quality for flammulated owl between 2012 and 2019 in SWCC project area.

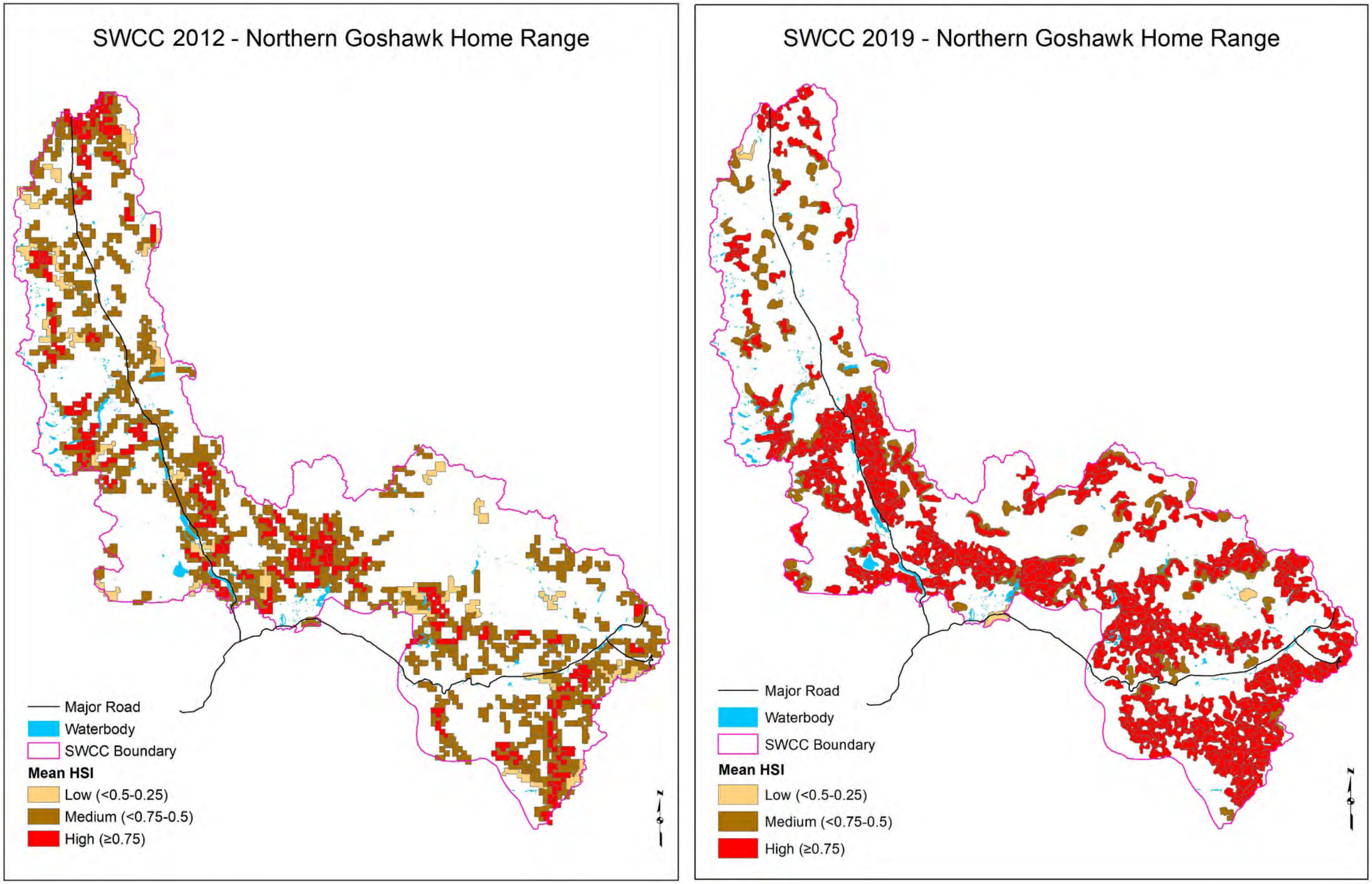


Figure 13. Comparison of potential home ranges of low, medium, or high quality for northern goshawk between 2012 and 2019 in SWCC project area.

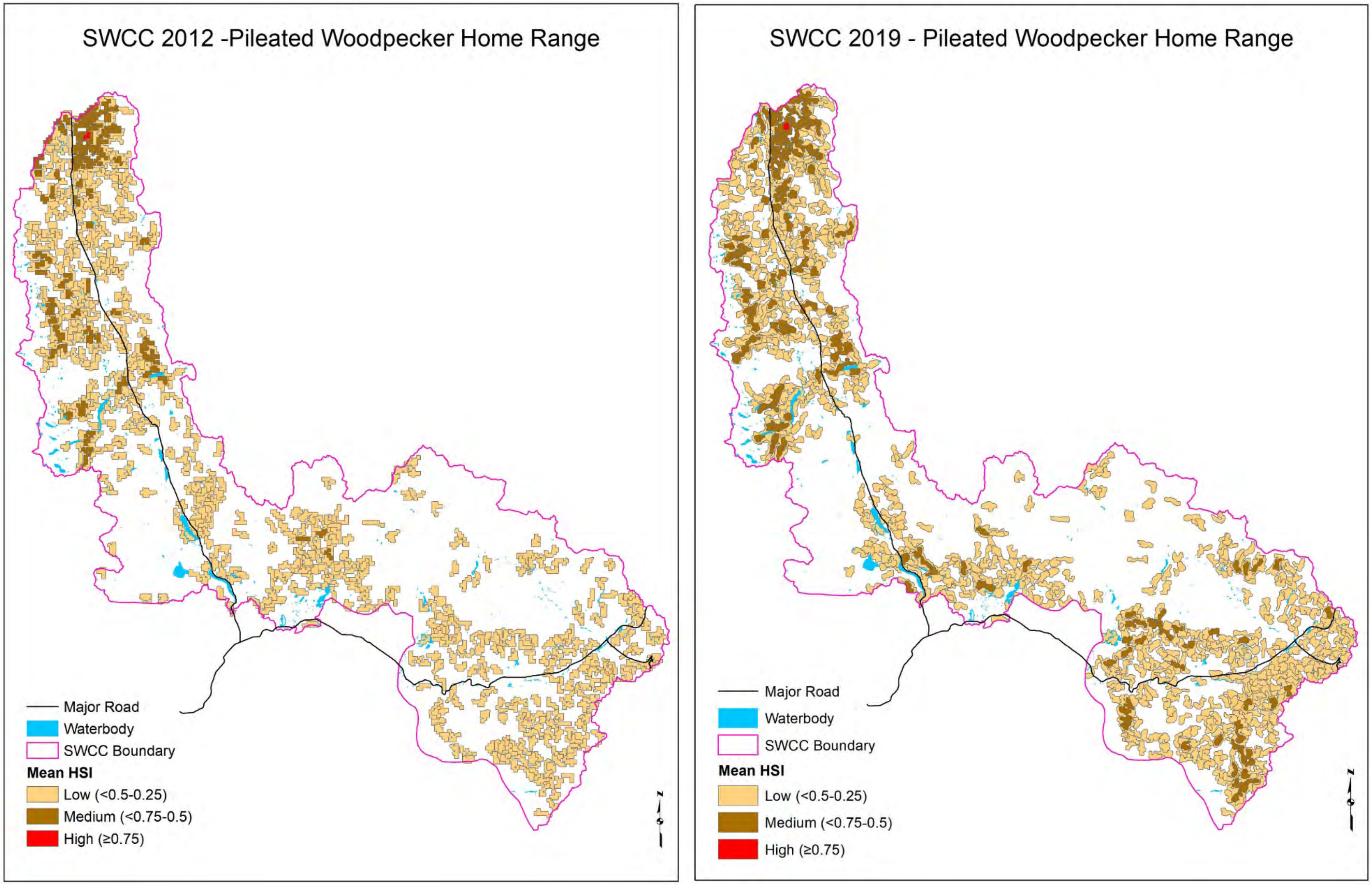


Figure 14. Comparison of potential home ranges of low, medium, or high quality for pileated woodpecker between 2012 and 2019 in SWCC project area.

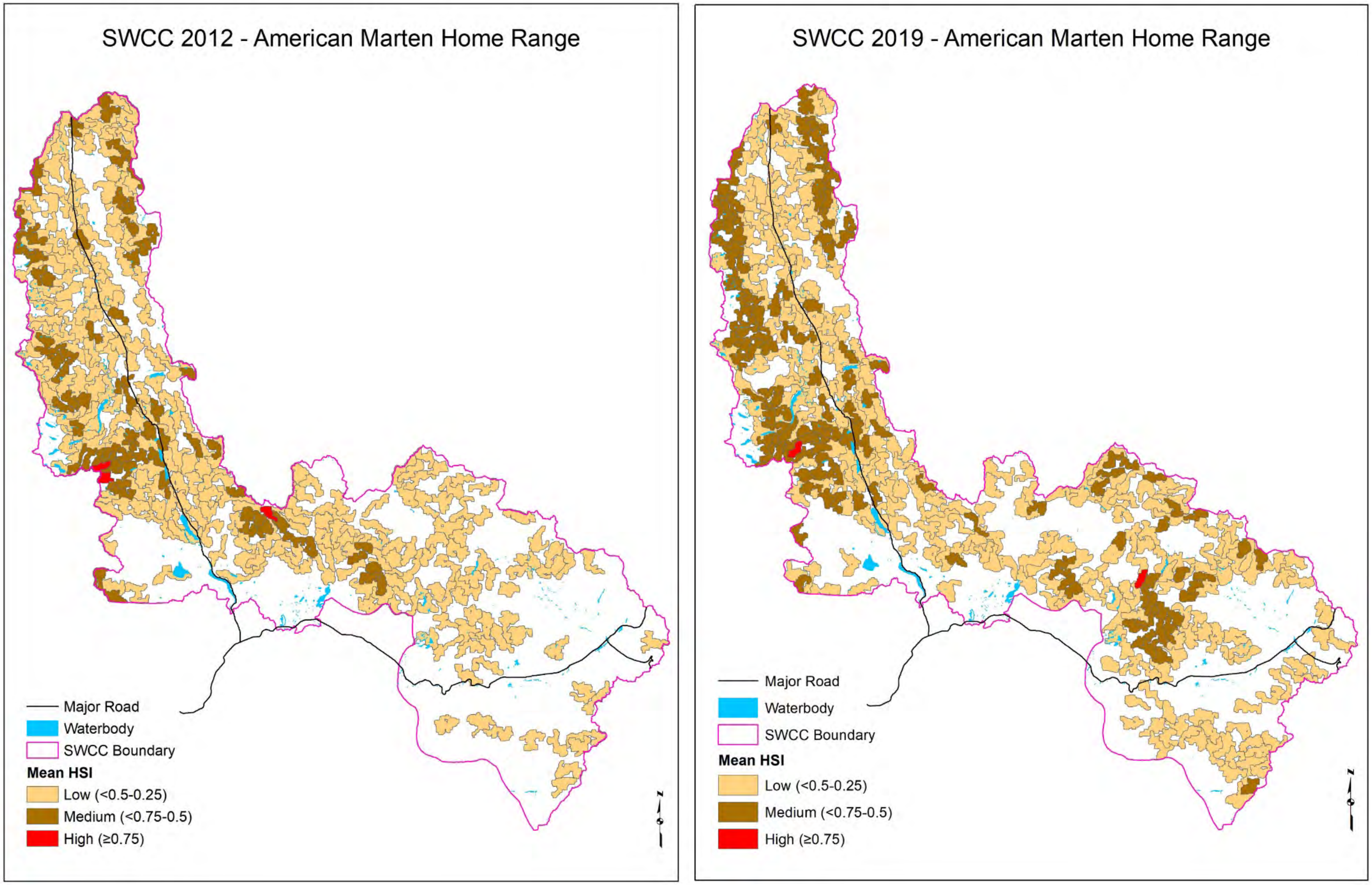


Figure 15. Comparison of potential home ranges of low, medium, or high quality for American marten between 2012 and 2019 in SWCC project area.

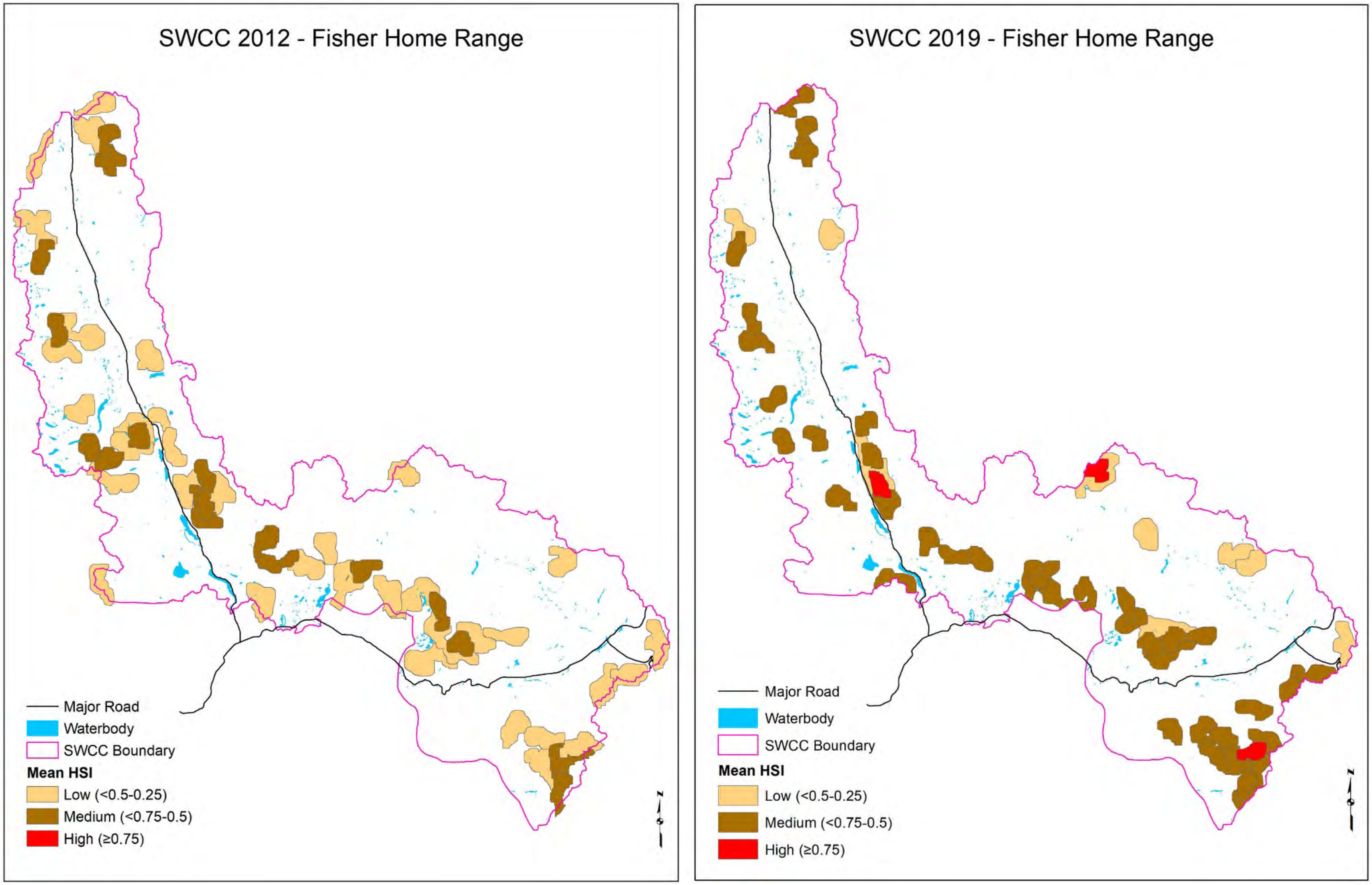


Figure 16. Comparison of potential home ranges of low, medium, or high quality for fisher between 2012 and 2019 in SWCC project area.

The confidence intervals for the home range estimates listed in Table 3 show some variability for a few of the estimates, such as the high-quality home ranges for pileated woodpeckers and fisher, but overall, they are not that large. In this analysis the main source of variability is the way HOMEGROWER generates expected home ranges. The starting points, or seeds, are random and result in slightly different results each time the model is run. However, HOMEGROWER pulls together a large number of pixels in the GIS, aggregating habitat quality values across these pixels. So the net effect of this process results in a more consistent evaluation of habitat quality expressed as number of home ranges of varying quality. These differences are magnified when the input data change as is the case when comparing the 2012 and 2019 results. Even though portions of the landscape (in this case the Flathead NF) did not change, the changes to the rest of the landscape result in variation in the starting seed locations. The tabular run results are the most important point of comparison as the home range maps are just a representation of potential home range locations. This finding is consistent with results of other studies that have reported on a dampening of variation extremes noted as map extents expand (Roloff et al. 2009, Wu 2004). One implication of this is that for species that have larger home ranges, the likely influence of potential variation in habitat attributes will have less influence on a final output, such as a home range quality rating than for species with smaller home ranges (Linden 2006). This means that species such as hairy woodpeckers can be expected to have more variability in the mapped home range qualities than species with larger home ranges, such as fisher.

Habitat Changes

As discussed previously, updated VMAP data were available for the portions of the project area located on the Lolo and Helena National Forests. There were not updated data available for the Flathead National Forest portion of the project area. In some cases the Lolo and Helena VMAP data still use the same base imagery as previous VMAP versions, but was updated using disturbance data through 2018 to result in new disturbance classes when compared to the VMAP data used in the 2012 SWCC analysis.

There were two notable differences that occurred throughout the landscape when comparing 2012 and 2019 HOMEGROWER results. These were the addition of many acres of recently disturbed areas due to wildland fire and an overall movement of many sites into more closed canopies due to forest growth projections. As can be seen in the results, habitat quality improved for most species in this analysis either due to the increases in small snag density (hairy woodpecker) or increases in stand size and canopy closure (fisher, marten), or an increased mixture of disturbed areas intermixed with larger trees and increased canopy closure (goshawk). Species such as flammulated owl and pileated woodpecker showed the least change between model years because of their reliance on lower elevation, drier stands with a very large tree component.

Conclusions

This update to the 2012 SWCC project shows that recent disturbances and other changes to the project area have generally resulted in higher quality habitat for most of the species of interest. We believe these changes are real and not just a byproduct of the data sources and analysis methods. However, the changes were undoubtedly driven primarily by the large wildfires that occurred between sampling years, and less so from CFLRP treatments. The woodpeckers in particular likely benefited from the recent fires. Species showing less change either require very large habitat patches (fisher and marten) or require habitat conditions that are more difficult to capture with remote imagery (flammulated owl and pileated woodpecker). The potential to better quantify fine scale vegetation characteristics could result in more accurate predictive models.

This update also reveals where a number of data gaps still exist. Many classification categories lack plot data that can provide estimates of the key habitat variables needed to evaluate habitat quality. Even when plots are available for a classification category, they typically lack sufficient numbers to generate good estimates of mean values and confidence intervals. For this analysis, the influence of VMAP data and GIS map accuracy on the variance of habitat variables could not be assessed because of the restrictions in use of FIA data. However, these results do suggest that additional field sampling of forest vegetation conditions are needed to produce reliable estimates of habitat variables to drive habitat suitability models.

As stated by Roloff et al. (2009:312), "Habitat maps generated by a GIS are often viewed by decision makers and the general public as absolute truth regardless of accuracy assessments, summaries of data use limitations, or spatial and temporal scale considerations. We caution wildlife-habitat relationship modelers to consider how the inherent properties of data sets and uncertainty from all these sources should be included and portrayed in their analyses." Evaluations of wildlife habitat suitability that fail to consider the potential variability resulting from vegetation classifications and sampling methods, especially those generated from remote-sensing, are likely to misrepresent the actual habitat quality occurring in a landscape. Habitat assessment tools are critical components of landscape analyses and are needed to evaluate potential changes to habitat quality that can result from vegetation treatment and management actions. However, careful attention to the best available information and a clear understanding of data limitations relative to the objectives of the habitat assessment tools is a requirement for their appropriate use.

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Appendices: Wildlife Habitat Suitability Models

Fisher (Pekania pennanti)

The fisher is a medium sized forest carnivore that was nearly extirpated from the intermountain west, but was reintroduced in the mid 1960's (Powell 1993, Powell and Zielinski 1994). The current range of fisher is shown in Figure A-1.

Figure A-1. Current range of the fisher in North America (Patterson et al. 2005).

Habitat Description

In general, fisher habitat quality is highest in late-successional conifer treatment sites (USFWS 2004). Specifically, fishers select for treatment sites with canopy cover >50% (preferably 80-100%), large diameter trees (>18.5 in. DBH), multi-story treatment sites, and high levels of coarse woody debris (Jones 1991, Powell 1993, Powell and Zielinski 1994, Purcell et al. 2009, Schwartz et al. 2013). Some studies, particularly in the southern Sierra, have found a preference for closer proximity to riparian areas (Jones 1991, Powell and Zielinski 1994, Zielinski et al. 2004). In north-central Idaho, old treatment sites dominated by grand fir and Engelmann spruce were preferentially selected in the summer (Jones 1991, Jones and Garton 1994) while both younger and older treatment sites were used in the winter. Fishers avoid non-forested areas (Jones and Garton 1994, USFWS 2004, Weir and Corbould 2010). Schwartz et al. (2013) studied fishers in eastern Idaho and western Montana. They reported that fishers selected areas that had the maximum DBH at treatment site scales and the largest proportion of treatment sites with large trees at the landscape scale as well as selecting areas with higher canopy cover at the landscape scale. They noted, however, that there was support for treatment sites with structural diversity including a mix of tree sizes. They also reported that fishers avoided ponderosa pine and lodgepole pine forests, and selected areas with snags and cavities present. Schwartz et al. (2013) did not find a preference for high canopy cover at the treatment site level in their study area, but noted that average treatment site conditions in their area had greater than 50% canopy cover.

Fishers require prey populations in proximity to two primary habitat features, resting sites and denning sites (Powell and Zielinski 1994). Resting sites provide protection from weather and predators and are preferred in close proximity to areas containing prey (Zielinski et al. 2004, Purcell et al. 2009, Aubry et al. 2013, Schwartz et al. 2013). It has been suggested that resting habitat is more important than foraging or traveling habitat with fisher consistently selecting mature trees even when in younger aged treatment sites (USFWS 2011). Aubry et al. (2013) conducted a meta-analysis of studies that collected habitat information for fisher resting sites in the western U.S. and Canada. They reported that "selected sites for resting had steeper slopes, cooler microclimates, denser overhead cover, a greater volume of logs, and a greater prevalence of large trees and snags than were generally available." Purcell et al. (2009) reported that in the southern Sierras that fisher resting sites occurred more frequently on steeper slopes and closer to streams. Zielinski et al. (2004) studied fisher resting sites in several locations in California. They found that fishers selected resting sites that had dense canopies, large maximum tree sizes, and steep slopes. Preferred nesting structures were large trees with live conifers averaging 117 cm DBH, conifer snags averaging 120 cm DBH, and hardwoods averaging 69 cm DBH. Zielinski et al. (2012) evaluated resting sites of fishers in northern California and reported that the best fit model included variables for canopy cover, tree age, total basal area, volume of large wood, and basal area of hardwoods.

Females use den sites for raising young. Den sites typically occur in large snags or live trees that offer a cavity or other structure for a female to den (Weir et al. 2012, Niblett et al. 2015). In Northeast British Columbia, all located den sites occurred in cavities in large aspen or balsam poplar (Weir et al. 2012), while a previous study in British Columbia (Weir and Harestad 2003) found den sites in large black cottonwoods.

Fisher et al. (2013) reported that fisher and marten appeared to avoid occupying the same area. The mechanism for this avoidance (i.e., was habitat selection different, was one species driving the other away, or was it mutual avoidance) was not determined, but if fisher were avoiding marten, this could be an additional factor to consider beyond habitat quality. Raine (Raine 1983) reported that fisher avoided soft thick snow, while marten was not limited by these conditions, providing one explanation of why fisher may not be found in some areas supporting marten, at least during those times and locations where deep soft snow may be present. Halsey et al. (2015) identified potential fisher reintroduction sites and reported that these sites should be outside of bobcat occurrence areas, as this felid predator on fisher could compromise reintroduction success. LaPoint et al. (2015) hypothesized that reductions in mesopredator populations that could effectively compete for larger food items of fisher could affect the ability of fisher to exist or expand into areas containing substantial competing mesopredator populations. Wengert et al. (2013) used molecular methods to confirm that bobcat, coyote, mountain lion, and domestic dogs were responsible for predation on fishers. These studies all point to additional factors that could influence fisher occurrence or densities in an area beyond the existing habitat quality identified for fisher. Similarly, trapping of fisher has been identified as a significant factor in their past extirpation from many areas, including Montana, and the residual influences of these past impacts can have a strong effect on fisher locations (Weckwert.Rp and Wright 1968, Vinkey et al. 2006).

Allen (Allen 1983) developed a fisher HSI model as did Olsen et al. (1999). Winter habitat is generally considered the limiting factor for fisher. Optimum winter habitat was assigned to mature treatment sites with high tree canopy cover, and a diverse understory. Proulx (2011) tested a winter fisher habitat model by examining locations of fisher tracks in British Columbia compared to estimated values of treatment sites. He rated the various cover types on British Columbia's vegetation mapping system for fisher habitat quality using a point system where he assigned to following values: "forest disturbance (presence: 0; absence: 4 points), age (≤ 60 years: 0; 61-80: 1; 81-100: 2; 101-120: 3; and >120: 5 points), presence of mature or old structural stages (2 points), basal area ≥ 20 m^2/ha in mature trees (1 point), ≥ 30% canopy closure (2 points), shrub cover (0%: 0; 5-20%: 1; 20-40%: 2; > 40: 3 points), and dbh ≥ 27.5 cm: 1 point." He found 66% of fisher tracks in polygons that were rated excellent quality and another 23% in polygons rated high quality and concluded that the rating system worked effectively for ranking fisher habitat quality within the project area. On average, treatment sites with fisher tracks were 138 years old, had canopy closures of 54%, 38 m^2/ha (166 ft^2/ac) of basal area, average DBH of 27.8 cm (10.9"), and 11% shrub cover.

Zielinski et al. (2006) developed a fisher habitat model for California designed to be used with forest inventory plot data. This model included the following variables: average canopy closure, basal area of trees <51 cm DBH, average hardwood DBH, maximum tree DBH, percentage slope, and the DBH of the largest conifer snag. The model worked fairly well on the modeled data set, but less well on an independent data set.

Sauder and Rachlow (2014) studied fisher habitat use in northcentral Idaho using radio-telemetry. They found that fishers selected areas with the largest patches of mature forest that were in close proximity to similar patches and avoided areas with open areas. They found a rapid decrease in habitat selection for home ranges with increasing amounts of open area, with 20% of an area in open conditions having a probability of use of less than 0.2, a similar finding to that of Weir and Corbould (2010). The variable they found to best explain occupancy was a proximity index for mature forest that considered not only amounts of mature forest but also its configuration within a landscape, selecting areas that had large patches of mature forest arranged in complex but highly connected patches. They suggested that high quality fisher habitat consists of >50% mature forest with less than 5% open areas. Weir and Corbould (2010) reported that a 5% increase in open areas (recently logged areas or wetlands in their study area) would decrease fisher occupancy by 50%. Sauder and Rachlow (2015) examined fisher core use areas within home ranges of fishers and found that while fishers heavily utilized mature forests, they selected for areas that had habitat heterogeneity rather than uniform conditions. They attributed this to the need for fishers to have areas containing good den or rest sites mixed with areas that supported good prey populations. A landscape edge variable best accounted for this core area heterogeneity. Weir and Harestad (2003) found that fishers selected habitat components at treatment site, patch, and element scales. When desired habitat at larger scales was lacking, they reported that fishers could compensate by selecting desired elements of higher quality at finer scales, for example selecting larger trees in a treatment site for resting or denning sites where the treatment site had smaller that optimal tree sizes.

Olson et al. (2014) modeled existing and predicted future fisher habitat in Idaho and Montana. They used LANDFIRE vegetation maps for their model mapping, and used selected variables including tree height in 3 classes, canopy cover in 5 classes, and existing vegetation in terms of dominant species. Their model found that tree heights were the best predictor of fisher occurrences, likely a relationship to mature forests required by this species as reported in numerous other studies. They also found measures of proximity to riparian areas to be important. The coarse scale of these variables and their mapping in this study reduces the applicability of their model to habitat interpretations. The primary focus of Olson et al. (2014) was to evaluate potential future habitat conditions as influenced by climate change, with their modeling outputs considered at broad landscape scales and evaluating the role of future climate changes.

Home ranges of fishers have been estimated in several studies. Sauder and Rachlow (2014) reported home ranges of males averaged 98 km^2 while females averaged 49 km^2 in their study in northcentral Idaho. Sauder and Rachlow (2015) found that "core areas" in their Idaho study area averaged 33 km^2 for males and 19 km^2 for females. Weir (1995) reported winter home ranges for female fishers in British Columbia to be 26 km^2 during one year and 25 km^2 the next, with what he described as core use areas averaged 4.4 km^2 and 5.4 km^2 for the two years. Weir et al. (2009) studied home range sizes for fishers in British Columbia and found that females ranged from 10-81 km^2 with a mean of 38 km^2, while male home ranges ranged from 49-225 km^2 with a mean of 161 km^2. Weir et al. (2003) estimated that winter home ranges of females averaged 25 km^2 in their study area in British Columbia. Davis et al. (2007) used a home range size of 10 km^2 in modeling and evaluating habitat for fishers in California.

HSI Model

We developed an HSI model for fisher habitat based on the variables most applicable to fisher habitat in the northern Rockies. The primary variables we selected were tree canopy cover, overstory DBH, shrub cover, percentage of true firs, spruce, larch, and western red cedar in a treatment site, and a measure of landscape edge as reported by Sauder and Rachlow (2015).

The first variable is tree canopy cover (Figure A-2). Fishers have been found to avoid open areas and their prevalence increases with decreasing amounts of open areas in the landscape (Weir and Corbould 2010). In a regional assessment of fisher that reviewed studies through the Pacific Northwest, including research conducted in Montana and Idaho, it was shown that the most consistent predictor of fisher occurrence has been a preference for areas with a minimum of 30% tree canopy cover with use increasing with amounts of canopy cover both in Idaho (Jones and Garton 1994) and in the lake states (Thomasma et al. 1994). Proulx (2006) reported that fisher in British Columbia used treatment sites with 30-60% canopy cover. Fisher in southeast British Columbia selected treatment sites with >40% canopy cover (mean of 53%) in both summer and winter (Fontana and Teske 2000). Purcell et al. (Purcell et al. 2009) found that canopy cover was the most important variable in identifying fisher resting sites in southern Sierras. They reported selected canopy cover in the 55-60% range depending upon method of measurement.

The second habitat variable for fisher is mean DBH of overstory trees (Figure A-3). This variable helps address treatment site age and successional state as fisher occurring in heavy snow regions have been shown to prefer older, mid- to late-successional treatment sites (Powell and Zielinski 1994). Fisher prefer treatment sites with mean tree diameter >8” and suitability increases with tree diameter (Jones 1991, Jones and Garton 1994, Thomasma et al. 1994, Proulx 2006). Niblett et al. (2015) similarly found that fishers selected large trees for den sites in their study area in California, but that only a small proportion of an area needed to be in a late successional state with these large trees present. In their area, home ranges of 5 female fisher contained 25% of the plots containing large trees compared to only 6% of the plots containing large trees in the overall forest. Swartz et al. (2013) found that landscapes selected for use by fisher in Idaho averaged 47% treatment sites of large trees while comparable available landscapes only had 29% of treatment sites with large trees. Purcell et al. (2009) found that fishers selected the largest available trees for resting sites in the southern Sierras.

The third variable is canopy cover of shrubs ≥3 ft. in height (Figure A-4). In the Pacific states and northern Rocky Mountains fisher have been shown to prefer multi-layer treatment sites and areas with high canopy cover (Powell and Zielinski 1994, Weir and Harestad 2003). Higher levels of horizontal cover created by shrubs and/or small trees are also important habitat for snowshoe hares (Litvaitis et al. 1985).

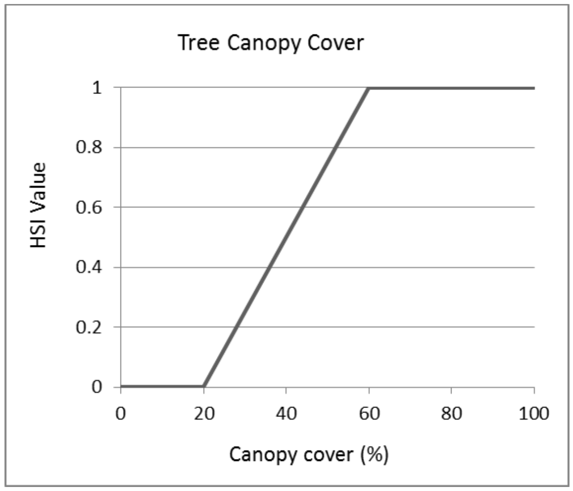

Figure A-2. Relationship between tree canopy cover and HSI values for fisher. The equation between 20% and 60% is $y=0.025x-0.5$.

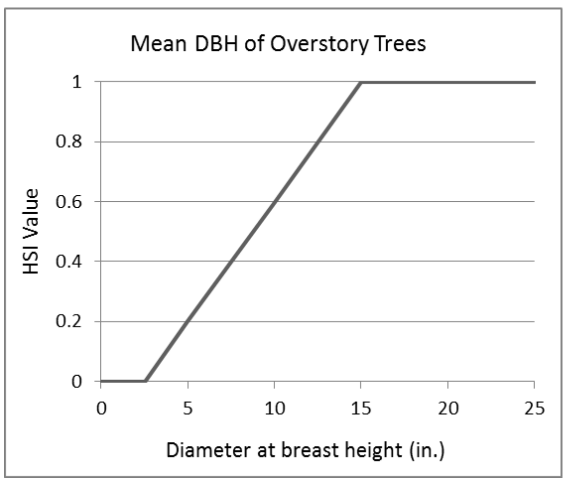

Figure A-3. Relationship between mean diameter at breast height of overstory trees and HSI values for fisher. The equation between 2.5 and 15 in. is $y=0.08x-0.2$.

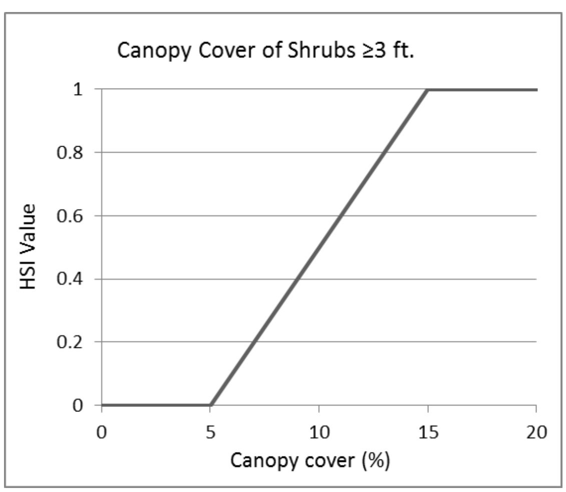

Figure A-4. Relationship between canopy cover of shrubs and HSI values for fisher. The equation between 5% and 15% is $y=0.1x-0.5$.

Snowshoe hares have been shown to be the primary food source for fisher in Idaho and Montana (Jones 1991). During the winter months in western Montana snowshoe hares are most abundant in early successional treatment sites and late successional heterogeneous treatment sites with high levels of horizontal cover (Carreker 1985, Koehler and Aubry 1994, Thomas et al. 1997, Griffin and Mills 2007, 2009). Hare use reaches the highest levels when horizontal cover of above snow vegetation is ≥50% (Carreker 1985). Dense cover provides the hares with critical food, cover, and thermal protection (Litvaitis et al. 1985, Hodges 2000). When horizontal cover drops below 10% the habitat is considered unsuitable (Thomas et al. 1997). Horizontal cover has not been included as a variable in the fisher model at this time, but it may be desirable to add this variable to the model to better assess the potential abundance of snowshoe hare in an area.

In the intermountain west riparian areas have been shown to be preferred by fisher due to decreased snow depths, prevalence of spruce, moderating temperatures, or topographic features (Jones 1991, Powell and Zielinski 1994, Olson et al. 2014). However, this relationship may not be as important in areas supporting mixed conifer forests including areas with spruce and fir outside of riparian zones, so a riparian variable is not included in the HSI model. An additional variable could be added if additional research on this relationship applicable to the northern Rockies shows a definite selection for riparian areas in this area. Similarly, steep slopes have been noted by some researchers to be selected for fisher resting sites. However, as with selection for riparian areas, this selection may be landscape specific, so was not included in this habitat model.

A fourth variable is the absolute canopy cover of spruce and true fir (Figure A-5). Ecosites that contain spruce and true fir have been shown to be preferred by fisher for foraging and resting (Powell and Zielinski 1994, Proulx 2006). These types of treatment sites have also been shown to provide good habitat for snowshoe hares (Griffin 2004).

In addition, based on the findings of Sauder and Rachlow (2015) and Weir and Corbould (2010) a variable was added to be applied at the home range scale that addresses the amount of openings within the area. This relationship (Figure A-6) was quantified by Sauder and Rachlow (2015), and would be a modification of the overall habitat quality value of the aggregate of treatment sites within the home range area.

The HSI grid for fisher was calculated with the following formula:

$$\textit{Fisher HSI} = (\textit{Tree Canopy HSI} \times \textit{DBH HSI} \times (\mathrm{Min}\ (1, [0.2 + 0.55 \times \textit{Shrub HSI} + 0.85 \times \textit{Spruce/Fir HSI}])^{0.333}$$

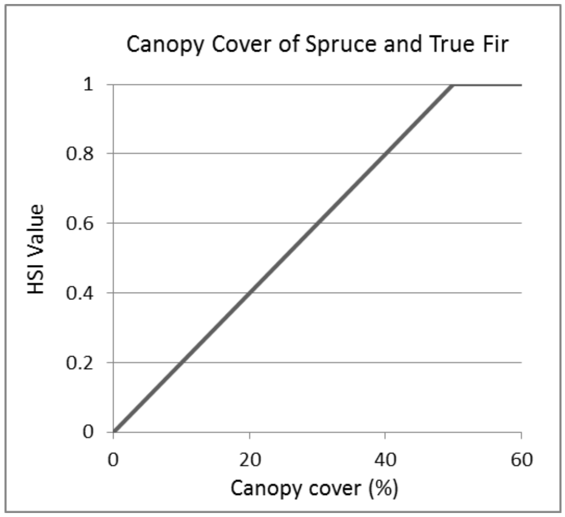

Figure A-5. Relationship between the absolute canopy cover of spruce and true fir and HSI values for fisher. The equation between 0% and 50% is $y=0.02x$.

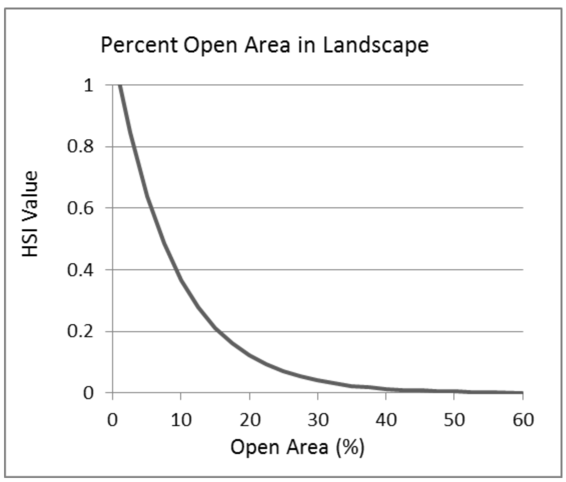

Figure A-6. Relationship between the percentage of open area in the landscape and HSI values for fisher. The equation between 0% and 50% is $y = 1.1146e^{-0.111x}$.

HSI values were determined for each vegetation category in the Southwest Crown of the Continent project area. Values for specific treatment site types were determined from FIA plot data applicable to the project area. For vegetation classes missing treatment site data values for each variable were estimated from the most similar vegetation conditions that had empirical data. HSI scores were then aggregated and contoured using a moving window analysis to produce the final input layer needed for HOMEGROWER. The size of the moving window is equal to the allometric home range (Roloff and Haufler 1997). The allometric home range for a 2.25 kg female fisher is 246 ha (Lindstedt et al. 1986).

Three iterations were done in HOMEGROWER. The target home range area was 5 times the allometric home range or 1233 ha. The number of seeds was 200,000 and the growth window was 40 cells.

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Flammulated Owl (*Otus flammeolus*)

Habitat Description

Flammulated owls are a small owl found throughout the lower elevation valleys of western Montana (Figure A-7), but typically are limited to dry, conifer dominated treatment sites (Groves et al. 1997). These low elevation treatment sites are dominated by mature (age 50 to 100 years) to old (age > 120 years) ponderosa pine and Douglas-fir with multiple canopies, low stocking rates, open canopies, and moderate shrub cover (McCallum 1994, Groves et al. 1997). Flammulated owls have also been documented nesting successfully in treatment sites dominated by Douglas-fir or mixed with trembling aspen and lacking ponderosa pine (Howie and Ritcey 1987, Powers et al. 1996). The mature trees are important for nesting while the younger trees and shrubs in the understory provide roosting areas and the openings facilitate foraging (Goggans 1986, Reynolds and Linkhart 1987). For example, tree densities in treatment sites where males responded to callback surveys (typically roosting areas) averaged 202 trees per acre with a mean diameter at breast height from 11.1-15 inches (Groves et al. 1997). Due to their preference for dry conditions and intolerance of high humidity, riparian areas are considered non-habitat (McCallum 1994).

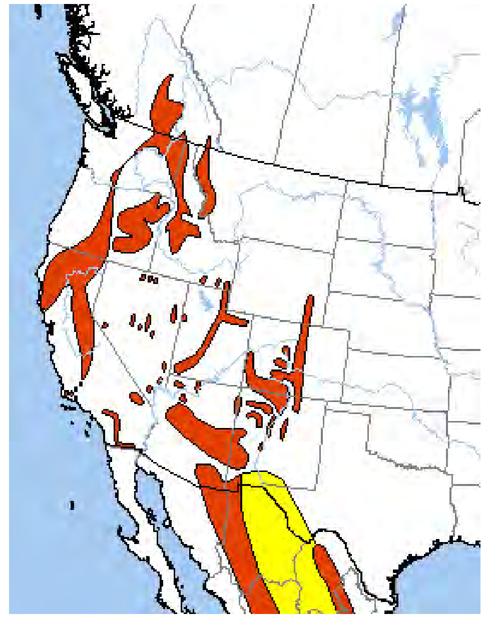

Figure A-7. Current range of the flammulated owl; red represents breeding resident (Ridgely et al. 2005).

HSI Model

The flammulated owl model is based on optimum conditions for nesting, roosting, and foraging, however nesting habitat is considered the primary determinant of flammulated owl habitat. Flammulated owls prefer xeric, open, mature-old ponderosa pine and Douglas-fir with scattered clumps of dense younger trees (roosting areas) and a component of large snags (Christie and van Woudenberg 1997, Linkhart 2001). The habitat variables selected for this model characterize treatment sites based on these optimum conditions.

The first habitat variable is ponderosa pine, western larch, and Douglas fir snags >20 in. DBH per acre (Figure A-8). As secondary cavity nesters it is critically important that flammulated owls have access to suitable nesting trees (McCallum 1994). Bull et al. (1990) found 88% of nest trees in Oregon (n=33) to be >20 in. DBH and 97% of nest trees to be either ponderosa pine or western larch. Occupied nest trees in south-central Idaho were found in either Douglas fir or aspen with a mean diameter of 19.6 in. (Powers et al. 1996). Douglas fir or ponderosa pine was the preferred nesting trees in both south-central British Columbia (Christie and van Woudenberg 1997) and Colorado (Linkhart et al. 1998). The purpose of this variable is to insure enough snags are present in a treatment site to keep the lack of nest sites from being a limiting factor.

The second variable is total canopy cover of the tree overstory (Figure A-9). Flammulated owls prefer open to semi-open treatment sites for both nesting and foraging (McCallum 1994). Treatment sites

Figure A-8. Relationship between ponderosa pine, Douglas fir, and western larch snags per acre > 20 in. DBH and HSI values for flammulated owl. The equation between 0 and 1.5 is $y=0.6667x$.

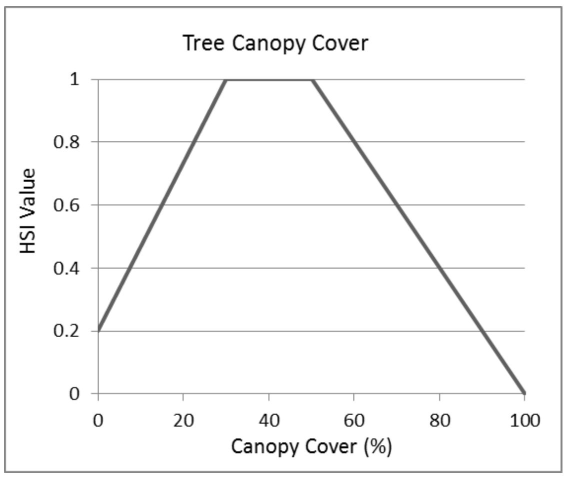

Figure A-9. Relationship between overstory tree canopy cover and HSI values for flammulated owl. The equation between 0 and 30 is $y=0.0267x+0.2$ and the equation between 50 and 100 is $y=-0.02x+2$.

surrounding nest sites in Oregon had a mean canopy cover of 55% (n=33) (Bull et al. 1990). In British Columbia canopy cover surrounding nest sites ranged from 30-50% (n=35) (Christie and van Woudenberg 1997). In the Blue Mountains of Oregon, treatment sites used by nesting owls all had canopy cover <50% (n=20) (Goggans 1986). Callback surveys in Idaho found male owls occupying treatment sites with 52-64% canopy cover (Groves et al. 1997), which was likely to represent roosting habitat. Samson (2006) considered treatment sites capable of supporting flammulated owls when the canopy cover was between 35-85%. Based on these studies, the canopy cover variable gives treatment sites an optimum suitability rating when cover is between 30% and 50%.

The third variable used in the flammulated owl model was percent of maximum treatment site density index ($SDI_{\%max}$) (Figure A-10). The $SDI_{\%max}$ is a variable that provides more detail about treatment site conditions than trees per acre or basal area (Woodall and Miles 2006). SDI is a function of treatment site density based on the average specific gravity of trees in the treatment site. Each treatment site has a maximum density. The percent of maximum of the treatment site's current condition provides an accurate measure of treatment site characteristics and treatment site potential (Long and Daniel

1990). This variable allows the model to assign higher suitability to treatment sites that are characterized by both large trees and open canopies. It also avoids assigning high suitability to treatment sites that meet one criteria, such as basal area, while not meeting another, such as trees per acre. A $SDI_{\%max}$ of 25 indicates the onset of inter-tree competition, a $SDI_{\%max}$ of 35 indicates the lower limit of full site occupancy, and a $SDI_{\%max}$ of 60 indicates the lower limit of self-thinning (Long 1985). For flammulated owl, optimum nesting and foraging conditions are found between 10 and 20 percent of $SDI_{\%max}$.

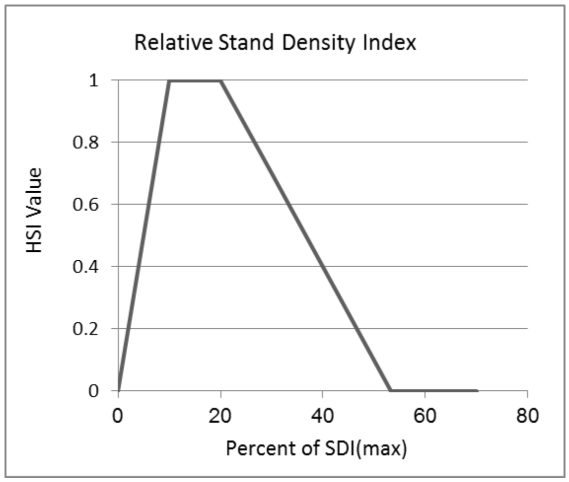

Figure A-10. Relationship between relative treatment site density index (for trees >6 in DBH) and HSI values for flammulated owl. The equation between 0 and 10 is $y=0.1x$ and the equation between 20 and 53.333 is $y=-0.03x+1.6$.

The final habitat variable is ecological site (Figure A-11). This variable identifies ecological sites and disturbance regimes that are used by flammulated owl. They are consistently found in low elevation treatment sites dominated by ponderosa pine and Douglas fir with open canopies and large trees (McCallum 1994, Christie and van Woudenberg 1997, Linkhart 2001). Sites can be further characterized by the lack of moist site indicator species such as *Salix* and *Vaccinium* (Wright et al. 1997). The habitat type HSI was based on the relative moisture of a site as indicated by the presence of understory species such as *Salix* and *Vaccinium*. In treatment sites where these species are present, the value for this variable is always zero. The final HSI grid was calculated by multiplying the geometric mean of the snag HSI, canopy cover HSI, and SDI HSI by the habitat type HSI.

Samson (2006) developed a regional wildlife habitat relationship model for flammulated owls designed to assign habitat values to mapped classes of vegetation. The model used dominance group, canopy cover, aspect, structure class, snag density, and a relationship between basal area and tree diameter as variables. The dominance group and aspect variables are captured by the ecological site variable used above. Snag density and canopy cover are used in both models. The SDI variable used above provides a similar measure of treatment site density and structural diversity as the basal area/tree diameter variable used in the Samson model.

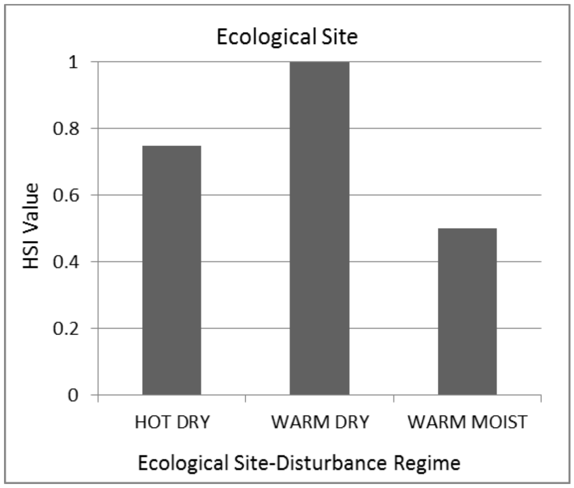

Figure A-11. Relationship between ecological site and HSI values for flammulated owl. Ecological sites (habitat type groupings) not listed received a rating of zero.

HSI values were determined for each vegetation category in the Southwest Crown of the Continent project area. Values for specific treatment site types were determined from FIA plot data applicable to the project area. For vegetation classes missing treatment site data values for each variable were estimated from the most similar vegetation conditions that had empirical data. HSI scores were then aggregated and contoured using a moving window analysis to produce the final input layer needed for HOMEGROWER. The size of the moving window is equal to the allometric home range (Roloff and Haufler 1997). The allometric home range for a 54 g female flammulated owl is 3.4 ha (Van Horne and Wiens 1991).

Three iterations were done in HOMEGROWER. The target home range area was 5 times the allometric home range or 17 ha. The number of seeds was 500,000 and the growth window was 5 cells.

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Northern Goshawk (*Accipiter gentilis*)

Habitat Description

Northern goshawks are a large accipiter found in forested areas throughout the Rocky Mountains (Figure A-12) (Squires and Reynolds 1997). Northern goshawks have long been considered sympatric with mid-aged to old (140 years to >240 years) conifer treatment sites and the bulk of available literature supports this (Reynolds et al. 1992, Daw and DeStefano 2001, Finn et al. 2002, Desimone and DeStefano 2005, Greenwald et al. 2005). The availability of small openings within mature treatment sites has been suggested as important for both prey densities and foraging efficiency (Reynolds et al. 1992, Daw and DeStefano 2001), (Reynolds et al. 2008). Nest sites in particular require mature treatment sites with high canopy cover, large trees, and multiple canopies (Crocker-Bedford and Chaney 1988, Hayward and Escano 1989, Squires and Reynolds 1997, Daw and DeStefano 2001, Greenwald et al. 2005) (Squires and Kennedy 2006, Reynolds et al. 2008). The size of the nest area varies considerably by region, but an area of 30 acres has been proposed

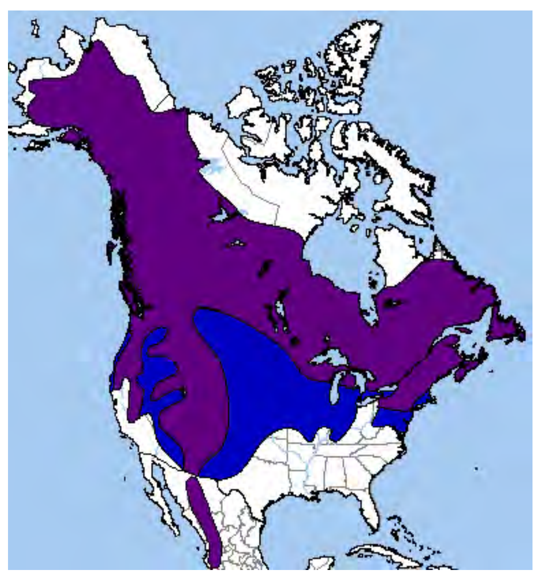

Figure A-12. Current range of the Northern Goshawk in North America; purple indicates permanent resident and blue indicates non-breeding range (Ridgely et al. 2005).

as an acceptable average (Reynolds et al. 1992). Daw and DeStefano (Daw and DeStefano 2001) found similar conditions for nesting in 30-60 ac areas surrounding nests they studied in Oregon. In northern Idaho, the mean nest height was 41 feet, in trees with a mean height of 85.3 feet and a mean diameter at breast height of 19.7 inches (Hayward and Escano 1989). Also, the canopy cover around the nest was higher than the mean cover for the treatment site.

Ideal conditions for foraging are treatment sites with a closed canopy, but an open understory that provides clear flight corridors (Reynolds et al. 1982, Hayward and Escano 1989). Goshawks in Oregon and Washington have been found to avoid open areas, such as meadows, shrublands, and logged early seral treatment sites (<30 years in age) (Austin 1993, Bloxton 2002). Avoidance of mature treatment sites with <40% canopy cover has also been documented (Austin 1993, Bright-Smith and Mannan 1994, Beier and Drennan 1997).

HSI Model

Separate nesting and foraging models were developed for goshawks. They were based on the framework described by Shaffer et al. (1999). Goshawk prefer mature treatment sites with complex canopies, high canopy cover, a mix of deciduous and conifer species, and minimal human disturbance.

The first variable used in the nesting model is mean overstory tree height (Figure A-13). The purpose of this variable is to help predict the availability of large trees in the treatment site that are suitable for nesting. It also provides a measure of treatment site maturity. Goshawks in western Montana and Idaho nested in trees ranging from 39.4-157.5 feet (n=17) in height (Hayward and Escano 1989). Further work in the interior Pacific Northwest found a similar range of heights for nest trees (40.4-157.5 ft; n=82) (McGrath et al. 2003). A study in the Yellowstone region of Wyoming measured the heights of 49 nest trees and found a range from 39.4-124.7 feet with a mean height of 82 feet (Patla 1997).

The second nesting variable is overstory tree canopy cover (Figure A-14). Goshawks have been found to nest in treatment sites with closed canopies (Crocker-Bedford and Chaney 1988, Hayward and Escano 1989, Squires and Reynolds 1997, Greenwald et al. 2005). Dense canopies provide protection both from predation (Reynolds et al. 1992) and weather extremes during the early portion of the nesting season (Moore and Henry 1983). Hayward and Escano (Hayward and Escano 1989) looked at 17 nest sites in Montana and Idaho that had mean canopy cover of 80% and a range from 65-90%. At 82 nest sites in Oregon and Idaho the mean canopy cover was 53.1% (McGrath (McGrath et al. 2003). In south-central Wyoming on higher elevation sites characterized by subalpine pine, Engelmann spruce, and lodgepole pine the mean cover at 39 nest sites was 66.7% (Squires and Ruggiero 1996). Also in Wyoming, Patla (1997) measured canopy cover at 44 nest sites and found average canopy cover to be 73%.

The third variable in the nesting model is basal area (Figure A-15). In eastern Oregon and Washington basal area was found to be a strong factor in the selection of nest sites, and was found to be more predictive of nest locations than treatment site structure (McGrath et al. 2003). Samson (2006) created a regional goshawk nesting model that used basal area as one variable. This study identified a range of basal areas from 104.5-257 ft^2/ac. A subsequent study (unpublished) on the Helena, Lewis and Clark, and Custer National Forests found that goshawks nest in treatment sites with both higher and lower amounts of basal area. For this model the range of 104.5-257 ft^2/ac will be considered optimal habitat while recognizing that values on either side of this range can still support successful nest sites.

The final nesting HSI grid was calculated by using the geometric mean of the three preceding habitat variables. The second component of the goshawk model is the foraging HSI grid. The variables used for the foraging grid are discussed below.

The first variable is overstory tree canopy cover (Figure A-16). Northern goshawk physiology requires them to have somewhat open forest conditions to forage effectively (Reynolds et al. 1992). As the bulk of most goshawk diets in North America consist of mammals (86-94%) an open understory promotes foraging efficiency by making ground based prey vulnerable to goshawk predation (Shaffer et al. 1999).

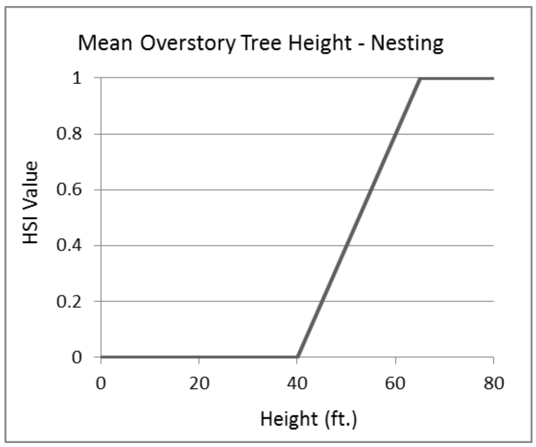

Figure A-13. Relationship between mean overstory tree height and HSI values for northern goshawk nesting. The equation between 40 and 65 ft. is *$y=0.04x-1.6$*.

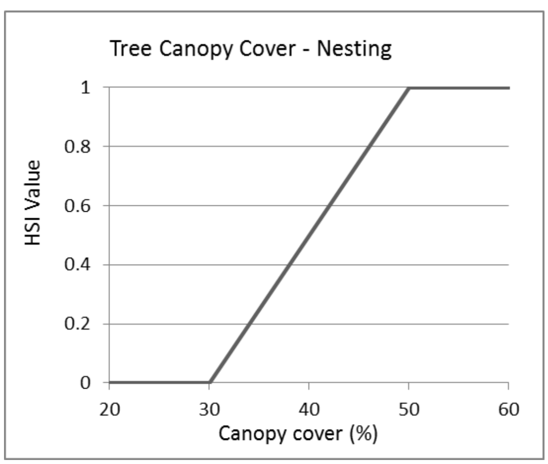

Figure A-14. Relationship between overstory tree canopy cover and HSI values for northern goshawk nesting. The equation between 30 and 50 is *$y=0.05x-1.5$*.

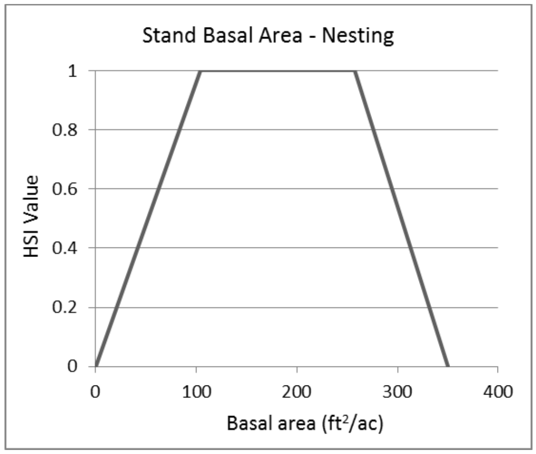

Figure A-15. Relationship between basal area and HSI values for northern goshawk nesting. The equation between 0 and 104.5 is $y=0.0096x$ and the equation between 257 and 350 is $y=-0.0108x+3.7636$.

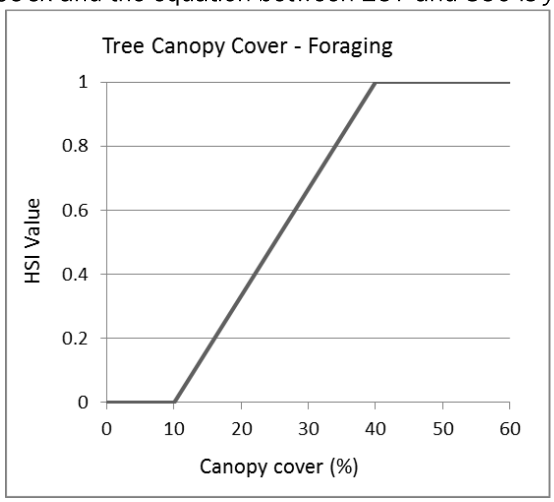

Figure A-16. Relationship between tree canopy cover and HSI values for northern goshawk foraging. The equation between 10 and 40 is $y=0.0333x-0.333$.

The second variable in the foraging model is mean overstory tree height (Figure A-17). The variable is also used to target older, more mature treatment sites that will have optimal habitat for goshawk foraging. The final foraging HSI grid was calculated by taking the geometric mean of these two variables.

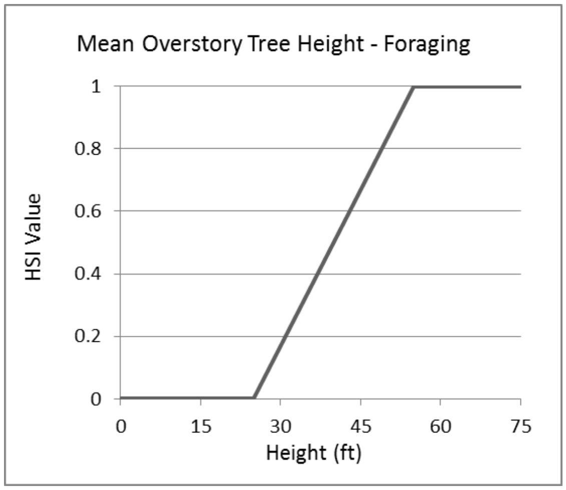

Figure A-17. Relationship between mean overstory tree height and HSI values for northern goshawk foraging. The equation between 25 and 55 ft. is *y=0.0333x-0.8333*

HSI values were determined for each vegetation category in the Southwest Crown of the Continent project area. Values for specific treatment site types were determined from FIA plot data applicable to the project area. For vegetation classes missing treatment site data values for each variable were estimated from the most similar vegetation conditions that had empirical data. HSI scores were then aggregated and contoured using a moving window analysis to produce the final input layer needed for HOMEGROWER. The size of the moving window is equal to the allometric home range (Roloff and Haufler 1997). The allometric home range for a 0.713 kg female goshawk is 67 ha (Van Horne and Wiens 1991).

Three iterations were done in HOMEGROWER. The target home range area was 5 times the allometric home range or 334 ha. The number of seeds was 800,000 and the growth window was 10 cells. For the nesting grid the target home range was 10 ha.

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Pileated Woodpecker (*Dryocopus pileatus*)

Habitat Description

Pileated woodpeckers are a primary cavity nester found throughout the United States (Figure A-18) in forested vegetation types (Bull and Jackson 1995). They generally occur in mature forests with partially closed to closed canopies and large diameter trees (Bull 1987, Aney and McClelland 1990). For nesting, tree size seems to be the most important variable with a variety of tree species used (Bull 1987, Aney and McClelland 1990, McClelland and McClelland 1999, Bonar 2001, Aubrey and Raley 2002).

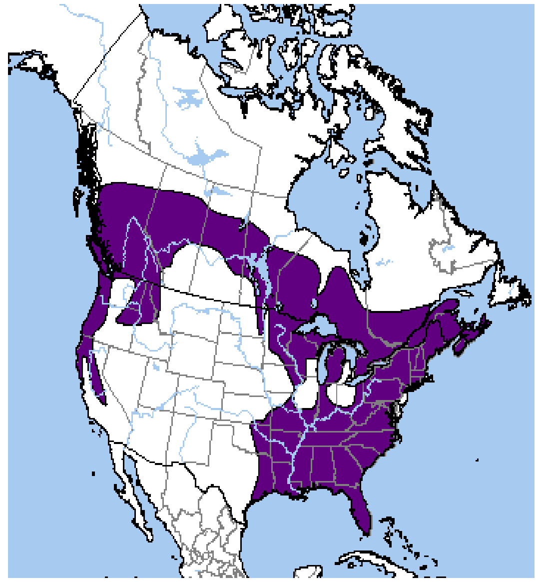

Figure A-18. Current range of the pileated woodpecker in North America (Ridgely et al. 2005).

Pileated woodpeckers primarily feed on ants (*Camponatus* and *Formica* spp.) (Beckwith and Bull 1985, Bull et al. 1992a, Bonar 2001). Thus, habitat that provides high suitability for ants should be suitable for woodpecker foraging, especially if it also provides overhead cover for protection from aerial predators. Avian predators are one of the leading causes of mortality for adult pileated woodpeckers (Bull et al. 1992b). Ideal ant habitat, and thus foraging habitat, consists of a mix of snags, stumps, and downed logs (Aney and McClelland 1990, Torgerson and Bull 1995).

The other important habitat characteristic for pileated woodpeckers is roost trees (Bull et al. 1992b, Aubrey and Raley 2002). Roost trees provide year round protection for mature birds and are important both for thermoregulation in the winter and protection from predation (Bull et al. 1992b). Roost trees differ from nest trees in that they can be completely hollow and have multiple entrances; however sizes are similar to nest trees (Bull et al. 1992b).

HSI Model

Roloff (2004) updated the pileated model developed by Aney and McClelland (1990) in order to account for new research and better integrate the requirements for roosting trees into a habitat model. The model presented here follows the framework of Roloff (2004). The first variable for the nesting component of the model is overstory tree canopy cover of preferred nesting species (Figure A-19). For the purpose of this model overstory trees are defined as trees ≥ 65 feet tall. Pileated woodpeckers require large trees for nesting and these are generally found in treatment sites with low to moderate canopy closure (Bull 1987, Aney and McClelland 1990, McClelland and McClelland 1999). Moderate amounts of canopy closure provide better protection from avian predators (Bull et al. 1992b). Preferred tree species for nesting are western larch and ponderosa pine, likely due to the fact they quickly lose their bark and lower branches (Bull 1987). Other tree species used for nesting include cottonwood, aspen, Douglas fir, western white pine, and grand fir (McClelland and McClelland 1999, Bonar 2001, Aubrey and Raley 2002).

The second and third nesting variables are the densities of small snags (Figure A-20) and large snags (Figure A-21). Pileated woodpeckers nest in snags and decadent trees with a range of diameters and seem to prefer a mix of available size classes (McClelland and McClelland 1999, Bonar 2001, Aubrey and Raley 2002). Having two size class variables insures there is a good diversity of size classes present in the landscape.

The fourth variable for the nesting portion of the model is the average size of suitable nesting trees (Figure A-22). This variable supports the snag density variable by insuring that the majority of dead and decadent trees are suitably sized for nesting. Pileated woodpecker nest tree selection has been positively correlated to increasing tree diameter (Bull 1987). A minimum size of 15 in has been used in other models (Samson 2006).

The nesting HSI value is calculated with the following formula:

Nesting HSI = (((Min 1,(Snag Density$_{small}$ *HSI + Snag Density*$_{large}$ *HSI) + Snag DBH HSI)/2) * Canopy Cover HSI)^0.5*

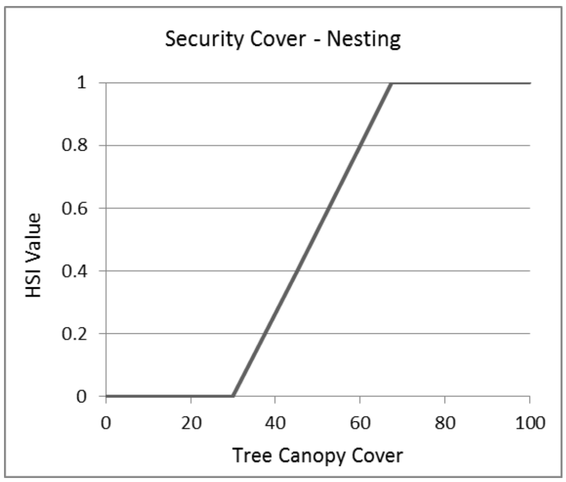

Figure A-19. HSI values for pileated woodpecker nesting based on overstory canopy cover of preferred tree species. The equation between 30 and 67 is *y=0.0267x-0.8*.

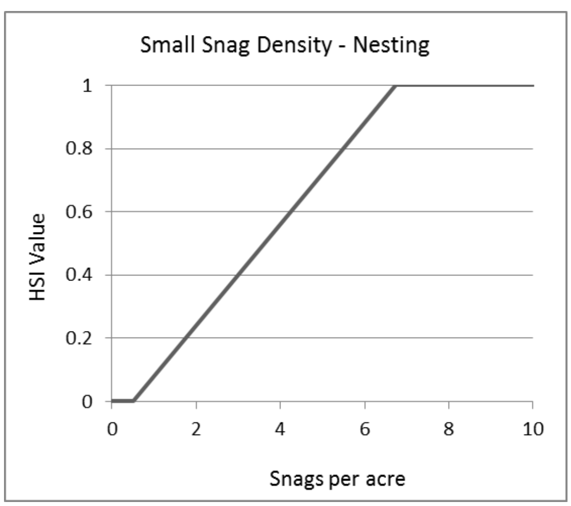

Figure A-20.HSI values for nesting habitat based on density of dead and defective larch, grand fir, ponderosa pine, and quaking aspen >15 in. DBH and >60 ft. tall. The equation between 0.5 and 6.75 is *y=0.16x-0.08*.

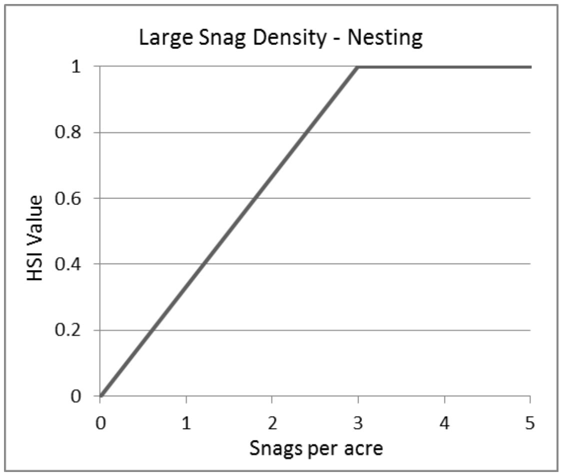

Figure A-21. HSI values for pileated woodpecker nesting based on density of dead and defective larch, grand fir, ponderosa pine, and quaking aspen >30 in. DBH and >60 ft. tall. The equation between 0 and 3 is *y=0.333x*.

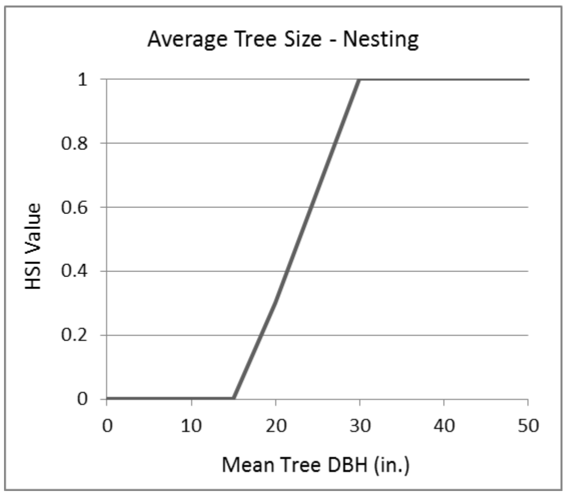

Figure A-22. HSI values for pileated woodpecker nesting based on average DBH (cm) of live and dead western larch, grand fir, ponderosa pine, and quaking aspen >20 in. DBH and >60 ft. tall. The equation between 15 and 30 in. is *y=0.07x-1.1*.

The second component of the pileated woodpecker model is foraging habitat. Ants have been shown to be the primary food source for pileated woodpeckers during the breeding season (Beckwith and Bull 1985, Bull et al. 1992a, Bonar 2001) thus ideal foraging habitat provides optimal conditions for ants while also providing some overhead cover to protect woodpeckers from aerial predation (Bull et al. 1992b). The foraging component is composed of three habitat variables. The first variable is tree canopy cover (Figure A-23). Moderate amounts of canopy cover provide cover from predation while allowing open flight lines to facilitate foraging. This variable also helps insure the site being rated has forest cover.

The second variable in the foraging model is the density of preferred foraging sites (Figure A-24). As the amount of treatment siteing snags and downed debris increases so does the population of ants (Torgerson and Bull 1995). Downed wood has been found to be as important for foraging as treatment siteing dead wood (Bull 1987, Aney and McClelland 1990).

The final foraging variable is average tree size (Figure A-25). Pileated woodpeckers have shown a preference for foraging or large treatment siteing trees, with preference increasing with tree diameter (Raley and Aubrey 2006). Woodpeckers in Alberta also selected large trees for foraging (Bonar 2001).

The final foraging HSI score is calculated by taking the geometric mean of the three foraging habitat variables. The final pileated HSI is calculated with the following formula:

*Final HSI = (((2 * Nest_HSI) * Forage_HSI)^0.33*

Samson (2006) developed a regional wildlife habitat relationship model for pileated woodpecker nesting and winter foraging. The model used dominance group, tree size (for nesting), and snag, log, and stump size (for winter foraging) as variables. The variables used in the habitat suitability model presented here are finer scale than those described for a Samson model, which was designed as a regional wildlife habitat relationship model.

HSI values were determined for each vegetation category in the Southwest Crown of the Continent project area. Values for specific treatment site types were determined from FIA plot data applicable to the project area. For vegetation classes missing treatment site data values for each variable were estimated from the most similar vegetation conditions that had empirical data. HSI scores were then aggregated and contoured using a moving window analysis to produce the final input layer needed for HOMEGROWER. The size of the moving window is equal to the allometric home range (Roloff and Haufler 1997). The allometric home range for a 0.30 kg female pileated is 24.7 ha (Van Horne and Wiens 1991).

Three iterations were done in HOMEGROWER. The target home range area was 5 times the allometric home range or 123.5 ha. The number of seeds was 600,000 and the growth window was 10 cells.

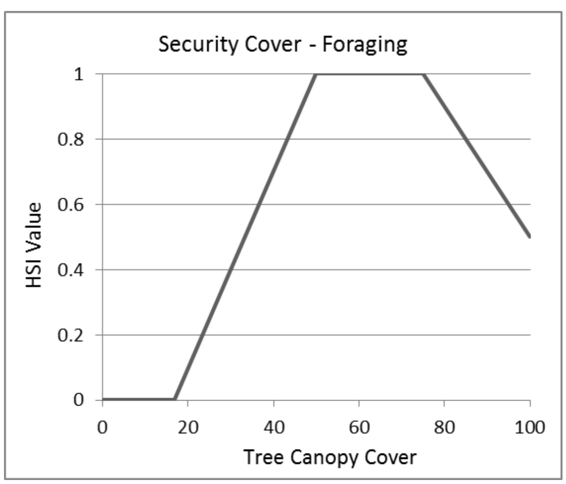

Figure A-23. HSI values for pileated woodpecker foraging based on overstory canopy cover. The equation between 16.67 and 50 is *y=0.03x-0.5* and the equation between 80 and 100 is *y=-0.02x+2.5*.

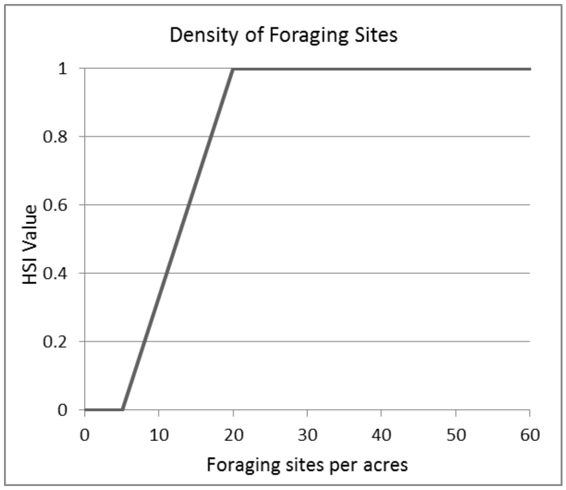

Figure A-24. HSI values for pileated woodpecker foraging based on density of dead trees >10 in. DBH plus logs >10 in. diameter and >6 ft. long. The equation between 5 and 20 is *y=0.0667x-0.3333*.

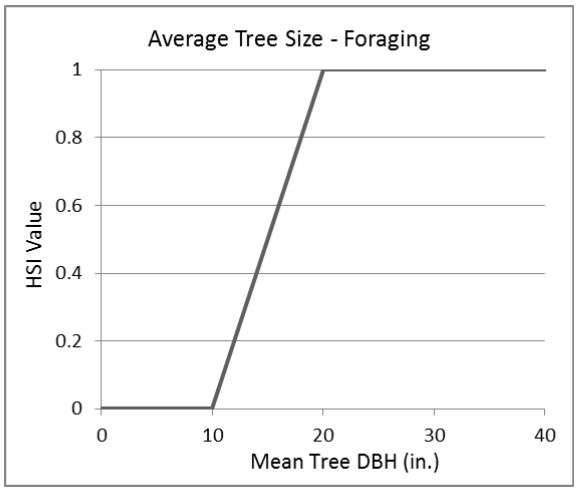

Figure A-25. HSI values for pileated woodpecker foraging based on the average DBH (in.) of overstory trees. The equation between 10 and 20 in. is *y=0.00394x-0.2333*.

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American Marten (*Martes americana*)

Habitat Description

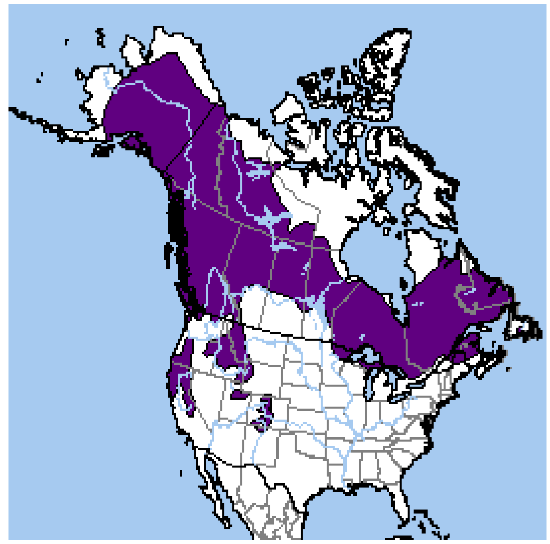

Figure A-26. Current range of the American marten in North America (Patterson et al. 2005).

The American marten is a medium-sized forest carnivore found throughout the coniferous forest region of northern and western North America (Hall 1981) (Figure A-26). Marten are strongly associated with conifers and intolerant of areas lacking overhead cover (Buskirk and Ruggerio 1994). Marten are generally considered to rely on late-successional, mesic forests with large amounts of treatment siteing and downed woody material that create complex cover near ground level (Buskirk and Powell 1994). Physical structure near ground level appears to be the most important habitat characteristic as it provides protection from predators, access to subnivean spaces for winter foraging, and thermal cover (Buskirk and Powell 1994). In locations with sufficient overhead cover marten can be found in young treatment sites (<40 years old) and deciduous dominated treatment sites (Poole et al. 2004). Primary prey species for marten include red-backed voles (*Clethrionomys* spp.), pine squirrels (*Tamiasciurus* spp.), and meadow voles (*Microtus* spp.) (Buskirk and Ruggerio 1994). Deer mice (*Peromyscus* spp.) are taken at a lower proportion than their availability and winter habitat with high numbers of deer mice generally have low habitat quality for martens (Buskirk and Ruggerio 1994).

HSI Model

The American marten model is based on the framework described by Allen (1982) with changes based on more recent literature. The first habitat variable in the model is tree canopy cover (Figure A-27). Marten in the Intermountain West have been shown to preferentially select areas with high overhead canopy cover and avoid areas with less than 30% canopy cover (Koehler and Hornocker 1977). The odds of detecting marten in southern British Columbia increased with canopy cover (Mowat 2006).

The second variable is relative percent cover of fir and spruce in the overstory tree canopy (Figure A-28). Marten most commonly associated with mesic sites characterized by spruce and true fir cover types (Buskirk and Powell 1994, Fecske et al. 2002, Baldwin and Bender 2008). The majority of resting sites used by marten in Oregon were in true fir or spruce (Bull and Heater 2000). Red-backed voles, a primary prey species, are also most common in mesic, spruce/fir dominated treatment sites (Raphael 1989). Ruggiero et al. (1998) found high densities of Engelmann spruce (*Picea engelmanni*) and subalpine fir (*Abies lasiocarpa*) to be significant for the selection of marten natal den sites.

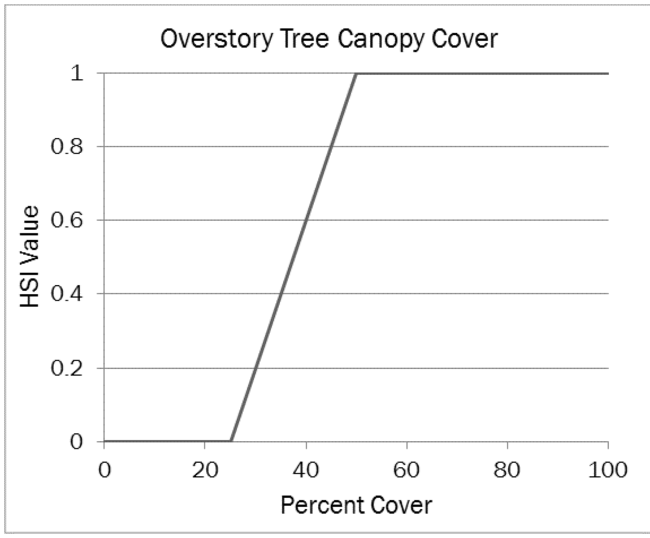

Figure A-27. HSI values for American marten based on the percent canopy cover of overstory trees. The equation between 25 and 50 is $y=0.04x-1$.

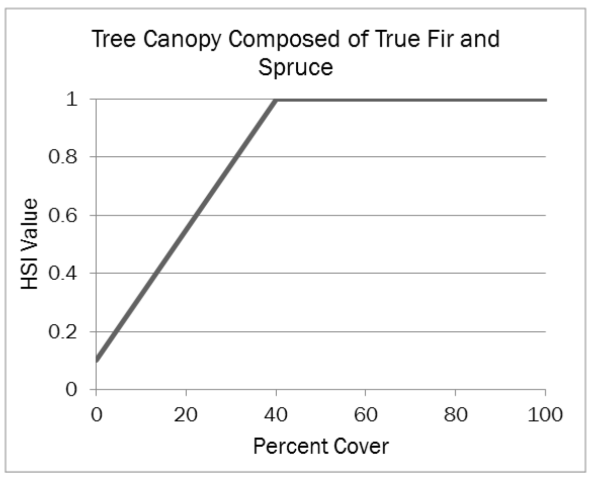

Figure A-28. HSI values for American marten based on the relative percent cover of true fir and spruce that comprise the overstory tree canopy. The equation between 0 and 40 is $y=0.0225x+0.1$.

The third habitat variable is the percent of ground covered by coarse woody debris (CWD) that has a diameter >6 inches (Figure A-29). CWD is particularly important in the winter because it creates access points to the subnivean spaces which are important for both foraging and resting sites (Buskirk and Powell 1994). Marten in Ontario have been shown to have higher foraging success rates on red-backed voles in uncut forests with high amounts of CWD compared to regenerating treatment sites with lower amounts of CWD, but similar population levels of voles (Andruskiw et al. 2008). Sherburne and Bissonette (1994) found both prey densities and marten occurrence in Yellowstone National Park increased with increasing percent cover of CWD. CWD is also important for providing natal and maternal den sites (Ruggiero et al. 1998, Bull and Heater 2000).

The fourth model variable is the average diameter of coarse woody debris (CWD) (Figure A-30). Larger diameter logs have been linked to increased foraging success for marten in Maine (Payer and Harrison

2003). Larger diameter logs also provide greater security for denning females (Patton and Escano 1990, Ruggiero et al. 1998).

The final variable is ecological site and disturbance regime (Figure A-31). The ecological site provides a measure of moisture at the site (Patton and Escano 1990) which is important because marten are associated with mesic sites and avoid xeric sites (Buskirk and Powell 1994). Red-backed voles were most common in mesic, spruce/fir dominated treatment sites in Wyoming (Raphael 1989).

The final HSI for American marten is calculated with the following formula:

*Final HSI = ((Canopy Cover HSI * Spruce/Fir HSI * CWD Cover HSI * CWD Diameter) ^ 1/2) * Ecosite HSI*

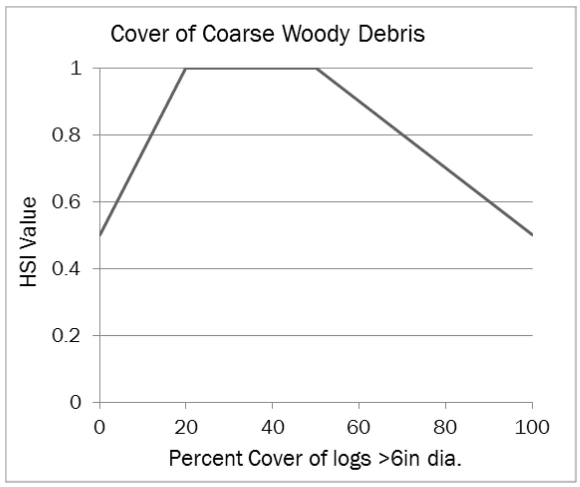

Figure A-29. HSI values for American marten based on the percent of ground covered by coarse woody debris >6 in. diameter. The equation between 0 and 20 is *y=0.025x+0.5*. The equation between 50 and 100 is *y=0.01x+1.5*.

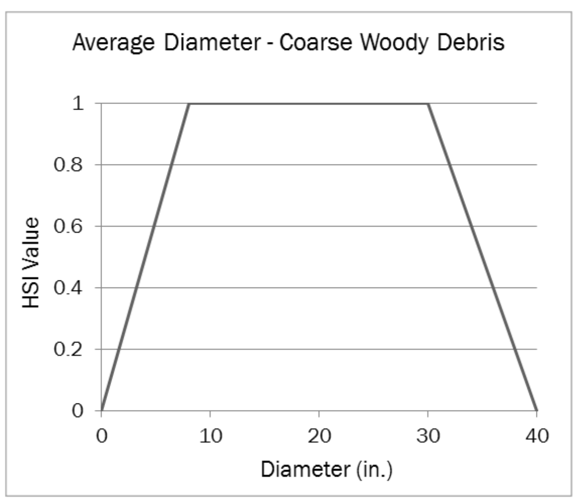

Figure A-30. HSI values for American marten based on the average diameter (in.) of coarse woody debris. The equation between 0 and 8 is *y=0.125x* and the equation between 30 and 40 is *y=-0.0833x+2.5*.

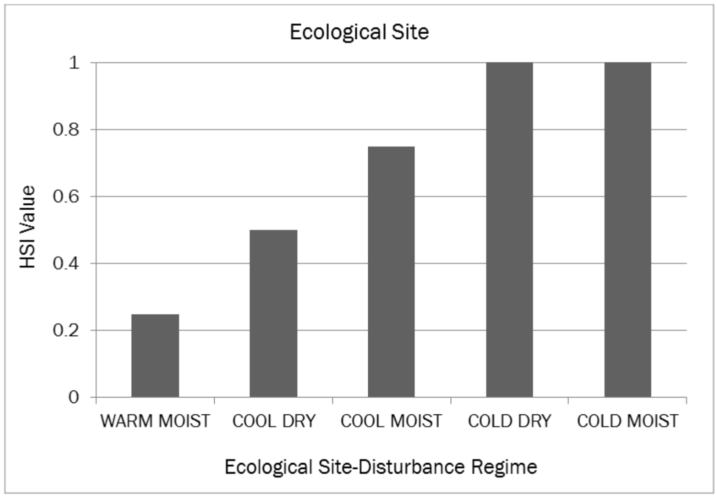

Figure A-31. HSI values for American marten based on the ecological site of the treatment site. Ecological sites not listed in the graph received an HSI score of zero.

HSI values were determined for each vegetation category in the Southwest Crown of the Continent project area. Values for specific treatment site types were determined from FIA plot data applicable to the project area. For vegetation classes missing treatment site data values for each variable were estimated from the most similar vegetation conditions that had empirical data. HSI scores were then aggregated and contoured using a moving window analysis to produce the final input layer needed for HOMEGROWER. The size of the moving window is equal to the allometric home range (Roloff and Haufler 1997). The allometric home range for a 0.80 kg female marten is 93 ha (Lindstedt et al. 1986).

Three iterations were done in HOMEGROWER. The target home range area was 5 times the allometric home range or 465 ha. The number of seeds was 500,000 and the growth window was 15 cells.

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Hairy Woodpecker (*Picoides villosus*)

Habitat Description

The hairy woodpecker is a year-round resident throughout most of North America (Figure A-32) and is a primary cavity nester in both coniferous and deciduous forests (AOU 1983). As a cavity nester hairy woodpeckers utilize both living and dead trees for potential nest sites (Thomas et al. 1979, Mannan et al. 1980, Zarnowitz and Manuwal 1985). Both hardwood and softwood snags provide important foraging habitat for breeding and overwintering woodpeckers (Mannan et al. 1980, Weikel and Hayes 1999, Ripper et al. 2007). Hairy woodpeckers have been associated with old forests (>200 years old) (Mannan 1980) and both nesting and foraging use is higher in treatment sites with larger, older trees (Thomas et al. 1979, Zarnowitz and Manuwal 1985, Ripper et al. 2007).

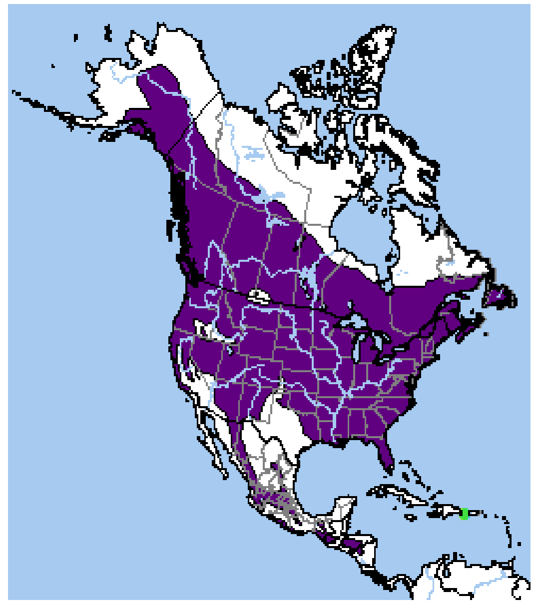

Figure A-32. Current range of the hairy woodpecker in North America (Ridgely et al. 2005).

HSI Model

The model for hairy woodpecker is based on initial work done by Sousa (1987) and modified by O'Neil et al. (1988). The model has both a nesting and foraging component. The first variable used in the nesting portion of the model is snag density (Figure A-33). Thomas et al. (1979) suggested that a density of 2 snags per acre >10 inches was necessary to provide for maximum occupancy by hairy woodpeckers for nesting.

The second variable for nesting is the mean diameter of overstory trees (Figure A-34). A minimum tree size of 10 inches has been suggested for nesting trees (Thomas et al. 1979). However, suitability for nesting has been shown to increase with the average diameter of snags (Mannan et al. 1980, Zarnowitz and Manuwal 1985). Older treatment sites with larger diameter trees also tend to have larger diameter snags and hairy woodpeckers have been shown to select older treatment sites for nesting (Ripper et al. 2007). In Washington the average nest tree had a diameter of 22.8 inches (n=16) (Zarnowitz and Manuwal 1985). Nest trees in Oregon had an average diameter of 36.2 inches (n=7) (Mannan et al. 1980).

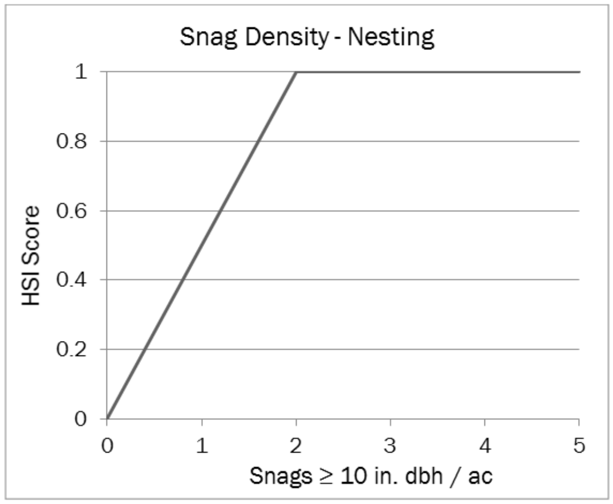

Figure A-33. HSI values for hairy woodpecker nesting based on the density of snags ≥ 10 in. dbh per acre. The equation between 0 and 2 is $y=0.5x$. For snag densities >2/ac the HSI value is 1.0.

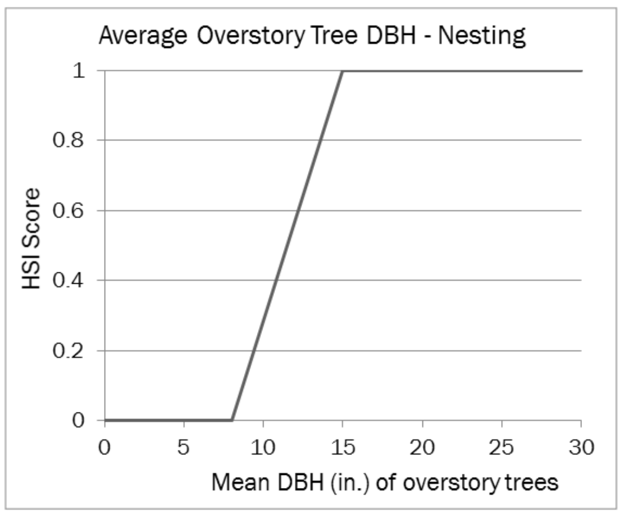

Figure A-34. HSI values for hairy woodpecker nesting based on the average DBH (in.) of overstory trees. The equation between 8 and 15 is $y=0.1429x-1.1429$.

The third variable is the canopy cover of the tree overstory (Figure A-35). Hairy woodpeckers forage in open areas with snags, but predominately nest in treatment sites with tree cover (Mannan et al. 1980, Zarnowitz and Manuwal 1985). However, they have been shown to avoid nesting in treatment sites that are extremely dense or have complete canopy closure (Verner 1980).

The HSI value for the nesting component was calculated with the following formula:

*Nesting HSI = Snag Density HSI * Canopy Cover HSI + (0.75 * DBH HSI)* - maximum value is 1.0

The foraging component of the hairy woodpecker model uses two habitat variables. The first variable is the mean diameter of overstory trees (Figure A-36). Hairy woodpeckers have been shown to selectively forage on larger diameter trees (Weikel and Hayes 1999, Ripper et al. 2007)). In Oregon, the mean diameter of tree used for foraging was >23.6 inches (Mannan et al. 1980).

The second variable in the foraging component is the density of foraging sites (Figure A-37). Hairy woodpeckers forage on live trees, treatment site snags, and downed wood (Thomas et al. 1979, Sousa 1987, Weikel and Hayes 1999). Large diameter, heavily decayed logs were the primary selected foraging sites in Oregon (Weikel and Hayes 1999). High densities of foraging sites provide year round food supply and increase the suitability of a site.

The foraging HSI value was calculated by taking the geometric mean of the two foraging variables.

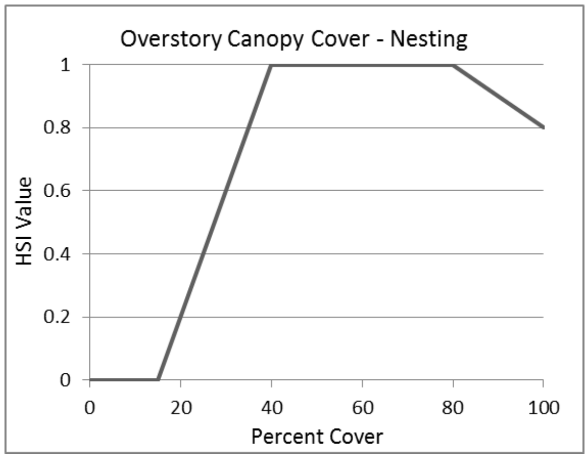

Figure A-35. HSI values for hairy woodpecker nesting based on the canopy cover of overstory trees. The equation between 15 and 40 is *y=0.04x-0.6*. The equation between 80 and 100 is *y=-0.01x+1.8*.

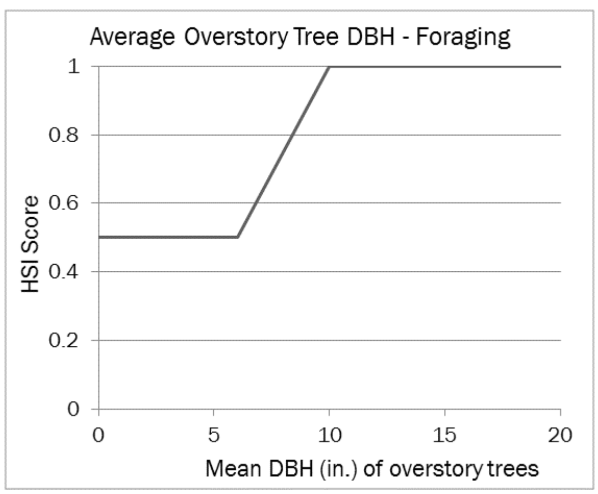

Figure A-36. HSI values for hairy woodpecker foraging based on the average DBH (in.) of overstory trees. The equation between 6 and 10 is *y=0.125x-0.25*.

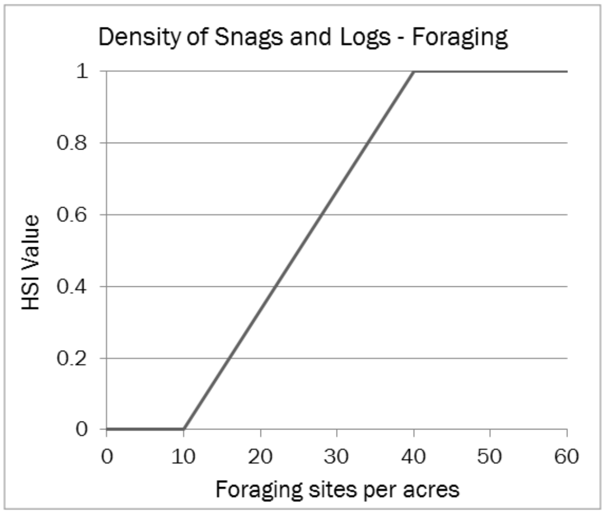

Figure A-37. HSI values for hairy woodpecker foraging based on the density of snags >10 in., stumps >1.5 ft. tall and >10 in. diameter and logs >10 in. diameter and >6 ft. long. The equation between 10 and 40 is *y=0.0333x-0.3333*.

HSI values were determined for each vegetation category in the Southwest Crown of the Continent project area. Values for specific treatment site types were determined from FIA plot data applicable to the project area. For vegetation classes missing treatment site data values for each variable were estimated from the most similar vegetation conditions that had empirical data. HSI scores were then aggregated and contoured using a moving window analysis to produce the final input layer needed for HOMEGROWER. The size of the moving window is equal to the allometric home range (Roloff and Haufler 1997). The allometric home range for a 68 g female hairy woodpecker is 4.5 ha (Van Horne and Wiens 1991).

Three iterations were done in HOMEGROWER. The target home range area was 5 times the allometric home range or 22.5 ha. The number of seeds was 500,000 and the growth window was 5 cells. For the nesting grid the target home range was 5 ha.

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