The Value of Second Growth Forests to Biodiversity

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Acknowledgements

This project was developed by Darryn McConkey and Val Schaefer and supervised by Val Schaefer, Academic Administrator of the Restoration of Natural Systems Program at the University of Victoria. I would like to thank Val for his assistance and guidance. Darryn McConkey of the B.C. Ministry of the Environment, Environmental Stewardship, Ecosystems Branch of Nanaimo, B.C., provided valuable assistance and input. Sara Duncan, Environmental Studies, University of Victoria, did the final revisions and editing, added potential roles of second growth forest sections in species accounts, and added the summary chapter. This project was funded jointly by the University of Victoria Office of Research Services, Knowledge Mobilization Unit, and the B.C. Ministry of the Environment. I would like to thank Joaquin Trapero, Special Programs Officer, for developing the partnership that made this project possible.
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Chapter 1 Introduction

1.1 Objectives

It is estimated that over half of all terrestrial plant and animal species live in forests (Millennium Ecosystem Assessment, 2005); therefore, the global trend of deforestation and forest fragmentation is a major threat to global biodiversity. However, the area of intensively managed second-growth forests is increasing, and there is some debate about the role these stands play in maintaining terrestrial biodiversity (Brockerhoff et al., 2008). The purpose of this report is:

a) To review relevant ecological concepts and theories as they apply to forest biodiversity;
b) To assess how second-growth forests contribute to the maintenance of biodiversity values; and,
c) To assess the use of second-growth forest in protecting species at risk on Vancouver Island and the Gulf Islands.

Relevant ecological concepts and theories as they apply to forest biodiversity conservation and management are discussed in Chapter 2., and the potential uses of second-growth forests for the conservation of biodiversity is discussed in Chapter 3. The role of second-growth forests in the conservation of species at risk is discussed in Chapter 4. Chapter 5 provides a summary of the value of second-growth forests for the preservation of biodiversity and species at risk in particular, including recommendations for second-growth management.

1.2 Defining Biodiversity

According to Article 2 of the Convention on Biological Diversity (1992), “biodiversity” is defined as “the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems, and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems.” Within forested ecosystems, biodiversity can be considered at different levels including ecosystems, landscapes, populations, species, and genetics. Complex interactions occur within and between these levels; this complexity allows organisms to adapt to changing environmental conditions and to maintain ecosystem functions (CBD, 2008). The evaluation of biodiversity requires measurable parameters that act as correlates or surrogates for biodiversity. Features of biodiversity can be
clearly defined by specific attributes (e.g., species at risk), and measures (e.g., number of individuals) (The Royal Society, 2003).

1.3 Causes of Declining Forest Biodiversity

Forest biological diversity is the result of evolutionary processes over thousands and even millions of years. An important point to remember during this discussion is that ecosystems are complex networks of interconnected organisms, the loss of any one component of an ecosystem can affect all remaining species, and this often occurs in ways we do not yet understand (Backhouse, 2000). Naturally, some existing species become extinct; the ‘background’ extinction rate is about one species per million per year (Wilson, 1992). The issue is that human activity has increased extinction rates to between 1000 and 10000 times this background level in places such as rainforests by reduction in area alone (Wilson, 1992).

1.3.1 Deforestation, Fragmentation, and Degradation of Forests

The three major anthropogenic categories of disturbance associated with declines in biological diversity in forest ecosystems are deforestation, fragmentation, and degradation. The mechanisms by which these disturbances occur include the conversion of forests to agricultural land, overgrazing, unmitigated shifting cultivation, unsustainable forest management, the introduction of invasive plant and animal species, infrastructure development, mining and oil exploration, anthropogenic forest fires, pollution, and climate change (CBD, 2008).

The case of the red-legged frog is an example of how these three categories of disturbance have affected a species at risk on Vancouver Island. Habitat loss and degradation have been suggested as the primary causes of ranid declines (Corn, 1994; Blaustein, 1994 in MWLAP, 2004k). Urbanization and agriculture has fragmented wetland habitat on southern Vancouver Island, significantly reducing the size and quality of breeding habitat. On Crown land, forest harvesting and road construction are likely the primary threats to red-legged frog habitat, as forest harvesting has been shown to affect many functions of wetlands required for breeding, including productivity, hydrology, species assemblages, and habitat (Richardson, 1994 in MWLAP, 2004k). Although red-legged frogs have been observed to use wetlands disturbed by harvesting and intersected by roads, it is unknown whether these sites produce sufficient offspring to ensure population viability, or whether they act as reproductive sinks (MWLAP, 2004k).
1.3.2 Climate Change

Climate change can have a significant effect on biodiversity because when environmental regimes and natural disturbances are pushed outside of their natural ranges of variability, changes in ecosystems will follow (Poiani et al., 2000). In terrestrial systems, enabling local and regional scale migrations will be critical for the survival of species in the face of climate change (Poiani et al., 2000); however, it has been suggested that the current trend of forest fragmentation may negatively affect the ability of species to undertake large-scale movements such as seasonal migrations and climate change-induced range shifts (Soule et al., 2004).

As an example of the type of range shifts that may occur, one climate change impact model projects a rapid expansion of the Coastal Douglas-fir (CDF) biogeoclimatic zone, with a 336 % increase in area by 2085 (Hamann and Wang, 2006). Paleoclimatic studies confirm that the CDF zone was much larger under warmer, drier historical climactic conditions (Brown and Hebda, 2002). These studies imply that the CDF zone will likely become an increasingly dominant ecosystem in B.C., and what little remains of the old-growth forests in the CDF should be protected to provide genetic source pools from which these ecosystems can expand (Wilson and Hebda, 2008). Given their current limited distribution, it is also imperative to consider the ecological restoration of degraded forested areas within this zone as a mechanism to increase the resiliency of B.C.’s ecosystems to climate change, as well as to mitigate greenhouse gas (GHG) emissions.

The appropriate stewardship of British Columbia’s forests is a key component for addressing climate change, not only for providing connectivity to enable species migration and range shifts in response to climate change, but also as a mechanism to combat climate change. Terrestrial systems absorb, cycle, store and release carbon through photosynthesis, respiration, decomposition, and burning. As a result of these processes, forests naturally remove carbon dioxide from the atmosphere and store large quantities of carbon in their biomass and soils. Forests in British Columbia have some of the highest carbon stores in Canada, averaging 311 tons per hectare; coastal forests have the highest carbon stores in B.C., containing between 600 and 1300 tons per hectare (Wilson and Hebda, 2008). The national forest inventory has also confirmed that the coastal temperate forests of B.C. have the highest biomass carbon (C) density in Canada (Power and Gillis, 2006). That said, managed harvest rotation cycles in the Pacific Northwest are typically much shorter than the pre-industrial natural disturbance return interval. This means there is a net release of carbon to the atmosphere when old-growth forests are transitioned into managed forests (Harmon et al., 1990). Most of the old-growth coastal temperate forests in the Pacific Northwest have been logged; approximately 48% of the CDF in B.C. has been converted from forests to other land cover types since settlement (Wilson and Hebda, 2008). In fact, all old-growth forest types that are either dominated or co-
dominated by Douglas-fir within the CDF are on the province’s list of rare and endangered ecosystems (Flynn, 1999).

Finally, the effects of climate change on forest biodiversity can act indirectly, cumulatively with other disturbances, or possess feedbacks or time lags. For example, climate change has been indirectly linked to changes in seabird populations via alterations in their prey species ecology (MWLAP, 2004t). In the case of the Cassin’s auklet (*Ptychoramphus aleuticus*), it has been hypothesized that declines in breeding populations are related to changes in the timing of availability and abundance of prey species caused by warming oceanic temperatures (Anderson and Piatt, 1999 in MWLAP, 2004t). Acting in conjunction with the introduction of mammalian predators on breeding colony islands that has rendered those islands unavailable for nesting, the effects of climate change have the potential to severely affect B.C.’s Cassin’s auklet populations. Unfortunately, these types of cumulative effects on B.C.’s species at risk are becoming increasingly common.

1.3.3 Invasive Species

Invasive alien species are considered the second largest threat to native species biodiversity next to habitat loss (Wilson, 1992). Ecosystem processes are likely to be affected by the invasion of novel organisms (Tilman *et al.*, 1997b). Non-native invasive species are able to alter ecosystem properties if: a) they have considerably different capacities to acquire and use resources than native species, b) they change the trophic organization at the invasion site, or c) they modify disturbance frequency and or intensity (Vitousek, 1990). For example, adding novel generalist herbivores to a system can depress producer populations and/or net primary productivity (Vitousek, 1990). Even if introduced species do not become invasive, novel species have the potential to alter community structure and ecosystem processes through predation, competition, their potential role as a pathogen, as vectors of diseases, and through their effects on water balance, productivity, and habitat structure (Drake *et al.* 1989). For example, soil nutrient availability may be diminished through the introduction of plants that produce low-quality acid litter (Vitousek 1990). Returning to the example of the Cassin’s auklet on Triangle Island (Section 1.3.2, above), the auklet may be also be threatened by introduced rabbits that may compete for or alter auklet burrowing habitat (MWLAP, 2004t).
1.4 Contributions of Second-growth Forests to Biodiversity

Biodiversity varies considerably with stand age, but not all species need old-growth forests to fulfill their habitat requirements. At different points in successional development, younger stands can provide sufficient structural complexity, species diversity, and ecological function to provide habitat for a variety of different species. For example, forest age does not appear to be important in determining the distribution of red-legged frogs at the stand level, so long as suitable vegetation and coarse woody debris are present to provide cover, and breeding wetlands are in close proximity (MWLAP, 2004k).

Although initial regrowth may be relatively homogenous, heterogeneous forest ecosystems emerge when underlying edaphic and microclimatic gradients (Samuels and Drake, 1997), combined with differential success of colonizing species in microsites, eventually cause local divergence in species composition and distribution (Harrelson and Matlack, 2006). The precise time scale of compositional divergence is unclear (Harrelson and Matlack, 2006); though, as time from disturbance increases, second-growth communities follow a trajectory of increasing richness and changing composition that approaches nearby primary forests (Flinn and Marks, 2004).

At a landscape scale, second-growth forests can contribute to biodiversity within an environmental matrix of different land use types through habitat supplementation or complementation, by improving connectivity between remnant forests, and by buffering remnant forests from edge effects (Brockerhoff et al., 2008). Furthermore, specific management tools can be applied to second-growth forests to recruit various habitat characteristics, such as wildlife trees, in a much shorter time frame than would occur with natural regeneration alone.

Habitat recruitment will be discussed in more detail in Section 5.

1.4.1 Habitat Supplementation or Complementation

Some species that survive in remnant forest patches may compensate for habitat loss by using resources in the surrounding landscape matrix (Brockerhoff et al., 2008). This is because, as stated by the patch-matrix-corridor model (Section 4.3), fragmented landscapes are composed of gradients in habitat conditions for forest dwelling species. If, for example, there is both mature second-growth forest and pasture land adjacent to a remnant patch of old-growth forest, forest fauna are more likely to hunt and/or breed in the second-growth forest than the pasture, since the second-growth forest has more structural similarity to primary forest (Gascon et al., 1999). In a study done by Gascon et al. (1999), a substantial number (40-80%) of
nominally primary forest species were observed using matrix habitats; however, most of these observations were made in areas close to large forest tracts which provide potential sources of immigration. Furthermore, the capacity of matrix habitat to support forest dwelling species is largely determined by the history and intensity of land use (Lawton et al., 1998).

1.4.2 Connectivity

Connectivity refers to the degree to which a site or landscape is connected for a specific species. Forest habitat value and threshold of connectivity for any given species depends on the abundance and spatial arrangement of their habitat, and the dispersal capabilities of the species (Dale et al. 2000); the species-specific ability of organisms to move, disperse, migrate, or recolonize is related to both life history characteristics and ecological processes (Hansen and Urban, 1992). Connectivity is a threshold dynamic; incremental losses of habitat will result in incremental effects on the presence and abundance of species; however, at some point, a threshold is passed and the effects on species presence and abundance are dramatic (Dale et al., 2000).

The presence of second-growth forests, both plantations and those that have naturally regenerated, can enhance indigenous biodiversity in fragmented landscapes by improving connectivity between indigenous forest remnants (Hampson and Peterken, 1998). For example, in the case of the red-legged frog, the most critical component for terrestrial habitat is likely sufficient cover and, on a larger scale, connectivity and distance between wetlands; i.e., having wetlands less than 850 m apart is considered to be of the utmost importance to the maintenance of metapopulation dynamics in the landscape. Networks of interconnected wetlands serve to increase dispersal of juvenile frogs and buffer against temporal variation in productivity of individual wetlands or stochastic events (MWLAP, 2004). Likewise, the protection of a network of natural forest remnants such as riparian areas may strengthen the likelihood of survival for species for which plantations and similar land cover types are less suitable habitat.

1.4.3 Buffering

Buffers mitigate potential negative impacts from incompatible land uses that occur adjacent to wildlife conservation areas. In addition, beach, estuary, and forest buffers can provide connectivity between watersheds, and riparian buffers can provide elevational linkages within watersheds for wildlife. For the purposes of this report, two types of buffers can be distinguished; riparian area buffers and forest buffers.
Riparian area buffers of sufficient width are important because they intercept sediment, nutrients, pesticides, and other materials in surface runoff and shallow subsurface water flow (USDA, 1997). The woody vegetation within these buffers provides food and cover for wildlife, helps lower water temperatures by shading waterbodies, and slows out-of-bank flood flows (USDA, 1997). In addition, the vegetation closest to the waterbody provides litter fall and large woody debris (LWD) important to aquatic organisms, while riparian root systems stabilize and increase the resistance of banks to erosion from high water flow events, including flash floods in urbanized environments (USDA, 1997).

Second-growth forests adjacent to old-growth remnants, sensitive ecosystems, and/or critical wildlife habitat can also be used to help maintain the attributes that characterize these areas. Forest buffers reduce edge effects, extend the effective size of core areas, reduce the potential for invasion by organisms adapted to edge environments, enhance forest interior habitat, reduce the likelihood of blowdown within core areas, and reduce disturbance of important sites such as nest and breeding areas. Other potential benefits may include a temporary refuge for plants and animals after natural disturbances within core areas, and a source of replacement species for old-growth core areas lost to catastrophic disturbances. Furthermore, if one assumes that many areas on crown land adjacent to critical wildlife habitat will eventually be cut, we can also assume there will be greater edge effects on those areas in the future than occur at present (Miadenoff et al., 1994). Forest buffering is a tool that can be employed to guard against this problem.
2 Ecological Concepts

2.1 Natural Variability

Landscapes are often characterized by spatial heterogeneity that is the result of physical setting, biological agents, processes of disturbance, and stress (Pickett and Rodgers, 1997), as well as human agency. The original physical template of the landscape reflects the geology, geomorphology, and soils of the site. Organisms affect landscape heterogeneity through their growth, interactions, and legacies or ecological memory. Disturbance affects the structure of a site through physical force, and humans affect the site in a number of both direct and indirect ways. Alternatively, spatial heterogeneity helps control biodiversity by both creating and closing opportunities for organisms, and influences nutrient cycling by generating barriers or pathways to the flow of energy and materials (Pickett et al., 1997).

Natural disturbances such as fire, wind throw, insect and disease outbreaks, mass wasting, surface erosion, and catastrophic events are particularly important because they control forest succession and affect soil productivity (Maynard, 2002). Ecosystem attributes vary naturally in response to fluctuations in natural disturbance and climate. For example, the dominant natural disturbance agent in Canada’s boreal forest is fire, and variable fire rates over time and space create a dynamic patchwork of forest types that provide habitat for a diverse assemblage of species. From the perspective of species at risk management, it is important to understand that within certain limits, changes in the population patterns of plant and animal life are normal. Management activities must recognize this and attempt to mimic this natural range of variability.

Natural ranges of variability can be determined by reconstructing historic patterns and processes. Methods include simulation modeling, historic accounts such as early land surveys, interpretation of historical air photos, and paleoecological studies of sediments, charcoal, tree rings and pollen (Poiani et al., 2000). Additionally, thermographs, rainfall hyetographs, hydrographs, and outputs from simulation models can be statistically summarized to describe natural ranges of variability (Baker, 1992; Morgan et al., 1994; Richter et al., 1996). However, when data are not available regarding historical patterns and processes, deductions with respect to cause and effect relationships may be drawn from reference ecosystems and similar
organisms (Arcese and Sinclair, 1997). For example, although little is known about the specific biology of the northern pygmy-owl on Vancouver Island, information can be inferred from the data available on other races of this species.

2.2 Ecosystem Composition, Structure and Function

An ecosystem can be defined as a physical environment and suite of organisms in a specific area that are functionally linked (Pickett et al., 1997). Forest ecosystems are typically quantified according to compositional, functional, and structural attributes. Forests as habitat may be broadly defined as the range of environments suitable for a given species (Fischer and Lindenmayer, 2007). Each species responds individualistically to a range of processes connected to its needs for food, shelter, space, climactic conditions, and interspecific processes such as competition, predation, and mutualism (Fischer and Lindenmayer, 2006). Thus, responses to landscape features are often species-specific.

Composition refers to the variety and proportion of various species present and represents a major component of biodiversity (Franklin et al., 2002). An example of its influence is clear in how the species and density of plants interact to drive local evapotranspiration rates (Eviner, 2004). The importance of composition will be discussed in more detail in Section 2.2.

Function refers to the work carried out by an ecosystem and is a general term used to describe a suite of processes such as primary production, ecosystem respiration, biogeochemical transformations, information transfer, and material transport that occur within ecosystems and link the structural components (Grimm et al., 2000). The function of an ecosystem, or rates of key ecosystem processes, is limited by the structure of the ecosystem. For example, primary production is limited by soil nutrients, temperature, and soil water availability, and these factors are mediated by local climate and weather conditions. Function can be thought of as an integrated measure of what a unit (ecosystem or part of ecosystem) does in the context of its surroundings (Grimm et al., 2000). Ecosystem function is generally quantified by measuring the magnitude and dynamics of ecosystem processes or rates and directions of energy transfer. For example, primary production can be measured through biomass accumulation.

Structure refers to the component parts of the system including both the variety of individual structures such as trees, snags, and coarse woody debris of various sizes and conditions, as well as the spatial arrangement of these structures, such as whether they are evenly spaced or clustered (Franklin et al., 2002). An alternative perspective is that forest structure is the physical stage on which ecological variables interact. Forest structure can mediate communities within the stand by providing resources for and influencing interactions between organisms;
stand structure can also influence ecosystem processes through its modification of environmental conditions and resource availability (Byrne, 2007). For example, it is known that above ground structure mediates soil temperature through interception and absorption of solar radiation and its ability to transfer heat energy into the soil (Geiger et al., 2003). It is hypothesized that forest stand structure affects resource pools both directly and indirectly. A direct relationship would be when the vegetation provides resources in the form of litter and roots. An indirect relationship would be when the vegetation influences the abundance of detrivores, reducing the amount of litter (Byrne, 2007). Furthermore, parameters of vegetation structural complexity such as vegetation type, height, and coverage may not be significant by themselves, but how they interact may be significant. For example, in carabid beetle diversity study (Brose, 2003), beetle diversity was not correlated with any one of these variables, yet the correlation with the multivariate structural gradient was highly significant.

Importantly, all three ecosystem attributes (composition, structure, and function) change during the successional development of a forest stand. It is necessary to keep in mind that patches are not static; they are dynamic entities that change through vegetation succession, plant and animal dispersal, physical disturbance etc. (Pickett et al., 1997). Ecosystem structure is dictated by ecosystem processes that control and limit the transformation of material, energy, organisms, and information in and across ecosystems, and ecological processes function at many different time scales, for example, decomposition occurs over hours to decades, whereas soil formation occurs on a scale from decades to centuries (Dale et al., 2000).

2.3 Functional Diversity vs. Species Diversity

There are three mechanisms by which species diversity, and by extension a significant component of composition, contributes to ecosystem function (Petchey, 2000). First, communities with many species are more likely to contain species with particularly unique traits. Second, communities with many species contain a greater range of species traits and therefore use resources more completely. Lastly, communities with many species are likely to have a higher frequency of facilitative interactions between species.

The biotic component of an ecosystem is often broken down into functional trophic groups; plant producers, consumers that feed on plants and each other, and decomposers. Functional groups are guilds of species that are classified on the basis of intrinsic physiological and morphological differences, resource requirement differences, seasonality of growth, or life history (Tilman et al., 1997b). If, for example, species are classified according to trophic groups, these groups may be further subdivided based on life history. This is because differences in life forms affect ecosystem properties and processes such as nutrient flow; perennials maintain
storage pools of energy and nutrients for subsequent growing seasons, while annuals only have seed storage and thus are wholly dependent on photosynthesis and nutrient uptake (Vitousek, 1990).

Recent studies by Tilman et al. (1997b) address the influence of functional diversity on ecosystem processes. They found that: the functional group component of diversity is a greater determinant of ecosystem function then the species component of diversity, factors that alter ecosystem composition are likely to impact ecosystem function, all species do not contribute to ecosystem function equally, and different ecosystem processes are likely to be affected by different functional groups and species. Petchey (2000) confirmed this, and found that communities containing species from different functional groups have higher levels of ecosystem function, as well as higher variation in ecosystem function. However, the research by Tilman et al. (1997b) shows that species diversity and functional diversity are correlated; each is significant by itself, as is species diversity within functional groups. The results of this research indicate that species losses from managed ecosystems may have significant effects on the productivity and sustainability of those ecosystems (Tilman et al. 1997a); thus species at risk should be protected.

The importance of both functional diversity and species diversity is evident in forest ecosystems. For example, tree species diversity contributes to ecosystem structure and function when species with different life forms and autecology are included, such as species of both evergreen and deciduous behaviors, and shade-tolerant and shade-intolerant habits (Franklin et al., 2002). Many forests have a lower tree stratum composed of species with limited height potential such as Pacific yew (Taxus brevifolia) in coastal Douglas-fir (Pseudotsuga menziesii) forests, and this lower stratum may make unique contributions to ecosystem function (Franklin et al., 2002). Furthermore, a variety of tree species produce snags and logs that differ widely in decomposition rates and patterns resulting in higher structural diversity (Harmon et al., 1986).

2.4 Deterministic and Stochastic Threats

Species declining as a result of human induced landscape change are generally affected by both deterministic and stochastic threats (Fischer and Lindenmayer, 2007). Exogenous deterministic threats encompass direct negative impacts on a species’ habitat and include landscape fragmentation, degradation, and deforestation. Whereas endogenous deterministic threatening processes are the indirect consequence of modified landscapes and include disruptions or changes in biology, behavior, and interactions with other species (Fischer and Lindenmayer, 2007). For example, with respect to the red-legged frog, roads can have both exogenous
deterministic population impacts, as well as endogenous deterministic indirect impacts. An example of an exogenous deterministic threat is road mortality, which may be common in urban environments (Waye, 1999 in MWLAP, 2004k), and where roads cross important dispersal or migration routes and are heavily used (Fahrig et al., 1995 in MWLAP, 2004k). An example of an endogenous deterministic effect is a change in wetland hydrology, which has important implications for breeding success (MWLAP, 2004k), due to adjacent road construction. Exogenous stochastic threats are related to environmental variability; examples are fluctuations in climate and natural disturbances such as hurricanes (Simberloff, 1988). Endogenous stochastic threats include yearly variability in reproductive success as well as genetic drift, and occur as part of a species’ life cycle (Fischer and Lindenmayer, 2007). Notably, endogenous stochastic threats are more pronounced in smaller populations (Keller and Waller, 2002). These concepts are important for understanding how anthropically-induced interacting factors may compromise the viability of a particular species or population, particularly in the context of management plans where the causes of species declines are multi-factoral and potentially interacting.

2.5 Habitat Degradation

Habitat degradation may be generally defined as the gradual deterioration of habitat quality for a given species (Fischer and Lindenmayer, 2007). Habitat degradation is significant because, for example, it may cause species to occur at lower densities (Felton et al., 2003), or may make them unable to breed (Battin, 2004). In forest ecosystems, habitat degradation may be difficult to detect because it can take a long time to manifest. For example, it can take decades, even centuries, for a tree to complete the cycle of germination, maturation and decay after a clear-cut (Thompson et al., 2005), and this has powerful implications for both snag and coarse woody debris recruitment.

Habitat degradation that is the result of forest harvesting is different than that caused by other events such as acid rain. Generally, native vegetation is cleared first in areas that are characterized by high primary productivity (Norton et al., 1995). The influence of harvesting on productivity varies greatly among species or forest type, time of harvest, and inherent soil properties (Maynard, 2002). Nutrient losses due to biomass removal may be substantial after harvesting. The magnitude of the loss depends on the type of harvest (whole tree vs. boles only), harvest system used (selective logging vs. clear cut), forest type, and time of year (Maynard, 2002). In general, the influence of forestry practices on nutrient loss is long-term, and harvesting rotation times that are less than the time it takes to replace lost nutrients will eventually reduce soil productivity (Maynard, 2002). Other harvesting impacts on soil include soil compaction, displacement, and organic matter loss associated with roads, skid trails, and
landings. Compaction is particularly important because it negatively affects soil structure, aeration, water infiltration, runoff, and surface erosion, all of which reduce soil productivity (Maynard, 2002). Furthermore, recovery after compaction has been found to vary from several years to several decades in boreal, temperate, and tropical forests (Grigal, 2000; Kozlowski, 1999). This is significant because soils are the medium on which most vegetation grows.

2.6 Edge Effects

Forest stand interiors have different conditions and favour different species than their edges; the amount of interior versus edge habitat is a function of stand size and shape. In order to assess the inputs and outputs of materials, energy and influences that can affect the structure and function of a forest ecosystem, it is necessary to define the boundary of that ecosystem. Likens (1992) states that boundaries between ecosystems can be delineated for convenience or at locations where the rates of key ecological processes change. Another potentially useful concept is landscape autonomy, which may be defined as the quality or condition of being independent or self-determining (Pulliam and Johnson, 2002). The degree of autonomy of a system is essentially determined by the relative magnitude of inputs and outputs in relation to the size of the system under consideration (Pulliam and Johnson, 2002). Regardless of how a boundary is defined, it is important to remember that boundaries in ecological systems do not encompass all of the influences important to that ecosystem.

Edge effects can be defined as changes in abiotic and biotic conditions at an ecosystem boundary or within adjacent ecosystems (Fischer and Lindenmayer, 2007). The statistical significance of edge effects on forest structure and composition depends on the degree to which edge habitat can be distinguished from interior forest habitat (Harper et al., 2005). Abiotic edge effects include changes in physical variables such as radiation, moisture, temperature, humidity, wind speed, and soil nutrients (Matlack, 1993). Biotic edge effects include changes in biological variables such as species composition, and patterns of competition, predation, and parasitism (Malcolm, 1994). An example of these potential edge effects on a species at risk can be seen in the case of the Queen Charlotte goshawk. In this case, habitat fragmentation may result in other raptors better suited to edge habitats - such as red-tailed hawks (Buteo jamaicensis), great horned owls (Bubo virginianus), and barred owls (Strix varia) - out-competing goshawks for nest sites. In addition, predation rates on adults and young may increase as nest and roost sites become more accessible to edge-dwelling predators such as great horned owls, raccoons (Procyon lotor), American marten (Martes americana), and fisher (Martes pennanti). In fragmented Wisconsin landscapes, Erdman et al. (1998 in MWLAP, 2004g) documented increased competition between red-tailed hawks and northern goshawk
(*Accipiter gentilis atricapillus*) populations, and increased goshawk nest predation rates by great horned owls and fisher.

The direct and immediate results of edge creation in forested ecosystems includes damage to vegetation, disruption of the forest floor and soil, increased dispersal of seeds and pollen, and changes in evapotranspiration rates, nutrient cycling, and decomposition (Harper *et al*., 2005). Secondary process responses subsequently affect understory structure and species composition (Harper *et al*., 2005). Edge influence has a larger effect on species composition than any other secondary response; typical compositional changes involve an increase in abundance of invasive exotic and shade intolerant species, and a reduction in abundance of shade tolerant species (Harper *et al*., 2005).

The degree of penetration and magnitude of edge influences are functionally independent and do not necessarily respond to edge creation and subsequent dynamics in the same way (Harper *et al*., 2005). For example: at edges with persistent wind effects, the distance of penetration of edge influences may increase over time even as the magnitude of edge effects declines (Laurance *et al*., 2002). Alternatively, in areas where the contrast between forested and non-forested communities are maintained, the secondary responses of edges between those environments often result in a ‘wall’ of dense vegetation which may reduce the depth of penetration of matter and energy into the remnant forest, while the magnitude of the edge influence remains strong (Laurance *et al*., 2002). If patch contrast is rapidly reduced due to regeneration, both the distance and magnitude of edge influences may be reduced (Harper *et al*., 2005). Thus, the magnitude of edge influences are likely to be enhanced when there is high structural contrast at the edge, high wind speeds and temperatures, and when pioneer, exotic, and invasive species are present that may be successful in these environments (Harper *et al*., 2005). Meanwhile, the penetration of edge effects is expected to be a function of the magnitude of edge influence, solar angle, and inherent heterogeneity of the forest community (Harper *et al*., 2005). In other words, the ecological impact of edge influences should be less important in forests that are more structurally heterogeneous (Harper *et al*., 2005).

3  **Forest Structure**

3.1  **Structural Attributes**

Structural attributes of forest stands are both theoretically and practically important for understanding and managing forest ecosystems. This is because structure is the attribute most often manipulated to achieve management objectives following stand establishment;
is a measurable surrogate for functions such as productivity, or organisms such as cavity dwelling birds that are difficult to measure directly; and has direct value in terms of products such as timber and in providing services such as carbon sequestration (Franklin et al., 2002). Some structural features of forest stands, including individual structural elements and spatial patterns of structural elements, are presented in Table 1.

Table 1. Important Structural Features of Forest Stands (Adapted from Franklin et al., 2002).

<table>
<thead>
<tr>
<th>Individual Structures</th>
<th>Important Attributes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Live trees</td>
<td>Species, density, mean diameter, range in diameter, height, canopy depth</td>
</tr>
<tr>
<td>Large-diameter live trees</td>
<td>Species, density, decadence including presence of decay columns, crown condition, bark characteristics</td>
</tr>
<tr>
<td>Large-diameter branches</td>
<td>Species, density, size, individual or arrays, presence of arboreal ‘soil’.</td>
</tr>
<tr>
<td>Lower-canopy tree community</td>
<td>Composition, density, height</td>
</tr>
<tr>
<td>Ground community</td>
<td>Composition, density, deciduous or evergreen</td>
</tr>
<tr>
<td>Standing dead trees (snags)</td>
<td>Species, size, decay state, density</td>
</tr>
<tr>
<td>Coarse woody debris (logs)</td>
<td>Species, density, decay state, volume, mass</td>
</tr>
<tr>
<td>Root wads and root holes</td>
<td>Density, size, age</td>
</tr>
<tr>
<td>Organic soil layers</td>
<td>Depth, chemical and physical properties, biota</td>
</tr>
<tr>
<td><strong>Spatial Patterns</strong></td>
<td></td>
</tr>
<tr>
<td>Vertical distribution of foliage/ canopy</td>
<td>Depth, continuity, cumulative distribution</td>
</tr>
<tr>
<td>Horizontal distribution of structures</td>
<td>Spatial pattern: random, dispersed, or aggregated</td>
</tr>
<tr>
<td>Gaps and anti-gaps</td>
<td>Size, shape, density</td>
</tr>
</tbody>
</table>
3.2 Old-growth Forests

Old-growth forests supply critical habitat for some wildlife species. Old-growth forests have been defined in terms of stand structure (e.g. Franklin et al., 1981), stand development processes (e.g. Oliver and Larson, 1996), and tree age (e.g. MacKinnon and Vold, 1998). For the purposes of this discussion, old-growth forest will be defined according to the MacKinnon and Vold (1998) B.C. inventory: old-growth forests are those that are greater than 250 years old on the coast and greater than 140 years old in the interior, except for lodgepole pine forests, which are considered to be old-growth at greater than 120 years old. These ages reflect the point at which structural and biological characteristics associated with old-growth forests begin to develop.

There are many pathways to natural forest development. However, many old-growth forests have developed under the influence of moderate to severe natural disturbances such as wildfires, windstorms, droughts, floods, and insect or disease outbreaks. After a natural disturbance, patches of living and many recently fallen trees remain, often preventing the regeneration of dense stands of uniform young trees. In old-growth forests, this means that the dominant trees vary considerably in age. This is because they started growing anywhere from several years before to decades after the disturbance that either killed the stand or created canopy gaps.

During the development of old-growth forests, young trees that were previously part of the understory generally grew rapidly after a disturbance due to wide spacing and the consequent lack of competition from other trees. This is supported by tree ring studies that show that in many old-growth forests the dominant trees frequently underwent rapid diameter and height growth in their first 50 to 80 years (Rapp, 2002). Furthermore, due to the open space around the tree crowns, these trees kept more branches then they would have if other trees had been growing close to them (Rapp, 2002). The result of this process is the development of crowns that are both wide and deep.

As these stands aged, the canopies began to close again. When old trees died, they created gaps in the canopy and, in some cases, deep crown redeveloped through epicormic branching (Rapp, 2002). For example, older Douglas-firs develop branches from dormant buds on their trunks when they are exposed to light (Rapp, 2002). These branches, along with shade tolerant saplings, contribute to the creation of a bottom-loaded canopy. Thus, the complex structure of old-growth forests is the result of variability. Variability in spacing allows some trees to grow rapidly and keep more live branches, and patchy mortality makes holes in the developing forest that allows other trees to grow. The result of these processes is a forest with many tree species,
ages, and sizes (Rapp, 2002). Table 2 shows some of the pertinent structural processes that are operational during the successional development of forest stands in the approximate order of their first appearance.

Table 2. Structural Process During Successional Development of Forest Stands (Adapted from Franklin et al., 2002)

<table>
<thead>
<tr>
<th>Structural Process</th>
<th>Additional Description</th>
<th>Typical Stand Age</th>
</tr>
</thead>
<tbody>
<tr>
<td>Disturbance and legacy creation</td>
<td></td>
<td>0</td>
</tr>
<tr>
<td>Establishment of new cohort of trees or plants</td>
<td></td>
<td>Between 0-10 years</td>
</tr>
<tr>
<td>Canopy closure of tree layer</td>
<td></td>
<td>Between 20-35 years</td>
</tr>
<tr>
<td>Competitive exclusion of ground flora</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lower tree canopy loss</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Biomass accumulation</td>
<td></td>
<td>Between 40-100 years</td>
</tr>
<tr>
<td>Density dependant tree mortality</td>
<td>Thinning mortality due to competition among tree life forms</td>
<td>Between 40-100 years</td>
</tr>
<tr>
<td>Density independent tree mortality</td>
<td>Mortality due to agents such as wind, disease or insects</td>
<td></td>
</tr>
<tr>
<td>Canopy gap initiation and expansion</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Generation of coarse woody debris and snags</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Uprooting of trees</td>
<td>Toppled trees cause soil disruption as well as the creation of structures such as root wades</td>
<td></td>
</tr>
<tr>
<td>Understory re-development</td>
<td>Shrub and herb layers</td>
<td></td>
</tr>
<tr>
<td>Establishment of shade tolerant tree species</td>
<td>Assuming the pioneer cohort are shade-intolerant species</td>
<td></td>
</tr>
<tr>
<td>Structural Process</td>
<td>Additional Description</td>
<td>Typical Stand Age</td>
</tr>
<tr>
<td>-------------------------------------------</td>
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<td>Shade patch development</td>
<td>Anti-gap development</td>
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<td>Maturation of pioneer tree cohort</td>
<td>Achievement of maximum crown height and crown spread</td>
<td>Between 100-200 years</td>
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<td>Canopy elaboration</td>
<td>Development of multi layered or continuous canopy</td>
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<td>through: a.) growth of shade tolerant species into co-</td>
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<td>dominant canopy position; b.) re-establishment of lower</td>
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<td>Development of large branches and large</td>
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<td>communities on large branches</td>
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<td>Pioneer cohort loss</td>
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<td>Between 800-1200 years</td>
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Older stands generally provide better habitat than younger stands because of their increased spatial and vertical heterogeneity, well-developed soil organic layers and associated fungal florals, increased large woody debris, better light environment, and inter-specific facilitation (Brockerhoff et al., 2008). Old-growth structural attributes include a diverse tree community, large dominant trees with substantial crowns, smaller shade tolerant trees, multilayered canopies, snags or wildlife trees, canopy gaps, and abundance of coarse woody debris, and understory patchiness with some shrub and herb development (Franklin et al., 1981).
3.3 Wildlife Trees

The most significant contribution of snags in forest ecosystems is their function as wildlife trees. “A wildlife tree is any standing dead or alive tree with special characteristics that provide valuable habitat for the conservation or enhancement of wildlife” (Thompson et al., 2005). In fact, more than 80 species of vertebrates and countless invertebrates depend on dead or deteriorating trees at some point in their life cycle (Thompson et al., 2005). Some of their uses of wildlife trees are nesting, feeding, communicating through drumming and marking, roosting, shelter, and over-wintering (Thompson et al., 2005). An example of a species at risk that requires wildlife trees is the Keen’s long-eared myotis (Myotis keenii). Tree cavities in wildlife trees with a decay class of 2 or greater, and loose bark on wildlife trees with a decay class of four or greater, are important natural roost sites for this species and may be limiting in part of their range (MWLAP, 2004d).

Within a natural stand, there are often multiple species of trees in many different sizes and conditions. It usually requires 100 or more years to recruit trees of sufficient size and condition to function as useful wildlife trees when relying on natural processes (Manning, 2006). Large old trees with features such as multiple or dead tops, bole or top decays and cavities contribute to the structural diversity within the stand. Examples of habitat features found in wildlife trees include natural cavities in bigleaf maples (Acer macrophyllum), the “chimney effect” in western redcedars (Thuja plicata), live hardwood with primary cavity excavation and feeding holes, heart rot and loose bark in grand fir (Abies grandis), and nesting and perching sites on broken topped Douglas-fir snags (Thompson et al., 2005). In other words, specific features such as decay cavities, large diameter branches, and distinctive bark features might be explicitly recognized because of unique functional and habitat roles (Franklin et al., 2002).

Wildlife trees are generally created through mortality agents such as fire, disease, insects, windthrow, snowpress, lightning, and wildlife excavation. Death by fire produces a very different type of wildlife tree than gradual death by insects or disease, and tree species and local climate also influence the way a tree deteriorates and decays (Thompson et al., 2005). The most significant indicators of wildlife tree quality are height, diameter, decay stage, location, distribution, and cause of death; but, the value of any particular tree for a given species depends on a variety of attributes, the most important of which are structure, age, condition, abundance, tree species, geographic location, and surrounding habitat features (Thompson et al., 2005).
3.4 Coarse Woody Debris

An important structural feature of a wide variety of old-growth forests in North America is a relatively large quantity of downed big logs (Sturtevant et al., 1997). Piece size, species (Harmon et al. 1986), and disturbance history (Spies et al., 1988) all affect the speed and type of decay of any CWD piece; while abundance, size, state of decay, and spatial distribution are all factors affecting the use of Coarse Woody Debris (CWD) by wildlife species (Keisker, 2000).

CWD is an important nutrient source and growing substrate for numerous species of bacteria, fungi, saprophytic plants, lichens, and mosses that are essential in decay, nitrogen production, and other nutrient and moisture cycling processes (Thompson et al., 2005). For example, in one study of 6 Biogeoclimatic Subzones in British Columbia, 70% of the 243 plant species recorded grew on CWD, with 23% of those species being restricted to CWD (Song, 1997). Other important functions of CWD include carbon storage, erosion control, buffering microclimates suitable for seedling establishment, cover from predators, shelter and access routes for small mammals during heavy snow cover, and contributions to stream stability, complexity, and geomorphology (Thompson et al., 2005).

Coarse woody debris also provides feeding, breeding, and shelter substrates for many invertebrates, small mammals, and amphibians (Thompson et al., 2005). Dupuis et al. (1995) found three to six times more *Plethodon vehiculum* salamanders in old-growth forests than in younger second-growth forests in coastal B.C.; this was partly attributed to the availability of CWD. Also, Davis (1998) found that the two salamander species observed in coastal B.C. used different decay classes of CWD. This suggests that old-growth forests are more likely to provide suitable habitat for both species than single-aged managed forests because old-growth forests generally provide a range of decay classes.

3.5 Mycorrhizal Fungi

Mycorrhizal fungi form symbiotic relationships with many forest plants. These fungi grow on plant roots and absorb sugars the plant produces by photosynthesis. In return, the plant gains access to water and nutrients that the fungi absorb from the soil through a network of filaments (Flynn, 1999). The importance of mycorrhizae cannot be overstated – plant growth is greatly increased in the presence of these fungi, and some plants cannot grow without their fungal partner. Mycorrhizal fungi are considered to be the keystone of coastal Douglas-fir forests (Flynn, 1999).
3.6 Riparian Areas

Riparian areas are included in this report because they are an important habitat component for several identified wildlife species. Riparian areas are recognized as unique habitats, and are frequently associated with specific flora, fauna, physiography, or microclimate processes not common in the adjacent forest. Riparian areas are characterized by high site productivity and a complex habitat structure; riparian areas usually containing downed wood, snags, shrubs and mixed tree species composition. Furthermore, riparian areas are often the most heavily used wildlife habitats. This is in part because they act as linear travel corridors for wildlife, providing valley bottom and cross-elevational connectivity. Additionally, riparian areas usually have a buffered microclimate compared to adjacent areas: they are often warmer in the winter and cooler in the summer compared to the surrounding forest stand.

3.7 Second-Growth Plantation Forests

Many second-growth forests are plantations. Logging and subsequent planting imitate natural forest development processes in some ways, but are not the same as natural processes (Rapp, 2002). This is in part because plantations are planned to maximize timber production. Plantation forests are created through the planting of one or more tree species and are typically of an even aged structure with an even spacing of trees. Hand-planted second-growth stands are typically much denser than stands natural processes would have created, and plantation forests generally contain few, if any, biological legacies such as snags and fallen trees or CWD (Rapp, 2002). This generally translates into less habitat diversity and structural complexity than more natural forests. Plantation forests can be expected to be more like natural forests if they are composed of locally occurring native tree species, however, even plantations of exotic trees may have an understory composed of native flora and fauna.

Recently, plantations have been explored as a mechanism for fixing carbon. Forest plantations may sequester some carbon while living; however, due to the simplified nature of these forests, it is more likely that at some point they will succumb to disease, insect outbreaks and/or fire and therefore may not be considered reliable carbon offsets (Wilson and Hebda, 2008). Alternatively, healthy, functioning, diverse ecosystems tend to be more resilient and therefore less vulnerable to normal periodic stress events in the local environment. In the case of forest plantations, there is the potential to restore these areas to a healthy functioning state, but an important question is: how do we restore a plantation forest so that it will simultaneously maximize carbon sequestration while maintaining biodiversity and ecosystem services? There appears to be a lack of strong relationship between carbon pools and individual variables across different ecosystem types. This suggests that there is a complex interplay between climate,
species composition, stand age, and soil properties on ecosystem carbon sequestration (Homann et al., 2005). Thus, any restoration plan that aims to maximize carbon sequestration while maintaining biodiversity values must be ecosystem specific.

Many plantations are intensively managed; they may have undergone site preparation such as ploughing, harrowing, use of fertilizers and herbicides, and may be subjected to silviculture practices such as thinning and the elimination of competitive woody understory regeneration (Brockerhoff et al., 2008). For example, forest fertilization is often employed because it accelerates overall stand development by increasing bole diameters and canopy closure, and by accelerating understory brush dieback and self-pruning below the canopy (Manning et al., 2006).

In the absence of management activities aimed at decreasing competition between the planted trees and natural regeneration, mono-specific plantations are often replaced by mixed forests comprised of both planted species and other floristic elements from the surrounding forest areas (Brockerhoff et al., 2008). Although natural forests generally offer superior habitat for native forest species than plantation forests, the degree to which plantation forests can provide suitable habitat for native species is dependent on management intensity and plantation species composition and structure in relation to surrounding natural forests (Brockerhoff et al., 2008). Planted forests often consist of fast-growing pioneer tree species that can potentially invade neighboring habitats (Ledgard, 2001). Furthermore, there are several groups associated with plantations such as weeds and feral animals, which can undergo similar invasion processes in surrounding remnant forests (Kupfer et al., 2006).

Plantation management practices that generally increase biodiversity values within a stand include:

a) Selecting tree species that provide resources and structures that favor native species;

b) Avoiding intensive site preparation that destroys herbaceous vegetation and coarse woody debris;

c.) Implementing wider tree spacing and heavy pre-commercial thinning to help maintain understory vegetation;

d) Increasing rotation length; and

e.) Maintaining some structural attributes such as old trees or snags (Brockerhoff et al., 2008).

Longer-rotation plantation forests managed for conservation values may have similar habitat value to managed natural forests (Suzuki and Olson, 2008). In contrast, the current trend of reduced age of forest harvesting, or decreased rotation periods, is expected to further reduce the availability of suitable breeding habitat for old-growth dependant species such as the
Queen Charlotte goshawk, since forests are harvested before they obtain the structural attributes that characterize goshawk nest stands (DeStefano, 1998 in MWLAP, 2004g). Furthermore, because Queen Charlotte goshawks typically do not forage within younger forests, access to most forest prey is reduced as the overall distribution of forest age classes across the landscape become younger (MWLAP, 2004g).

Despite the limitations of plantation forests, however, there are an increasing number of instances where species at risk have been recorded in plantations. Examples include the locally threatened hoopoe (Upupa epops), in plantation forests in the Landes region in France (Barbaro et al., 2008), and the endangered brown kiwi (Apteryx mantelli) in exotic pine plantations in New Zealand (Kleinpaste, 1990).

4 Landscape Ecology Theory

Landscape ecology provides the theoretical scientific basis for reconnecting and restoring fragmented habitats (Forman and Godron, 1986). In terrestrial ecosystems, habitat fragmentation can occur when changes in land use or land cover transform a contiguous habitat patch into disjunct patches. Southern and eastern Vancouver Island have become extensively urbanized and developed, and much of the forest in the interior of the Island and north of Vancouver has been fragmented. This is important because particular landscape patterns, such as the size, shape, connectivity and configuration of habitat remnants, have certain implications for both biotic and abiotic processes. Patch dynamics focus on the creation of spatial heterogeneity within landscapes and how that heterogeneity influences the flow of energy, matter, species, and information across the landscape (Zipperer et al., 2000).

Fragmentation is significant because it reduces the available area of forest habitat, increases the isolation of forest patches, and increases the edge effects in the remaining patches. There are two key theoretical frameworks in community and population ecology that have been used to study habitat fragmentation; the theories of island biogeography (MacArthur and Wilson, 1967), and metapopulation dynamics (Levins, 1969). The former has been used as a guide to assess the influence of patch size and isolation on species composition, whereas the latter has focused attention on connectivity and interchange between spatially distributed populations (Collinge, 1996).
4.1 Island Biogeography Theory

According to the island biogeography theory, the number of species in a remnant patch of forest is not only controlled by the habitats and resources present on site, but also by the balance of immigration and local extinction (MacArthur and Wilson, 1967). Patterns of immigration are primarily determined by distance from other sources of potential colonists. Habitat patches that are relatively close to other patches are more likely to be occupied than more isolated patches because they are likely to be recolonized after a local extinction event (Pulliam and Johnson, 2002). Larger patches can support larger populations, which are less vulnerable to extinction (MacArthur and Wilson, 1967). In other words, all other things being equal, smaller remnant patches support fewer native species than larger patches (Bellamy et al., 1996). This is not only because larger forest remnants are likely to have a higher ratio of colonizations to extinctions, but also because they are more likely to have undisturbed components necessary to some species (Harris, 1984), and are more likely to contain a range of habitats for different species (Fox, 1983).

4.2 Metapopulation Theory

Metapopulation theory can be thought of as an extension of island biogeography from habitat patches to population patches. Patchy populations are true metapopulations only if movement between sub-populations is neither very common nor very uncommon (Hanski and Simberloff, 1997). Clusters of populations may interact over time through the exchange of individuals or genetic material, and individual populations may frequently go extinct and the same area recolonized at a later time by immigrants from extant populations (Pulliam and Johnson, 2002). The dynamic nature of local extinctions and recolonization dictate that any particular patch of habitat may or may not be occupied at a given point in time, however, the metapopulation as a whole persists because some patches are always populated (Pulliam and Johnson, 2002). In addition, large populations are less likely to go extinct than small populations, and large habitat patches, which are more likely to support large populations, are more likely to be occupied than small patches. An example of the application of metapopulation theory is evident in the case of the red-legged frog on Vancouver Island. At the watershed scale, the loss of small wetlands can affect metapopulation dynamics of pond-breeding amphibians and increase the probability of extinction of populations in the remaining wetlands (Gibbs, 1993, 2000; Semlitsch, 1998 in MWLAP, 2004k). Even though small wetlands do not comprise a large area in the land base, they are often numerically dominant to large wetlands. For example, over 97% of all wetlands surveyed on the west side of Vancouver Island were <0.1 ha, and red-legged frogs were present in 26% of these wetlands (Beasley et al., 2000 in MWLAP, 1994k). In this situation, the loss of
small unclassified wetlands not only decreases the number of aquatic breeding sites, reducing the abundance or density of organisms, it also increases the nearest neighbor distance between sites, impeding source-sink processes (Gibbs, 1993, 2000; Semlitsch, 1998 in MWLAP, 1994k). This is significant because for a number of species of ranid frogs, the occupancy of wetlands is related to proximity of other breeding ponds (Laan and Verboon, 1990; Gulve, 1994; Pope et al., 2000 in MWLAP, 2004k). These results suggest nearby population sources are also important in maintaining metapopulations of pond-breeding amphibians.

4.3 Patch-Matrix-Corridor Model

An extension of the aforementioned fragmentation theories is the patch-matrix-corridor model (Forman, 1995). Within large patch and matrix landscapes, disturbances create a diverse, shifting mosaic of successional stages and physical settings of different origin and size (Bormann and Likens, 1979). The patch-matrix-corridor model is significant because it recognizes that the ability of a species to reach remnant forest patches depends on how inhospitable, or permeable, the landscape matrix surrounding the patch is (Forman, 1995). With this model we move away from the often misleading conceptualization of landscapes as areas of forest/ habitat or non-forest/ non-habitat, toward the idea that the landscape matrix surrounding remnant forest patches may be neither uniformly unsuitable as habitat nor serve as a complete barrier to the dispersal of forest taxa (Kupfer et al., 2006). Thus, the extent to which fragmentation affects a given species depends on how the landscape has been modified, what constitutes suitable habitat for the species, mode and scale of movement, and dispersal behavior (Fischer and Lindenmayer, 2007). Furthermore, the rate of recovery of an ecosystem or species at any scale following a disturbance is not only strongly influenced by the availability of nearby organisms or propagules, but also by biological legacies, such as seed banks, for recolonization (Holling, 1973).

4.4 Habitat Heterogeneity Hypothesis

The habitat heterogeneity hypothesis suggests that structurally complex patches may provide more niches and different opportunities to exploit environmental resource and thus species diversity will increase with patch complexity (Bazzaz, 1975). This concept is supported by the research done by Tilman et al. (1997a). However, there are discrepancies in the literature regarding the relationship between habitat heterogeneity and fauna diversity. This is because the relationship between patch heterogeneity in terms of vegetation architecture and animal species diversity depends on:
a) How habitat heterogeneity is perceived by the animal guild studies;
b) How species diversity is measured;
c) How habitat heterogeneity is defined;
d) How vegetation structure is measured; and,
e) The spatio-temporal scale of the study (Tews et al. 2004).

Furthermore, species diversity patterns show year-to-year and season-to-season variations, which have important implications for across-study comparisons (Tews et al., 2004).

### 4.5 Intermediate Disturbance Hypothesis

The intermediate disturbance hypothesis suggests that disturbance strongly influences patterns of species diversity, and, that species diversity is at a maximum at intermediate levels of disturbance (Huston, 1994). This is because physical disturbance prevents competitively dominant species from excluding other species from the community, and there is a trade-off between species’ ability to compete and their ability to tolerate disturbance. In other words, at low levels of disturbance only the best competitors persist; when disturbances are very intense or very frequent few species persist or repeatedly colonize after each disturbance; and, at intermediate intensities or frequencies of disturbance, conditions favor the co-existence of competitive species and disturbance tolerant species (Huston, 1994). However, while disturbance history sometimes explains a lot of the observed variation in species diversity, it does not do so consistently, and it has been suggested that the strength of the species diversity and disturbance relationship actually reflect sampling artifacts (Mackey and Currie, 2001). For example, Mackey and Currie (2000) argue that with less intense sampling (e.g. point counts), one is less likely to detect rare species: as sampling intensity increases, rare species are detected and disturbance is predicted to have less effect on richness. Thus it appears that sampling methodology, including the number of individuals censused, number of disturbance levels examined, and quadrat area sampled, has a significant influence on the relationship being observed.

Furthermore, the power of disturbance to explain variation in species diversity is greater for plants (sessile producers), then for animals (motile fauna) (Mackey and Currie, 2001). Fuentes and Jaksic (1988) suggest that strong species diversity-disturbance relationships are not frequently found among terrestrial vertebrates because of their ability to move in and out of disturbed patches and consume of many different resources. The generally high heterogeneity of patch environments also means that many terrestrial fauna do not meet the conditions required for disturbances to have a strong effect on species diversity. However, in the context
of species at risk, particularly those species that are habitat specialists versus habitat generalists, this may not be the case.

Anecdotal evidence suggests that almost all of the studies supporting the intermediate disturbance hypothesis took place in systems with high rates of population growth; i.e., systems with high levels of productivity (Huston, 1994). However, Mackey and Currie (2001) found that the odds of observing a peaked richness-disturbance relationship decreased by 97% for each 10-fold increase in actual evapotranspiration (which they used as a surrogate for terrestrial productivity), rejecting the hypothesis that intermediate disturbance relationships are associated with highly productive systems.

4.6 Conservation Area Functionality

One of the most important tasks in conserving a species at risk is determining what areas should be set aside for that species. This decision should be based, in part, on the area’s functionality or ecological integrity. Four attributes that can be examined to assess a potential conservation area’s functionality have been suggested: composition and structure of the focal ecosystems and species; dominant environmental regimes, including natural disturbance; minimum dynamic area; and connectivity (Poiani et al., 2000). Key compositional and structural components for a given species may include age structure, evidence of reproduction, population size or abundance, genetic diversity, and minimum viable population (Poiani et al., 2000).

Other important compositional and structural components for ecosystems may include abundance of invasive species, presence of species that indicate unaltered ecological processes, abundance of important prey species, evidence of reproduction of dominant species, existence of characteristic species diversity, and evidence of vertical or strata layering (Poiani et al., 2000). Important dominant environmental regimes may include grazing or herbivory, hydrologic and water chemistry regimes (e.g. surface and groundwater), geomorphic processes, climatic regimes (e.g. temperature and precipitation), fire regimes, and many types of natural disturbances (Poiani et al., 2000). The area required to ensure survival or recolonization of a given species has been termed the minimum dynamic area (Picket and Thompson, 1978). An important consideration in the creation of minimum dynamic areas is disturbance size. For example, Baker (2004) suggests that conservation areas should be large relative to maximum disturbance size to minimize their vulnerability to fatal loss of organisms, to reduce the chance of disturbance spreading into surrounding developed lands, and to minimize the influences of adjacent lands on the size and spread of the disturbance. Developing scientific estimates of
minimum dynamic area and metapopulation structure for biodiversity at different scales is one of the critical frontiers of applied conservation biology (Poiani et al., 2000).

4.7 Hierarchal Structures and Models for Understanding Complex Systems

Ultimately, a comprehensive plan for the protection of biodiversity must include all elements of biodiversity from genes to landscapes and is thus hierarchical both in spatial scales and biological levels of organization (Noss and Cooperrider, 1994). Several different conceptual hierarchies have been developed to facilitate the understanding of complex systems such as forest ecosystems. A hierarchy explains relationships within a system by ranking levels of organization. Levels may be defined by a variety of attributes including physical or spatial structure, or interaction rates. Furthermore, two types of hierarchies can be distinguished: structural and control.

Control hierarchies exist when components at one level exert control on components at a lower level that may not be a subsystem of the controlling unit and thus are considered non-nested (O’Neill, 1989). This type of hierarchy might be employed in the study of relationships between plants, herbivores, and carnivores. An example of this concept in the context of species at risk is evident in the case of the Vancouver Island Marmot (Marmota vancouverensis), which is hypothesized to be declining as a result of increased predation. At least 80% of marmot mortality since 1992 has been attributed to predation, largely by wolves (Canis lupis), cougars (Puma concolor), and golden eagles (Aquila chrysaetos) (Bryant and Page, 2005). Although these are natural predators of the Vancouver Island marmot, numbers of cougars and wolves on the island have increased dramatically since the early 1980s, likely in response to changing dear populations (Bryant and Page, 2005). This is notable because Roehmer et al. (2001) have shown that, in some cases, when the abundance of naturally occurring predators and their primary prey species change, abnormal mortality can be inflicted on secondary prey species, in this case the Vancouver Island Marmot. That said, increased levels of predation on marmots may also reflect a functional response by predators; predators may hunt more often or more successfully in marmot habitats surrounded by clear-cuts (Bryant, 1998).

A structural hierarchy, on the other hand, focuses on subsystems within systems and thus is nested (O’Neill, 1989). An example of a structural hierarchy is the relationships between genes, organisms, and populations. The most useful hierarchy to employ in classifying and analyzing the ecology of forest remnants within landscapes is the structural hierarchy because it allows for the examination of faster processes at a fine scale in site-specific environments, and the clustering of detail to expose more general slower processes at the coarser landscape scale.
The concept of a structural hierarchy has been built on by Hollings (2001), who presents the concept of “panarchy” to describe complex adaptive socioecological systems. Within this framework, systems are interlinked in infinite four-phase adaptive cycles of growth, accumulation, restructuring, and renewal. These transformational cycles occur in nested sets among variables that share similar speeds and spatial attributes at various scales in both space and time. This hierarchy lends itself well to the examination of succession in forest ecosystems. This concept is important because it helps define the role of biological legacies. That is, cycles are interlinked because at each level in the hierarchy, small amounts of information and or material are communicated to the next higher level. In the case of forests, this information is in the form of biological legacies such as snags and seed banks. Importantly, the sustainability of an adaptive forest ecosystem is determined by the functioning of the cycles, or successional stages, as well as the communication between them.
Chapter 3  Management Options to Promote Biodiversity in Second-growth Forests

5  Habitat Recruitment

Forest managers can use the data on density, growth rates, and ages of old-growth stands to design silvicultural options that could put second-growth forests on different pathways likely to lead to greater forest complexity and habitat diversity (Rapp, 2002). Management strategies employed in young forest to increase biodiversity values are based on the idea that affinity of those wildlife species occurring in late-seral forests is more likely attributable to ecological characteristics such as structure rather than the age of the forest (Hayes et al., 1997).

Second-growth stands will likely develop old-growth characteristics as a result of natural events; however, this process can take considerable time. Disturbances such as windstorms, ice storms, root rot, insect infestations, and fire will likely occur in unmanaged plantations over the course of a century; trees in very dense stands – like plantations - are often not very sturdy and are prone to blowdown (Rapp, 2002), while root rot and wind throw could create small openings leading to structural complexity if seeds of shade tolerant trees are present in the seed bank.

These somewhat unpredictable disturbances could either put these plantations on a pathway that leads to structural diversity, or kill enough trees that new stands regenerate naturally; specific stand response depends on many factors including the plantation size and the characteristics of the stands around it (Rapp, 2002). If the trees over a large area are destroyed, it can be a major setback in forest succession, thus no management is an option for forest managers that may have negative consequences for biodiversity.

5.1  Partial Cut Harvesting Systems

Partial cut harvesting systems are designed to retain individual trees or groups of trees. Examples of partial cut harvesting systems include variable retention, sheltered, seed tree, and clear-cut with reserve systems. In areas where these systems have been employed in the past, both patches and individual leave trees may be considered for long-term retention to enhance recruitment of large diameter wildlife trees.
5.2 Thinning for Specific Structural Attributes

It is possible to speed up the recruitment of late seral structural features and complexity through silviculture strategies such as pre-commercial thinning (PCT) and variable-density thinning (VDT) (Carey and Wilson, 2001). For example, in dense, uniform conifer plantations, one or more VDTs is likely to accelerate the development of some old-growth characteristics, perhaps by decades, in comparison to stands where no action is taken (Rapp, 2002). This is supported by computer simulations which suggest that heavy thinning of young Douglas-fir stands at age 15 or 30, accompanied by under planting, can accelerate the development of aspects of stand structure found in late-seral stage forests such as tree diameter, crown depth, and limb diameter (Barbour et al., 1997). Furthermore, By delaying thinning until a stand is 30 years of age and increasing residual stand density from 75 to 150 trees per hectare, it may be possible to grow stands with multi-storied structure, abundant large snag-candidate trees (in excess of 85 cm DBH at 100 years), and sufficient residual stems to carry out a second thinning to recover moderate quality wood (Barbour et al., 1997).

Thinning to develop old-growth characteristics is different than thinning to maximize timber production. This is because timber management uses evenly spaced thinning to produce uniform stands and removes just enough trees to maximize the growth in volume of the stand as a whole, rather than maximize the growth in volume per tree (Rapp, 2002). That said, stand tending such as pre-commercial thinning and pruning can have beneficial effects on wildlife habitat by affecting the volume and diameter of snags and CWD recruited into the stand. That is, stand tending decreases the future volume of CWD but increases its average future size, and spacing or thinning increases tree incremental growth, thus recruiting trees to become larger snags at an earlier age (Manning et al, 2006). Importantly, dense plantation trees are most responsive to thinning when they are less than 80 years old; options for accelerating forest structure development may diminish substantially if stands are not thinned when they are young (Rapp, 2002).

5.2.1 Pre-Commercial Thinning

PCT reduces tree competition and increases the amount of growing space per tree, which concentrates the growth potential of the site on the remaining trees (Sullivan et al., 2006). PCT produces high vigour trees with deep crowns and relatively rapid individual tree growth (Sullivan et al., 2006). Furthermore, PCT leads to an increase in abundance of understory tree classes, shrubs, and herbs, which produce an enhanced array of habitats and microhabitats in the understory of these stands (Sullivan et al., 2006).
This is evident in a study in the Oregon Coast Range where two out of three stands between 25 to 30 years of age were thinned to different densities. A few years after thinning, the stands exhibited major differences in tree size, range of variability, and understory development (Rapp, 2002). The unthinned stand tree density was 491 trees per acre, and the average diameter at breast height (DBH) was eight inches. In comparison, the most heavily thinned stand tree density was 105 trees per acre, and the average DBH was 15 inches: almost double the average in the untreated stand (Rapp, 2002). Additionally, both of the thinned stands had a greater range of variability in tree size than the unthinned stand, which was generally uniform in size (Rapp, 2002). The mixture of large and small trees in the treated stands is important because it will provide increased structural diversity as the stand ages. Finally, understory development in the unthinned stand was almost non-existent, with essentially no green plants on the ground. Alternatively, the moderately thinned stand (185 trees/acre) had a few plants on the ground, where as the heavily thinned stand had many herbs, shrubs, and saplings growing (Rapp, 2002). Thus, it is the response of crop trees to silviculture practices, in terms of diameter and height growth as well as crown architecture, which drives the development of habitat attributes such as understory composition, abundance, and stand level structural diversity (Sullivan et al., 2006). The influence of PCT on successional development may be temporary; however, the temporal scale depends on the range of thinning intensity (Sullivan et al., 2006).

5.2.2 Variable Density Thinning

Given that old-growth forests are the product of variability, spacing should vary in thinning treatments intended to accelerate the development of old-growth structural characteristics. When it comes to VDT, there is no standardized procedure because thinning designs are site-specific and depend on the characteristics and landscape context of each stand. However, some suggestions have been made: VDT thinning can be done by thinning to different densities in quarter- to one-acre patches, by leaving small quarter- to half-acre unthinned patches, and in other areas creating very small gaps, up to a quarter acre in size at most (Rapp, 2002). The reason for the small gap size is to prevent susceptibility to wind throw while simultaneously encouraging understory development and large, open-grown trees. In addition, the unthinned and lightly thinned areas will be suitable for shade-tolerant seedlings such as hemlock.
5.3 Snag/Wildlife Tree Recruitment

As discussed in Section 3.3, snags, particularly those with heart rot and/or cavities, are an integral structural element for many wildlife species; and yet, this valuable forest attribute is often lost from stands when short-rotation forestry is practiced. Operationally, there are three types of wildlife tree management strategies that can be employed to maintain snags across a landscape: wildlife tree patches (WTPs), individual live tree retention, and artificially created wildlife trees (Stone et al., 2002). Regardless of which wildlife tree retention and or recruitment technique is used, there are several variables that need to be examined on a species-by-species basis in order to ensure that the habitat goals for each species are reached. These variables include size (area), composition (tree species, decay class, basal area), distribution and density (stems/ha), and condition (age class, decay class) (Manning et al., 2006). Furthermore, wildlife tree retention areas should be large enough to buffer key wildlife trees (those containing nest cavities, broken tops, stem scars, hollows or cracks) from adjacent harvesting areas, and provide some undisturbed habitat and interior forest-like conditions (Manning et al., 2006). In other words, a patch should be centered on a well-used wildlife tree or group of wildlife trees (Stone et al., 2002), and, where ecologically appropriate (i.e. the patch contains the desired habitat features), a roughly circular patch shape will optimize forest interior habitat.

Management tools that might be employed to create snags include topping trees with chainsaws or explosives, girdling trees, cavity creation using chainsaws, and inoculating trees with native fungus.

Fungal inoculation is a relatively new technique and is still under development. The ecological and operational feasibility of fungal inoculation as a habitat enhancement tool was still being evaluated; however, as of 2006, interim results indicate that it is an efficient and effective means for creating a tree that contains heart rot (Manning, 2006). Furthermore, it is predicted that when these trees are excavated for use by wildlife, they will eventually become hollow trees, which further increases their habitat value (Manning, 2006).

Two inoculation procedures were investigated to determine which produces the best results. These include: a) injecting the native heart-rot fungus, Phellinus pini, by climbing a tree, drilling a hole, and inserting a wooden dowel that has been cultured with a locally collected fungal strain; and b) shooting the tree trunk with a bullet that contains a smaller wooden dowel cultured with the same fungus (Manning, 2006). Both techniques have resulted in the spread of fungal decay within the tree both above and below the point of inoculation. Four years after the treatment, no wildlife activity had been seen in the treated trees; however, it is expected that usable heartwood decay columns, sufficient for nest cavity construction by primary cavity-excavating bird species, will be created between 5-15 years post-treatment (Manning, 2006).
Importantly, there appears to be little risk of spread to non-targeted trees using this technique because of the reproductive history of the fungi.

Recommended species for inoculation include Douglas-fir and spruce (*Picea* spp) (Manning *et al.*, 2006).

Fungal inoculation is used on live, healthy trees. Results indicate that the fungus does not usually kill the tree; rather, a compartmentalized decay column is produced within the trunk within three to six years. Inoculated trees generally maintain their foliage and growth form, and continue to put on new incremental growth and function as seed sources. This is significant because trees in this condition provide habitat for a longer period of time than dead snags, are likely to provide fewer worker safety or operational concerns, and are less likely to be felled by firewood cutters (Manning, 2006).

**5.4 Coarse Woody Debris Retention and Recruitment**

Eventually, snags will break apart and become coarse woody debris, which is another important ecosystem component. Alternatively, CWD can be recruited by leaving a mixture of coniferous and deciduous thinned trees on the forest floor. The reason for using a mixture is that coniferous trees decay more slowly than deciduous trees (Manning *et al.*, 2006). This means that deciduous CWD can provide important short-term ecological benefits where as coniferous CWD provides ecological benefits for a greater period of time. Ideally, larger CWD pieces will be either recruited or maintained because larger pieces decay more slowly, hold more moisture, present less of a fire hazard, and provide more habitat value to a greater number of wildlife species (Manning *et al.*, 2006). Additionally, when recruiting CWD, it should be left on site in a way that mimics its natural distribution of randomness and connectivity, with some clumping and layering (Manning *et al.*, 2006).

Regardless of which CWD retention and/or recruitment technique is used, there are several variables that need to be examined on a species-by-species basis in order to ensure that the habitat goals for each species are reached for CWD. Variables to consider include the amount (volume), condition (species and decay class), and distribution of CWD. Furthermore, forest health variables such as insects and fuel loading must be considered in the context of this evaluation (Manning *et al.*, 2006).
5.5 Riparian Area Restoration

General strategies for maintaining and/or recruiting riparian habitat and function (Manning et al., 2006) are:

1. Choose silviculture treatments and equipment that minimize ground disturbance within riparian areas in order to minimize introduction of non-native plant species and to maintain natural water movement.

2. Maintain and/or recruit natural levels of coarse woody debris within the riparian area. CWD has additional value in riparian areas as habitat for a number of wildlife and plant species.

3. Within riparian areas, leave all dead wildlife trees that do not pose a risk to worker safety to provide future in stream large woody debris (LWD). If low value wildlife trees and danger trees have to be felled for worker safety reasons, then these stems should be retained on site as CWD.

More detailed information on riparian management area guidelines and recommended management practices for riparian zone silviculture and restoration treatments can be found in Recommended Riparian Zone Silviculture Treatments (Bancroft and Zielke, 2002) and the Riparian Management Area Guidebook (MOF, 1995).

5.6 Additional Forest Restoration Techniques

Some additional common forest restoration techniques include prescribed burning, underplanting, and mycorrhizal fungus recruitment. In some cases, low intensity under-burning might be used to help diversify the stand structure; and under-planting with several tree species, particularly shade tolerant conifers when they are absent, may be employed to increase vertical heterogeneity where required (Rapp, 2002). Although these methods are not within the scope of this document, additional information can be found in The Once and Future Forests: a Guide to Forest Restoration Strategies (Sauer, 1998), Silviculture for Structural Diversity: a New Look at Multiaged Systems (O'Hara, 1998), and Restoring Fire-Dependant Ponderosa Pine Forests in Western Montana (Arno et al., 1995).
Chapter 4  The Potential Role of Second-growth Forest in the Management of Species at Risk

6  Species At Risk

6.1  Definition

Species at risk in Canada include species that are categorized as Extirpated, endangered, threatened, of Special Concern, or data deficient by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). ‘Extirpated’ refers to a species that no longer exists in the wild, or is no longer present in parts of historic range. ‘Endangered’ refers to a species facing imminent extirpation or extinction. ‘Threatened’ refers to a species that is likely to become endangered if nothing is done to reverse the factors leading to its extirpation or extinction, and ‘Special Concern’ refers to a species with characteristics that make it particularly sensitive to human activities or natural events (Parks Canada, 2007). Species at risk are a focal point for individuals and organizations that are attempting to address declines of forest biodiversity.

6.2  Extinction Proneness

The conceptual foundation of species prioritization recognizes that not all species are equally prone to extinction. For example, species with small population sizes are more at risk than those with large populations due to unpredictable changes in birth and death rates, environmental fluctuations, and random genetic processes (Shaffer, 1981). Factors that have been related to the extinction proneness of species in modified landscapes include habitat or niche specialization, home range size, mobility, population density or rarity, edge sensitivity, body size, dietary specialization (Fischer and Lindenmayer, 2007), and reproductive rates (Pimm et al., 1988). For example, specialized species are more likely to lose their habitat as a result of deforestation then generalist species; body size may be used as a proxy for area requirements; and high density populations contain many individuals in a given area and thus are more resilient to stochastic threats (Fischer and Lindenmayer, 2007). Furthermore, even large populations may be prone to extinction if they are sensitive to habitat change and are narrowly distributed, require two or more habitats, or are disrupted by habitat fragmentation (Pimm et al., 1988).
6.3 Identified Wildlife Management Strategy

The Identified Wildlife Management Strategy (IWMS) is an initiative of the Ministry of Environment in partnership with the Ministry of Forests and Range. Under the Forest and Range Practices Act, the Minister responsible for the Wildlife Act - the Minister of Environment - is authorized to establish two categories of wildlife that require special management attention to address the impacts of forest and range activities on Crown land. The Species at Risk category, which is the focus of this document, is a sub-set of the provincial species at risk list. It includes endangered, threatened, or vulnerable species of vertebrates and invertebrates, and endangered or threatened plants and plant communities that are negatively affected by forest or range management on Crown land and are not adequately protected by other mechanisms. The Identified Wildlife Management Strategy aims to minimize the effects of forest and range practices on Identified Wildlife, and to maintain their critical habitats throughout their critical ranges, and, where appropriate, their historic ranges. In some cases, this will require the restoration of previously occupies habitats. Wildlife habitat areas (WHAs) are mapped areas that have been approved by the Ministry of Water, Land, and Air protection as requiring special management; the purpose of WHAs is to conserve habitats considered to be limiting for identified wildlife elements.

6.4 Species and Ecosystem Inventory Standards

Fundamental to species and ecosystems at risk management is the need to identify, map, and monitor where they occur, and to encourage land use decisions that will ensure their continued integrity over the long term. It is necessary to underline the importance of the identification of critical habitat based on scientific considerations. The Provincial Resources Information Standards Committee (RISC) (ILMB, undated) is responsible for establishing standards for natural and cultural resources inventories including collection, storage, analysis, interpretation, and reporting of inventory data, and should be used as guidance for undertaking inventory of species and ecosystems in general, where species specific guidance is not yet available from Recovery Teams.

6.5 Ranking Species At Risk in British Columbia

The rankings highlight species and ecological communities that have particular threats, declining population trends, or restricted distributions that indicate that they require special
attention. These lists serve as a practical method to assist in making conservation and land-use decisions and prioritize research, inventory, management, and protection activities. For example, Operational Planning Regulations in the Forest Practices Code of British Columbia Act use the Red and Blue lists in the development of the list of Identified Wildlife.

The RED and BLUE lists serve two purposes:

1. To provide a list of species for consideration for more formal designation as Endangered or Threatened, either provincially under the British Columbia Wildlife Act, or nationally by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC).

2. To help inform setting conservation priorities for species/ecological communities considered at risk in British Columbia.

6.5.1 Red List

Includes any ecological community, indigenous species, or subspecies that is extirpated, endangered, or threatened in British Columbia. Extirpated elements no longer exist in the wild in British Columbia, but do occur elsewhere. Endangered elements are facing imminent extirpation or extinction. Threatened elements are likely to become endangered if limiting factors are not reversed. Red-Listed species and sub-species may be legally designated as, or maybe considered candidates for, legal designation as Extirpated, Endangered or Threatened under the Wildlife Act. Not all Red-Listed taxa will necessarily become formally designated. Placing taxa on these lists flags them as being at risk and requiring investigation.

6.5.2 Blue List

Includes any ecological community, indigenous species, or subspecies considered to be of Special Concern (formerly vulnerable) in British Columbia. Elements are of Special Concern because of characteristics that make them particularly sensitive to human activities or natural events. Blue-listed elements are at risk, but are not extirpated, endangered or threatened.

6.5.3 The Relationship of Red and Blue lists to CDC Ranks

Species and ecological communities are assigned to the Red and Blue list based on the provincial Conservation Status Rank (SRANK) assigned by the Conservation Data Center, these ranks are presented in Table 3.
Table 3. The Relationship of Red and Blue lists to CDC Ranks

<table>
<thead>
<tr>
<th></th>
<th>Red List</th>
<th>Blue List</th>
</tr>
</thead>
<tbody>
<tr>
<td>Animals*</td>
<td>SX, SH, S1, S1S2, S2, S2?, S1S3</td>
<td>S2S3, S2S4, S3S3?, S3S4, S3S5</td>
</tr>
<tr>
<td>Plants</td>
<td>SX, SH, S1, S1S2, S1S3, S2, S2?</td>
<td>S2S3, S2S4, S3, S3?</td>
</tr>
<tr>
<td>Ecological Communities</td>
<td>SX, SH, S1, S1S2, S2</td>
<td>S2S3, S3</td>
</tr>
</tbody>
</table>

* Note: this information applies to regularly occurring breeding animal species only. Regularly occurring non-breeding animals with Global ranks of G1, G2, G1G2 (for species), T1, T2, or T1T2 (for subspecies) are placed on the Red List. Those with Global ranks of G2G3, G3, G3G4 (for species); T2T3, T3, or T3T4 (for subspecies) are placed on the Blue List.

6.6 Conservation Status Definitions

### Table 4. Basic NatureServe Global Conservation Status Ranks

<table>
<thead>
<tr>
<th>Rank</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>GX</td>
<td>Presumed Extinct (species)— Not located despite intensive searches and virtually no likelihood of rediscovery. Eliminated (ecological communities)—Eliminated throughout its range, with no restoration potential due to extinction of dominant or characteristic species.</td>
</tr>
<tr>
<td>GH</td>
<td>Possibly Extinct (species)— Missing; known from only historical occurrences but still some hope of rediscovery. Presumed Eliminated— (Historic, ecological communities)-Presumed eliminated throughout its range, with no or virtually no likelihood that it will be rediscovered, but with the potential for restoration, for example, American Chestnut (Forest).</td>
</tr>
<tr>
<td>G1</td>
<td>Critically Imperiled—At very high risk of extinction due to extreme rarity (often 5 or fewer populations), very steep declines, or other factors.</td>
</tr>
<tr>
<td>G2</td>
<td>Imperiled—At high risk of extinction due to very restricted range, very few populations (often 20 or fewer), steep declines, or other factors.</td>
</tr>
<tr>
<td>G3</td>
<td>Vulnerable—At moderate risk of extinction due to a restricted range, relatively few populations (often 80 or fewer), recent and widespread declines, or other factors.</td>
</tr>
<tr>
<td>G4</td>
<td>Apparently Secure—Uncommon but not rare; some cause for long-term concern due to declines or other factors.</td>
</tr>
<tr>
<td>G5</td>
<td>Secure—Common; widespread and abundant.</td>
</tr>
</tbody>
</table>
### Table 5. Variant NatureServe Global Conservation Status Ranks

<table>
<thead>
<tr>
<th>Rank</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>G#G#</td>
<td>Range Rank—A numeric range rank (e.g., G2G3) is used to indicate the range of uncertainty in the status of a species or community. Ranges cannot skip more than one rank (e.g., GU should be used rather than G1G4).</td>
</tr>
<tr>
<td>GU</td>
<td>Unrankable—Currently unrankable due to lack of information or due to substantially conflicting information about status or trends. Whenever possible, the most likely rank is assigned and the question mark qualifier is added (e.g., G2?) to express uncertainty, or a range rank (e.g., G2G3) is used to delineate the limits (range) of uncertainty.</td>
</tr>
<tr>
<td>GNR</td>
<td>Unranked—Global rank not yet assessed.</td>
</tr>
<tr>
<td>GNA</td>
<td>Not Applicable—A conservation status rank is not applicable because the species is not a suitable target for conservation activities.</td>
</tr>
</tbody>
</table>

### Table 6. NatureServe Global Conservation Status Rank Qualifiers

<table>
<thead>
<tr>
<th>Rank</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>?</td>
<td>Inexact Numeric Rank—Denotes inexact numeric rank (e.g., G2?)</td>
</tr>
<tr>
<td>Q</td>
<td>Questionable taxonomy—Taxonomic distinctiveness of this entity at the current level is questionable; resolution of this uncertainty may result in change from a species to a subspecies or hybrid, or the inclusion of this taxon in another taxon, with the resulting taxon having a lower-priority conservation priority.</td>
</tr>
<tr>
<td>C</td>
<td>Captive or Cultivated Only—At present extant only in captivity or cultivation, or as a reintroduced population not yet established.</td>
</tr>
</tbody>
</table>
Table 7. NatureServe National Conservation Status Ranks

<table>
<thead>
<tr>
<th>Rank</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>N#N#</td>
<td>A numeric range rank (e.g., N2N3) is used to indicate the range of uncertainty about the status of the species or community. Ranges cannot skip more than one rank (e.g., NU should be used rather than N1N4).</td>
</tr>
<tr>
<td>N?</td>
<td>Nation conservation status not yet assessed</td>
</tr>
<tr>
<td>N1</td>
<td>Critically imperiled in the nation because of extreme rarity (often 5 or fewer occurrences) or because of some factor(s) such as very steep declines making it especially vulnerable to extirpation from the nation</td>
</tr>
<tr>
<td>N2</td>
<td>Imperiled in the nation because of rarity due to very restricted range, very few populations (often 20 or fewer), steep declines, or other factors making it very vulnerable to extirpation from the nation</td>
</tr>
<tr>
<td>N3</td>
<td>Vulnerable in the nation due to a restricted range, relatively few populations (often 80 or fewer), recent and widespread declines, or other factors making it vulnerable to extinction</td>
</tr>
<tr>
<td>N4</td>
<td>Apparently Secure; Uncommon but not rare; some cause for long-term concern due to declines or other factors</td>
</tr>
<tr>
<td>N5</td>
<td>Secure-Common, widespread, and abundant in the nation</td>
</tr>
</tbody>
</table>

7 Prioritizing Identified Wildlife Protection

Very limited resources are available for species at risk recovery planning, underscoring the need to establish priorities. The priority setting process should be clearly established in legislation to determine which species require the most immediate action. Difficult and competing recovery decisions need to be made with limited funding and capacity. The Ministry of Environment’s (MOE) conservation framework is intended to help coordinate and align conservation efforts across government and non-government sectors, and more effectively focus the allocation of resources to yield the best conservation outcomes across the province. The three goals of the conservation framework are: a) to contribute to global efforts for species and ecosystem conservation; b) prevent species and ecosystems from becoming at risk; and c) maintain the diversity of native species and ecosystems (MOE, 2009). Within this context, species are assessed according to their global and provincial status, population trends, threats, stewardship responsibility, and feasibility of recovery. Each species receives a rank of 1 (highest) through 6 (lowest) under each of the three goals and is placed under the goal in which it receives the highest score. The Ecosystem Component of the Conservation Framework completed in 2009 focuses on: a) identifying and prioritizing ecological communities of conservation concern; b) adapting management action to address key threats, including climate change; and c) informing
landscape-level actions to support ecological functions and processes while maintaining options for species migration and adaptation (MOE, 2009). Given that the ecosystem component of the conservation framework was still being developed as of the time of completion of this report, the remainder of this discussion will focus on the species component of the framework.

In the conservation framework, a species that is globally secure, yet drastically declining in B.C. may be given the same priority for conservation as a species that is endemic to B.C. and has small fragmented populations that are threatened. It is evident that species at the edges of their ranges are important with respect to species adaptation and evolutionary processes; however, it is imperative to recognize that range shifts can be a natural occurrence and are predicted to increase in magnitude in response to climate change. If there are large source populations outside of B.C. from which species can recolonize B.C., should these species be given the same level of prioritization as those species that only exist in B.C.? In other words, should species at the edge of their ranges be given the same level of priority as those with very small populations? Should individual species at risk be given priority over ecosystems at risk? And finally, should preventing species from becoming at risk have the same priority as saving species that are already at risk? There has been some debate over these questions, and in response it is appropriate to explore alternative methods for prioritizing species and ecosystem conservation. Table 8 presents several different approaches for prioritizing and managing ecosystems and species at risk.
Table 8. Current approaches for prioritizing and managing ecosystems and species at various spatial scales (adapted from A.J. Hansen et al., 1999).

<table>
<thead>
<tr>
<th>Name</th>
<th>Scale</th>
<th>Concept</th>
<th>Method</th>
<th>Example</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gap Analysis/Critical</td>
<td>Province to continental</td>
<td>Identify ecosystems that contain species at risk or are poorly protected so that local management can be applied</td>
<td>Rank ecosystems based on native species, threats to ecosystem, and other factors</td>
<td>Nature Conservancy (1982), Noss and Cooperrider (1994)</td>
</tr>
<tr>
<td>Ecosystems</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Species Prioritization</td>
<td>Province to continental</td>
<td>Identify species most at risk so that management can be directed towards them</td>
<td>Use existing life history and other data to rank species viability</td>
<td>Millsap et al. (1990), Hunter et al. (1993)</td>
</tr>
<tr>
<td>Ecological Process</td>
<td>Watershed to Province</td>
<td>Maintain key ecological processes (such as disturbance and succession) and landscape structures to maintain species adapted to these conditions</td>
<td>Analyze interactions among ecological processes and structures and manage to maintain them</td>
<td>Boyce (1991), Cissel et al. (1994)</td>
</tr>
<tr>
<td>Management</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dynamic Habitat Modeling</td>
<td>Watershed to province</td>
<td>Use configuration of suitable habitat for each species as a measure of extinction risk; assumes species abundance is related to habitat suitability</td>
<td>Quantify/project change in suitable habitats for each species under varying management</td>
<td>Hansen et al. (1993), White et al. 1997)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Name</td>
<td>Scale</td>
<td>Concept</td>
<td>Method</td>
<td>Example</td>
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<td>-------------------------------</td>
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<td>----------------------------------------------</td>
<td>-------------------------------------------------------------------------</td>
<td>-------------------------------</td>
</tr>
<tr>
<td>Population Viability Analysis</td>
<td>Watershed to range</td>
<td>Analyze population demography to assess risk of extinction</td>
<td>Use complex demographic models to assess species viability under varying management strategies</td>
<td>Shaffer (1981), Murphy and Noon (1992)</td>
</tr>
<tr>
<td>Dynamic Habitat and Population Analysis</td>
<td>Watershed to province</td>
<td>Identify the species and places most at risk and focus research and management on these</td>
<td>Use a hierarchical set of filters to identify and manage the species and places most at risk</td>
<td>Hansen et al. (1999)</td>
</tr>
</tbody>
</table>

With the exception of Population Viability Analysis, all of these strategies are coarse filter approaches. They assume that the population status of any given species is correlated with habitat availability (Hansen et al., 1999). With these approaches the area and spatial patterning of habitat types is quantified and rule-based or statistical functions are applied that predict the presence or abundance of a given species as a function of the habitat (Hansen et al., 1999). The limitations of coarse filter approaches include the fact that the accuracy of habitat modeling in predicting species abundances, etc., is seldom qualified, and is likely to be variable, or low, (or both) if local field data is not used (Hansen et al., 1999). Also, the abundance of a species in a habitat may not be indicative of rates of survival and reproduction in that habitat; source/sink dynamics are probably common in landscapes where resources or habitat conditions are relatively heterogeneous in space (Pulliam, 1996). Thus, using the coarse filter approach, species can undergo population declines and even extinctions without biodiversity managers knowing (Hansen et al., 1999). A fine filter approach, on the other hand, requires demographic data on individual species that are often difficult and expensive to obtain. This is because population vital rates and densities vary spatially, requiring spatially explicit field study, knowledge of dispersal behaviour, and population modeling in order to make useful estimates (Dunning et al., 1995). However, an increasing number of studies are finding that spatially-
mediated population dynamics are prevalent in nature and strongly influence population viability (Pulliam, 1996). Thus, there is an apparent need to bridge coarse filter studies of habitats with species-specific studies of population demography and viability at local spatial scales (Hansen et al., 1999). The following sections briefly describe selected techniques that can be employed to prioritize species at risk.

7.1 Selected Species at Risk Prioritization Strategies

7.1.1 GAP Analysis

The GAP method uses geographic information system (GIS) maps of vegetation cover, vertebrate species ranges, existing reserves, and land ownership to determine which species are adequately protected in reserves and which species require further protection (Kiester et al., 1996). This approach requires ecosystem-level data such as vegetation classes, and species information such as presence/absence or density data (Kiester et al., 1996; Jennings, 2000). Although this method is relevant for land use decisions made at a landscape scale, it is only the first stage in a biodiversity protection effort. This is because it only provides direction to those areas that should be considered a high priority, rather than prescriptions for reserve boundaries and management techniques required to maintain viable populations and ecosystem processes. In other words, Gap Analysis provides a way of calculating the positive cumulative impact of choosing various areas for protection. The intent of gap analysis is to be proactive, to preserve biodiversity before it becomes rare, and to consider all native species throughout an entire region (Kiester et al., 1996).

7.1.2 Species Prioritization

To prioritize species, traits thought to relate to extinction proneness are compiled for each species within the area of interest; managers select variables and spatial scales consistent with their location and objectives. A typical constraint on the selection of variables is the quality of available data. Methods for scoring sensitivity to extinction vary; however, combining scores for several variables into one index of sensitivity to extinction is often done by summing the scores of individual variables or weighting some variables more heavily than others. Multivariate statistical approaches have been described as more objective and informative than just combining scores across traits (Reed, 1995); however, knowledge of the species and type and quality of the data available often determine which approach is used.
7.1.3 Dynamic Habitat Analysis

It is clear that habitat attributes set an upper limit on population abundance and fitness, and species habitat associations often vary among ecosystems and over time. For this reason, local studies of habitat are needed to accurately predict species abundance. Habitat is generally quantified in terms of vegetation structure and composition at either the stand or landscape scale. Habitat analysis entails: a) determining the habitat requirements of the species; b) mapping the distribution of these habitat factors over the planning area; c) extrapolating species abundance over the planning area based on habitat associations; and d) analyzing the predicted patterns of species distributions (Hansen et al., 1999). Abiotic factors such as climate, topography, soil, and moisture regimes can sometimes be used to improve the predictive ability of habitat models because these factors may directly influence the demography of a population or indirectly influence organisms by altering disturbance regimes or vegetation structure (Hansen and Rotella, 1999). Estimating species abundance by habitat type from field studies is labour intensive and can be expensive. That said, cost-effective methods include quadrat sampling for plants, sticky traps and sweep netting for invertebrates, pitfall traps for herptiles, live traps for small mammals, and point counts for birds (Sutherland, 1996). Decisions about sample sizes, season of sampling, and temporal duration of sampling must be made based on knowledge of the ecology of the organisms, ecosystem variability, and resources available for sampling (Hansen et al., 1999). Stratifying sample locations by the environmental variables that are most likely to be related to species abundances, and randomizing relative to other predictor variables allows for exploratory analyses of the influence of these variables on the organisms of interest (Hansen et al., 1999). Statistical methods for determining the strengths of association among predictor variables and species abundance, and for extrapolating species abundance over the planning area can be found in the following publications: Capen (1981), Verner et al. (1986), Morrison et al. (1992), and Mauer (1994).

7.1.4 Population Viability Analysis

Given that the abundance of a local population may not be a good measure of the species extinction proneness, a demographic analysis should be conducted for species determined to be most at risk in the previous steps. Demographic analysis takes into consideration a population’s birth, death, and dispersal rates. The assessment of a population’s risk of extinction, or Population Viability Analysis (PVA), estimates the probability of a population attaining an undesirable status within a specified time period given the model’s assumptions. Analysis can cover both long and short time periods, and can be deterministic or stochastic. Approaches available for performing PVAs can be found in the following publications: Starfield and Blelock (1986), Renshaw (1991), and Burgman et al. (1993). Model selection is somewhat
determined by the availability of data to parameterize the model, and range from simple exponential population growth models to complex spatially explicit metapopulation models (Burgman et al., 1993). When data is limited, exponential population growth models may be adequate for modeling populations when densities are well below the carrying capacity of the management area, and in the short term when growth is not extreme and critical rates are not changing dramatically due to density changes (Burgman et al., 1993). The drawback of many simple models is that they do not indicate which aspects of demography, such as survival vs. reproduction, most strongly influence population changes (Hansen et al., 1999). If habitat-specific data is available, relatively more complex source-sink models (e.g. Pulliam, 1988) can be used to identify those habitats most important to population persistence (Pulliam and Danielson, 1991). If adequate data are available, even more complex models, such as spatially explicit population models, can be used to study the effects of landscape configuration, dispersal ability, and metapopulation structure (Hanski and Gilpin, 1991; Dunning et al., 1995). A wide array of software (e.g., Legendre and Clobert, 1995; Mills et al., 1996) is available for population analysis.

7.1.5 Dynamic Habitat and Population (DHP) Analysis

Dynamic Habitat and Population (DHP) Analysis integrates habitat-based and population-based methods at local spatial scales to focus research and management on the species in a community that are most at risk of extinction, and on the places in the landscape that are most important for these species. The steps involved in DHP analysis include:

1) Determining which species in the planning area most merit field study based on existing data;

2) Using local field data to select species that most merit demographic study;

3) Using demographic data to model population viability of the species deemed the most at risk; and

4) Designing and evaluating alternative management strategies for key species and landscape settings (Hansen et al., 1999).

DHP allows for a combination of conservation approaches and examination of alternative management scenarios because it provides data and knowledge on key relationships between organisms and their local environments. This includes biophysical features, distribution of species richness, distribution of abundance for species at risk, and distribution of population sources and sinks for species highly at risk (Hansen et al., 1999). This data can also be used to
parameterize simulation models that evaluate the probable consequences that result from each alternative for habitat patterns, distributions of species abundance, and species population dynamics (e.g. Hansen et al., 1993; White et al., 1997). This approach is particularly useful for hierarchical management scenarios that aim to actively restore populations and habitats of species most at risk while maintaining key ecosystems and processes to prevent other species from becoming at risk (Hansen et al., 1999).

7.2 New Combination Approaches

7.2.1 Incorporating Evolutionary Measures into Species Prioritization

Although it is not intended as a prioritization metric, threat status measures are often seen as being synonymous with conservation priority (Possingham et al., 2002). This is an issue because this type of prioritization assumes that all species are of equal worth except for their threat status, yet, it is known that species differ substantially in the amount of unique genetic information they embody (Crozier, 1997). However, evolutionary importance is difficult to quantify because of problems such as difficulties in determining what constitutes “evolutionarily significant units” (Erwin, 1991) and the relationship between phenotypic and genetic variation (Diniz, 2004). Nevertheless, by calculating the expected loss of genetic information for a group of species, which is the product of the probabilities of extinction and a value of genetic diversity, the evolutionary importance can be combined with threat status to prioritize species (Witting et al., 1994; Redding and Mooers 2006).

7.2.2 Multispecies Conservation Values

This method involves combining maps of habitat suitability for each species with the extinction risk faced by each species in a single map of multispecies conservation values (MCVs). Risk of extinction is used as a weighting factor so that the priority given to a species habitat requirements is proportional to how imperiled the species is. Two measures of risk are used, endangerment indices and extinction risk probabilities from metapopulation modeling; these produce an index-based multispecies conservation value and a risk-based multispecies conservation value, respectively (Root et al., 2003). This method provides a quantitative and spatially explicit conservation value useful for such applications as a multispecies recovery plan, a regional habitat conservation plan, or an evaluation of local management alternatives. The strengths of this approach are that it:
a) Is a quantitative estimate of conservation value, and provides an objective measure of the ecological worth of a given site based on models that can be validated;

b) Is multivariate and incorporates in a single value as many factors as possible about the species of interest (e.g. presence/absence, density, habitat requirements, and spatial data);

c) Is flexible; and

d) Evaluates a site in terms of all selected component species of its ecosystem rather than in terms of a single threatened or endangered species (Root et al., 2003).

The limitations of this approach include:

a) The threat indices used are only an approximate estimate of the extinction risk of each species, and they may not agree completely with one another (e.g. global vs. provincial ranking); and

b) The method does not take into account interactions among the species considered in the analysis (e.g. competitive or trophic relationships) (Root et al., 2003).

7.2.3 The Nature Conservancy’s Approach

The Nature Conservancy aims for the conservation of multiple examples of each ecosystem target, stratified across its geographic and ecological ranges, as is necessary to capture the ecological variability of the target and to provide sufficient redundancy to ensure persistence in the face of environmental stochasticity (Valutis and Mullin, 2000). Conservation targets also include species and communities at risk, and critical and declining habitats that need be protected. There are five principles that drive the prioritization process for selecting conservation sites: conservation status, complementarity, conservation value, threats and feasibility, and leverage (Valutis and Mullin, 2000). The Conservancy also considers future impacts such as climate change in order to ensure that various habitats are connected, for instance through corridors, in order to allow animals to adapt to varying conditions.

Conservation Status

Due to limited resources and the huge challenge of conserving biodiversity, the Conservancy has determined that it will select sites where it can make an immediate difference. In this context, areas within which some or all of the targets have reduced viability or are highly threatened are prime candidates for conservation.
Complementarity

Complementary approaches assess the conservation targets of any existing site and then proceed in a stepwise fashion to next select the site that is most complementary to the existing set of sites. Complementarity is closely related to the concept of efficiency; the most efficient reserve design is one that captures the conservation targets and achieves the conservation goals in the least area of land or at the lowest costs. Approaches that are complementary generally result in more efficient reserve designs.

Conservation Value

The Conservancy uses two factors to determine conservation value: the number or diversity of conservation targets within the site, and the viability of the conservation target occurrences within a conservation area. Conservation areas with multiple targets, and targets at different spatial scales are given a higher ranking than other areas. The viability of the targets is based upon three criteria: size, condition, and landscape context. For each site, the presence or absence of large-scale natural disturbances and ecological processes is used to determine if the site is large enough. Typically, larger areas allow for diverse successional stages of various communities, as well as natural disturbances (e.g. disease, fire, insect outbreaks, hurricanes and ice storms). Two factors are considered to determine the condition of a site: anthropogenic impacts and biological legacies. Anthropogenic impacts include fragmentation, introduction of exotic species, alteration of natural disturbance regimes, and pollution. Conservation areas that contain relatively continuous cover of natural vegetation (i.e. less fragmentation) are more likely to have intact ecological processes, fewer anthropogenic threats, and be free of invasive exotic species. Biological legacies are the critical features of a site that may sometimes take hundreds to thousands of years to develop. As a general rule, the presence of a well-developed structure and species composition that include characteristic but also uncommon species implies good habitat quality and some historical continuity. Although landscape context is important for all areas, those patch and matrix types that depend on easily disrupted ecological processes occurring at a scale larger than the individual community are most at risk by what happens in the surrounding landscape (e.g. altered fire regime, ground water pumping). In general, conservation areas that are connected to, or in proximity to, other natural habitats are usually preferable to isolated examples of communities.
**Threats and Feasibility**

Threats to a conservation target are defined as a combination of the stress (e.g. habitat fragmentation) and the source of the stress (e.g. development of roads). A stress is the impairment or degradation that reduces the viability (i.e. the size, condition, or landscape context) of a conservation target. A source of stress is an extraneous factor, either human or biological, that infringes upon a conservation target in a way that results in stress. Threats are ranked based upon how likely the threat is to exist within a given amount of time. Feasibility is ranked based upon capacity, the probability of success and the costs of implementing strategies at a site.

**Leverage**

Frequently, the most effective strategies are catalytic in nature; a little bit of effort or a small investment triggers positive work or resources from others and new opportunities. Working in some areas may lead to developing new resources, partners, or tools for other areas in the ecoregional portfolio. Leverage is an exceptional opportunity to use the actions at one site to influence many other areas. Conservation areas with high, clearly specified, demonstrable leverage building partnerships, tools, or funding to conserve other areas are given a higher ranking than other areas.

**7.2.4 The Nature Conservancy of Canada’s Approach**

Conservation at the Nature Conservancy of Canada is a four-step process (NCC, 2009). This includes the development of ecoregional assessments, site conservation planning, conservation action and measuring success. For the purposes of this discussion we will focus on the ecoregional assessment as a prioritization tool. This component of the process has four parts: setting conservation goals, mapping existing conservation lands, designing a network of sites, and setting priorities (NCC, 2009). The first step in the process is used to define how much of the habitat of a target species or natural ecosystem must be conserved to ensure its long-term survival. In the second step, as much as possible, all data on the existing places where the native biodiversity is already conserved are mapped and considered in terms of the conservation goals they already meet. In the third step, ecoregional planning teams select a portfolio of sites where conservation should be considered in order to meet the targeted needs. In addition to the core information about the locations of critical habitats, practical information regarding linkages among target sites, connectivity with existing protected areas
and land ownership patterns are considered. In the fourth step, high-priority target sites are selected based on their inherent biological values, threats to their integrity, the feasibility of conservation action, and potential leverage for conserving other key sites on the landscape (NNC, 2009).

### 7.2.5 Recommendations from the Species at Risk Working Group (2000)

According to the Species at Risk Working Group (2000), factors that should be considered when setting priorities include:

a) Immediacy of threat;

b) Importance of the species to the ecosystem;

c) Rate of population decline;

d) Genetic and taxonomic uniqueness;

e) Continental status of the species and importance of a Canadian effort for its survival;

f) Reasons for population decline; and

g) Degree of knowledge and control of key factors affecting the species.

Once these factors have been assessed, consideration should be given to the direct and indirect program costs of each recovery plan in order to optimize the use of public funds in achieving a prompt recovery of the highest possible number of species at risk. Furthermore, it is considered essential that the ministries responsible for identified wildlife management and recovery planning include the necessary conservation expertise to inform effective species recovery decisions. Given that all of these ministries are currently significantly understaffed and underfunded, appropriate staff should be employed and resources allocated without delay.

### 8 Landscape Level Planning

#### 8.1 Geographical Scales of Biodiversity Management

In the same way that nature operates at different levels of biological organization from genes to landscapes, natural systems also operate at different spatial scales ranging from a few square meters to very large areas (Environment Canada, 2005). For example, some vegetation communities are restricted to very specific physical conditions such as seepage areas and thus
naturally occur as small patch systems; whereas other vegetation communities cover vast areas based on wide-spread soil types and other physical conditions, and form a dominant or matrix habitat type across a landscape. Table 9 highlights the different geographical scales of vegetation and wildlife species.

Understanding these relationships is important for identifying conservation needs, particularly related elements of scale, and key conservation strategies (Environment Canada, 2005). For species or communities that occur as small patch systems, small isolated nature reserves may provide effective protection, Species and communities that require a larger geographical area may require a series of reserves protecting key habitat areas in conjunction with effective linkages or corridors.
Table 9. Geographic Scales of Vegetation Communities and Species (Adapted from Environment Canada, 2005)

<table>
<thead>
<tr>
<th>Geographic Scale</th>
<th>Communities</th>
<th>Species</th>
</tr>
</thead>
</table>
| Local; <2 000 acres or 800 ha | Small patch systems occurring under very specific physical conditions  
E.G. Douglas-fir/ Alaska Oniongrass plant community  
(*Pseudotsuga menziesii/ Melica subulata*) | Limited dispersal ability and generally restricted to a specific community type  
E.G. Quatsino Cave Amphipod  
(*Stygarobromus quatsinensis*),  
Scouler’s Corydalis  
(*Corydalis scouleria*),  
Vancouver Island Marmot  
(*Marmota vancouverensis*) |
| Intermediate; 10 000-50 000 acres or 400-20 200 ha | large patch systems defined by distinct physical factors and environmental regimes  
E.G. Garry Oak ecosystems  
(*Quercus garryana*) | Species that depend on large patch systems or several different types of small patch systems  
E.G. Keen’s Long-eared Myotis  
(*Myotis keenii*), Red-Legged Frog  
(*Rana aurora aurora*) |
| Coarse; 50 000-1 million acres or 20 200-405 000 ha | Matrix communities that are, or historically were, the dominant habitat between patches. Matrix systems are defined by a broader range of physical conditions such as moisture and topography.  
E.G. Douglas-fir/ Dull Oregon Grape plant community  
(*Pseudotsuga menziesii/ Mahonia nervosa*) | Species that require large areas to access the habitat required  
E.G. Queen Charlotte Goshawk  
(*Accipiter gentilis laingi*), Marbled Murrelet  
(*Brachyramphus marmoratus*) |
| Regional; >1 million acres or >405 000 ha | Applies to species only | Populations where individuals have either very large home ranges or species that migrate over large areas  
E.G. Short-eared Owl  
(*Asio flammeus flammeus*) |
8.2 Identified Wildlife Conservation Rankings, Geographical Scales, and Cross Referenced Species, for Species Inhabiting Vancouver Island, the Gulf Islands, and the Queen Charlottes Islands

As stated at the beginning of Section 6.5: one method for prioritizing conservation for species or ecological communities considered at risk in British Columbia is to examine and compare their conservation status provincially, nationally, and globally. The conservation status designations listed in Table 10 are the product of NatureServe assessments. For ecological communities these rankings provide an estimate of the risk of elimination, while for species these ranks provide an estimate of extinction risk (NatureServe, 2009). Conservation status ranks are based on a scale from one to five, ranging from critically imperiled (S1) to demonstrably secure (S5). Ten factors are used to assess conservation status, and these factors are grouped into three major categories: rarity, trends, and threats. Factors in the rarity category include population size, range extent, area of occupancy, number of occurrences (distinct populations), percent area with good viability or ecological integrity, and environmental specificity. Trend factors include long- and short-term trends in population size or area. Threat factors include overall threat impact, which is determined by considering the scope and magnitude of major threats and intrinsic vulnerability (NatureServe, 2009). Based on weightings assigned to each factor, and some condition rules, the ‘rank calculator’ developed by NatureServe assigns a conservation status rank. In this scenario, the species that are most imperiled have the highest priority for protection and recovery.
<table>
<thead>
<tr>
<th>Identified Wildlife Species and Ecosystems</th>
<th>Provincial Status</th>
<th>National Status</th>
<th>Global Status</th>
<th>Geographic Scale</th>
<th>Cross Referenced Identified Wildlife Species</th>
</tr>
</thead>
<tbody>
<tr>
<td>Douglas-fir/ Alaska Oniongrass&lt;br&gt;(Pseudotsuga menziesii/ Melica subulata)</td>
<td>Red List (S1): Critically imperiled/high risk of extinction</td>
<td>N/A</td>
<td>Unknown</td>
<td>Local</td>
<td>Douglas-fir/dull Oregon-grape, Keen’s Long-Eared Myotis, “Queen Charlotte” Goshawk</td>
</tr>
<tr>
<td>Sandhill Crane (A)-Georgia Depression population&lt;br&gt;(Grus canadensis)</td>
<td>Red List (S1): Critically imperiled/high risk of extinction</td>
<td>N?: Unranked</td>
<td>G5T1Q: Critically imperiled subspecies of an otherwise common species, questionable taxonomy</td>
<td>Local to intermediate</td>
<td>Nelson’s Sharp-tailed Sparrow, Pacific Water Shrew</td>
</tr>
<tr>
<td>Vananda Creek Limnetic Stickleback, Vananda Creek Benthic Stickleback&lt;br&gt;(Gasterosteus species 16 and 17 respectively)</td>
<td>Red List (S1): Critically imperiled/Very high risk of extinction</td>
<td>N1: Critically imperiled</td>
<td>G1: Critically imperiled</td>
<td>Local</td>
<td>N/A</td>
</tr>
<tr>
<td>Keen’s Long-eared Myotis&lt;br&gt;(Myotis keenii)</td>
<td>Red List (S1S3): Uncertainty, ranges from critically imperiled to rare or</td>
<td>N1N3: Uncertainty, ranges from critically imperiled to rare or</td>
<td>G2G3: Uncertainty, ranges from imperiled to rare or</td>
<td>Intermediate</td>
<td>Marbled Murrelet, Quatsino Cave Amphipod,</td>
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<tr>
<td>Identified Wildlife Species and Ecosystems</td>
<td>Provincial Status</td>
<td>National Status</td>
<td>Global Status</td>
<td>Geographic Scale</td>
<td>Cross Referenced Identified Wildlife Species</td>
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<tr>
<td>“Vancouver Island” Common Water Shrew</td>
<td>Imperiled to rare or uncommon</td>
<td>Rare or uncommon</td>
<td>Uncommon</td>
<td></td>
<td>“Queen Charlotte” Goshawk, “Vancouver Island” Common Water Shrew</td>
</tr>
<tr>
<td><em>(Sorex palustris brooksi)</em></td>
<td>Red List (S2): Imperiled/very vulnerable to extirpation</td>
<td>N2: Imperiled</td>
<td>G5T2: Imperiled subspecies of an otherwise common species</td>
<td>Local</td>
<td>Douglas-fir/Alaska onion grass, Keen’s Long-eared Myotis, Marbled Murrelet, “Queen Charlotte” Goshawk, Scouler’s corydalis</td>
</tr>
<tr>
<td>Douglas-fir/ Dull Oregon Grape</td>
<td>Red List (S2): Imperiled/very vulnerable to extirpation</td>
<td>N/A</td>
<td>Unknown, Assumed G2G3-G3: Imperiled to vulnerable</td>
<td>Coarse</td>
<td>Keen’s Long-eared Myotis, Red-Legged Frog</td>
</tr>
<tr>
<td><em>(Pseudotsuga menziesii/Mahonia nervosa)</em></td>
<td>Red List (S2B): Breeding populations are imperiled/very vulnerable to extirpation</td>
<td>N2: Imperiled</td>
<td>G5T2: Imperiled subspecies of an otherwise common species</td>
<td>Coarse</td>
<td>Great Blue Heron, Marbled Murrelet, “Queen Charlotte” Hairy Woodpecker, “Vancouver Island” Northern Saw-whet</td>
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<tr>
<td>Identified Wildlife Species and Ecosystems</td>
<td>Provincial Status</td>
<td>National Status</td>
<td>Global Status</td>
<td>Geographic Scale</td>
<td>Cross Referenced Identified Wildlife Species</td>
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<tr>
<td>Marbled Murrelet <em>(Brachyramphus marmoratus)</em></td>
<td>Red List (S2B, S4N): Breeding populations are imperiled/very vulnerable to extirpation, non-breeding populations are apparently secure</td>
<td>N2: Imperiled</td>
<td>G3G4: Uncertainty regarding status, ranges from vulnerable to apparently secure</td>
<td>Coarse</td>
<td>Great Blue Heron, Grizzly Bear, Keen’s Long-eared Myotis, “Queen Charlotte” Goshawk, “Queen Charlotte” Northern Saw-whet Owl</td>
</tr>
<tr>
<td>Quatsino Cave Amphipod <em>(Stygobromus quatsinensis)</em></td>
<td>Blue List (S2S3): Uncertainty, ranges from imperiled to vulnerable</td>
<td>N3: Vulnerable</td>
<td>G3: Vulnerable</td>
<td>Local</td>
<td>Keen’s Long-eared Myotis</td>
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<tr>
<td>Ancient Murrelet <em>(Synthliboramphus antiquus)</em></td>
<td>Blue List (S2S3B, S4N): Uncertainty regarding breeding populations status, ranges from imperiled to vulnerable, non-breeding populations are apparently secure</td>
<td>N3: Vulnerable</td>
<td>G4: Apparently Secure</td>
<td>Coarse</td>
<td>Cassin’s Auklet, Keen’s Long-eared Myotis, “Queen Charlotte” Northern Saw-whet Owl, “Queen Charlotte” Hairy Woodpecker</td>
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<tr>
<td>Identified Wildlife Species and Ecosystems</td>
<td>Provincial Status</td>
<td>National Status</td>
<td>Global Status</td>
<td>Geographic Scale</td>
<td>Cross Referenced Identified Wildlife Species</td>
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<tr>
<td>Identified Wildlife Species and Ecosystems</td>
<td>secure</td>
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<tr>
<td><strong>Short-eared Owl</strong> <em>(Asio flammeus flammeus)</em></td>
<td>Blue List (S3B, S2N): Breeding populations are vulnerable, non-breeding populations are imperiled</td>
<td>N4N, N5B: Breeding populations are secure, non-breeding populations are apparently secure</td>
<td>G5: Secure</td>
<td>Regional</td>
<td>Sandhill Crane</td>
</tr>
<tr>
<td><strong>Scouler’s Corydalis</strong> <em>(Corydalis scouleria)</em></td>
<td>Blue List (S3): Vulnerable/Rare or uncommon</td>
<td>N3: Vulnerable</td>
<td>G4: Apparently secure</td>
<td>Local</td>
<td>Marbled Murrelet, “Queen Charlotte” Goshawk, “Vancouver Island” Common Water Shrew</td>
</tr>
<tr>
<td><strong>“Vancouver Island” Northern Pygmy-Owl</strong> <em>(Glaucidium gnoma swarthi)</em></td>
<td>Blue List (S3): Vulnerable/Rare or uncommon</td>
<td>N3: Vulnerable</td>
<td>G5T3Q: Vulnerable subspecies of an otherwise common species, questionable taxonomy</td>
<td>Local to intermediate</td>
<td>Marbled Murrelet</td>
</tr>
<tr>
<td><strong>“Vancouver Island” White-Tailed Ptarmigan</strong> <em>(Lagopus leucurus saxatilis)</em></td>
<td>Blue List (S3): Vulnerable</td>
<td>N3: Vulnerable</td>
<td>G5T3: Vulnerable subspecies of an otherwise common species</td>
<td>Intermediate</td>
<td>Vancouver Island Marmot</td>
</tr>
<tr>
<td><strong>“Queen Charlotte”</strong></td>
<td>Blue List</td>
<td>N3:</td>
<td>G5T3:</td>
<td>Intermediate</td>
<td>Ancient</td>
</tr>
<tr>
<td>Identified Wildlife Species and Ecosystems</td>
<td>Provincial Status</td>
<td>National Status</td>
<td>Global Status</td>
<td>Geographic Scale</td>
<td>Cross Referenced Identified Wildlife Species</td>
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<tr>
<td>Northern Saw-whet Owl (Aegolius acadicus brooksi)</td>
<td>(S3): Vulnerable/ Rare or uncommon</td>
<td>Vulnerable</td>
<td>Vulnerable subspecies of an otherwise common species</td>
<td></td>
<td>Murrelet, “Queen Charlotte” Goshawk, “Queen Charlotte” Hairy Woodpecker</td>
</tr>
<tr>
<td>“Queen Charlotte” Hairy Woodpecker (Picoides villosus picoides)</td>
<td>Blue List (S3): Vulnerable/ Rare or uncommon</td>
<td>N3: Vulnerable</td>
<td>G5T3: Vulnerable subspecies of an otherwise common species</td>
<td>Intermediate</td>
<td>“Queen Charlotte” Northern Saw-whet Owl</td>
</tr>
<tr>
<td>Great Blue Heron (Ardea herodias fannini)</td>
<td>Blue List (S3B, S5N): Breeding populations are vulnerable, non-breeding populations are secure</td>
<td>N?: Unranked</td>
<td>G5T4: Apparently secure subspecies of an otherwise common species</td>
<td>Intermediate</td>
<td>Marbled Murrelet, “Queen Charlotte” Goshawk, Spotted Owl, “Vancouver Island” Northern Pygmy-Owl</td>
</tr>
<tr>
<td>Red-Legged Frog (Rana aurora aurora)</td>
<td>Blue List (S3S4): Uncertainty, ranges from rare or uncommon (vulnerable) to apparently secure</td>
<td>N?: Unranked</td>
<td>G4: Apparently secure</td>
<td>Local</td>
<td>Keen’s Long-eared Myotis, Pacific Water Shrew</td>
</tr>
<tr>
<td>Sandhill Crane (B)- All Other</td>
<td>Blue List (S3S4B):</td>
<td>N5B: Breeding</td>
<td>G5: Secure</td>
<td>Regional</td>
<td>Nelson’s Sharp-tailed</td>
</tr>
</tbody>
</table>
Cross-referenced identified wildlife species and geographic scale were included in Table 10 to build on the concept of complementarity presented in Section 7.2.3. Ideally, priority for wildlife habitat areas will be given to high quality occurrences of limiting habitat for the maximum number of identified wildlife elements. That is, high value areas that meet the needs of multiple identified wildlife elements should be given priority over areas that meet the needs of only one identified wildlife element. Those identified wildlife elements that operate at a local scale may require ‘single species’ WHA’s due to dispersal abilities, etc.; in these instances the concepts presented in the conservation area functionality section (Section 4.6) should be applied and complementarity should still be considered. Landscape level planning will be discussed in more detail in the following section.

### 8.3 Landscape Level Planning for Specific Identified Wildlife Elements

Landscape level management objectives such as patch size distribution, landscape connectivity, and seral stage targets are described in higher-level plans such as Sustainable Forest Management Plans, Land Use Plans, and Land and Resource Management Plans. Higher-level plan objectives may assist in the implementation of the desired habitat objectives for a given species and vice versa. For example, the establishment of a Queen Charlotte goshawk WHA can also contribute to old seral targets for a particular landscape unit. However, these higher-level plans are inadequate in the context of protecting several identified wildlife elements; in particular, species that have large home ranges, occur at low densities, have widely and sparsely distributed limiting habitats, or are sensitive to landscape level disturbances. These species include, but are not limited to, the badger, bull trout, caribou, fisher, grizzly bear,
marbled murrelet, Queen Charlotte goshawk, spotted owl, and wolverine. This situation requires the examination of special management areas, such as Old Growth Management Areas (OGMAs), Ungulate Winter Ranges (UWRs), Wildlife Habitat Areas (WHAs), and Riparian Reserves (RRs), plus a determination of how these may be linked to achieve key habitat objectives for a given identified wildlife element.

One way to accomplish this is to use forest inventory data, aerial photographs, and GIS mapping tools to determine current and projected landscape level targets (spatially and temporally) for:

1. Special management area distribution (includes OGMAs, UWRs, WHAs, RRs)
2. Seral stage distribution (i.e., relative proportions of young, mature and old forest);
3. Patch size distribution (includes opening sizes and forested patch sizes);
4. Landscape level connectivity (includes inter-patch connectivity and cross-elevational and cross-valley connectivity);
5. Access (i.e., road density, amount of active road systems) which influences habitat fragmentation and human disturbance; and
6. Visual breaks and barriers, to provide security cover for wildlife, especially in areas with abundant human access.

An analysis of these six landscape variables is important because it will help determine methods for maintaining natural habitats on a larger scale, which significantly influences the achievement of habitat objectives for the previously mentioned Identified Wildlife elements. Furthermore, there may be benefits from planning for the requirements of elements at the strategic and landscape level, in that it may be possible to effectively plan for a greater number of species and accommodate connectivity requirements while simultaneously reducing the incremental impacts to resource industries.

Importantly, although the Biogeoclimatic Ecosystem Classification (BEC) program initiated by the B.C. Ministry of Forests and Range has made substantial headway with mapping B.C.’s landscapes, there are significant knowledge gaps in the fine filter site-specific habitat attributes required by many species at risk. For example, there are several identified wildlife bird species in the Coast Forest Region that are limited by the density, distribution, and decay status of wildlife trees, yet this habitat element is not adequately mapped at the site series level to account for these limiting elements during the wildlife habitat area assessment process. In other words, a more comprehensive land inventory will be necessary in many potential conservation areas.
# Plant Community Specific Information

Within the Vancouver Island and Gulf Islands component of the coast forest region there are two ecosystem types that have been identified as needing special consideration under the IWMS. These include the Douglas-fir/Alaska Oniongrass and Douglas-fir/Dull Oregon-grape plant communities; both of which fall within the CDFmm BEC zone. Significantly, one 1995 study estimated that only one half of one percent (about 1100 hectares) of the low coastal plain is covered by relatively undisturbed old coastal Douglas-fir forests (Flynn, 1999). This is far below what scientists consider to be the minimum area required for the continued survival of these forest types. Furthermore, even if efforts to protect all remaining old-growth stands are successful, additional areas of older second-growth forest will have to be protected and allowed to recover to an old-growth state in order to ensure adequate representation of these forest types in the future, and to provide a continuous network of wildlife habitat (Flynn, 1999).

## 9.1 Douglas-fir/ Dull Oregon Grape (*Pseudotsuga menziesii*/ *Mahonia nervosa*)

### 9.1.1 Sources


### 9.1.2 Conservation Status

This community is restricted to low elevations along southern Vancouver Island from Bowser to Victoria, the Gulf Islands south of Cortes Island, a narrow strip along the Sunshine Coast between Powel River and Lund, Halfmoon Bay, and on the Fraser River delta. Less than 1% (possibly < 0.5%) of the entire CDF zone remains in mature or old-growth conditions in B.C.. and all of these old remnants are small fragments (<40 ha). Conservation rankings for the Douglas-fir/dull Oregon grape community are listed in Table 11.

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S2</td>
</tr>
<tr>
<td>Global</td>
<td>Unknown; similar communities are ranked G3 and G2G3</td>
</tr>
</tbody>
</table>

9.1.3 Douglas-fir/ Dull Oregon Grape Community Composition and Habitat Characteristics

This plant community was originally the modal community type for the CDFmm; it was the most common and widespread community type of the subzone and may have covered as much as 135 000 ha. Today, most of the remaining occurrences are young secondary forests. Table 12 presents community layers and their characteristic species composition; Table 13 outlines important habitat components and their typical characteristics.

Table 12. Characteristic Species Composition in Relation to Community Layers

<table>
<thead>
<tr>
<th>Community Layers</th>
<th>Characteristic Species Composition</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moss Layer</td>
<td>Oregon beaked moss (<em>Kindbergia oregana</em>), electrified cat’s-tail moss (<em>Rhytidiadelphus triquetrus</em>), step moss (<em>Hylocomium splendens</em>)</td>
</tr>
<tr>
<td>Herb Layer</td>
<td>Broad-leaved star flower (<em>Trientalis borealis</em> ssp. <em>latifolia</em>), sword fern (<em>Polystichum munitum</em>), bracken fern (<em>Pteridium aquiline</em>)</td>
</tr>
<tr>
<td>Shrub Layer</td>
<td>Dull Oregon grape (<em>Mehonia nervosa</em>), salal (<em>Gaultheria shallon</em>), oceanspray (<em>Holodiscus discolor</em>), trailing blackberry (<em>Rubus ursinus</em>)</td>
</tr>
<tr>
<td>Tree Layer</td>
<td>Douglas-fir (<em>Pseudotsuga menziesii</em>), grand fir (<em>Abies grandis</em>), western redcedar (<em>Thuja plicata</em>)</td>
</tr>
</tbody>
</table>
Table 13. Douglas-fir/ Dull Oregon Grape Plant Community Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Structural Stage</td>
<td>• The more structurally complex stands are usually &gt; 80 years, mature forest, and old-growth</td>
</tr>
<tr>
<td>Disturbance Regime</td>
<td>• Infrequent stand initiating events (~150-200 years); Primarily medium to high intensity small scale crown fires (~5-50 ha)</td>
</tr>
<tr>
<td></td>
<td>• Small surface fires due to aboriginal and post contact burning (~20-100 years)</td>
</tr>
<tr>
<td></td>
<td>• Occasionally windthrow</td>
</tr>
<tr>
<td></td>
<td>• Direct tree mortality due to root rot, and bark beetles</td>
</tr>
<tr>
<td></td>
<td>• Indirect tree mortality via predisposition of attacked trees to blowdown</td>
</tr>
<tr>
<td>Aspect</td>
<td>• All Aspects</td>
</tr>
<tr>
<td>Slope position</td>
<td>• Middle Slope</td>
</tr>
<tr>
<td>Elevation</td>
<td>• 0-200 m asl</td>
</tr>
<tr>
<td>Parent Material</td>
<td>• Mostly morainal, occasionally colluvial or marine</td>
</tr>
<tr>
<td>Soils</td>
<td>• Typically Orthic Dystric Brunisols; Mostly sandy loams, some gravelly, sandy, and silty loams</td>
</tr>
<tr>
<td>Moisture Regime</td>
<td>• Moderately dry moisture regime (relative within subzone)</td>
</tr>
<tr>
<td>Nutrient Regime</td>
<td>• Very poor to medium</td>
</tr>
</tbody>
</table>

9.1.4 Important Conservation Considerations

**Distribution:** Due to urbanization and industrialization in both B.C. and the U.S. Pacific Northwest, very little of this kind of coastal old growth is left in the world. Approximately 3% (6700ha) of the entire CDFmm subzone occurs in protected areas. What little remains outside of protected areas is primarily on private land and may continue to be lost to urban development and forest harvesting.
Climate Change: One climate change impact model projects a rapid expansion of the CDF zone, with a 336 percent increase by 2085 (Hamann and Wang, 2006). Paleoecological studies confirm that the CDF zone was much larger under warmer, drier historical climactic conditions (Brown and Hebda, 2002). These studies imply that the CDF zone will likely become an increasingly dominant ecosystem in B.C., and what little remains of the old-growth forests in the CDF should be protected to provide genetic source pools from which these ecosystems can expand (Wilson and Hebda, 2008). Given their current limited distribution, it is also imperative to consider the ecological restoration of degraded forested areas within this zone as a mechanism to increase the resiliency of B.C.’s ecosystems to climate change, as well as to mitigate greenhouse gas (GHG) emissions.

Disturbance Regime: The small surface fires due to aboriginal and post-contact burning that were historically common across the landscape likely contributed to the maintenance of a moderately open forest canopy and Douglas-fir regeneration. Mature Douglas-fir trees have thick bark, which protects them from the low-intensity fires that used to occur approximately every 100 to 300 years (Flynn, 1999). Such fires maintained the dominance of Douglas-fir by controlling the growth of competing trees, and they also reduced the risk of high-intensity fires by preventing tinder-dry debris from building up on the forest floor (Flynn, 1999). The ecological integrity of all occurrences of the CDF zone has been compromised by the unnatural ecosystem dynamics resulting in part from decades of fire suppression and prevention: now that forest fires are suppressed, the Douglas-firs are in danger of being replaced by other conifers or killed by high-intensity fires that humans will not be able to control.

Fragility: Moderately Fragile. Soils around rocky outcrops and ridges may be shallow and thus susceptible to soil degradation through erosion and nutrient loss. However, these ecosystems may recover quite quickly after disturbances providing there are biological legacies such as snags, CWD, and some remnant large older trees, and there has been no displacement or damage to the soil. Summer droughts can delay forest regeneration after a disturbance.

Invasive Species: These forests are very susceptible to the invasion and spread of exotic species. Introduced garden species such as English ivy (Hedera helix), spurge-laurel (Daphne laureola), Scotch broom (Cytisus scoparius), and gorse (Ulex europaeus) are particularly prevalent in stands adjacent to urban or farm areas.

Cross References: There are several identified wildlife elements that are associated with this ecosystem type and should be considered during the WHA selection process and subsequent restoration process (where applicable). These include the Douglas-fir/Alaska oniongrass community, Lewis’s woodpecker, “Interior” western screech-owl, Keen’s long-eared myotis, and the red-legged frog.
9.1.5 Potential Roles of Second-growth Forest in Conservation

As previously discussed, most of the remaining occurrences of this plant community are younger second-growth forests. Ways in which these second-growth forests can contribute to the conservation of the Douglas fir/dull Oregon grape plant community are shown in Table 14.


<table>
<thead>
<tr>
<th>Community Type</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment/Restoration</th>
</tr>
</thead>
<tbody>
<tr>
<td>Douglas-fir/ Dull Oregon Grape</td>
<td>X</td>
<td></td>
<td></td>
<td>X</td>
</tr>
</tbody>
</table>

Priority for protection should be:

a) Any old or mature (structural stage 6-7) occurrences that are in a relatively natural state;
b) The most structurally complex stands containing biological legacies such as old residual conifers;
c) Relatively lightly damaged sites that can recover to a more natural state;
d) Occurrences adjacent to other natural plant communities;
e) Occurrences that will contribute to a network of reserve areas; and
f) Occurrences in areas where the forest community has been severely depleted.

The size of WHAs should be based on the extent of the plant community and include a minimum 80m buffer within adjacent natural areas to help maintain interior forest conditions. WHAs should be approximately 50 ha when there is a relatively pure composition, and approximately 200 ha when the understory community or tree layer has a relatively patchy distribution, or when the plant community occurs in a matrix with other at-risk plant communities. In addition, boundaries should be designed to minimize edge effects and be wind firm to the extent possible.
In some cases it may be necessary to restore the plant communities to a more natural state, i.e., the same species composition, physical structure, and ecological processes as natural examples of this community. Structures that are particularly important include large old trees, a range of tree sizes, large snags, CWD, canopy depth and roughness, multiple vegetation layers, and understory patchiness (Section 3). Methods for recruiting these structural characteristics can be found in the habitat recruitment section (Section 5). In addition to physical structure recruitment, it may be necessary to reduce fuel accumulations and shade-tolerant understory vegetation through prescribed burning to mimic historical ground fires (where feasible), and manual or mechanical removal that could potentially be combined with piling and burning (Section 5.6).

9.2 Douglas-fir/Alaska Oniongrass (*Pseudotsuga menziesii*/Melica subulata)

9.2.1 Sources


9.2.2 Conservation Status

This community is restricted to low elevations along southern Vancouver Island from Bowser to Victoria, and on the Gulf Islands south of Hornby and Lasqueti islands. Less than 1% (possibly < 0.5%) of the entire CDF zone remains in mature or old growth conditions in B.C. This community has a very restricted range and has been depleted to near extirpation in B.C. The conservation rankings for the Douglas-fir/Alaska oniongrass community are listed in Table 15.

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S1</td>
</tr>
<tr>
<td>Global</td>
<td>Unknown</td>
</tr>
</tbody>
</table>
9.2.3 Douglas-fir/ Alaska Oniongrass Community Composition and Habitat Characteristics

Depending on disturbance history, successional stage and chance, this plant community can have several manifestations. However, it usually includes an open canopy of both Douglas-fir and Garry oak as dominant or frequent trees. This community historically occurred infrequently and mostly in small patches within the CDF matrix. Table 16 presents community layers and their characteristic species composition; Table 17 outlines important habitat components and their typical characteristics.

Table 16. Characteristic Species Composition in Relation to Community Layers

<table>
<thead>
<tr>
<th>Community Layers</th>
<th>Characteristic Species Composition</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moss Layer</td>
<td>Electrified cat’s-tail moss (<em>Rhytidiadelphus triquetrus</em>)</td>
</tr>
<tr>
<td>Shrub Layer</td>
<td>Hairy honeysuckle (<em>Lonicera hispidula</em>)</td>
</tr>
<tr>
<td>Tree Layer</td>
<td>Douglas-fir (<em>Pseudotsuga menziesii</em>), Garry oak (<em>Quercus garryana</em>)</td>
</tr>
</tbody>
</table>
Table 17. Douglas-fir/Alaska Oniongrass Plant Community Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Structural Stage</td>
<td>• 3-7</td>
</tr>
<tr>
<td>Disturbance Regime</td>
<td>• A variant of frequent stand-maintaining fires (NDT4)</td>
</tr>
<tr>
<td></td>
<td>• Infrequent stand initiating events (~150-200 years); Primarily medium to high intensity small scale crown fires (~5-50 ha)</td>
</tr>
<tr>
<td></td>
<td>• Small surface fires due to aboriginal and post contact burning (~20-100 years)</td>
</tr>
<tr>
<td></td>
<td>• Occasionally windthrow</td>
</tr>
<tr>
<td></td>
<td>• Direct tree mortality due to root rot, defoliator insects, bark beetles, and occasionally severe drought</td>
</tr>
<tr>
<td></td>
<td>• Indirect tree mortality via predisposition of attacked trees to blowdown</td>
</tr>
<tr>
<td>Aspect</td>
<td>• Typically southerly aspects</td>
</tr>
<tr>
<td>Elevation</td>
<td>• 0-150 m asl</td>
</tr>
<tr>
<td>Parent Material</td>
<td>• Inactive colluvial, sometimes morainal</td>
</tr>
<tr>
<td>Soils</td>
<td>• Sombric Brunisols; shallow, mostly sandy loams, often with moderate coarse fragment content</td>
</tr>
<tr>
<td>Moisture Regime</td>
<td>• Very dry</td>
</tr>
<tr>
<td>Nutrient Regime</td>
<td>• Rich to very rich</td>
</tr>
</tbody>
</table>

9.2.4 Important Considerations

**Distribution:** Intact remnants of this community are all small fragments, including those occurrences in protected areas. The few known occurrences in parks and protected areas are in active recreation areas and are fragmented by trails. Most of what occurs outside of protected areas is on private land, and there are few if any high quality occurrences left.

**Climate Change:** One climate change impact model projects a rapid expansion of the CDF zone, with a 336 percent increase by 2085 (Hamann and Wang, 2006). Paleoecological studies confirm that the CDF zone was much larger under warmer, drier historical climactic conditions (Brown and Hebda, 2002). These studies imply that the CDF zone will likely become an
increasingly dominant ecosystem in B.C., and what little remains of the old-growth forests in the CDF should be protected to provide genetic source pools from which these ecosystems can expand (Wilson and Hebd, 2008). Given their current limited distribution, it is also imperative to consider the ecological restoration of degraded forested areas within this zone as a mechanism to increase the resiliency of B.C.’s ecosystems to climate change, as well as to mitigate greenhouse gas (GHG) emissions.

**Disturbance Regime:** The small surface fires due to aboriginal and post-contact burning that were historically common across the landscape likely contributed to the maintenance of a moderately open forest canopy and Douglas-fir regeneration. Mature Douglas-fir trees have thick bark, which protects them from the low-intensity fires that used to occur approximately every 100 to 300 years (Flynn, 1999). Such fires maintained the dominance of Douglas-fir by controlling the growth of competing trees, and they also reduced the risk of high-intensity fires by preventing tinder-dry debris from building up on the forest floor (Flynn, 1999). The ecological integrity of all occurrences of the CDF zone has been compromised by the unnatural ecosystem dynamics resulting in part from decades of fire suppression and prevention: now that forest fires are suppressed, the Douglas-firs are in danger of being replaced by other conifers.

**Fragility:** Very Fragile. Soils around rocky outcrops and ridges may be shallow and susceptible to soil degradation through erosion and compaction. These ecosystems recover slowly after stand replacing disturbances due to droughty soils, invasion by exotic species, and slow recruitment of structural elements such as snags, large living trees, and CWD. Additionally, moisture stress can delay forest regeneration after a disturbance.

**Grazing:** Grazing and browsing by both domestic livestock (goats and sheep in particular), as well as deer (native and introduced), is a significant threat to this ecosystem.

**Invasive Species:** These forests are very susceptible to the invasion and spread of exotic species. Introduced garden species such as spurge-laurel (*Daphne laureola*), Scotch broom (*Cytisus scoparius*), and gorse (*Ulex europaeus*) are particularly prevalent in this ecosystem type.

**Cross References:** There are several identified wildlife elements that are associated with this ecosystem type and should be considered during the WHA selection process and subsequent restoration process (where applicable). These include the Douglas-fir/ Dull Oregon grape plant community, Lewis’s woodpecker, Keen’s long-eared myotis, and the “Queen Charlotte“ goshawk.
9.2.5 Potential Roles of Second-growth Forest in Conservation

Most of the remaining occurrences of this plant community are younger second-growth forests. Ways in which these second-growth forests can contribute to the conservation of the Douglas fir/Alaska Oniongrass plant community are shown in Table 18.

Table 18. Potential Roles of Second-growth Forest in the Conservation of the Douglas-fir/Alaska Oniongrass Plant Community

<table>
<thead>
<tr>
<th>Community Type</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment/Restoration</th>
</tr>
</thead>
<tbody>
<tr>
<td>Douglas-fir/Alaska Oniongrass plant community</td>
<td>X</td>
<td>X</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

All remaining occurrences greater than 3ha in any structural stage and in a relatively natural state should be designated as WHAs. Priority for protection should be:

a) The oldest occurrences that are in a relatively natural state;

b) The most structurally complex stands containing biological legacies such as old residual Douglas-fir and Garry oak;

c) Relatively lightly damaged sites that can recover to a more natural state;

d) Occurrences adjacent to other natural plant communities; and

e) Occurrences that will contribute to a network of reserve areas.

The size of WHAs should be based on the extent of the plant community, most of which are <20 ha. WHAs should include a 60m buffer (approximate). In addition, boundaries should be designed to minimize edge effects and to the extent possible be wind-firm. That said, this community tends to be adapted to strong winds.
In some cases it may be necessary to restore the plant communities to a more natural state, i.e., the same species composition, physical structure, and ecological processes as natural examples of this community. Structures that are particularly important include large living trees, a range of tree sizes, large snags, CWD, canopy depth and roughness, open canopy conditions or a range from open to closed conditions, multiple vegetation layers, and understory patchiness with a sparse shrub layer (Section 3). Methods for recruiting these structural characteristics can be found in the habitat recruitment section (Section 5). In addition to physical structure recruitment, it may be necessary to implement silviculture and prescribed burning practices that reduce conifer ingress, fuel accumulation, and shade-tolerant understory vegetation. It may also be necessary to maintain or recruit Garry oak seedlings as well as minimize the introduction and spread of invasive species.

10 Species Specific Information

10.1 Scouler’s Corydalis (*Corydalis scouleria*)

10.1.1 Sources

COSEWIC, 2006
RS- Recovery Strategy

10.1.2 Conservation Status

Scouler’s corydalis is limited to the Pacific Northwest, where it occurs west of the Cascades, from northwestern Oregon, through the Olympic Peninsula to southwestern Vancouver Island. It is frequent to common in Oregon and Washington (>100 extant populations), but rare in B.C. (approximately 24 extant populations). Within B.C., this species is restricted to the southwestern portion of Vancouver Island. It is currently Blue-listed under the Species at Risk Act; however, it was reassessed as Not at Risk by COSEWIC in 2006 after larger populations were discovered after intensive searching, and the extent of potential habitat was increased (COSEWIC, 2006). Table 19 shows the conservation rankings for Scouler’s corydalis throughout its range in North America.
Table 19. Conservation Rankings for the Scouler’s Corydalis

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S3S4</td>
</tr>
<tr>
<td>Canada</td>
<td>N3N4</td>
</tr>
<tr>
<td>WA</td>
<td>S?</td>
</tr>
<tr>
<td>OR</td>
<td>S?</td>
</tr>
<tr>
<td>Global</td>
<td>G4</td>
</tr>
</tbody>
</table>

10.1.3 Scouler’s Corydalis Habitat Characteristics

The Scouler’s corydalis is one of five Corydalis species indigenous to Canada, four of which occur in B.C. Scouler’s corydalis is a tall perennial herbaceous understory plant with thick rhizomes. Although all of the known B.C. occurrences are adjacent to watercourses, it is classified as a facultative, rather than obligate, wetland species. Scouler’s corydalis can be cultivated in shady or woodland situations and is grown by gardeners in Europe and the Pacific Northwest. Typical stand characteristics corresponding with each habitat component of the Scouler’s corydalis are listed in Table 20.
Table 20. Scouler’s Corydalis Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Stand Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Structural stage</td>
<td>• Early seral stage forest</td>
</tr>
<tr>
<td></td>
<td>• Occasionally mature to old forest</td>
</tr>
<tr>
<td>Forest Type</td>
<td>• Deciduous or mixed forests</td>
</tr>
<tr>
<td>Elevation</td>
<td>• 0-200 m asl</td>
</tr>
<tr>
<td>Slope</td>
<td>• 0-45%</td>
</tr>
<tr>
<td>Aspect</td>
<td>• West, south-west, and north to north-east</td>
</tr>
<tr>
<td>Structural Features</td>
<td>• Cool, moist, shady riparian areas</td>
</tr>
<tr>
<td></td>
<td>• From moderately large rivers to small tributaries and roadside ditches draining into streams</td>
</tr>
<tr>
<td>Soil</td>
<td>• Prime Habitat: Fine silts and sediments of floodplains, alluvial flats, and river benches</td>
</tr>
<tr>
<td></td>
<td>• Other Habitat: silty soil combined with coarser floodplain material or river smoothed rocks on stream banks, river terraces, islands, and bottomlands</td>
</tr>
<tr>
<td></td>
<td>• Humus Form: Nitrogen rich moder and mull humus</td>
</tr>
</tbody>
</table>

10.1.4 Important Considerations

Reproduction and Dispersal: This plant species reproduces well vegetatively, generating annual stems apically from the rhizome. Although a number of pollinators also visit this species, it appears to have low sexual reproductive success. Dispersal occurs when ripe seed capsules explode elastically after a disturbance, through vectors such as ants, and when periodic flooding facilitates the movement of both rhizome fragments and seeds. That said, seeds are
short lived, dry out easily, and possibly have limited dispersal. These characteristics, combined with the limited number of populations in the province, may result in a lack of genetic heterogeneity.

**Population Trends:** The 36 extant populations in B.C. are restricted to two major watersheds (Klanawa, Nitinat, and immediately east of the Nitinat near Cowichan Lake) on southwestern Vancouver Island. These populations range in size from one stem to 462,000 stems growing over a 3.4 ha area.

**Habitat Threats:** Forest recreational activity, such as camp sites and hiking trails, tend to be situated within riparian areas (e.g., Nitinat Lake, River, and Provincial Park) and may impact the habitat of Scouler’s corydalis. In addition, logging and forest management practices such as road building adjacent to riparian areas may damage individuals, alter suitable habitat by removing shade cover, or cause downstream erosion or flooding which could result in alterations or loss of habitat.

**Forest Structure:** Forest succession will lead to changes in forest structure, which is predicted to eventually shade out the corydalis.

**Protection:** Nearly half of the total known stems of Scouler’s corydalis (over 400,000 out of 848,000) found in eight locations in the Nitinat River drainage are within Wildlife Habitat Areas to be managed for this species. In addition, two of the Scouler’s corydalis populations in British Columbia occur in Provincial Parks and are protected by the Provincial Park Act, and one occurs in an Ecological Reserve and is protected under the Ecological Reserves Act (COSEWIC, 2006).

**Cross References:** Forest management practices designed to benefit the Marbled murrelet, “Queen Charlotte” goshawk, and “Vancouver Island” common water shrew will also benefit the Scouler’s corydalis, particularly efforts to maintain or restore riparian habitats.

10.1.5 **Potential Roles of Second-growth Forest in Conservation**

Ways in which second-growth forest can contribute to the conservation of the Scouler’s corydalis are shown in Table 21.
Given that most known populations occur within riparian areas, the most important mechanism currently in place to protect this species is the riparian management recommendations under the results-based code. New locations of this species should be designated as WHAs to ensure their long-term persistence and remove any discretionary powers the riparian management recommendations provide. WHAs for this species will typically be <10 ha, however, the size of the WHA will ultimately depend on the extent of the population, available suitable habitat in the vicinity of the population, and surrounding conditions. Most known populations are less than 0.5 ha in size, but at least one population has been mapped over 3.4 ha (COSEWIC, 2006). The core of the WHA should coincide with the parameter of the population, and management zones will typically be approximately 50 m. However, the management zone is intended to act as a buffer (Section 1.4.3) to preserve ambient and hydrological conditions and may extend up to 250 m depending on site-specific characteristics. Furthermore, although large canopy gaps should be avoided, it may be necessary to employ treatments within the WHA with the aim of maintaining or improving stand characteristics for Scouler’s corydalis (Section 5.5).

### 10.2 Keen’s Long-eared Myotis (*Myotis keenii*)

#### 10.2.1 Sources

B.C. Ministry of Water, Land and Air Protection, 2004d.

#### 10.2.2 Conservation Status

The Keen’s long-eared myotis is restricted to the Pacific Northwest coast; British Columbia is in the center of its range with records extending from western Washington to southeast Alaska. Within B.C., this species is found on Vancouver Island, the Queen Charlotte Islands, and the
mainland coast. There is only one maternity colony and 18 occurrences known in B.C., but more are likely to exist. The Keen’s long-eared myotis has been Red-listed in B.C. and designated as a species of Special Concern in Canada. Additional information on conservation rankings of this species can be found in Table 22.

### Table 22. Conservation Rankings for the Keen’s Long-eared Myotis

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S1S3</td>
</tr>
<tr>
<td>Alaska</td>
<td>SH</td>
</tr>
<tr>
<td>Washington</td>
<td>SH</td>
</tr>
<tr>
<td>Canada</td>
<td>N1N3</td>
</tr>
<tr>
<td>Global</td>
<td>G2G3</td>
</tr>
</tbody>
</table>

#### 10.2.3 Keen’s Long-eared Myotis Habitat Characteristics

Very little is known regarding the home range of Keen’s long-eared myotis, however, preliminary research indicates that they may not move large distances in the summer and they may have small home ranges. There is little information on site fidelity for this species; however, other species of *Myotis* show high fidelity to maternity roost and hibernation sites. Although foraging behavior for this species is largely unknown, its small size, low wing-loading ratio, and very low intensity echolocation call makes it well adapted for flying and foraging within structurally complex old forests. This indicates that low elevation coastal forests and riparian areas may be important foraging areas. Structural complexity is also important in natural roosting sites; tree cavities in wildlife trees (decay class 2 or greater) and loose bark (decay class 4 or greater) are important roosting sites and may be limiting in some parts of their range. Keen’s long-eared myotis appears to be associated with cool wet coastal montane forests and karst features. Typical habitat characteristics of the Keen’s Long-eared myotis are listed in Table 23.
Table 23. Keen’s Long-eared Myotis Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Stand Characteristics</th>
</tr>
</thead>
</table>
| Forest Structural Stages    | • Mature forest  
|                             | • Old forest  |
| Roosting Sites              | • Southwest facing rock crevices  
|                             | • Geothermally heated rocks  
|                             | • Tree cavities  
|                             | • Bark crevices  |
| Summer Maternity Roosts     | • <240m asl  
|                             | • Rock faces and knolls with crevices  
|                             | • Solar or geothermal heating  |
| Summer Foraging Area        | • <240m asl  
|                             | • Ponds and riparian areas with high insect productivity  
|                             | • Structurally complex forests  |
| Winter Hibernation Sites    | • >400m asl  
|                             | • Caves >100m long  
|                             | • Caves with stable temperatures between 2.4-4.0°C  
|                             | • Caves with 100% relative humidity  |

10.2.4 Important Considerations

**Habitat Threats:** Although information on habitat trends is generally lacking, mineral extraction, logging and road building are known to negatively impact wildlife trees and other potential roosting sites, thus it is assumed that habitat quality and quantity is generally declining.

**Sensitivity:** Disturbance during hibernation and while raising young is a major concern. Disturbances may result from recreational activities such as caving, or industrial activities such as blasting for road construction.

**Distribution:** This species is sparsely distributed and has a limited distribution, which could increase its risk of extirpation or extinction.
Protection: The Keen’s long-eared myotis is protected from killing, wounding, hunting or removal from its habitat under the Wildlife Act. The only known maternity colony is protected within Gwaii Haanas National Park Reserve, and one other known female and young roost site is managed within an existing WHA (MWLAP, 2004d).

Cross References: Forest management practices designed to benefit the marbled murrelet, Quatsino cave amphipod, “Queen Charlotte” goshawk and “Vancouver Island” common water shrew will benefit the Keen’s long-eared myotis. The recruitment of wildlife trees in particular will benefit both this species and the “Queen Charlotte” goshawk in second-growth forests.

10.2.5 Potential Roles of Second-growth Forest in Conservation

Ways in which second-growth forest can contribute to the conservation of the Keen’s long-eared myotis are shown in Table 24.

Table 24. Potential Roles of Second-growth Forest in the Conservation of the Keen’s Long-eared Myotis

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Keen’s Long-eared Myotis</td>
<td>X</td>
<td>X</td>
<td></td>
<td>X</td>
</tr>
</tbody>
</table>

Known colonies, roosting sites, adjacent foraging areas and movement corridors should be protected through the development of WHAs. WHAs will typically range from 30 to 50 ha; however, this will depend on site-specific factors such as the type of feature being protected, location of roosting trees, presence of wetlands or other riparian areas, and potential movement corridors. For example, if a cave entrance is the focal point of a WHA, there should be a 100 m core area and a 200 m buffer management zone (total 300m). In addition, the WHA should include a minimum 20 m buffer on either side of any water body within 500 m of the core area. This is important because riparian areas adjacent to core areas may be movement corridors and or foraging grounds and may serve as sources of connectivity between the core area and other potential foraging grounds. In addition, if there are no natural roosting trees
within acceptable proximity to the core area, it may be possible to recruit these structural elements (see Section 5.3).

10.3 “Vancouver Island” Northern Pygmy-Owl (*Glaucidium gnomas swarthi*)

10.3.1 Sources

B.C. Ministry of Water, Land and Air Protection, 2004e.

10.3.2 Conservation Status

Three subspecies of northern pygmy-owl breed in B.C., including *Glaucidium gnomas swarthi*, which is endemic to Vancouver Island and possibly the adjacent Gulf Islands. The “Vancouver island” northern pygmy-owl is on the provincial Blue List in B.C., and its status in Canada has yet to be determined. Additional information on conservation rankings of this species can be found in Table 25.

Table 25. Conservation Rankings for the “Vancouver Island” Northern Pygmy-Owl

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S3</td>
</tr>
<tr>
<td>Canada</td>
<td>N3</td>
</tr>
<tr>
<td>Global</td>
<td>G5T3Q</td>
</tr>
</tbody>
</table>

10.3.3 “Vancouver Island” Northern Pygmy-Owl Habitat Characteristics

Little is known about the specifics of the biology of the northern pygmy-owl on Vancouver Island, so much of the following information is inferred from the limited data available for other races of this species. Although home range details for this species are lacking, northern pygmy-owls in Oregon have been documented as little as 600 m apart, in Colorado home range size is approximately 75 ha, and in Washington nests have been found as little as as 1.25 km apart; thus, it can be assumed that “Vancouver Island” northern pygmy-owls are usually sparsely distributed within appropriate habitat. Typical habitat characteristics of the “Vancouver Island” northern pygmy-owl are listed in Table 26.
Table 26. “Vancouver Island” Northern Pygmy-Owl Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Stand Characteristics</th>
</tr>
</thead>
</table>
| Nest Areas         | • Elevation: 50-950 m asl (likely occur at higher elevations)  
|                    | • Structural Stages: young forest, mature forest, old forest  
|                    | • Coniferous forests  
|                    | • Mixed riparian forests  
|                    | • Pure deciduous stands |
| Nest Cavities      | • Cavities excavated by northern flickers (*Colaptes auratus*), hairy woodpeckers (*Picoides villosus*), and to a lesser extent pileated woodpeckers (*Dryocopus pileatus*) are likely the most useful for the “Vancouver Island” northern pygmy-owl |
| Foraging Areas     | • Structural Stages: shrub/herb, pole/sapling, young forest, mature forest, old forest  
|                    | • Forest edges  
|                    | • Along roads through forested areas  
|                    | • Openings within continuous forest  
|                    | • Open stands  
|                    | • Riparian corridors  
|                    | • Mosaics of forested and open habitats  
|                    | • Lakeshores  
|                    | • Higher elevations where stands tend to thin |

10.3.4 Important Considerations

**Population Trends:** The population size of this species is unknown, but presumed to be small, and generally thought to be declining. Sufficient data is lacking and long-term trends cannot be estimated; there are very few breeding records and no data on breeding ecology.

**Nest Sites:** Little information exists for pair formation, nest site selection, or nest building behavior; however, the northern pygmy-owl is an obligate secondary cavity nester and is dependent on woodpecker nest cavities or natural cavities as nest sites. Information about the nesting habitats for the northern flickers, hairy woodpeckers, and the pileated woodpecker may assist in identifying potential nesting habitats for the “Vancouver Island” northern pygmy-owl. The diameter at breast height (dbh) of nest trees for these three species of woodpeckers differ depending on the species of woodpecker and nesting location; dbh may also differs depending on species of woodpecker and species of nest tree. Northern flickers select wildlife trees of decay classes 3-6 and trees with all their bark intact for nesting. Hairy woodpeckers use wildlife tree classes 2-7 for nesting, and a high proportion of nests have been found in classes 4 and 5;
suggesting they prefer to nest in trees in a considerable state of decay. That said, hairy woodpecker nests have been found more often than expected in snags that had all of their bark present. Pileated woodpeckers nest and roost sites are usually in the upper canopy of trees with broken tops. These trees range from alive and healthy to dead with most of the branches absent (decay classes 2-5). In one study, all of the nests of the “Vancouver Island” northern pygmy-owl were found in old woodpecker cavities in coniferous trees and were most often found on steep hillsides. Nest cavities have been reused by this species, although it is not known if this was by the same or different individuals. Furthermore, this owl tends to breed near the edge of forest openings rather than in interior forest. So although this subspecies may be a habitat generalist because it uses a variety of forest types during the breeding season, it is likely that the availability of suitable nesting cavities is the limiting factor influencing “Vancouver Island” pygmy-owl abundance and distribution.

Foraging Behavior: Northern pygmy-owls are crepuscular or diurnal, use a perch and pounce hunting method, and prey on a wide range of small mammals, birds, invertebrates, reptiles, and amphibians. This species is a prey generalist, so although the availability of prey could affect its distribution, this is not likely to be a limiting factor. Foraging is usually associated with forest edges including road edges and regenerating clearcuts. It appears that this species prefers habitats with diverse understory structure that provides habitat for a variety of small mammal and bird prey.

Movement: Although this is a resident species, there may be some seasonal altitudinal movement with birds descending to lower elevations in the fall.

Predation: Anecdotal evidence suggests that the increasing population of larger barred owls (Strix varia) following forest fragmentation may contribute to local declines in northern pygmy-owls; however, data on the impact of barred owls is not available.

Protection: The northern pygmy-owl, its nests, and its eggs are protected under the provincial Wildlife Act.

Cross References: This species of owl is largely dependent on woodpecker cavities for nest sites, so management practices that benefit woodpeckers (see “Queen Charlotte” hairy woodpecker, Section 10.17) will also enhance habitat for the “Vancouver Island” northern pygmy-owl.
10.3.5 Potential Roles of Second-growth Forest in Conservation

Ways in which second-growth forest can contribute to the conservation of the “Vancouver Island” northern pygmy-owl are shown in Table 27.

Table 27. Potential Roles of Second Growth Forest in the Conservation of the “Vancouver Island” Northern Pygmy-Owl

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment</th>
</tr>
</thead>
<tbody>
<tr>
<td>“Vancouver Island” northern pygmy-owl</td>
<td></td>
<td></td>
<td>X</td>
<td>X</td>
</tr>
</tbody>
</table>

With smaller clearcuts, wildlife tree retention areas, and riparian reserve zones, large scale population declines are unlikely. In fact, given that this owl prefers edge habitat to continuous forest, current forest practices might actually be increasing the available habitat for this species. Under the results-based code, OGMAs, riparian reserves, and wildlife tree retention areas may partially address the requirements of this owl. However, local population fluctuations can be expected if owl territories are logged without adequate retention of wildlife tree nesting habitat. Given that foraging for this species can be accommodated in younger stands, WHAs should focus on patches of mature or old forest that include potential nest trees and recruitment trees to maintain breeding pairs. Alternatively, it may be possible to recruit wildlife trees in known nest or occupied residence areas where this habitat component has been diminished; methods for recruiting wildlife trees can be found in Section 5.3. Because this species of owl is largely dependent on woodpecker cavities for nest sites, management practices that benefit woodpeckers will also enhance habitat for the “Vancouver Island” northern pygmy-owl. Pertinent information regarding woodpecker requirements can be found in Tables 28 and 29. In general, WHAs for this subspecies should typically be 80 to 100 ha, however, the size will depend on site-specific factors. WHAs should be designed to minimize disturbance and maintain suitable foraging habitat. This will likely include a 12 ha core area surrounding known nests, and a 300-400 m management zone. The management zone boundary should be defined based on the remaining home range, which should be estimated based on suitable habitat. In areas where the exact location of nest trees is unknown, core areas should include highly suitable nest trees or known roost sites.
Table 28. Dbh (mean ± SD) (cm) of Nest Trees of Hairy Woodpeckers (HAWO), Northern Flickers (NOFL), and Pileated Woodpeckers (PIWO) in Four Locations (MWLAP, 2004e).

<table>
<thead>
<tr>
<th>Forest</th>
<th>Location</th>
<th>n</th>
<th>HAWO</th>
<th>NOFL</th>
<th>PIWO</th>
<th>Citation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Western Hemlock</td>
<td>Oregon Coast Ranges</td>
<td>23</td>
<td>72.2 ± 48.0</td>
<td>9</td>
<td>95.8 ± 30.0</td>
<td>68.9 ± 25</td>
</tr>
<tr>
<td>Mixed Conifer to Douglas-fir</td>
<td>South Cascades</td>
<td>18</td>
<td>73.9 ± 33.4</td>
<td>3</td>
<td>127.7 ± 38.5</td>
<td>88.0 ± 19.8</td>
</tr>
<tr>
<td>CWHxm, CWHvm, MHmm</td>
<td>Northern Vancouver</td>
<td>73</td>
<td>78.6 ± 28.1</td>
<td>85</td>
<td>73.1 ± 3.4</td>
<td>84.2 ± 17.5</td>
</tr>
<tr>
<td>CWHxm, CDF</td>
<td>SE Vancouver Island</td>
<td>7</td>
<td>82 ± 42</td>
<td>85</td>
<td>73.1 ± 3.4</td>
<td>84.2 ± 17.5</td>
</tr>
</tbody>
</table>

Table 29. Dbh (mean ± SD) (cm) of Nest Trees of Hairy Woodpeckers (HAWO), and Northern Flickers (NOFL) by Tree Species Found in the Nimpkish Valley (after Deal and Setterington, 2000 in MWLAP, 2004e)

<table>
<thead>
<tr>
<th>Species</th>
<th>HAWO</th>
<th>n</th>
<th>NOFL</th>
<th>n</th>
</tr>
</thead>
<tbody>
<tr>
<td>Amabilis fir</td>
<td>66.4 ± 23.0</td>
<td>8</td>
<td>71.0 ± 36.6</td>
<td>11</td>
</tr>
<tr>
<td>Douglas-fir</td>
<td>95.1 ± 37.6</td>
<td>15</td>
<td>110.6 ± 37.2</td>
<td>15</td>
</tr>
<tr>
<td>Western Hemlock</td>
<td>76.6 ± 22.9</td>
<td>48</td>
<td>64.7 ± 18.7</td>
<td>39</td>
</tr>
<tr>
<td>Western White Pine</td>
<td>77.5 ± 34.6</td>
<td>2</td>
<td>60.1 ± 18.1</td>
<td>9</td>
</tr>
</tbody>
</table>

10.4 Short-eared Owl (*Asio flammeus flammeus*)

10.4.1 Sources

B.C. Ministry of Water, Land and Air Protection, 2004f
COSEWIC, 2008a.
10.4.2 Conservation Status

Short-eared owls breed locally on the south mainland coast, through the Fraser River delta east to Fort Langley, in the south and central Interior north through the Thompson and Chilcotin-Cariboo basins to Prince George, and in the Peace Lowland. It is an uncommon migrant throughout the province. The Fraser River delta is the main wintering area in the province although a few birds winter on southeastern Vancouver Island and in the southern Interior. Population sizes and trends are difficult to assess because this owl is cyclic and nomadic, an unknown portion of the population nests in remote, unsurveyed regions, and even within easily accessible known owl habitat, there has been a lack of consistent standardized census effort. Estimating population size is further complicated by migration patterns because wintering, migrating and resident bird populations overlap. However, Christmas Bird Count data from the Lower Mainland has shown a decline of approximately 3% per year over the last 40 years (COSEWIC, 2008). Within B.C., the short-eared owl is on the provincial Blue List; within Canada it is considered a species of Special Concern. Additional information regarding the conservation status of the short-eared owl can be found in Table 30.

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S3B, S2N</td>
</tr>
<tr>
<td>Alaska</td>
<td>S3N, S5B</td>
</tr>
<tr>
<td>California</td>
<td>S3</td>
</tr>
<tr>
<td>Idaho</td>
<td>S5</td>
</tr>
<tr>
<td>Montana</td>
<td>S4</td>
</tr>
<tr>
<td>Oregon</td>
<td>S4?</td>
</tr>
<tr>
<td>Washington</td>
<td>S4B, S4N</td>
</tr>
<tr>
<td>Canada</td>
<td>N4N, N5B</td>
</tr>
<tr>
<td>Global</td>
<td>G5</td>
</tr>
</tbody>
</table>
10.4.3 Short-eared Owl Habitat Characteristics

Although the majority of this species’ habitat within B.C. is on the mainland, a few birds winter on southeastern Vancouver Island, which is the focus of this report. For this reason, the remainder of this discussion will concentrate on winter habitat. In winter, this species is non-territorial, congregating where there is suitable habitat, adequate roost sites, and a good prey supply; these attributes are likely limiting factors for wintering populations in B.C. Typical habitat characteristics of the short-eared owl are listed in Table 31.

Table 31. Short-eared owl Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Stand Characteristics</th>
</tr>
</thead>
</table>
| Elevation             | • Generally <975m asl  
                        | • Occasionally <2165m asl                                                                          |
| Foraging Area         | • Open areas with patchy vegetation  
                        | • Structural Stage 2-3a  
                        | • Grass lands  
                        | • Marshes  
                        | • Fallow fields                                                   |
| Winter Habitat        | • Structural Stage 2-3a  
                        | • Old-growth fields (variable grass heights and shrub patches)  
                        | • Marine foreshores  
                        | • Estuaries  
                        | • Marshes  
                        | • Grasslands  
                        | • Fallow fields  
                        | • Hay fields  
                        | • Pastures  
                        | • Airports  
                        | • Golf courses                                                   |
| Roosting Sites        | • Mid-height to tall grass, shrubs, or hedgerows  
                        | • Trees where snow depth >5cm  
                        | • Close to foraging areas  
                        | • Protection from weather  
                        | • Concealment from predators  
                        | • Free from human disturbance                                   |
10.4.4 Important Considerations

**Habitat Threats:** In B.C., the primary threat to this species is the loss or degradation of old-field winter habitat. On the coast, estuarine marshes have been eliminated by industrial development and fallow fields have been converted to housing, industry or more intensive agricultural practices. This species is also threatened by forest encroachment into natural grassland habitat. Habitat loss leads directly to a reduction in food availability causing an increase in intra- and interspecific competition. Furthermore, the ongoing loss and fragmentation of habitat make new prey supplies harder to find.

**Prey Abundance:** Prey abundance and accessibility are critical for wintering populations and seem to be strongly linked to old-field habitat; small mammals tend to be more accessible to owls in old-field habitat than in the uniform vegetation of cultivated fields. Short-eared owls are prey specialists concentrating on small rodents (primarily microtines); however, other small mammals, insects, and birds are taken in lesser quantities.

**Protection:** The short-eared owl, its nests, and its eggs are protected under the provincial Wildlife Act.

**Cross References:** Due to similar nesting and foraging habitat needs, measures to maintain early seral stage habitats that benefit the sandhill crane will benefit the short-eared owl, as well.

10.4.5 Potential Roles of Second-growth Forest in Conservation

Ways in which second-growth forest can contribute to the conservation of the short-eared owl are shown in Table 32.

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Short-eared Owl</td>
<td></td>
<td></td>
<td>X</td>
<td>X</td>
</tr>
</tbody>
</table>
Although Short-eared Owls tend to be nomadic, they may reuse areas for breeding, roosting and wintering. Areas that have more than 8 owls using them as traditional communal roosting sites, nest sites, or winter areas should be protected through the creation of WHAs. Depending on site-specific factors, WHAs for traditional roost sites (used for several years) should generally be approximately 5 ha, and WHAs for traditional wintering grounds should generally be approximately 10 ha. WHAs for this species are not intended to encompass the entire area used by the owl, but rather, are meant to maintain key areas used for nesting, roosting, or foraging. The objective of WHAs for this species is to maintain a mosaic of grasslands and old-field habitat in suitable condition to ensure a continued supply of nesting and wintering habitat. Although prey may be more abundant in fallow fields, surrounding second-growth forests with a structural stage of 2-3a may be included to compliment the main foraging area. Furthermore, given the fact that it is the early seral stages that are used by this species, it may be necessary to control forest encroachment into known natural grassland habitat and known forest clearing habitat with controlled prescribed burning or other methods (Section 5.6). It may also be desirable to enhance habitat for voles and other microtines within WHAs to encourage a continued supply of prey for the owl. The suitability of clear cuts for foraging habitat is not known; further research is needed to assess the benefits and drawbacks of including recently clearcut areas in the foraging component of WHAs for this species.

10.5 Queen Charlotte Goshawk (*Accipiter gentilis laingi*)

10.5.1 Sources

B.C. Ministry of Water, Land and Air Protection, 2004g.

10.5.2 Conservation Status

Globally there are less than 1000 Queen Charlotte goshawk individuals. Canada contains 54% of the Queen Charlotte goshawk range, and 27% of the B.C. population range is on Vancouver Island. There are approximately 165 breeding pairs on Vancouver Island. Additional information on conservation rankings of this species can be found in Table 33. Although information regarding historical distribution patterns of the Queen Charlotte goshawk is lacking, it is likely that these patterns have changed over the last 100 years due to harvesting, reduced harvest rotation length, and changes in the distribution and abundance of prey species. Recovery of this species is considered both biologically and technically feasible.
Table 33. Conservation Rankings for the Queen Charlotte Goshawk

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S2B, SZN</td>
</tr>
<tr>
<td>Alaska</td>
<td>S2</td>
</tr>
<tr>
<td>Canada</td>
<td>N2</td>
</tr>
<tr>
<td>Global</td>
<td>G5T2</td>
</tr>
</tbody>
</table>

10.5.3 Queen Charlotte Goshawk Habitat Characteristics

Quantitative information on the Queen Charlotte goshawk’s breeding home range size for Vancouver Island is lacking, and winter habitat requirements are almost completely unknown. For this reason, winter habitat will not be assessed in this report; however, nesting density and distance between adjacent active nests can be used as a proxy for breeding home range size. The breeding home range of this species encompasses a nest area, post-fledgling area, and foraging area (Figure 1).

Fig 1. Graphical representation of the hierarchical components within home ranges of Queen Charlotte goshawks (revised from Northern Goshawk Recovery Team, 2008). The location of nest trees and post-fledging areas are not necessarily centered within foraging areas.

Nest areas contain several alternative nest trees, roost trees, and plucking posts, and vary in size and shape depending on topography and habitat suitability. In general, they are approximately 200 ha, which is comparable with the post-fledgling area size when multiple
PFA`s around alternative nesting sites and some buffering around edges is included. The post-fledgling area includes the active nest site and corresponds with the adult female core use area. Because the nest area has a similar biological role as the post fledgling area, the two areas can be considered the same for management purposes. The foraging area may or may not include the post-fledgling area and nest area; it has been suggested that males do not hunt in these areas to conserve food for the female and fledglings. Foraging areas vary in size depending on the experience of the individual, her hunting efficiency, brood size, and the availability of food. Thus, there is no defined foraging area patch size, but breeding home range size may be used as a proxy.

Although Queen Charlotte goshawks are generally associated with mature and old-growth forests, selection of breeding habitat is based on stand structure, not age or composition *per se*. Typical stand characteristics corresponding with each habitat component of the Queen Charlotte goshawk are listed in Table 34.
Table 34. Queen Charlotte Goshawk Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Stand Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>General</strong></td>
<td>• Young, mature, and old forests&lt;br&gt;• Structural stage ≥ 5&lt;br&gt;• Elevation &lt; 900m asl&lt;br&gt;• Multi layered canopy&lt;br&gt;• Structurally diverse&lt;br&gt;• No urban areas</td>
</tr>
<tr>
<td><strong>Nest Trees</strong></td>
<td>• Western hemlock, Douglas-fir, Sitka spruce, red alder&lt;br&gt;• Multiple leaders, deformities, sometimes snags&lt;br&gt;• Large diameter trees; must support large stick nests</td>
</tr>
<tr>
<td><strong>Nest Area/ Post Fledgling Area</strong></td>
<td>• Mature and old forests&lt;br&gt;• May use forests &gt; 50-60 years&lt;br&gt;• May nest in riparian areas&lt;br&gt;• &gt; 200 m from forest edges; competitor and predator avoidance&lt;br&gt;• Stands &gt; 100 ha&lt;br&gt;• Mesoslope position, &lt; 40°&lt;br&gt;• &gt; 50 % canopy closure; fledgling predator avoidance</td>
</tr>
<tr>
<td><strong>Foraging Area</strong></td>
<td>• Generally do not forage in young forests&lt;br&gt;• Snags and CWD; important for prey abundance&lt;br&gt;• Closed canopy&lt;br&gt;• Large diameter trees&lt;br&gt;• Open understory; flight space and access to prey</td>
</tr>
<tr>
<td><strong>Winter Habitat</strong></td>
<td>• Mature and old-growth forests&lt;br&gt;• May use high elevations, sub alpine areas, estuaries, and riparian areas&lt;br&gt;• Within 10-100 km of nest areas</td>
</tr>
</tbody>
</table>

10.5.4 Important Considerations

**Prey Abundance:** The Queen Charlotte goshawk is a food-limited species. Although they appear to be opportunistic hunters, feeding on both birds and small mammals, their dominant prey during the breeding season is the red squirrel. Alterations in prey availability associated with landscape alterations are likely to have a significant effect on the Queen Charlotte goshawk. Furthermore, foraging does not necessarily occur in areas where prey is most abundant; rather, foraging occurs in areas where forest structure is the most conducive to catching prey. This situation is compounded for those populations that reside on islands, and can be explained through the examination of prey species dynamics through the lens of island
biogeography theory. As described in Section 4.1, species diversity is lower on small islands then large islands, and diversity will decrease as the distance between an island and mainland increases. This means that Vancouver Island and the Queen Charlotte Islands have a lower diversity of prey species then the mainland. Furthermore, a reduction in the annual abundance of a prey species on these islands may impact the ability of the Queen Charlotte goshawks to obtain enough food to meet their annual energy requirements.

**Breeding Dispersal:** The Queen Charlotte goshawk has a high fidelity to breeding areas. This means that habitat loss in a given home range has the potential to have long-lasting effects on breeding pairs. Furthermore, juvenile dispersal is thought to be between 4-12 km from the nest site, and large distances between suitable breeding sites are likely to reduce successful dispersal.

**Protection:** The Queen Charlotte goshawk, its nests and eggs are protected under the provincial Wildlife Act. Current wildlife management provisions under the Forest Practices act such as UWRs, OGMAs, wildlife tree retention areas, and riparian management areas are not considered likely to be large enough to provide suitable breeding habitat. (MWLAP, 2004g)

Cross References: Second-growth forest management techniques that benefit the great blue heron, marbled murrelet, “Queen Charlotte” hairy woodpecker, and “Vancouver Island” northern saw-whet owl will also benefit the Queen Charlotte goshawk.

### 10.5.5 Potential Roles of Second-growth Forest in Conservation

Ways in which second-growth forest can contribute to the conservation of the Queen Charlotte goshawk are shown in Table 35.

**Table 35. Potential Roles of Second Growth Forest in the Conservation of the Queen Charlotte Goshawk**

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Queen Charlotte Goshawk</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
</tbody>
</table>
Human-induced landscape changes have resulted in a decrease in suitable breeding site availability and juvenile dispersal. It has been suggested that suitable breeding habitat should be available every 6-8 km. Second-growth forest can be used to facilitate juvenile dispersal through the creation or maintenance of connectivity between late structural stage forest patches, particularly those patches that contain suitable nesting, post-fledgling, and foraging habitat. In areas where there are identified active breeding areas surrounded by second-growth forests, the second-growth forests can be used as buffers against three significant threats. First, buffers can help maintain interior forest microclimatic conditions. Second, buffers can reduce the susceptibility of the breeding habitat to natural and anthropogenic disturbances such as windthrow, snowpress, and forests fires. Third, buffers can increase the distance between the breeding sites and forest edges; thereby reducing predation by edge-adapted species such as raccoons, and competition from species such as the red-tailed hawk, which is better suited for hunting in edge environments. Finally, given that Queen Charlotte goshawk habitat preferences are based on structure versus age, it is possible to employ silviculture techniques such as low density selective thinning to increase the rate of recruitment of important breeding habitat characteristics such as large diameter trees (Section 5). Furthermore, stand treatment activities might also be employed in second-growth forests adjacent to breeding sites to assist with the recovery, maintenance, and diversity of prey species populations. Stand treatment activities conducted with the Queen Charlotte goshawk stop-go short-stay perch-hunting method in mind will also supplement existing foraging habitat.

10.6 Great Blue Heron (*Ardea herodias fannini*)

10.6.1 Sources


10.6.2 Conservation Status

Populations of *Ardea Herodias fannini* are non-migratory and occur year round on the Pacific Coast, and occasionally inland to the Bulkley Valley. The highest concentration of breeding herons (>100 breeding pairs) occurs in the Georgia Depression Ecoprovince. The *fannini* subspecies in B.C. is estimated at between 4000-5000 breeding adults (COSEWIC, 2008b), with an estimated 3326 adults breeding in the Strait of Georgia and 300 breeding elsewhere on the coast. In total, 59% of all active nests in the Georgia Depression are currently protected. Little data is available on population trends on the coast; however, over the past 30 years the
population has been reported to be generally stable or declining. This subspecies is on the provincial Blue List in B.C., and is considered a species of Special Concern in Canada. Additional information on conservation rankings of this species can be found in Table 36.

Table 36. Conservation Rankings for the Great Blue Heron (subspecies *fannini*)

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S3B, S5N</td>
</tr>
<tr>
<td>Alaska</td>
<td>S4</td>
</tr>
<tr>
<td>Washington</td>
<td>?</td>
</tr>
<tr>
<td>Canada</td>
<td>N?</td>
</tr>
<tr>
<td>Global</td>
<td>G5T4</td>
</tr>
</tbody>
</table>

10.6.3 Great Blue Heron Habitat Characteristics

In British Columbia, most herons occur near sea level on the coast, or in the lowlands and valley bottoms of the interior. Great Blue Herons are prey generalists, although they primarily forage for fish. On the coast of B.C., *A. herodias fannini* is chiefly non-migratory, with most birds wintering close to breeding areas. Large colonies are associated with extensive estuarine mudflats and eelgrass beds, and colony size has been linked with available foraging area. In B.C., breeding colonies range in size from two to about 400 nests with some pairs nesting alone. Large colonies in deciduous trees or small and dispersed colonies can encompass several hectares. During the breeding season, adult herons range within about 30 km of their colonies, although most stay within 10 km. During the winter, some adults maintain small foraging territories, but little is known of how frequently alternate sites are used. Colonies are dynamic, and although some are used for many years (known up to 28 years), most colonies containing fewer than 50 nests are relocated more frequently. Furthermore, once a colony has been abandoned for more than 1 year, recolonization occurs infrequently. Typical characteristics corresponding with each habitat component of the great blue heron are listed in Table 37.
Table 37. Great Blue Heron Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Stand Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>General Habitat</strong></td>
<td>• Elevation: 0-1100m asl&lt;br&gt; • Roost trees&lt;br&gt; • Ground barriers that exclude mammalian predators (e.g. Ditches)</td>
</tr>
<tr>
<td><strong>Foraging Habitat</strong></td>
<td>• Shallow water; tidal mudflats, riverbanks, lakeshores, and wetlands&lt;br&gt; • Fallow agricultural fields and other vegetated areas</td>
</tr>
<tr>
<td><strong>Breeding Habitat</strong></td>
<td>• Structural Stages: Young forest, mature forest, old forest&lt;br&gt; • Abundant and accessible prey within 10km&lt;br&gt; • Contiguous forest around nest trees&lt;br&gt; • Common nest trees: red alder (<em>Alnus rubra</em>), black cottonwood (<em>Populus balsamifera</em>), bigleaf maple (<em>Acer macrophyllum</em>), lodgepole pine (<em>Pinus contorta</em>), Sitka spruce (<em>Picea sitchensis</em>), and Douglas-fir (<em>Pseudotsuga menziesii</em>)</td>
</tr>
</tbody>
</table>

10.6.4 Important Considerations

**Foraging:** Shallow water fish are the most important prey group for great blue herons during the breeding and non-breeding seasons. Estuaries and other low-grade intertidal habitats are prime foraging grounds for the heron. During winter on the coast, when aquatic prey are less abundant due to the reduced duration of daytime low tides, fallow agricultural fields become important foraging areas. Suitable foraging habitat has likely declined in B.C., and this decline is considered to be equally or more important than the loss of breeding habitat on the status of the herons.

**Nest Sites:** The size of great blue heron populations has been correlated with the area of locally available foraging habitat. Thus, it is paramount that sufficient nesting habitat is maintained near important feeding areas; especially since suitable nesting habitat has declined in B.C. over the past century, and continues to decline, due to increases in human populations and industry, especially in southern coastal areas including Vancouver Island. Since herons frequently relocate colonies, it is also important that alternate forest nest sites are available.

**Population Threats:** Direct threats to the great blue heron populations in B.C. include disturbance and mortality from humans and predators, food supply limitations, contamination
of food supplies, and weather. Bald eagle (*Haliaeetos leucocephalus*) depredation and human disturbance have been highlighted as the most important threats because of their impact on breeding productivity.

**Habitat Threats:** Threats to great blue heron habitat in B.C. include the loss of breeding and foraging areas to urban development, forestry, hydroelectric power development, and natural processes. Forestry can impact heron habitat through the removal of active or potential nest trees. Furthermore, forest fragmentation may increase access to or visibility of breeding colonies for predators such as Bald Eagles, thereby reducing the amount of suitable breeding habitat available.

**Protection:** The great blue heron, its nests and eggs are protected under the Wildlife Act, as well as the Migratory Birds Convention Act. Two nesting colonies are currently protected within existing parks or wildlife management areas on Vancouver Island and the Gulf Islands; a total of 59% of all active nests in the Georgia Depression are within currently protected areas.

**Cross References:** Second-growth forest management options that benefit the marbled murrelet, “Queen Charlotte” goshawk, and “Vancouver Island” northern pygmy-owl will also benefit the great blue heron.

### 10.6.5 Potential Roles of Second-growth Forest in Conservation

Ways in which second-growth forest can contribute to the conservation of the great blue heron are shown in Table 38.

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Great Blue Heron</td>
<td></td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
</tbody>
</table>

Under the results-based code, some critical foraging and nesting habitat on Crown land could be protected through the establishment of Old-growth Management Areas, Riparian Management Areas, wildlife tree retention areas, and ‘wildlife habitat feature’ designation of
known nest sites. WHAs should be established at nesting areas and nesting colonies. WHAs will typically be 80 ha but will ultimately depend on site-specific factors such as the number of individuals using each location for breeding or foraging, and intensity of use. Other important factors include location, topography, proximity to foraging sites, relative isolation, and degree of habituation to disturbance. The core area is typically 12 ha and should ideally encompass known nest sites and potential nest areas, and, where appropriate, foraging areas and flight paths. The boundary of the core area should be no closer than approximately 200 m from the colony or ‘wildlife habitat feature’. Beyond this, where second-growth forests exist around known nest sites, 300 m buffers might be employed as management areas to minimize disturbance and maintain the integrity of nesting habitat. In management zones that have few trees other than nest trees, restocking or silviculture techniques can be used to enhance rapid development and protection of the stand. Where possible, the trees and shrubs that are planted should be a mixture of conifers and deciduous trees, and half should be of the same species currently used for nesting in that area.

10.7 Sandhill Crane (*Grus canadensis*)

10.7.1 Sources

B.C. Ministry of Water, Land and Air Protection, 2004i.

10.7.2 Conservation Status

The sandhill crane has a widespread breeding distribution in B.C., although the breeding distribution of the three subspecies within B.C. is not well understood. The three subspecies within B.C. include the lesser (*G. canadensis canadensis*), Canadian (*G. canadensis rowani*), and greater (*G. canadensis tabida*). The lesser sandhill crane is a common migrant through B.C.; the greater sandhill crane is thought to breed in the Lower Mainland, the Queen Charlotte Islands, Vancouver Island, the Hecate Lowlands, and the interior of the province; and the Canadian sandhill crane is thought to breed on the coast but may also breed in the central interior as well. Most breeding populations of sandhill crane are on the provincial Blue List; however, the Georgia Depression population is on the provincial Red List. The greater sandhill crane is considered Not at Risk in Canada; other subspecies have not been assessed. Additional information on conservation rankings of this species can be found in Table 39.
### Table 39. Conservation Rankings for the Sandhill Crane

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking (Georgia Depression Population)</th>
<th>Conservation Ranking (All Others)</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S1</td>
<td>S3S4B, SZN</td>
</tr>
<tr>
<td>Alaska</td>
<td>N/A</td>
<td>S5B</td>
</tr>
<tr>
<td>AB</td>
<td>N/A</td>
<td>S4B</td>
</tr>
<tr>
<td>ID</td>
<td>N/A</td>
<td>S5B, SZN</td>
</tr>
<tr>
<td>MT</td>
<td>N/A</td>
<td>S2N, S5B</td>
</tr>
<tr>
<td>NWT</td>
<td>N/A</td>
<td>S?</td>
</tr>
<tr>
<td>OR</td>
<td>N/A</td>
<td>S3B</td>
</tr>
<tr>
<td>Washington</td>
<td>N/A</td>
<td>S1B, S3N</td>
</tr>
<tr>
<td>YK</td>
<td>N/A</td>
<td>S?</td>
</tr>
<tr>
<td>Canada</td>
<td>N?</td>
<td>N5B</td>
</tr>
<tr>
<td>Global</td>
<td>G5T1Q</td>
<td>G5</td>
</tr>
</tbody>
</table>

#### 10.7.3 Sandhill Crane Habitat Characteristics

The key habitat requirements for cranes include water, nesting cover, and feeding meadows. Different structural stages associated with roosting, nesting, escape and screen behavior can be found in Table 40. Throughout most of the province, there have been few changes in habitat suitability and availability. However, in the Georgia depression, populations have declined as spreading urbanization and intensive agriculture have encroached on wetlands. In other parts of B.C., land use practices such as logging up to the edge of wetlands, draining wetlands for agriculture, and the trampling of emergent vegetation by livestock has resulted in a loss of habitat.
### Table 40. Structural Stages Used by the Sandhill Crane

<table>
<thead>
<tr>
<th>Structural Stage</th>
<th>Roosting</th>
<th>Nesting</th>
<th>Escape</th>
<th>Screen</th>
</tr>
</thead>
<tbody>
<tr>
<td>1: non-vegetated or sparsely vegetated</td>
<td>X</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2: herb</td>
<td>X</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>3a: low shrub</td>
<td>X</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>3b: tall shrub</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>4: pole/sapling</td>
<td></td>
<td></td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>5: young forest</td>
<td></td>
<td></td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>6: mature forest</td>
<td></td>
<td></td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>7: old forest</td>
<td></td>
<td></td>
<td>X</td>
<td>X</td>
</tr>
</tbody>
</table>

Some of the most important characteristics of sandhill crane habitat are an unobstructed view of surrounding areas and isolation from disturbance. Additional typical stand characteristics important to the sandhill crane can be found in Table 41.
<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Stand Characteristics</th>
</tr>
</thead>
</table>
| **Breeding Habitat** | • Structural stages 1-3b  
• Elevation: 0-1220 m asl  
• Isolated bogs, wet and dry heathland, marshes, swamps, meadows, and brackish estuaries  
• Shallow freshwater wetlands (>1 ha) surrounded by forest cover  
• Emergent vegetation for nesting and brood rearing including: sedges (*Carex* spp.), Cattail (*Typha latifolia*), bulrush (*Scirpus* spp.), Hardhack (*Spiraea douglasii*), willows (*Salix* spp), and Labrador Tea (*Ledum groenlandicum*) |
| **Foraging Habitat** | • Elevation: 0-1510 m asl  
• Shallow wetlands, marshes, swamps, fens, bogs, ponds, meadows, estuarine marshes, intertidal areas, and dry upland areas such as grasslands and agricultural fields |
| **Roosting/Staging Habitat** | • Structural stages 1-3b  
• Elevation: 0-1510 m asl  
• Level terrain  
• Shallow water (critical)  
• Shoreline devoid of vegetation or sparsely vegetated  
• Isolated location |

### 10.7.4 Important Considerations

**Diet:** Sandhill cranes are opportunistic foragers, feeding on both animal and plant foods. Although invertebrates and plants make up the majority of this species diet, other foods include crayfish, voles, mice, frogs, toads, snakes, nestling birds, bird eggs, berries, and carrion.

**Site Fidelity:** Although preliminary data in central B.C. suggests that site fidelity of breeding pairs is not strong, studies conducted on the Rocky Mountain population indicated strong site fidelity to summer and winter grounds during successive years.

**Migration Routes:** Three migration routes are known in B.C., each of which is used in spring and autumn: coastal, central Interior, and northeastern Interior. The remainder of this discussion will focus on the coastal route. Cranes migrating along the coastal route enter B.C. over the Juan de Fuca Strait and are occasionally seen in the Barkley Sound and Johnston Strait.
regions. Birds using the coastal route (approximately 3500 in number) are suspected of nesting in the coastal islands of B.C. and southeast Alaska.

**Population Trends:** B.C., population trend data are lacking. Most populations are likely stable., however, the Fraser Lowland populations have declined significantly and are endangered, and the South Okanagan populations have been extirpated.

**Protection:** The sandhill crane, its nests, and its eggs are protected under the federal Migratory Birds Convention Act and the provincial Wildlife Act. Several known nesting areas are currently protected in existing parks and wildlife management areas on the Queen Charlotte Islands.

**Cross References:** Due to similar nesting and foraging habitat needs, measures to maintain early seral stage habitats that benefit the short-eared own will benefit the sandhill crane.

10.7.5 **Potential Roles of Second-growth Forest in Conservation**

Ways in which second-growth forest can contribute to the conservation of the sandhill crane are shown in Table 42.

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sandhill Crane</td>
<td>X</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Forest buffers around nesting marshes are likely critical for relatively small (1-10 ha) wetlands. This is because forests are used as escape cover by young, and provide a buffer against disturbance. Structural stages capable of fulfilling these roles include tall shrub (3b) through to old forest (7). Although the sandhill crane has been occasionally reported as nesting in revegetated clearcuts, clearcuts are generally not considered suitable habitat alternatives to wetlands. Research needs to be conducted to determine the tolerance of sandhill cranes to logging adjacent to wetland breeding habitat, and the effectiveness of forested buffer strips in protecting this habitat. An inventory of potential breeding areas is also required.
Prioritization for WHA establishment is for the Red-Listed Georgia Depression population; WHAs should be created in areas where breeding is known to occur and yet is not protected through other provisions. WHAs will vary in size depending on site-specific factors such as the size and isolation of the wetland; in general, they should be approximately 20 ha excluding the wetland area. WHAs might also be created for primary migratory stop-over points; in this situation, WHAs will also be approximately 20 ha depending on the specific habitat conditions of the site. Within the WHA, the core area should include the entire stand of emergent vegetation around the wetland plus 50 meters. The management zone surrounding the core area will likely be 200-350 m depending on site-specific factors such as potential disturbances, existing tree density, and characteristics of the adjacent upland. The key purpose of the management zone is to maintain seclusion of the wetland and minimize disturbance.

10.8 “Vancouver Island” White-Tailed Ptarmigan (*Lagopus leucurus saxatilis*)

10.8.1 Sources


10.8.2 Conservation Status

There are five subspecies of white-tailed ptarmigan that occur in western Alaska, south and central Yukon, and mountain ranges from northern British Columbia to New Mexico. The Vancouver Island white-tailed ptarmigan is considered endemic to Vancouver Island. Historically the Vancouver Island white-tailed ptarmigan ranged from Mount Brenton to Tsitika Mountain; recent studies suggest that this subspecies still occupies most of its historic range. The total population size of this species is unknown. The Vancouver Island white-tailed ptarmigan is on the provincial Blue List in B.C., and its status in Canada has yet to be determined. Conservation rankings for the Vancouver Island white-tailed ptarmigan can be found in Table 43.
Table 43. Conservation Rankings for the White-tailed Ptarmigan

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S3</td>
</tr>
<tr>
<td>Canada</td>
<td>N3</td>
</tr>
<tr>
<td>Global</td>
<td>G5T3</td>
</tr>
</tbody>
</table>

10.8.3 “Vancouver Island” White-Tailed Ptarmigan Habitat Characteristics

Vancouver Island ptarmigan are year-round residents of a variety of alpine, subalpine, and upper montane habitats during the year. This subspecies occurs at lower elevations and uses a wider range of habitats than its mainland cousins. For the Vancouver Island subspecies, habitat elevation ranges differ between the breeding and winter seasons, and between the south, central, and north parts of the island. Some of this subspecies migrate to lower elevations during the winter, whereas others remain close to breeding areas. The distance that adult birds migrate between winter locations and breeding areas is, on average, 1.4 km in the southern portion of the island, and 2.0 km in the northern portion of the island. Although some habitat use data are available, habitat requirements are difficult to determine because of limited data on multiple sightings for individuals. Furthermore, habitat requirements may vary between south and central island populations.
Table 44. “Vancouver Island” White-Tailed Ptarmigan Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Stand Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Foraging Habitat</td>
<td>• Elevation: &gt;822 m asl</td>
</tr>
<tr>
<td></td>
<td>• Food plants include: Vaccinium, Poa, and Carex species, Empetrum nigrum, Arctostaphylos alpina, Cassiope mertensiana, Phyllodoce empetriformis, and Sedum oregonum</td>
</tr>
<tr>
<td>Breeding Habitat</td>
<td>• Elevation: &gt;822 m asl</td>
</tr>
<tr>
<td></td>
<td>• Alpine and subalpine mountains</td>
</tr>
<tr>
<td></td>
<td>• Alpine heather communities</td>
</tr>
<tr>
<td></td>
<td>• Subalpine heather communities</td>
</tr>
<tr>
<td></td>
<td>• Fir or hemlock tree islands</td>
</tr>
<tr>
<td></td>
<td>• Snowfields</td>
</tr>
<tr>
<td></td>
<td>• Exposed rocky areas/ tundra</td>
</tr>
<tr>
<td></td>
<td>• Sparse vegetation</td>
</tr>
<tr>
<td></td>
<td>• Overhead cover from trees and shrubs</td>
</tr>
<tr>
<td>Wintering Habitat</td>
<td>• Elevation: &gt;822 m asl</td>
</tr>
<tr>
<td></td>
<td>• Above and below the tree line</td>
</tr>
<tr>
<td></td>
<td>• Alpine bowls</td>
</tr>
<tr>
<td></td>
<td>• Hemlock and cedar forests (Median tree height 4m)</td>
</tr>
<tr>
<td></td>
<td>• Clearcuts</td>
</tr>
<tr>
<td></td>
<td>• Un-vegetated rocky outcrops</td>
</tr>
<tr>
<td></td>
<td>• Cliffs</td>
</tr>
<tr>
<td></td>
<td>• South, southeast, and southwest facing slopes</td>
</tr>
</tbody>
</table>

10.8.4 Important Considerations

**Site Fidelity:** The “Vancouver Island” white-tailed ptarmigan exhibits strong site fidelity to breeding territories after the first breeding season.

**Habitat Trends:** The amount of alpine habitat on Vancouver Island has remained fairly constant, although ski resort developments in the central and southern portion of the island may have impacted localized areas. Forest harvesting in the southern part of the island may have changed habitat conditions in winter and early spring.

**Population Threats:** The subspecies is vulnerable to population extinction processes because the birds exist in very low densities in patchy habitats with stochastic population dynamics and environmental conditions.

**Habitat Threats:** Vancouver Island white-tailed ptarmigan habitat has four main threats: recreation, air- and ground-based pollutants, forest harvesting, and climate change. Human
presence in the alpine can be associated with the introduction of generalist predators and exotic plant species. Regional air and water pollution is an increasing concern for high elevation species because pollutants are carried by wind from urban and industrial centers and deposited at high elevations in many areas. Logging is also a significant concern because it decreases the amount of mature forest and increases fragmentation. Removing forest cover changes microclimate conditions including wind and insolation patterns, which may influence the rate of snowmelt. Snowfields are important for cooling, providing food resources, and enabling birds to remain cryptic when their plumage is changing. Thus, fewer or smaller snowfields restrict birds to a smaller amount of snowfield habitat making them vulnerable to increased risk of predation and increasing travel distances between snowfield patches. Increased fragmentation of montane forest could result in longer seasonal migrations with predicted higher mortality. Finally, climate change has the potential to alter the amount of alpine and subalpine habitat and to increase alpine fragmentation because of rising subalpine tree lines that may accompany higher temperatures.

**Dispersal:** On Vancouver Island, chicks have dispersed up to 34 km to other mountains.

**Protection:** The white-tailed ptarmigan, its nests, and its eggs are protected under the provincial Wildlife Act. An important white-tailed ptarmigan habitat is currently protected in Strathcona Provincial Park. Additional areas to recognize the importance of the ptarmigan, but that do not provide legal protection, (Important Bird Areas) are proposed in Strathcona Provincial Park and Mount Arrowsmith Area Mountains.

**Cross References:** Due to their use of montane meadows, measures to maintain early seral stage habitat for the Vancouver Island marmot will benefit the Vancouver Island white-tailed ptarmigan as well.

**10.8.5 Potential Roles of Second-growth Forest in Conservation**

Ways in which second-growth forest can contribute to the conservation of the Vancouver Island white-tailed ptarmigan are shown in Table 45.
Table 45. Potential Roles of Second-growth Forest in the Conservation of the Vancouver Island White-tailed Ptarmigan

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vancouver Island White-tailed Ptarmigan</td>
<td>X</td>
<td>X</td>
<td>X</td>
<td></td>
</tr>
</tbody>
</table>

Given that white-tailed ptarmigan use montane forest, some white-tailed ptarmigan habitat may be conserved where wildlife tree retention areas or old-growth management areas are implemented under the results-based code. Outside of these provisions, WHAs should be established in upper montane areas where concentrations of white-tailed ptarmigan are known to occur regularly during the winter. WHAs will typically be between 1 and 7 ha, however, this will depend on site-specific factors such as the maintenance of wind firmness, and the maintenance of microclimatic conditions such as low local temperatures and local wind patterns. WHAs should be 50–250 m wide on southerly aspects (S, SE, SW) and 25–250 m on northerly aspects (N, NE, NW), and a minimum length of 250 m. The WHA should include upper montane forest that will create a continuous zone around the adjacent subalpine and alpine habitat to provide cover, buffer and maintain microclimatic conditions suitable for retaining snowfields, and allow access, or connectivity, to lower elevations.

10.9 Red-Legged Frog (*Rana aurora aurora*)

10.9.1 Sources


10.9.2 Conservation Status

The global distribution of the red-Legged frog includes the coastal lowlands of southwestern British Columbia, Washington, Oregon, and northern California. Within B.C., this species occurs on the mainland west of the Coast Mountains in the Fraser Valley, and adjacent to the Strait of
Georgia, as well as on Vancouver Island and the Gulf Islands. The red-legged frog is Blue-Listed in B.C., and a species of Special Concern in Canada. Conservation rankings for the red-legged frog can be found in Table 46.

Table 46. Conservation Rankings for the Red-legged Frog

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S3S4</td>
</tr>
<tr>
<td>Washington</td>
<td>S5?</td>
</tr>
<tr>
<td>Oregon</td>
<td>S3</td>
</tr>
<tr>
<td>California</td>
<td>S2</td>
</tr>
<tr>
<td>Global</td>
<td>G4</td>
</tr>
</tbody>
</table>

10.9.3 Red Legged Frog Habitat Characteristics

Red-legged frogs require standing water to breed. Known breeding sites exhibit large variation in size, water depth, degree of permanency, and community structure. The highest occurrences of the red-legged frog have been found in bogs and fens. The terrestrial environment surrounding occupied wetlands is of consequence because this is where up to 90% of feeding and growth occurs. At the stand level, the presence of this species does not appear to be associated with the age of the forest surrounding the wetland; however, red-legged frogs appear to be negatively affected by clear cutting and non-forested successional stages. Results from studies conducted in disturbed landscapes suggest that proximity to water, including streams, appears to be an important determinant of terrestrial red-legged frog distribution. However, what constitutes high versus low quality terrestrial habitat remains unknown. The spatial distribution of red-legged frogs in terrestrial landscapes is likely related to proximity to suitable breeding habitat rather than forest age. Typical characteristics corresponding with each habitat component of the red-legged frog are listed in Table 47.
### Table 47. Red-Legged Frog Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Stand Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Aquatic Breeding Habitat</strong></td>
<td>• Structural complexity; High percent cover of thin stemmed emergent vegetation and coarse woody debris</td>
</tr>
<tr>
<td></td>
<td>• Low water flow</td>
</tr>
<tr>
<td></td>
<td>• Humus substrate</td>
</tr>
<tr>
<td></td>
<td>• Vegetation cover surrounding wetland</td>
</tr>
<tr>
<td></td>
<td>• No vertebrate predators such as fish and bullfrogs</td>
</tr>
<tr>
<td></td>
<td>• &lt;850 m asl</td>
</tr>
<tr>
<td></td>
<td>• Small wetlands &lt;0.5 ha</td>
</tr>
<tr>
<td></td>
<td>• Water retention until Aug. 31st</td>
</tr>
<tr>
<td><strong>Adult Terrestrial Foraging Habitat</strong></td>
<td>• &lt;850 m asl</td>
</tr>
<tr>
<td></td>
<td>• Low slope</td>
</tr>
<tr>
<td></td>
<td>• A deciduous element (USA)</td>
</tr>
<tr>
<td></td>
<td>• Coarse woody debris for cover</td>
</tr>
</tbody>
</table>

#### 10.9.4 Important Considerations

**Population Declines:** Although there is currently no information on population trends for red-legged frog populations in B.C., populations in the Willamette Valley in Oregon have declined drastically, and populations of the California subspecies (*R. aurora draytoni*) have declined severely throughout their range.

**Exotic Species:** The introduction of exotic species has been suggested as one of the reasons for the decline of red-legged frogs in western North America; however, the response of red-legged Frog populations to introduced predators including bullfrogs and fish are not predictable. Although red-legged frog tadpoles have variable survival in the presence of bullfrog larvae, metamorphosing frog survival is <5% in the presence of adult bullfrogs. This implies that bullfrog induced terrestrial mortality may have a considerable effect on the number of successful metamorphs leaving a wetland. Furthermore, recent studies have shown that the non-native North American bullfrog populations are commonly vectors of the chytrid fungus *Batrachochytrium dendrobatidis*, which causes a disease called chytridiomycosis that has been linked to amphibian declines around the world.

**Habitat Loss and Degradation:** Habitat loss and degradation have been suggested as the primary causes of ranid declines. Watershed urbanization and subsequent increasing water-
level fluctuations within wetland habitats, and likely agricultural and urban runoff, can have significant negative impacts on red-legged frog populations. Although the specific effects of landscape fragmentation on this species are unknown, structural components of development, such as roads, can have both direct population impacts when they cross important dispersal routes, and indirect impacts through habitat alteration. Furthermore, forest harvesting has been shown to affect wetland variables such as productivity, hydrology, species assemblage, and habitat availability. Notably, the degree to which wetland functions are altered depends on a number of factors such as the size of the wetland and the type of forest harvesting and management practices used. For the red-legged frog, it appears that one the most significant forest harvesting impacts is the reduction of canopy cover. This is important because removing forest canopy increases the rate of evaporation from wetlands, which can strand eggs when a shoreline recedes too early in the spring, and/or strand larvae if the wetland dries up before the tadpoles have a chance to metamorphose.

**Metapopulation Dynamics:** At the landscape scale, the loss of small wetlands can affect metapopulation dynamics of pond-breeding amphibians and increase the probability of extinction of populations in the remaining wetlands. Small, unclassified wetlands tend to be numerically dominant to large wetlands. The loss of small wetlands not only decreases the number of breeding sites and thus the density of these amphibians, it also increases the nearest neighbour distance between sites, which impedes source sink processes. Although little is known about the metapopulation dynamics of red-legged frogs, studies on other ranids suggest that nearby population sources are important in maintaining metapopulations of pond-breeding amphibians.

**Protection:** The red-legged frog is protected under the *Wildlife Act* and cannot be killed, collected, or held in captivity.

**Cross References:** Measures to maintain forest connectivity between wetlands for the red-legged frog will also benefit the “Vancouver Island” common water shrew and Keen’s long-eared myotis.

10.9.5 Potential Roles of Second-growth Forest in Conservation

Ways in which second-growth forest can contribute to the conservation of the red-legged frog are shown in Table 48.
Under current legislation, ephemeral wetlands and wetlands <0.5 ha in area receive no protection. Unclassified wetlands that are known breeding sites and the terrestrial riparian areas surrounding these wetlands should be protected. This would serve four purposes: maintenance of canopy cover, maintenance of adult foraging ground, maintenance of suitable microclimatic conditions for emerging juveniles, and buffering of the wetland from surrounding land use changes. It has been suggested that the terrestrial component should consist of a 30 m reserve of adjacent riparian habitat beyond the high water mark. The most critical component for terrestrial habitat is likely sufficient cover; provided that forest succession surrounding the wetland has resulted in a structural stage of 3b Tall Shrub or higher, there should be sufficient cover for the red-legged frog. In addition, red-legged frog presence has been correlated with high amounts of coarse woody debris, indicating that this habitat element may be important for cover. Providing there is no negative effect on percent canopy cover, silviculture practices that increase the recruitment of this habitat element might be employed. Furthermore, given the importance of metapopulation dynamics, wetlands that are surrounded by terrestrial structural stages of less than 3b Tall Shrub might be considered for protection if they are in close proximity to known breeding ponds and there are no known impediments to suitable terrestrial habitat recruitment such as repeated disturbance. In addition, second-growth forests between these wetlands should be protected to provide connectivity for juvenile dispersal, provided that the distance between the wetlands does not exceed 1 km. The creation of networks of interconnected wetlands will increase the connectivity and dispersal of juvenile frogs, possibly maintain metapopulation dynamics, buffer against temporal variation in productivity of individual wetlands, and buffer against stochastic events that may change population source and sink dynamics.

10.10 Quatsino Cave Amphipod (*Stygobromus quatsinensis*)

10.10.1 Sources

B.C. Ministry of Water, Land and Air Protection, 2004I.
10.10.2 Conservation Status

The Quatsino Cave Amphipod has only been found in subterranean karstic waters of coastal northwest North America from Vancouver Island to southeastern Alaska. Within British Columbia, the only known locations of this species are in the limestone caves in the Quatsino Formation on Vancouver Island. Within these caves there are between 10-20 known occurrences; however, each occurrence with one exception, represents only one or two individuals. At the exceptional site, 11 individuals were observed. It has been suggested that recent occurrences may represent relict populations stranded after the recession of ice sheets in association with the end of the Wisconsin Glaciation. Table 49 lists the conservation status of this species for all relevant jurisdictions.

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S2S3</td>
</tr>
<tr>
<td>Alaska</td>
<td>S?</td>
</tr>
<tr>
<td>Canada</td>
<td>N3</td>
</tr>
<tr>
<td>Global</td>
<td>G3</td>
</tr>
</tbody>
</table>

10.10.3 Quatsino Cave Amphipod Habitat Characteristics

Cave habitats typically have low levels of productivity and sparse food sources. Although the diet and foraging behavior of this species is unknown, it is thought to be similar to other *Stygobromus* amphipods, which are detritivores. Potential food sources include bacteria, microfungi, organic particles on ingested sediments, and animals that wash into cave pools. Typical habitat characteristics of the Quatsino cave amphipod can be found in Table 50.
Table 50. Quatsino Cave Amphipod Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Characteristics</th>
</tr>
</thead>
</table>
| Cave Habitat        | • Karst caves and interstitial waters  
|                     | • Shallow, mud-bottomed pools  
|                     | • Underground stream gravel  |
| Adjacent Forest     | • Highest densities below mature and old-growth forests  
|                     | • Has been found below second-growth forests  |

10.10.4 Important Considerations

**Knowledge Gaps:** The biology of this species is unstudied. It is possible that it occurs in the hyporheic habitats of surface streams. Additional localities for this species may exist in karst in remote areas on the mainland, on the Queen Charlotte Islands, and other island exposures of the Quatsino Formation limestone.

**Dispersal:** Movement of subterranean aquatic species occurs through small, continuous water filled cracks and fissures in suitable bedrock. The small size of these amphipods in conjunction with the discontinuous nature of carbonate bedrock likely limits the current dispersal of this species. This is significant because a restricted distribution and limited dispersal ability may translate into a lack of genetic exchange and may result in increased risk of extirpation.

**Protection:** This species is not legally protected under any Acts, but several known localities are protected within provincial parks (e.g., Weymer Creek Karst, Horne Lake Caves).

**Cross References:** Forest management practices to prevent removal of critical forest habitat that protect the Quatsino cave amphipod’s habitat are likely to benefit the Keen’s long-eared myotis, but there are no second-growth management practices that would be particularly likely to benefit other species as well as this one.

10.10.5 Potential Roles of Second-growth Forest in Conservation

This species is threatened by habitat alterations as a result from forest harvesting and road construction in the forests adjacent to the caves. Ways in which second-growth forest might be used to conserve the Quatsino cave amphipod are shown in Table 51.
Table 51. Potential Roles of Second-growth Forest in the Conservation of the Quatsino Cave Amphipod

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment/Restoration</th>
</tr>
</thead>
<tbody>
<tr>
<td>Quatsino Cave Amphipod</td>
<td></td>
<td>X</td>
<td></td>
<td>X</td>
</tr>
</tbody>
</table>

Surface activity such as forest harvesting and road construction can negatively affect Quatsino Cave Amphipod habitat, particularly water quality, through infilling from logging debris, changes in surface hydrological conditions, increased soil erosion, and the shattering of cave roofs. Adherence to the results-based code best management practices for karst features, particularly recommendations for buffers around swallets and harvesting restrictions that minimize soil loss and infilling of epikarst, may provide sufficient protection at sites within the timber harvesting land base. A minimum 100 m radius buffer is recommended around the point where streams go underground. Furthermore, in areas where infilling and smothering by suspended sediment is an issue, silviculture treatments (Section 5.2) in adjacent second-growth forests that open up the canopy to increase understory vegetation may be employed to help control surface runoff and siltation.

10.11 Vancouver Island Marmot (*Marmota vancouverensis*)

10.11.1 Sources

B.C. Ministry of Water, Land and Air Protection, 2004m.
Vancouver Island Marmot Recovery Team, 2008.
COSEWIC, 2008c.

10.11.2 Conservation Status

The Vancouver Island marmot is endemic to Vancouver Island, and is one of the rarest mammals in North America. Recent decades have seen a drastic reduction in the area of occupied range and population sizes. With one known exception, the remaining marmot colonies occur in south-central Vancouver Island at the headwaters of the adjacent drainages of the Nanaimo, Chemainus, Nitinat, Cameron, and Cowichan rivers. The other small, isolated
colony occurs on Mount Washington in east-central Vancouver Island. As of 2007, there were approximately 50 wild-born individuals in the wild, distributed among five mountains, as well as a few dozen released marmots and approximately 100 individuals in captivity (COSEWIC, 2008c). The conservation rankings for this species are presented in Table 52.

Table 52. Conservation Rankings for the Vancouver Island Marmot

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S1</td>
</tr>
<tr>
<td>Canada</td>
<td>N1</td>
</tr>
</tbody>
</table>

10.11.3 **Vancouver Island Marmot Habitat Characteristics**

Habitat scarcity is the fundamental reason for the rarity of the Vancouver Island marmot. The Vancouver Island marmot exhibits high site fidelity, using the same natal, escape, and hibernation burrows and adjacent meadows and cliffs from year to year.

Vancouver Island marmots require three essential habitat features: grasses and forbs to eat; suitable soil for constructing overnight and overwintering burrows; and microclimatic conditions that permit summer foraging, thermoregulation, and successful hibernation. More detailed habitat characteristics can be found in Table 53.
Table 53. Vancouver Island Marmot Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Characteristics</th>
</tr>
</thead>
</table>
| **Subalpine Meadows-Natural Habitat** | • Natural meadows maintained by avalanches, snow-creep, and potentially wildfire  
• Vegetation structural stages: non-vegetated/ sparse, herb, low shrub  
• Colluvial soil  
• Spring food plants: oatgrass (*Danthonia intermedia*), *sedges* (*Carex* spp.) and spreading phlox (*Phlox diffusa*)  
• Summer and fall food plants: forbs, especially broadleaf lupin (*Lupinus latifolius*), woolly sunflower (*Eriophyllum lanatum*), and purple peavine (*Lathyrus nevadensis*)  
• Elevation between 1040 and 1450 m asl  
• South to west facing slope |
| **Subalpine Created Habitat**       | • Meadows created through clearcuts, ski-run development, and mine tailings  
• Within 1km of natural colonies  
• Vegetation structural stages: non-vegetated/ sparse, herb, low shrub  
• Colluvial soil  
• Food plants: grasses, *Anaphalis margaritacea*, *Fragaria* spp., *Epilobium angustifolium*, and *Lupinus latifolius*.  
• Elevation between 730-1140 m asl  
• South to west facing slope |

10.11.4 Important Considerations

**Home Range:** The home range of individual adult marmots within a colony has been documented as several hectares. Marmots commonly shift their areas of use between early spring/ late fall and summer. In natural colonies, several habitat patches may be used.

**Climate Change:** Re discussion with Dan Smith from tree lab, also talk with Eric Higgs re tree invasion in subalpine meadows

**Clearcuts:** In clearcuts, marmots often travel along logging roads, and daily movements range from 500-1000m. Most colonizations occur within 5 to 15 years after harvest, and maximum occupancy at logged sites is 21 years. Importantly, all known marmot colonies in clearcuts appear to have gone extinct.
**Predator/Prey Dynamics:** Most of the decline in marmot populations since the mid 1980s is due to distinct “episodes” of high mortality in a particular colony. The spatial and temporal pattern of these episodes is consistent with a predation and disease hypothesis. Circumstantial evidence suggests that predation pressure on marmot populations has increased, particularly in clearcuts.

**Protection:** The Vancouver Island marmot is protected under the Wildlife Act. One marmot habitat area is protected under the B.C. Ecological Reserves Act (Haley Lake Ecological Reserve) and one is protected under the B.C. Wildlife Act (Green Mountain Wildlife Critical Habitat Area).

**Cross References:** Due to their use of montane meadows, measures to maintain early seral stage habitat for the “Vancouver Island” white-tailed ptarmigan will benefit the Vancouver Island marmot as well.

### 10.11.5 Potential Roles of Second-growth Forest in Conservation

Ways in which second-growth forests can contribute to the conservation of the Vancouver Island marmot are listed in Table 54.

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vancouver Island Marmot</td>
<td></td>
<td></td>
<td>X</td>
<td></td>
</tr>
</tbody>
</table>

This species does not use forested habitats to meet any of its requirements, so there is no potential role for second-growth forests over approximately 20 years following harvest in the conservation of the Vancouver Island marmot. These marmots will colonize new clearcuts, and may be able to use them as supplemental habitat until a forest cover regenerates.
10.12 Marbled Murrelet (*Brachyramphus marmoratus*)

10.12.1 Sources


10.12.2 Conservation Status

The marbled murrelet’s range extends from the Aleutian Islands along the southern coast of Alaska, south to central California. The population in 2002 was estimated to be 54,700-77,700, but large parts of the range have no count data, making these numbers uncertain. The small amount of long-term data available on population trends indicates that the population in British Columbia is declining, especially in eastern Vancouver Island and the southern mainland. The conservation rankings for marbled murrelet in all relevant jurisdictions are presented in Table 55.
Table 55. Conservation Rankings for the Marbled Murrelet

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alaska</td>
<td>S2S3</td>
</tr>
<tr>
<td>British Columbia</td>
<td>S2B, S4N</td>
</tr>
<tr>
<td>California</td>
<td>S1</td>
</tr>
<tr>
<td>Oregon</td>
<td>S2</td>
</tr>
<tr>
<td>Washington</td>
<td>S3</td>
</tr>
<tr>
<td>Canada</td>
<td>N2</td>
</tr>
<tr>
<td>Global</td>
<td>G3G4</td>
</tr>
</tbody>
</table>

10.12.3 Marbled Murrelet Habitat Characteristics

Marbled murrelets forage at sea, and are fast fliers; able to commute over 100 km to feed at prey concentrations. Still, they are not very maneuverable; difficulty landing and taking off affects their choice of nest sites. They show diurnal and seasonal movements among foraging sites, and, though they are not colonial, usually aggregate at favoured sites. Suitable nesting habitat for the marbled murrelet is late seral stage coniferous forest containing large trees with suitable platforms accessible through a variable canopy structure; studies have shown significant positive correlations between the number of marbled murrelet in a watershed and the area of large-tree late seral habitat available. Population size seems to be more correlated with the total area and quality of nesting habitat available than the sizes and shapes of the remaining habitat patches.

Some nests have been found in different habitats including cliffs, deciduous trees, and isolated veterans in low productivity areas, but these habitats are less likely to be inhabited. Most of the 200 nests found in British Columbia to date have been located within 30-50 km of marine capture sites in old conifers scattered throughout suitable habitat: predominantly yellow-cedar, western hemlock, Sitka spruce, Douglas-fir, and western redcedar trees. Trees on steep slopes are preferable because they enable easy access to tree canopies and reduce the risk of nest predation by terrestrial predators.
A portion of the marbled murrelet population remains near the breeding grounds throughout the winter, but many of the adults and newly fledged juveniles migrate from breeding areas on the outer west coast to more sheltered wintering areas in the Strait of Georgia and Puget Sound at the end of the breeding season.

An accurate assessment of the amount of nesting habitat that has been lost due to logging is not available, but preliminary reports suggest a 35-49% loss of habitat since 2000. Large declines in suitable habitats were evident in Port Alberni, Campbell River, Duncan, Port McNeill, and the Sunshine Coast. The most notable loss is in southeast Vancouver Island and the southern mainland. More detailed habitat characteristics can be found in Table 56.

Table 56. Marbled Murrelet Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Foraging</strong></td>
<td>• Shallow near shore and sheltered waters known to support concentrations of prey schools.</td>
</tr>
<tr>
<td><strong>Wintering</strong></td>
<td>• Shallow near shore and sheltered waters.</td>
</tr>
</tbody>
</table>
| **Nesting**        | • Late seral stage coniferous forest.  
|                    | • Large trees with suitable horizontal platforms made with limbs >15 cm diameter to provide nest site and landing pad.  
|                    | • Trees with sufficient height (> 40 m) to allow stall-landings and jump-off departures.  
|                    | • Nest trees with suitable branch density and crown characteristics to provide shelter and reduce detection by predators.  
|                    | • Openings in canopy for flight access.  
|                    | • Soft substrate (moss, epiphytes, duff and leaf litter) to provide nest cup. |
| **Elevation**      | • 0-900 m asl preferred nesting habitat.  
|                    | • 0-600 m asl in watersheds with more intact old stands. |

10.12.4 Important Considerations

**Habitat Threats:** Loss of nesting habitat is a major threat causing reduced recruitment throughout the species’ range. Murrelets do not increase nest densities in remaining old forest patches in response to less available habitat; radar studies in Clayoquot Sound showed reduced populations in watersheds subject to intensive logging. The results of field studies on the effect of small forest patches, forest edges, and fragmentation of habitat on nesting murrelets are unclear and contradictory, but murrelet population size seems to be more correlated with the
total area and quality of nesting habitat available than the sizes and shapes of the habitat patches.

**Climate Change:** Altered climate might affect nesting murrelets through thermal stress and reducing the availability of epiphyte mats used as nest substrate. Thermal stress effects are most likely to occur in areas bordered by clearcuts or young regenerating forests.

**Predators:** Human activity in marbled murrelet habitat increases predation risk, as nest predators such as Stellar’s jays and ravens favour forest edges bordering clearcuts and roads.

**Protection:** Marbled Murrelets and their nests and eggs are protected under Canadian Migratory Birds Convention Act, the provincial Wildlife Act, and the Species at Risk Act. Marbled murrelet habitat is currently protected within the Carmanah-Walbran Provincial Park, Pacific Rim National Park, Strathcona Provincial Park, Gwaii Haanas National Park Reserve, and coastal protected areas in Clayoquot Sound.

**Cross References:** Second-growth forest management practices to retain snags and recruit wildlife trees that benefit the great blue heron, Keen’s long-eared myotis, “Queen Charlotte” goshawk, and “Queen Charlotte” northern saw-whet owl will also likely benefit the marbled murrelet.

### 10.12.5 Potential Roles of Second-growth Forest in Conservation

This species is threatened by loss of nesting habitat and habitat fragmentation as a result of forest harvesting and road construction in coastal forests. The creation of forest edge habitat in clearcuts and along roads also increases the density of predators that favour edge habitats, potentially increasing predation risk on eggs and nestlings.

Ways in which second-growth forest might be used to conserve the marbled murrelet are shown in Table 57.
Table 57. Potential Roles of Second-growth Forest in the Conservation of the Marbled Murrelet

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment/Restoration</th>
</tr>
</thead>
<tbody>
<tr>
<td>Marbled Murrelet</td>
<td>X</td>
<td></td>
<td></td>
<td>X</td>
</tr>
</tbody>
</table>

The marbled murrelet is not likely to use second-growth forests until they reach late successional stages and trees have achieved the heights, branch characteristics, and epiphyte biomass loadings required for nesting. Partial cut harvesting and thinning to maintain or promote large trees may enhance the restoration of marbled murrelet habitat attributes in second-growth forests. Second-growth forests may also potentially provide dispersal corridors between epiphyte (plant, moss, and lichen) population sources in old forests and maturing forests.

10.13 Ancient Murrelet (*Synthliboramphus antiquus*)

10.13.1 Sources


10.13.2 Conservation Status

The range of the ancient murrelet spans an arc around the rim of the northern Pacific Ocean, and their breeding colonies and at-sea distribution are found on offshore islands south to the Haida Gwaii. Approximately one-half of the birds in this region nest at three large colonies off the northwest side of Graham Island and another 44% breed at 17 colonies off the coast of Moresby Island. The breeding population is approximately 256,000 pairs; at best guess, the B.C. population represents one half of the world population. Estimates from colonies without introduced predators indicate breeding populations have increased by 0.2-9.5% per year, while colonies with introduced predators have decreased by 1-23% per year. The conservation rankings for the ancient murrelet in all relevant jurisdictions are presented in Table 58.
Table 58. Conservation Rankings for the Ancient Murrelet

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alaska</td>
<td>S4</td>
</tr>
<tr>
<td>British Columbia</td>
<td>S2S3B, S4N</td>
</tr>
<tr>
<td>California</td>
<td>S?</td>
</tr>
<tr>
<td>Oregon</td>
<td>SZN</td>
</tr>
<tr>
<td>Washington</td>
<td>S3S4N</td>
</tr>
<tr>
<td>Canada</td>
<td>N3</td>
</tr>
<tr>
<td>Global</td>
<td>G4</td>
</tr>
</tbody>
</table>

10.13.3 Ancient Murrelet Habitat Characteristics

The ancient murrelet requires islands without alien mammalian predators or human disturbances to nest in, and marine areas with no night lighting or gill fishing to feed in. They nest in colonial burrows dug into the ground beneath Sitka spruce or western hemlock on seaward slopes or flats. Two types of site fidelity are practiced: fidelity to natal colony and fidelity to nest site. Ancient murrelets may desert their nest if it is disturbed, and pairs are more likely to return the following year to a burrow where they bred successfully, than a burrow where they previously deserted their eggs. Family groups immediately leave the waters around the breeding colony once the chicks have joined their parents at sea.

Ancient murrelet populations have declined due to the introduction of mammals including rats, raccoons, and foxes on nesting islands.

More detailed habitat characteristics can be found in Table 59.
Table 59. Ancient Murrelet Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Characteristics</th>
</tr>
</thead>
</table>
| Nesting            | • Low, flat forested islands offshore from main islands.  
|                    | • No mammalian predators or human disturbances.  
|                    | • Trees, stumps, or fallen logs within 300 m of the ocean to burrow under. |
| Marine             | • Continental shelf and slope in waters with sea surface temp between 4 and 20 °C.  
|                    | • Sheltered water adjacent to breeding colonies to gather at dusk and dawn.  
|                    | • Areas of tidal upwelling above the continental shelf where prey concentrate in winter. |

10.13.4 Important Considerations

**Predation:** Presence of introduced mammalian predators on colony islands has rendered those islands unsuitable for nesting.

**Climation Change:** Changes in the timing, abundance and locations of prey concentrations due to warming ocean temperatures may affect ancient murrelet populations, and rising ocean levels may decrease the area of available nesting sites.

**Habitat Degradation:** Exploitation of ocean resources through commercial fishing, human recreation such as camping and boating near nesting grounds, oil pollution/oil spills in marine environments, and forest harvesting at breeding colonies have all contributed to the degradation of nesting sites and feeding grounds.

**Protection:** The ancient murrelet, its nests, and eggs are all protected from hunting and collecting in Canada. Only members of the Haida nation can still legally hunt the birds for subsistence purposes. In B.C., 16 of the 31 active nesting colonies are protected.

**Cross References:** The retention or recruitment of wildlife trees on offshore islands that may benefit the Keen’s long-eared myotis, “Queen Charlotte” northern saw-whet owl, and “Queen Charlotte” hairy woodpecker may also benefit the ancient murrelet.
10.13.5 Potential Roles of Second-growth Forest in Conservation

This species is threatened primarily through increased predation on nests by introduced mammalian predators on nesting islands (e.g. rats). Forest harvesting and human recreation has also contributed to the degradation of nesting habitat. Ways in which second-growth forest might be used to conserve the ancient murrelet are shown in Table 60.

Table 60. Potential Roles of Second-growth Forest in the Conservation of the Ancient Murrelet

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment/Restoration</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ancient Murrelet</td>
<td></td>
<td>X</td>
<td></td>
<td>X</td>
</tr>
</tbody>
</table>

The ancient murrelet burrows under tree roots, stumps, and fallen logs present within 300 m of shore on flat, offshore islands in the Haida Gwaii. Ancient murrelet nesting habitat can be maintained by the retention of large trees, logs, and stumps within 300 m of shoreline on suitable islands during logging operations, or by the placement of logs and stumps in second-growth forests.

Second-growth forests may also be used to provide a buffer between ancient murrelet nesting sites and human recreational sites.

10.14 “Queen Charlotte” Northern Saw-whet Owl (*Aegolius acadicus brooksi*)

10.14.1 Sources


10.14.2 Conservation Status

The Queen Charlotte northern saw-whet owl subspecies is Blue Listed in British Columbia, and its status in Canada has not been determined. This species is a non-migratory resident on the Queen Charlotte Islands and there is no record of *A. acadicus brooksi* from the mainland. The
only population data available suggests that there may be 2775 males keeping territories in the Queen Charlotte Islands; this assumes that all habitats are of equal quality and saturated, that the species is restricted to coastal western hemlock habitats, and that habitat coverage was representative of the entire Queen Charlotte Islands in both surveys conducted to date. The conservation rankings for this species in all relevant jurisdictions are presented in Table 61.

Table 61. Conservation Rankings for the “Queen Charlotte” Northern Saw-Whet Owl

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S3</td>
</tr>
<tr>
<td>Canada</td>
<td>N3</td>
</tr>
<tr>
<td>Global</td>
<td>G5T3</td>
</tr>
</tbody>
</table>

10.14.3   “Queen Charlotte” Northern Saw-Whet Owl Habitat Characteristics

The northern saw-whet owl is a secondary cavity-nester that uses old woodpecker nest sites in either coniferous or deciduous wildlife trees, though they will use suitable nest boxes when available. Because of a lack of woodpeckers on the Queen Charlotte Islands, this subspecies may only use natural cavities in old trees and snags; though cavities excavated and made bigger by smaller woodpeckers and northern flickers may also be suitable. Continued presence of owls at nest sites and other evidence from the mainland subspecies suggest reuse of nesting areas, though it is not known whether the Queen Charlotte subspecies reuse nest sites. Home ranges for breeding males on the Queen Charlotte Islands are likely similar to those in the Okanagan Valley; ranging from 125 to 150 ha.

Queen Charlotte northern saw-whet owls feed almost exclusively on small mammals such as deer mice, but have been known to eat small birds, insects, and intertidal invertebrates. Thus, they require a mixture of terrestrial forested habitats that support a diversity of prey items, and are able to take advantage of some shore habitats as well.

More detailed habitat characteristics can be found in Table 62.
Table 62. “Queen Charlotte” Northern Saw-Whet Owl Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Characteristics</th>
</tr>
</thead>
</table>
| **Nesting**        | • Coniferous forests with mixtures of early seral and mature forest habitat types within the home range.  
                     • Proximity to riparian habitat may be important.  
                     • Known trees for nesting include coniferous and deciduous: particularly western hemlock, Sitka spruce, lodgepole pine, yellow-cedar, western redcedar, and red alder.  
                     • Trees should be > 83 cm dbh or larger, with visible woodpecker or natural cavities and evidence of heart rot (decay class 5)  
                     • Known nest trees were 15 and 28 m in height.  
                     • Stand nest trees were > 83 cm dbh or larger, with visible woodpecker or natural cavities and evidence of heart rot (decay class 5)  
                     • Known daytime roost sites located in the upper third section of the canopy of large western hemlock trees within old-growth forest stands.  
                     • Mixtures of tree sizes for singing: trees with larger height and diameter than surrounding trees, free of surrounding vegetation. |
| **Foraging**       | • Uneven-aged forest structure with openings  
                     • May be able to use small clearcuts, but not young forests with dense regrowth.  
                     • Edge habitat in forest openings.  
                     • Intertidal zones. |
| **Roosting**       | • Known daytime roost sites located in the upper third section of the canopy of large western hemlock trees within old-growth forest stands.  
                     • Mixtures of tree sizes for singing: trees with larger height and diameter than surrounding trees, free of surrounding vegetation. |

10.14.4 Important Considerations

**Habitat Loss and Degradation:** The primary threat to the Queen Charlotte northern saw-whet owl is the loss and degradation of breeding and foraging habitat through logging. Forest harvesting and the creation of even-aged plantations reduces the amount of uneven-aged stand characteristics, number of old trees with cavities available for nesting, and the number of forest openings for hunting. Suitable habitat is likely declining between 2-4% per year, but the actual rate of decline is likely in the lower end of the range due to the creation of new suitable habitats in second-growth stands.

**Limited Distribution:** The risk of extinction for the Queen Charlotte northern saw-whet owl is also increased to due its small range, which makes it susceptible to stochastic events as well as changes in limited habitat.
**Cross References:** This species shares habitat characteristics with the “Queen Charlotte” hairy woodpecker and wildlife tree recruitment strategies in second-growth forests may benefit both of these species.

### 10.14.5 Potential Roles of Second Growth Forest in Conservation

This species is primarily threatened by habitat loss and degradation through logging. Ways in which second-growth forest might be used to conserve the Queen Charlotte northern saw-whet owl are shown in Table 63.

Table 63. Potential Roles of Second-growth Forest in the Conservation of the “Queen Charlotte” Northern Saw-whet Owl.

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment/Restoration</th>
</tr>
</thead>
<tbody>
<tr>
<td>“Queen Charlotte” Northern Saw-whet Owl</td>
<td></td>
<td></td>
<td>X</td>
<td>X</td>
</tr>
</tbody>
</table>

The northern saw-whet owl hunts in forests with mixed seral stage patches and clearings, so second-growth forests can be used as foraging habitat for this owl at early seral stages, before the dense regrowth of trees and understory vegetation. Thinning and brushing techniques may enhance foraging habitat during forest regeneration. Management practices that increase the retention of large trees or thinning practices that promote uneven tree size distributions can also provide roost trees (which are usually large trees surrounded by smaller trees) and provide future nest trees in the second-growth forest.

Management practices that benefit woodpeckers will also enhance habitat for the “Queen Charlotte” northern saw-whet owl, including the artificial planting of dead snags in second growth and infection of live trees with heartrot fungus (see Section 5.3) to create nesting habitats in young forests.
10.15 Vananda Creek Limnetic Stickleback (*Gasterosteus* species 16) and Vananda Creek Benthic Stickleback (*Gasterosteus* species 17)

10.15.1 Sources

National Recovery Team for Stickleback Species Pairs. 2007.

10.15.2 Conservation Status

The Vananda Creek stickleback only lives in Emily, Priest, and Spectacle Lakes in the Van Anda Creek watershed on Texada Island, British Columbia. The two Vananda Creek stickleback species occur together in their localized state as a pair of closely related species, so they are discussed together. Both species are Red Listed in British Columbia, and Endangered in Canada due to their susceptibility to extinction from the introduction of new species or changes in habitat characteristics in their small range. Current total population sizes and trends are unknown, but one report states that the Vananda Creek stickleback has been found in abundance in all three lakes (Hatfield, 2001 in MWLAP, 2004q). The conservation rankings for this species in all relevant jurisdictions are presented in Table 64.

Table 64. Conservation Rankings for the Vananda Creek Stickelback

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S1</td>
</tr>
<tr>
<td>Canada</td>
<td>N1</td>
</tr>
<tr>
<td>Global</td>
<td>G1</td>
</tr>
</tbody>
</table>

10.15.3 Vananda Creek Stickleback Habitat Characteristics

The Vananda Creek stickleback remains in the lacustrine environment year round. As the names suggest, the limnetic species forms loose schools to forage on zooplankton in the open waters of the lakes, while the benthic species generally forages for invertebrates on the aquatic vegetation in the shallows close to shore and for benthos on the lake bottom. Both species
build nests in the shallows near the shore to breed. The limnetic species nests in relatively shallow open areas, while the benthic species chooses nesting sites at slightly deeper depths with denser aquatic vegetation. The males of both species eat benthos while they tend their nests, while juveniles of both species forage on vegetation for invertebrates before leaving the shallows to assume their respective adult foraging habits. More detailed habitat characteristics can be found in Table 65.

Table 65. Vananda Creek Stickleback Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Characteristics</th>
</tr>
</thead>
</table>
| Foraging           | • Limnetics: clear open water with abundant zooplankton  
                     • Benthics, both species during breeding: shallow lake edges with aquatic vegetation and benthos-rich sediments. |
| Breeding           | • Shallow, vegetated littoral zones.  
                     • Limnetics: gravel and rock substrates or submerged logs at depths <1m.  
                     • Benthics: aquatic vegetation near shore at depths >1 m and <2 m. |

10.15.4 Important Considerations

Habitat Loss and Degradation: The breeding habitats of the Vananda Creek stickleback are highly sensitive: more sensitive to change in habitat and water quality than other populations of sticklebacks. The aquatic habitats of the three lakes can be impacted by sedimentation derived from erosion events on land within the watershed surrounding the lakes. Road building, forest harvesting, mining, clearing for building sites, and natural disturbances have all been known to cause an increase in suspended sediments in water bodies. Changes in water transparency due to suspended sediments have been known to decrease production of zooplankton required by adults of the limnetic species, and associated changes in water quality causing a decline in prey abundance can negatively affect the growth and fitness of benthic juveniles.

A dam currently regulates the water levels in Priest and Spectacle lakes. The water level of Priest Lake and the riparian channel between Priest and Spectacle Lake has already been raised; further changes in water level due to dam management decisions could cause:

- Less suitable littoral habitat and erosion of riparian soils;
- Erosion of exposed littoral areas during rain events in periods of drawdown, and increased sediment loading in the lakes as a result;
- Lower reproductive success if lake level changes during the spawning period; and
• A decrease in gene flow between Emily Lake and Priest and Spectacle Lakes, and an increase in gene flow between Priest and Spectacle Lakes.

**Hybridization:** Changes in suspended sediments can also interfere with mate recognition; reducing the ability of females to discriminate between males of the other species and their own, and causing an increase in the rate of hybridization between the species. An increase of only 3% in the hybridization rate would be sufficient to cause the two species to collapse into a hybrid swarm.

**Introduced Species:** One of the greatest threats to the stickleback species pair remaining on Texada Island is introduced species, which have already rendered the species pair on Lasqueti Island extinct.

**Protection:** The Vananda Creek stickleback species are not protected under the provincial Wildlife Act, but are protected by the provincial Fish Protection Act and the habitat provisions of the federal Fisheries Act.

**Cross References:** None.

**10.15.5 Potential Roles of Second Growth Forest in Conservation**

This species is primarily threatened by habitat loss and degradation from increased siltation and sedimentation due to land management practices in their watershed, and potential changes to habitat structure and water quality due to operation of the Priest Lake dam. Ways in which second-growth forest might be used to conserve the Vandana Creek stickleback are shown in Table 66.
Surface activity such as forest harvesting and road construction can negatively affect water quality in the lake system inhabited by the Vandana Creek stickleback through increased soil erosion leading to siltation and sedimentation. Adherence to the results-based code best management practices for protecting streams and riparian areas in the Van Anda watershed may protect Emily, Priest and Spectacle Lakes from siltation. A WHA is in development for the land around the sticklebacks’ lakes (as of 2007: National Recovery Team for Stickleback Species Pairs. 2007).

Furthermore, silviculture treatments in second-growth forests that open up the canopy to increase understory vegetation may be employed to help control surface runoff and siltation (Section 5.2).

### 10.16 “Vancouver Island” Common Water Shrew (Sorex palustris brooksi)

**10.16.1 Sources**

B.C. Ministry of Water, Land and Air Protection, 2004r

**10.16.2 Conservation Status**

The common water shrew is a widespread species found throughout most of Canada, but the subspecies restricted to Vancouver Island is the only island population of common water shrew on the entire Pacific Coast of America. Even though there are only 67 known records from 38 locations, the Vancouver Island subspecies is believed to live throughout much of Vancouver Island. It has been documented as far north as Quatse River, near the north end of Port Hardy,
along the east coast at Quinsam River, inland at Robertson Creek and Lowry Lake, along the west coast at Lost Shoe Creek, and near Victoria at Veitch Creek. The population is believed to be declining due to habitat loss on southeast Vancouver Island. The conservation rankings for the Vancouver Island common water shrew are provided in Table 67.

Table 67. Conservation Rankings for the “Vancouver Island” Common Water Shrew

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S2</td>
</tr>
<tr>
<td>Canada</td>
<td>N2</td>
</tr>
<tr>
<td>Global</td>
<td>G5T2</td>
</tr>
</tbody>
</table>

10.16.3 “Vancouver Island” Common Water Shrew Habitat Characteristics

The extent of this species home range is not known, but individuals have been captured almost exclusively at the land/water interface, less than 1 metre from the water’s edge. They are presumed to be concentrated near or within the banks of the inhabited stream, with ranges shaped like long, linear strips along the water body. “Vancouver Island” common water shrews have been captured along riparian corridors in all vegetated structural stages, in forests of all ages. They consume terrestrial invertebrates found on the forest floor amongst litter and decomposed coarse woody debris and hollow logs, though up to 50% of their diet consists of aquatic animals and invertebrates. Based on successful capture locations, suitable aquatic habitat for the “Vancouver Island” common water shrew includes intermittent or permanent watercourses, still pools, slow- or swift-flowing streams, marshes, ponds, lakes, and small seepages. Streams with gravel and cobble substrate appear to be preferred to those with bedrock substrates. Suitable aquatic habitat appears to be more important than the structural stage of the surrounding terrestrial habitat.

If riparian zones near habitable waterways are harvested, then water shrews will likely not be present until stream water quality and riparian zone vegetation recovers; thus, urban development and forestry practices occurring within riparian habitats are degrading and reducing the amount of preferred common water shrew habitat. Typical habitat characteristics can be found in Table 68.
Table 68. “Vancouver Island” Common Water Shrew Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aquatic</td>
<td>• Cool, swift flowing streams at high elevations (mainland subspecies.)</td>
</tr>
<tr>
<td></td>
<td>• Low gradient, low elevation watercourses &lt;10 m wide (most sampled for Vancouver Island subspecies.)</td>
</tr>
<tr>
<td></td>
<td>• Abundant rocks and boulders within and around the watercourse.</td>
</tr>
<tr>
<td></td>
<td>• Gravel or cobble stream substrate; especially not bedrock.</td>
</tr>
<tr>
<td>Terrestrial</td>
<td>• Any vegetated riparian habitat is suitable.</td>
</tr>
<tr>
<td></td>
<td>• Complex riparian habitat with overhanging vegetation, undercut banks, exposed tree roots, crevices, and in-stream woody debris preferred.</td>
</tr>
<tr>
<td></td>
<td>• Litter, decomposed coarse woody debris, and hollow logs.</td>
</tr>
<tr>
<td></td>
<td>• Elevations up to 558 m (highest elevation captured).</td>
</tr>
</tbody>
</table>

10.16.4 Important Considerations

**Limited Distribution:** A restricted distribution makes this subspecies vulnerable to extinction from environmental changes that further restrict distribution and the ability to recolonize habitats.

**Habitat Loss and Degradation:** Removal of riparian vegetation; erosion and siltation from forest harvesting; water contamination from urban sources, industrial waste dumping, and pesticide runoff have all had a detrimental effect on habitat quality and the diversity of aquatic food sources.

**Protection:** The “Vancouver Island” common water shrew is protected under the provincial Wildlife Act. Some known habitats exist in provincial parks, but because the true extent of this subspecies populations is unknown, it is not known what percentage of the shrew’s habitat is represented in these parks (e.g. Goldstream Provincial Park, including the Greater Victoria Water District adjacent to the park, Pacific Rim National Park, Miracle Beach Provincial Park, Veitch Regional Park and Niagara Catchment, Dudley Marsh, and Marble River Provincial Park).

**Cross References:** Strategies to maintain habitat connectivity in second-growth forest stands that benefit the red-legged frog are likely to benefit the “Vancouver Island” common water shrew.
10.16.5  Potential Roles of Second-growth Forests in Conservation

This species is primarily threatened by habitat loss and degradation from human activities in riparian areas and removal of riparian vegetation. Ways in which second-growth forest might be used to conserve the “Vancouver Island” common water shrew are shown in Table 69.

Table 69. Potential Roles of Second-growth Forest in the Conservation of the “Vancouver Island” Common Water Shrew

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment/Restoration</th>
</tr>
</thead>
<tbody>
<tr>
<td>“Vancouver Island” Common Water Shrew</td>
<td>X</td>
<td>X</td>
<td></td>
<td>X</td>
</tr>
</tbody>
</table>

Removal of riparian vegetation during logging can reduce the amount of common water shrew habitat available. Adherence to the results-based code best management practices for protecting streams and riparian areas on Vancouver Island may help to conserve common water shrew habitat in logged areas and prevent the degradation of downstream water quality. Since water shrew home ranges are likely highly linear and strongly associated with stream edges, maintaining riparian vegetation during logging and replanting of riparian vegetation in second-growth forests may help to maintain connectivity between shrew habitats and allow for dispersal.

10.17 “Queen Charlotte” Hairy Woodpecker (*Picoides villosus picoides*)

10.17.1  Sources

B.C. Ministry of Water, Land and Air Protection, 2004s.
10.17.2 Conservation Status

The “Queen Charlotte” hairy woodpecker is endemic to the Queen Charlotte Islands. There are no data on breeding territory size, home ranges, or population and habitat trends for this subspecies. Breeding Bird Surveys for other subspecies found that population sizes increased in British Columbia between 1966 and 1996; though in other regions hairy woodpecker populations have been declining. Conservation rankings for the “Queen Charlotte” hairy woodpecker are show in Table 70.

<table>
<thead>
<tr>
<th>Location</th>
<th>Conservation Ranking</th>
</tr>
</thead>
<tbody>
<tr>
<td>British Columbia</td>
<td>S3</td>
</tr>
<tr>
<td>Canada</td>
<td>N3</td>
</tr>
<tr>
<td>Global</td>
<td>G5T3</td>
</tr>
</tbody>
</table>

Table 70. Conservation Rankings for the “Queen Charlotte” Hairy Woodpecker

10.17.3 “Queen Charlotte” Hairy Woodpecker Habitat Characteristics

Hairy woodpeckers nest and roost in cavities excavated in live trees with heart rot or dead trees. Many species of coniferous or deciduous trees may be used, though most nests have been found in western hemlock (60%), Douglas-fir (20%), red alder (4%), and western white pine (3%). Hairy woodpeckers can inhabit a variety of forest types and successional stages: mature conifer stands, mature hardwood stands, meadow edges, riparian zones, recent burns, and old growth. Mature to old conifer stands, or younger, diseased conifer stands are preferred nesting habitats when a mix of decay classes (2-6) is present. Typical habitat characteristics can be found in Table 67.
Table 71. “Queen Charlotte” Hairy Woodpecker Habitat Characteristics

<table>
<thead>
<tr>
<th>Habitat Components</th>
<th>Typical Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nesting</td>
<td>• Tall, dead conifers with large dbh or live trees infected with heart rot.</td>
</tr>
<tr>
<td></td>
<td>• Trees in decay classes 2-7; classes 4 and 5 preferred (&gt;95% bark remaining).</td>
</tr>
<tr>
<td></td>
<td>• Forests on slopes &lt;20% preferred.</td>
</tr>
<tr>
<td>Foraging</td>
<td>• Mature coniferous forests</td>
</tr>
<tr>
<td></td>
<td>• Deciduous and mixed forests</td>
</tr>
<tr>
<td></td>
<td>• Forests with meadow openings, marshes, ponds, logged areas, and burns</td>
</tr>
<tr>
<td></td>
<td>• Douglas-fir and western hemlock second growth (Washington Cascades).</td>
</tr>
<tr>
<td></td>
<td>• Large diameter trees (&gt;50 cm) are preferred in old growth.</td>
</tr>
<tr>
<td></td>
<td>• 11-50 cm dbh trees in second growth.</td>
</tr>
<tr>
<td></td>
<td>• Residential gardens and feeders (winter).</td>
</tr>
<tr>
<td>Wintering</td>
<td>• Use old tree cavities or excavate new cavities for roosting.</td>
</tr>
<tr>
<td></td>
<td>• Variety of forest types essential: similar to nesting habitat.</td>
</tr>
<tr>
<td></td>
<td>• Mixture of coniferous trees in old growth and deciduous trees in second growth.</td>
</tr>
</tbody>
</table>

10.17.4 Important Considerations

**Habitat Loss and Degradation:** A primary threat is the loss of coniferous and hardwood forest habitats of all ages required for breeding and foraging through logging. Snag removal and even-aged stand development associated with forest management can also contribute to population reduction.

**Limited Distribution:** A restricted range makes this subspecies vulnerable to extinction due to stochastic occurrences and changes in limited habitat.

**Protection:** The hairy woodpecker, its nests, and its eggs are protected under the Wildlife Act and the Migratory Birds Convention Act. Nesting habitat is currently protected in the Gwaii Haanas National Park Reserve, Naikoon Park, and several smaller reserves within the Queen Charlotte Islands. Most of the remaining nesting habitat is on Crown land.

**Cross References:** Wildlife tree recruitment and retention strategies in second-growth forests that benefit the “Queen Charlotte” northern saw-whet owl will also benefit the “Queen Charlotte” hairy woodpecker.
10.17.5 Potential Role of Second-growth Forests in Conservation

This species is primarily threatened by habitat loss and degradation through logging. Ways in which second-growth forest might be used to conserve the Queen Charlotte hairy woodpecker are shown in Table 58.

Table 72. Potential Roles of Second-growth Forest in the Conservation of the “Queen Charlotte” Hairy Woodpecker.

<table>
<thead>
<tr>
<th>Species</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment/Restoration</th>
</tr>
</thead>
<tbody>
<tr>
<td>“Queen Charlotte” Hairy Woodpecker</td>
<td></td>
<td></td>
<td>X</td>
<td>X</td>
</tr>
</tbody>
</table>

Hairy woodpeckers have been recorded in second-growth Douglas-fir and western hemlock stands in the Washington Cascades, and may use the same species plantations in the Queen Charlottes. Second-growth forests with deciduous tree species present are also preferred.

The hairy woodpecker utilizes forests with mixed seral stage patches and clearings, so second-growth forests can be used to increase the amount of foraging habitat. Smaller clearcuts with wildlife tree retention areas could also create more favorable edge habitats used for foraging. Management practices that increase the retention of large trees or thinning practices that promote uneven tree size distributions may provide future nest trees in the second-growth forest.

Management practices that benefit the “Queen Charlotte” northern saw-whet owl will also enhance habitat for woodpeckers, including the artificial planting of dead snags in second growth and infection of live trees with heartrot fungus (see Section 5.3) to create nesting habitats in young forests.
Chapter 5  Summary and Recommendations

11 Summary and Recommendations

Although natural forests generally offer superior habitat for native forest species than plantation forests, the degree to which plantation forests can provide suitable habitat for native species is dependent on management intensity. The magnitude of the impact to site biodiversity will depend on the harvest methods used as well as the management of the resulting second-growth stand. Harvest method influences include the type of harvest, harvesting system, initial forest type, and time of year of the harvest. A forest may take years to decades to recover from impacts such as soil compaction and changes to soil structure, and reductions in soil organic matter and nutrient pools. In terms of stand management, hand-planted second-growth stands are typically much denser than stands natural processes would have created, and plantation forests generally contain few, if any, biological legacies such as snags and fallen trees or coarse woody debris. This generally translates into less habitat diversity and structural complexity than naturally regenerated forests.

Despite the limitations of plantation forests, however, there are an increasing number of instances where species at risk have been recorded in plantations, and second-growth forests can contribute to maintenance and enhancement of overall biodiversity through:

1. Habitat supplementation or complementation,
2. By buffering remnant forests from edge effects, and
3. By improving connectivity between remnant forests.

Second-growth forests can provide habitat supplementation because some species that survive in remnant forest patches may compensate for habitat loss by using resources in the surrounding landscape. The landscape matrix surrounding remnant forest patches may be neither uniformly unsuitable as habitat nor serve as a complete barrier to the dispersal of forest species. When their primary forest habitat is in short supply, forest fauna are more likely to use second-growth forests than land under other uses, as even managed stands will have some structural similarities to their preferred forest habitats.

Second-growth forests adjacent to primary forest habitats such as old-growth remnants, or sensitive ecosystems can also be used to help maintain the attributes that characterize these areas by buffering them from edge influences, which have a larger effect on species composition than any other aspect of forest structure. Forest buffers can:
1. Reduce edge effects,
2. Extend the effective size of core areas,
3. Reduce the potential for invasion by organisms adapted to edge environments,
4. Enhance forest interior habitat,
5. Reduce the likelihood of blowdown within core areas, and
6. Reduce disturbance of important sites such as nest and breeding areas.

Other potential benefits may include a temporary refuge for plants and animals after natural disturbances within core areas, and a source of replacement species for old-growth core areas lost to catastrophic disturbances.

In addition to buffering and supplementing primary forest or sensitive habitats, second-growth forests can also enhance indigenous biodiversity in fragmented landscapes by improving connectivity between forest remnants. Southern and eastern Vancouver Island have become extensively urbanized and developed, and much of the forest in the interior of the Island and north of Vancouver has been fragmented. This connectivity will likely become increasingly important under a changing climate regime, when plants and animals may require these forests in order to migrate to new ranges. In addition, using second-growth forests to maintain connectivity on the landscape can help prevent species from becoming endangered or extirpated; large populations are less likely to go extinct than small populations, and large habitat patches, which are more likely to support large populations, are more likely to be occupied than small patches.

Specific management tools can also be applied on a site level in second-growth forests to recruit habitat characteristics, such as wildlife trees, to increase the habitat value of second-growth stands for specific species. Management strategies employed in young forest to increase biodiversity values are based on the idea that affinity of those wildlife species occurring in late-seral forests is more likely attributable to forest structure than the age of the forest.

Plantation management practices that generally increase biodiversity values within a stand include:

a) Selecting tree species that provide resources and structures that favour native species;
b) Avoiding intensive site preparation that destroys herbaceous vegetation and coarse woody debris;
c) Implementing wider tree spacing and heavy pre-commercial thinning to help maintain understory vegetation;
d) Increasing rotation length; and
e) Maintaining some structural attributes such as old trees or snags.
Wildlife trees are generally created over decades through mortality agents such as fire, disease, insects, windthrow, snowpress, lightning, and wildlife excavation; management tools that might be employed to create wildlife trees in a shorter time frame include topping trees with chainsaws or explosives, girdling trees, cavity creation using chainsaws, and inoculating trees with native fungus. Coarse woody debris can also be recruited by leaving a mixture of coniferous and deciduous thinned trees on the forest floor.

Forest managers could design silvicultural options to put second-growth forests on different pathways likely to lead to greater forest complexity and habitat diversity using techniques such as partial cut harvesting, pre-commercial thinning, and variable-density thinning. Tree spacing should vary to accelerate the development of old-growth structural characteristics, such as a wide range of tree sizes and abundant understorey vegetation, which are crucial to maintaining a diversity of species. Some additional common forest restoration techniques to increase habitat heterogeneity and mimic natural successional patterns include prescribed burning, under-planting, and mycorrhizal fungus recruitment.

On a landscape scale, second-growth forests can be used to maintain a patchwork of seral stages, manage patch size and distribution (using second growth as buffers to increase old growth and riparian patch sizes), increase forest connectivity (including inter-patch, cross-elevational, and cross-valley connectivity), create barriers to human access to sensitive areas, and provide visual breaks and security cover for wildlife, especially in areas with abundant human access. Furthermore, even if efforts to protect all remaining old-growth stands are successful, additional areas of older second-growth forest will have to be protected and allowed to recover to an old-growth state in order to ensure adequate representation of these forest types in the future, and to provide a continuous network of wildlife habitat.

The potential contributions of second-growth forest to management for Identified Wildlife species and communities on Vancouver Island and the Gulf Islands are summarized in Table 73, below. In general, where second-growth forests can be used to maintain connectivity or provide buffers, decisions will need to be made on a landscape scale as to which forest areas to include in management for species at risk. Where second-growth forests can be used to provide habitat supplementation or complementation, management decisions on how to enhance wildlife habitat can be made at the stand level as well as in a landscape context (i.e., provided the stand to be managed is adjacent to known or potential habitat for the target species).
<table>
<thead>
<tr>
<th>Species/Community</th>
<th>Connectivity</th>
<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment/Restoration</th>
<th>Forest Management Scale</th>
<th>Management Attributes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Douglas-fir/dull Oregon grape</td>
<td></td>
<td>X</td>
<td></td>
<td>X</td>
<td>Stand and Landscape</td>
<td>• Variable thinning&lt;br&gt;• Snag recruitment&lt;br&gt;• CWD recruitment&lt;br&gt;• Prescribed burning&lt;br&gt;• Brush removal</td>
</tr>
<tr>
<td>Douglas-fir/Alaska oniongrass</td>
<td></td>
<td>X</td>
<td></td>
<td>X</td>
<td>Stand and Landscape</td>
<td>• Tree species choice&lt;br&gt;• Variable thinning&lt;br&gt;• Snag recruitment&lt;br&gt;• CWD recruitment&lt;br&gt;• Prescribed burning&lt;br&gt;• Brush removal</td>
</tr>
<tr>
<td>Scouler’s corydalis</td>
<td></td>
<td>X</td>
<td></td>
<td>X</td>
<td>Stand and Landscape</td>
<td>• Tree species choice&lt;br&gt;• Variable thinning</td>
</tr>
<tr>
<td>Keen’s long-eared myotis</td>
<td>X</td>
<td>X</td>
<td></td>
<td>X</td>
<td>Stand and Landscape</td>
<td>• Wildlife tree recruitment&lt;br&gt;• Appropriate buffer size maintenance</td>
</tr>
<tr>
<td>“Vancouver Island” northern pygmy owl</td>
<td></td>
<td>X</td>
<td>X</td>
<td>X</td>
<td>Stand</td>
<td>• Maintain early seral stages&lt;br&gt;• Recruit wildlife trees</td>
</tr>
<tr>
<td>Species/Community</td>
<td>Connectivity</td>
<td>Buffering</td>
<td>Habitat Supplementation/Complementation</td>
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<td>Forest Management Scale</td>
<td>Management Attributes</td>
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</tbody>
</table>
| Short-eared owl            |              |           | X                                      | X                              | Stand                    | • Maintain early seral stages  
|                            |              |           |                                        |                                 |                          | • Recruit wildlife trees                  |
| Queen Charlotte goshawk    | X            | X         |                                        | X                              | Stand and Landscape      | • Recruit wildlife trees  
|                            |              |           |                                        |                                 |                          | • Variable thinning (enhance prey populations)  
|                            |              |           |                                        |                                 |                          | • Appropriate buffer size maintenance |
| Great blue heron           | X            |           |                                        | X                              | Stand and Landscape      | • Tree species selection  
|                            |              |           |                                        |                                 |                          | • Restocking  
|                            |              |           |                                        |                                 |                          | • Appropriate buffer size maintenance |
| Sandhill crane             |              | X         |                                        |                                | Landscape                | • Appropriate buffer size maintenance |
| “Vancouver Island” white-tailed ptarmigan | X            | X         |                                        | X                              | Stand and Landscape      | • Appropriate buffer size maintenance  
<p>|                            |              |           |                                        |                                 |                          | • Wildlife tree recruitment  |</p>
<table>
<thead>
<tr>
<th>Species/Community</th>
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<th>Forest Management Scale</th>
<th>Management Attributes</th>
</tr>
</thead>
</table>
| Red-legged frog   | X            | X         | X                                      | X                             | Stand and Landscape    | • Appropriate buffer size maintenance  
|                   |              |           |                                        |                               |                        | • CWD recruitment         
|                   |              |           |                                        |                               |                        | • Plan to maintain corridors connecting wetlands |
| Quatsino cave amphipod | X          |            |                                        | X                             | Stand and Landscape    | • Appropriate buffer size maintenance  
|                   |              |           |                                        |                               |                        | • Understory enhancement    |
| Vancouver Island marmot |          |            |                                        | X                             | Stand                  | • Early seral stage maintenance  |
| Marbled murrelet  | X            |           |                                        | X                             | Stand and Landscape    | • Partial cut harvesting/thinning to promote tree size  
<p>|                   |              |           |                                        |                               |                        | • Plan to maintain connections with epiphyte source population |
| Ancient murrelet  | X            |           |                                        | X                             | Stand and Landscape    | • CWD recruitment          |</p>
<table>
<thead>
<tr>
<th>Species/Community</th>
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<th>Buffering</th>
<th>Habitat Supplementation/Complementation</th>
<th>Habitat Recruitment/Restoration</th>
<th>Forest Management Scale</th>
<th>Management Attributes</th>
</tr>
</thead>
</table>
| “Queen Charlotte” northern saw-whet owl | | X | X | | Stand | • Thinning  
• Wildlife tree recruitment  
• Brushing/understory management |
| Vananda Creek Stickleback Species Pair | X | | X | | Landscape | • Appropriate buffer size maintenance  
• Understory enhancement |
| “Vancouver Island” common water shrew | X | X | | | Stand and Landscape | • Selection of areas to be maintained to provide riparian corridors  
• Riparian species selection |
| “Queen Charlotte” hairy woodpecker | | | X | X | Stand and Landscape | • Wildlife tree retention/recruitment  
• Variable thinning |
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MWLAP, see B.C. Ministry of Water, Land and Air Protection.

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