Appendix A. Focal Species Modeling Background

Contents

Sharp-tailed Grouse (*Tympanuchus phasianellus*)

Prepared by Leslie Robb (Independent Researcher)

Justification for Selection

Sharp-tailed Grouse (*Tympanuchus phasianellus*) were chosen as a focal species to represent the Semi-desert vegetation class in eastern Washington. Describing the relationship between landscape pattern and how Sharp-tailed Grouse perceive that pattern will help identify habitat concentration areas (HCAs) and linkages between those areas. Connectivity analyses that further our understanding of how landscape patterns influence Sharp-tailed Grouse mobility, and ultimately gene flow, are critical for conservation of Sharp-tailed Grouse in Washington.

Sharp-tailed Grouse were historically found throughout steppe, grassland and mixed-shrub habitats of northern and central North America (Connelly et al. 1998). Populations were originally found in 21 states and 8 Canadian provinces. Numbers of Sharp-tailed Grouse have declined range-wide since the 1900s, especially in the eastern and southern portions of the range (Connelly et al. 1998). Primary factors attributed to this decline include but are not limited to, conversion of native habitat to cropland, excessive grazing by livestock, herbicide treatments, removal of trees in riparian areas, invasion of conifers, urban development and fire suppression (Giesen & Connelly 1993; Connelly et al. 1998).

There are seven recognized sub-species (one extinct) of Sharp-tailed Grouse in North America (Connelly et al. 1998); the subspecies found in Washington, Idaho, Oregon and British Columbia is the Columbian Sharp-tailed Grouse (*T. p. columbianus*; Aldrich & Duvall 1955). The distribution of Columbian Sharp-tailed Grouse has severely contracted range-wide and <10% of the historical range is currently occupied (Bart 2000). Range contraction has been particularly severe in Washington where Columbian Sharp-tailed Grouse inhabit <3% of their 79,865 km^2 historical range (Schroeder et al. 2000; Stinson & Schroeder 2010).

Sharp-tailed Grouse are classified as a game bird under Washington state law but hunting has been closed since 1988 (Stinson & Schroeder 2010). They are listed as Threatened by the state of Washington and are designated a Priority Species and their habitats Priority Habitats, by the Washington Department of Fish and Wildlife (WDFW) Priority Habitats and Species Program (Hays et al. 1998). In 1995 the U.S. Fish and Wildlife Service (USFWS) was petitioned to list Sharp-tailed Grouse as Threatened under the federal Endangered Species Act; listing was not considered warranted (USFWS 2000). The Columbian subspecies was again petitioned for federal listing as Threatened or Endangered in 2004; the USFWS (2006) determined that listing was not warranted. However, Bart (2000) in a status review for the USFWS predicted that without federal protection most of the small remnant populations of Columbian Sharp-tailed Grouse, like those in Washington, would likely be extirpated in a decade or two.

Distribution

The historical range of the Columbian subspecies of Sharp-tailed Grouse (hereafter Sharp-tailed Grouse) included central-interior British Columbia south to California, Nevada and Utah, east to western Montana, Wyoming, Colorado and possibly New Mexico (Aldrich & Duvall 1955; Bart

2000). Habitat loss and degradation are the primary factors responsible for population declines (Bart 2000). By the mid-1900s Sharp-tailed Grouse were extirpated from California, Nevada and Oregon; remnant populations (≤1000 birds) occur in Washington, Montana and Wyoming. Populations of ≥5000 birds occur in Idaho, Utah, Colorado, and British Columbia (Schroeder et al. 2000).

Beginning in the early 1990s, efforts were made to reestablish populations of Sharp-tailed Grouse in extreme northeastern Oregon and northeastern Nevada. These translocation projects still continue. Since the early 1900s numbers of Sharp-tailed Grouse have declined an estimated 70% in British Columbia, especially in grassland habitats in the south-central interior. They have expanded their range and numbers in the forested areas of the central interior in response to large-scale timber harvesting that has created suitable habitat. This increase is predicted to be temporary as clear cuts regenerate (Leupin 2003).

Historical evidence indicates that Sharp-tailed Grouse were abundant and widely distributed in eastern Washington (Schroeder et al. 2000; Stinson & Schroeder 2010). Significant population declines were observed in the late 1800s and continued steadily throughout the 1900s; the overall estimated decline was 91.5% between 1954 and 1998 (Schroeder et al. 2000). The current distribution in the state encompasses 2234 km^2 or 2.8% of the historical range (Schroeder et al. 2000). There are an estimated 800 Sharp-tailed Grouse in Washington (M. Schroeder, personal communication) distributed among seven small, isolated populations in Okanogan, Douglas, and Lincoln counties (Hays et al. 1998; Schroeder et al. 2000; Stinson & Schroeder 2010). More than half (51.1%) of the active breeding complexes associated with these remnant populations are located on private land, 24.4% are on state or federal land, and 24.4 % are on lands of the Colville Confederated Tribes (Schroeder et al. 2000).

Habitat Associations

GENERAL

Sharp-tailed Grouse use a variety of habitats in Washington including shrubsteppe, meadowsteppe, mountain shrub, and riparian/deciduous habitats (Hays et al. 1998). The presence of dense herbaceous vegetation and shrubs is of key importance. Plant species composition is secondary to structural characteristics of the habitat (Connelly et al. 1998). Grassland habitats provide breeding and nesting areas while deciduous trees and shrubs in upland and riparian areas provide essential food and cover in winter (Giesen & Connelly 1993). Historical densities of Sharp-tailed Grouse in Washington are believed to have been highest where steppe habitats were interspersed with riparian, forest-edge and mountain-shrub habitats (Schroeder et al. 2000; Stinson & Schroeder 2010). Topographic features such as elevation and slope vary throughout the range. In Washington, Sharp-tailed Grouse are found at elevations of 300–1350 m but in other parts of their range occur as high as 2900 m (Hays et al. 1998). Slope tends to be used in proportion to availability. In southwestern Idaho, Saab and Marks (1992) noted that in summer birds used habitats with 0–47% slope, however most observations (>95%) were <30% slope.

Conservation Reserve Program (CRP) lands are owned by farmers but set aside as wildlife habitat with compensation from the federal government. A review of prairie grouse and CRP lands concluded that more than any other species, Sharp-tailed Grouse responded positively (both populations and range) to CRP in 10 of 12 states where they occur. CRP is considered integral to conservation of Sharp-tailed Grouse in Colorado, Utah and Idaho (Rodgers & Hoffman 2005). Bart (2000) believed termination or modification of the CRP so that it no longer supports Sharp-tailed Grouse would result in severe declines and extirpation of populations. McDonald (1998) monitored radio-tagged birds on the Swanson Lakes Wildlife Area in Washington and found that CRP lands were used more than expected by availability; 11 of 17 nests he found at SLWA were situated in CRP. However, nest success was significantly lower (18%) in CRP lands planted mainly with crested wheatgrass (*Agropyron cristatum*) than in two other sites where cover type consisted of native bunch-grasses and forbs (38% and 47%; McDonald 1998).

BREEDING

In spring, Sharp-tailed Grouse congregate on traditional sites termed leks for courtship and breeding. Leks are typically situated on elevated areas such as knolls or ridgetops but lower sites are also used (Giesen & Connelly 1993). Leks are often located in disturbed areas or on sites with less vegetation than surrounding areas (Connelly et al. 1998). Factors important for nesting and brood rearing habitat include (1) vegetation density, (2) vegetation height, and (3) diversity of forbs and bunchgrasses (Giesen & Connelly 1993). Abundance of forbs and insects, high shrub density, and interspersion of cover types are features of good brood habitat (Connelly et al. 1998).

WINTER

In winter Sharp-tailed Grouse rely on deciduous trees and shrubs in upland and riparian areas for food and cover (Connelly et al. 1998). In western Idaho Mark and Marks (1988) found 88% of all winter locations of Sharp-tailed Grouse were <50 m from mountain shrub or riparian cover types.

SENSITIVITY TO DEVELOPMENT

Initial declines of Sharp-tailed Grouse occurred in the early 1900s when much of eastern Washington was dramatically altered by livestock grazing, agriculture and development. Since then numerous factors have contributed to the continued conversion, fragmentation, and degradation of habitat and isolation of populations (Stinson & Schroeder 2010). Sharp-tailed Grouse currently occupy small portions of Douglas, Grant, and Lincoln counties, about 3% of their historical distribution (Stinson & Schroeder 2010). Occupied range exists in higher elevation steppe habitats where ranching and farmland are predominant land use practices and human population density is relatively low. Conversion of these areas to residential development, even at a low density, potentially decreases habitat suitability and connectivity among occupied areas (Stinson & Schroeder 2010).

SENSITIVITY TO TRAFFIC

Evidence from radio-marked birds introduced to the Scotch Creek Wildlife Area in Washington indicates that Sharp-tailed Grouse crossed U.S. Highway 97 in Okanogan County. Birds tended to cross the highway where it was situated in a valley between ridges (M. Schroeder, personal communication) suggesting that landscape resistance from highways may be mitigated by topography, and that crossing areas may be limited. The extent to which Sharp-tailed Grouse avoid roads is unknown. However, Connelly et al. (2004) examined the distribution of 804 Greater Sage-Grouse (*Centrocercus urophasianus*) leks in southern Wyoming and found no leks within 2 km (4 km band) of Interstate 80 (I-80), and only 9 leks within 2–4 km (>4–8 km band) of I-80. Additionally, analysis of Greater Sage-Grouse populations between 1970 and 2003 indicated that leks within 7.5 km of I-80 appeared to decline at a higher rate than leks 7.5–15.0 km from I-80.

FOREST STRUCTURE

Sharp-tailed Grouse in eastern Washington are often associated with edge habitats where steppe vegetation transitions to dry forest. In some parts of the current distribution ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) have invaded and increased in density, potentially decreasing habitat suitability (Stinson & Schroeder 2010). Sharp-tailed Grouse occupy forest clear cuts in British Columbia however these areas are large-scale harvesting operations (M. Schroeder, personal communication).

Dispersal

Little is known about natal dispersal (movement from hatch location to place of first breeding or attempted breeding) by juvenile Sharp-tailed Grouse. Gratson (1988) recorded natal dispersal for one Sharp-tailed Grouse in Wisconsin; a juvenile female nested 1.4 km from the range it used as a chick. Seasonal movement information for Sharp-tailed Grouse is limited to data collected from radio-marked birds captured at leks and monitored throughout the year (Table A.1).

From spring through fall Sharp-tailed Grouse move fairly short distances; females in Washington nested an average 1.3 km from the leks where they were captured (Schroeder 1994). Distances moved by Sharp-tailed Grouse are likely influenced by landscape features. For instance, Schroeder (1996) found seasonal distances moved by Sharp-tailed Grouse differed significantly by county in Washington, with migration distance largest in Lincoln County. By comparison, Boisvert et al. (2005) monitored Sharp-tailed Grouse on CRP and Mine Reclamation lands in northwestern Colorado: during winter birds were a median distance of 21.5 km from lek sites where they were captured. The shorter distances moved by Sharp-tailed Grouse in Washington may be influenced by the fragmented nature of the populations (M. Schroeder, personal communication).

Table A.1. Seasonal movements of Sharp-tailed Grouse.

	Distance (km)			
Category/location	*mean*	*median*	*maximum*	*Citation*
Females				
hunter return juveniles/SD	21.6			Robel et al. 1972
breeding to winter range/WA	4.4			Schroeder 1996
breeding to winter range/CO	21.4			Boisvert et al. 2005
from capture, autumn/CO		1.5		Giesen 1997
from capture lek, spring/WA	2.0			McDonald 1998
from capture lek, spring/WA	1.0			McDonald 1998
from capture lek, winter/WA	2.3			McDonald 1998
from capture lek, winter/WA	5.6			McDonald 1998
from capture lek, winter/WA			11.5	McDonald 1998
Males				
hunter return juveniles/SD	13.6			Robel et al. 1972
breeding to winter range/WA	2.8			Schroeder 1996
breeding to winter range/CO	21.5			Boisvert et al. 2005
from capture lek, autumn/CO		0.6		Giesen 1997
from capture lek, spring/WA	0.6			McDonald 1998
from capture lek, spring/WA	0.7			McDonald 1998
from capture lek, winter/WA	2.8			McDonald 1998
from capture lek, winter/WA	1.0			McDonald 1998
from capture lek, winter/WA			9.7	McDonald 1998

Sharp-tailed Grouse were released on the Scotch Creek Wildlife Area by WDFW in 1998 as part of a population augmentation project. Based on 1300 observations of 24 radio-marked individuals, birds moved an average of 12–17 km from the wildlife area during the initial weeks following release (Schroeder & Peterson 1998). Movement behavior suggested that the released birds "explored" potential habitat in the region. All the birds eventually returned to the wildlife area except for 2 females who moved 34 and 31 km respectively and established in the Tunk Valley and Siwash Creek areas after "finding" local populations of Sharp-tailed Grouse. M. Schroeder (personal communication) suggested that the released birds had a search distance of about 16 km and returned to the Scotch Creek Wildlife Area when they did not encounter other Sharp-tailed Grouse.

Modeling Efforts

Meints et al. (1992) and Ashley (2006) have constructed habitat suitability models for Sharp-tailed Grouse. In general, there is a lack of specific modeling analyses examining anthropogenic disturbances and their impact on Sharp-tailed Grouse.

Conceptual Basis for Statewide Connectivity Model Development

OVERVIEW

Habitats used by Sharp-tailed Grouse are well documented (Connelly et al. 1998; Stinson & Schroeder 2010). Optimal habitat consists of a mosaic of shrubsteppe, meadowsteppe, mountain shrub, and riparian/deciduous plant communities. Grassland type habitats are used for nesting and brood rearing while deciduous trees and shrubs in upland and riparian areas are essential for food and cover in winter. What is less understood is how various habitat types, especially altered habitats, influence movement of Sharp-tailed Grouse through the landscape.

To characterize landscape resistance for Sharp-tailed Grouse we used whenever possible documented habitat associations. Behavioral responses to constructed habitat features, such as buildings and roads, are more difficult to document and most evidence comes from research on other species of prairie grouse. For instance, Greater Sage-Grouse, Lesser Prairie-Chickens (*Tympanuchus pallidicinctus*) and Greater Prairie-Chickens (*Tympanuchus cupido*) tend to avoid vertical structures in the landscape (Braun 1998; Pitman et al. 2005; Pruett et al. 2009). When information was lacking for Sharp-tailed Grouse, we relied upon the professional judgment and knowledge of grouse biologists to score resistance values. Urban development, human population density, and roads were considered major factors contributing to landscape resistance for Sharp-tailed Grouse.

Sharp-tailed Grouse select habitat at multiple scales. Following Johnson’s (1980) orders of habitat selection these scales have a nested hierarchy for instance: feeding site, seasonal use area, home range, population area, and range. Movements of Sharp-tailed Grouse can also be thought of in a similar way: daily movements, home range, seasonal migration, and dispersal. Ecological and human caused disturbances at one scale can affect disturbances at other scales. Our model of habitat connectivity takes a broad-scale approach. We examine connectivity among (sub) populations of Sharp-tailed Grouse at the statewide scale.

MOVEMENT DISTANCE

Based on daily and seasonal movement distances (Table A.1), and additional information regarding Sharp-tailed Grouse in Washington (See Dispersal subsection) we used a cost-weighted distance of 40 km to indicate available habitat suitable for movement outside of the identified HCAs.

Landscape Features and Resistance Values

The modeled resistance surface for Sharp-tailed Grouse (Table A.2) is based on the following spatial data layers:

1) Land cover/land-use
2) Elevation
3) Slope
4) Housing density

5) Road type and distance
6) Forest structure

Habitat Concentration Areas

Habitat concentration areas for Sharp-tailed Grouse within the assessment area have been spatially defined by extensive surveys: we obtained corresponding GIS data from appropriate sources. Washington Department of Fish and Wildlife GIS distribution data were used to delineate most HCAs and correspond to the occupied areas identified as Chesaw, Tunk Valley, Scotch Creek, Greenaway Springs, Nespelem, Dyer Hill, and Swanson Lakes in the WDFW Sharp-tailed Grouse Recovery Plan (Stinson & Schroeder 2010). Boundaries of these occupied areas were based on active lek locations, movements of radio-marked birds, observations of birds year-round, and distribution of occupied habitat. The Methow Sharp-tail Recovery Unit has been designated by WDFW as an area important for Sharp-tailed Grouse in the state (D. Stinson & M. Schroeder, personal communication), thus we included it in the model. This recovery unit has been assigned the highest ranking by WDFW for potential reintroduction of Sharp-tailed Grouse for the following reasons: (1) the area is within the historical distribution of Sharp-tailed Grouse, (2) occasional birds were observed in the recovery unit as late as the 1980s, (3) the unit has suitable habitat for Sharp-tailed Grouse and 31,000 ac is currently managed by WDFW, and (4) habitat restoration efforts have improved the vegetation in recent years (Stinson & Schroeder 2010). The Methow Recovery Unit HCA was delineated using WDFW GIS data. Core areas in Oregon were delineated by Christian Hagen, Oregon Department of Fish and Wildlife (ODFW). Core areas in British Columbia and Idaho were based on WDFW Sharp-tailed Grouse range-wide mapping products (Stinson & Schroeder 2010).

Linkage Modeling

Based on Sharp-tailed Grouse movement information (see Dispersal and Movement Distance subsections) a cost-weighted distance of 80 km was used to define the maximum corridor length in the normalized least-cost corridor analysis and a corridor mapping cutoff of 25 km cost-weighted distance was used.

Considerations for Future Modeling

Future modeling at the ecoregional scale should: (1) parse agricultural lands into categories that include CRP, dryland and irrigated agricultural cover types, and buffers extending into croplands from adjacent shrubsteppe, (2) examine barrier effects of powerlines and wind energy facilities, and (3) examine potential changes in precipitation zones as a result of projected climate change.

Table A.2. Landscape features and resistance values used to model habitat connectivity for Sharp-tailed Grouse.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	6
urban/developed	1000
water	10
sparsely vegetated	6
alpine	20
riparian	1
wetland	8
grass-dominated	0
shrub-dominated	0
dry forest	4
wet forest	50
Elevation (meters)	
0–250	0
> 250–500	0
> 500–750	0
> 750–1000	0
> 1000–1500	0
> 1500–2000	0
> 2000–2500	10
> 2500–3300	50
> 3300	500
Slope (degrees)	
0–20	0
> 20–40	1
> 40	2
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	10
> 20 ≤ 40	100
> 10 ≤ 20	1000
< 10	1000
Road type and distance (meters)*	
freeway > 500–1000 buffer	0
freeway > 0–500 buffer	4
freeway centerline	50
major highway > 500–1000 buffer	0
major highway > 0–500 buffer	3
major highway centerline	30
secondary highway > 500–1000 buffer	0
secondary highway > 0–500 buffer	1
secondary highway centerline	5
local road > 500–1000 buffer	0
local road > 0–500 buffer	0
local road centerline	2
Forest structure (density and height [meters])	
nonforest	0
sparse low (0–40%, ≤ 25)	2
sparse high (0–40%, > 25)	4
open low (> 40–70%, ≤ 25)	6
open high (> 40–70%, > 25)	8
dense low (> 70–100%, ≤ 25)	10
dense high (> 70–100%, > 25)	10

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Acknowledgements

Brian Cosentino (WDFW), Kelly McAllister (WSDOT), Andrew Shirk (Independent Researcher) and Brad McRae (TNC) provided assistance with model development and review. In addition the following persons provided input and review of the model: Mike Schroeder (WDFW), Derek Stinson (WDFW), Mike Livingston (WDFW), Mike Atamian (WDFW), Jeff Bernatowicz (WDFW), Richard Whitney (CCT), and Joanne Schuett-Hames (WDFW).

Literature Cited

Aldrich, J. W., and A. J. Duvall. 1955. Distribution of American gallinaceous game birds. Circular 34. U. S. Department of Interior, Fish and Wildlife Service.

Ashley, P. 2006. Draft Sharp-tailed Grouse habitat suitability model. Washington Department of Fish and Wildlife, Olympia, Washington.

Bart, J., 2000. Status assessment and conservation plan for Columbian Sharp-tailed Grouse. Forest and Rangeland Ecosystem Science Center, U.S. Geological Survey, Boise, Idaho.

Boisvert, J. H., R. W. Hoffman, and K. P. Reese. 2005. Home range and seasonal movements of Columbian sharp-tailed grouse associated with Conservation Reserve Program and mine reclamation. Western North American Naturalist 65:36–44.

Braun, C. E. 1998. Sage grouse declines in western North America: what are the problems? Proceedings of the Western Association of State Fish and Wildlife Agencies 78:139–156.

Connelly, J. W., M. W. Gratson, and K. P. Reese. 1998. Sharp-tailed grouse (*Tympanuchus phasianellu*s). in A. Poole and F. Gill, editors. The Birds of North America, No. 354. The Birds of North America, Philadelphia, PA.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of Greater Sage-Grouse and sagebrush habitats. Unpublished report. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming.

Giesen, K. M. 1997. Seasonal movements, home ranges, and habitat use by Columbian sharp-tailed grouse in Colorado. Special Report No. 72. Colorado Division of Wildlife, Fort Collins, Colorado.

Giesen, K. M., and J. W. Connelly. 1993. Guidelines for management of Columbian sharp-tailed Grouse habitats. Wildlife Society Bulletin 21:325–333.

Gratson, M. W. 1988. Spatial patterns, movements, and cover selection by sharp-tailed grouse. Pages 158–192 in A. T. Bergerud and M. W. Gratson, editors. Adaptive strategies and population ecology of northern grouse. University of Minnesota Press, Minneapolis, Minnesota.

Hays, D. W., M. J. Tirhi, and D. W. Stinson. 1998. Washington State status report for the Sharp-tailed grouse. Washington Department of Fish and Wildlife, Olympia, Washington.

Johnson, D. H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology 61:65–71.

Leupin, E. E. 2003. Status of the Sharp-tailed grouse (*Tympanuchus phasianellus*) in British Columbia. Wildlife Bulletin No. B-104. B. C. Ministry of Water, Land and Air Protection, Biodiversity Branch, Victoria, British Columbia, Canada.

Marks, J. S., and V. A. Marks. 1988. Winter habitat use by Columbian sharp-tailed grouse in Western Idaho. Journal of Wildlife Management 52:743–745.

McDonald, M. W. 1998. Ecology of Columbian sharp-tailed grouse in eastern Washington. Master's thesis. University of Idaho, Moscow.

Meints, D. R., J. W. Connelly, K. P. Reese, A. R. Sands, and T. P. Hemker. 1992. Habitat suitability index procedure for Columbian sharp-tailed grouse. Bulletin 55. Idaho Forest, Wildlife and Range Experiment Station, University of Idaho, Moscow.

Pitman, J. C., C. A. Hagen, R. J. Robel, T. M. Loughin, and R. D. Applegate. 2005. Location and success of lesser prairie-chicken nests in relation to vegetation and human disturbance. Journal of Wildlife Management 69:1259–1269.

Pruett, C. L., M. A. Patten, and D. H. Wolfe. 2009. It's not easy being green: wind energy and a declining grassland bird. BioScience 59:257–262.

Robel, R. J., F. R. Henderson, and W. Jackson. 1972. Some sharp-tailed grouse population statistics from South Dakota. Journal of Wildlife Management 36:87–98.

Rodgers, R. D., and R. W. Hoffman. 2005. Prairie grouse population response to Conservation Reserve Program grasslands: an overview. Pages 120–128 in A. W. Allen and M. W. Vandever, editors. The Conservation Reserve Program - planting the future: proceedings of a national conference, Scientific Investigation Report 2005-5145. U. S. Geological Survey, Biological Resources Division, Fort Collins, Colorado.

Saab, V. A., and J. S. Marks. 1992. Summer habitat use by Columbian sharp-tailed grouse in western Idaho. Great Basin Naturalist 52:166–173.

Schroeder, M. A. 1994. Productivity and habitat use of sharp-tailed grouse in north-central Washington. Job Progress Report. Washington Department of Wildlife, Olympia, Washington.

Schroeder, M. A. 1996. Productivity and habitat use of sharp-tailed grouse in north-central Washington. Job Progress Report. Washington Department of Wildlife, Olympia, Washington.

Schroeder, M. A., D. W. Hays, M. A. Murphy, and D. J. Pierce. 2000. Changes in the distribution and abundance of Columbian sharp-tailed grouse in Washington. Northwestern Naturalist 81:95–103.

Schroeder, M. A., and D. J. Peterson. 1998. Translocation of sharp-tailed grouse to the Scotch Creek Wildlife Area in north-central Washington. Job Progress Report. Washington Department of Wildlife, Olympia, Washington.

Stinson, D. W., and M. A. Schroeder. 2010. Draft Washington Sate recovery plan for the Columbian Sharp-tailed Grouse. Washington Department of Fish and Wildlife, Olympia, Washington.

USFWS (U. S. Fish and Wildlife Service). 2000. 12-Month finding on a petition to list the Columbian Sharp-tailed Grouse as threatened. Federal Register 65(197):60391–60396.

USFWS (U. S. Fish and Wildlife Service). 2006. 90-Day finding on a petition to list the Columbian Sharp-tailed grouse as threatened. Federal Register 71(224):67318–67325.

Personal Communication

Michael Schroeder, Upland Game Bird Research Scientist
Washington Department of Fish and Wildlife
Olympia, Washington

Derek Stinson, Biologist
Washington Department of Fish and Wildlife
Olympia, Washington

Greater Sage-Grouse (*Centrocercus urophasianus*)

Prepared by Leslie Robb (Independent Researcher)

Justification for Selection

Greater Sage-Grouse were chosen as a focal species to represent the Semi-desert vegetation class in eastern Washington. Describing the relationship between landscape pattern and how Greater Sage-Grouse perceive that pattern will help identify HCAs and linkages between these areas. Connectivity analyses that further our understanding of how landscape patterns influence Greater Sage-Grouse mobility, and ultimately gene flow, are critical for conservation of Greater Sage-Grouse populations in Washington.

Greater Sage-Grouse are considered a shrubsteppe obligate species because of their year round dependence on sagebrush (*Artemisia* spp.) dominated habitats for food and cover (Schroeder et al. 1999). They are a landscape species that have large home ranges, are capable of extensive movements, and use a mosaic of habitat patch sizes within the sagebrush ecosystem (Connelly et al. 2004). Greater Sage-Grouse are sensitive to disturbance from human activities as well as the configuration and juxtaposition of suitable habitat in the landscape (Braun 1986; Lyon & Anderson 2003; Connelly et al. 2004; Aldridge 2005; Aldridge & Boyce 2007). An analysis of Greater Sage-Grouse population trends and sagebrush habitat concluded that intensity of the human footprint was higher in areas where Greater Sage-Grouse were extirpated than areas still occupied (Connelly et al. 2004). Validation of landscape resistance models for Greater Sage-Grouse is possible because the WDFW has committed to long term monitoring of this species. As well, Greater Sage-Grouse behavior can be monitored using radio-telemetry techniques (Stinson et al. 2004).

Habitat loss, degradation, and fragmentation of native shrubsteppe vegetation resulting from altered fire regimes, conversion of shrubsteppe to agriculture, urban development, energy development, grazing, mining, military activity, noise, powerlines, roads, fences and encroachment by invasive plant species threaten the persistence of Greater Sage-Grouse populations in Washington (Schroeder et al. 2003; Stinson et al. 2004). Additional threats include loss of genetic diversity through population isolation (Stinson et al. 2004); genetic evidence suggests that these populations have already undergone a genetic bottleneck (Benedict et al. 2003; Oyler-McCance et al. 2005).

Greater Sage-Grouse are listed as Threatened by the state of Washington and are considered a Priority Species by the WDFW Priority Habitats and Species Program (Hays et al. 1998; Schroeder et al. 2003; Stinson et al. 2004). In 2001, the USFWS determined that Greater Sage-Grouse in Washington constituted a distinct population segment and listing under the federal Endangered Species Act was warranted but precluded due to higher listing priorities (USFWS 2001). Petitions for listing Greater Sage-Grouse range-wide were filed in 2002 and 2003; in 2005 the USFWS determined that listing was not warranted. In 2008, a status review was initiated by the USFWS to address new information available since the 2005 finding. In March 2010 the USFWS determined that the range-wide listing of Greater Sage-Grouse under the federal

Endangered Species Act was warranted but precluded due to higher listing priorities; range-wide they are considered a Candidate species with an assigned listing priority of 8 (USFWS 2010).

Distribution

The historical distribution of Greater Sage-Grouse coincided with that of sagebrush in western North America (Schroeder et al. 1999; Schroeder et al. 2004). At one time Greater Sage-Grouse populations were found in parts of 14 states in the western U.S. and 3 Canadian provinces (Connelly et al. 2004; Schroeder et al. 2004). Currently Greater Sage-Grouse occur in 11 states and 2 provinces with an estimated total range of 668,412 km^2, approximately 56% of the pre-settlement distribution of potential habitat (Schroeder et al. 2004).

Greater Sage-Grouse were once widely distributed throughout central and eastern Washington, parts of north-central and eastern Oregon, southern Idaho and in the extreme southern portion of British Columbia following the Okanagan valley (Campbell et al. 1990; Schroeder et al. 2000; Aldridge & Brigham 2003; Schroeder et al. 2004). Initial declines of Greater Sage-Grouse distribution in Washington were related to cultivation of shrubsteppe habitat, primarily for production of wheat, and continued as cultivation expanded throughout the Columbia Basin (Schroeder et al. 2000). Between 1900 and 1940 Greater Sage-Grouse numbers declined in Oregon and by 1955 birds were extirpated from the northern parts of the state (Stinson et al. 2004). Habitat along the Snake River in Idaho likely once supported populations of Greater Sage-Grouse but is currently unoccupied (Schroeder et al. 2004); Greater Sage-Grouse continue to decline in the Snake River region (Connelly et al. 2004). The number of Greater Sage-Grouse in British Columbia was probably always low, they were considered extirpated in 1918 (Campbell et al. 1990; Aldridge & Brigham 2003).

The estimated range of Greater Sage-Grouse in Washington is approximately 4683 km^2 or 8% of the historical range (Schroeder et al. 2000). Connelly et al. (2004) concluded that populations in Washington declined at a rate of 4.8% per year from 1965 to 2003 and current estimates place the state population at approximately 1100 birds (2010 estimate; M. Schroeder, personal communication). Greater Sage-Grouse are found in two populations in Washington. One population is located in the Moses Coulee area in Douglas/Grant counties and one is on the U.S. Army's Yakima Training Center (YTC) in Yakima/Kittitas counties (Schroeder et al. 2000; Stinson et al. 2004). These populations are isolated from each other by approximately 50 km and from populations in Oregon and Idaho by about 250 km and 350 km respectively.

Greater Sage-Grouse were extirpated from Lincoln County in 1987 (Hayes et al. 1998). In 2008 WDFW initiated a translocation project (project timeline from 2008 to 2011) to release Greater Sage-Grouse in the Swanson Lakes Wildlife Area, Lincoln County, in east-central Washington (Schroeder et al. 2008). It is too early to know if this translocation project is successful. Greater Sage-Grouse were introduced to lands managed by the Yakama Nation in 2006. Two leks were observed the year following translocation efforts; these leks have since disappeared (N. Burkepile, personal communication).

Habitat Associations

GENERAL

The distribution of Greater Sage-Grouse is closely allied to the distribution of sagebrush, particularly big sagebrush (*A. tridentata*) in the western U.S. Sagebrush habitat types demonstrate considerable variation across the range in terms of vegetative composition, fragmentation, topography, substrate, weather, and frequency of fire (Schroeder et al. 1999). Because Greater Sage-Grouse use a variety of habitat patches within a larger landscape the juxtaposition and quality of these habitat types is critical.

In Washington, Greater Sage-Grouse habitat includes the shrubsteppe and meadowsteppe plant communities (Stinson et al. 2004). Shrubsteppe plant communities are characterized by bunchgrasses, big sagebrush, three-tipped sagebrush (*A. tripartita*), bitterbrush (*Purshia tridentata*) and forbs. Meadowsteppe habitat is characterized by dense grass and forb cover and fewer shrubs (Stinson et al. 2004). Quality of shrubsteppe is critical as many uncultivated areas are not suitable for Greater Sage-Grouse because of lack of sagebrush, perennial grasses, and forbs (Schroeder et al. 1999). Greater Sage-Grouse may use alfalfa (*Medicago sativa*), wheat (*Triticum* spp.) and crested wheatgrass but use of these altered habitats depends primarily on their configuration with native habitat (Schroeder et al. 1999).

Greater Sage-Grouse in Washington are found in suitable habitat at 300–900 m elevation (Cadwell et al. 1997; Livingston 1998; M. Schroeder, personal communication). Greater Sage-Grouse in Oregon and Idaho are found at 1200–2400 m and 1200–2900 m elevation respectively (WDFW 1995). Within suitable habitats elevation is not likely a significant factor limiting movements by Greater Sage-Grouse (M. Schroeder, personal communication). Similarly, although slope is a component of suitable Greater Sage-Grouse habitat it is not likely to be a factor impeding movement. In Washington, Greater Sage-Grouse generally use habitat where the slope is <7 degrees (Cadwell et al. 1997; Livingston 1998) and ≤18 degrees (M. Schroeder, personal communication).

BREEDING

Leks are traditional breeding areas where males congregate in the spring and perform courtship displays. They are typically situated near nesting habitat and close to relatively dense stands of sagebrush that are used for cover and feeding (Connelly et al. 2004). Leks tend to be located in natural openings such as ridge-tops, grassy swales and dry stream channels as well as openings created by human disturbance such as cultivated fields, airstrips, gravel pits, roads, burned areas and edges of stock ponds (Schroeder et al. 1999; Connelly et al. 2004).

Sagebrush/bunchgrass habitat is used for nesting (Stinson et al. 2004); nests tend to be situated under the tallest sagebrush within a stand (Connelly et al. 2000). Conservation Reserve Program (CRP) lands are owned by farmers but set aside as wildlife habitat with compensation from the federal government. In the Douglas/Grant population female Greater Sage-Grouse nest in CRP areas more than expected by its availability (Schroeder & Vander Haegen 2010). These areas had relatively thick cover of perennial grasses and/or shrubs and tended to be more "mature" (Schroeder & Vander Haegen 2010). Good quality brood habitat is characterized by abundant forbs, insects and high plant diversity (Connelly et al. 2000).

WINTER

Winter habitat for Greater Sage-Grouse consists of large stands of good quality sagebrush that provide food and cover. Presence of sagebrush is essential for survival as it is 100% of the winter diet (Schroeder et al. 1999). Spatial distribution of Greater Sage-Grouse in winter is related to snow depth as sagebrush must be exposed to be accessible for forage (Connelly et al. 2004). Sagebrush stands with canopy cover 10–30% and heights of at least 25–35cm are considered minimal for winter habitat (Connelly et al. 2000).

SENSITIVITY TO DEVELOPMENT

Greater Sage-Grouse are sensitive to the impacts of human related disturbance of sagebrush communities (Johnson et al. 2011; Knick & Hanser 2011; Wisdom et al. 2011). Occupied range in Washington occurs in areas where ranching and farming are the major land uses practices and human population density is low. Sagebrush habitats tend to be characterized by few naturally occurring vertical features. Consequently, man-made structures, like buildings, have the potential to impact Greater Sage-Grouse movements and patterns of habitat use. For instance, studies of Lesser Prairie-Chickens in Kansas indicate that use sites average 336 m further from buildings than non-use sites (Hagen et al. 2004). The ecological effect of vertical structures can be complex as, for example, the footprint of a feature like a housing site is an accumulation of stressors such as buildings, vehicles, trailers, pets, trees, fences, transmission lines, noise and activity.

Aldridge et al. (2008) concluded that throughout Greater Sage-Grouse range the extirpation of a population was most likely in an area where human density in 1950 exceeded four persons per square kilometer. Studies of the impact of oil and gas development indicate that birds abandon leks when oil and gas pads occur at 40-acre spacing (Taylor et al. 2007). Doherty et al. (2008) studied winter habitat selection by Greater Sage-Grouse and concluded that birds were 1.3 times more likely to occupy sagebrush habitats that lacked coal bed/natural gas wells within a 4 km^2 area, compared to those that had the maximum density of 12.3 wells per 4 km^2. In Washington, 15 known leks in the Douglas/Grant population are on average 3.5 km from the nearest dwelling. Anecdotal evidence suggests that noise from human activity is a potential factor affecting Greater Sage-Grouse movement behavior. In the last decade the single lek that has been abandoned was located about a half mile from a gravel crushing pit that operated through the spring breeding season (M. Schroeder, personal communication).

SENSITIVITY TO TRAFFIC

Roads can negatively impact Greater Sage-Grouse by increasing mortality through collisions with vehicles (Stinson et al. 2004). It is more difficult to evaluate how roads affect Greater Sage-Grouse behavior. Lyon and Anderson (2003) concluded that light vehicle traffic (1–12 vehicles/day) in a gas-well development area might reduce nest-initiation rates (65% for disturbed leks versus 89% for undisturbed leks) and increase distances moved from leks during nest-site selection by female Greater Sage-Grouse. Connelly et al. (2004) examined the distribution of 804 leks in southern Wyoming and found no leks within 2 km (4 km band) of I-80, and only 9 leks 2–4 km from the interstate. Additionally, analysis of Greater Sage-Grouse populations between 1970 and 2003 indicated that leks within 7.5 km of I-80 appeared to decline at a higher rate than leks 7.5 to 15.0 km from I-80.

Dispersal

Little information is available regarding dispersal of juvenile Greater Sage-Grouse from their natal territory (hatch area) to location of first breeding (Connelly et al. 2004); one study in Colorado recorded juvenile females (*n* = 12) moving an average 8.8 km and males (*n* = 12) moving 7.4 km (Dunn & Braun 1985). These distances are considerably less than what has been recorded for Greater Sage-Grouse movements in Washington where females often move 13 km between lek sites during a single breeding season. In at least two cases females moved 27 km and 33 km between nest locations within the same nesting season (Schroeder & Robb 2003).

Three types of seasonal migration patterns have been defined for Greater Sage-Grouse: (1) non-migratory (individual movements <10 km between or among seasonal ranges), (2) one-stage migratory, birds move between two distinct seasonal ranges, and (3) 2-stage migratory, birds move among three distinct seasonal ranges (Connelly et al. 2000). Schroeder and Vander Haegen (2003) concluded that the Greater Sage-Grouse population in Douglas/Grant counties demonstrated the characteristics of all three patterns. Distance moved between traditional breeding and winter ranges averaged 16 km for females and 11 km for males; one female Greater Sage-Grouse moved 85 km (Table A.3; Schroeder & Vander Haegen 2003). Greater Sage-Grouse on the YTC are more localized on a seasonal basis (Livingston 1998) possibly because the YTC is surrounded by unsuitable habitat (Schroeder & Vander Haegen 2003).

Migratory corridors for Greater Sage-Grouse are determined by the relationship among habitat requirements, configuration of required habitat and seasonal patterns of Greater Sage-Grouse movement (Connelly et al. 2004). Greater Sage-Grouse intensively monitored during seasonal migration followed shrubsteppe corridors at higher elevations, close to breeding habitat. Birds tended to deviate from a minimal “straight-line” route, instead choosing longer routes in or close to shrubsteppe vegetation (Schroeder & Vander Haegen 2003). Based on occasional sightings, a few Greater Sage-Grouse may occur on the periphery of the current range (Stinson et al. 2004). However, genetic analysis indicates that the YTC population has only 1 haplotype and the Moses Coulee population 3 haplotypes (2 unique) compared to an average of 6.4 for other populations range-wide reflecting little gene flow between these populations (Benedict et al. 2003; Oyler-McCance et al. 2005).

Modeling Efforts

Connelly et al. (2004) modeled anthropogenic disturbance factors in sagebrush habitats throughout Greater Sage-Grouse range; variables included railroads, powerlines, three road layers, campgrounds, rest stops, landfills, irrigation canals, oil-gas wells, human-induced fires, agricultural land, and populated areas. These spatial data sets were used to develop a human footprint model. The Columbia Plateau in Washington had high human footprint influence compared to other parts of Greater Sage-Grouse range. They also concluded that footprint intensity was higher in areas where Greater Sage-Grouse were extirpated. Aldridge et al. (2008) modeled Greater Sage-Grouse persistence range-wide. Extirpation was related to sagebrush habitat, cultivated cropland, human density in 1950, prevalence of severe droughts and historical range periphery. Extirpation of Greater Sage-Grouse was most likely in areas having at least four persons/km^2 in 1950, 25% cultivated cropland in 2002 or the presence of three or more severe droughts per decade. Extirpation was most often explained by the combined effects of peripherality (within 30 km of range edge) and lack of sagebrush cover, less than 25% within 30 km. Knick and Hanser (2011) examined the range-wide connectivity of Greater Sage-Grouse leks. They concluded that Greater Sage-Grouse leks have the lowest level of connectivity in the Columbia Basin Greater Sage-Grouse management zone (Washington) due to the small number of leks and the longer than average distance and few linkages among leks.

Table A.3. Seasonal movements of Greater Sage-grouse.

	Distance (km)			
Category/location	*mean*	*median*	*maximum*	*Citation*
Females				
natal dispersal/CO		8.8		Dunn & Braun 1985
breeding to winter range/WA	16.0			Schroeder & Vander Haegen 2003
breeding to winter range/WA			85.0	Schroeder & Vander Haegen 2003
spring to summer range/ID	12.3			Connelly et al. 1988
winter to breeding range/ID			72.0	Connelly et al. 1988
winter to summer range/ID	32.7			Connelly et al. 1988
winter to summer range/ID	12.3			Connelly et al. 1988
winter to summer range/ID	8.8			Connelly et al. 1988
Males				
natal dispersal/CO		7.4		Dunn & Braun 1985
breeding to winter range/WA	11.0			Schroeder & Vander Haegen 2003
spring to summer range/ID	16.1			Connelly et al. 1988
winter to breeding range/ID			82.0	Connelly et al. 1988
winter to summer range/ID	48.6			Connelly et al. 1988
winter to summer range/ID	16.1			Connelly et al. 1988
winter to summer range/ID	31.5			Connelly et al. 1988
Unknown juveniles				
summer to winter range/ID	14.9			Connelly et al. 1988
Unknown adults				
summer to winter range/ID	11.3			Connelly et al. 1988

Conceptual Basis for Model Development

OVERVIEW

Habitats used by Greater Sage-Grouse are well documented (Schroeder et al. 1999; Connelly et al. 2004; Stinson et al. 2004). What is less understood is how various habitat types, especially altered habitats, influence movement of Greater Sage-Grouse through the landscape. Urban development, human population density and roads were considered major factors contributing to landscape resistance for Greater Sage-Grouse. Although not addressed in our model, we were aware that agriculture varies with respect to resistance to movement for Greater Sage-Grouse; for example CRP habitats are less resistant than cropland. Slope and elevation were not considered significant impediments to movement.

To characterize landscape resistance for Greater Sage-Grouse we used, whenever possible, documented habitat associations. Behavioral responses to man-made habitat features, such as buildings and roads, are more difficult to document and most evidence comes from research on other species of prairie grouse. For instance, Lesser Prairie-Chickens and Greater Prairie-Chickens tend to avoid vertical structures in the landscape (Pitman et al. 2005; Pruett et al. 2009). When information was lacking for Greater Sage-Grouse we relied upon the professional judgment and knowledge of grouse biologists to score resistance values.

Greater Sage-Grouse select habitat at multiple scales. Following Johnson's (1980) orders of habitat selection these scales have a nested hierarchy for instance: feeding site, seasonal use area, home range, population area, and range. Movements of Greater Sage-Grouse can also be thought of in a similar way: daily movements, home range, seasonal migration, and dispersal. Ecological and human caused disturbances at one scale can affect others. Our model of habitat connectivity takes a broad-scale approach. We examine connectivity among (sub) populations of Greater Sage-Grouse at the statewide scale to help identify and prioritize areas for conservation.

MOVEMENT DISTANCE

Most Greater Sage-Grouse in Washington move less than 30 km between seasonal breeding and wintering areas (Table A.3; Schroeder & Vander Haegen 2003). Some birds have been recorded moving considerably further (as far as 85 km), these birds are the ones important for maintaining connectivity among/between (sub) populations. Based on daily and seasonal movement distances (Table A.3), and additional information regarding Greater Sage-Grouse in Washington (See Dispersal) we used a cost-weighted distance of 100 km to indicate available habitat suitable for movement outside of the identified HCAs.

Landscape Features and Resistance Values

The modeled resistance surface for Greater Sage-Grouse (Table A.4) is based on the following spatial data layers:

1) Land cover/land-use
2) Elevation
3) Slope
4) Housing density
5) Road type and distance

Habitat Concentration Areas

Within the assessment area core habitat for Greater Sage-Grouse has been spatially defined by extensive surveys; we obtained corresponding GIS data from appropriate sources. The

Douglas/Grant counties and the YTC core areas were based on WDFW GIS distribution data of known Greater Sage-Grouse populations. These occupied areas were based on active lek locations, movements of radio-marked birds, observations of birds year-round, and distribution of occupied habitat. The Swanson Lakes Wildlife Area polygon was delineated by D. Stinson and represents habitat occupied by reintroduced Greater Sage-Grouse. The Yakama Nation polygon is based on the WDFW Toppenish Ridge Greater Sage-Grouse management unit (Stinson et al. 2004). Greater Sage-Grouse have been reintroduced to this area but there is no known population (Stinson et al. 2004). Core areas in Oregon and Idaho are based on WDFW GIS Greater Sage-Grouse distribution mapping products (Schroeder et al. 2004).

Linkage Modeling

Based on Greater Sage-Grouse movement information (See Dispersal and Movement Distance) a cost-weighted distance of 200 km was used to define the maximum corridor length in the normalized least-cost corridor analysis, we used a corridor mapping cutoff of 25 km cost-weighted distance.

Considerations for Future Modeling

Future modeling at the eco-regional scale should: (1) address the complex nature of agricultural land-cover by considering CRP land as a separate class or by assigning a buffer to agricultural areas (buffer would extend into the agricultural cover), (2) differentiate dryland and irrigated agricultural land-cover types, and (3) include resistance from powerlines and wind energy facilities.

Powerlines can increase Greater Sage-Grouse mortality through collisions and by providing perch sites for raptors (Connelly et al. 2004). Additionally, Greater Sage-Grouse, and other species of prairie grouse tend to avoid vertical structures in the landscape. In Colorado, use of areas by Greater Sage-Grouse, as measured by pellet transects, increased as distance from powerlines increased, up to 600 m (Braun 1998). Pitman et al. (2005) seldom found Lesser Prairie-Chicken nests within 400 m of transmission lines even though the habitat was similar. Preliminary data from radio-tagged Greater Prairie-Chickens in Oklahoma show that individuals appear to stay at least 0.5 km away from powerlines (Pruett et al. 2009). Concern that large vertical structures, like those associated with powerlines, may negatively impact Greater Sage-Grouse prompted the USFWS to recommend that wind turbine construction maintain an 8 km buffer from occupied prairie grouse leks (Manville 2004). In Washington, 19 of 20 leks (95%) documented within 7.5 km of 500 kV powerlines are now vacant, the vacancy rate for leks further away is 59% (22 of 37 leks) (Schroeder 2008). A large powerline corridor crosses the northern portion of the YTC. This area has been rested from grazing since the early 1990s. Although habitat north of the powerline appears suitable, there has been little documented use of this area by Greater Sage-Grouse. Additionally, radio-marked Greater Sage-Grouse released on the YTC have been documented crossing Interstate 90 (I-90) and the Columbia River but no birds have been documented crossing the powerlines (M. Livingston, personal communication).

Wind power is an emerging issue contributing to the overall risk of energy development to sage-grouse populations. Impacts are largely unknown because development is so recent immediate and lag effects have not been identified (Doherty et al. 2011; Knick et al. 2011). Wind energy development within sage-grouse range in Washington was 2.2 gigawatts in 2009, capacity is projected to increase to 5 to 10 gigawatts by 2030 (USFWS 2010). The ecological footprint of large-scale wind energy development has the potential to adversely impact sage-grouse through direct habitat loss, habitat fragmentation, and increased human presence (USFWS 2010).

Table A.4. Landscape features and resistance values used to model habitat connectivity for Greater Sage-Grouse.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	6
urban/developed	1000
water	10
sparsely vegetated	4
alpine	50
riparian	4
wetland	10
grass-dominated	1
shrub-dominated	0
dry forest	100
wet forest	1000
Elevation (meters)	
0–250	0
> 250–500	0
> 500–750	0
> 750–1000	0
> 1000–1500	0
> 1500–2000	0
> 2000–2500	10
> 2500–3300	50
> 3300	500
Slope (degrees)	
0–20	0
> 20–40	1
> 40	2
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	10
> 20 ≤ 40	100
> 10 ≤ 20	1000
< 10	1000
Road type and distance (meters)*	
freeway > 500–1000 buffer	0
freeway > 0–500 buffer	4
freeway centerline	50
major highway > 500–1000 buffer	0
major highway > 0–500 buffer	3
major highway centerline	30
secondary highway > 500–1000 buffer	0
secondary highway > 0–500 buffer	1
secondary highway centerline	5
local road > 500–1000 buffer	0
local road > 0–500 buffer	0
local road centerline	2

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Acknowledgements

Brian Cosentino (WDFW), Kelly McAllister (WSDOT), Andrew Shirk (Independent Researcher), and Brad McRae (TNC) provided assistance with model development. In addition the following persons provided model review and input: Mike Schroeder (WDFW), Derek Stinson (WDFW), Mike Atamian (WDFW), Mike Livingston (WDFW), Jeff Bernatowicz (WDFW), Richard Whitney (CCT), Jack Connelly (IDFG).

Literature Cited

Aldridge, C. L. 2005. Identifying habitats for persistence of Greater Sage-Grouse (*Centrocercus urophasianus*). PhD dissertation. University of Alberta, Edmonton, Alberta.

Aldridge, C. L., and M. S. Boyce. 2007. Linking occurrence and fitness to persistence: habitat based approach for endangered Greater Sage-Grouse. Ecological Applications 17:508–526.

Aldridge, C. L., and R. M. Brigham. 2003. Distribution, abundance, and status of the Greater Sage-Grouse, *Centrocercus urophasianus,* in Canada. Canadian Field Naturalist 117:25–34.

Aldridge, C. L., S. E. Nielsen, H. L. Beyer, M. S. Boyce, J. W. Connelly, S. T. Knick, and M. A. Schroeder. 2008. Range-wide patterns of Greater Sage-Grouse persistence. Diversity and Distributions 14:983–994.

Benedict, N. G., S. J. Oyler-McCance, S. E. Taylor, C. E. Braun, and T. Quinn. 2003. Evaluation of the Eastern (*Centrocercus urophasianus urophasianus*) and Western (*Centrocercus urophasianus phaios*) subspecies of Sage-grouse using mitochondrial and control-region sequence data. Conservation Genetics 4:301–310.

Braun, C. E. 1986. Changes in Sage Grouse lek counts with advent of surface coal mining. Proceedings of Issues and Techniques in the Management of Impacted Western Wildlife 2:227–231.

Braun, C. E. 1998. Sage grouse declines in western North America: what are the problems? Proceedings of the Western Association of State Fish and Wildlife Agencies 78:139–156.

Cadwell, L. L., M. A. Simmons, J. J. Nugent, and V. I. Cullinan. 1997. Sage-grouse habitat on the Yakima Training Center: part II, habitat modeling. PNNL-11758. Pacific Northwest National Laboratory, Richland, Washington.

Campbell, R. W., N. K. Dawe, I. M. Cowan, J. M. Cooper, G. W. Kaiser, and M. C. E. McNall. 1990. The birds of British Columbia. Vol. 2. Royal British Columbia Museum, Victoria.

Connelly, J. W., H. W. Browers, and R. J. Gates. 1988. Seasonal movements of Sage Grouse in southeastern Idaho. Journal of Wildlife Management 52:116–122.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation assessment of Greater Sage-Grouse and sagebrush habitats. Unpublished report. Western Association of Fish and Wildlife Agencies, Cheyenne, Wyoming.

Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967–985.

Doherty, K. E., D. E. Naugle, H., Copeland, A. Pocewicz, and J. Kiesecker. Forthcoming 2011 April. Energy development and conservation tradeoffs. Pages 505–516 in S. T. Knick and J. W. Connelly, editors. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology Series (vol. 38), University of California Press, Berkeley, California.

Doherty, K. E., D. E. Naugle, B. L. Walker, and J. M. Graham. 2008. Greater sage-grouse winter habitat selection and energy development. Journal of Wildlife Management 72:187–195.

Dunn, P. O., and C. E. Braun. 1985. Natal dispersal and lek fidelity of Sage Grouse. Auk 102:621–627.

Hagen, C. A., B. E. Jamison, K. M. Giesen, and T. Z. Riley. 2004. Guidelines for managing lesser prairie-chicken populations and their habitats. Wildlife Society Bulletin 32:69–82.

Hays, D. W., M. J. Tirhi, and D. W. Stinson. 1998. Washington State Status Report for the Sage Grouse. Washington Department of Fish and Wildlife, Olympia.

Johnson, D. H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology 61:65–71.

Johnson, D. H., M. J. Halloran, J. W. Connelly, S. E. Hanser, C. L. Amundson, and S. T. Knick. Forthcoming 2011 April. Influences of environmental and anthropogenic features on Greater Sage-Grouse populations, 1997–2007. Pages 407–450 in S. T. Knick and J. W. Connelly, editors. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology Series (vol. 38), University of California Press, Berkeley, California.

Knick, S. T., and S. E. Hanser. Forthcoming 2011 April. Connecting pattern and process in greater sage-grouse populations and sagebrush landscapes. Pages 383–406 in S. T. Knick and J. W. Connelly editors. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology Series (vol. 38), University of California Press, Berkeley, California.

Knick, S. T., S. E. Hanser, R. F. Miller, D. A. Pyke, M. J. Wisdom, S. P. Finn, E. T. Rinkes, and C. J. Henny. Forthcoming 2011 April. Ecological influence and pathways of land use in sagebrush. Pages 203–251 in S. T. Knick and J. W. Connelly editors. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Studies in

Avian Biology Series (vol. 38), University of California Press, Berkeley, California. In press.

Livingston, M. A. 1998. Western Sage grouse management plan. Department of the Army, Directorate of Environmental and Natural Resources, Yakima Training Center, Yakima, Washington.

Lyon, A. G., and S. H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin 31:486–491.

Manville, A. M., II. 2004. Prairie grouse leks and wind turbines: U.S. Fish and Wildlife Service justification for a 5-mile buffer from leks; additional grassland songbird recommendations. Peer-reviewed briefing paper. Division of Migratory Bird Management, USFWS, Arlington, Virginia.

Oyler-McCance, S. J., S. E. Taylor, and T. W. Quinn. 2005. A multilocus population genetic survey of the Greater Sage-Grouse across their range. Molecular Ecology 14:1293–1310.

Pitman, J. C., C. A. Hagen, R. J. Robel, T. M. Loughin, and R. D. Applegate. 2005. Location and success of lesser prairie-chicken nests in relation to vegetation and human disturbance. Journal of Wildlife Management 69:1259–1269.

Pruett, C. L., M. A. Patten, and D. H. Wolfe. 2009. It’s not easy being green: wind energy and a declining grassland bird. BioScience 59:257–262.

Schroeder, M. A. 2008. Greater sage-grouse and the proposed Withrow wind farm. Unpublished Report. Washington Department of Fish and Wildlife, Olympia, Washington.

Schroeder, M. A., C. L. Aldridge, A. D. Apa, J. R. Bohne, C. E. Braun, S. D. Bunnell, J. W. Connelly, P. A. Deibert, S. C. Gardner, M. A. Hilliard, G. D. Kobriger, S. M. McAdam, C. W. McCarthy, J. J. McCarthy, D. L. Mitchell, E. V. Rickerson, and S. J. Stiver. 2004. Distribution of sage-grouse in North America. Condor 106:363–376.

Schroeder, M. A., D. W. Hays, M. F. Livingston, L. E. Stream, J. E. Jacobson, and D. J. Pierce. 2000. Changes in the distribution and abundance of sage grouse in Washington. Northwestern Naturalist 81:104–112.

Stinson, D. W., D. W. Hays, and M. A. Schroeder. 2004. Washington State recovery plan for the greater sage-grouse. Washington Department of Fish and Wildlife, Olympia, Washington.

Schroeder, M. A., and L. A. Robb. 2003. Fidelity of greater sage-grouse *Centrocercus urophasianus* to breeding areas in a fragmented landscape. Wildlife Biology 9:369–377.

Schroeder, M. A., D. Stinson, H. Ferguson, M. Atamian, and M. Finch. 2008. Re-introduction of sage-grouse to Lincoln County, Washington. Progress Report. Washington Department of Fish and Wildlife, Olympia, Washington.

Schroeder, M. A., D. Stinson, and M. Tirhi. 2003. Greater sage-grouse (*Centrocercus urophasianus*). Priority Habitat and Species Management Recommendations Vol. IV: Birds. Washington Department of Fish and Wildlife, Olympia, Washington.

Schroeder, M. A., and W. M. Vander Haegen. 2003. Migration Patterns of Greater Sage-Grouse in a fragmented landscape. Unpublished Report. Washington Department of Fish and Wildlife, Olympia, Washington.

Schroeder, M. A., and W. M. Vander Haegen. 2010. Response of greater sage-grouse to the Conservation Reserve Program in Washington State. Studies in Avian Biology. Monograph. Available from http://sagemap.wr.usgs.gov/monograph.aspx (accessed June 2010).

Schroeder, M. A., J. R. Young, and C. E. Braun. 1999. Sage grouse (*Centrocercus urophasianus*). Pages 1–28 in A. Poole and F. Gill, editors. The Birds of North America No. 425. The birds of North America, Philadelphia, Pennsylvania.

Taylor, R. C., M. R. Dzialak, and L. D. Hayden-Wing. 2007. Greater sage-grouse populations and energy development in Wyoming. Available from http://bogc.dnrc.state.mt.us/PDF/WYsageGrouseNov2paper.pdf (accessed January 2010)

USFWS (U.S. Fish and Wildlife Service). 2001. 12-month finding for a petition to list the Western population of Western Sage-Grouse (*Centrocercus urophasianus phaios*). Federal Register 66(88):22984–22994.

USFWS (U.S. Fish and Wildlife Service). 2010. 12-month findings for petitions to list the Greater Sage-grouse (*Centrocercus urophasianus*) as threatened or endangered. Federal Register 75(55):13910–14014.

WDFW (Washington Department of Fish and Wildlife). 1995. Washington State management plan for sage grouse. Game Division. Washington Department of Fish and Wildlife, Olympia, Washington.

Wisdom, M. J., C. W. Meinke, S. T. Knick, and M. A. Schroeder. Forthcoming 2011 April. Factors associated with extirpation of Sage-Grouse. Pages 451–472 in S. T. Knick and J. W. Connelly, editors. Greater Sage-Grouse: ecology and conservation of a landscape Species and its habitats. Studies in Avian Biology Series (vol. 38), University of California Press, Berkeley, California.

Personal Communications

Nathan Burkepile, Biologist
Yakama Nation Wildlife
Washington

Mike Livingston, Biologist
Washington Department of Fish and Wildlife
Olympia, Washington

Michael Schroeder, Upland Game Bird Research Scientist
Washington Department of Fish and Wildlife
Olympia, Washington

American Badger (*Taxidea taxus*)

Prepared by Chris Sato (Washington Department of Fish and Wildlife)

Justification for Selection

The American badger (*Taxidea taxus*) was selected as a good representative of wildlife connectivity habitat needs within the Semi-desert vegetation class. It also occurs in the Rocky Mountain Forests classification. Badgers are vulnerable to loss of habitat connectivity from three of the four main connectivity threats: development, roads and traffic, and the presence of people and domestic animals. Badgers tolerate human proximity well, but are at risk for increased mortality from vehicle collisions and persecution. Disturbed conditions such as those created by road right-of-ways and energy transmission corridors can be attractive to badgers. Road cuts and road fill provide good conditions for burrows, both for badgers and their preferred prey. Utility corridors through forested areas can also create preferred conditions for badgers and their prey. Literature shows that they are relatively easily monitored. Their movement scale is appropriate for a statewide modeling effort based on recorded dispersal distances. They are a good choice to represent the Semi-desert vegetation class for the following reasons: they are specialists in that class, they inhabit a wide range of habitat types in that class, and they have relatively large home range and dispersal distances. Badgers are listed as a Species of Greatest Conservation Need in Washington due to habitat loss and human-related threats. In Canada, the subspecies *jeffersonii* is on the British Columbia red list.

Distribution

The American badger (subspecies *jeffersonii*) ranges from British Columbia, Canada to southern California and across the western United States. In Washington, it is an eastside species. South of Vantage (Kittitas County), its range extends up valleys penetrating the East Cascades and across the southern part of the state. North of Vantage, the western edge of its range is east of the Methow Valley in Okanogan County. In the northeast it occurs primarily in the Okanogan Highlands and in the bottoms of the major river drainages (Johnson & Cassidy 1997).

Habitat Associations

American badgers require deep soils and adequate fossorial (burrowing) prey (Messick & Hornocker 1981). Optimal soil types are silty and sandy loams (Apps et al. 2002; Eldridge 2004; Diamond 2006). Soils are pivotal to badger prey species; thus, they are important to badgers as well (Lindzey 1976; T. A. Kinley, personal communication). Badgers may occur in open habitats in eastern Washington including semi-desert, sagebrush, grassland, and pasture/CRP (Conservation Reserve Program) land. They can also be present in open dry forest however forest canopy cover is a significant deterrent (T. Kinley, personal communication). In Washington, except for one record in the Kettle Mountains, they are not known from alpine or subalpine parkland (WDFW 2010). Johnson and Cassidy (1997) designated core zones as steppe, ponderosa pine and oak, and in the Blue Mountains, grand fir and high basalt ridges. However, all Washington recorded badger occurrence points are located in dry shrub, grassland habitat, or on the fringes of agricultural lands with one exception in the Kettle Mountains which was likely a dispersing animal (WDFW 2010). In studies conducted in northern Utah and southern Idaho, badgers were found in big sagebrush and crested wheatgrass patches (Lindzey 1978; Messick &

Hornocker 1981). In Wyoming they occurred in shrub steppe habitats (Goodrich & Buskirk 1998). Paulson (2007) found that at Swanson Lakes in eastern Washington, badgers preferred dense cover of big sagebrush and basin wildrye (*Leymus cinereus*). In southern British Columbia, badgers will use other habitat types such as montane forest, subalpine forest, and alpine areas, although their occurrence in montane and subalpine forests is almost entirely limited to early seral stages; for example cutblocks, burns and roadsides (Newhouse & Kinley 2000). They are typically found at lower elevations, but have been recorded at elevations up to 3660 m (Lindzey 1982). In Washington they occur up to 1981 m (Johnson & Cassidy 1997); nearly all recorded points are in shrubland or agriculture edge habitats where the elevation is at or below 1500 m, and the slope is <20° (WDFW 2010).

Dispersal and Home Range

Young of year disperse in July and August (Case 1978). The longest recorded dispersal distances were 110 km for a male and 52 km for a female (Messick & Hornocker 1981). The authors believe these distances to be well below actual dispersal capability, since nearly all juveniles were sampled during only a part of the dispersal period. The dispersal routes in the Messick and Hornocker study required the female to cross the Snake River; the male crossed several drainages as well as the Snake River. Young of year dispersers will readily move across farmland and other disturbed sites (Messick & Hornocker 1981). Most elevations are suitable for badger movements. They are able to move through a wide variety of conditions but may avoid expanses of rock or ice/snow (T. Kinley, personal communication). They can travel from 30 to 50 km per day (T. Kinley, personal communication).

In eastern Washington, home range sizes were 9.2 km^2 for males and 5.7 km^2 for females (Paulson 2007). During the breeding season (June-August), males expand their home ranges. Goodrich and Buskirk (1998) recorded home ranges up to 12.4 km^2 in southeastern Wyoming. Home ranges of both male and female badgers frequently overlap; in one study, overlaps ranged from 62% to 98%, with a 1 km center-point shift (Paulson 2007). Home ranges averaging 69 km^2 were recorded in highly fragmented habitat in British Columbia (Newhouse & Kinley 2000). In southeastern British Columbia, maximum home range sizes of 19 km^2 for a female and 64 km^2 for a male were recorded (Kinley & Newhouse 2008). Home range size varies throughout badger distribution, and is correlated with prey density, female availability and habitat attributes (Hoodicoff & Larsen 2009).

Modeling Efforts

There are a number of suitable habitat models for the American badger. For example, Penrod et al. (2004) conducted a landscape permeability analysis incorporating weighted values for land cover, road density, topography, and elevation. They also conducted a patch size and configuration analysis based on habitat suitability models of vegetation, terrain, home range, and dispersal distance. Because the majority of methods used to document dispersal distance for most species underestimate the true value (LaHaye et al. 2001), Penrod et al. (2004) assumed that the American badger could disperse twice as far as the longest recorded distance of 110 km (Messick & Hornocker 1981). Apps and Newhouse (2000) identified soil parent material and texture, vegetation cover (i.e., forested versus non-forested), and topographical features as the variables that most affected where badgers occurred throughout the landscape. Diamond (2006) demonstrated a least-cost model that included soil types, vegetation, elevation, roads, and urban areas.

Parameters such as distance to water bodies and measured distances to roads/road density were not found in any models. Paulson (2007) evaluated these features using GIS, but his results were inconclusive due to the nature and small size of the study area. This study did, however, link badger movements with vegetation type and prey availability. Studies that primarily dealt with road issues addressed mortality rates based on traffic volume and speed, and mortality risk related to culverts and Jersey barriers (Case 1978; Kinley & Newhouse 2009).

Conceptual Basis for Model Development

OVERVIEW

American badgers are open habitat specialists that occupy shrub/grassland and occasionally open forest habitats, although they are capable of traversing a variety of habitats if necessary, as long as friable soils and suitable prey are present. Resistance values for landscape features were derived from descriptions in the literature of badger habitat and movements. In cases where little published information was available we relied upon the professional judgment of expert reviewers. Movement routes used by badgers are expected to be influenced by availability of rodent prey, land-cover type, and human disturbance (persecution and vehicle traffic). Factors impeding their movement throughout the landscape include vehicle traffic, urban land-use, and human population density**.**

MOVEMENT DISTANCE

Maximum recorded dispersal distance is 110 km, (Messick & Hornocker 1981); badgers are capable of traveling 30 to 50 km per day (T. Kinley, personal communication).

Landscape Features and Resistance Values

The resistance surface for the badger (Table A.5) is based on the following parameterized spatial data layers:

1) Land cover/land-use
2) Slope
3) Housing density
4) Road use type and distance
5) Forest structure

Habitat Concentration Areas

These are the steps we used to identify the American badger HCAs:

1) Develop a binary habitat surface where each grid cell in a raster is designated as either habitat or non-habitat. We defined badger habitat as any grid cell where resistance is 1.

2) Calculate the proportion of habitat within a circular moving window of a size representative of each species' home range radius. A home range radius of 3 km was used based on estimates found in badger movement studies. This step generates a surface which identifies the areas where the habitat is most concentrated.

3) Delete habitat grid cells in areas where habitat is sparse. We removed habitat grid cells from the binary habitat raster if the proportion of habitat within home range radius was <0.8. This prevented habitat concentrations from forming in areas where habitat is not concentrated to the degree required to be considered core habitat.

4) Join remaining habitat grid cells together if they are within a home range movement distance. We expanded the designated habitat area outwards (from the remaining habitat grid cells after step 3) up to a total cost-weighted distance equal to a badger's home range movement radius (3 km). This has the effect of joining nearby habitat grid cells together if the intervening landscape supports within-home range connectivity.

5) Eliminate small patches unlikely to contribute significantly to a species' core habitat. We eliminated aggregates of habitat less than 200 km^2 and retained the largest blocks of core habitat for HCAs. This 200 km^2 cutoff was chosen to represent the minimum area required to support core populations of the large carnivores (American badger and American black bear [*Ursus americanus*]) we chose as focal species.

Linkage Modeling

Linkages were modeled when the least-cost distance between a pair of HCAs was less than 301 km. This distance provided a best-fit model based on cost-weighted corridor maps and early core habitat modeling, as well as recorded Washington occurrence points (Johnson & Cassidy 1997; WDFW 2010).

Considerations for Future Modeling

Some significant badger habitat areas were smaller than the threshold used in the statewide model. Therefore, modeling at the ecoregional scale should: (1) adjust the minimum size requirement for badger HCAs, (2) include soils as a parameter for HCAs, (3) include spatial information for prey resources, and (4) differentiate between types of agriculture and allow for lower resistance where agricultural lands are in close proximity to shrubsteppe.

Table A.5. Landscape features and resistance values used to model habitat connectivity for American badgers.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	15
urban/developed	1000
water	40
sparsely vegetated	100
alpine	25
riparian	5
wetland	40
grass-dominated	0
shrub-dominated	0
dry forest	50
wet forest	100
Slope (degrees)	
0–20	0
> 20–40	10
> 40	25
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	0
> 20 ≤ 40	0
> 10 ≤ 20	25
< 10	35
Road type and distance (meters)*	
freeway > 500–1000 buffer	0
freeway > 0–500 buffer	0
freeway centerline	1000
major highway > 500–1000 buffer	0
major highway > 0–500 buffer	0
major highway centerline	200
secondary highway > 500–1000 buffer	0
secondary highway > 0–500 buffer	0
secondary highway centerline	35
local road > 500–1000 buffer	0
local road > 0–500 buffer	0
local road centerline	5
Forest structure (density and height [meters])	
nonforest	0
sparse low (0–40%, ≤ 25)	50
sparse high (0–40%, > 25)	50
open low (> 40–70%, ≤ 25)	100
open high (> 40–70%, > 25)	100
dense low (> 70–100%, ≤ 25)	100
dense high (> 70–100%, > 25)	100

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Acknowledgements

Brian Cosentino (WDFW), Andrew Shirk (Independent Researcher), Kelly McAllister (WSDOT), Peter Singleton (USFS), Brad McRae (TNC), and Albert Perez (WSDOT) provided assistance with model development. Brian Cosentino built the American badger resistance surface, and Andrew Shirk developed the American badger HCA methods. In addition, the following persons provided model review: Trevor Kinley (Sylvan Consulting Ltd.), Rich Weir (Artemis Consulting), Richard Finger (WDFW), and Jeff Bernatowicz (WDFW).

Literature Cited

Apps, C. D., and N. J. Newhouse. 2000. Habitat modeling for American badgers in the East Kootenay Region of British Columbia. Prepared for Columbia Basin Fish & Wildlife Comp. Program, Nelson, British Columbia, Crestbrook Forest Industries, Cranbrook, British Columbia, East Kootenay Environmental Society, Kimberley, British Columbia, Parks Canada, Radium Hot Springs, British Columbia.

Apps, C. D., N. J. Newhouse, and T. A. Kinley. 2002. Habitat associations of American badgers in southeastern British Columbia. Canadian Journal of Zoology 80:1228–1239.

Case, R. M. 1978. Interstate highway road-killed animals: a data source for biologists. Wildlife Society Bulletin 6:8–13.

Diamond, T. 2006. Identification of potential wildlife corridors utilized by the North American badger (*Taxidea taxus*) in the San Francisco Bay Area & Monterey County. Paper presented at: Sierra Azul Wildlife Connectivity Workshop, 2006 Oct 11, San Jose State University.

Eldridge, D. J. 2004. Mounds of the American badger (*Taxidea taxus*): significant features of North American shrub-steppe ecosystems. Journal of Mammalogy 85:1060–1067.

Goodrich, J. M., and S. W. Buskirk. 1998. Spacing and ecology of North American badgers (*Taxidea taxus*) in a prairie dog (*Cynomys leucurus*) complex. Journal of Mammalogy 79:171–179.

Hoodicoff, S. C., and K. W. Larsen. 2009. Home range size and attributes for badgers (*Taxidea taxus jeffersonii*) in south-central British Columbia, Canada. American Midland Naturalist 162:305–317.

Johnson, R. E., and K. M. Cassidy. 1997. Terrestrial mammals of Washington State: location data and predicted distributions. Vol. 3 in Washington State Gap Analysis – Final Report. K. M. Cassidy, C. E. Grue, M. R. Smith, and K. M. Dvornich, editors. Washington Cooperative Fish and Wildlife Research Unit, University of Washington, Seattle, Washington.

Kinley, T. A., and N. J. Newhouse. 2008. Ecology and translocation-aided recovery of an endangered badger population. Journal of Wildlife Management 72:113–122.

Kinley, T. A., and N. J. Newhouse. 2009. Badger roadkill risk in relation to the presence of culverts and jersey barriers. Northwest Science 83:148–153.

LaHaye, W. S., R. J. Gutierrez, and J. R. Dunk. 2001. Natal dispersal of the spotted owl in southern California: dispersal profile of an insular population. Condor 103:691–700.

Lindzey, F. G. 1976. Characteristics of the natal den of a badger. Northwest Science 50:178–180.

Lindzey, F. G. 1978. Movement patterns of badgers in northwestern Utah. Journal of Wildlife Management 42:418–422.

Lindzey, F. G. 1982. The North American badger. Pages 653–663 in J. A. Chapman and G. A. Feldhamer, editors. Wild mammals of North America: biology, management and economics. John Hopkins University Press, Baltimore, Maryland.

Messick, J. P., and M. G. Hornocker. 1981. Ecology of the badger in southwestern Idaho. Wildlife Monographs N-76.

Newhouse, N. J., and T. A. Kinley. 2000. Ecology of American badgers near their range limit in southeastern British Columbia. Columbia Basin Fish and Wildlife Compensation Program, Nelson, British Columbia, Crestbrook Forest Industries, Cranbrook, British Columbia, East Kootenay Environmental Society, Kimberley, British Columbia, and Parks Canada, Radium Hot Springs, British Columbia.

Paulson, N. J. 2007. Spatial and habitat ecology of North American badgers (*Taxidea taxus*) in a native shrub-steppe ecosystem of eastern Washington. Master's thesis. University of Washington, Seattle, Washington.

Penrod, K., C. Cabañero, P. Beier, C. Luke, W. Spencer, , W. Rubin, S. Loe, and K. Meyer. 2004. South Coast missing linkages project: a linkage design for the San Gabriel-San Bernardino connection. South Coast Wildlands, Idyllwild, Californaia.

WDFW (Washington Department of Fish and Wildlife). 2010. Wildlife Surveys and Data Management Database. Washington Department of Fish and Wildlife, Olympia, Washington.

Personal Communications

J. Bernatowicz, Wildlife Biologist
Washington Department of Fish and Wildlife
Olympia, Washington

R. Finger, Wildlife Biologist
Washington Department of Fish and Wildlife
Olympia, Washington
WDFW

T. A. Kinley
Sylvan Consulting Ltd.
British Columbia, Canada

R. Weir
Artemis Wildlife Consultants
British Columbia, Canada

Black-tailed Jackrabbit (*Lepus californicus*)

Prepared by Howard L. Ferguson, Karin A. Divens, and Michael Atamian (Washington Department of Fish and Wildlife)

Justification for Selection

The black-tailed jackrabbit (*Lepus californicus*) was selected as a focal species because it is a good representative of wildlife habitat and connectivity needs within the Semi-desert vegetation class. Black-tailed jackrabbits are closely associated with this habitat in Washington and throughout their range. Black-tailed jackrabbits are vulnerable to loss of habitat connectivity from all four major connectivity threats: clearing and vegetation removal, development, roads and traffic, and the presence of people and domestic animals. They are at considerable risk for increased mortality from roads, persecution from farmers, and harassment by pets. Population monitoring is a challenge as no reliable census method exists. Individual monitoring, however, is fairly easy when using radio telemetry. Their movement scale is appropriate for a statewide modeling effort based on recorded dispersal distances. Black-tailed jackrabbits are listed as a Species of Greatest Conservation Need in Washington due to habitat loss and human-related threats.

Distribution

The black-tailed jackrabbit's historical range encompasses an area from the Pacific Ocean on the west to Arkansas and Missouri on the east. In the north it ranges from southern Washington to South Dakota and in the south it is found throughout Baja California and well into south-central Mexico (Chapman & Flux 1990). This species has been successfully introduced into various states in the eastern U.S. including: Massachusetts, Maryland, New Jersey, Virginia and Florida. In Washington, it is an eastside species, inhabiting the Columbia Plateau of central Washington, and extending south into Oregon. There are some indications that black-tailed jackrabbits are recent colonizers of Washington State (Couch 1927; Larrison 1976). Larrison (1976) mentions, "…in the years preceding their sweep into Eastern Oregon and Washington…"

Habitat Associations

Black-tailed jackrabbits are found in the arid Columbia Plateau region of Washington. Compared to white-tailed jackrabbits (*Lepus townsendii*), black-tailed jackrabbits prefer areas with more shrub growth, less abundant grasses, and often, where grazing has been severe (Best 1996).They do not readily move into areas of tall grass, where visibility is obscured. Associated plant species include: *Artemisia tridentata*, *Agropyron spicatum*, *Chrysothamnus viscidiflorus*, *Sitanion hystrix*, *Orzyopsis hymenoides*, and *Atriplex* sp. (French et al. 1965). Washington Gap analysis (Johnson & Cassidy 1997) noted that black-tailed jackrabbits are usually in sagebrush and sometimes rabbitbrush (*Chrysothamnus* sp.), including areas of mixed grass and sagebrush and/or rabbitbrush, but generally not in grassland lacking shrub cover. They can also be found in and around inland sand dunes (Hallock et al. 2007). In Idaho, they are primarily associated with shrub cover dominated by *A. tridentata*. Their diet varies seasonally; they eat a higher percentage of shrubs in winter, forbs in spring, and mostly grasses with almost no shrub ingestion in summer (Grant 1987). They are known to also use juniper (*Juniperus osteosperma*), shadscale (*Atriplex confertifolia*) and greasewood (*Sarcobatus vermiculatus*) for feeding and cover (Gross et al.

1974). Thompson and Gese (2007) found that presence of lagomorphs was positively correlated with shrub density ($r^2 = 0.33$, $p > 0.006$).

Dispersal

Size of home range varies from 20–300 ha (Lechleitner 1958; Harestad & Bunnell 1979; Smith 1990). In one study in Idaho, black-tailed jackrabbits seldom moved >0.4 km (home range of < 16.2 ha; French et al. 1965). In southeastern Idaho, polygon sizes of nightly movements ranged from 4.7 to 17.2 ha and home ranges varied from 11.5 to 57.6 ha (Grant 1987). In California, Lechleitner (1958) reported that black-tailed jackrabbits tended to remain within relatively well defined areas of 17.5 ha. Chapman & Flux (1990) summarized data from California and found that home ranges averaged 20 ha; female black-tailed jackrabbits generally have larger home range sizes than males. From a telemetry study in Utah, Smith (1990) reported black-tailed jackrabbit home ranges up to 300 ha (150–300 ha) and noted that the shape of most home ranges tended to be elliptical. A similar elliptical shape was noted by Rusch (1965) and Donoho (1971). Donoho (1971) reported an average home range size of 258 ha in Colorado. Amount of daily movement varies; black-tailed jackrabbits have been recorded moving up to 16 km round trip to feed in an alfalfa field (Best 1996). Grant (1987) recorded a monthly movement of 10 km from April 8 to May 6. Smith et al. (2002) found that more than 43% of seasonal movements that were ≥5 km were between 5 and 7 km while 21% were between 8 and 10 km. These movements most commonly occurred between March and April, and October and December, indicating that the movements were likely migrations to and from wintering habitat areas.

Literature suggests that no regular seasonal migration occurs, however, most large movements appear to be between the fall and winter ranges and also winter and spring ranges (Rusch 1965; Grant 1987; Smith et al. 2002). Lechleitner (1958) reported a maximum dispersal distance of 1.6 km (Table A.6). Porth (1995) observed long-distance movements by both male and female black-tailed jackrabbits in all seasons, but found that movements were more common in fall (39% versus 28% in summer and 17% in winter). Porth (1995) reported an average movement distance of 12 km along the Utah-Idaho border. Smith et al. (2002) monitored radio-marked black-tailed jackrabbits and reported a maximum dispersal distance of 35 km during a severe winter; 35% of 393 jackrabbits moved ≥5 km at least once during their lifetimes. In Idaho, dispersal movements occurred in approximately 18% of the population with the greatest distance being a 45 km movement in a 17-week period (French et al. 1965). Grant (1987) reported a maximum movement of 57.3 km occurring during early winter, with nightly movement polygons from 2.1 to 26.4 ha (average 10.8 ha) and a home range average of 31.0 ha (SD 12.4). In this study, distances traveled averaged 16.2 km with a range of 2.2 to 57.3 km. From a very early observation in Washington, Couch (1927) reported that this species expanded its range 64 km from 1908–1912, colonizing the area from western Walla Walla County up to Grant County.

Table A.6. Maximum dispersal distance reported for black-tailed jackrabbits.

Maximum dispersal distance (km)	*Citation*
1.6	Lechleitner (1958)
12.0	Porth (1995)
45.0	French et al. (1965)
57.3	Grant (1987)

Modeling Efforts

Washington Gap (Johnson & Cassidy 1997) modeled black-tailed jackrabbits in all steppe zones in the Columbia Plateau with the most arid zones (areas with *Agropyron* sp. or *Artemisia tridentata* as climax species) as their core habitats and other steppe zones within its range limits as peripheral habitats. All non-forested steppe habitats were good. All others were excluded.

Conceptual Basis for Model Development

OVERVIEW

Black-tailed jackrabbits are found in the arid Columbia Plateau and prefer shrub dominated habitat for both browse and cover. They do not utilize areas with tall grass cover. In Washington, black-tailed jackrabbits typically inhabit areas of relatively flat topography but may use slopes that exceed 40%. Although slope is a component of suitable jackrabbit habitat it is not likely to be a factor impeding movement or dispersal. Black-tailed jackrabbits are at risk from the conversion of native shrubsteppe and grassland habitats for development. Black-tailed jackrabbits are known to use undeveloped roads for movement between habitat areas and will cross all roads, including major highways if encountered. However, black-tailed jackrabbits are at risk from roads due to mortality associated with vehicles; road size and traffic volume are key factors influencing risk associated with roads. It is assumed that local roads and secondary highways have a minimal impact, while freeways and major highways pose a higher resistance to black-tailed jackrabbit movement. Black-tailed jackrabbits select habitat at multiple scales. Ecological and human caused disturbances at one scale can affect others. Our model of habitat connectivity takes a broad-scale approach. We examine connectivity among (sub) populations of black-tailed jackrabbits at the statewide scale.

Landscape Features and Resistance Values

The resistance surface is based on the following parameterized spatial data layers:

1) Land cover/land-use

2) Elevation

3) Slope

4) Housing density

5) Road type and distance

Values for these parameters are provided in Table A.7. The resistance value for each 100 m by 100 m cell is the sum of layer cell values.

Habitat Concentration Areas

General Definition — Habitat concentrations areas (HCAs) are core habitat areas that are expected or known to be important habitat for focal species. These areas are used as the focus locations for running dispersal models that identify connectivity pathways.

Technical Definition — Habitat concentration areas are aggregations of habitat grid cells that are connected to each other by a species-specific home range diameter or short range dispersal distance. These aggregations must also meet a minimum size requirement designed to select only those areas that are capable of supporting multiple individuals. Connectivity between HCAs will be evaluated based on a species-specific dispersal distance and a landscape resistance model.

Due to a lack of sufficient studies and occurrence data for black-tailed jackrabbits, the core habitat in Washington was not well defined. Many of the field observations are made from vehicles and thus clustered around roads. Because of these shortcomings, models were developed to define the core habitat areas. Jackrabbits are corridor dwellers and have a relatively small home range movement radius compared to some of the other focal species used in the statewide connectivity analysis. This factor made it difficult to model jackrabbit HCAs on a statewide scale. A movement radius of only 1.1 km, led to a model output of many small patches of core habitat areas. Consequently, for coarse scale modeling purposes, the home range movement radius was increased to 2 km. This process merged many small HCAs into larger HCAs, more appropriate for the statewide scale. The modeling process for jackrabbit HCA identification involves the following steps:

1) Develop a binary habitat surface where each grid cell in a raster is designated as either habitat or non-habitat. A resistance threshold of 1 was chosen from the species resistance surface. Cells with a value of 1 were designated habitat while cells >1 were designated non-habitat.

2) Calculate the proportion of habitat surrounding each grid cell using a circular moving window of a size representative of each species' home range. A home range radius of 2 km was used for black-tailed jackrabbits. This step generates a habitat density surface with each grid cell representing the proportion of habitat around it.

3) Limit HCA formation to those areas where the largest concentrations of habitat exist. A habitat density cutoff of 75% was applied to black-tailed jackrabbits. Those habitat grid cells with lower habitat density than the cutoff value were not used in HCA formation.

4) Create habitat areas by joining the remaining (after step 3) habitat density grid cells together if they were within a home range movement distance. Habitat areas were expanded outwards up to a total cost-weighted distance equal to the species' home range movement radius (2 km). This had the effect of joining nearby cells together if the intervening landscape supported within-home range connectivity.

5) Eliminate small habitat area patches unlikely to contribute significantly to a species' core habitat. Habitat patches less than 50 km^2 were eliminated because they were unlikely to support a viable population of jackrabbits. The remaining habitat areas became the black-tailed jackrabbit HCAs.

The modeling process resulted in HCAs occurring outside of the historical range of the black-tailed jackrabbit, specifically in the Okanogan Highlands. These HCAs were retained because these areas do possess suitable habitat, and in sufficient quantities, to support black-tailed jackrabbit populations. Given that the range of black-tailed jackrabbits in Washington is expanding, and that there have not been any recent extensive surveys specific to jackrabbits completed in the Okanogan Highlands, we decided to keep these HCAs in the statewide analysis.

Linkage Modeling

The black-tailed jackrabbit is considered to be a linkage dweller. If suitable habitat is available between defined HCAs, jackrabbits may utilize it for feeding, cover, movement and reproduction. For this reason, no maximum dispersal distance was assigned to jackrabbits. Some nearby HCAs were combined for the linkage modeling. All defined HCAs that were completely contained within Washington State were connected during linkage modeling. This resulted in a total of 75 corridors linking the 31 black-tailed jackrabbit HCAs identified with the modeling efforts.

Considerations for Future Modeling

There is a general lack of studies and occurrence data for black-tailed jackrabbits in Washington. This factor, along with the fact that black-tailed jackrabbits have a relatively small home-range movement radius, made them challenging to model on a statewide scale. Future modeling at the eco-regional scale should: (1) analyze the resistance surfaces applied to the model and ensure that species appropriate resistance surfaces are used; (2) collect more location data and population information to improve delineation of HCAs; (3) apply a more species appropriate home range movement relative to the habitat being utilized; (4) use finer scale vegetation layers, and (5) possibly incorporate other factors, such as precipitation or soils layers, to better define species range.

Table A.7. Landscape features and resistance values used to model habitat connectivity for black-tailed jackrabbits.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	2
urban/developed	1000
water	50
sparsely vegetated	1
alpine	100
riparian	2
wetland	5
grass-dominated	5
shrub-dominated	0
dry forest	50
wet forest	500
Elevation (meters)	
0–250	0
> 250–500	0
> 500–750	0
> 750–1000	3
> 1000–1500	5
> 1500–2000	10
> 2000–2500	25
> 2500–3300	50
> 3300 meters	500
Slope (degrees)	
0–20	0
> 20–40	0
> 40	10
Housing density (acres per dwelling unit)	
> 80	0
> 40 < 80	0
> 20 < 40	2
> 10 < 20	10
< 10	25
Road type and distance (meters)*	
freeway > 500–1000 buffer	0
freeway > 0–500 buffer	1
freeway centerline	250
major highway > 500–1000 buffer	0
major highway > 0–500 buffer	0
major highway centerline	50
secondary highway > 500–1000 buffer	0
secondary highway > 0–500 buffer	0
secondary highway centerline	10
local road > 500–1000 buffer	0
local road > 0–500 buffer	0
local road centerline	5

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Acknowledgements

Brian Cosentino (WDFW), Andrew Shirk (Independent Researcher), Kelly McAllister (WSDOT), Joanne Schuett-Hames (WDFW), Peter Singleton (USFS), Brad McRae (TNC), and Albert Perez (WSDOT) provided assistance with model development. Mike Livingston (WDFW), Mike Schroeder (WDFW), Derek Stinson (WDFW), and Jeff Bernatowicz (WDFW) provided model review and technical input.

Literature Cited

Best, T. L. 1996. *Lepus californicus*. Mammalian Species 530:1–10.

Chapman, J. A., and J. E. C. Flux. 1990. Rabbits, Hares and Pikas. Status Survey and Conservation Action Plan. IUCN, Gland, Switzerland.

Couch, L. K. 1927. Migrations of the Washington black-tailed jackrabbit. Journal of Mammalogy 8:313–314.

Donoho, H. S. 1971. Dispersion and Dispersal of white-tailed and black-tailed jackrabbits, Pawnee National Grasslands. U.S. International Biological Program.

French, N. R., R. McBride, and J. Detmer. 1965. Fertility and population density of the black-tailed jackrabbit. Journal of Wildlife Management 29:14–26.

Grant, J. C. 1987. Ecology of the black-tailed jackrabbit near a solid radioactive waste disposal site in southeastern Idaho. Master’s thesis. University of Montana, Missoula, Montana.

Gross, J. E., L. Charles Stoddart, and Frederic H. Wagner. 1974. Demographic analysis of a northern Utah jackrabbit population. Wildlife Monographs N-40, The Wildlife Society.

Hallock, L. A., R. D. Haugo, and R. Crawford. 2007. Conservation Strategy for Washington State Inland Sand Dunes. WA State Department of Natural Resources, Natural Heritage Report 2007-05. Prepared for BLM, Spokane, WA.

Harestad, A. S., and F. L. Bunnell. 1979. Home range and body weight – a reevaluation. Ecology 60:389-402.

Johnson, R. E., and K. M. Cassidy. 1997. Terrestrial mammals of Washington State: location data and predicted distributions. Vol. 3 in Washington State Gap Analysis – Final Report, K. M. Cassidy, C. E. Grue, M. R. Smith and K. M. Dvornich, editors. Washington Cooperative Fish and Wildlife Research Unit, University of Washington, Seattle.

Larrison, E. J. 1976. Mammals of the Northwest. Seattle Audubon Society.

Lechleitner, R. R. 1958. Movements, density, and mortality in a black-tailed jackrabbit population. Journal of Wildlife Management 22:371–384.

Porth, A. T. 1995. Movements of black-tailed jackrabbits (*Lepus californicus*) and effects of high population densities on the nitrogen budget of sage-brush steppe. Master's thesis, Idaho State University, Pocatello, Idaho.

Rusch, D. H. 1965. Some movements of black-tailed jackrabbits in northern Utah. Master's thesis, Utah State University, Logan, Utah.

Smith, G. W. 1990. Home range and activity patterns of black-tailed jackrabbits. Great Basin Naturalist 50:249–256.

Smith, G. W., L. C. Stoddart, and F. F. Knowlton. 2002. Long-distance movements of black-tailed jackrabbits. Journal of Wildlife Management 66:463–469.

Thompson, C. M., and E. M. Gese. 2007. Food webs and intra-guild predation: community interactions of a native mesocarnivore. Ecology 88:334–346.

White-tailed Jackrabbit (*Lepus townsendii*)

Prepared by Howard L. Ferguson, Karin A. Divens, and Michael Atamian (Washington Department of Fish and Wildlife)

Justification for Selection

The white-tailed jackrabbit was selected as a focal species because it is a good representative of wildlife habitat and connectivity needs within the Semi-desert vegetation class. White-tailed jackrabbits are closely associated with this habitat in Washington and throughout their range with the exception of occupying ecologically equivalent alpine regions in Colorado (Braun & Streeter 1968). White-tailed jackrabbits are vulnerable to loss of habitat connectivity from all four major connectivity threats: clearing and vegetation removal, development, roads and traffic, and the presence of people and domestic animals. They are at considerable risk for increased mortality from roads, persecution, and harassment by pets. Population monitoring is a challenge as no reliable census method exists for all population levels. Individual monitoring, however, is fairly easy when using radio telemetry. Their movement scale is appropriate for a statewide modeling effort based on recorded dispersal distances. White-tailed jackrabbits are listed as a Species of Greatest Conservation Need in Washington due to habitat loss and human-related threats.

Distribution

The white-tailed jackrabbit is distributed in west-central North America from the prairies of southern Saskatchewan and Alberta to the Rocky Mountains of northern New Mexico, and from Lake Michigan in Wisconsin to east of the Cascade Mountains of Washington and the Sierra Nevada Mountains of California (Wilson & Ruff 1999). Historically, the distribution of this species spread as far east as Wisconsin, Iowa and Missouri in areas of cleared forest. However, when its prairie habitat became more and more cultivated, the range of the species contracted, and it became extinct in Kansas and southern Nebraska (Chapman & Flux 1990). In Washington, it is an eastside species, with distribution throughout the Columbia Plateau shrubsteppe habitat from British Columbia to Oregon. However, there is concern for this species in Washington because of their shrinking range which may be a result of overgrazing combined with the more aggressive and/or efficient nature of the black-tailed jackrabbit. Larrison in 1976 expressed his concern for the decline of the white-tailed jackrabbit when he wrote, “The white-tailed jackrabbit was once common in the sagebrush-bunch grass habitat of Central and Eastern Washington, but at the present time it is restricted and scarce in that state except in the Okanogan Valley… the splendid large white-tailed jackrabbit may well disappear from Washington, and even large parts of the remainder of the Northwest.”

Habitat Associations

White-tailed jackrabbits feed primarily on succulent grasses and forbs, eating some shrubs in winter if they are the only food item available. They tend to consume grasses and forbs that are young and have not yet produced seed. When feeding on shrubs, woody stems are eaten more frequently than leaves or young shoots (Lim 1987). They are known to feed on alfalfa when available. In Colorado, diet items in summer consisted of 70% forbs, 19% grasses, and 7% shrubs. The four most commonly ingested plants were clover (*Trifolium* sp.), dandelion

(*Taraxacum officinale*), dryland sedge (*Carex obtusata*), and Indian paintbrush (*Castilleja integra*). In autumn, white-tailed jackrabbit diet consisted of 43% grasses, 34% forbs, and 14% shrubs. In winter, diet items consisted of 76% shrubs and 12% forbs (Bear & Hansen 1966). Washington Gap (Johnson & Cassidy 1997) reported that white-tailed jackrabbits inhabit areas with bunchgrass and rabbitbrush and are less frequently found in open, low sagebrush. They noted that they are usually in more open habitat than black-tailed jackrabbits. Throughout the Columbia Basin habitat quality for white-tailed jackrabbits has declined as livestock grazing has resulted in replacement of bunchgrass with sagebrush. In Washington, the white-tailed jackrabbit tends to occur at higher elevations (on grassy hills and plateaus) than the black-tailed jackrabbit, and its distribution extends farther north (up the Okanogan drainage into British Columbia) than the black-tailed jackrabbit.

In some of its historical range, where cultivation, drought or overgrazing has affected the habitat, white-tailed jackrabbits have been replaced by black-tailed jackrabbits (Armstrong 1972). Where sympatric, black-tailed jackrabbits are possibly more efficient than white-tailed jackrabbits in foraging and may possess a competitive advantage (Flinders & Hansen 1972). In these conditions, the two species of jackrabbits use different habitats; black-tailed Jackrabbits occur primarily in sagebrush with open grass, while white-tailed jackrabbits are most common in bunchgrass habitats with less open cover (Anthony 1913; Couch 1927). In eastern Washington, Dalquest (1948) found white-tailed jackrabbits on arid, hilly bunchgrass sites during the summer and in lower sagebrush valleys during winter. Dalquest (1948) also noted the gradual disappearance of bunchgrass as the result of overgrazing speculating that this caused the reduction of numbers and confinement of the white-tailed jackrabbit to arid grasslands in the Okanogan. Both species of jackrabbits may frequent agricultural land where they can become a pest of crops and fruit trees (Lechleitner 1958).

Dispersal

The home range of the white-tailed jackrabbit has been reported as 2 to 3 km in diameter (Seton 1928; Jackson 1961) however, research on movement and other aspects of this species is scant. Donoho (1971) reported that white-tailed jackrabbits had an average home range size of 258 ha (2.58 km^2) in Colorado.

Modeling Efforts

Washington Gap (Johnson & Cassidy 1997) modeled white-tailed jackrabbits in all steppe zones within its current range limits within the Columbia Basin. Steppe zones were considered core habitat, except for Central Arid Steppe, which was considered peripheral. In steppe zones, non-forested habitats were included; all others were excluded. Peripheral habitat included Ponderosa and Oak zones at the edges of the steppe. In these two forested zones good habitat included sparsely vegetated cover, grassland, and tree savannah; all others were excluded.

Conceptual Basis for Model Development

OVERVIEW

White-tailed jackrabbits prefer grasses and forbs for browse. In Washington, this species inhabits areas with bunchgrasses and rabbitbrush and are less frequently found in open, low sagebrush (Johnson & Cassidy 1997). White-tailed jackrabbits seasonally inhabit higher elevation habitat (grassy hills and plateaus) and their distribution extends farther north than black-tailed jackrabbits. In Washington, white-tailed jackrabbits commonly inhabit areas of relatively flat topography but may use areas with slopes that exceed 40%. Although slope is a component of suitable white-tailed jackrabbit habitat, it is not likely to be a factor impeding movement or dispersal. White-tailed jackrabbits are at risk from the conversion of native shrubsteppe and grassland habitats for development; they are especially sensitive to a housing density greater than 20 acres/dwelling unit. White-tailed jackrabbits are known to use undeveloped roads for movement between habitat areas and will cross all roads, including major highways if encountered. However, white-tailed jackrabbits are at risk from roads due to mortality associated with vehicles; road size and traffic volume are key factors influencing risk associated with roads. It is assumed that local roads and secondary highways have a minimal impact while freeways and major highways pose higher resistance to white-tailed jackrabbit movement. White-tailed jackrabbits select habitat at multiple scales. Ecological and human caused disturbances at one scale can affect others. Our model of habitat connectivity takes a broad-scale approach. We examine connectivity among (sub) populations of white-tailed jackrabbits at the state-wide scale.

Landscape Features and Resistance Values

The resistance surface is based on the following parameterized spatial data layers:

1) Land cover/land-use
2) Elevation
3) Slope
4) Housing density
5) Roads type and distance

Values for these parameters are provided in Table A.8. The resistance value for each 100 m by 100 m cell is the sum of layer cell values.

Habitat Concentration Areas

General Definition — Habitat concentration areas (HCAs) are core habitat areas that are expected or known to be important habitat for focal species. These areas are used as the focus locations for running dispersal models that identify connectivity pathways.

Technical Definition — Habitat concentration areas are aggregations of habitat grid cells that are connected to each other by a species-specific home range diameter or short range dispersal distance. These aggregations must also meet a minimum size requirement designed to select only those areas that are capable of supporting multiple individuals. Connectivity between HCAs will be evaluated based on a species-specific dispersal distance and a landscape resistance model.

Due to a lack of sufficient studies and occurrence data for white-tailed jackrabbits, their core habitat in Washington is not well defined. Many of the field observations are from vehicles and thus clustered around roads. Models were developed to define the core habitat areas for the statewide analysis. White-tailed jackrabbits are corridor dwellers and have a relatively small home range movement radius compared to some of the other focal species used in the statewide connectivity analysis. This factor made it more challenging to model white-tailed jackrabbit HCAs on a statewide scale. A movement radius of 1.1 km, led to a model output of many small patches of core habitat areas. Consequently, for coarse scale modeling purposes, the home range movement radius was increased to 2 km. This process merged many small HCAs into larger HCAs, more appropriate for the statewide scale.

These are the steps that we used to identify the white-tailed jackrabbit HCAs:

1) Develop a binary habitat surface where each grid cell in a raster is designated as either habitat or non-habitat. Habitat is defined as any grid cell with a resistance value threshold of 1.

2) Calculate the proportion of habitat surrounding each cell using a circular moving window of a size representative of each species' home range. A home range radius of 2 km was used for the white-tailed jackrabbit. This step generates a habitat density surface with each cell representing the proportion of habitat around it.

3) Limit HCA formation to those areas where the largest concentrations of habitat exist. A habitat density cutoff of 85% was applied to white-tailed jackrabbits. Those grid cells with lower habitat density than the cutoff value were not used in HCA formation.

4) Create habitat areas by joining the remaining (from the remaining habitat cells after step 3) habitat density cells together if they were within the 2 km home range movement distance. Habitat areas were expanded outwards up to a total cost-weighted distance equal to the species' home range movement radius (2 km). This had the effect of joining nearby cells together if the intervening landscape supported within-home range connectivity.

5) Eliminate small habitat area patches unlikely to contribute significantly to a species' core habitat. Habitat patches less than 50 km^2 were eliminated because they were unlikely to support a viable population of jackrabbits. The remaining habitat areas became the white-tailed jackrabbit HCAs.

Table A.8. Landscape features and resistance values used to model habitat connectivity for white-tailed jackrabbits.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	2
urban/developed	1000
water	50
sparsely vegetated	2
alpine	75
riparian	2
wetland	5
grass-dominated	0
shrub-dominated	0
dry forest	50
wet forest	500
Elevation (meters)	
0–250	3
> 250–500	0
> 500–750	0
> 750–1000	1
> 1000–1500	1
> 1500–2000	7
> 2000–2500	25
> 2500–3300	50
> 3300 meters	500
Slope (degrees)	
0–20	0
> 20–40	0
> 40	10
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	0
> 20 ≤ 40	5
> 10 ≤ 20	25
< 10	100
Road type and distance (meters)*	
freeway > 500–1000 buffer	0
freeway > 0–500 buffer	1
freeway centerline	250
major highway > 500–1000 buffer	0
major highway > 0–500 buffer	0
major highway centerline	50
secondary highway > 500–1000 buffer	0
secondary highway > 0–500 buffer	0
secondary highway centerline	10
local road > 500–1000 buffer	0
local road > 0–500 buffer	0
local road centerline	5

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Linkage Modeling

The white-tailed jackrabbit is considered to be a linkage dweller. They are a species that disperses between HCAs over the course of multiple generations by living and reproducing within linkage zones.

If good habitat is available between defined HCAs, jackrabbits may utilize it for home range movement, feeding, cover, movement, and reproduction. For this reason no maximum dispersal distance was assigned to jackrabbits. During linkage modeling, some nearby HCAs were combined. During linkage modeling exercise, all HCAs modeled wholly within Washington were connected.

Considerations for Future Modeling

There is a general lack of studies and occurrence data for white-tailed jackrabbits in Washington. This, along with the fact that white-tailed jackrabbits have a relatively small home-range movement radius, made them challenging to model on a statewide scale. Future modeling at the eco-regional scale should: (1) analyze the resistance surfaces applied to the model and ensure that species appropriate resistance surfaces are used; (2) collect more location data and population information to improve delineation of HCAs; (3) apply a more species appropriate home range movement relative to the habitat being utilized; (4) use finer scale vegetation layers; and (5) possibly incorporate other factors, such as precipitation or soils layers, to better define species range.

Acknowledgements

Brian Cosentino (WDFW), Andrew Shirk (Independent Researcher), Kelly McAllister (WSDOT), Joanne Schuett-Hames (WDFW), Peter Singleton (USFS), Brad McRae (TNC), and Albert Perez (WSDOT) provided assistance with model development. Mike Livingston (WDFW), Mike Schroeder (WDFW), Derek Stinson (WDFW), and Jeff Bernatowicz (WDFW) provided model review and technical input.

Literature Cited

Anthony, H. E. 1913. Mammals of northern Malheur County, Oregon. Bulletin of the American Museum of Natural History 32:1–27.

Armstrong, D. M. 1972. Distribution of mammals in Colorado. Monograph, University of Kansas Museum of Natural History 3:1–415.

Bear, G. D., and R. M. Hansen 1966. Food habits, growth and reproduction of white-tailed jackrabbits in southern Colorado. Technical Bulletin Agricultural Experimental Station, Colorado State University 90:1–59.

Braun, C. E., and R. G. Streeter. 1968. Observations on the occurrence of white-tailed jackrabbits in the alpine zone. Journal of Mammalogy 49:160–161.

Chapman, J. A., and J. E. C. Flux. 1990. Rabbits, hares and pikas. Status survey and conservation action plan. IUCN, Gland, Switzerland.

Couch, L. K. 1927. Migrations of the Washington black-tailed jackrabbit. Journal of Mammalogy 8:313–314.

Dalquest, W. W. 1948. Mammals of Washington. University of Kansas Publication, Museum of Natural History 2:1–444.

Donoho, H. S. 1971. Dispersion and dispersal of white-tailed and black-tailed jackrabbits, Pawnee National Grasslands. U.S. International Biological Program.

Flinders, J. T., and R. M. Hansen. 1972. Diets and habitats of jackrabbits in northeastern Colorado (Science Series 12). Range Science Department, Colorado State University, Colorado.

Jackson , H. H. T. 1961. Mammals of Wisconsin. University of Wisconsin Press, Madison.

Johnson, R. E., and K. M. Cassidy. 1997. Terrestrial mammals of Washington State: location data and predicted distributions. Vol. 3 in K. M. Cassidy, C. E. Grue, M. R. Smith, and K. M. Dvornich, editors. Washington State Gap Analysis – Final Report, Washington Cooperative Fish and Wildlife Research Unit, University of Washington, Seattle.

Larrison, E. J. 1976. Mammals of the Northwest. Seattle Audubon Society.

Lechleitner, R. R. 1958. Certain aspects of behavior of the black-tailed jack rabbit. American Midland Naturalist 60:145–155.

Lim, B. K. 1987. *Lepus townsendii*. Mammalian Species 288:1–6.

Seton, E. T. 1928. Lives of game animals. Doubleday, Doran and Co., Inc., Garden City, New York.

Wilson, D. E., and S. Ruff, editors. 1999. The Smithsonian Book of North American Mammals. Smithsonian Institution Press, Washington D.C.

Mule Deer (*Odocoileus hemionus*)

Prepared by Kelly R. McAllister (Washington Department of Transportation)

Justification for Selection

Mule deer (*Odocoileus hemionus*) are important members of the wildlife community, serving a number of key ecological functions as herbivores and prey for large carnivores such as cougars (*Felis concolor*) and wolves (*Canis lupus*). Some local populations are migratory, exploiting productive mountain meadow habitat in summer but retreating to low elevation valleys in winter. As such, migratory mule deer often move long distances on a seasonal basis.

Mule deer were selected as good representatives of connectivity needs in the Semi-desert and Northern Rocky Mountain Forest vegetation classes. They were considered to be detrimentally affected by development, roads and traffic, and the presence of people and domestic animals. Mule deer can be dispersal-limited. Considerable mule deer research has produced adequate information for developing a movement model. Lastly, it's a species that can be monitored to evaluate efforts to maintain or improve habitat connectivity.

Distribution

Mule deer are found throughout much of western North America, extending as far east as Nebraska, Kansas, and western Texas. In Washington, mule deer are found throughout the state, except for the larger agricultural blocks of the Columbia Plateau that lack adequate forest or shrub cover (Johnson & Cassidy 1997).

Habitat Associations

Mule deer require a mosaic of habitat types of different age classes to meet their life history requirements. They use forest, woodland, brush, and meadow habitats, reaching their highest densities in open pine forests, riparian strips within arid and agricultural lands, and along edges of meadows and grasslands. They also occur in open scrub, young chaparral and low elevation coniferous forests. A variety of brush cover and tree thickets interspersed with meadows and shrubby areas are important for food and cover. Thick cover can provide escape from predators, shade in the summer, or shelter from wind, rain, and snow. Varying slopes and topographic relief are important for providing shade. Fawning occurs in moderately dense shrub, forest, riparian or meadow edge cover. Meadows are particularly important as fawning habitat.

Dispersal

Home ranges typically comprise a mosaic of habitat types that provide deer with various life history requirements. Home range estimates vary from 39 ha to 3379 ha. Harestad and Bunnell (1979) calculated mean home range from several studies as 285 ha. Doe and fawn groups have smaller home ranges, averaging 100–300 ha, but can vary from 50 to 500 ha. Bucks usually have larger home ranges and are known to wander greater distances. A recent study of 5 different sites throughout California recorded home range sizes from 49 to 1138 ha (Kie et al. 2002).

Where deer are seasonally nomadic, winter and summer home ranges tend to largely overlap in consecutive years. Elevational migrations are observed in mountainous regions in response to extreme weather events in winter, or to seek shade and perennial water during summer. Distances travelled between winter and summer ranges vary from 8.6 to 29.8 km. Robinette (1966) observed natal dispersal distances ranging from 97 to 217 km (Table A.9).

Table A.9. Long-range mule deer movements or dispersal distances.

Description	*Distance*	*Citation*
Max juvenile dispersal	217 km	Robinette 1966
Max seasonal range migration	115 km	Thomas & Irby 1990
Home range maximum	17,090 ha	Myers 2001
Typical female home range	100–300 ha	W. Myers, personal communication

Modeling Efforts

Mule deer were included in a linkage design project for the San Bernadino-San Jacinto area of Southern California (Penrod et al. 2005). Large blocks of protected lands were the source areas between which they sought to identify corridors. Their model used vegetation/land-use, topography, and road density to establish landscape resistance to movements.

Conceptual Basis for Model Development

Mule deer require a mosaic of habitat types of different age classes to meet their life history requirements. They use forest, woodland, brush, and meadow habitats, reaching their highest densities in open coniferous forests, riparian strips within arid and agricultural lands, and along edges of meadows and grasslands.

The mule deer dispersal habitat suitability resistance surface was created using the most suitable habitat conditions to represent the low range of resistance to movement. We used this resistance surface as the basis for identifying areas of concentrated suitable habitat. Apparently suitable habitat was eliminated from consideration if it fell outside of documented mule deer range (North American Mule Deer Foundation, unpublished data). Patches of 100 km^2 or greater were retained as HCAs.

OVERVIEW

Mule deer utilize a broad range of habitats. They require access to perennial water. Core areas potentially supporting 50 or more deer are equal to or greater than 260 ha. Patch size was classified as >100 km^2 (10,000 ha). Maximum dispersal distance was defined as 217 km Euclidean distance.

MOVEMENT DISTANCE

Mule deer have been observed dispersing in excess of 217 km in Euclidean distance (Robinette 1966).

Landscape Features and Resistance Values

The modeled resistance surface for mule deer (Table A.10) is based on the following spatial data layers:

1) Land cover/land-use
2) Elevation
3) Slope
4) Housing density
5) Road type and distance

Habitat Concentration Areas

General Definition — Habitat concentration areas (HCAs) are core habitat areas that are expected or known to be important habitat for focal species. These areas are used as the focus locations for running dispersal models that identify connectivity pathways.

Technical Definition — Habitat concentration areas are aggregations of habitat grid cells that are connected to each other by a species-specific home range movement radius. These aggregations must also meet a minimum size requirement designed to select only those areas that are capable of supporting multiple individuals. Connectivity between HCAs will be evaluated based on a species-specific dispersal distance and a landscape resistance model.

For mule deer, HCAs were identified using a combination of empirical knowledge of distribution and a habitat identification model. Steps used in the habitat identification model were as follows:

1) Develop a binary habitat surface where each grid cell in a raster is designated as either habitat or non-habitat. Habitat grid cells had resistance values of 3 or less. Mule deer range, as mapped by the North American Mule Deer Foundation (NAMDF) was used to eliminate areas that met the habitat definition but were outside the mapped range of mule deer (this included elimination of areas that the NAMDF classified as “Limited Range”).

2) Calculate the proportion of habitat within a circular moving window of a size representative of each species’ home range radius. A home range radius of 2 km was used. This step generates a surface which identifies the areas where the habitat is most concentrated.

3) Delete habitat grid cells in areas where habitat is sparse. Habitat grid cells were removed from the binary habitat raster if the proportion of habitat within a home range radius was <0.7. This prevented habitat concentrations from forming in areas where habitat is not concentrated to the degree required to be considered core habitat.

4) Join remaining habitat grid cells together if they are within a home range movement distance. Habitat areas were expanded outwards (from the remaining habitat grid cells

after step 3) up to a total cost-weighted distance equal to the species' home range movement radius (2 km). This had the effect of joining nearby habitat grid cells together if the intervening landscape supports within-home range connectivity.

5) Eliminate small patches unlikely to contribute significantly to a species' core habitat. Habitat patches less than 100 km^2 were eliminated.

Linkage Modeling

Mule deer are the most ubiquitous of the sixteen focal species chosen for the statewide analysis. They are expected to occur in many areas that lie outside of HCAs, sometimes at low densities. In recognition of this, we chose a liberal distance, 250 km, as the maximum cost-weighted distance at which habitat would be connected.

Considerations for Future Modeling

At more local scales, barrier effects that contribute to poor connections between seasonal ranges should be examined.

Table A.10. Landscape features and resistance values used to model habitat connectivity for mule deer.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	5
urban/developed	100
water	20
sparsely vegetated	5
alpine	0
riparian	0
wetland	1
grass-dominated	2
shrub-dominated	2
dry forest	0
wet forest	0
Elevation (meters)	
0–250	0
> 250–500	0
> 500–750	0
> 750–1000	0
> 1000–1500	0
> 1500–2000	1
> 2000–2500	2
> 2500–3300	25
> 3300 meters	25
Slope (degrees)	
0–20	0
> 20–40	0
> 40	30
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	0
> 20 ≤ 40	1
> 10 ≤ 20	2
< 10	10
Road type and distance (meters)*	
freeway > 500–1000 buffer	0
freeway > 0–500 buffer	0
freeway centerline	200
major highway > 500–1000 buffer	0
major highway > 0–500 buffer	0
major highway centerline	50
secondary highway > 500–1000 buffer	0
secondary highway > 0–500 buffer	0
secondary highway centerline	20
local road > 500–1000 buffer	0
local road > 0–500 buffer	0
local road centerline	2

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Acknowledgements

The author is grateful to W. Myers (WDFW) for his extraordinary knowledge of Washington's mule deer and willingness to share that knowledge. Others who made valuable contributions include: S. McCorquodale (WDFW), J. Nelson (WDFW), D. Ware (WDFW), T. Clausing (WDFW), J. Bernatowicz (WDFW), J. Stephenson (YNW), D. Reynolds (BCME), S. Fitkin (WDFW), D. Anderson (WDFW), A. Reid (BCME), L. Svancara (UI), Gregg Servheen (IDFG), T. Szkorupa (BCME), and H. Ferguson (WDFW)

Literature Cited

Harestad, A. S., and F. L. Bunnell. 1979. Home range and body weight – a reevaluation. Ecology 60:389–402.

Johnson, R. E., and K.M. Cassidy. 1997. Terrestrial mammals of Washington State: Location data and predicted distributions. Volume 3 in Washington State Gap Analysis – Final Report. K. M. Cassidy, C. E. Grue, M. R. Smith, and K. M. Dvornich, editors. Washington Cooperative Fish and Wildlife Research Unit, University of Washington, Seattle.

Kie, J. G., R. T. Bowyer, M. C. Nicholson, B. B. Boroski, and E. R. Loft. 2002. Landscape Heterogeneity at differing scales: effects on spatial distribution of mule deer. Ecology 83:530–544.

Myers, W. L. 2001. Job Progress Report Federal Aid in Wildlife Restoration: Ungulate Ecology and Population Dynamics in Washington. Progress Report. Washington Department of Fish and Wildlife, Olympia, Washington.

Penrod, K., C. Cabañero, P. Beier, C. Luke, W. Spencer, and E. Rubin. 2005. South Coast Missing Linkages Project: A Linkage Design for the San Bernardino-San Jacinto Connection. South Coast Wildlands, Idyllwild, California. Available from www.scwildlands.org (accessed June 2010).

Robinette, W. L. 1966. Mule deer home range and dispersal in Utah. Journal of Wildlife Management 30:335–349.

Thomas, T. R., and L. R. Irby. 1990. Habitat use and movement patterns by migrating mule deer in southeastern Idaho. Northwest Science 64:19–27.

Bighorn Sheep (*Ovis canadensis*)

Prepared by William L. Gaines (U.S. Forest Service)

Justification for Selection

Bighorn sheep (*Ovis canadensis*) were selected as focal species to represent grassland/shrubland habitats of the Semi-desert vegetation class. Many of the bighorn sheep herds are a result of re-introduction efforts throughout eastern Washington following their extirpation. Historically, California bighorn sheep occurred on the eastern slopes of the Cascades from the Canadian border south to the Columbia River. Most of the herds were gone before 1900; the last known survivors, on Chopaka Mountain, died in 1925 (Johnson 1999a). Rocky Mountain bighorn sheep occurred historically in the Selkirk Mountains but were extirpated from this area by the late 1800s (Johnson 1999b). Currently there are approximately 1000–1500 bighorn sheep statewide.

Distribution

Bighorn sheep are distributed across eastern Washington in 19 herds (14 California bighorn herds, 5 Rocky Mountain bighorn herds) each with a limited range. Rocky Mountain bighorn sheep occur in the Canadian Rockies adjacent to the U.S. border (Shackleton et al. 1999) and California bighorn sheep occur in and adjacent to the Okanagan Valley with three populations in the assessment area: Granby River, Ashnola, and Vaseaux Lake (Demarchi 1999). In the assessment area in Idaho there are three populations of Rocky Mountain bighorn sheep that include the Craig Mountain/Redbird herd, Nez Perce National Park herd, and Hells Canyon herd (Toweill 1999). In Oregon there are two populations of California bighorns, the Lower John Day herd and the Deschutes River herd (Van Dyke 1999). The Columbia River forms the northern boundary of these herds and is likely a barrier to Washington populations. There are also three populations of Rocky Mountain bighorns in Oregon within the assessment area: Lower Hells Canyon, Upper Joseph Creek, and Wenaha/Haas/Cottonwood (Coggins 1999).

Habitat Associations

Source habitat has been modeled for bighorn sheep in northeastern Washington by Begley (2008) and Gaines et al. (in prep.). They developed a model of habitat using logistic regression and telemetry data from the Swakane bighorn sheep herd in central Washington. They then extrapolated their model across northeastern Washington and used telemetry data from the Tieton and Vulcan herds to test the extrapolated model. The variables that were identified in the "best model" of bighorn sheep habitat included escape cover (mapped using digital elevation data), canopy closure, and vegetation zones.

Bighorn sheep have been reported to respond to human disturbance (Hicks & Elder 1979; MacArthur et al. 1979, 1982; Leslie & Douglas 1980; King & Workman 1986; Smith et al. 1991; Papouchis et al. 2001). MacArthur et al. (1979) showed that the heart rate of bighorn sheep varies inversely with the distance from a road. MacArthur et al. (1982) reported that sheep are affected by a human approaching within 50 m, and Papouchis et al. (2001) found that bighorn sheep respond to hikers at an average distance of 200 m. They also showed avoidance of roads was greater for high-use (5–13 vehicles per hour) versus low-use (1 vehicle per hour). On average, radio-collared sheep were 490 m from high-use roads compared to 354 m from low-use roads (Papouchis et al. 2001). Smith et al. (1991) developed a habitat suitability model for bighorn sheep and considered areas within 100 m of low to moderate human use (<500 visitors per year) trails and roads as unsuitable, and areas within 150 m of high human use (>500 visitors per year) trails and roads as unsuitable. Based on the above information, Gaines et al. (2003) developed a bighorn sheep habitat disturbance index used to assess habitat effectiveness. The index identified a zone of influence on each side of roads or trails (Table A.11) and then intersected this with the bighorn sheep habitat to assess the proportion of habitat within a zone of influence.

Table A.11. The zone of influence applied to each side of a trail or road based on road type and use level for bighorn sheep.

Trail or road type and status	*Zone of influence (m)*
Nonmotorized trail (ski or hiking)	200
Motorized trail	350
Road ≤ 1 vehicle per day	350
Road > 1 vehicle per day	500

Dispersal

Dispersal rates in female bighorn sheep have been reported to be very low (Festa-Bianchet 1991; Jorgenson et al. 1997). Epps et al. (2005) reported that the distance between populations of desert bighorn sheep appeared to be a prevailing natural barrier, as evidenced by the strong correlation of genetic diversity and gene flow with distance. They estimated a “barrier effect distance” to be about 40 km.

Modeling Efforts

A dispersal habitat suitability model was created by Begley (2008) following methods described in Singleton et al. (2002). The variables used in this model included: canopy closure, slope (Cassier et al. 1997), housing density (Johnson & Swift 2000), roads and trails (Johnson & Swift 2000; Gaines et al. 2003), and water-bodies >5 acres (Johnson & Swift 2000).

Conceptual Basis for Statewide Connectivity Model Development

OVERVIEW

Factors related to human land-use can limit or restrict bighorn movement and habitat by creating barriers (Epps et al. 2005). These barriers can include water systems (e.g., canals and reservoirs), high-use roads and highways, fences, and centers for human activity (e.g., agricultural lands and residential areas; Smith et al. 1991; Johnson & Swift 2000; Epps et al. 2005). Epps et al. (2005) found that barriers (highways, waterways, and human development) greatly reduced genetic diversity in desert bighorn sheep populations. The suppression of migration by barriers was most detectable within distances at which high relative gene flow was most detectable, in their case at <15 km between populations.

One approach put forth by Chetkiewicz et al. (2006) was to use resource selection functions (RSF) to determine permeability values (1/RSF). Begley (2008) provided relative RSF for bighorn sheep that we used to inform the development of resistance values (Table A.12); they could not be used directly due to differences in the study area specific spatial data the RSFs were based on and the broader scale spatial data used in our statewide assessment.

MOVEMENT DISTANCE

The distance we used for determining which HCAs to link was a cost-weighted distance of 1000 km.

Landscape Features and Resistance Values

The modeled resistance surface for bighorn sheep (Table A.12) is based on the following spatial data layers:

1) Land cover/land-use
2) Elevation
3) Slope
4) Housing density
5) Road type and distance
6) Forest structure

Habitat Concentration Areas

Habitat concentration areas for bighorn sheep were identified using a GIS layer of herd ranges compiled for the western U.S. and Canada (D. Martorello, personal communication). Habitat connectivity was then evaluated between these areas.

Herd ranges and HCAs were limited to south-central British Columbia, eastern Washington, northern and central Idaho, and the Blue Mountains in northeastern Oregon and southeastern Washington. A total of 37 HCAs were identified within the project area.

Linkage Modeling

Potential linkages were modeled between HCAs when the least-cost distance was <1000 km.

Table A.12. Landscape features and resistance values used to model habitat connectivity for bighorn sheep.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	7
urban/developed	500
water	10
sparsely vegetated	0
alpine	5
riparian	5
wetland	5
grass-dominated	0
shrub-dominated	1
dry forest	0
wet forest	5
Elevation (meters)	
0–250	0
> 250–500	0
> 500–750	0
> 750–1000	1
> 1000–1500	1
> 1500–2000	5
> 2000–2500	5
> 2500–3300	10
> 3300 meters	1000
Slope (degrees)	
0–20	5
> 20–40	0
> 40	2
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	2
> 20 ≤ 40	5
> 10 ≤ 20	10
< 10	100
Road type and distance (meters)*	
freeway > 500–1000 buffer	100
freeway > 0–500 buffer	200
freeway centerline	1000
major highway > 500–1000 buffer	5
major highway > 0–500 buffer	10
major highway centerline	500
secondary highway > 500–1000 buffer	5
secondary highway > 0–500 buffer	10
secondary highway centerline	50
local road > 500–1000 buffer	2
local road > 0–500 buffer	5
local road centerline	10
Forest structure (density and height [meters])	
nonforest	0
sparse low (0–40%, ≤ 25)	0
sparse high (0–40%, > 25)	0
open low (> 40–70%, ≤ 25)	10
open high (> 40–70%, > 25)	10
dense low (> 70–100%, ≤ 25)	25
dense high (> 70–100%, > 25)	25

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Considerations for Future Modeling

Future modeling should include fine-scale features that would influence habitat connectivity for bighorn sheep, such as fences. Finer-scale linkage modeling will allow more specific identification of areas where habitat connectivity can be restored or enhanced. Finally, landscape genetic techniques or radio-telemetry may be used to evaluate the linkage models.

Acknowledgements

The initial resistance values and HCAs were reviewed by David Wallin (WWU), Scott McCorquodale (WDFW), Jeff Bernatowitz (WDFW), Jim Stephenson (YNW), and James Begley (WTI). A special thanks to Andrew Shirk (Independent Researcher) and Brian Cosentino (WDFW) for their GIS and analytical support.

Literature Cited

Begley, J. 2008. Habitat assessment for bighorn sheep. Master's thesis. Central Washington University, Ellensburg, Washington.

Cassirer, F., et al. 1997. Restoration of bighorn sheep to Hells Canyon: the Hells Canyon initiative. Idaho Department of Fish and Game, Oregon Department of Fish and Wildlife, Washington Department of Fish and Wildlife, U.S. Forest Service, Bureau of Land Management, and the Foundation for North American Wild Sheep.

Chetkiewicz, C. B., C. C. St. Clair, and M. S. Boyce. 2006. Corridors for conservation: integrating pattern and process. Annual Review of Ecology, Evolution, and Systematics 37:317–342.

Coggins, V. 1999. Rocky Mountain Bighorns: Oregon. Pages 100–103 in D. E. Toweill and V. Geist, editors. Return of royalty: wild sheep of North America. Boone and Crockett Club and Foundation for North American Wild Sheep, Missoula, Montana.

Demarchi, R. 1999. California Bighorns: British Columbia. Pages 126–129 in D. E. Toweill and V. Geist, editors. Return of royalty: wild sheep of North America. Boone and Crockett Club and Foundation for North American Wild Sheep, Missoula, Montana.

Epps, C. W., P. J. Palsboll, J. D. Wehausen, G. K. Roderick, R. R. Ramey, and D. R. McCullough. 2005. Highways block gene flow and cause a rapid decline in genetic diversity of Desert bighorn sheep. Ecology Letters 8:1029–1038.

Festa-Bianchet, M. 1991. The social system of bighorn sheep: grouping patterns, kinship, and dominance rank. Animal Behaviour 42:71–82.

Gaines, W. L., P. H. Singleton, and R. C. Ross. 2003. Assessing the cumulative effects of linear recreation routes on wildlife habitats on the Okanogan and Wenatchee National Forests. General Technical Report N-586, Pacific Northwest Research Station, Corvallis, Oregon.

Hicks, L. L., and J. M. Elder. 1979. Human disturbance of Sierra Nevada bighorn sheep. Journal of Wildlife Management 43:909–915.

Johnson, R. 1999a. California Bighorns: Washington. Pages 152–153 in D. E. Toweill and V. Geist, editors. Return of royalty: wild sheep of North America. Boone and Crockett Club and Foundation for North American Wild Sheep, Missoula, Montana.

Johnson, R. 1999b. Rocky Mountain Bighorns: Washington. Pages 112–113 in D. E. Toweill and V. Geist, editors. Return of royalty: wild sheep of North America. Boone and Crockett Club and Foundation for North American Wild Sheep, Missoula, Montana.

Johnson, T. L., and D. M. Swift. 2000. A test of habitat evaluation procedure for Rocky Mountain bighorn sheep. Restoration Ecology 8:47–56.

Jorgenson, J. T., M. Festa-Bianchet, J. M. Gaillard, and W. D. Wishart. 1997. Effects of age, sex, disease, and density on survival of bighorn sheep. Ecology 78:1019–1032.

King, M. M., and G. W. Workman. 1986. Response of desert bighorn sheep to human harassment: management implications. Pages 74–85 in Transactions of the 51st North American Wildlife and Natural Resources Conference, Reno, Nevada.

Leslie, D. M., and C. L. Douglas. 1980. Human disturbance at water sources of desert bighorn sheep. Wildlife Society Bulletin 8:284–290.

MacArthur, R. A., R. H. Johnston, and V. Geist. 1979. Factors influencing heart rate in free-ranging bighorn sheep: a physiological approach to the study of wildlife harassment. Canadian Journal of Zoology 57:2010–2121.

MacArthur, R. A., V. Geist, and R. H. Johnston. 1982. Cardiac and behavioral responses of mountain sheep to human disturbance. Journal of Wildlife Management 46:351–358.

Papouchis, C. M., F. J. Singer, and W. B. Sloan. 2001. Responses of desert bighorn sheep to increased human recreation. Journal of Wildlife Management 65:573–582.

Shackleton, D., I. Hatter, and H. Schwantje. 1999. Rocky Mountain Bighorns: British Columbia. Pages 74–77 in D. E. Toweill and V. Geist, editors. Return of Royalty: wild sheep of North America. Boone and Crockett Club and Foundation for North American Wild Sheep, Missoula, Montana.

Singleton, P. H., W. L. Gaines, and J. F. Lehmkuhl. 2002. Landscape permeability for large carnivores in Washington: A geographic information system weighted-distance and least-cost corridor assessment. General Technical Report N-549. Pacific Northwest Research Station, Corvallis, Oregon.

Smith, T. S., J. T. Flinders, and D. S. Winn. 1991. A habitat evaluation procedure for Rocky Mountain bighorn sheep in the intermountain West. Great Basin Naturalist 51:205–225.

Toweill, D. 1999. Rocky Mountain Bighorns: Idaho. Pages 82–85 in D. E. Toweill and V. Geist, editors. Return of royalty: wild sheep of North America. Boone and Crockett Club and Foundation for North American Wild Sheep, Missoula, Montana.

Van Dyke, W. 1999. California bighorn sheep – Oregon. Pages 144–149 in D. E. Toweill and V. Geist, editors. Return of royalty: wild sheep of North America. Boone and Crockett Club and Foundation for North American Wild Sheep, Missoula, Montana.

Personal Communication

D. Martorello
Washington Department of Fish and Wildlife
Olympia, Washington

Western Gray Squirrel (*Sciurus griseus*)

Prepared by Kelly R. McAllister (Washington Department of Transportation)

Justification for Selection

The western gray squirrel (*Sciurus griseus*) was selected as a good representative of wildlife habitat connectivity needs within the Rocky Mountain Forests vegetation class. The species was considered vulnerable to loss of habitat connectivity from all four overarching connectivity threats: land clearing and vegetation removal, development, roads and traffic, and the presence of people and domestic animals. Western gray squirrels are well distributed within the Rocky Mountain Forests vegetation class for which they were selected. Their populations are limited by human-created barriers to dispersal and their movement scale is appropriate for a statewide modeling effort. Information on the species is believed adequate for model development.

Distribution

Western gray squirrels range from north-central Washington south to the southern border of California. Within Washington, their range consists of three geographically distinct areas: South Puget Sound (primarily Joint Base Lewis-McChord), Klickitat County extending into Yakima County, and the Lake Chelan and Methow Valley region.

Habitat Associations

Western gray squirrels inhabit mast-producing conifer-hardwood forest types. In Washington, these forest types include transitional forests of ponderosa pine, Oregon white oak (*Quercus garryana*), Douglas-fir and various riparian tree species. Most occupied forest habitats contain pine or oak, though the presence of both is not essential. The most prevalent mast-producing trees and shrubs, within western gray squirrel ranges, are Douglas maple (*Acer douglasii*), vine maple (*Acer circinatum*), bigleaf maple (*Acer macrophylum*), hazelnut (*Corylus cornuta*), oceanspray (*Holodiscus discolor*), blue elderberry (*Sambucus cerulea*), huckleberry (*Vaccinium* spp.), snowberry (*Symphoricarpos albus*), and serviceberry (*Amelanchier alnifolia*). Mixed stands of Oregon white oak and either ponderosa pine or Douglas-fir with a sparse or open understory are preferred.

The geographic range of extant populations in Washington overlap with the following forested Ecological Systems:

1) Columbia Basin Foothill riparian woodland and shrubland

2) East Cascades Mesic Montane Mixed-Conifer Forest and Woodland

3) East Cascades Oak Ponderosa Pine Forest and Woodland *

4) Great Basin foothill and lower Montane Riparian Woodland and Shrubland

5) North Pacific Dry Douglas-fir (Madrone) Forest

6) North Pacific Lowland Riparian Forested Shrubland
7) North Pacific Oak Woodland*
8) North Pacific Maritime Mesic-wet Douglas-fir Western Hemlock forest
9) North Pacific Maritime Dry-Mesic Douglas-fir Western Hemlock forest
10) Northern Rocky Mountain Dry-Mesic Montane Mixed Conifer forest
11) Northern Rocky Mountain Ponderosa Pine Woodland and Savannah*
12) Rocky Mountain Lodgepole Pine forest

**Ecological System with which western gray squirrels have a close association.*

Dispersal/Movements

Western gray squirrels regularly move 4–5 km in brief time intervals (Table A. 13). Juveniles have been tracked dispersing an average of 2862 m from their natal site (Vander Haegen et al. 2005). The longest recorded movement distance was noted in an adult squirrel fitted with a radio collar in Chelan County. This animal moved 19.2 km in a two week time span (M. Vander Haegen, personal communication).

Table A.13. Western gray squirrel movement or dispersal distances.

Description	*Distance*	*Citation*
Adult female, Klickitat County	4 km	Linders & Stinson 2007
Male mating season movements	>5 km/day	Cross 1969; Linders 2000
Juvenile dispersal	2862 m (mean, *n* = 6)	Vander Haegen et al. 2005
Adult, Chelan County	19.2 km/2 wks	W. M. Vander Haegen, personal communication

Modeling Efforts

Suitable habitat models have been developed for Washington (Rodrick 1999; Linders & Stinson 2007). These models identify forest types with Oregon white oak and ponderosa pine inclusions that are within the range limits of the species.

Conceptual Basis for Model Development

OVERVIEW

Identifying suitable western gray squirrel habitat for use in modeling benefitted from a considerable amount of survey data available as a GIS point layer. Patterns evident from these occurrence data helped, along with written descriptions of habitat, to identify other potentially suitable habitat areas that are not known to be occupied by western gray squirrels. The view that these areas could eventually be inhabited by western gray squirrels is critical to a future with connected populations of western gray squirrels along the entire north-south axis of the eastern Cascade foothills. The forest types used for identifying HCAs were those with ponderosa pine or Oregon white oak, two tree species that are present in the majority of occupied western gray squirrel habitat in the project area.

The resistance surface for the western gray squirrel assumes that many forest conditions are suitable for western gray squirrel movements, without regard for the presence of any particular tree species.

MOVEMENT DISTANCE

Western gray squirrels are relatively mobile. However, information that adequately represents their movement capabilities is far from complete and much is learned during each passing year. Connectivity across the broad scale represented by our project area will require corridor-dwelling animals in habitat patches distributed along much of the length of identified corridors.

Landscape Features and Resistance Values

Habitat concentration areas were identified from: (1) known occupied habitat, and (2) areas with concentrations of ponderosa pine or Oregon white oak forests within the historical range of the species.

Resistance parameters were derived, primarily, from literature describing suitable habitat characteristics (Table A.14). Resistance parameters for non-habitat conditions were based on professional judgment and vetted with experts attending a workshop in Cle Elum, Washington on 10 November 2009. The modeled resistance surface for western gray squirrel is based on the following spatial data layers:

1) Land cover/land-use
2) Elevation
3) Slope
4) Housing density
5) Road type and distance

Habitat Concentration Areas

General definition — Habitat concentration areas (HCAs) are core habitat areas that are expected or known to be important habitat for focal species. These areas are used as the focus locations for running dispersal models that identify connectivity pathways.

Technical Definition — Habitat concentration areas are aggregations of habitat grid cells that are connected to each other by a species-specific home range movement radius. These aggregations must also meet a minimum size requirement designed to select only those areas that are capable of supporting multiple individuals. Connectivity between HCAs will be evaluated based on a species-specific dispersal distance and a landscape resistance model.

For western gray squirrels, HCAs were identified using a combination of survey results and identification of concentrations of suitable habitat within the species' historical range. The steps used for HCA identification were as follows:

1) Develop a binary habitat surface where each grid cell in a raster is designated as either habitat or non-habitat. Habitat grid cells were a subset of grid cells the Regap Ecological Systems data layer classified as Northern Rocky Mountain Ponderosa Pine Woodland & Savannah, North Pacific Oak Woodland or East Cascades Oak-Ponderosa Pine Forest & Woodland. To be included, a grid cell had to be one of these three Ecological Systems types and have a resistance value of 10 or less. Screening out grid cells with resistance values higher than 10 prevented grid cells with centerlines of busy roads or dense in human dwellings, from being considered as suitable quality habitat.

2) Calculate the proportion of habitat within a circular moving window of a size representative of the western gray squirrel's home range radius. A home range radius of 2 km was used based on estimates found in western gray squirrel movement studies (Table A.13). This step generates a surface which identifies the areas where the habitat is most concentrated.

3) Delete habitat grid cells in areas where habitat is sparse. Habitat grid cells were removed from the binary habitat raster if the proportion of habitat within a home range radius was <0.1. This prevented habitat concentrations from forming in areas where habitat is not concentrated to the degree required to be considered core habitat.

4) Join remaining habitat grid cells together if they are within a home range movement distance. Habitat areas were expanded outwards (from the remaining habitat grid cells after step 3) up to a total cost-weighted distance equal to the species home range movement radius (2 km). This had the effect of joining nearby habitat grid cells together if the intervening landscape supports within home-range connectivity.

5) Eliminate small patches unlikely to contribute significantly to a species core habitat. Habitat patches less than 50 km^2 were eliminated.

6) Eliminate areas outside the documented range of the species. Screen out areas outside the historical range of the western gray squirrel.

7) Manually alter or adjust as necessary. Add in the known occupied habitat area at Joint Base Lewis-McChord, the only known occupied area that failed to be identified by the model, likely due to the somewhat atypical forest communities inhabited by western gray squirrels there, usually including a codominance by Douglas-fir.

Linkage Modeling

Linkages were modeled between all HCAs within 200 km cost-weighted distance of each other. Linkages approaching the maximum cost-weighted distance length are unlikely to be useful for connectivity without linkage dwellers and connectivity achieved over multiple generations.

Considerations for Future Modeling

Modeling at more local scales should identify and address important partial or complete barriers within the three western gray squirrel recovery zones (Puget Trough, North Cascades, and South Cascades).

Table A.14. Landscape features and resistance values used to model habitat connectivity for western gray squirrels.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	100
urban/developed	200
water	500
sparsely vegetated	5
alpine	500
riparian	0
wetland	100
grass-dominated	5
shrub-dominated	5
dry forest	0
wet forest	0
Elevation (meters)	
0–250	0
> 250–500	0
> 500–750	0
> 750–1000	0
> 1000–1500	5
> 1500–2000	10
> 2000–2500	20
> 2500–3300	100
> 3300 meters	100
Slope (degrees)	
0–20	0
> 20–40	0
> 40	5
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	0
> 20 ≤ 40	5
> 10 ≤ 20	10
< 10	50
Road type and distance (meters)*	
freeway > 500–1000 buffer	0
freeway > 0–500 buffer	0
freeway centerline	200
major highway > 500–1000 buffer	0
major highway > 0–500 buffer	0
major highway centerline	100
secondary highway > 500–1000 buffer	0
secondary highway > 0–500 buffer	0
secondary highway centerline	50
local road > 500–1000 buffer	0
local road > 0–500 buffer	0
local road centerline	5

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Acknowledgements

Matt Vander Haegen (WDFW) was generous with his considerable knowledge of western gray squirrels. Matt, along with John Lehmkuhl (USFS), Winston Smith (USFS PNW), Gina King (YNW), and Dave Werntz (CNW), provided a valuable critique of an earlier version of this document and the associated model. I depended heavily on the work of Mary Linders (WDFW), Derek Stinson (WDFW) and others who have completed field studies and assembled and interpreted data from throughout the range of the species. To all of these generous scientists, I express my sincere gratitude.

Literature Cited

Cross, Stephen Paul. 1969. Behavioral aspects of western gray squirrel ecology. PhD dissertation, University of Arizona, Tucson, Arizona.

Linders, M. J. 2000. Spatial ecology of the western gray squirrel, (*Sciurus griseus*) in Washington: The interaction of season, habitat and home range. Master's thesis, University of Washington, Seattle, Washington.

Linders, M. J., and D. W. Stinson. 2007. Washington State recovery plan for the western gray squirrel. Washington Department of Fish and Wildlife, Olympia, Washington.

Rodrick, E. A. 1999. Western gray squirrel habitat mapping and surveys in Washington State, 1994-1996. Unpublished Report. Washington Department of Fish and Wildlife, Olympia, Washington.

Vander Haegen, M., G. R. Orth, and L. M. Aker. 2005. Ecology of the western gray squirrel in south-central Washington. Progress Report. Washington Department of Fish and Wildlife, Olympia, Washington.

American Black Bear (*Ursus americanus*)

Prepared by William L. Gaines (U.S. Forest Service)

Justification for Selection

American black bears (*Ursus americanus*) were selected as a focal species due to their broad distribution within the assessment area, association with forested habitats, and wide-ranging space-use patterns. Several characteristics of bear ecology make American black bears an ideal study species to evaluate techniques for delineating habitat linkages (Kindall & Van Manen 2007). These characteristics include large home ranges and relative sensitivity to landscape fragmentation (Beier & Noss 1998). Cushman et al. (2009) evaluated the potential for American black bears to be used as a surrogate for the federally threatened grizzly bear (*Ursus arctos*) in the identification of regional conservation corridors. They found considerable overlap in the areas they identified as important corridors for American black bears compared to areas that have been identified as important “linkage zones” for grizzly bears (Mietz 1994; Sandstrom 1996; Waller & Servheen 2005).

Distribution

American black bears are wide-spread throughout the forested portions of the assessment area (Hummel et al. 1991; Johnson & Cassidy 1997). Sizeable gaps in their distribution include the Columbia Basin and the Okanogan Valley (WDFW 1996).

Habitat Associations

Studies of the habitat associations of American black bears have been conducted on both the east and west sides of the Cascades and on the Olympic Peninsula (Poelker & Hartwell 1973; Lindzey & Meslow 1977; Koehler & Pierce 2003; Lyons et al. 2003; Gaines et al. 2005). American black bears in all study areas used a variety of forested and non-forested cover types. However, on the east-side of the Cascades important habitats included riparian forest, deciduous forest and montane-high elevation forests. Generally, dry non-forest habitats such as shrubsteppe were ranked as low-use by bears (Lyons et al. 2003; Gaines et al. 2005). Important variables in all models of American black bear habitat use were distances to roads of varying use levels and open road density (Gaines et al. 2005). Generally, areas adjacent to low use (<1 vehicle/hr) roads were used more than areas adjacent to moderate (1–10 vehicles/hr) and high (>10 vehicles/hr) use roads (Gaines et al. 2005). In addition, areas with no open roads >200 ha in size consistently ranked as important predictors of American black bear habitat use (Gaines et al. 2005).

The willingness of American black bears to cross roads of different traffic volumes has not been extensively studied. However, Beringer et al. (1989) did study road crossing by black bears in North Carolina and found significantly low crossing of Class I roads which had >10,000 vehicles/day compared to Class II (50–100 vehicles/day) and Class III (5–20 vehicles/day) roads.

Dispersal

Studies of dispersal by American black bears show that subadult males are the most likely to move into new areas away from their natal home ranges (Rogers 1987; Beck 1991; Beecham & Rohlman 1994; Wertz et al. 2000; Table A.15). Males tend to disperse greater distances than females. Reported dispersal distances for females range from 12 to 55 km (Elowe & Dodge 1989; Samson & Huot 1998) while male distances range from 30 to 200 km (Elowe & Dodge 1989; Wertz et al. 2000). In the Blue Mountains of Oregon subadult males (ages 2–4 years) dispersed 33.4–95.8 km. Interstate 84 (I-84), which bordered the east and north of the Oregon study area functioned to limit dispersal. Only 1 bear attempted to cross the freeway during the 7 year study and was killed along with one of her cubs (Wertz et al. 2000). Based on the available information, it is recommended that a maximum cost-weighted distance of 400 km be used in the modeling of habitat connectivity.

Table A.15. American black bear dispersal distances.

Description	*Distance(km)*	*Citation*
Subadult males	33.4–95.8	Wertz et al. 2000
2.5 year old female	15	Elowe & Dodge 1989
2 year old males	30–200	Elowe & Dodge 1989
Females	12–55	Samson & Huot 1998

Modeling Efforts

Cushman et al. (2006) tested landscape-resistance hypotheses using genetic distance metrics. The variables they used to develop the hypotheses included elevation, slope, roads and land cover. The models most fully supported by the genetic distance data showed strong relationships with forest cover and mid-elevations, with equivocal support for different levels of road factors and no relationship with slope (Cushman et al. 2006). The best supported model, however, had high road resistance, which was used to identify corridors. Cushman et al. (2008) extrapolated the model to a broader geographic area to identify potential corridors for American black bears between the forested portions of the Canadian border (between the Idaho-Washington state border and the eastern edge of Glacier National Park) down to the northern boundary of Yellowstone National Park. They identified three categories of potential barriers along the potential bear movement corridors: gaps in federal ownership that contain freeways and major highways; areas within federal ownership where major highways cross the corridor; and areas where major corridors parallel highways (Cushman et al. 2008).

Singleton et al. (2004) assessed landscape permeability for grizzly bears across the state of Washington and adjacent areas in British Columbia and Idaho. They used land cover, human population density, road density, elevation, and slope variables. Four potential fracture zones for grizzly bears were identified between habitat concentration areas: Stevens Pass, Fraser-Coquihalla, Okanogan Valley-Upper Columbia, and Snoqualmie Pass (Singleton et al. 2004).

Conceptual Basis for Statewide Connectivity Model Development

OVERVIEW

The data presented about American black bear habitat associations and connectivity were used to determine how HCAs would be identified and dispersal habitat suitability modeled. One approach put forth by Chetkiewicz et al. (2006) was to use resource selection functions (RSF) to determine permeability values (1/RSF). Gaines et al. (2005) provided relative RSF for American black bears that were used to inform the development of resistance values (Table A.16) but they could not be used directly due to differences in the study area specific spatial data the RSFs were based on and the broader scale spatial data used in the statewide assessment.

Movement Distance

The maximum movement distance to model connectivity between habitat concentration areas was a cost-weighted distance of 400 km.

Landscape Features and Resistance Values

Information was used from published habitat connectivity models (Singleton et al. 2002; Cushman et al. 2006) and modified with local research on resource selection (Koehler & Pierce 2003; Lyons et al. 2003; Gaines et al. 2005) to derive resistance values for dispersal habitat suitability modeling (Table A.16). The resistance surface for American black bears is based on the following spatial data layers:

1) Land cover/land-use
2) Elevation
3) Slope
4) Housing density
5) Road type and distance
6) Forest structure

Habitat Concentration Areas

Habitat concentration areas were identified using a resistance value of ≤6, a home range radius of 2.6 km, a moving window threshold of 0.5, and a minimum patch size of 200 km^2. The minimum patch size of 200 km^2, designated for large carnivores, represents 10 average female home ranges reported in Koehler and Pierce (2003; Table A.17).

Habitat concentration areas are well distributed throughout the known distribution of American black bears within the project area (See Chapter 3; WDFW 1996; Johnson & Cassidy 1997). Areas that are within the distribution of the American black bear but are not included within HCAs include southwestern Washington where high concentrations of human activities, such as roads, resulted in high resistance values. Other notable gaps in the distribution of HCAs occurs along the Okanogan and Upper Columbia River valleys where a combination of low-elevation dry vegetation types, rivers, highways, and other human activities precluded inclusion within a HCA. These patterns are relatively consistent with those presented for the general forest carnivore model by Singleton et al. (2002) and represent a reasonable approximation of the distribution of high quality habitat for American black bears across the project area.

Table A.16. Landscape features and resistance values used to model habitat connectivity for American black bears.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	100
urban/developed	200
water	100
sparsely vegetated	1
alpine	0
riparian	0
wetland	0
grass-dominated	1
shrub-dominated	1
dry forest	1
wet forest	0
Elevation (meters)	
0–250	5
> 250–500	5
> 500–750	4
> 750–1000	3
> 1000–1500	2
> 1500–2000	1
> 2000–2500	0
> 2500–3300	1
> 3300 meters	100
Slope (degrees)	
0–20	0
> 20–40	1
> 40	3
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	10
> 20 ≤ 40	10
> 10 ≤ 20	10
< 10	100
Road type and distance (meters)*	
freeway > 500–1000 buffer	10
freeway > 0–500 buffer	50
freeway centerline	1000
major highway > 500–1000 buffer	5
major highway > 0–500 buffer	10
major highway centerline	100
secondary highway > 500–1000 buffer	4
secondary highway > 0–500 buffer	8
secondary highway centerline	50
local road > 500–1000 buffer	1
local road > 0–500 buffer	2
local road centerline	3
Forest structure (density and height [meters])	
nonforest	1
sparse low (0–40%, ≤ 25)	0
sparse high (0–40%, > 25)	0
open low (> 40–70%, ≤ 25)	0
open high (> 40–70%, > 25)	0
dense low (> 70–100%, ≤ 25)	0
dense high (> 70–100%, > 25)	0

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Table A.17. Reported American black bear homes ranges for Washington.

Study area	*Home range size (km^2)*	*Citation*
Olympic Pennisula	125.5 (male) 28.3 (female)	Koehler & Pierce 2003
Westside of Cascades	90.8 (male) 18.0 (female)	Koehler & Pierce 2003
Eastside of Cascades	73.5 (male) 25.9 (female)	Koehler & Pierce 2003
Eastside of Cascades	289.7 (male) 37.1 (female)	Lyons et al. 2003
Long Island	5.05 (male) 2.35 (female)	Lindzey & Meslow 1977

Linkage Modeling

Linkages were modeled between HCAs using information from the resistance surfaces and American black bear dispersal ecology. Potential linkages for American black bears were modeled when the least-cost distance between a pair of HCAs was <400 km.

Considerations for Future Modeling

Future modeling should include more detailed vegetation information, in particular, the ability to discriminate between low-elevation grasslands and shrub fields that occur in dry environments versus those that occur in more moist environments. These habitats have very different value to American black bears but were difficult to separate with the broad-scale vegetation data used in our assessment. Finer-scale linkage modeling will allow more specific identification of areas where habitat connectivity can be restored or enhanced. Finally, the American black bear, in part because it is relatively common, provides an important opportunity to evaluate our habitat connectivity modeling efforts, both the identification of HCAs and potential linkages, through the use of landscape genetic techniques (Excoffier & Heckel 2006; Frantz et al. 2009).

Acknowledgements

Initial resistance values and HCAs were reviewed by Gary Koehler (WDFW), Robert Long (WTI), James Begley (WTI), and David Wallin (WWU). A special thanks to Andrew Shirk (Independent Researcher) and Brian Cosentino (WDFW) for their GIS and analytical support.

Literature Cited

Beck, T. D. 1991. Black bears of west-central Colorado. Colorado Division of Wildlife Technical Bulletin N-39, Fort Collins, Colorado.

Beecham, J. J., and J. Rohlman. 1994. A shadow in the forest, Idaho’s black bear. University of Idaho Press, Moscow, Idaho.

Beier, P., and R. F. Noss. 1998. Do habitat corridors provide connectivity? Conservation Biology 12:1241–1252.

Beringer, J. J., S. G. Siebert, and M. R. Pelton. 1989. Incidence of road crossing by black bears on Pisgah National Forest, North Carolina. International conference on bear research and management 8:85–92.

Chetkiewicz, C. B., C. C. St. Clair, and M. S. Boyce. 2006. Corridors for conservation: integrating pattern and process. Annual Review of Ecology, Evolution, and Systematics 37:317–342.

Cushman, S. A., K. S. McKelvey, J. Hayden, and M. K. Schwartz. 2006. Gene flow in complex landscapes: testing multiple hypotheses with causal modeling. The American Naturalist 168: 486–499.

Cushman, S. A., K. S. McKelvey, and M. K. Schwartz. 2009. Use of empirically derived source-destination models to map regional conservation corridors. Conservation Biology 23:368–376.

Elowe, K. D., and W. E. Dodge. 1989. Factors affecting black bear reproductive success and cub survival. Journal of Wildlife Management 53:962–968.

Excoffier, L., and G. Heckel. 2006. Computer programs for population genetics data analysis: a survival guide. Nature Reviews 7:745–758.

Frantz, A. C., S. Cellina, A. Krier, L. Schley, and T. Burke. 2009. Using spatial Bayesian methods to determine the genetic structure of a continuously distributed population: clusters or isolation by distance? Journal of Applied Ecology 46:493–505.

Gaines, W. L., A. L. Lyons, J. F. Lehmkuhl, and K. J. Raedeke. 2005. Landscape evaluation of female black bear habitat effectiveness and capability in the North Cascades, Washington. Biological Conservation 125:411–425.

Hummel, M., S. Pettigrew, and J. Murray. 1991. Wild hunters: predators in peril. Roberts Rhinehart Publishers, Niwot, Colorado.

Johnson, R. E., and K. M. Cassidy. 1997. Terrestrial mammals of Washington State: location data and predicted distributions. Volume 3 in Washington State Gap Analysis – Final Report. K. M. Cassidy, C. E. Grue, M. R. Smith, and K. M. Dvornich, editors. Washington Cooperative Fish and Wildlife Research Unit, University of Washington, Seattle.

Kindall, J. L., and F. T. Van Manen. 2007. Identifying habitat linkages for American black bears in North Carolina, USA. Journal of Wildlife Management 71:487–495.

Koehler, G. M., and J. Pierce. 2003. Black bear home-range sizes in Washington: climatic, vegetative, and social influences. Journal of Mammalogy 84:81–91.

Lindzey, F. G., and E. C. Meslow. 1977. Home range and habitat use by black bears in southwestern Washington. Journal of Wildlife Management 41:413–425.

Lyons, A. L., W. L. Gaines, and C. Servheen. 2003. Black bear resource selection in the northeastern Cascades, Washington. Biological Conservation 113:55–62.

Mietz, S. N. 1994. Linkage zone identification and evaluation of management options for grizzly bears in the Evaro Hill area. Master's thesis. University of Montana, Missoula, Montana.

Poelker, R. J., and H. D. Hartwell. 1973. Black bear of Washington: its biology, natural history, and relationship to forest regeneration. Washington State Department of Game, Biological Bulletin N-14, Olympia, Washington.

Rogers, L. L. 1987. Effects of food supply and kinship on social behavior, movements, and population growth of black bears in northeastern Minnesota. Wildlife Monographs N-97.

Samson, C., and J. Huot. 1998. Movements of female black bears in relation to landscape vegetation type in southern Quebec. Journal of Wildlife Management 62:718–727.

Sandstrom, P. L. 1996. Identification of potential linkage zones for grizzly bears in the Swan-Clearwater valley using GIS. Master's thesis. University of Montana, Missoula, Montana.

Singleton, P. H., W. L. Gaines, and J. F. Lehmkuhl. 2002. Landscape permeability for large carnivores in Washington: A geographic information system weighted-distance and least-cost corridor assessment. Research Paper N-549. U. S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Singleton, P. H., W. L. Gaines, and J. F. Lehmkuhl. 2004. Landscape permeability for grizzly bear movements in Washington and southwestern British Columbia. Ursus 15:90–103.

Waller, J. S., and C. Servheen. 2005. Effects of transportation infrastructure on grizzly bears in northwestern Montana. Journal of Wildlife Management 69:985–1000.

WDFW (Washington Department of Fish and Wildlife). 1996. Environmental impact statement for black bear management. Washington Department of Fish and Wildlife, Olympia, Washington.

Wertz, T. L., J. J. Akenson, M. G. Henjum, and E. L. Bull. 2000. Home range and dispersal patterns of subadult black bears in northeastern Oregon. Western black bear Workshop 7:93–100.

Elk (*Cervus elaphus*)

Prepared by Kelly R. McAllister (Washington Department of Transportation)

Justification for Selection

Elk (*Cervus elaphus*) are important members of the wildlife community, serving a number of key ecological functions as herbivores and prey for large carnivores such as cougars and wolves. Some local populations are migratory, exploiting productive mountain meadow habitat in summer but retreating to low-elevation valleys in winter. As such, migratory elk often move long distances on a seasonal basis.

Elk are known to be affected by development, roads and traffic, and the presence of people and domestic animals. Elk avoid human activity unless in an area that is secure from predation in which case they are often tolerant of human proximity (Morgantini & Hudson 1979; Geist 2002; Lyon & Christensen 2002). Landowners, though, are frequently intolerant of the presence of elk. When confronted with elk grazing in their gardens and pastures and interacting with their livestock, horses or pets, landowners frequently seek relief from their local department of Fish and Wildlife.

The Department of Fish and Wildlife's Landscape Priority Habitats and Species Project characterized elk as having high sensitivity to the effects of roads and development. One population, the Nooksack elk population, is a considered a Species of Greatest Conservation Need in Washington State.

Distribution

Elk, or red deer, were once found throughout much of the Northern Hemisphere, from Europe through northern Africa, Asia, and North America. Extensive hunting and habitat destruction have limited elk to a portion of their former range. Elk populations in eastern North America were extirpated largely as a result of overhunting. Today large populations in North America are found only in the western United States from Canada through the Eastern Rockies to New Mexico, and in a small region of the northern lower peninsula of Michigan. Elk have been introduced to Ireland, Argentina, Chile, Australia, and New Zealand.

In the Pacific Northwest, elk are common to abundant in most mountainous regions and are present in many low-lying valleys, particularly during winter. The only extensive areas with few to no elk are the arid desert regions.

Habitat Associations

In the Pacific Northwest, elk are associated with a wide variety of habitat conditions including forest habitats spanning the full range of moisture conditions and even shrubsteppe environments where there are no trees within the herd's range. In general, though, elk are associated with open woodlands or a mosaic of mature forest, meadow, and early successional forest conditions. They avoid dense, unbroken forests, largely due to a lack of adequate forage. Elk can be found in coniferous swamps, clear cuts, aspen-hardwood forests, and coniferous-hardwood forests. They are found over a wide range of elevations. In our project area, they occur from sea level to nearly 3000 m, with the highest elevations occupied seasonally, when snowpack allows.

Dispersal

Seasonal movement scale was based on studies listed in Table A.18. Telemetry studies of the migratory elk herd near Yakima (S. McCorquodale, personal communication) indicate that the average distance between winter and summer home ranges is about 30 km. Most of the elk near Yakima had winter and summer activity centers that were separated by ~25–40 km. Maximum distances between winter and summer activity centers were in the range of 70–80 km. Elk are known to move as much as 100 km between seasonally important habitats (Boyce 1991).

Table A.18. Long range elk movements or dispersal distances.

Description	*Distance*	*Citation*
Seasonal migratory movements	25–40 km (max 70–80 km)	S. McCorquodale, personal communication
Seasonal migratory movements	100 km	Boyce 1991
Dispersal of translocated females	10–110 km (most 10–30 km)	Fryxell et al. 2008

Modeling Efforts

Many elk habitat association models have been developed. In addition, elk distribution and seasonal ranges within Washington and adjacent jurisdictions have been mapped using empirical data and expert knowledge (e.g., Rocky Mountain Elk Foundation, Washington Department of Fish and Wildlife's Priority Habitats and Species, B. C. Ministry of Environment). So, there is considerable information available that can be applied to identifying HCAs in the project area. However, there are no known attempts at modeling landscape permeability for elk in our project area.

Outside of our project area, there are at least two modeling efforts that attempt to identify elements of landscape permeability. Beier and Majka (2007) applied habitat suitability scores in their development of a least-cost distance model to define biologically best corridors for elk in Arizona. Their scoring of landscape suitability for movements used the same parameter values as would be appropriate for identifying core or source habitats. Walker and Craighead (1997) developed a least-cost distance formula for elk that utilized three parameters applied to a raster grid of one square kilometer cells. The three parameters were habitat quality (based on vegetative cover and scored from 1 to 3), total length of forest/shrub-land and forest/grassland edge, and road density.

Conceptual Basis for Statewide Connectivity Model Development

OVERVIEW

Elk HCAs were largely identified based on vegetative cover conditions that indicated adequate forage and cover within the typical daily movement range of an individual elk. We defined habitat as having a minimum of at least 25% of area in forage and 25% in cover within a 500 m radius. We applied a GIS moving window of 2.25 km radius to identify areas with at least 70%

suitable habitat conditions. This model identified suitable areas outside of the primary range of elk or in areas where other landscape features, i.e., high human population density or busy roads, limit elk use. Therefore, polygons representing the outer bounds of elk range were used to constrain outputs from the habitat model. Highway corridors and areas of human population density greater than one dwelling unit per 40 ac (based on 2000 census data) were also removed from the map of concentrated habitat areas.

To characterize the suitability of the landscape for elk movements, resistance parameters were developed from descriptions of optimal elk habitat conditions and features of the landscape that are avoided. While there is ample information on elk habitat associations and preference, there is little published information on conditions suitable for elk movements, with the exception of research in Arizona to determine the barrier effect of highways (Dodd et al. 2007a; Dodd et al. 2007b; Gagnon et al. 2007). Despite this lack of published information, the ungulate biologists queried for opinions often related observations of elk moving through a wide variety of landscape conditions that could not be considered as suitable elk habitat.

Habitat suitability based on documented habitat associations was used to assign low resistance scores. Since road avoidance is a recurring theme in the elk literature, this aspect of elk behavior was built into the model. Road effects distances vary considerably in the published literature. Research has noted a measurable decline in elk habitat use up to 1.8 km from roads (Rowland et al. 2000). Elk have additionally been documented avoiding roads at distances of 400 m (Ward et al. 1980), 800 m (Lyon 1979), and 2.2 km (Brown et al. 1980; Rowland et al. 2004). Scoring resistance for landscape attributes that fell short of documented preferred conditions was based on professional judgment with the knowledge that elk will move through a wide variety of conditions that offer little or nothing in the form of security cover or forage.

MOVEMENT DISTANCE

We used a conservative movement distance of 500 m to identify forage and cover conditions in adequate proximity to be considered habitat for elk. To identify concentrations of suitable habitat, we used an estimated average home range radius of 2.25 km. Table A.19 provides documented elk movements reflective of daily movement scales.

Table A.19. Daily elk movements or distances between forage and cover patches.

Description	*Distance*	*Citation*
Distance between successive daily locations	720 m	Vore & Schmidt 2001
Daily movement scale	2.4 km (mean)	Bowyer 1981
Daily movements between patches	1.6 km (mean)	Frair et al. 2005
Distance between successive daily locations	720 m	Vore & Schmidt 2001

Landscape Features and Resistance Values

The resistance surface for elk (Table A.20) is based on the following parameterized spatial data layers:

1) Land cover/land-use
2) Elevation
3) Slope
4) Housing density
5) Road type and distance

Habitat Concentration Areas

General Definition — Habitat concentration areas (HCAs) are core habitat areas that are expected or known to be important habitat for focal species. These areas are used as the focus locations for running dispersal models that identify connectivity pathways.

Technical Definition — Habitat concentration areas are aggregations of habitat grid cells that are connected to each other by a species-specific home range movement radius. These aggregations must also meet a minimum size requirement designed to select only those areas that are capable of supporting multiple individuals. Connectivity between HCAs will be evaluated based on a species-specific dispersal distance and a landscape resistance model.

For elk, HCAs were identified using a combination empirical knowledge of distribution and a habitat identification model. Steps used in the habitat identification model were as follows:

1) Develop a binary habitat surface where each grid cell in a raster is designated as either habitat or non-habitat. Habitat grid cells had at least of 25% of the area within a 500 m radius in forage (open or sparse forest, alpine, grass, shrub, or wetland) and at least 25% of the same area, in cover (dense forest). A 500 m radius circle represented the distance at which both cover and forage conditions were within a typical daily movement range.

2) Calculate the proportion of habitat within a circular moving window of a size representative of each species' home range radius. A home range radius of 2.25 km was used based on estimates found in elk movement studies (Table A.19). This step generates a surface which identifies the areas where the habitat is most concentrated.

3) Delete habitat grid cells in areas where habitat is sparse. Habitat grid cells were removed from the binary habitat raster if the proportion of habitat within a home range radius was <0.5. This prevented habitat concentrations from forming in areas where habitat is not concentrated to the degree required to be considered core habitat.

4) Join remaining habitat grid cells together if they are within a home range movement distance. Habitat areas were expanded outwards (from the remaining habitat grid cells

after step 3) up to a total cost-weighted distance equal to the species' home range movement radius (2.25 km). This had the effect of joining nearby habitat grid cells together if the intervening landscape supports within-home range connectivity.

5) Eliminate small patches unlikely to contribute significantly to a species' core habitat. Habitat patches less than 100 km^2 were eliminated because they were below the minimum size requirement for large herbivores.

The habitat areas identified by the above model were further refined using known elk range. To accomplish this, we clipped the habitat map using elk range polygons from the Rocky Mountain Elk Foundation's 1999 mapping effort spanning the western United States. These polygons represented ranges of elk during all seasons, reduced by the exclusion of areas of moderate to high human population densities and the immediate vicinities of freeways and high-use roads.

Linkage Modeling

Linkages were modeled when the cost-weighted distance between a pair of HCAs was less than 250 km.

Considerations for Future Modeling

At more local scales, greater consideration should be given to connections between seasonal ranges and shifting distributions of croplands and human populations in response to a changing climate.

Table A.20. Landscape features and resistance values used to model habitat connectivity for elk.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	20
urban/developed	100
water	20
sparsely vegetated	5
alpine	0
riparian	0
wetland	0
grass-dominated	0
shrub-dominated	0
dry forest	0
wet forest	0
Elevation (meters)	
0–250	0
> 250–500	0
> 500–750	0
> 750–1000	0
> 1000–1500	0
> 1500–2000	0
> 2000–2500	0
> 2500–3300	25
> 3300 meters	25
Slope (degrees)	
0–20	0
> 20–40	0
> 40	30
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	0
> 20 ≤ 40	5
> 10 ≤ 20	10
< 10	20
Road type and distance (meters)*	
freeway > 500–1000 buffer	10
freeway > 0–500 buffer	30
freeway centerline	100
major highway > 500–1000 buffer	5
major highway > 0–500 buffer	10
major highway centerline	80
secondary highway > 500–1000 buffer	3
secondary highway > 0–500 buffer	8
secondary highway centerline	50
local road > 500–1000 buffer	0
local road > 0–500 buffer	1
local road centerline	5

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Acknowledgements

This work would have suffered had it not been for the assistance of B. Johnson (ODFW), H. Zahn (WDFW), S. McCorquodale (WDFW), J. Nelson (WDFW), D. Ware (WDFW), T. Clausing (WDFW), J. Bernatowicz (WDFW), J. Stephenson (YNW), P. Singleton (USFS), A. Shirk (Independent Researcher), D. Reynolds (BCME), S. Fitkin (WDFW), A. Reid (BCME), M.

Wood (ODFW), J. Germond (ODFW), G. Servheen (IDFG), T. Hamilton (BCME), M. Knapik (BCME), T. Szkorupa (BCME), W Myers (WDFW), and H. Ferguson (WDFW).

Literature Cited

Beier, P., and D. Majka. 2007. Arizona Missing Linkages – Hualapai – Peacock Linkage Design. Northern Arizona University, Flagstaff, Arizona.

Bowyer, R. T. 1981. Activity, movement, and distribution of Roosevelt elk during rut. Journal of Mammalogy 62:574–582.

Boyce, M. S. 1991. Migratory behavior and management of elk (Cervus elaphus). Applied Animal Behavior Science 29:239–250.

Brown, G. et al. 1980. Elk habitat: timber management relations. Central Zone, Northern Region. USDA Forest Service, Montana Department of Fish, Wildlife and Parks, Confederated Salish and Kootenai Tribes.

Dodd, N. L., J. W. Gagnon, A. L. Manzo, and R. E. Schweinsburg. 2007b. Video surveillance to assess highway underpass use by elk in Arizona. Journal of Wildlife Management 71:637–645.

Dodd, N. L., J. W. Gagnon, S. Boe, and R. E. Schwenburg. 2007a. Assessment of elk highway permeability by using global positioning system telemetry. Journal of Wildlife Management 71:1107–1117.

Frair, J. L., E. H. Merrill, D. R. Visscher, D. Fortin, H. L. Beyer, and J. M. Morales. 2005. Scales of movement by elk (*Cervus elaphus*) in response to heterogeneity in forage resources and predation risk. Landscape Ecology 20:273–283.

Fryxell, J. M., M. Hazell, L. Borger, B. D. Dalziel, D. T. Haydon, J. M. Morales, T. McIntosh, and R. C. Rosatte. 2008. Multiple movement modes by large herbivores at multiple spatiotemporal scales. Proceedings of the National Academy of Sciences 105:19114–19119.

Gagnon, J. W., R. E. Schweinsburg, and N. L. Dodd. 2007. Effects of roadway traffic on wild ungulates: a review of the literature and case studies of elk in Arizona. Pages 449–458 in C. L. Irwin, D. Nelson, and K. P. McDermott, editors. Proceedings of the 2007 International Conference of Ecology & Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, North Carolina.

Geist, V. 2002. Adaptive and behavioral strategies. Pages 389–433 in D. E. Toweill and J. W. Thomas, editors. North American elk: Ecology and Management. Smithsonian Institute Press Washington DC.

Lyon, L. J. 1979. Habitat effectiveness for elk as influenced by roads and cover. Journal of Forestry 77:658–660.

Lyon, L. J., and A. G. Christensen. 2002. Elk and land management. Pages 557–581 in D. E. Toweill and J. W. Thomas, editors. North American elk: Ecology and management. Smithsonian Institution Press, Washington DC.

Morgantini, L. E., and R. J. Hudson. 1979. Human disturbance and habitat selection in elk. Pages 132–139 in M. S. Boyce and L. D. Hayden-Wing, editors. North American elk: Ecology, Behavior and Management. University of Wyoming, Laramie, Wyoming.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and J. G. Kie. 2000. Elk distribution and modeling in relation to roads. Journal of Wildlife Management 64:672–684.

Rowland, M. M., M. J. Wisdom, B. K. Johnson, and M. A. Penninger. 2004. Effects of roads on elk: implications for management in forested ecosystems. Transactions of the 69th North American Wildlife and Natural Resources Conference,

Vore, J. M. and E. M. Schmidt. 2001. Movements of female elk during calving season in northwest Montana. Wildlife Society Bulletin 29:720–725.

Walker, R., and F. L. Craighead. 1997. Analyzing wildlife movement corridors in Montana using GIS. Paper presented at: 1997 ESRI International User Conference, San Diego, California.

Ward, A. L., N. E. Fornwalt, S. E. Henry, and R. A. Hodorff. 1980. Effects of highway operation practices and facilities on elk, mule deer, and pronghorn antelope. Report FHWA-RD-79-143. Rocky Mountain Forest and Range Experiment Station, Ft. Collins, Colorado.

Personal Communications

S. McCorquodale, Ungulate Specialist
Washington Department of Fish and Wildlife, Wildlife Program
Olympia, Washington

Northern Flying Squirrel (*Glaucomys sabrinus*)

Prepared by Chris Sato (Washington Department of Fish and Wildlife)

Justification for Selection

The northern flying squirrel (*Glaucomys sabrinus*) was selected as a good representative of wildlife habitat connectivity needs within both Vancouverian and Rocky Mountain Forests. Flying squirrel abundance has been correlated with old-growth forest structure and processes, and the species requires certain habitat requirements for efficient movement. Flying squirrels are vulnerable to loss of habitat connectivity from all four of the main connectivity threats: land clearing and vegetation removal, development, roads and traffic, and people and domestic animals.

Distribution

Northern flying squirrels occur in all coniferous and mixed forests within their range. They are also likely to be found in pure deciduous forests, such as pure oak stands (Bayrakci et al. 2001). They do not occur in the San Juan Islands, or Guemes, Cypress, or Lummi Islands, and are not found in isolated conifer stands in the Palouse. They are present in several cities including Walla Walla, Seattle, Dayton, and Tacoma (Johnson & Cassidy 1997).

Habitat Associations

The northern flying squirrel is associated with several features of late-seral forest structure, and thus is a good indicator of forest ecosystem health (Carey et al. 1999; Carey 2000). It is generally affiliated with mature and old-growth forest types and associated structural components. Across its range, abundance is correlated with large-diameter trees, large snags, coarse woody debris (particularly decayed logs), and fungi (Carey et al. 1999; Smith 2007). These associations may have more to do with the resources available in a structurally complex forest than a specific need for these individual components (Wilson 2010).

Along the West Coast, northern flying squirrel abundance generally increases with stand age, with local variations (Carey 2000). Forests that support high densities of northern flying squirrels are generally characterized as having dense multi-layered mid and over-story canopies, low to moderate amounts of understory, and few canopy gaps (Wilson 2010). These characteristics are typically found in mature and old-growth forests but can also be found in some young forests (Rosenberg & Anthony 1992; Buck & Woodworth 2008). Several studies have shown that northern flying squirrels are more abundant in second-growth stands in which legacy trees, coarse woody debris and snags were retained during the prior harvest. These second-growth stands were more structurally complex than second-growth stands lacking these components (Carey et al. 1992; Witt 1992; Carey 1995; Carey 2000). Flying squirrel density in young thinned stands in the Oregon Coast Range was positively correlated with the density of large logs and conifers exceeding 50 cm in diameter at breast height (dbh; Gomez et al. 2005).

On the east slope of the Washington Cascades, northern flying squirrels were found at high densities only in stands with greater than 55% canopy cover. Squirrels preferred mixed-conifer forest over ponderosa pine forests (Lehmkuhl et al. 2006). Hypogeous sporocarps (fungi) comprise much of the diet of northern flying squirrels. Species richness and biomass of hypogeous sporocarps, were correlated with increased coarse woody debris cover (Lehmkuhl et al. 2004). Survival of northern flying squirrels has been strongly linked to understory species richness and forage lichen biomass (Lehmkuhl et al. 2006). At the stand level, northern flying squirrels were associated with decadence, especially coarse woody debris, and less so with foliage-height diversity. At the point level, decadence was the most dominant predictor, with canopy stratification next (Carey et al. 1999). In northern California, northern flying squirrels were found in 60 to 80% of stands exceeding 57 ha, and in less than 10% of stands smaller than 7 ha (Rosenberg & Raphael 1986). Forests that supported high squirrel abundance generally exhibited high amounts of multi-dimensional structure in the mid-story and over-story layers, low to moderate amounts of understory, and few canopy gaps (Wilson 2010).

Dispersal

For an arboreal species such as the northern flying squirrel, home range size may not be the best reflection of actual habitat use. For example, home range estimates only reflect movement within a two-dimensional horizontal plane, yet flying squirrels use all of the multiple dimensions of a forest, from denning in substrates that can be high in the canopy of trees, to gliding between trees from the canopy to the forest floor, to digging truffles on the forest floor (Wilson 2010). The parameters used in the statewide connectivity project do not allow for these distinctions.

Most northern flying squirrel space-use studies of have been based on estimating home ranges using periodic triangulation of radio-collared squirrels (Mowrey & Zasada 1984; Weigl et al. 1992; Witt 1992; Martin & Anthony 1999; Menzel et al. 2006; Holloway & Malcolm 2007). Reported home range estimates have varied widely, with males generally having larger home ranges than females (Witt 1992; Martin & Anthony 1999; Menzel et al. 2006; Holloway & Malcolm 2007). In the Pacific Northwest, home range size was found similar between two old-growth forests and two second-growth forests in the western Oregon Cascades where population densities were similar (Martin & Anthony 1999; Wilson 2010).

Home ranges in Pacific Northwest studies tend to vary depending on forest structure and composition; from 2.5 ha to 4.6 ha (Lehmkuhl et al. 2006), and 4.1 ha to 5.8 ha (Martin & Anthony 1999). The largest home range found for males was 11.2 ha in unlogged coniferous forest in Ontario, Canada (Holloway & Malcolm 2007). Holloway and Malcolm (2007) determined a home range overlap for males of 61% and 74% for females.

In spring and early summer, males greatly expand their area of use and make wide forays in search of females in estrus (Wilson 2010). Circuit distances of up to 5 km have been reported in the Pacific Northwest (Wilson 2010; Smith et al. in prep).

Juvenile dispersal occurs in early fall—September and October (Pyare & Smith 2005). Northern flying squirrel dispersal rates and distances may depend on population densities in a given source site and on habitat quality, however there are few studies that address this topic. T. Wilson (personal communication) suggests that dispersal may be more prevalent in complex, high-quality habitat where there is increased intraspecific competition for resources and more social interaction than in low-density forest. In high-quality habitat, it may benefit juveniles to disperse rather than to face high density-dependent pressures at their birth sites, whereas squirrels have very little motivation to move in lower density, self-sustaining populations. He adds that corridor width and degree of structural occlusion (e.g., dense multi-layered canopy) may be key factors in how far and how safely an individual can move through corridors.

Research using radio telemetry on a closely related species in Europe, Siberian flying squirrels (*Pteromys volans*), has shown that this species uses corridors and matrix habitats to move upward of 400 m between habitat fragments during dispersal events in which squirrels disperse a total distance of up to approximately 8 km (Selonen & Hanski 2004). According to a 2003 paper by the same authors, Siberian flying squirrel juveniles dispersed over long distances in fragmented forest landscapes up to a mean of 5.5 km. The authors found that there was a very clear directional bias in the dispersal path, nearly a straight line over a broad scale; but wide-open areas obstructed the straightness of the path. In a study conducted in Alaska, northern flying squirrel juveniles dispersing in a landscape of complex old growth islands in the Tongass National Forest moved from 0.8 km to 1.1 km. Other juveniles in the same study readily crossed two-lane roads, moving between 1 and 2 km (Smith et al. in prep). During a pilot/feasibility study conducted by Smith et al. (in prep), one juvenile moved about 7 km in 48 hours.

Modeling Efforts

Habitat models are available for the northern flying squirrel, in part because it is the primary prey for Northern Spotted Owls (*Strix occidentalis*) throughout much of their range. Ritchie et al. (2009) assessed the independent effects of landscape composition and configuration on the occurrence of northern flying squirrels. Buck and Woodworth (2008) created a Bayesian model using six input nodes (average tree dbh, snag density, canopy cover, coarse woody debris cover, shrub cover and patch size), Lehmkuhl et al. (2006) quantified habitat characteristics and squirrel density, population trends, and demography in three common forest cover types in eastern Washington, and in separate studies, Carey et al. (1999) and Wilson (2010) focused on the structural complexity of western Washington forests.

There have been few efforts to assess habitat connectivity for the northern flying squirrel. Smith et al. (in prep) evaluated data from telemetry experiments to determine effective distances moved by dispersers through landscapes composed of old-growth and managed forests. Pyare and Smith (2005) created a GIS-based model of movement capability to evaluate effective distance traveled and movements in fragmented landscapes. Both studies were conducted in medium to high quality old growth habitat in the Tongass National Forest in Alaska. In Finland, Selonen and Hanski (2003, 2004) studied dispersal of juvenile Siberian Flying Squirrels by looking at the effects of landscape structure on selected dispersal direction, distance, and straightness of dispersal path.

Conceptual Basis for Statewide Connectivity Model Development

MOVEMENT DISTANCE

Northern flying squirrels can be considered linkage dwellers. Connectivity is frequently accomplished as a slow, multi-generational progression over the landscape. Because of this, considering movement over a temporal as well as spatial scale is more practical than focusing on an individual’s capacity to move. Squirrel genes, carried by multiple generations of squirrels, travel much further than any single individual. It is therefore more appropriate to consider patch connectivity over a period of 10 to 20 years or more rather than the lifetime, or any portion of the life span, of any individual squirrel.

Landscape Features and Resistance Values

The resistance surface is based on the following parameterized spatial data layers:

1) Housing density
2) Road use type and distance
3) Forest structure

Values for these factors are provided in Table A.21.

Habitat Concentration Areas

General definition — Habitat concentration areas (HCAs) are core habitat areas that are expected or known to be important habitat for focal species. These areas are used as the focus locations for running dispersal models that identify connectivity pathways.

Technical definition — Habitat concentration areas are aggregations of habitat grid cells that are connected to each other by a species-specific home range diameter. These aggregations must also meet a minimum size requirement designed to select only those areas that are capable of supporting multiple individuals. Connectivity between HCAs will be evaluated based on a species-specific dispersal distance and a landscape resistance model.

These are the steps we used to identify the northern flying squirrel HCAs.

1) Develop a binary habitat surface where each grid cell in a raster is designated as either habitat or non-habitat. We defined northern flying squirrel habitat as any grid cell with a resistance value of 1.

2) From that binary habitat map, we passed a moving window sized to a typical home range radius of 2 km over the landscape and masked out habitat in areas where it was less than 10% of the window (this eliminates habitat in areas where it is sparse). We chose the 10% threshold to capture major concentrations of habitat while still leaving room in the landscape to model corridors.

3) Join remaining habitat cells together if they are within a home range movement distance. We expanded the designated habitat area outwards (from the remaining habitat grid cells after step 3) up to a total cost-weighted distance equal to the northern flying squirrel's home range movement radius (2 km). This has the effect of joining nearby habitat grid cells together if the intervening landscape supports within-home range connectivity.

4) Eliminate small patches unlikely to contribute significantly to a species core habitat. We standardized HCA size to >50 km^2 to make it relevant at a statewide scale.

Linkage Modeling

The maximum distance chosen for linkage modeling is 126 km. This distance provides a best-fit model based on cost-weighted corridor maps, as well as Washington GAP distribution data (Johnson & Cassidy 1997). Recorded dispersal distances for the flying squirrel do not provide an accurate depiction of actual movement across the landscape. Dispersal is a slow, multi-generational process. Because of this, considering movement over a temporal as well as spatial scale is more practical than focusing on an individual's capacity to move. When considering forest dynamics, natural as well as human influenced, choosing a large maximum distance allows for gradual movement through a constantly changing landscape. The maximum distance of 126 km takes into account slow, multigenerational dispersal over fragmented landscapes and provides some flexibility until better mapping products for the northern flying squirrel are available.

Considerations for Future Modeling

Parameters used in this modeling exercise were not specific enough to accurately depict the somewhat complex within-stand habitat requirements of northern flying squirrels. In addition, updated distribution data are needed. Future modeling at the ecoregional scale should: (1) apply more detailed data reflecting structural and forest type and seral stage, (2) use updated GIS habitat layers, (3) consider land ownership, and (4) consider effects of dynamic changes on the landscape (i.e., ongoing forestry activity).

Table A.21. Landscape features and resistance values used to model habitat connectivity for northern flying squirrels.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	0
urban/developed	0
water	0
sparsely vegetated	0
alpine	0
riparian	0
wetland	0
grass-dominated	0
shrub-dominated	0
dry forest	0
wet forest	0
Elevation (meters)	
0–250	0
> 250–500	0
> 500–750	0
> 750–1000	0
> 1000–1500	0
> 1500–2000	0
> 2000–2500	0
> 2500–3300	0
> 3300 meters	0
Slope (degrees)	
0–20	0
> 20–40	0
> 40	0
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	0
> 20 ≤ 40	5
> 10 ≤ 20	50
< 10	100
Road type and distance (meters)*	
freeway > 500–1000 buffer	0
freeway > 0–500 buffer	0
freeway centerline	1000
major highway > 500–1000 buffer	0
major highway > 0–500 buffer	0
major highway centerline	500
secondary highway > 500–1000 buffer	0
secondary highway > 0–500 buffer	0
secondary highway centerline	10
local road > 500–1000 buffer	0
local road > 0–500 buffer	0
local road centerline	0
Forest structure (density and height [meters])	
nonforest	100
sparse low (0–40%, ≤ 25)	20
sparse high (0–40%, > 25)	20
open low (>40–70%, ≤ 25)	5
open high (>40–70%, > 25)	5
dense low (>70–100%, ≤ 25)	0
dense high (>70–100%, > 25)	0

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Acknowledgements

Brian Cosentino (WDFW), Andrew Shirk (Independent Researcher), Kelly McAllister (WSDOT), Peter Singleton (USFS), Brad McRae (TNC), and Albert Perez (WSDOT) provided assistance with model development. Brian Cosentino ran the northern flying squirrel resistance surface, and Andrew Shirk developed the northern flying squirrel HCA methods. In addition, the following persons provided model review: Todd Wilson (USFS PNW), Winston Smith (USFS PNW), and John Lehmkuhl (USFS).

Literature Cited

Bayrakci, R., A. B. Carey, and T. M. Wilson. 2001. Current status of the western gray squirrel population in the Puget Trough, Washington. Northwest Science 75:333–341.

Buck, W., and D. R. Woodworth. 2008 (draft). Bayesian Belief Network Model: upland mature forest associations in forests of Washington. Washington landscape-level wildlife assessment project. Biota Pacific Environmental Sciences, Bothell, Washington.

Carey, A. B. 1995. Sciurids in Pacific Northwest managed and old-growth forests. Ecological Applications 5:648–661.

Carey, A. B. 2000. Effects of new forest management strategies on squirrel populations. Ecological Applications 10:248–257.

Carey, A. B., J. Kershner, B. Siswell, and L. Dominguez de Toledo. 1999. Ecological scale and forest development: squirrels, dietary fungi, and vascular plants in managed and unmanaged forests. Wildlife Monographs 142:1–71.

Carey, A. B., S. P. Horton, and B. L. Biswell. 1992. Northern spotted owls: influence of prey base and landscape character. Ecological Monographs 62:223–250.

Gomez, D. M., R. G. Anthony, and J. P. Hayes. 2005. Influence of thinning of Douglas-fir forests on population parameters and diet of northern flying squirrels. Journal of Wildlife Management 69:1670–1682.

Holloway, G. L., and J. R. Malcolm. 2006. Sciurid habitat relationships in forests managed under selection and shelterwood silviculture in Ontario. Journal of Wildlife Management 70:1735–1745.

Holloway, G. L., and J. R. Malcolm. 2007. Northern and southern flying squirrels use of space within home ranges in central Ontario. Forest Ecology and Management 242:74–755.

Johnson, R. E., and K. M. Cassidy. 1997. Terrestrial mammals of Washington State: location data and predicted distributions. Volume 3 in K. M. Cassidy, C. E. Grue, M. R. Smith and K. M. Dvornich, editors. Washington State Gap Analysis – Final Report. Washington Cooperative Fish and Wildlife Research Unit, University of Washington, Seattle.

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

Lehmkuhl, J. F., L. E. Gould, E. Cázares, D. R. Hosford. 2004. Truffle abundance and mycophagy by northern flying squirrels in eastern Washington forests. Forest Ecology and Management 200:49–65.

Martin, K. J., and R. G. Anthony. 1999. Movements of northern flying squirrels in different-aged forest stands of western Oregon. Journal of Wildlife Management 63:291–297.

Menzel, J. M., W. M. Ford, J. W. Edwards, and T. M. Terry. 2006. Home range and habitat use of the vulnerable Virginia northern flying squirrel *Glaucomys sabrinus fuscus* in the central Appalachian Mountains, U.S.A. Oryx 40:204–210.

Mowrey, R. A., and J. C. Zasada. 1984. Den tree use and movements of northern flying squirrels in interior Alaska and implications for forest management. Pages 351–356 in W. R. Meehan, T. R. Merrell, and T. A. Hanley, editors. Fish and Wildlife Relationships in Old-growth Forests: Proceedings of a Symposium. American Institute of Fishery Research Biologists, Moorhead City, North Carolina.

Pyare, S., and W. P. Smith. 2005. Functional connectivity of Tongass Old-Growth Reserves: an assessment based on flying squirrel movement capability. Progress Report for Tongass Monitoring Grants.

Ritchie, L. E., M. G. Betts, G. Forbes, and K. Vernes. 2009. Effects of landscape composition and configuration on northern flying squirrels in a forest mosaic. Forest Ecology and Management 257:1920–1929.

Rosenberg, K. D., and R. G. Anthony. 1992. Characteristics of northern flying squirrel populations in young second - and old-growth forests in western Oregon. Canadian Journal of Zoology 70:161–166.

Rosenberg, K. V., and M. G. Raphael. 1986. Effects of forest fragmentation on vertebrates in Douglas-fir forests. Pages 263–272 in J. Verner, M. L. Morrison, and C. J. Ralph, editors. Wildlife 2000: Modeling habitat relationships of terrestrial vertebrates. University of Wisconsin Press, Madison, Wisconsin.

Selonen, V., and I. K. Hanski. 2003. Movements of the flying squirrel *Pteromys volans* in corridors and in matrix habitat. Ecography 26:641–651.

Selonen,V., and I. K. Hanski. 2004. Young flying squirrels (*Pteromys volans*) dispersing in fragmented forests. Behavioral Ecology 15:564–571.

Smith, W. P. 2007. Ecology of *Glaucomys sabrinus*: habitat, demography, and community relations. Journal of Mammalogy 88:862–881.

Smith, W. P., D. K. Person, and S. Pyare. In review. Source-sinks, metapopulations, and forest reserves: conserving northern flying squirrels in the temperate rainforests of Southeast Alaska.

Weigl, P. D., T. W. Knowles, and A. C. Boynton. 1992. The distribution and ecology of the northern flying squirrel, *Glaucomys sabrinus coloratus*, in the Southern Appalachians. Rep. 2. [Place of publication unknown]: North Carolina Wildlife Resources Commission, Morgantown, North Carolina.

Wilson, T. M. 2010. Limiting factors for northern flying squirrels (*Glaucomys sabrinus*) in the Pacific Northwest; a spatio-temporal analysis. PhD dissertation. Union Institute and University.

Witt, J. W. 1992. Home range and density estimates for the northern flying squirrel, Glaucomys sabrinus, in western Oregon. Journal of Mammalogy 73:921–929.

Personal Communications

T. M. Wilson, Research Scientist
USDA USFS PNW Research Station
Corvallis, Oregon

Western Toad (*Anaxyrus boreas*)

Prepared by Joanne P. Schuett-Hames (Washington Department of Fish and Wildlife)

Justification for Selection

Western toads (*Anaxyrus boreas*) were chosen as a representative focal species for the Vancouverian, Subalpine, and Rocky Mountain Forests vegetation classes. They additionally occur within Alpine and Semi-desert vegetation classes overall providing a broad representation function for much of Washington State.

Within the Vancouverian, Subalpine, and Rocky Mountain Forests the toads were scored “excellent” for: (1) representation of the vegetation class; (2) connectivity threats related to development, roads and traffic, and people and domestic animals; (3) information for modeling; (4) dispersal limitations; and (5) ability to be monitored. The toads were scored “acceptable” for the remaining selection criteria: the connectivity threat from land clearing/vegetation removal, and movement scale.

Although the scale of individual animal movement is less than that of most species chosen for the statewide analysis, the toad’s broad-coverage across the landscape, reliance on connectivity between populations, and in particular, its association with wetlands and aquatic systems led to inclusion in the statewide analysis.

In Washington State the western toad has a Candidate conservation status. It is a federal Species of Concern.

Distribution

Western toads range across much of western North America. They are found throughout the Pacific Northwest, north through western Canada and into the southeastern portion of Alaska (summarized by Muths & Nanjappa 2005). With the exception of the most arid portions of the Columbia Basin, western toads are found throughout Washington from near sea level to 1988 m (Leonard et al. 1993). They occur up to 2247 m in elevation throughout Oregon except for the northern Coast Range and most of the Willamette Valley (Leonard et al. 1993). The toads are found in Idaho up to 2865 m (Idaho Cooperative Fish and Wildlife Research Unit 2009).

This species can be locally common, however rapid losses and declines have been observed in numerous populations across its range (NatureServe 2009), including within Washington State. For example, K. Richter (personal communication) observed the disappearance of western toads following residential development of the East Lake Sammamish Plateau. Western toads are described in Leonard et al. (1993) as now uncommon in the lowlands of western Washington, and the mountain meadows of the North Cascades, for unknown reasons. In addition, recent surveys in Mt. Rainier National Park have documented western toad breeding at only 12 sites in stark contrast to past levels of abundance throughout the 300 lakes and ponds within the park (Adams no date).

Habitat Associations

During the year western toads move across the landscape accessing breeding areas, active season habitats, and over-wintering locations. They have been called a "linkage species" as these movements provide ecological connection between aquatic and terrestrial ecosystems (Pyare et al. 2005). These connections may often occur in the context of a positive association with landscape and ecological processes that include: (1) volcanic eruptions, (2) fire, (3) river dynamics that maintain off-channel habitats and alluvial channels and floodplains in first to third order channels, (4) snow and ice environments that provide open habitats in higher elevation third order and higher channels, (5) grazers that maintain open habitats, and (6) burrow creation by mammals such as gophers[1].

FRESHWATER HABITAT

Western toads use an array of freshwater habitats for breeding. These habitats are summarized as "…still or barely moving water, typically ponds and small lakes, streams, rain pools, and ditches…" (Muths & Nanjappa 2005), and as "…slow portions of streams, springs, ponds, lakes, reservoirs or stock ponds…" (Jones et al. 2005). More specifically, within Washington State breeding locations at Fort Lewis have occurred within National Wetland Inventory wetland types (Cowardin et al. 1979) of Lacustrine, Scrub-shrub edges, Emergent and Open Water wetlands (Lynch 2006). Western toad riverine breeding sites in the Satsop Basin of Washington are fairly specific to river sections within the Lincoln siltstone formation (Peterson 2004).

ACTIVE SEASON HABITAT

Post-breeding toads and toadlets leave the breeding site and move to active-season habitats. In northeast Oregon 81% of post-breeding toads studied in mountainous forested terrain were in terrestrial locations away from breeding sites, while the remaining 19% were in water away from breeding sites (Bull 2006). These toads selected forest habitats as follows: (1) with little to no canopy; (2) south facing slopes; (3) near water; and (4) with high densities of potential refugia such as burrows, rocks, and logs (Bull 2006). Western toads on Fort Lewis were found in forest and prairie ecosystems (Lynch 2006). In the forest they were often in association with either sword ferns (*Polystichum munitum*) or woody debris; in the prairie they were frequently found underground in Mazama pocket gopher (*Thomomys mazama*) and Townsend mole (*Scapanus townsendi*) burrows (Lynch 2006).

Western toads in British Columbia actively chose to enter clear cuts (≤5 ha in size) from neighboring forests; they also remained longer in clear cuts than in forests (Deguise 2007). This research was performed during May and June leading Deguise (2007) to speculate, "I believe clear cuts may be favourable during the early summer months only. Later in the season, this habitat could potentially be very inhospitable when temperatures are extremely high and there is very little precipitation, resulting in a significant risk of desiccation for toads…further research is

---

[1] *Based on a western toad model review meeting held in Olympia 17 August 2009, attended by: Saunders Freed (TNC), Marc Hayes (WDFW), Eric Lund (WDFW), Jim Lynch (Fort Lewis), Aimee MacIntyre (WDFW), Kelly McAllister (WDOT), John Richardson (Fort Lewis), and Joanne Schuett-Hames (WDFW).*

needed before any definitive conclusions can be made in terms of the suitability of clear cuts for western toads."

OVERWINTERING HABITAT

Bull (2006) studied overwintering habitat use by western toads in northeast Oregon. Of 26 toads, most overwintered in rodent burrows (38%), others overwintered under large rocks (27%), under logs or root wads (19%), and under banks adjacent to streams or lakes (15%).

SLOPE

Steep terrain does do not appear to be an impediment to western toad movements. Bull (2006) includes the following description of steep terrain traversed by toads "Some toads crossed rugged terrain to reach summer habitat; two females traveled 2.4 km down steep cliffs to the Imnaha River which was 530 m lower in elevation. Two toads climbed 200 m in elevation up rock hillsides with 60 percent slopes."

SENSITIVITY TO DEVELOPMENT

The western toad is categorized as a species with very high sensitivity to development (WDFW 2008). Species experts indicated persistence for this species is expected at ≤1 dwelling unit per 40 acres; persistence is indicated as unclear at 10 to 20 acres per dwelling unit. Although toads might be observed on occasion in areas with a development density of ≥1 dwelling unit per five acres, population persistence would not be expected (WDFW 2008). Western toad vulnerabilities to development are noted in WDFW (2008) as follows: (1) habitat configuration is an important variable as western toad breeding and active habitats differ, (2) the extensive movement scale of this species necessitates considerations for habitat connectivity, and (3) this species is extremely sensitive to road mortality.

Similarly, Ovaska et al. (2004) indicate toads are compatible with rural lands; however overall compatibility of western toads with development in British Columbia is low to moderate. Western toad recommendations are noted as follows, "Reduce road mortality during mass migrations" and provide "access to extensive terrestrial habitat…" (Ovaska et al. 2004).

SENSITIVITY TO TRAFFIC

Western toads are considered highly vulnerable to traffic mortality (WDFW 2008) as they are slow moving and travel relatively long distances. Lynch (2006) provides these descriptions of western toad road mortality at Fort Lewis, "...in one instance, nine adult toads were found dead on a 1 km stretch of road near Fiander Lake at the onset of the breeding season" and, "Road killed adult toads are frequently seen individually on roads within the Rainier Training Area, whereas recent metamorphs can be found dead in large numbers when vehicles travel through a mass of them that are still piled together near a breeding site."

Road mortality rates have not been reported for western toads. However, published information for the common toad (*Bufo bufo*) indicates that even low levels of traffic may be a concern. Hels & Buchwald (2001) estimate that the probability of amphibians being killed on a secondary road (134 vehicles/hour) is between 34 and 61%, and on a busy highway (625 vehicles/hour) between 89 and 98%. Kuhn (1987) reported that 24–40 vehicles/hour is sufficient to kill 50% of migrating common toads, while Heine (1987) calculated a 0% survival rate for common toads when traffic volume reached 26 vehicles/hour. van Gelder (1973) estimated 29% of common toad females were killed over one year while crossing a paved road that had an estimated traffic volume of 14 vehicles/hour.

Complicating the issue of road mortality for western toads is the tendency for toads to remain on roads they encounter rather than moving across them. While studying western toads in managed forest landscapes in British Columbia, Deguise (2007) found "Every toad that encountered a road would stay on or near the road, where they would either bask along its edges and burrow in the ditches, or use the roads to move long distances. The longest movement bouts recorded during this study occurred on roads, including one individual that moved 982 m in approximately 24 hours." Deguise (2007) concluded that forest roads facilitated movement, and that abandoned logging roads may be beneficial as movement corridors for the toads by potentially encouraging dispersal and helping to maintain connectivity in the landscape.

Dispersal

Dispersal, "…demographic and genetic exchange among populations, as opposed to migration, which is annual upland movement within a population" (based on Compton et al. 2007) is poorly documented for western toads. Western toads become sexually mature at 2 to 4 years (Jones et al. 2005). There appear to be no studies that have followed dispersing toads throughout this interval, thus I have inferred potential dispersal distances from adult toad movement data (Table A.22.).

Table A.22. Movement distances of adult western toads in the Pacific Northwest.

Mean distance traveled (km)		*Maximum distance traveled (km)*		*State*	*Citation*
		5.3	(male; 5 months)	WA	Lynch 2006
2.5	(female; summer)	6.2	(female; summer)	OR	Bull 2006
1.0	(male; summer)	3.9	(male; summer)	OR	Bull 2006
2.0	(mean distance of winter hibernacula from breeding sites)			OR	Bull 2006
0.6	(males; distance traveled from breeding site)	1.0	(males; greatest distance from breeding site)	ID	Bartelt et al. 2004
1.1	(females; distance traveled from breeding site)	2.4	(females; greatest distance from breeding site)	ID	Bartelt et al. 2004
2.9	(median distance moved; summer; <1 mo.)	13.0	(summer; <1 mo.)	MT	Schmetterling & Young 2008
		2.0	(male; 19 days; post-breeding)	BC	Deguise 2007
		1.0	(male; ca. 24 hrs)	BC	Deguise 2007

Mean or median western toad movement distances in the Pacific Northwest range from 0.6 km to 2.9 km; maximum travel distances range from 1.0 km to 13.0 km (Bartelt et al. 2004; Deguise 2007; Schmetterling & Young 2008). These results are largely based on research that tracked toad movements from breeding sites; little is known about movements of western toads throughout their full yearly cycle.

Modeling that predicts toad population persistence may also be of use for considering dispersal distance. Although not available for western toads, modeling for common toads indicates populations in initially unoccupied ponds would have >95% certainty of persistence if they were located within 4.0 km of a source pond that could support >50 adult females (Halley et al. 1996).

Modeling Efforts

Modeling efforts for western toads as well as other anurans provide support to the design of this model. Washington GAP (Dvornich et al. 1997) modeled western toads as follows, "All ecoregions throughout Washington were selected. In western Washington, all zones up to and including Silver Fir were core zones. Mountain Hemlock and Alpine/Parkland zones were marginal. In eastern Washington, the steppe zones, Ponderosa Pine, Mountain Hemlock and Alpine/Parkland were marginal. The rest of the forested zones were core. Good habitats were open water, rivers and riparian areas. All non-forested and forested habitats were considered suitable if appropriate microhabitats existed."

MacCracken (2008) developed a Bayesian belief network model for western toads and other pond-breeding amphibians in Washington State. General premises of this model are: (1) forested stands are generally higher quality habitat than agricultural areas, which are better than urban areas; (2) maximum distance from water (breeding sites) influences use of the surrounding upland stands; and (3) wetland buffer regulations provide varying degrees of habitat protection. The maximum distance from breeding sites was set at 5 km for the long distance moving amphibians based on maximum movements of northern red-legged frogs (*Rana aurora*) and western toads.

Models non-specific to western toads additionally are useful. Ray et al. (2002) modeled landscape connectivity using cost-distance GIS analysis to predict occurrence of the common toad in western Switzerland. Importantly, the area of a migration zone and the number of ponds within the zone were positively related to common toad presence. Significant models were found utilizing both 1500 m and 3000 m migration zones; the best predictive model of toad presence used number of colonized ponds, sparse vegetation area, migration zone area, and cultivated fields area. Similar methodologies have been used to model northern red-legged frog habitat connectivity in King County, Washington (Mathias 2008).

Conceptual Basis for Statewide Connectivity Model Development

Western toads are found in many habitats from sea-level to subalpine areas and have life cycles that involve extensive adult and juvenile seasonal movements between lentic breeding areas and forested or open summer and winter habitats. Road traffic, human population density, and urban land-use are top factors impacting landscape permeability for this species.

Populations of pond-breeding amphibians such as the western toad operate at multiple scales: (1) the individual breeding pool or stream, (2) the breeding pool or stream with surrounding upland habitat, (3) neighboring breeding locations and upland habitat, and (4) clusters of neighboring populations in a regional framework where the focus is on long-term connectivity of metapopulations at a regional scale (after Compton et al. 2007). The latter scale is the focus for this statewide modeling effort.

Movement Distance

Based on consideration of movement distances presented in Table A.22, other information contained in this report, and informal sensitivity analyses, the statewide connectivity model used a 2 km distance to link breeding habitat to upland habitat. For our cost-weighted distance map, we used weighted distances outside of habitat concentration areas in categories of 1–5 km, 6–10 km, 11–15 km, and 16–20 km to provide an indication of additional habitat that may be available within reasonable movement distances for the western toad. These distance categories provide a variety of opportunities to evaluate connections between modeled neighboring populations.

Landscape Features and Resistance Values

The modeled resistance surface for western toads is based on the following parameterized spatial data layers:

1) Land cover/land-use
2) Elevation
3) Housing density
4) Road use type and distance

Values for these factors are provided in Table A.23.

Table A.23. Landscape features and resistance values used to model habitat connectivity for the western toad.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	50
urban/developed	500
water	10
sparsely vegetated	2
alpine	10
riparian	0
wetland	0
grass-dominated	0
shrub-dominated	0
dry forest	0
wet forest	0
Elevation (meters)	
0–250	0
>250–500	0
>500–750	0
>750–1000	0
>1000–1500	0
>1500–2000	0
>2000–2500	20
>2500–3300	40
>3300 meters	500
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤80	0
> 20 ≤40	20
> 10 ≤20	50
≤ 10	500
Road type and distance (meters)*	
freeway > 500–1000 buffer	25
freeway > 0–500 buffer	50
freeway centerline	500
major highway > 500–1000 buffer	10
major highway > 0–500 buffer	20
major highway centerline	100
secondary highway > 500–1000 buffer	0
secondary highway > 0–500 buffer	2
secondary highway centerline	5
local road > 500–1000 buffer	0
local road > 0–500 buffer	1
local road centerline	2

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Habitat Concentration Areas

We used the following steps to identify western toad habitat concentration areas (HCAs):

1) Identify potential breeding habitat.

    a. Use the land cover/land-use layer for the full study area and from this layer use the grid cells classified as water, wetland, or riparian.

    b. Additionally use the National Wetlands Inventory (NWI) layer for Washington. Include: all Palustrine types; all Lacustrine Littoral; all Riverine Upper and Lower Perennial. For eastern Washington additionally use the Riverine Intermittent streambed. (We used the NWI layer to incorporate better details for areas of Washington where the land cover/land-use layer was weak on detail. This layer does not exist for British Columbia, and Idaho and Oregon NWI data included inconsistencies that were not compatible with the Washington NWI).

    c. Reclassify and merge the layers such that all grid cells classified as potential breeding habitat have a value of 1 and all other grid cells have a value of 0.

2) Determine terrestrial habitats that are connected (i.e., complementary) to breeding habitats.

    a. Run a 2 km moving window over the potential breeding habitat identified in step 1.c. to calculate the average proportion of habitat within the window.

    b. Remove any habitat grid cells that fall within areas where presumed breeding habitat density is <0.05 thus eliminating areas where the breeding habitat density is very scarce.

3) Determine neighboring breeding locations and connected terrestrials habitats.

    a. Use the western toad resistance surface to run a 2 km cost-weighted distance from the areas identified in step 2.

    b. Reclassify areas identified in 3.a. as habitat (value = 1).

4) Review map. At this step we identified that the draft map provided extensive connections between presumed breeding habitat, complementary terrestrial habitat, and neighboring breeding locations and upland habitat. This map became the preliminary HCA map.

5) Complete HCA processing.

    a. Habitat concentration areas below a minimum size of 50 km^2 were removed. (This size is a rough approximation for an area comprised of four adjoining complementary breeding and terrestrial habitat areas each with a radius of 2 km. The radius of 2 km is chosen to approximate the use of a 2 km radius movement distance from presumed breeding habitat).

    b. In our final step we removed habitat concentration areas that were outside of the toad's eastern Washington range identified in Dvornich et al. (1997).

Linkage Modeling

During linkage modeling we used a maximum cost-weighted distance of 50 km. This distance was based on review of draft linkage products, and ensures most HCAs are connected to at least one other HCA while maintaining a distance that toads might move as linkage dwellers over multiple generations. We used zero for the minimum modeling distance between HCAs to ensure linkages would be modeled for all HCAs in close proximity.

To display linkage widths we used cost-weighted distances of 10 km on both sides of the least cost distance path. Our objective in using this distance was to provide broad linkages that could be further refined during finer-scale modeling, or implementation planning.

Considerations for Future Modeling

Five areas stand out as future opportunities for improving the modeling of broad-scale habitat use and movement for western toads. They are:

1) The Salish Sea Islands (e.g., Whidbey Island, Vancouver Island) were not included in the statewide analysis; these are important areas for western toads, and should be included in future efforts.

2) We received feedback that our results for lands east of the Okanogan River in northeast Washington may under-represent the amount of western toad habitat. Our data layers may have under-represented water, wetland, and/or riparian features for this part of the state. For future modeling, the accuracy of these layers should be reviewed, and additional factors that might improve model quality should be considered.

3) Conversely, we received feedback that our model results may be over estimating toad habitat for the Willapa Hills. A possible improvement would be to provide finer detail to how the riverine breeding habitat is classified.

4) Road traffic is likely to be among the most significant factors affecting toad survival. In this model we purposely applied low values to the resistance factors for secondary highways and local roads. Had we done otherwise, the coarse road data we used for the statewide scale would have indicated much of the landscape as inhospitable. At finer scales of modeling, obtaining road layers with traffic data should enhance model performance.

5) Fire, flooding, volcanic eruptions, and other large landscape processes create open habitat conditions associated with toad use. It would be useful to determine whether GIS layers are available for landscape processes that create openings, and if so, to consider how to effectively use such layers in western toad modeling.

In addition, we need basic research that can be used to validate these model results. This research is necessary to answer data gaps throughout the spectrum of population scales, from the breeding site and associated uplands, to the broad, metapopulation scale. Such work would benefit from landscape genetics to understand barrier effects, and to document population characteristics.

Acknowledgements

Brian Cosentino (WDFW), Marc Hayes (WDFW), Andrew Shirk (Independent Researcher), Kelly McAllister (WSDOT), Peter Singleton (USFS), Brad McRae (TNC), and Albert Perez

(WSDOT) provided assistance with model development. Brian Cosentino produced the western toad resistance surface and linkage map, and Andrew Shirk assisted with methods development and performed informal sensitivity analyses to test data layers and refine methods to produce the HCA map product.

In addition, the following persons provided model review: Scott Fitkin (WDFW), Saunders Freed (TNC), Rose Gerlinger (CCT), Lisa Hallock (WDFW), Audrey Hatch (WDFW), Marc Hayes (WDFW), Jeff Heinlen (WDFW), Gina King (YNW), Bill Leonard (WSDOT), Eric Lund (WDFW), Jim Lynch (Fort Lewis), Aimee MacIntyre, (WDFW), Kelly McAllister (WSDOT), Ruth Milner (WDFW), Dede Olson (USFS), John Pierce (WDFW), John Richardson (Fort Lewis), Leona Svancara (UI), and Elke Wind (E. Wind Consulting).

Literature Cited

Adams, M. J. [date unknown]. Study description: Space use of western toads at Tipsoo Lake in Relation to roads and trails. USGS. Available from http://fresc.usgs.gov/research/StudyDetail.asp?Study_ID=618 (accessed June 2010)

Bartelt, P. E., C. R. Peterson, and R.W. Klaver. 2004. Sexual differences in the post-breeding movements and habitats selected by western toads (*Bufo boreas*) in southeastern Idaho. Herpetologica 60:455–467.

Bull, E. L. 2006. Sexual differences in the ecology and habitat selection of western toads (*Bufo boreas*) in northeastern Oregon. Herpetological Conservation and Biology 1:27–38.

Compton, B.W., K. McGarigal, S. A. Cushman, and L. R. Gamble. 2007. A resistant-kernel model of connectivity for amphibians that breed in vernal pools. Conservation Biology 21:788–799.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Fish and Wildlife Service. Washington D.C.

Deguise, I. E. 2007. Movements of adult western toads, *Bufo boreas*, in a managed forest landscape and the incidence of a disease in southwestern British Columbia. Master’s thesis. University of British Columbia.

Dvornich, K. M., K. R. McAllister, and K. B. Aubry. 1997. Amphibians and reptiles of Washington State: Location data and predicted distributions. Volume 2 in K. M. Cassidy, C. E. Grue, M. R. Smith, and K.M. Dvornich, editors. Washington State Gap Analysis – Final Report, Washington Cooperative Fish and Wildlife Research Unit, University of Washington, Seattle.

Halley, J. M., R. S. Oldham, and J. W. Arntzen. 1996. Predicting the persistence of amphibian populations with the help of a spatial model. Journal of Applied Ecology 33:455–470.

Heine, G. 1987. Einfache MeB- und Rechemnethode sur Ermittlumg der Uberlebenschance wandernder Amphibien beim Uberquieren von StraBen. Naturschutz und

Landschaftspflege in Baden-Wurttemberg. 41:473–479. in Reh, W. and A. Seitz. 1990. The influence of land use in the genetic structure of populations of the common frog *Rana temporaria*. Biological Conservation 54:239–249.

Hels, T., and E. Buchwald. 2001. The effect of road kills on amphibian populations. Biological Conservation 99:331–340.

Idaho Cooperative Fish and Wildlife Research Unit. 2002. Idaho Vertebrate Modeling Database. Available from http://www.wildlife.unidaho.edu/idvmd/idvmd_habitat.asp?elcode=AAABB01030 (accessed June 2009).

Jones, L. L. C., W. P. Leonard, and D. H. Olson, editors. 2005. Amphibians of the Pacific Northwest. Seattle Audubon Society, Seattle, Washington.

Kuhn, J. 1987. StraBentod der Erdkrote (*Bufo bufo* L.): Verlustquoten und Verkehsaufkommen, Verhalten auf der StraBe. Naturschutz und Landschaftspflege in Baden-Wurttemberg. 41:175–186. in Reh, W. and A. Seitz. 1990. The influence of land use in the genetic structure of populations of the common frog *Rana temporaria*. Biological Conservation 54:239–249.

Leonard, W. P., H. A. Brown, L. L. C. Jones, K. R. McAllister, and R. M. Storm. 1993. Amphibians of Washington and Oregon. Seattle Audubon Society, Seattle, Washington.

Lynch, J. 2006. Western toad (*Bufo boreas*) habitat use, distribution, and conservation at Fort Lewis, Washington. Master's thesis. The Evergreen State College, Olympia, Washington.

MacCracken, J. G. 2008. Bayesian belief network model: Pond-breeding amphibians. Washington Landscape-level Wildlife Assessment Project. Longview Timberlands, LLC, and Washington Department of Fish and Wildlife, Olympia, Washington.

Mathias, M. 2008. Assessing functional connectivity for the northern red-legged frog (*Rana aurora*) in an urbanizing landscape. Master's thesis. University of Washington, Seattle, Washington.

Muths, E., and P. Nanjappa. 2005. Species account: western toad (*Bufo boreas*). in M. Lannoo, editor. Amphibian declines: conservation status of United States species. The University of California Press, Berkeley, California.

NatureServe. 2009. NatureServe Explorer: an online encyclopedia of life [web application]. Version 7.1. NatureServe, Arlington, Virginia. Available from http://www.natureserve.org/explorer (accessed June 2009).

Ovaska, K., L. Sopuck, C. Engelstoft, L. Matthias, E. Wind, and J. MacGarvie. 2004. Best management practices for amphibians and reptiles in urban and rural environments in

British Columbia. British Columbia Ministry of Water, Land and Air Protection, Nanaimo, British Columbia.

Peterson, N. P. 2004. Western toad monitoring plan. Prepared for Eric Beach, Green Diamond Resource Company. Stillwater Sciences, Olympia, Washington.

Pyare, S., R. E. Christensen III, R. Carstensen, and J. J. Adams. 2005. Preliminary assessment of breeding site occupancy and habitat associations for western toad (*Bufo boreas*) monitoring in Glacier Bay. in J. F. Piatt and S. M. Gende, editors. Proceedings of the Fourth Glacier Bay Science Symposium, 2004. U.S. Geological Survey, Information and Technology Report USGS/BRD/ITR-2005-00XX, Washington, D.C.

Ray, N., A. Lehmann, and P. Joly. 2002. Modeling spatial distribution of amphibian populations: a GIS approach based on habitat matrix permeability. Biodiversity and Conservation 11:2143–2165.

Schmetterling, D. A., and M. K. Young. 2008. Summer movements of boreal toads (*Bufo boreas boreas*) in two western Montana basins. Journal of Herpetology 42:11–123.

van Gelder, J. J. 1973. A quantitative approach to the mortality resulting from traffic in a population of *Bufo bufo* L. Oecologia (Berl.) 13:93–95.

WDFW (Washington Department of Fish and Wildlife). 2008. Draft Species and Development Database. Diversity Division, Olympia, Washington.

Personal Communication

K. Richter
King County Department of Natural Resources
Washington

American Marten (*Martes americana*)

Prepared by William L. Gaines (U.S. Forest Service)

Justification for Selection

American marten (*Martes americana*) were selected as a focal species for this assessment because of their relatively wide distribution and association with late-successional forests. Marten have also been identified as a species of management focus on national forest lands (USDA-FS 2006) and are considered a Species of Greatest Conservation Need in Washington State (WDFW 2005).

Distribution

American marten have a relatively wide distribution across the forested portions of the assessment area (Johnson & Cassidy 1997), though they are relatively rare within the lower-elevation dry forests of eastern Washington (Bull et al. 2005; Munzing & Gaines 2008).

Habitat Associations

Marten prefer riparian habitats throughout their range (Martin 1987; Buskirk et al. 1989; Anthony et al. 2003) and habitats near water (Bull et al. 2005). Fecske et al. (2002) characterized this relationship by distinguishing areas less than and greater than 100 m from streams.

Percentage of the landscape in openings is a primary factor in determining the quality of marten habitat. Hargis and Bissonette (1997) and Hargis et al. (1999) reported very little use of landscapes with ≥25% in openings. Potvin et al. (2000) also reported that marten home ranges contained less than 30–35% clear cut openings. Clear cuts supported 0–33% of population levels of marten in nearby uncut forest (Soutiere 1979; Snyder & Bissonette 1987; Thompson et al. 1989). Marten population reductions of 67% were reported following removal of 60% of timber (Soutiere 1979) and 90% with 90% timber removal (Thompson 1994). Chapin et al. (1998) reported that marten tolerated 20% (median value) of their home range in regenerating forest.

Martin and Barrett (1991) found 39 logs/ha within habitats used by marten. Resting stands had a mean density of coarse woody debris (logs >0.2 m in diameter) of 13/ha (Buskirk et al. 1989). Coarse woody debris was 117/ha at natal den sites and 90/ha at maternal den sites (Ruggiero et al. 1998). Gilbert et al. (1997) found 150 logs/ha at den and rest sites. Marten avoided plots with low densities of coarse woody debris whereas plots with high, to very high, densities were selected by marten (Spencer et al. 1983). Log densities of 20–50/ha were considered optimum (Martin 1987). Therefore, removal of snags and/or coarse woody debris through fire-wood gathering or through silvicultural practices will diminish the value of a stand as source habitat for marten.

Hodgman et al. (1994) reported 90% of marten mortality resulted from trapping on an area with a road density of 1.09 km/km^2. Thompson (1994) also reported that trapping was the major source of mortality for marten. He also observed that predation and trapping mortality rates were higher in logged forests (with road development) than in uncut forests. Alexander and Waters (2000) observed avoidance by marten of areas within 50 m of roads.

Dispersal

Limited information is available concerning the size of American marten home ranges. Information from Bull and Heater (2001) from the Blue Mountains in Oregon represents the best source of home range estimates for American Marten in the assessment area. They reported American Marten home range sizes of 27.2 km^2 for males and 14.2 km^2 for females.

Modeling Efforts

There have been two efforts to assess habitat connectivity for the American marten within portions of Washington State. W. Gaines (unpublished data) developed a dispersal habitat suitability model for American marten in northeastern Washington using the following variables: vegetation zone/cover-type, road and trail density, housing density, elevation, and slope. Begley and Long (2009) developed a similar model for the area around Steven Pass using the variables: elevation, slope, vegetation zone, vegetation structure, roads and trails, and building density.

Conceptual Basis for Statewide Model Development

OVERVIEW

Source habitat was modeled in northeastern Washington for American marten and considered to be cold-moist and cold-dry forests dominated by subalpine fir (*Abies lasiocarpa*), grand fir (*Abies grandis*), Pacific silver fir (*Abies amabilis*), Englemann spruce (*Picea englemanni*), western hemlock (*Tsuga heterophylla*), mountain hemlock (*Tsuga mertensiana*), whitebark pine (*Pinus albicaulis*), and western redcedar (*Thuja plicata*). Key attributes of forest structure include multi-story mid and late seral stages, a quadratic mean diameter >38 cm, and a closed canopy (i.e., > 50%; W. Gaines, unpublished data). This designation of source habitat is based on the association of American marten with medium and large trees (i.e., >40 cm dbh) and closed-canopy over-story vegetation in coniferous forests (Koehler et al. 1975; Campbell 1979; Martin 1987; Buskirk et al. 1989; Bull & Heater 2000; Wilbert et al. 2000; Nams & Bourgeois 2004; Gosse et al. 2005).

Source habitat for American marten is typically forests without a history of timber harvest and high densities of snag and coarse woody debris. American marten habitat values associated with varying snag densities are documented in Martin and Barrett (1991), Gilbert et al. (1997), Payer and Harrison (1999), and Ruggiero et al. (1998). Snyder and Bissonette (1987) reported limited use by American marten of patches <15 ha. Patches used by resident American marten were 18 times larger (median = 27 ha) than patches that were not used (median = 1.5 ha) and were closer to adjacent forest preserves (Chapin et al. 1998). Median size of the largest forest patch in American marten home ranges was 150 ha for females and 247 ha for males (Chapin et al. 1998). Potvin et al. (2000) recommended that uncut forest patches be >100 ha to maximize core area and to minimize edge.

MOVEMENT DISTANCE

Based on dispersal and home range information we used a within home-range movement distance of 10 km and a cost-weighted distance of 30 km. For dispersal we used a maximum distance of 30 km and a cost-weighted distance of 300 km.

Landscape Features and Resistance Values

The modeled resistance surface for American marten (Table A.24) is based on the following spatial data layers:

1) Land cover/land-use
2) Elevation
3) Slope
4) Housing density
5) Road type and distance
6) Forest structure

Habitat Concentration Areas

Bull et al. (2001) presents the best home range estimates for American marten in the assessment area. They reported American marten home range sizes of 27.2 km^2 for males and 14.2 km^2 for females.

To identify habitat concentration areas (HCAs) for American marten we used a minimum patch size of 100 km^2, a resistance cutoff of ≤8, a home range radius of 2.5 km, and a moving window threshold of ≥0.7.

Linkage Modeling

Potential linkages were modeled for American marten when the least-cost distance between HCAs was <300 km.

Considerations for Future Modeling

Future modeling should include more detailed vegetation information, in particular, the ability to identify late-successional habitat. These habitats are important for marten conservation but were difficult to accurately identify with the broad-scale vegetation data used in our assessment. Finer-scale linkage modeling will allow more specific identification of areas where habitat connectivity can be restored or enhanced. Finally, the American marten, in part because it is relatively common, provides an important opportunity to evaluate our habitat connectivity modeling efforts, both the identification of HCAs and potential linkages, through the use of landscape genetic techniques (Excoffier & Heckel 2006; Frantz et al. 2009).

Table A.24. Landscape features and resistance values used to model habitat connectivity for American marten.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	100
urban/developed	200
water	100
sparsely vegetated	50
alpine	5
riparian	0
wetland	1
grass-dominated	50
shrub-dominated	50
dry forest	10
wet forest	0
Elevation (meters)	
0–250	5
> 250–500	5
> 500–750	5
> 750–1000	5
> 1000–1500	2
> 1500–2000	2
> 2000–2500	1
> 2500–3300	1
>3300 meters	100
Slope (degrees)	
0–20	0
> 20–40	1
> 40	3
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	5
> 20 ≤ 40	10
> 10 ≤ 20	15
< 10	50
Road type and distance (meters)*	
freeway > 500–1000 buffer	1
freeway > 0–500 buffer	50
freeway centerline	1000
major highway > 500–1000 buffer	1
major highway > 0–500 buffer	10
major highway centerline	100
secondary highway > 500–1000 buffer	1
secondary highway > 0–500 buffer	6
secondary highway centerline	50
local road > 500–1000 buffer	1
local road > 0–500 buffer	1
local road centerline	1
Forest structure (density and height [meters])	
nonforest	10
sparse low (0–40%, ≤ 25)	7
sparse high (0–40%, > 25)	7
open low (>40–70%, ≤ 25)	5
open high (>40–70%, > 25)	5
dense low (>70–100%, ≤ 25)	1
dense high (>70–100%, > 25)	0

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Acknowledgements

Initial resistance values and HCAs were reviewed by Gary Koehler (WDFW), Robert Long (WTI), James Begley (WTI), and David Wallin (WWU). And special thanks to Andrew Shirk (Independent Researcher) and Brian Cosentino (WDFW) for their GIS and analytical support.

Literature Cited

Alexander, S. M., and N. M. Waters. 2000. Modeling wildlife movement requisites in the Banff-Bow Valley transportation corridor. International Conference on Integrating GIS and Environmental Modeling (GIS/EM4) 4 Available from http://www.colorado.edu/research/cires/banff/pubpapers/24/ (accessed 2010).

Anthony, R. G., W. A. O'Connell, M. M. Pollock, and J. G. Hallett. 2003. Association of mammals with riparian ecosystems in Pacific Northwest forests. Pages 510–563 in C. J. Zabel and R. G. Anthony, editors. Mammal community dynamics: management and conservation in the coniferous forests of North America. Cambridge University Press, New York, New York.

Begley, J. S., and R. A. Long. 2009. Modeling of wildlife dispersal habitat at Stevens Pass, Washington. Western Transportation Institute, College of Engineering, Montana State University, Bozeman, Montana.

Bull, E. L., and T. W. Heater. 2000. Resting and denning sites of American martens in northeastern Oregon. Northwest Science 74:179–185.

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

Bull, E. L., T. W. Heater, and J. F. Shepard. 2005. Habitat selection by the American marten in northeastern Oregon. Northwest Science 79:37–43.

Buskirk, S. W., S. C. Forrest, M. G. Raphael, and H. J. Harlow. 1989. Winter resting site ecology of marten in the central Rocky Mountains. Journal of Wildlife Management 53:191–196.

Campbell, T. M., III. 1979. Short-term effects of timber harvests on pine marten ecology. Master's thesis, Colorado State University, Fort Collins, Colorado.

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

Excoffier, L., and G. Heckel. 2006. Computer programs for population genetics data analysis: a survival guide. Nature Reviews 7:745–758.

Fecske, D. M., J. A. Jenks, and V. J. Smith. 2002. Field evaluation of a habitat-relation model for the American Marten. Wildlife Society Bulletin 30:775–782.

Frantz, A. C., S. Cellina, A. Krier, L. Schley, and T. Burke. 2009. Using spatial Bayesian methods to determine the genetic structure of a continuously distributed population: clusters or isolation by distance? Journal of Applied Ecology 46:493–505.

Gilbert, J. H., J. L. Wright, D. J. Lauten, and J. R. Probst. 1997. Den and rest-site characteristics of American Marten and Fisher in Northern Wisconsin. Pages 135–145 in G. Proulx, H. N. Bryant, and P. M. Woodard, editors. *Martes*: taxonomy, ecology, techniques, and management. Provincial Museum of Alberta, Edmonton, Alberta, Canada.

Gosse, J. W., R. Cox, and S. W. Avery. 2005. Home-range characteristics and habitat use by American Martens in eastern Newfoundland. Journal of Mammalogy 86:1156–1163.

Hargis, C. D., and J. A. Bissonette. 1997. Effects of forest fragmentation on populations of American Marten in the intermountain West. Pages 437–451 in G. Proulx, H. N. Bryant, and P. M. Woodard, editors. *Martes*: taxonomy, ecology, techniques, and management. Provincial Museum of Alberta, Edmonton, Alberta, Canada.

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

Hodgman, T. P., D. J. Harrison, D. D. Katnik, and K. D. Elowe. 1994. Survival in an intensively trapped marten population in Maine. Journal of Wildlife Management 58:593–600.

Johnson, R. E., and Cassidy, K. M. 1997. Terrestrial mammals of Washington State: Location data and predicted distributions. Volume 3 in K. M. Cassidy, C. E. Grue, M. R. Smith, and K. M. Dvornich, editors. Washington State Gap Analysis – Final report. Washington Cooperative Fish and Wildlife Research Unit, University of Washington, Seattle.

Koehler, G. M., W. R. Moore, R.A. Taylor. 1975. Preserving the Pine Marten: management guidelines for western forests. Western Wildlife 2:31–36.

Martin, S. K. 1987. The ecology of the Pine Marten (*Martes americana*) at Sagehen Creek, California. PhD dissertation, University of California, Berkeley, California.

Martin, S. K., and R. H. Barrett. 1991. Resting site selection by marten at Sagehen Creek, California. Northwest Naturalist 72:37–42.

Munzing, D., and W. L. Gaines. 2008. Monitoring American marten on the east-side of the North Cascades of Washington. Northwestern Naturalist 89:67–75.

Nams, V. O., and M. Bourgeois. 2004. Fractal dimension measures habitat use at different spatial scales: an example with marten. Canadian Journal of Zoology 82:1738–1747.

Payer, R. D., and D. J. Harrison. 1999. Effects of forest structure on spatial distribution of American Marten. National Council of the Paper Industry for Air and Stream Improvement Technical Bulletin 787.

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

Ruggiero, L. F., D. E. Pearson, and S. E. Henry. 1998. Characteristics of American Marten den sites in Wyoming. Journal of Wildlife Management 62:663–673.

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

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

Soutiere, E. C. 1979. Effects of timber harvesting on marten in Maine. Journal of Wildlife Management 43:850–860.

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

Thompson, I. D., I. J. Davidson, S. O'Donnell, and F. Brazeau. 1989. Use of track transects to measure the relative occurrence of some boreal mammals in uncut forest and regeneration stands. Canadian Journal of Zoology 67:1816–1823.

USDA-FS (United States Department of Agriculture-Forest Service). 2006. Terrestrial species assessments: Region 6 forest plan revisions. U.S. Forest Service, Pacific Northwest Region, Portland, Oregon.

WDFW (Washington Department of Fish and Wildlife). 2005. Washington Natural Heritage and Priority Habitat and Species Databases. WDFW Wildlife Program, Olympia, Washington.

Wilbert, C. J., S. W. Buskirk, and K. G. Gerow. 2000. Effects of weather and snow on habitat selection by American Martens. Canadian Journal of Zoology 78:1691–1696.

Canada Lynx (*Lynx canadensis*)

Prepared by William L. Gaines (U.S. Forest Service)

Justification for Selection

The Canada lynx (*Lynx canadensis*) was selected as a focal species due to its association with boreal forests, roughly equivalent to our subalpine vegetation class (Koehler & Aubry 1994; Aubry et al. 2000; Maletzke et al. 2008; Koehler et al. 2008). The Canada lynx is federally listed as a Threatened species within this portion of its range.

Winter recreation has been identified as a potential risk factor (Koehler & Aubry 1994; Buskirk et al. 2000; Ruediger et al. 2000). Snow trails that are groomed for snowmobiling and skiing potentially provide travel routes for competitors of Canada lynx such as coyotes (*Canis latrans*), bobcats (*Lynx rufus*), and cougars, allowing them to access areas where Canada lynx would normally have competitive advantage due to their large, snowshoe-like feet (Koehler & Aubry 1994; Buskirk et al. 2000).

Distribution

Canada lynx occur in most boreal forest habitats in North America, including the upper elevation coniferous forests of the Rocky Mountains and Cascade Ranges (Aubry et al. 2000). Lynx distribution is associated with boreal and sub-boreal forest conditions, including upper elevation, coniferous forests in western mountains. Lynx occur predominately in habitats where snowshoe hares (*Lepus americanus*) are abundant, especially early successional stands with high stem densities (Koehler & Aubry 1994; Maletzke et al. 2008). Throughout their range, Canada lynx are absent or uncommon in dense, wet forests along the Pacific coast (Aubry et al. 2000).

The distribution of Canada lynx within Washington State has been stratified into core, secondary and peripheral habitat areas based on known occurrence records (USFWS 2005). Core areas occur on the Okanogan National Forest, on the Methow Valley Ranger District and the western portion of the Tonasket Ranger District. The eastern portion of the Okanogan National Forest, on the Tonasket Ranger District, and western portion of the Colville National Forest is in a peripheral area. Core areas on the Colville include the Kettle Range. The Selkirks on the northeast portion of the Colville is a secondary area. Most of the Wenatchee National Forest north of I-90 is considered secondary, while the area south of I-90 is peripheral.

Habitat Associations

The southernmost extent of the boreal forest that supports Canada lynx occurs in the contiguous United States and includes the northern Rockies and the northern Cascades (McKelvey et al. 2000). The boreal forest in these areas transitions into other vegetation communities and becomes more patchily distributed. Key habitat components include foraging habitat for Canada lynx where understory stem densities and structure provide forage and cover for snowshoe hare (Koehler 1990; Agee 2000; Hodges 2000). Such habitat structure is common in early seral stages but may also occur in coniferous forests with mature but relatively open overstories (Hodges 2000).

Habitat selection by Canada lynx within Washington populations has been studied on the Okanogan National Forest. Lynx selected for Engelmann spruce and subalpine forest, moderate canopy cover, flat to moderate slopes, and relatively high elevations. They selected against Douglas-fir and ponderosa pine forest, forest openings, recent burns, sparse canopy and understory, and relatively steep slopes (Koehler et al. 2008; Maletzke et al. 2008). Probability of use by Canada lynx was 19.4 times greater in Engelmann spruce and subalpine fir forest than other vegetation types, 4.9 times greater in areas with moderate canopy cover than for other cover classes, 5.0 times greater at elevations ranging from 1525 m to 1829 m than other elevations, and 48.8 times greater on flat to moderate slopes than on steep slopes (Koehler et al. 2008).

Several researchers have expressed concerns over the effects of winter recreational activities on Canada lynx (Koehler & Aubry 1994; Buskirk et al. 2000; Bunnell et al. 2006; Kolbe 2006). Specifically, snow compaction associated with grooming for snowmobiling and cross-country skiing may provide travel routes for competitors and predators such as coyotes, bobcats, and cougars (Koehler & Aubry 1994; Buskirk et al. 2000; Bunnell et al. 2006).

Dispersal

Intra-home range movements vary seasonally and depend on the availability of prey, mainly snowshoe hares. Daily movement distances range from 2.6 to 10 km (Parker et al. 1983; Ward & Krebs 1985). Long-distance movements beyond home range boundaries have been documented in Montana and ranged from 15 to 40 km (Squires & Laurion 2000). However, this type of movement was not documented in the study in north-central Washington (Koehler 1990). In more northerly habitats Canada lynx can make long-distance movements during periods of prey scarcity; up to 1000 km (Slough & Mowat 1996; Poole 1997; Mech 1980). Information on lynx dispersal in southern boreal forests is scanty and anecdotal in nature. There has been no successful dispersal where breeding has been documented after moving to a new location in the southern part of Canada lynx range (Aubry et al. 2000). Dispersal distances reported from southern boreal forests are comparable to those in populations farther north, >100 km are considered typical (Mowat et al. 2000). Four of five lynx that dispersed in Montana, Washington, and Minnesota crossed either two- or four-lane highways and major rivers before they were trapped (Aubry et al. 2000).

Table A.25. Canada lynx within home-range and long-distance movements.

Description	*Distance*	*Citation*
Within home-range movement	2.7–5.4 km/day	Ward & Krebs 1985
Within home-range movement	8.8–10.0 km/day	Parker et al. 1983
Long-distance movements	15–40 km	Squires & Laurion 2000
Long-distance movements	up to 1000 km	Mech 1980; Poole 1997; Slough & Mowat 1997

Modeling Efforts

There have been at least three efforts to evaluate habitat connectivity for Canada lynx, each at a different spatial scale. Singleton et al. (2002) evaluated landscape permeability for Canada lynx across the state of Washington, including adjacent areas in Canada and Idaho. They used land cover class, human population density, road density, elevation, and slope as variables in their model.

Gaines et al. (in prep) conducted an assessment of habitat connectivity for Canada lynx by developing dispersal habitat suitability maps (similar to this project's resistance surfaces) for the northeastern portion of the state. They used vegetation zone/cover-type, road density, housing density, elevation, and slope in their model of dispersal habitat suitability.

Begley and Long (2009) developed a dispersal habitat suitability model for Canada lynx for the area around Stevens Pass. They developed the model using the following variables: elevation, slope, vegetation zone, vegetation structure, roads and trails, and building density.

Conceptual Basis for Statewide Connectivity Model Development

OVERVIEW

The information summarized in the previous sections was used to inform the development of resistance surfaces used to evaluate dispersal habitat suitability. This spatial information was then used to identify concentrations of high quality Canada lynx habitat referred to as habitat concentration areas (HCAs).

MOVEMENT DISTANCE

For the purpose of establishing which pairs of HCAs to link, we used a cost-weighted distance of 1350 km (Table A.26).

Table A.26. Home range information for Canada lynx from studies conducted within or adjacent to the assessment area and that had female home range information.

Location	*Home Range Size (km^2)*	*Citation*
S. Canadian Rocky Mtn.	277 (male) 135 (female)	Apps 2000
N-central Washington	49 (male) 37 (female)	Brittell et al. 1989
N-central Washington	69 (male) 39 (female)	Koehler 1990
N-central Montana	122 (male) 43 (female)	Brainerd 1985
N-central Montana	238 (male) 115 (female)	Squires & Laurion 2000

Landscape Features and Resistance Values

The resistance values assigned to various habitat features were based on the information summarized in the Habitat Associations section of this species summary and were reviewed and revised based on input from panels of experts (Table A.28.). The resistance surface for Canada lynx is based on the following spatial data layers:

1) Land cover/land-use
2) Elevation
3) Slope
4) Housing density
5) Road type and distance
6) Forest structure

Habitat Concentration Areas

The distribution of HCAs within the U.S. portion of the assessment area was constrained by the location of the core and secondary areas identified by the USFWS (2005), which were based on Canada lynx distribution.

Home range information was used for 17 female Canada lynx from the studies in Table A.26 to calculate a weighted average (weighted by the number of females in the study) of female home range size of 60.4 km^2. The information used to identify HCAs for Canada lynx included: a resistance value of ≤8, a home range radius of 4.4 km, a minimum patch size of 400 km^2, and a habitat threshold of 0.5.

Habitat concentration areas were identified for Canada lynx within the northern and eastern portions of the project area. These occurred within core and secondary areas identified for Canada lynx recovery (USFWS 2005) and within the highest quality Canada lynx habitat in the remainder of the project area. Habitat concentration areas occurred primarily within the North Cascades, Kettle Range, and Selkirk Mountains. The pattern of HCAs for Canada lynx are similar to those identified by Singleton et al. (2002) except that we constrained the distribution of HCAs by the core and secondary areas (as described above).

Linkage Modeling

Potential linkages were modeled for Canada lynx when the least-cost distance between a pair of HCAs was <1350 km.

Table A.27. Landscape features and resistance values used to model habitat connectivity for Canada lynx.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	100
urban/developed	200
water	100
sparsely vegetated	50
alpine	10
riparian	0
wetland	0
grass-dominated	50
shrub-dominated	50
dry forest	1
wet forest	0
Elevation (meters)	
0–250	50
> 250–500	20
> 500–750	10
> 750–1000	5
> 1000–1500	0
> 1500–2000	0
> 2000–2500	0
> 2500–3300	20
> 3300 meters	1000
Slope (degrees)	
0–20	0
> 20–40	2
> 40	10
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	2
> 20 ≤ 40	4
> 10 ≤ 20	10
< 10	100
Road type and distance (meters)*	
freeway > 500–1000 buffer	10
freeway > 0–500 buffer	100
freeway centerline	1000
major highway > 500–1000 buffer	5
major highway > 0–500 buffer	10
major highway centerline	100
secondary highway > 500–1000 buffer	3
secondary highway > 0–500 buffer	5
secondary highway centerline	20
local road > 500–1000 buffer	1
local road > 0–500 buffer	1
local road centerline	1
Forest structure (density and height [meters])	
non-forest	10
sparse low (0–40%, ≤ 25)	5
sparse high (0–40%, > 25)	5
open low (> 40–70%, ≤ 25)	2
open high (> 40–70%, > 25)	2
dense low (> 70–100%, ≤ 25)	0
dense high (> 70–100%, > 25)	0

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Acknowledgements

Initial resistance values and HCAs were reviewed by Gary Koehler (WDFW), Robert Long (WTI), James Begley (WTI), Robert Naney (USFS), and David Wallin (WWU). A special thanks to Andrew Shirk (Independent Researcher) and Brian Cosentino (WDFW) for their GIS and analytical support.

Literature Cited

Agee, J. K. 2000. Disturbance ecology of North American boreal forests. Pages 39–82 in Ecology and conservation of lynx in the United States. General Technical Report N-30WWW. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Apps, C. D. 2000. Space-use, diet, demographics, and topographic associations of lynx in the southern Canadian Rocky Mountains: A study. Pages 351–371 in Ecology and conservation of lynx in the United States. General Technical Report N-30WWW. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Aubry, K. B., G. M. Koehler, and J. R. Squires. 2000. Ecology of Canada lynx in southern boreal forests. Pages 373–396 in Ecology and conservation of lynx in the United States. General Technical Report N-30WWW. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Begley, J. S., and R. A. Long. 2009. Modeling of wildlife dispersal habitat at Stevens Pass, Washington. Western Transportation Institute, College of Engineering, Montana State University, Bozeman, Montana.

Brainerd, S. M. 1985. Reproductive ecology of bobcats and lynx in western Montana. Master's thesis. University of Montana, Missoula, Montana.

Brittell, J. D., R. J. Poekler, S. J. Sweeney, and G. M. Koehler. 1989. Native cats of Washington, Section III: lynx. Washington Department of Wildlife, Olympia, Washington.

Bunnell, K. D., J. T. Flinders, and M. L. Wolfe. 2006. Potential impacts of coyotes and snowmobiles on lynx conservation in the intermountain west. Wildlife Society Bulletin 34:828–838.

Buskirk, S. W., L. F. Ruggiero, and C. J. Krebs. 2000. Habitat fragmentation and interspecific competition: implications for lynx conservation. Pages 83–100 in Ecology and conservation of lynx in the United States. General Technical Report N-30WWW. USDA Forest Service, Rocky Mountain Research Station. Fort Collins, Colorado.

Gaines, W. L., J. S. Begley, B. C. Wales, L. H. Suring, K. Mellen-McLean, and S. Mohoric. in prep. Terrestrial species sustainability assessments for national forests in northeastern Washington. General Technical Report, USDA Forest Service, Pacific Northwest Research Station, Corvallis, Oregon.

Hodges, K. E. 2000. Ecology of snowshoe hares in southern boreal and montane forests. Pages 163–206 in Ecology and conservation of lynx in the United States. General Technical Report N-30WWW. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Koehler, G. M. 1990. Population and habitat characteristics of lynx and snowshoe hares in north central Washington. Canadian Journal of Zoology 68:845–851.

Koehler, G. M., and K. B. Aubry. 1994. Lynx. Pages 74–98 in The scientific basis for conserving forest carnivores: American marten, fisher, lynx and wolverine in the western United States. General Technical Report N-254. USDA Forest Service, Rocky Mountain Research Station, Missoula, Montana.

Koehler, G. M., B. T. Maletzke, J. A. VonKienast, K. B. Aubry, R. B. Weilgus, and R. H. Naney. 2008. Habitat fragmentation and the persistence of lynx populations in Washington State. Journal of Wildlife Management 72:1518–1524.

Kolbe, J. A. 2006. The effect of snowmobile trails on coyote movements within lynx home ranges. Master’s thesis. University of Montana, Missoula, Montana.

Maletzke, B. T., G. M. Koehler, R. B. Wielgus, K. B. Aubry, and M. A. Evans. 2008. Habitat conditions associated with lynx hunting behavior during winter in northern Washington. Journal of Wildlife Management 72:1473–1478.

McKelvey, K. S., Y. K. Ortega, G. M. Koehler, K. B. Aubry, and J. D. Brittell. 2000. Canada lynx habitat and topographic use patterns in north central Washington: a reanalysis. Pages 307–336 in Ecology and conservation of Lynx in the United States. General Technical Report N-30WWW. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Mech, L. D. 1980. Age, sex, reproduction, and spatial organization of lynxes colonizing northeastern Minnesota. Journal of Mammalogy 61:261–267.

Mowat, G., K. G. Poole, M. O’Donoghue. 2000. Ecology of lynx in northern Canada and Alaska. Pages 265–306 in Ecology and conservation of Lynx in the United States. General Technical Report N-30WWW. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Parker, G. R., J. W. Maxwell, L. D. Morton, and G. E. J. Smith. 1983. The ecology of the lynx (*Lynx canadensis*) on Cape Breton Island. Canadian Journal of Zoology 61:770–786.

Poole, K. G. 1997. Dispersal patterns of lynx in the Northwest Territories. Journal of Wildlife Management 61:497–505.

Ruediger, B., et al. 2000. Canada lynx conservation assessment and strategy. 2nd edition. Forest Service Publication #R1-00-53. USDA Forest Service, USDI Fish and Wildlife Service, USDI Bureau of Land Management, and USDI National Park Service. Missoula, Montana.

Singleton, P. H., W. L. Gaines, and J. F. Lehmkuhl. 2002. Landscape permeability for large carnivores in Washington: A geographic information system weighted-distance and least-cost corridor assessment. Research Paper N-549. USDA Forest Service, Pacific Northwest Research Station, Corvallis, Oregon.

Slough, B. G., and G. Mowat. 1996. Lynx population dynamics in an untrapped refugium. Journal of Wildlife Management 60:946–961.

Squires, J. R., and T. Laurion. 2000. Lynx home range and movements in Montana and Wyoming: preliminary results. Pages 337–349 in Ecology and conservation of lynx in the United States. General Technical Report N-30WWW. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

USFWS (U. S. Fish and Wildlife Service). 2005. Recovery outline: contiguous United States Distinct Population Segment of the Canada lynx. U.S. Fish and Wildlife Service, Montana Field Office, Helena, Montana.

Ward, R. P. M., and C. J. Krebs. 1985. Behavioural responses of lynx to declining snowshoe hare abundance. Canadian Journal of Zoology 63:2817–2824.

Mountain Goat (*Oreamnos americanus*)

Prepared by Andrew Shirk and Cliff Rice (Independent Researcher and Washington Department of Fish and Wildlife)

Justification for Selection

Mountain goats (*Oreamnos americanus*) were selected as focal species to represent alpine and subalpine habitat specialists. In these environments, mountain goats play key roles in ecosystem structure and function. Specifically, their foraging can strongly shape alpine plant communities (Pfitsch 1981) and they also serve as prey for large carnivores such as cougars, American black bears, grizzly bears, wolves, and wolverines (*Gulo gulo*; Festa-Bianchet et al. 1994). Mountain goats were also selected due to their avoidance of populated areas, which makes them sensitive to anthropogenic landscape changes, including roads and development. The regional population is patchily distributed among islands of habitat that are linked together by dispersal. Modeling dispersal is possible for this species as they are capable of long-distance movements which are evidently affected by coarse-scale landscape features such as elevation, land-cover type, and roads (Shirk et al. 2010). Moreover, validation of models is possible for this species as their movements are readily observed by tracking with radio collars and GPS (Global Positioning System) tagging (Rice 2008). Washington Department of Fish and Wildlife (WDFW) and the National Park Service have GPS movement data from 80 goats in Washington.

Distribution

Mountain goats are native to alpine and subalpine areas throughout the Washington Cascade Range, the Coastal Range of British Columbia, as well as certain subranges of the Rockies in Idaho and Montana. Populations have also been re-established in Idaho, Montana, and northeast Oregon after their extirpation. In addition, there are non-native populations in the region that have been established in areas that did not historically support mountain goats, including the Olympic Peninsula of Washington and portions of Montana and Idaho. The total population of Washington, British Columbia, Idaho, and Montana is thought to number approximately 2500, 36,000–63,000, 2700, and 2295–3045 respectively (Côté et al. 2003; Rice & Gay 2010). In Washington, the native population numbered approximately 8500 (excluding populations within Mount Rainier National Park and Yakama Nation lands) in 1961, but has been greatly depleted over the past several decades, likely due to over-harvest. Though harvest is now strictly limited, the current population has yet to recover to its former size, and some areas of formerly occupied habitat are now sparsely populated. Currently, the largest Washington concentrations exist in the Mount Baker area (442) and in the Goat Rocks area (378; C. R., unpublished data).

Habitat Associations

In low snowpack months, mountain goats are associated with alpine and subalpine meadows (which provide forage) in proximity to steep slopes (which provide escape terrain for predator avoidance). The percent slope required for escape terrain has been variably described to range from 37% to 50% (Smith 1994; Gross et al. 2002; Wells 2006). In high snowpack months, mountain goat habitat varies between coastal North American and interior populations. On the west side of the Coastal Range of British Columbia and the Cascade Range, mountain goats tend to move down in elevation during high snowpack months to areas where the maritime climate reduces snow depth and thereby improves foraging success. Mountain goats also favor steep slopes, southern aspects, and areas with a forest canopy in winter months, all of which potentially reduce snow depth and improve foraging success. In periods of mild weather, however, or on windswept ridges where snow does not accumulate, coastal mountain goats may move to forage in higher elevations during the winter. By contrast, interior mountain goats have no low elevation refuge, and tend to stay at higher elevations with steep slope and southern aspect in winter (Festa-Bianchet & Cote 2007).

Dispersal

Mountain goats are capable of long-distance movement (>60 km) through areas of poor habitat (Fielder & Keesee 1988). Recent studies describing mountain goat habitat (Wells 2006), genetic structure, and gene flow (Shirk et al. 2010) reveal that connectivity between the north and south Cascades is greatly reduced due to the effect of I-90. This interstate does not appear to be a complete barrier to dispersal, however, as several offspring of migrants that crossed I-90 have been detected by genetic means (Shirk et al. 2010). Aside from I-90, primarily geographic distance but also highways, urban and agricultural areas, very high and low elevations, and bodies of water reduce landscape connectivity for this species (Shirk et al. 2010). Mainly due to the large geographic extent of the North Cascades, the north subpopulation appears to be somewhat internally differentiated along a north-south gradient, indicating connectivity is not sufficiently high to unite all occupied habitat in the North Cascades into a single panmictic breeding pool. In the South Cascades (a much smaller geographic extent), however, mountain goats appear largely undifferentiated, suggesting high connectivity and panmixia (Shirk et al. 2010).

Modeling Efforts

A recent study of mountain goats within the Cascade Range of Washington sought to infer the resistance various landscape attributes pose to gene flow (Shirk et al. 2010). The conceptual basis behind this study was to identify an expert-opinion based hypothesis relating landscape resistance to four variables, including elevation, land-cover type, distance to escape terrain, and road traffic volume. This expert-opinion hypothesis was then tested using microsatellite genotypes of 135 mountain goat samples collected from individuals throughout much of the occupied habitat in the Cascade Range. This test was predicated on the assumption that the ecological distance (as opposed to Euclidean distance) between individuals should be strongly related to the genetic distance between sampled individuals. Specifically, circuit theory (McRae 2006) was used to infer the ecological distance between locations of all sampled individuals given a raster representation of the expert-opinion hypothesis and genetic distances that were determined based on principal components analysis of the microsatellite genotypes. The expert-opinion hypothesis was then modified one parameter at a time to determine if an alternate “optimized” resistance assigned to each of the four variables resulted in a model that fit better to the observed genetic distance between individuals. In this way, a new model (Figure A.1a) was identified that outperformed the expert opinion model as well as null models of isolation by Euclidean distance or isolation by the barrier effect of I-90. This approach also accounted for interactions between variables, allowed for non-linear responses, and excluded variables that reduce model performance. Circuit theory was also used to predict how gene flow moved along a north-south axis through the Cascade Range (Figure A.1b).

Surprisingly, the addition of a variable accounting for distance to escape terrain did not improve the fit of the optimized model (Shirk et al. 2010). There are several explanations as to why this important habitat requirement for mountain goats did not appear to be important in a model of gene flow within the study area. First, the spatial scale of the analysis (450 m grid cells) may not have been appropriate to capture the influence of escape terrain. Subsequent efforts to evaluate the importance of this variable demonstrated that reducing the resolution of the raster resistance surface to 100 m did not improve the fit of the model, however (A.S., unpublished data). Second, it may be that mountain goats do not have the perceptual ability to maximize their proximity to escape terrain while dispersing into unfamiliar territory. Third, as the Cascade Range possesses an abundance of steep slopes, it may be that escape terrain is not a limiting factor in habitat connectivity, and therefore does not appear necessary in models of gene flow.

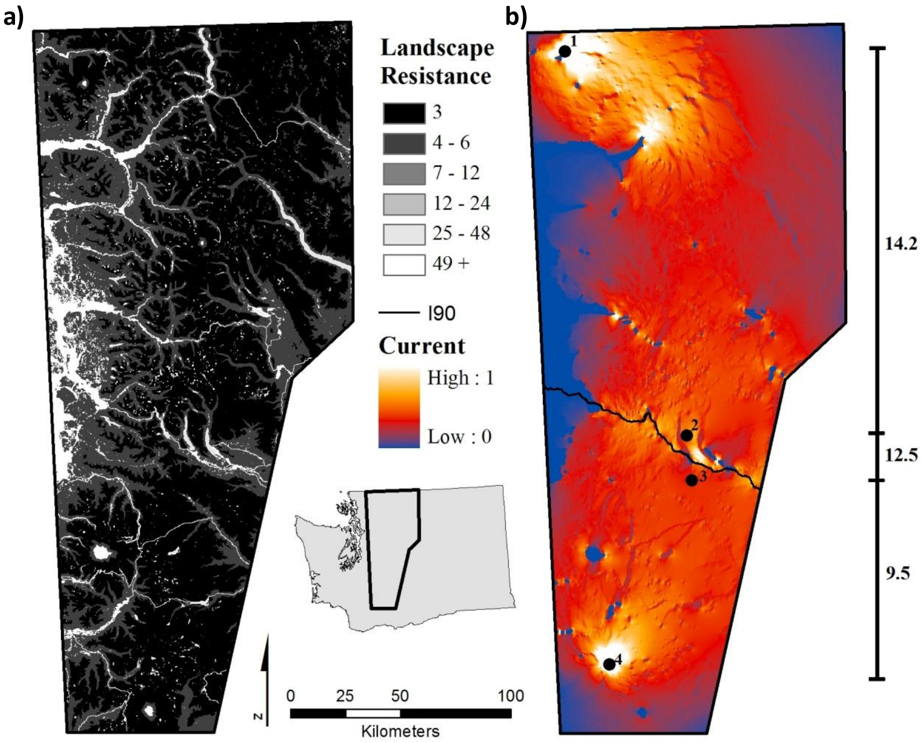

Figure A.1. (a) The optimized resistance model is depicted, with the highest resistance in white and lowest resistance in black. The values represent the sum of resistance due to roads, land cover, and elevation. (b) The predicted current after injecting one ampere into point 1 and connecting point 4 to ground. Zero current flows through areas where landscape resistance is very high (shown in blue). Moderate levels of current flow through areas where gene flow is relatively unconstrained (such as the north-central Cascades). In areas where highly resistive features constrain flow, current increases. Because current is constrained by the location where it is injected and the ground location where it exits, current is also high near points 1 and 4. The total circuit-theory resistance between points 1 and 4 is 25.8. Circuit theory resistance between points 1 and 4, and two intermediate points (point 2 and 3) flanking I-90 (black line), are shown to the right of the current flow map.

The optimized landscape resistance model from Shirk et al. (2010) represents the landscape's influence on gene flow rather than an index of habitat suitability. This dynamic is clearly illustrated by the optimized resistance model in that forested areas do not appear to limit gene flow (i.e., they confer no additional resistance), even though forested areas do not represent high quality habitat during low snowpack months that permit dispersal. Mountain goats appear to be capable of long distance movements across poor habitat. This is anecdotally supported by the limited GPS data available from radio-collared individuals attempting long-distance movements. For example, one individual (033GPF) underwent a seasonal movement to a salt lick on the northeast side of Glacier Peak, directly crossing a heavily forested valley (Figure A.2, blue GPS points and blue arrow). In the optimized resistance model, modified slightly for this WHCWG connectivity study (as described below), the entire track of 033GPF occurred in grid cells with the minimum resistance (one).

In theory, high-quality habitat should be a subset of the lowest resistance portions of the study area (based on the logic that habitat that is suitable for home range movements is also suitable for long-distance movement). Consistent with this notion, other GPS collared individuals moving within their home ranges also favored areas with minimum resistance (Figure A.2, black circles).

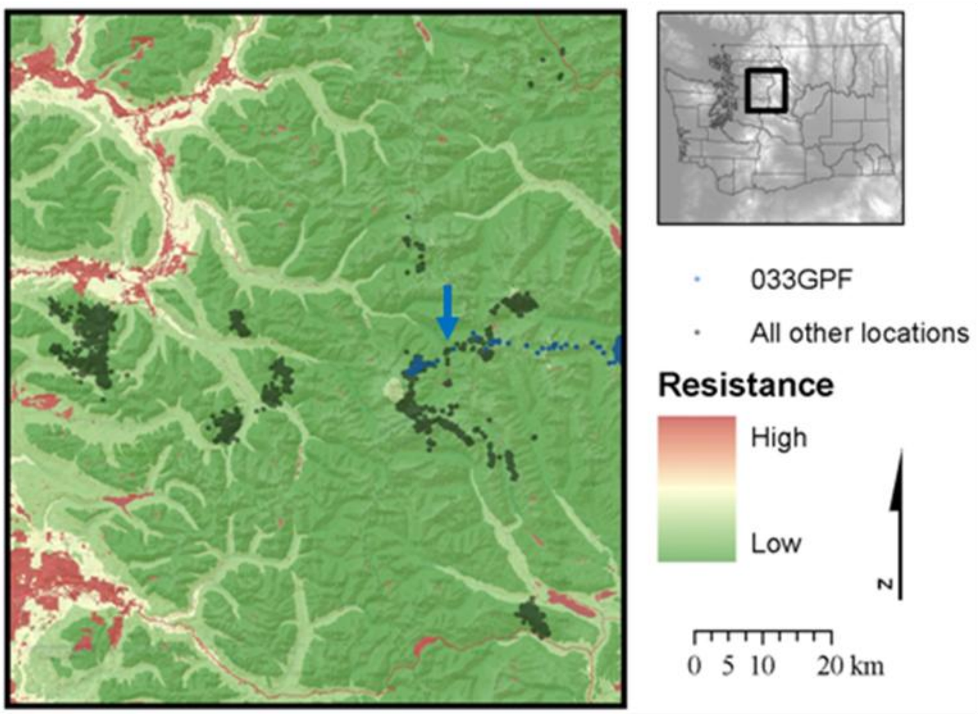

Figure A.2. Locations of GPS collared mountain goats are shown (black circles) relative to the WHCWG mountain goat landscape resistance model. A long-distance movement by individual 033GPF is also shown (blue circles), including a crossing of a broad valley of montane forest (blue arrow). Resistance varies from a minimum of one in dark green to 6 or more in red.

Conceptual Basis for Statewide Connectivity Model Development

OVERVIEW

Mountain goats are distributed in fairly discrete patches of habitat consisting of terrain that permits ground and aerial survey. Nearly all of the HCAs identified within the study area were delineated by surveys conducted by state and provincial wildlife agencies. The few that weren't were identified by regional wildlife biologists with general knowledge of the species' distribution within their district.

We based landscape resistances for the statewide connectivity model largely on the Shirk et al. (2010) study with three modifications. First, we divided resistance values of Shirk et al. (2010) by three to account for the fact that our minimum resistance was a value of 1 compared to 3 in the original study. Second, we partitioned 10% of the resistance due to interstate highways into the 100–500 m buffer zone on either side of the centerline (with the remaining 90% attributed to the centerline grid cells). The original study did not include road proximity buffers, however we considered this an important factor to consider given the aversion of this species to human disturbance. Lastly, we added the WHCWG housing density layer as a variable in the model. We considered adding this layer to be important, given this species' sensitivity to human population density.

MOVEMENT DISTANCE

Mountain goats have been observed dispersing in excess of 60 km in Euclidean distance (Fielder 1988; Cote et al. 2003). To conservatively represent the full capacity of this species to move between habitat areas, we chose 100 km as a maximum Euclidean distance at which habitat would be connected. The corridor models require distance to be expressed in cost-weighted distance, however. To achieve this, we multiplied 100 km by 2.17 (for a total of 217 km cost-weighted movement distance) based on the average resistance of the study area after excluding high-resistance (>10) grid cells which would presumably be rare in least-cost mountain goat movement corridors.

Landscape Features and Resistance Values

The resistance surface for mountain goats (Table A.28) is based on the following spatial data layers:

1) Land cover/land-use
2) Elevation
3) Housing density
4) Road type and distance

Table A.28. Landscape features and resistance values used to model mountain goat habitat connectivity.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use	
agriculture	1
urban/developed	0
water	8
sparsely vegetated	0
alpine	0
riparian	0
wetland	8
grass-dominated	0
shrub-dominated	0
dry forest	0
wet forest	0
Elevation (meters)	
0–250	2
> 250–500	1
> 500–750	1
> 750–1000	0
> 1000–1500	0
> 1500–2000	0
> 2000–2500	0
> 2500–3300	1
> 3300 meters	10000
Slope (degrees)	
0–20	0
> 20–40	0
> 40	0
Housing density (acres per dwelling unit)	
> 80	0
> 40 ≤ 80	10
> 20 ≤ 40	100
> 10 ≤ 20	1000
< 10	10000
Road type and distance (meters)*	
freeway > 500–1000 buffer	0
freeway > 0–500 buffer	10
freeway centerline	900
major highway > 500–1000 buffer	0
major highway > 0–500 buffer	0
major highway centerline	8
secondary highway > 500–1000 buffer	0
secondary highway > 0–500 buffer	0
secondary highway centerline	8
local road > 500–1000 buffer	0
local road > 0–500 buffer	0
local road centerline	0

**Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells.*

Habitat Concentration Areas

Mountain goat HCAs for Washington were delineated mainly by ground and aerial surveys and, to a lesser extent, by expert knowledge from district wildlife biologists (C.R., unpublished data). HCAs for portions of British Columbia, Idaho, Oregon, and Montana that fall within the study area have also been delineated by surveys and were been made available to this study by the

British Columbia Ministry of the Environment, Idaho Department of Fish and Game, Oregon Department of Fish and Game, and the Montana Department of Fish, Wildlife and Parks.

Linkage Modeling

Corridors linking mountain goat HCAs were limited to cost-weighted distances of less than 217 km (See Movement Distance subsection).

Considerations for Future Modeling

It may be insightful to consider the addition of slope as a factor in future efforts to model mountain goat habitat connectivity within the study area. We omitted this variable in the statewide model because the genetic analysis of mountain goat gene flow we based our resistance assignments on (Shirk et al. 2010) suggested it may not be a limiting factor. However, the broader extent of the WHCWG study area encompasses many areas of low slope, raising the notion that this important factor for mountain goat survival be included in future corridor models.

In future ecoregional models at finer spatial resolutions, it would be insightful to consider the impacts of planned I-90 wildlife crossing structures on connectivity between the north and south Cascades, and to identify new locations to maximize connectivity for this species. Interstate 90 was implicated as a major barrier to dispersal in Shirk et al. (2010). Continued habitat fragmentation by I-90 may reduce the population’s ability to recover from recent decline. Moreover, major highways are also likely to have greatly diminished connectivity between Washington, coastal British Columbia, and the Rockies. As such, identifying appropriate locations for crossing structures, and their effects, would increase our understanding of the potential to maintain connections between the peripheral population of Washington and the core populations in North America.

Acknowledgements

We thank David Wallin (WWU) for input and model review. We also thank the British Columbia Ministry of the Environment, Idaho Department of Fish and Game, Oregon Department of Fish and Game, and the Montana Department of Fish, Wildlife and Parks for providing spatial data pertaining to mountain goat habitat within their jurisdictions.

Literature Cited

Côté, S. D., M. Festa-Bianchet, and K. G. Smith. 2003. Wild mammals of North America: biology, management and conservation. 2nd edition. Johns Hopkins University, Baltimore, Maryland.

Festa-Bianchet, M., and S. Cote. 2007. Mountain goats: ecology, behavior, and conservation of an alpine ungulate. Island Press, Washington, D.C.

Festa-Bianchet, M., M. Urquhart, and K. G. Smith. 1994. Mountain goat recruitment - kid production and survival to breeding age. Canadian Journal of Zoology-Revue Canadienne de Zoologie 72:22–27.

Fielder, P. C., and B. G. Keesee. 1988. Results of a mountain goat transplant along Lake Chelan, Washington. Northwest Science 62:218–222.

Gross, J. E., M. C. Kneeland, D. F. Reed, and R. M. Reich. 2002. GIS-based habitat models for mountain goats. Journal of Mammalogy 83:218–228.

McRae, B. H. 2006. Isolation by resistance. Evolution 60:1551–1561.

Pfitsch, W. A. 1981. The effect of mountain goats on the subalpine plant communities of Klahhane ridge: Olympic National Park, Washington. University of Washington, Seattle, Washington.

Rice, C. G. 2008. Seasonal altitudinal movements of mountain goats. Journal of Wildlife Management 72:1706–1716.

Rice, C. G., and D. Gay. 2010. Mountain Goat harvest in Washington State: Effects on historic and contemporary populations. Northwestern Naturalist 91:40–57.

Shirk, A. J., D. O. Wallin, S. A. Cushman, C. R. Rice, and K. I. Warheit. 2010. Inferring landscape effects on gene flow: a new model selection framework. Molecular Ecology 19:3603–3619.

Smith, C. A. 1994. Bi-level analysis of habitat selection by mountain goats in coastal Alaska. Proceedings of the Biennial Symposium of the Northern Wild Sheep and Goat Council 5:366–379.

Wells, A. G. 2006. Global Positioning System (GPS) bias correction and habitat analysis of mountain goats *Oreamnos americanus* in the Cascades of Washington State, USA. Master's thesis. Western Washington University, Bellingham, Washington.

Wolverine (*Gulo gulo*)

Prepared by Karl C. Halupka and Andrew J. Shirk (U.S. Fish and Wildlife Service and Independent Researcher)

Justification for Selection

We selected the wolverine as a focal species to represent species that breed in subalpine and alpine habitats. The subgroup of the WHCWG that selected focal species gave the wolverine an “excellent” rating for all selection criteria as a representative of the Subalpine Forests and Alpine vegetation classes.

Wolverines are predators and scavengers that currently reproduce only in isolated, high-elevation refugia within our analysis area. Although wolverines appear to prefer to move through higher elevation areas (Copeland & Yates 2008; Schwartz et al. 2009; Copeland et al. 2010) they show a remarkable capacity for long-distance dispersal across a variety of forested and unforested habitat types (Inman et al. 2009). Wolverines also avoid human developments within home ranges (May et al. 2006) and during dispersal (Packila et al. 2007). Thus the wolverine represents breeding habitat specialists that are sensitive to human disturbance and dispersal habitat generalists that are highly mobile. The wolverine tends to have large spatial requirements, making it well suited for evaluating landscape permeability at large extents and coarse scales such as this statewide assessment (Begley & Long 2009).

Wolverines are known to travel great distances, begging the question of whether they are connectivity limited. Analysis of genetic diversity and population-genetic structure has suggested restricted female gene flow (Cegelski et al. 2006), especially compared to wolverines at more northern latitudes. The factors responsible for limiting gene flow are not well understood, but available evidence suggests both topographic and anthropogenic features may be involved (Cegelski et al. 2006). Together, these factors may sufficiently disrupt connectivity among wolverine populations that their likelihood of persistence is reduced. It is reasonable to expect that contemporary gene flow into the Washington wolverine population is limited to rare, long-distance, dispersal events.

Wolverine home ranges in Washington are large for mammals weighing less than 20 kg (Banci 1994; Rohrer et al. 2008), and may indicate that wolverines have high energetic requirements and at the same time occupy relatively unproductive niches (Inman et al. 2007). In Washington, near the southern extent of the wolverine’s range, suitable breeding habitat is largely confined to high-elevation areas. Large home-ranges combined with topographic and climatic constraints on habitat suitability result in relatively large areas being capable of supporting only a few individuals. In this context, wolverine population structure may be best considered a metapopulation dependent upon linkage among small subpopulations for sustainability. We expect that wolverines in Washington exist at low population densities and have low genetic diversity, suggesting that regular exchange of individuals among habitat islands is essential to avoiding further loss of genetic variation or local extirpation due to demographic stochasticity.

The association between wolverines and areas of persistent spring snow cover suggests the wolverine is also representative of species sensitive to climate changes that influence snow depth and persistence (Brodie & Post 2010; Copeland et al. 2010).

Finally, the wolverine is a rare carnivore that is a candidate for listing under the Endangered species Actand has been petitioned for listing under the Endangered Species Act. It is currently a Species of Concern at both the Washington State and federal levels.

Distribution

Wolverine habitat is defined by a tundra-subalpine signature that effectively restricts the species' distribution (Inman et al. 2008). At far northern latitudes, wolverine habitat occurs virtually everywhere, but in Washington, the subalpine life-zone necessary for wolverine presence becomes restricted to an elevational band, resulting in a naturally fragmented distribution (Copeland & Yates 2008). Populations in the Cascades and Rocky Mountains have been described as peninsular extensions of a more widespread population in Canada (Banci 1994).

In Washington, Oregon, and Idaho, wolverine sightings suggest the species' current distribution is clustered in the Cascade Ranges of Washington and Oregon, and the northern Rocky Mountains of Idaho (Edelmann & Copeland 1999). In British Columbia, the highest density of wolverines was predicted to occur in interior mountainous regions, with moderate densities in interior plateau and boreal forest regions, and low densities in mainland coastal regions and drier interior plateaus (Lofroth & Krebs 2007). Wolverines are rare on Vancouver Island and may be extirpated (Lofroth & Krebs 2007, citing Cannings et al. 1999).

The pattern of wolverine distribution in Washington has varied through time. Before 1919, wolverine presence was reported often in the Cascade Range and northern parts of Washington State (Johnson 1977). From 1919 to 1959 reports were rare, but increasing reports in the 1960s and 1970s suggested re-colonization was occurring (Johnson 1977). Wolverines have not been reliably reported from the Olympic Peninsula and coastal areas to the south (Johnson 1977). Recent work suggests breeding is occurring in the North Cascades of Washington (Rohrer et al. 2008).

Beyond the Washington statewide analysis area, the nearest large patches of wolverine habitat in the contiguous United States are in central Idaho and in the southern Cascades of Oregon (Edelmann & Copeland 1999). In southeast British Columbia, a broad band of high-quality habitat runs from the Monashee, Selkirk, and Purcell Mountain ranges, northwestward along the Rocky Mountains. North of the Yellowhead Highway, the extent of high quality habitat reaches from the Coast Mountains to the Rocky Mountains (Lofroth & Krebs 2007).

Wolverines likely occupied a wider variety of habitats during pre-settlement times as evidenced by historical presence in upper mid-western states, and fossil evidence of extant representatives in Great Basin habitats of southern Idaho (White et al. 1984). Records of the wolverine within the upper mid-west apparently pre-date human settlement, with the animal most likely absent by the early 1900s (Banci 1994). Human encroachment into historically occupied wolverine habitat may have forced the wolverine into its present distribution (Wolverine Foundation 2009).

Habitat Associations

GENERAL

Wolverine habitat in the contiguous United States consists of small, isolated ‘‘islands’’ or linear bands of high-elevation, alpine habitats following major mountain chains. These areas contain sufficient depth of snow during the denning period to support wolverine reproduction (Aubry et al. 2007; Schwartz et al. 2009; Copeland et al. 2010). These patches of suitable habitat are typically separated from each other by low valleys containing unsuitable habitats. Large distances between suitable wolverine habitat patches result in wolverines existing in an archipelago of semi-isolated, suitable habitats near mountain tops, surrounded by a sea of unsuitable habitats. Wolverines typically occupy habitat in a high-elevation band from 2100 m to 2600 m in Washington, Idaho, Montana and Wyoming. The intervening valleys in this area range from 975 m to 1500 m in elevation and are unsuitable for long-term wolverine habitat because they do not have the snow conditions or other habitat features required by wolverines (Aubry et al. 2007).

A preliminary analysis of elevational use by 4 radio-collared wolverines in Washington indicated that 3 of them used elevations <1350 m less than expected, elevations 1800 to 2250 m more than expected, and other elevation zones (1350 to 1800 m and >2250 m) at levels comparable to availability. These early results suggest a preference for relatively high elevation habitats near treeline (Rohrer et al. 2008).

The availability and distribution of food are likely the primary factors determining wolverine movements and home range size (Hornocker & Hash 1981; Banci 1994). Ungulate carrion is a primary food item and a close relationship exists between wolverine and ungulate presence. Wounding mortality of ungulates from hunting and livestock losses on public grazing allotments most likely provide consistent carrion sources. Individual wolverines display a movement pattern that suggests a patchy distribution of food resources; individuals move long distances to food patches and remain there for a period of time (Copeland & Yates 2008). Movement to and from these patches occurs at a rate of about 4 km/hour indicating that little time is spent hunting or pursuing other interests between patches. GPS telemetry points tend to show as clusters at these patches. Adult wolverines move repeatedly among these patches throughout the year (Copeland & Yates 2008).

Valley bottoms in Washington and other western states occupied by wolverines are increasingly dominated by human residential and commercial developments and transportation corridors, and represent semi-permeable barriers to wolverines. Although crossings of developed valleys, primarily by males (e.g., Packila et al. 2007, Figs. 2, 3), have been documented, these crossings are not common, and movements within valleys occur less frequently than movements in higher elevation wolverine habitats (Packila et al. 2007). Nonetheless, wolverines have been documented crossing a number of significant natural and artificial discontinuities in the landscape (Inman et al. 2004, 2009; Moriarity et al. 2009).

Roads are unlikely to be a complete barrier to wolverine movement, but depending on the amount of traffic and location, may affect wolverine behavior (Austin 1998; Moriarity et al. 2009; Inman et al. 2009) and survival (Krebs et al. 2004). During a long-term study in the

Greater Yellowstone region, researchers documented a total of 43 crossings of U.S. and state highways by 12 wolverines (Packila et al. 2007). Crossings of U.S. or State highways occurred in less than 8% of the annual home ranges of resident adults (n = 28 annual home ranges; Packila et al. 2007). During long-distance dispersal, individuals may cross numerous roads (Moriarity et al. 2009). Roads with rights-of-way that exceed 100 m may be a deterrent to wolverine movement (Austin 1998).

Denning habitat free from human disturbance may be a critical habitat feature for wolverines. Wolverines locate their home ranges away from human development and avoid human structures within home ranges (May et al. 2006). Idaho wolverines selected specific natal and kit rearing habitat and responded negatively to human disturbance near these sites (Magoun & Copeland 1998). Technological advances in over-snow vehicles and increased interest in winter recreation has likely displaced wolverines from potential denning habitat and will continue to threaten what may be a limited resource (Wolverine Foundation, 2009).

Vegetative characteristics appear less important to wolverines than physiographic structure of the habitat. Montane coniferous forests, suitable for winter foraging and summer kit rearing, may only be useful if connected with subalpine cirque habitats required for natal denning, security areas, and summer foraging. In addition, these habitats must be available during the proper season.

Preliminary analysis of fine-scale GPS telemetry data recorded at 5-minute intervals suggests that the wolverine generally maximizes its efficiency traversing rugged topography by operating as a “flat-land” species (Copeland & Yates 2008). Wolverines in Glacier National Park avoid uphill slopes greater than 5%, and downhill descents greater than about 3% slope as they contour the landscape. This apparent avoidance of steep slopes is confounded by the occasional crossings over formidable obstacles such as mountain peaks and high passes (Copeland & Yates 2008).

Dispersal

Wolverines are noted for having large home ranges and making long distance movements (Hornocker & Hash 1981; Lofroth & Krebs 2007). Daily movements of 30 to 40 km are common (reviewed in Singleton et al. 2002). Movements greater than 150 km have repeatedly been reported for both sexes (Copeland 1996; Vangen et al. 2001; Inman et al. 2004, 2007; Copeland & Yates 2008), and prodigious dispersal movements exceeding 350 km occasionally occur (e.g., Gardner et al. 1986; Inman et al. 2009; Moriarity et al. 2009).

Scaled to a typical home range, few dispersers moved more than 5 home-range diameters in Scandinavia (Vangen et al. 2001). Dispersal distances there averaged 51 km for males (range = 11–101 km, SD = 30 km) and 60 km for females (range = 15–178 km, SD = 48 km; Vangen et al. 2001). These values are likely to be underestimates because longer-distance dispersers were less likely to be detected during radio-tracking efforts (Vangen et al. 2001). Maximum dispersal distances of about 170 km from the center of maternal home ranges were recorded in a study of Yellowstone wolverines (Inman et al. 2007).

Pre-dispersal exploratory movements are common and can extend more than 100 km from natal territories (Vangen et al. 2001; Inman et al. 2004). Long-distance exploratory movements can occur in as few as seven days or be several months in duration (Inman et al. 2004).

Wolverines appear to prefer areas with spring snow cover during dispersal movements (Schwartz et al. 2009; Copeland et al. 2010). We do not know whether this movement pattern reflects strong preference for subalpine and alpine habitats, or strong avoidance of human development and activity at lower elevations.

In Scandinavia, the average age of dispersing wolverines was about 13 months for both males and females (Vangen et al. 2001). In the Greater Yellowstone Area, dispersal typically began at about 12 months of age, and occurred across multiple years (Inman et al. 2007). Dispersal rates (Vangen et al. 2001) and distances of male wolverines tend to be greater than for female wolverines (Banci 1994), but relatively little information about wolverine dispersal is available. Parallel analyses of both mitochondrial and nuclear DNA support predominant dispersal by males (Cegelski et al. 2006).

HOME RANGE SIZE AND WITHIN-TERRITORY MOVEMENTS

Wolverines have exceptionally large and variable home ranges (Table A.29). In Scandinavia, home ranges varied from 25 to 1246 km^2 (Vangen et al. 2001; Persson et al. 2010; 95% fixed kernel), and in North America home ranges are typically larger, sometimes exceeding 2000 km^2 (Copeland 1996; 95% minimum convex polygon). Home ranges of independent sub-adults can be even larger (Copeland 1996; Inman et al. 2004). Persson et al. (2010) provide a compilation of published home-range size estimates for wolverines across their holarctic distribution.

In North America, adult male home ranges are typically 3 to 5 times larger than adult female home ranges. Intrasexual territoriality is nearly exclusive, especially in males (Inman et al. 2007; Persson et al. 2010). Home range sizes have not been calculated for wolverines in the North Cascades of Washington, but preliminary analyses indicate they have activity areas that range from about 1150 to 2000 km^2 (Rohrer et al. 2008).

Relatively few estimates of average wolverine density are available for western North America (Table A.30), but all estimates indicate wolverines exist at remarkably low densities. Nonetheless, density-dependent factors likely regulate wolverine populations (Sæther et al. 2005). These low density estimates are likely representative of the wolverine population in Washington. These estimates also support the idea that connectivity among wolverine populations at the southern end of the species' range in North America is critical to maintaining sufficient gene flow to counteract reduction of genetic diversity due to genetic drift, and to provide demographic support to counteract stochastic impacts on small populations.

Movement studies using collars that provide location information at short intervals have revealed that wolverines often follow traditional routes within their home ranges, moving rapidly among patches where food is located (Copeland & Yates 2008). Movement between patches in Glacier National Park occurred at an average rate of about 4 km/hr (Copeland & Yates 2008). Adult male wolverines typically traveled about 150 km in a seven day period, for an average daily movement of about 21 km. These high rates of movement led to large portions of the annual home range being used over a relatively short time period (2 to 4 weeks) (Inman et al. 2007, 2008).

Table A.29. Annual home range estimates (km^2) for wolverine from western North America.

Sex and age class	*Location*	*Number of individuals*	*Mean area*	*Range*	*Estimation method*	*Citation*
Female adult	Greater Yellowstone	15*	453		95% fixed kernel	Inman et al. 2007
	Glacier NP	7	139	72–207	100% MCP	Copeland & Yates 2008
	Idaho (reproductive)	2	273	107–438	95% MCP	Copeland 1996
	Idaho (non-reproductive)	4	320	108–413	95% MCP	Copeland 1996
Male adult	Greater Yellowstone	13*	1160		95% fixed kernel	Inman et al. 2007
	Glacier NP	4	521		100% MCP	Copeland & Yates 2008
	Idaho	5	1506	953–2400	95% MCP	Copeland 1996
	Montana	3	422		100% MCP	Hornocker & Hash 1981

* Sample size expressed as “wolverine years.”

Table A.30. Wolverine population density estimates from western North America.

Location	*Density Estimate (per 1000 km^2)*	*Citation*	*Comments*
Greater Yellowstone	3.7	Inman et al. 2007	
British Columbia	6.2	Lofroth & Krebs 2007	High-quality habitat estimate
Northwestern Montana	15.4	Hornocker & Hash 1981	
Southwestern Montana	1.4 to 1.8	Squires et al. 2007	

Conceptual Basis for Statewide Connectivity Model Development

This section describes the results of previous modeling efforts and descriptive field investigations that provide the basis for estimating parameters for our statewide models of dispersal habitat suitability and habitat concentration areas (HCAs). A growing body of information derived from telemetry studies of wolverines is improving our understanding of wolverine habitat selection at coarse and fine scales, as well as their sensitivity to anthropogenic disturbances.

New information largely refines and affirms the conceptual underpinning of past modeling efforts regarding dispersal habitat suitability. New information about the tight association between wolverines and persistent spring snow cover (Copeland et al. 2010) considerably simplifies modeling of HCAs. Much of the information we describe under the “Dispersal Habitat Suitability” heading below also pertains to delineation of HCAs.

DISPERSAL HABITAT SUITABILITY

The evaluation of dispersal habitat suitability conducted by Singleton et al. (2002) served as the primary resource for our model development. These authors estimated the contribution to landscape resistance made by a set of variables very similar to the variables used in our analysis (Table A.31). The most pronounced difference between our model and that of Singleton et al. (2002) is that we evaluated the effects of roads using classes of road size and traffic volume rather than road density. We considered local roads- and secondary highways to contribute little to landscape resistance for dispersing wolverines. Due to anecdotal evidence of wolverine mortality associated with major roads and freeways, we considered these features to have a very strong effect on landscape resistance to wolverine dispersal. We also relied on a similar, small-scale modeling effort done by Begley and Long (2009) in the central Washington Cascades for ideas about how to translate information about wolverine habitat selection and response to human development into coefficients of habitat resistance. Overall, our parameterization of habitat resistance also benefitted greatly from expert review and from cross comparison with proposed parameters for other carnivore focal species.

Persistent spring snow cover is strongly correlated with wolverine den locations, year-round movement patterns, and is correlated with gene flow (Schwartz et al. 2009; Copeland et al., 2010). Persistent spring snow cover may provide a reliable and spatially explicit model of preferred wolverine habitat. This landscape feature is likely to strongly influence breeding habitat suitability for wolverine, especially when these areas are distant from roads or protected from human intrusion by regulations, distance, or inaccessibility. In Glacier National Park, female wolverines selected den sites where deep snow (2 to 3 meters) persisted until weaning in early May (Copeland & Yates 2008). Den sites located in Idaho and Alaska all were covered with at least 1 m of snow (Magoun & Copeland 1998). We describe our application of persistent spring snow cover to delineate HCAs in the next section of this account.

Topography may also influence habitat suitability. Denning habitat models that include alpine cirque habitat (landscape curvature) are better predictors of wolverine presence (Carroll et al. 2000). We did not include this feature in our modeling because a base layer of land curvature was not developed for our statewide modeling effort, and because the strength of the association with this land form was not sufficiently strong and independent of spring snow cover to warrant development of a custom layer for the wolverine model.

Low road densities (<1 mi/mi^2) also have been correlated with wolverine presence and used in models of source habitat (C. Carroll et al. 2001; Rowland et al. 2003; Copeland et al. 2007; Krebs et al. 2007). Winter recreational access may diminish habitat effectiveness. Incorporating overlays of groomed and designated winter recreation routes and distance from winter recreation trailheads may improve models of source habitat (Gaines in prep). We did not include road densities or winter recreation routes into our models due to the challenges associated with developing a base layer for these features across the entire analysis area.

Cover types and structural stages in montane forest, subalpine forest, and alpine tundra have also been included in source habitat models (Rowland et al. 2003). Preliminary source habitat models for wolverine on the Okanogan and Wenatchee National Forests included areas with road densities <1 mi/mi^2 and alpine, parkland, subalpine fir, Pacific silver fir, Engelmann spruce, western hemlock, western redcedar, mountain hemlock, lodgepole pine (*Pinus contorta*), western larch (*Larix occidentalis*), mixed conifer, Douglas-fir, and grand fir cover types (W. Gaines, unpublished data). We used this information to inform our resistance values for different cover types.

Broad-scale habitat association models for wolverine included Kuchler's potential natural vegetation types, Holdridge's climatic life zones, and EASE snow cover maps during the latter portion of the denning period (15 Apr to 14 May; Aubry et al. 2007). In Washington and Oregon, wolverine records are closely related to Kuchler's Alpine Meadows type, but in Idaho, Holdridge's Alpine stratum is more prevalent (Aubry et al. 2007). Spring snow cover was the only habitat layer that fully accounted for the distribution of historical wolverine records in the western mountains (Aubry et al. 2007). This information was instrumental to our parameterization of cover types and delineation of HCAs.

In British Columbia, wolverine habitat quality was modeled as a combination of prey (ungulates and marmots [*Marmota* spp.]) abundance, cover type (alpine tundra, Engelmann spruce, subalpine fir, mountain hemlock, spruce-willow-birch), persistent April snow cover, and road density (Lofroth & Krebs 2007). Multi-scale habitat use was analyzed using multi-model inference procedures including human use, land cover, winter prey distribution, and terrain variables (human use = snowmobiling, backcountry skiing, heli-skiing, roadless; terrain = ruggedness index; prey = % winter range, % avalanche, % alpine, % recently logged, % buffered riparian [beaver, *Castor canadensis*], and % interior cedar hemlock [small mammals]; Krebs et al. 2007). Male wolverine habitat use was positively associated with food-related habitat variables in both summer and winter (Krebs et al. 2007). Habitat associations for female wolverine were more complex, likely involving responses to food, predation risk, and human disturbance (Krebs et al. 2007). We did not develop base layers for prey, but used the results described here to inform our parameterization of resistance for cover types, human development, and roads.

In Idaho, seasonal habitat associations were modeled based on 9 habitat cover types (Douglas-fir, Douglas-fir–lodgepole pine, Douglas-fir–ponderosa pine, grass-shrub, lodgepole pine, mixed subalpine fir, montane–park, rock–barren, and whitebark pine [*Pinus albicaulis*]), 3 topographic variables (elevation, slope, and aspect), and distance to trails, roads, and streams (Copeland et al. 2007). Elevation was the strongest and most consistent variable distinguishing wolverine presence across all logistic regression models (Copeland et al. 2007). In summer, elevations >2400 m were used more than expected and elevations <2200 m were used less than expected. During winter, use shifted to the 2400 to 2600 m elevation zone (Copeland et al. 2007). Preferred cover type shifted from whitebark pine in summer to Douglas-fir and lodgepole pine in winter.

Steeper slopes were preferred by adult wolverine but not by subadults. Grass-shrub and rock were consistently strong negative indicators of wolverine presence. Wolverine stayed near streams in winter, but moved away from streams in summer. No relationship was apparent regarding distance to trails, possibly due to low rates of trail use. Wolverine appeared to avoid roads, but this may be an artifact of most roads being at lower elevations and the periphery of the study area (Copeland et al. 2007). It is unclear whether wolverine avoid human-related infrastructure, or simply prefer to reside in areas that are generally inhospitable to human development (Copeland et al. 2007). We used these results to inform our resistance values for cover types, elevation, slope, human development, and roads.

In an analysis of wolverine habitat across the Rocky Mountain States, including the eastern edge of our analysis area, a model which included latitude-adjusted elevation, a terrain ruggedness index, conifer cover, snow depth, forest edge, and road density was found to be robust to extrapolation and a potential foundation for conservation planning for wolverines (Brock et al. 2007).

After reviewing this information, we found that patterns of habitat association for wolverines were relatively consistent and made intuitive sense. However, the influence of specific features on landscape resistance, population substructure, and gene flow of wolverines are not well understood (Schwartz et al. 2009). Detailed analyses of short-interval telemetry data to determine fine-scale responses of wolverines to different habitat features are underway, but not currently available to inform parameterization of our model. Consequently, we sought to reflect the high level of uncertainty in our understanding of habitat resistance for wolverines by generally using resistance coefficients that roughly corresponded to low (0–10), medium (11–100), and high (>100) levels of resistance (Table A.31). We also attempted to round values to an order of magnitude whenever possible to avoid implying an unwarranted level of precision in our estimates. In the “footnotes” of Table A.31 we provide details about our rationale for estimating the resistance contributions of different landscape features.

Habitat Concentration Areas

To model HCAs for wolverine, we sought to incorporate factors found by other modeling efforts to be useful predictors of primary wolverine habitat. We began our attempts to model wolverine habitat concentration areas using layers the WHCWG had developed for modeling dispersal habitat resistance for multiple focal species. Input from wolverine experts and new literature, however, suggested that spring snow depth could also be a critical element defining wolverine habitat. Consequently, our early attempts to delineate HCAs for the wolverine included landcover, elevation, and road variables, as well as spring snow depth. Peer review of these efforts persuaded us to investigate simpler models in order to achieve a better correspondence with known wolverine locations and to reduce redundancy and apparent correlations among variables. We ultimately found that a model combining spring snow depth with low cumulative landscape resistance provided what we felt was the best approximation of wolverine HCAs.

Our basis for selecting the best model for wolverine HCAs relied on a few, relatively simple, qualitative criteria. First, we looked for conformance with known activity areas of wolverines fitted with radio telemetry collars in the North Cascades of Washington. This is the best recent information available about wolverine habitat use in our analysis area. For most of our area outside the North Cascades, sighting reports or anecdotes are the only types of information available about wolverine habitat use. Second, where our analysis area overlapped other efforts to model wolverine habitat, we looked for concordance between our proposed HCAs and areas found to be high quality habitat in previous modeling efforts. In particular, this included the analysis of Lofroth and Krebs (2007) for British Columbia, Singleton et al. (2002) for our entire analysis area, and Brock et al. (2007) for the western Rocky Mountains. Finally, we assessed the degree to which our proposed HCA models captured patches of concentrated sighting records.

Because our objective was to include in HCAs only the highest quality habitat, we did not use comprehensive coverage of sighting records as a criterion for evaluating quality of our candidate HCA delineation models. Note that for most of our analysis area, the current distribution of wolverines is unknown and our maps of HCAs represent predictions or hypotheses about where wolverines are most likely to be found.

The GIS layer describing spring snow depth we developed to delineate wolverine HCAs was a custom layer used only for this purpose. To achieve continuous coverage across the analysis area, we combined data from the Snow Data Assimilation System (SNODAS) and a broader effort to estimate monthly snow depth across North America (Brown et al. 2003). The SNODAS dataset has a resolution of 1 km and covers all of the analysis area except portions of the Coast Range in British Columbia (T. Carroll et al. 2001; Barrett 2003; NOHRSC 2004). We used SNODAS data collected from 2003 to 2009. This interval included record droughts (winter of 2004–2005), as well as years when winter precipitation was greater than 130% of normal (2008–2009) over much of the analysis area. Although the SNODAS time series is relatively short, we believe it is a representative sample. To fill gaps in SNODAS coverage in British Columbia we used estimates of snow depth interpolated to a 0.25 degree grid of latitude and longitude (about 25 km resolution), which were provided by the National Snow and Ice Data Center (NSIDC). These snow-depth estimates are based on data collected from 1980 to 1997 by the Canadian Meteorological Centre (R. Brown, personal communication; Brown et al. 2003). We resampled both the SNODAS and NSIDC data to a resolution of 100 m and reprojected it in the Albers NAD83 projection to be consistent with other layers used in our study.

SNODAS was developed by the National Weather Service's National Operational Hydrologic Remote Sensing Center to support hydrologic modeling and analysis. The aim of SNODAS is to provide a physically consistent framework to integrate snow data from satellite, airborne platforms, and ground stations with model estimates of snow cover (T. Carroll et al. 2001). SNODAS simulates snow cover using a physically based, spatially-distributed energy-and-mass-balance snow model. SNODAS also includes procedures to assimilate satellite-derived, airborne, and ground-based observations of snow-covered area (NOHRSC 2004).

Table A.31. Landscape features and resistance values used to model habitat connectivity for wolverines.

Spatial data layers and included factors	*Resistance value*
Land cover/land-use[a]	
agriculture	100
urban/developed	1000
water	100
sparsely vegetated	0
alpine	0
riparian	1
wetland	1
grass-dominated	1
shrub-dominated	1
dry forest	0
wet forest	0
Elevation[b] (meters)	
0–250	10
> 250–500	10
> 500–750	10
> 750–1000	1
> 1000–1500	1
> 1500–2000	0
> 2000–2500	0
> 2500–3300	10
> 3300 meters	100
Slope[c] (degrees)	
0–20	0
> 20–40	1
> 40	5
Housing density[d] (acres per dwelling unit)	
> 80	1
> 40 ≤ 80	5
> 20 ≤ 40	10
> 10 ≤ 20	200
< 10	200
Road type and distance[e] (meters)	
freeway > 500–1000 buffer	100
freeway > 0–500 buffer	200
freeway centerline	400
major highway > 500–1000 buffer	5
major highway > 0–500 buffer	10
major highway centerline	70
secondary highway > 500–1000 buffer	2
secondary highway > 0–500 buffer	4
secondary highway centerline	8
local road > 500–1000 buffer	1
local road > 0–500 buffer	1
local road centerline	1

[a] *Limited evidence of avoidance of infrastructure and disturbance. Resistance value is for open water. Frozen lakes and rivers likely have resistance values near 1(Banci 1994; Singleton et al. 2002; May et al. 2006; Begley & Long 2009).*

[b] *Elevation is a very strong predictor of wolverine presence and suitability of denning habitat. Increasing evidence that wolverines prefer high-elevation areas during dispersal as well (Schwartz et al. 2009; Copeland et al. 2010). Lack of food resources on volcanic summits and high risk and physiologic cost of crossing these summits contributes to the medium/high resistance value for this elevation band (see Singleton et al. 2002).*

[c] *Traversing 5 grid cells to avoid one steep grid cell seems consistent with movement patterns described by Copeland and Yates (2008; Singleton et al. 2002).*

[d]Limited specific information in literature about relative avoidance of areas of human development and activity. Current pattern of habitat selection suggests a strong pattern of avoidance; our opinion is that avoidance of human infrastructure is a comparatively strong factor influencing habitat resistance.

[e]Roads include one centerline cell and ten cells on each side of the centerline grid cell. Distance values are measured from margins of the centerline cells. Although wolverines are known to successfully cross highways, evidence of mortality and population genetic structuring due to highways suggest they may have high resistance (total of 1000 for a perpendicular crossing).

To model wolverine HCAs, we combined mean snow depth on May 1 across all years with the suite of layers we used to estimate landscape resistance to wolverine dispersal. Based on our review of literature about wolverine habitat selection, we specified that snow depth on May 1 must be 1 m or greater to be considered for inclusion in a wolverine HCA. We also specified that cumulative resistance for a grid cell must be 10 or less. Both criteria needed to be met before a grid cell could be mapped as part of a potential wolverine HCA.

The layer depicting the intersection of high spring snow depth and low landscape resistance became our base layer for delineation of wolverine HCAs. To complete the delineation process we needed to complete three additional steps. First, we wanted to remove from HCA consideration areas with a low concentration of habitat. Second, we sought to consolidate patches of habitat that were in close enough proximity to be effectively linked by the typical within-territory movements of a wolverine. And third, we needed to remove from HCA consideration patches of habitat that were not large enough to support at least one pair of wolverines.

To identify areas with concentrated habitat, we began by passing a circular moving window with an area equivalent to the mean home range of female wolverines over the resistance surface for the analysis area (area = 296.3 km^2, radius = 9.7 km; see Table A.29). Habitat grid cells in windows whose area included at least 50% habitat were retained. To consolidate nearby habitat patches, we expanded the patches defined by the moving window analysis by a cost-weighted distance equal to the home range radius of a female wolverine (9.7 km). We considered a home-range diameter (19.4 km) to be an index of a typical within-territory movement distance for a wolverine. Because expansion of habitat patches occurs from the perimeter of all patches, expansion between two adjacent patches combined would be equal to the cost-weighted home range diameter. If this expansion led to overlap in adjoining patches, we annealed these patches into one patch. Finally, habitat patches that were less than 400 km^2 in area were removed from consideration for HCA status. This process led to our final delineation of wolverine HCAs.

Increasing size of suitable habitat patches likely increases the conservation value of the patch for wolverines. Probability of occupancy may be related to both patch size and proximity to other suitable patches. In an analysis of primary wolverine habitat in the northern Rocky Mountain states, a minimum patch size of 100 km^2 was used (Inman et al. 2008), corresponding to the size of the minimum adult female home range in the coterminous U.S. (Hornocker & Hash 1981). Resulting patches were subsequently aggregated into 14 “major demes,” of which four were large enough to support greater than 50 wolverines (total area of primary habitat greater than 10,600 km^2). Aggregation was based on consideration of linkage habitat quality, location of major roads, and distribution of geographic features (Inman et al. 2008). The four large areas were designated “regional population centers” that have the potential to sustain the regional meta-population if adequate connectivity among these centers can be maintained or restored (Inman et al. 2008).

We found it challenging to determine a minimum size for HCAs. The difficulties were largely associated with conflict between emphasizing the role of smaller patches of habitat that could serve as stepping stones for dispersing wolverines (about 100 km^2) or focusing on larger patches of high quality habitat that were more likely to sustain populations of wolverines through time (10,000 km^2). We compromised at a smallish patch size of 400 km^2 that we felt was appropriate for both of the focal species in our analysis that are wide ranging carnivores (wolverine and lynx). We felt this patch size would meet our objectives of seeding our landscape with nodes of high-quality habitat for connectivity analysis, while accommodating uncertainty about the spatial requirements of wolverines in our analysis area.

Linkage Modeling

An important step in our linkage modeling process was to use information about the mobility of each focal species to identify both a maximum distance beyond which we believe linkage modeling is inappropriate (isolation by distance), and a minimum distance within which we believe separate HCAs should be annealed because linkage modeling will be uninformative. For the wolverine, we concluded an appropriate maximum distance was about 150 km, based on multiple studies that found maximum dispersal events for both sexes of about this distance. Longer movements have been reported, but they are rare. This distance spans the gaps between HCAs in the Washington Cascades and HCAs in the Coast Range and Rocky Mountains of British Columbia. These linkages are likely to be the primary routes for dispersal and gene flow for the wolverine. Regarding a minimum distance for annealing HCAs, we believe that the diameter of a female home range, about 20 km (Table A.29) is appropriate. This distance is the index of intra-territorial movements we used in HCA delineation.

In order for these distances to be used in our linkage modeling approach, we needed to express them in terms of cost-weighted distance. Because the relationship between Euclidean distance and cost-weighted distance can be highly variable, we sought a logical framework for translating Euclidean dispersal distances from the literature into cost-weighted distances. Given we had defined high-quality wolverine habitat to be included in HCAs as grid cells with a total resistance of 10 or less, we reasoned that a wolverine dispersing through HCA-quality habitat could accumulate a cost-weighted distance of up to 10 times the corresponding Euclidean distance. Thus, we expected 1500 km to be the maximum cost-weighted distance a wolverine would traverse if they moved through the landscape preferentially selecting the most suitable habitat available.

Considerations for Future Modeling

We feel that the movement patterns and large home range requirements of the wolverine make it most suitable for modeling at the state-wide scale of analysis. Connections among populations at this scale are likely necessary to achieve a sufficiently large effective population to reduce the negative effects of genetic drift and inbreeding and provide for demographic rescue of populations that decline due to stochastic events.

If the wolverine is used as a focal species for linkage analysis at the ecoregional scale, we suggest incorporating factors demonstrated to be useful for predicting wolverine habitat selection that could be compiled more readily at a smaller scale. These factors include prey resources, landscape curvature, and recreational routes (especially groomed winter recreation routes). Modeling at smaller scales could also seek to refine resistance values associated with transportation corridors, especially major roads. These efforts at the ecoregional scale could begin zooming-in on locations where wildlife crossing structures could be positioned near to preferred wolverine habitat. Because wolverines rely on deep spring snow for denning habitat, impacts of climate change on snowpack will be an important feature to consider in future modeling efforts.

Acknowledgements

Keith Aubry (USFS), Jeff Copeland (USFS), Cathy Raley (USFS), and John Rohrer (USFS) educated us about key aspects of wolverine biology and provided references. William Gaines (USFS) and Peter Singleton (USFS) provided valuable suggestions throughout the model development process. Brian Consentino (WDFW) and Albert Perez (WSDOT) compiled and managed GIS data and produced a profusion of maps. Gary Koehler (WDFW) provided a clear-eyed critique of preliminary parameter values for landscape resistance factors. Patty Garvey-Darda (USFS), Tristan Nuñez (UW), Robert Long (WTI), and Mark Nuetzmann (YNW) improved our ability to translate information about wolverines into a model of landscape resistance.

Literature Cited

Aubry, K. B., K. S. McKelvey, and J. P. Copeland. 2007. Distribution and broadscale habitat relations of the wolverine in the contiguous United States. Journal of Wildlife Management 71:2147–2158.

Austin, M. 1998. Wolverine winter travel routes and response to transportation corridors in Kicking Horse Pass between Yoho and Banff National Parks. Master’s thesis, University of Calgary, Calgary, Alberta, Canada.

Banci, V. A. 1994. Wolverine. Pages 99–127 in L. F. Ruggiero, K. B. Aubry, S. W. Buskirk, L. J. Lyon, and W. J. Zielinski, editors. The scientific basis for conserving forest carnivores, American marten, fisher, lynx and wolverine in the western United States. General Technical Report RM-254. USDA, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Barrett, A. 2003. National operational hydrologic remote sensing center snow data assimilation system (SNODAS) products at NSIDC. NSIDC Special Report 11. Boulder, Colorado USA: National Snow and Ice Data Center.

Begley, J. S., and R. A. Long. 2009. Modeling of wildlife dispersal habitat at Stevens Pass, Washington. Unpublished report prepared by the Western Transportation Institute for the Stevens Pass Resort.

Brock, B. L., R. M. Inman, K. H. Inman, A. J. McCue, M. L. Packila, and B. Giddings. 2007. Broad-scale wolverine habitat in the conterminous Rocky Mountain States. Chapter 2 in Greater Yellowstone wolverine study, cumulative report, May 2007. Wildlife Conservation Society, North America Program, General Technical Report, Bozeman, Montana.

Brodie, J. F., and E. Post. 2010. Nonlinear responses of wolverine populations to declining winter snowpack. Population Ecology 52:279–287.

Brown, R. D., B. Brasnett, and D. Robinson. 2003. Gridded North American monthly snow depth and snow water equivalent for GCM evaluation. Atmosphere-Ocean 411:1–14.

Cannings, S. G., L. R. Ramsay, D. F. Fraser, and M. A. Fraker. 1999. Rare amphibians, reptiles and mammals of British Columbia. Wildlife Branch and Resource Inventory Branch, British Columbia Ministry of Environment, Lands and Parks, Victoria, British Columbia, Canada.

Carroll, C, R. F. Noss, and P. C. Paquet. 2001. Carnivores as focal species for conservation planning the Rocky Mountain region. Ecological Applications 11:961–980.

Carroll, C., P. C. Paquet, and R. F. Noss. 2000. Modeling carnivore habitat in the Rocky Mountain region: a literature review and suggested strategy. World Wildlife Fund, Toronto, Ontario, Canada.

Carroll, T., D. Cline, G. Fall, A. Nilsson, L. Li, and A. Rost. 2001. Pages1–14 in, NOHRSC operations and the simulation of snow cover properties for the conterminous U.S. Proceedings of the 69th Annual Meeting of the Western Snow Conference, Las Vegas, New Mexico.

Cegelski, C. C., L. P. Waits, N. J. Anderson, O. Flagstad, C. Strobeck, and C. J. Kyle. 2006. Genetic diversity and population structure of wolverine (*Gulo gulo*) populations at the southern edge of their current distribution in North America with implications for genetic viability. Conservation Genetics 7:197–211.

Copeland, J. P. 1996. Biology of the wolverine in central Idaho. Master's thesis, University of Idaho, Moscow, Idaho.

Copeland, J. P., and R. E. Yates. 2008. Wolverine population assessment in Glacier National Park: comprehensive summary update. USDA, Forest Service, Rocky Mountain Research Station, Missoula, Montana.

Copeland, J. P., K. S. McKelvey, K. B. Aubry, A. Landa , J. Persson, R. M. Inman, J. Krebs, E. Lofroth, H. Golden, J. R. Squires, A. Magoun, M. K. Schwartz, J. Wilmot, C. L. Copeland, R.E. Yates, I. Kojola, and R. May. 2010. The bioclimatic envelope of the wolverine (*Gulo gulo*): do climatic constraints limit its geographic distribution? Canadian Journal of Zoology 88:233–246.

Copeland, J. P., J. Peak, C. Groves, W. Melquist, K. S. McKelvey, G. W. McDaniel, C. D. Long, and C. E. Harris. 2007. Seasonal habitat associations of the wolverine in Central Idaho. Journal of Wildlife Management 71:2201–2212.

Edelmann, F, and J. Copeland. 1999. Wolverine distribution in the northwestern United States and a survey in the Seven Devils Mountains of Idaho. Northwest Science 73:295–300.

Gardner, C. L., W. B. Ballard, and R. H. Jessup. 1986. Long distance movement by an adult wolverine. Journal of Mammalogy 67:603.

Hornocker, M. G., and H. S. Hash. 1981. Ecology of the wolverine in northwestern Montana. Canadian Journal of Zoology 59:1286–1301.

Inman, R. M., K. H. Inman, A. J. McCue, M. L. Packila, G. C. White, and B. C. Aber. 2007. Wolverine space use in Greater Yellowstone. Chapter 1 in, Greater Yellowstone wolverine study, cumulative report, May 2007. Wildlife Conservation Society, North America Program, General Technical Report, Bozeman, Montana.

Inman, R. M., M. L. Packila, K. H. Inman, R. S. Spence, and D. McCauley. 2008. Greater Yellowstone wolverine program: progress report–November 2008. Wildlife Conservation Society.

Inman, R. M., M. L. Packila, K.H. Inman, B. C. Aber, R.S. Spence, and D. McCauley. 2009. Greater Yellowstone wolverine program: progress report – December 2009. Wildlife Conservation Society.

Inman, R. M., R. R. Wigglesworth, K. H. Inman, M. K. Schwartz, B. L. Brock, and J. D. Rieck. 2004. Wolverine makes extensive movements in the Greater Yellowstone Ecosystem. Northwest Science 78:261–266.

Johnson, R. E. 1977. An historical analysis of wolverine abundance and distribution in Washington, U.S.A. Murrelet 58:13–16.

Krebs, J., E. C. Lofroth, J. Copeland, V. Banci, H. Golden, A. Magoun, R. Mulders, and B. Shults. 2004. Synthesis of survival rates and causes of mortality in North American wolverine. Journal of Wildlife Management 68:493–502.

Krebs, J., E. C. Lofroth, and I. Parfitt. 2007. Multiscale habitat use by wolverines in British Columbia, Canada. Journal of Wildlife Management 71:2180–2192.

Lofroth, E. C., and J. Krebs. 2007. The abundance and distribution of wolverines in British Columbia, Canada. Journal of Wildlife Management 71:2159–2169.

Magoun, A. J., and J. P. Copeland. 1998. Characteristics of wolverine reproductive den sites. Journal of Wildlife Management 62:1313–1320.

May, R., A. Landa, J. van Dijk, J. D. C. Linnell, and R. Andersen. 2006. Impact of infrastructure on habitat selection of wolverines *Gulo gulo*. Wildlife Biology 12:285–295.

May, R., J. van Dijk, A. Landa, R. Andersen, and R. Andersen. 2010. Spatio-temporal ranging behavior and its relevance to foraging strategies in wide-ranging wolverines. Ecological Modeling 221:936–943.

Moriarity, K. M., W. J. Zielinski, A. G. Gonzales, T. E. Dawson, K. M. Boatner, C. A. Wilson, F. V. Schlexer, K. L. Pilgrim, J. P. Copeland, and M. K. Schwartz. 2009. Wolverine confirmation in California after nearly a century: native or long-distance immigrant? Northwest Science 83:154–162.

NOHRSC (National Operational Hydrologic Remote Sensing Center). 2004. Snow Data Assimilation System (SNODAS) Data Products at NSIDC, 2003 to 2009. Boulder, Colorado USA: National Snow and Ice Data Center. Available from http://www.nohrsc.noaa.gov/archived_data/ (accessed June 2010).

Packila, M. L., R. M. Inman, K. H. Inman, and A. J. McCue. 2007. Wolverine road crossings in western Greater Yellowstone. Chapter 7 in, Greater Yellowstone wolverine study, cumulative report, May 2007. Wildlife Conservation Society, North America Program, General Technical Report, Bozeman, Montana.

Persson, J., P. Wedholm, and P. Segerström. 2010. Space use and territoriality of wolverines (*Gulo gulo*) in northern Scandinavia. European Journal of Wildlife Research 56:49–57.

Rohrer, J., K. Aubry, and C. Raley. 2008. Distribution and ecology of wolverines in the North Cascades. Year 2 (winter 2007/08) of a 5-year study. FY 2008 Status Report for the Interagency Special Status/Sensitive Species Program. USDA Forest Service.

Rowland, M. M., M. J. Wisdom, D. H. Johnson, B. C. Wales, J. P. Copeland, and F. B. Edelmann. 2003. Evaluation of landscape models for wolverines in the interior Northwest, United States of America. Journal of Mammalogy 84:92–105.

Sæther, B. E., S. Engen, J. Persson, H. Brøseth, A. Landa, and T. Willebrand. 2005. Management strategies for the wolverine in Scandinavia. Journal of Wildlife Management 69:1001–1014.

Schwartz, M. K., J. P. Copeland, N. J. Anderson, J. R. Squires, R. M. Inman, K. S. McKelvey, K. L. Pilgrim, L. P. Waits, and S. A. Cushman. 2009. Wolverine gene flow across a narrow climatic niche. Ecology 90:3222–3232.

Singleton, P. S., W. L. Gaines, and J. F. Lehmkuhl. 2002. Landscape permeability for large carnivores in Washington: a geographic information system weighted-distance and least-cost corridor assessment. Research Paper N-549. Pacific Northwest Research Station, U.S. Department of Agriculture, Forest Service, Portland, Oregon.

Squires, J. R., J. P. Copeland, T. J. Ulizio, M. K. Schwartz, and L. R. Ruggiero. 2007. Sources and patterns of wolverine mortality in western Montana. Journal of Wildlife Management 71: 2213–2220.

Vangen, K. M., J. Persson, A. Landa, R. Andersen, and P. Segerström. 2001. Characteristics of dispersal in wolverines. Canadian Journal of Zoology 79:1641–1649.

White, J. A., H. G. McDonald, E. Anderson, and J. M. Soiset. 1984. Lava blisters and carnivore traps. Special Publication of the Carnegie Museum of Natural History 8:241–256.

Wolverine Foundation. 2009. Available from (http://www.wolverinefoundation.org/specacct.htm#distribution, accessed June 2009).

Personal Communication

R. Brown
Meteorological Service of Canada
Climate Research Branch
2121 TransCanada Highway
Dorval, Quebec
H9P 1J3