A STRATEGY FOR MAINTAINING WELL-DISTRIBUTED, VIABLE POPULATIONS OF WILDLIFE ASSOCIATED WITH OLD-GROWTH FORESTS IN SOUTHEAST ALASKA

Report of an Interagency Committee

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Abstract: Sub-specific variation and unique distribution patterns of wildlife are characteristic of the Tongass National Forest on the mainland and Alexander Archipelago of southeast Alaska. Maintaining viable, well-distributed populations of wildlife across this 17 million-ac landscape is required by the 1976 National Forest Management Act and offers a significant challenge to the USDA Forest Service. In this document an interagency committee, appointed by the Forest Service, proposes conservation measures necessary to meet this requirement. A screening process was used to identify and evaluate wildlife species which were potentially most sensitive to reasonably foreseeable land management actions. The natural histories of 11 species were summarized for which there is a high level of concern. Five species were determined to require large tracts dominated by old-growth forest, of varying size, distributed across the Tongass National Forest to maintain viability and current distribution. The most restrictive elements for each species were combined to develop a single strategy for all 5 species across the Forest (i.e., 40,000+ ac tracts, ≤20 mi apart; 10,000+ ac tracts, ≤8 mi apart; and a 1,600 ac tract in each major watershed). This was done to reduce the cumulative effect of species-specific requirements on the commercial timber base. Specific management standards necessary for maintaining viability and distribution of all 11 species, but not associated with tracts of old-growth forest, are also proposed.

THE SETTING

The Tongass National Forest, at 17 million acres, is the largest in the National Forest System. The Tongass includes thousands of islands, known as the Alexander Archipelago, and a narrow strip of mainland that rises abruptly to glaciers and icefields capping the Coast Range (Figure 1) (Harris et al. 1974). The landscape is exceptionally steep and rugged, particularly in the north, with mountains reaching 3-4,000 ft on larger islands, and over 10,000 ft on the mainland. The climate is strongly maritime, with cool summers, mild winters, and abundant precipitation (100-200 in annually) distributed throughout the year.

The coastal forests of southeast Alaska are part of the temperate rainforest biome. Defined by the distribution of Sitka spruce (Picea sitchensis) and western hemlock (Tsuga heterophylla), the biome extends along the Pacific coast from northern Kodiak Island to southern Oregon. In southeast Alaska, western hemlock-Sitka spruce forest types predominate on 96% of all productive land, with 63% of those stands having western hemlock dominant, 23% having Sitka spruce dominant, and 35% classified as mixed (Hutchison 1967). Minor amounts of western red cedar (Thuja plicata), Alaska cedar (Chamaecyparis nootkatensis), mountain hemlock (Tsuga mertensiana), and shore pine (Pinus contorta) occur primarily on poorly-drained or high-elevation sites. Alder (Alnus spp.) is common along streams, beach fringes, avalanche slopes, and recently disturbed soils. The forest understory is characterized by a wide variety of shrubs and forbs. Common plants include blueberry (Vaccinium spp.), devil's club (Oplopanax horridus), bunchberry (Cornus canadensis), skunk cabbage (Lysichiton americanum), and numerous ferns and mosses (Alaback and Juday 1989).

The vast majority of productive forest land in southeast Alaska is classified as "old growth" and is typical of forests which develop in the absence of large-scale catastrophic disturbance such as wild fire. Individual trees can attain ages well in excess of 500 years and trees older than 300 years are common. Blowdown is the most important natural disturbance process in these forests (Harris 1989). The high-frequency, low-intensity disturbance regime typically affects individual trees or small patches of trees. As old trees die and fall to the ground, the new canopy gap allows sunlight to reach the forest floor, prompting a response in understory shrubs and forbs, and young trees. This results in an irregular patchwork of all-aged trees, uneven canopy, diverse understory, and large woody material on the ground. All these components contribute to the structural, compositional, and functional diversity typical of old-growth forests (Franklin et al. 1981). Ten distinct types of temperate old-growth forests are recognized on the Tongass National Forest (USDA Forest Service, Alaska Region, unpubl. data).

These forests are dynamic, steady-state plant communities where the death of single or small groups of trees are balanced by the growth of new trees. This has been termed a "shifting-mosaic steady state" (Bormann and Likens 1979). Wildlife in southeast Alaska have developed natural history patterns that are closely linked with this complex of steady-state old-growth forests (Schoen et al. 1988).

Although commercial logging has occurred in southeast Alaska since the early 1900s, large-scale utilization of timber resources did not begin until the early 1950s. Since 1954 clearcut logging has removed approximately 6.7% of the "productive old-growth" (i.e., $\geq$8,000 board ft per ac) on the Forest; slightly more than 5,000,000 ac of productive old-growth forest remain (USDA Forest

Service 1991). Alaska is one of the few places in the world where large tracts of pristine temperate rainforest still remain (Alaback in press).

THE NEED

Sub-specific variation and distribution patterns of wildlife in southeast Alaska have resulted from the discontinuous nature of habitat in southeast Alaska. Many of the subspecies endemic to coastal rainforest in southeast Alaska are found on only a few islands (e.g., Hall 1981). The complexity of habitats and frequency of subspecies with limited distribution provide a significant challenge to the USDA Forest Service to maintain biological diversity within the context of ongoing land management activities under a multiple-use mandate.

Timber harvests in this area under a 100-year rotation result in an essentially permanent change from the steady-state forest condition (Alaback 1984). Although clearcut logging has affected a relatively small percentage of southeast Alaska, it has and will have significant impacts in the more productive areas of the Forest (e.g., Prince of Wales Island). Where logging does occur, it is typically concentrated in the rare, highly productive old-growth stands at low elevations. In general, these same stands are the most valuable for wildlife; their loss results in disproportionate impacts on certain species (Schoen et al. 1988).

Management of landscapes through application of the principles of conservation biology provides land managers an opportunity to maintain biological diversity in southeast Alaska (e.g., Suring and Crocker-Bedford 1992). Conservation of biological diversity requires specific actions to ensure that viable populations of all wildlife are maintained and are well distributed over the landscape (Keystone Center 1991). The revision of the Tongass National Forest Land Management Plan provides the opportunity to develop and implement management standards and guidelines that reduce the risk of additional species being listed under the Endangered Species Act.

THE PROCESS

Background

Rules and regulations were developed to facilitate implementation of the National Forest Management Act of 1976 on National Forests (USDA Forest Service 1982). These rules and regulations direct the USDA Forest Service to manage wildlife habitats to maintain viable populations of existing native and desired non-native vertebrate species on National Forests. A viable population is defined as "...one which has the estimated numbers and distribution of reproductive individuals to insure its continued existence is well distributed..." throughout a National Forest (USDA Forest Service 1982:43048). "Well distributed" has been defined "throughout the existing range of the subspecies" (USDA Forest Service 1984).

The rules and regulations further state that habitat must be provided to support viable populations and that "...habitat must be well distributed so that...individuals can interact with others..." on National Forests (USDA

Forest Service 1982:43048). "Well distributed" is more specifically defined for the purposes of this document to mean that a species has a high likelihood of occurring within each third-order watershed (e.g., $\geq$10,000 ac) within its current range. Precedence for this definition was set in the resolution of the appeal of the Flathead National Forest Land Management Plan in Montana.

An interagency committee was assembled in October 1990 by the Tongass Land Management Plan Revision Interdisciplinary Team. The charge to this committee was to develop and recommend management standards that provided a high likelihood of maintaining viable, well-distributed populations of old-growth associated species on the Tongass National Forest over the long term (i.e., 100 years).

To accomplish this the committee implemented the following process:

1.) reviewed prior efforts to address species with viability or distribution concerns on the Tongass National Forest;

2.) identified species associated with old-growth forest communities that may have viability or distribution concerns either in the next 10 years or as a result of the cumulative impacts of proposed management actions over the long term (i.e., 100 yrs);

3.) documented the best information available on taxonomy, population status, demographics, and habitat relationships of identified species and the need for research to fill significant data gaps;

4.) developed management standards to maintain viable, well-distributed populations for species associated with old-growth forests for which a concern for viability or distribution has been identified; and

5.) consolidated management standards for each species into a conservation strategy that satisfies the overall charge.

Selection of Species for Review

Southeast Alaska provides habitat for 275 bird species, 73 mammal species, and 8 species of amphibians and reptiles (Taylor 1979). Of these, 44 bird species and 3 species of amphibians and reptiles are on the geographic edge of their range or occur here only accidentally. Of the remaining 309 species, 103 were associated with old-growth forests. The assumption was made that these species differentiate among habitats on the basis of forest age, composition, and/or structure. These 103 species were previously evaluated for viability and distribution concerns using 17 criteria developed and used by another task group in 1988 (Table 1) (Orme 1988).

This screening phase was repeated using information not available during the previous evaluation. These criteria were also weighted from 1 to 5 during this exercise to reflect their importance in determining whether a species should be considered in this analysis. Conservation planning was considered necessary if a species exhibited specialization for habitats that are declining in abundance and the species experienced a documented population decline or has a high likelihood for a population decline. Eleven species associated with old-growth forest habitats were identified as having potential viability and/or distribution concerns (Table 2). A level of concern associated with each evaluation criteria for each of the 11 species was also recorded (Table 3). Levels of concern were multiplied by the weighting factors of the evaluation criteria and the products summed to provide a ranking of concern for the 11 species (Table 4).

Development of Conservation Strategies

Literature and unpublished records concerning the distribution, taxonomic status, and natural history of each of these 11 species were reviewed. This information was examined to determine if specific management practices to maintain habitat capability could be implemented to assure their continued viability and distribution in southeast Alaska. Detailed information on the habitat requirements, reproductive biology, sensitivity to impacts, and standards needed to maintain distribution and viability of each species is included in the individual species reports (Appendix B).

Management strategies were developed for each of the 11 species. Five species required tracts of varying size dominated by old-growth forest distributed across the Tongass National Forest to maintain their current distribution and ensure long-term viability (Table 5). The specific requirements for each species were compared and the most restrictive elements were integrated to develop an overall strategy for maintaining habitat for all 5 species across the Forest. This was done to minimize the cumulative effect of management standards for each of the species.

These species are landscape or community level species as defined by Orme et al. (1990). Implementation of conservation strategies for species at these levels tends to ensure that viability and distribution of all species associated with old-growth forest habitats will be assured. Species-specific management standards important for maintaining viability and distribution but not associated with tracts of old-growth forest are also proposed.

Numerous publications provide a quantitative basis for the development of conservation strategies (see Suring and Crocker-Bedford [1992] for a summary). However, before the development of the conservation strategy for the northern spotted owl (Strix occidentalis caurina) (Thomas et al. 1990), a step-by-step application of biological data to the development of a habitat reserve system had not been accomplished (Murphy and Noon 1992). The effort described in this document followed the basic approach taken in the conservation strategy for the northern spotted owl.

In many cases comprehensive, local information was not available for critical habitat and population factors required to develop risk-free management standards. Consequently, the management standards presented here are often based on information from other areas within the species range. Additional local information is needed to verify if these standards are adequate and effective for southeast Alaska.

Application of the Conservation Strategy

The standards developed for maintaining tracts of old-growth habitat were applied on the Tongass National Forest to demonstrate their implementation.

This application was based on the steps for mapping Habitat Conservation Areas (HCAs) described by Murphy and Noon (1992). Maps were generated at a 1:500,000 scale by a Geographic Information System (GIS).

1.) A land ownership map layer was generated to ensure that HCAs were placed on National Forest lands.

2.) Wilderness areas and other areas legislatively removed from timber harvest were delineated on the map. These areas were used for placement of HCAs, to the greatest extent possible within established criteria, to minimize the effect of HCAs on lands available for timber harvest. Lands not suitable for timber harvest or difficult to harvest were also used to the extent possible for HCA placement.

3.) Old-growth forest communities and existing clearcuts were delineated to provide a basis for locating HCAs in areas that met habitat specifications.

4.) Size and habitat composition of the HCAs were evaluated through GIS analysis following initial and subsequent delineations. Adjustments were made in the size, shape, and location of the HCAs through an iterative process to meet the criteria more closely. We also adjusted the HCA boundaries to take advantage of old-growth habitat that would meet the requirements of wildlife, but for a variety of reasons, was unsuited or less economic to log. By doing this, we hoped to minimize the effects of the HCA withdrawal on the Allowable Sale Quantity

(ASQ), and minimize adverse economic consequences on the timber industry.

5.) In calculating the additive impact of the proposed standards on the ASQ, we tabulated the number of tentatively suitable acres withdrawn in the HCAs. Not included in this total were tentatively suitable areas already withdrawn for other reasons, including

i.) Wilderness areas,

ii.) legislated roadless areas,

iii.) beach fringe and estuary buffer strips (common to all alternatives), and

iv.) legislated buffers along streams.

Using a regression equation (USDA Forest Service, unpubl. data), the opportunities foregone in terms of ASQ were computed using the number of acres of tentatively suitable forest land withdrawn for the HCAs.

PROPOSED MANAGEMENT STANDARDS

OLD-GROWTH FORESTS

Goal: Maintain sufficient habitat to ensure that species which require large tracts of old-growth forest have a high likelihood of continued existence throughout their range in southeast Alaska.

These proposed standards were developed to locate tracts dominated by relatively undisturbed, old-growth forest habitat. These tracts should be close enough together across the landscape so that the local population of species of concern (e.g., brown bear, marten, Queen Charlotte goshawk, boreal owl) occupying each tract can adequately interact with nearby populations.

Such interaction provides for the essential interchange of individuals among populations or demes. However, recolonization of vacant habitats from occupied habitats may be more critical for viability and distribution. Any local population may disappear; recolonization counteracts such localized extinctions. The rate of recolonization is associated with the rate at which dispersers happen upon unoccupied habitat. The distance to occupied habitats relative to a species' dispersal capabilities, the presence of suitable travel corridors, and the productivity of nearby occupied habitats all affect the recolonization rate. Without recolonization, interchange of individuals is not achieved and the maintenance of a well-distributed population is not possible.

These standards are intended to minimally ensure continued distribution of wildlife species over most of their current range in southeast Alaska. Game species and furbearers will typically require additional old-growth forest habitat if their populations are intended to provide for recreational and subsistence harvests and wildlife viewing.

Large Habitat Conservation Areas

Large tracts of habitat dominated by old-growth forest are intended to ensure that populations of marten, boreal owls, goshawks, wolves and brown bears will be secure (Figure 2). These Large HCAs are intended to produce enough marten and boreal owls to recolonize vacant, suitable habitats within their dispersal range. The Large HCAs are intended to support enough goshawks that the chance of local extinction is less than in more fragmented habitats. The goshawks produced there may also disperse to other suitable habitats. Because of minimal road access within the tracts Large HCAs are also intended to provide critical refugia for wolves and brown bears. Ensuring long-term viability of brown bear and wolf populations will also require management actions beyond the establishment of HCAs (see management standards for individual species).

Objectives:

1.) Maintain (i.e., limit timber harvest, minimize roads and clearing widths, and minimize vehicle access) one contiguous tract capable of supporting at least 5 female brown bears, 25 female marten during winters of poor prey, 8 pairs of goshawks, and 24 pairs of boreal owls

(Crocker-Bedford 1992, Flynn 1992, Suring 1992a, Titus and Schoen 1992).

2.) The best available information indicates that to meet the population objectives for a Large HCA, a tract should include at least 20,000 ac of old-growth with over 8 thousand board feet (mbf) per ac, including at least 10,000 ac with over 20 mbf per ac, and at least 1 Class I, anadromous fish stream (if the HCA is within the range of brown bears), within a total area of at least 40,000 ac. HCAs with largely circular shapes are preferable as they provide a greater amount of interior old-growth forest environment than more linear HCAs.

3.) Large HCAs should be not more than 20 mi apart, edge to edge, to ensure that dispersal effectively occurs between them.

4.) Often an area with another Forest Plan prescription (e.g., old-growth, Wilderness, Primitive Recreation, or Municipal Watershed Prescription) will serve as a Large HCA. In such cases the prescriptions should be co-designated on maps to clearly indicate the intent to manage for both purposes.

5.) Monitoring should be implemented to determine whether the Large HCAs are meeting their population objectives.

Medium Habitat Conservation Areas

Medium HCAs are intended to provide habitat for small, local populations that may be prone to frequent, local extinctions (Figure 2). However, the Medium HCAs should be located close enough to the Large HCAs or to other Medium HCAs for recolonization to occur.

Objectives:

1.) At intervals of approximately 8 mi, retain Medium HCAs capable of supporting at least 5 female marten during winters of poor prey, 2 pairs of goshawks, and 9 pairs of boreal owls (Crocker-Bedford 1992, Flynn 1992, Suring 1992a).

2.) The best available information indicates that a Medium HCA should encompass at least 5,000 ac of old-growth forest with over 8 mbf per ac, including at least 2,500 ac of old-growth forest with over 20,000 mbf per ac, within an area of at least 10,000 ac. HCAs that are somewhat circular are preferable to linear ones because of the smaller area of edge habitat.

3.) Often an area with another Forest Plan prescription (e.g., old-growth, Wilderness, Primitive Recreation, or Municipal Watershed Prescription) will serve as a Medium HCA. In such cases the prescriptions should be co-designated on maps to clearly indicate the intent to manage for both purposes.

4.) Monitoring should be implemented to determine whether the Medium HCAs are meeting their population objectives.

Small Habitat Conservation Areas

Small HCAs are maintained to provide temporary functional habitat for animals dispersing between Large and Medium HCAs and to ensure that species of concern have a relatively high likelihood of occurring in each third-order watershed (e.g., $\geq$10,000 ac) at least on a temporary basis (Figure 2). The Small HCAs also contribute to the landscape matrix between Large and Medium HCAs. Small HCAs help reduce risk of mortality to dispersers and enhance population stability.

Objectives:

1.) Maintain 1 Small HCA capable of supporting at least 1 female marten during winters of poor prey and 20 to 40 flying squirrels within each major watershed ($\geq$10,000 ac) (Flynn 1992, Suring 1992b).

2.) A Small HCA is estimated to include at least 800 ac of old-growth forest having over 8 mbf per ac within an area of at least 1,600 ac.

3.) Small HCAs should be desginated at the project level. Lands not suitable for timber harvest, existing buffers, and other lands removed from timber harvest should be used to the extent practicable for Small HCAs.

Travel Corridors

Objective: Provide corridors of old-growth forest habitats to increase the likelihood of dispersal of the species of concern throughout the landscape.

Few studies exist that demonstrate the effectiveness of corridors (e.g., Fahrig and Merriam 1985, Henderson et al. 1985, Soule et al. 1988). However, biological intuition suggests that vegetation between HCAs similar to that within the HCAs will enhance the survival of dispersing individuals (Fahrig and Merriam 1985, Noss 1987, Murphy and Noon 1992).

A beach buffer, at least 500 feet wide, should be maintained wherever the coastline is forested. Old-growth riparian buffers are critical for brown bears, act as corridors, and are also assumed to aid in the dispersal of old-growth associated species. Additional biological corridors may need to be designated during project level analyses to assure sufficient movement of old-growth associated species between HCAs. Breaks in old-growth travel corridors should not exceed 65 ft to ensure that flying squirrels can glide across the openings.

Management Within HCAs and Travel Corridors

Standards:

1.) Harvesting of old-growth timber should not be permitted within designated areas, unless an alternate HCA or travel corridor is first designated elsewhere which would provide the same ecological function for brown bears, marten, goshawks, and boreal owls.

2.) Harvesting of existing second growth forests may be permitted within designated HCAs if new roads are not constructed and existing roads are closed to general public access.

3.) Salvage harvesting of downed or dead trees is permitted only in the case of catastrophic events larger than 100 ac if:

i. salvaging is accomplished without new roads; and

ii. all standing living trees are left uncut, except as necessary for safety.

4.) Roads should be located outside of HCAs and old-growth travel corridors, except where no other reasonable and prudent routing alternative exists.

If roads must be built in such areas, clearing widths should be kept to a minimum and roads should be closed to general public access, except when needs to keep roads open are identified through environmental analyses. A wildlife biologist should be consulted to evaluate routing alternatives and should assist with locating the road.

Habitat capability of a HCA that is lost to road construction or disturbance as a result of road use should be replaced by increasing the size of the HCA. If this is not possible because of the absence of suitable habitat, then the habitat capability loss should be compensated in the nearest HCA.

ADDITIONAL STANDARDS FOR SPECIES OF CONCERN

The above standards for size and distribution of old-growth tracts form the core of the conservation strategy for most species associated with old-growth forest habitat. However, several species-specific standards are also necessary to ensure that viable, well-distributed populations exist for species with identified concerns.

Northwestern Great Blue Heron (Schenck and Suring 1992a)

Active nests of northwestern great blue herons are rare in southeast Alaska (<10 locations ever reported). Nests are considered to be active if breeding, nesting, or fledging activities are known to have occurred within the previous 2 years, after which it can be assumed the site will probably not be reused.

1.) Within 2 years following known nest activity, development (e.g., timber harvest, road, campground, or trail construction) is not allowed within 1/8 mi of heron nests during the nesting season (1 March - 31 July).

2.) Forest Service permitted aircraft flights are not allowed within 660 feet in elevation within 1/4 mi of an active nest from 1 March to 31 July.

Vancouver Canada Goose (Iverson 1992)

1.) Establish 1,000-ft buffers of old-growth forest adjacent to estuaries throughout the Tongass National Forest.

2.) Maintain or enhance the current habitat capability of Vancouver Canada geese on the Yakatat Ranger District because the population is low and disjunct in this area.

Queen Charlotte Goshawk (Crocker-Bedford 1992)

Protect individual pairs of Queen Charlotte goshawks wherever they are found.

1.) Inventory proposed timber sale areas for active goshawk use prior to, or as part of, EA's or EIS's for timber sales.

2.) If a nesting territory (as evidenced by the nest itself, defensive adults, fledglings, or frequent sightings of foraging birds between May and August) is discovered, a committee of biologists should identify their best estimate of the pair's home range (generally about 5,000 ac) and the 1,600 ac core area of the male within the home range. Vegetational disturbance should not be permitted within the core area of the male, except roading and recreational development may occur over 1/4 mi from nest. Outside of the 1,600 ac core area, but within the home range, no more than 5% of the productive forest land should be harvested in a decade (including road construction). Harvesting should be proportional to the occurrence of volume classes or emphasis should be placed on the lower volume classes.

Boreal Owl (Suring 1992a)

1.) Nest sites located outside of HCAs should be protected by a 2600-ft buffer of old-growth forest where ground disturbing activities, including logging, would not be allowed.

Northern Hawk Owl (Suring 1992c)

1.) Implement a forest-wide snag management policy in association with timber harvest which ensures the continued presence of snags in clearcuts and second growth forests. This policy will help to provide nest sites and perches for this species.

Alexander Archipelago Wolf (Kirchhoff 1992)

1.) Where roads are accessible to medium or large-sized communities (i.e., ferry and/or road access to communities greater than 1000 people), open road density should not exceed 1 mi per mi^2 within a Wildlife Analysis Area (WAA) where wolves occur. Roads which are closed and made unusable for motorized traffic by administrative closure and gating, ditching, or barricading after timber harvest should not be included in calculating open-road density. Because the marine coastline provides access to wolves that is comparable to road access, the coastline accessible by skiff should be added to open road length when computing road density. In WAAs that adjoin wilderness or roadless areas of greater than 40,000, road densities of up to 1.25 mi per mi^2 may be allowed.

2.) Habitat capability necessary to provide for equilibrium populations of predators and prey should be maintained wherever possible. As a general rule, sufficient habitat capability for deer should be maintained to support at least 5 deer per mi^2 where deer are the primary prey item for wolves (i.e., on most islands and the southern half of Cleveland Peninsula).

Brown Bear (Titus and Schoen 1992)

1.) Bear-human conflicts should be minimized through careful waste management. Food and solid waste should be handled and disposed of using appropriate and approved methods (e.g., State of Alaska Department of Environmental Conservation, U.S. Environmental Protection Agency) to minimize attracting bears. Fuel-fired incineration should be required in all communities and permanent and seasonal camps.

2.) Bear-human conflicts are minimized by keeping people away from bears. Seasonal and permanent camps, mineral exploration and operational facilities, log dumps and transfer facilities should be located more than 1 mi from sites of seasonal brown bear concentrations to the extent possible.

3.) Operating plans for mineral exploration and development, concessionaire special use permits, and timber/road construction contracts should include specific plans for protecting brown bear habitat and reducing bear-human conflicts. Exploration and development should be seasonally restricted to avoid times and seasons when bear-human encounters are likely. This should be determined an a case by case basis in consultation with the Alaska Department of Fish and Game.

4.) Roads that must be constructed through HCAs, to access timber or minerals, should be closed except to timber harvest or mineral development operations. Use of motorized vehicles (e.g., cars, trucks, off-road vehicles) within HCAs for brown bear hunting should not be allowed on Admiralty, Baranof and Chichagof islands. Seasonal exceptions may be allowed following appropriate analysis through a committee of biologists.

5.) A minimum of 300-ft buffers (best management practices would be 600 ft) of uncut timber should be retained adjacent to pink salmon (Oncorhynchus gorbuscha) and chum salmon (Oncorhynchus keta) spawning areas that are important feeding areas for brown bears. These important brown bear feeding areas are generally Class I streams with pink and chum salmon runs that are less than 80 ft wide within key watersheds. Specific pink and chum salmon spawning areas requiring the 300-ft buffer will be identified by biologists during project-level planning.

6.) Roads should not be built within 300 ft (ideally 600 ft) of important salmon-bear streams, except as necessary to cross the stream at a nearly perpendicular angle to the stream.

Prince of Wales River Otter (Suring and Larsen 1992)

1.) Forest-wide application of 500-ft old-growth buffers along the marine coastline, 1,000-ft old-growth buffers along estuaries, and riparian old-growth buffers should ensure that forest and mineral management activities do not disrupt the distribution of this endemic subspecies.

Mountain Goat (Schenck and Suring 1992b)

1.) Site-specific project planning should identify cliffs used by mountain goats during critical winter periods and for kidding through pre-project surveys and inventories in conjunction and consultation with ADF&G biologists. The mountain goat habitat capability model should be used to estimate winter mountain goat habitat capability surrounding cliffs identified within the project area. Model results should be verified in the field by biologists. At least 80% of the potential winter habitat capability available to discrete mountain goat populations, as determined by the habitat capability model, should be maintained and protected from disturbance from 1 November to 1 May. Kidding areas should be protected from disturbance from 1 May to 1 August.

2.) Siting of camps, roads, trails, mineral exploration and operational facilities, and log dump and transfer facilities should be located more than 1 mi from sites of critical winter habitat or kidding areas. Alternatively, operating plans could include specific plans for protecting mountain goat wintering habitat and kidding areas and reducing goat-human conflicts through seasonal restrictions that avoid goat-human encounters.

IMPLEMENTING THE STRATEGY - AN EXAMPLE

An important test of the committee's proposed approach was to apply it forestwide over a real landscape. We needed to know, for example, if adequate stands of old-growth forest of high enough quality were available to technically meet the proposed standards where the species of concern occur. As a result we mapped 1 possible layout for the proposed HCAs. The committee decided that for practical purposes, the mapping exercise at the forest-wide scale would include only the Large and Medium HCAs. Mapping of Small HCAs and buffer strips affected relatively small land areas and required more site-specific knowledge. It is therefore proposed that mapping of the Small HCAs be deferred to project-level planning.

In developing the map of the HCAs, we attempted to simultaneously meet the spacing, size, composition, and shape requirements. Some compromises were necessary when all constraints could not be met simultaneously. For example, some areas met the size, spacing, and shape criteria, but did not meet the composition criteria. Other areas had to be made more linear to better meet the composition criteria, or spacings were changed slightly to locate HCAs in areas where composition guidelines could be met. A "perfect" application of this conservation strategy does not exist, but through repeated iterations, using the GIS to supply information on size, spacing, and composition, it is possible to improve the final product. Since this exercise was intended as an example of one way (and not the only way) to lay out these HCAs, we have not invested the effort to do the repeated iterations necessary to reach an optimal solution.

The resultant map (see inside back cover) identifies 40 Large HCAs and 109 Medium HCAs throughout the region. Approximately 25% of these fall within existing wilderness or legislated LUD II (roadless) areas. The total size and composition of each HCA, the ASQ, and the number of tentatively suitable acres (i.e., those eligible for logging) involved are shown in table 6.

There are some areas in which the prescribed HCAs do not meet the draft standards (e.g., on very sparsely forested areas of the mainland). Although old-growth associated species may exist at relatively low numbers in some of these areas, those populations are presumably at greater risk of local extirpation. The recorded presence of an animal in an area, even in occasionally large numbers, cannot be equated with the existence of high-quality habitat (Van Horne 1983) or that the habitat available can support a viable population. This is particularly true for animals that are highly territorial and disperse widely as juveniles (e.g., brown and black bears, martens, goshawks, and wolves).

The small HCAs, which have not been mapped, will also affect some area of tentatively suitable forest land. To quantify that effect we made the following assumptions about the number needed and the area affected. First, small HCAs are not needed where:

1.) the VCU is less than 10,000 ac in size,

2.) there are existing legislated areas (Wilderness and Lud II), and

3.) fewer than 1,000 acres of old-growth forest exist in the VCU.

To the maximum extent possible, we expect these project level allocations to use old-growth forest already protected or unavailable for timber harvest, including beach fringe buffers, estuary buffers, riparian buffers, and nonsuitable timber. Based on the maps the Forest Service provided, on average, we estimate that 20% of old-growth forest required by each Small HCA will have to come from the tentatively suitable timber base. Because each Small HCA requires 800 ac of old-growth forest, approximately 160 tentatively suitable ac will be affected for each small HCA.

VIABILITY/DISTRIBUTION RISK ASSESSMENT

One of the general precepts of conservation biology is that small, isolated populations which result from habitat fragmentation face higher risks of maintaining their viability and distribution than large, interacting populations (Iwasa and Mochizuki 1988, Suring and Crocker-Bedford 1992).

Current and proposed management strategies for maintaining viable populations that are well distributed on the Tongass National Forest were evaluated using the following criteria to provide perspective among strategies (Thomas et al. 1990) (Table 7):

1.) habitat tracts are of sufficient size and high enough quality to ensure occupancy and high rates of reproduction,

2.) habitat tracts are close enough together and large enough to ensure recolonization following extirpation in habitat tracts, and

3.) habitat tracts are distributed across the landscape to ensure distribution of species throughout their range on the Forest.

The current Tongass Land Management Plan provides for the retention of over 8% of lands outside of designated wilderness and other lands not available for timber harvest for wildlife and fish habitat and visual management (USDA Forest Service 1979). However, guidance was not provided in the Plan on the size, distribution, or quality of habitats to be retained. A procedure was also not established to locate and designate specific areas. These conditions led to an assessment of low likelihood for maintaining viability and distribution under the current Forest Plan.

The draft Environmental Impact Statement for the revision of the Tongass Land Management Plan called for maintaining 24% of the forest area of each Wildlife Analysis Area (WAA) in an old-growth condition (USDA Forest Service 1990). At least 1 tract of old-growth habitat in each WAA was to be 5,000 ac or larger;

75% of the designated old-growth habitat was to be in tracts 1,000 ac or larger. This management strategy assured distribution of habitats across the Forest. However, it was not established that the size and spacing of the old-growth tracts would assure viability and maintain distribution of the species across the Forest. This management strategy was judged to have a moderate likelihood of maintaining viable populations of old-growth wildlife species distributed throughout their range in southeast Alaska.

The Supplement to the draft Environmental Impact Statement for revision of the Tongass Land Management Plan also proposes a strategy for maintaining viable, well-distributed populations on the Forest (USDA Forest Service 1991). That strategy suggests that habitat within designated Wilderness/Monuments and other areas where timber harvest is prohibited by legislation will assure continued viability and distribution of old-growth associated species throughout the Forest. The discussion of the strategy in that document did not demonstrate that those legislated areas provide high quality habitats, in large enough tracts and in close enough proximity across the landscape to ensure that viable populations will continue to be well distributed across the Forest over the long term (i.e., 100 years). To some extent these areas serve to maintain viable populations of wildlife well distributed across the Forest. However, because they were not planned with that specific need in mind, some areas fail because they are mostly rock and ice, and contain very little productive habitat. Other areas provide productive habitat but do not provide for distribution across the Forest. This is of particular concern on those areas of the Forest with few, or without any, legislatively protected areas (e.g., north Prince-of-Wales Island, northeast Chichagof Island). This strategy was assessed as having a very low likelihood of maintaining viability and distribution.

The initial management strategy drafted by this committee provided for the maintenance and distribution of high quality old-growth forest habitats across the Forest (Crocker-Bedford et al. 1991). The strategy was based on conservative assessments of the habitat needs and dispersal abilities of the species that were evaluated. When the strategy was applied to the Forest it became evident that in some areas the standards protected habitat beyond what was assumed to be needed to maintain viability and distribution. That approach was assessed as having a very high likelihood of maintaining viability and distribution of old-growth wildlife on the Forest.

The management standards proposed in this document are based on the work of an interagency committee of biologists most familiar with the species and habitat conditions in southeast Alaska. The reviews conducted during this effort established that the viability or distribution of several species may be threatened within the next 100 yrs, or sooner, unless specific management actions are implemented. Information from the literature, interim results of on-going research, and professional judgement were used to develop a set of proposed management standards. If these proposed standards are implemented, the committee believes that there is a high likelihood that viable, well-distributed populations of species associated with old-growth forests will be maintained on the Tongass National Forest.

INFORMATION NEEDS

The approach presented here and the design suggested for HCA composition and distribution in southeast Alaska may not be the only solution to the conservation of wildlife associated with old-growth forests. This conservation strategy should be considered a series of hypotheses that have been constructed from information on the distribution, abundance, habitat relationships, and natural history of wildlife species. While the committee has confidence that implementation of this conservation strategy will result in the maintenance of viable, well-distributed populations of wildlife on the Tongass National Forest, we also believe that these hypotheses require additional testing and evaluation. The structure and components of this strategy should be tested more thoroughly with statistical analysis of empirical data, predictions from ecological theory and population models, and inferences drawn from studies of related species (Murphy and Noon 1991). Implementing the process suggested by Murphy and Noon (1991) will allow adjustment of the structure of this conservation strategy, where necessary, to reshape and strengthen it.

The reviews associated with this effort documented the paucity of information concerning critical habitat and population factors upon which a relatively risk-free conservation strategy should be based. Research, administrative studies, and monitoring efforts should be directed toward verifying and improving this conservation strategy. Additional information is needed on:

1.) the minimum number of reproductive pairs that should be supported within a tract of old-growth forest for it to be considered a functional component of the species' habitat,

2.) the vegetation and structural characteristics required within tracts of old-growth forest for them to function as habitat for species of concern,

3.) the dispersal capabilities of the species of concern, and

4.) the size and distribution of habitat tracts and corridors necessary to assure viable, interacting subpopulations of wildlife throughout their range.

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Table 1. Criteria used to evaluate species for viability and distribution concerns in southeast Alaska.

	Criteria	Weighting Factor
1.	Breeding habitat occurs in southeast Alaska	1
2.	Essential winter range occurs in southeast Alaska	3
3.	Essential migratory range occurs in southeast Alaska	2
4.	Habitats are vulnerable to land management activities	4
5.	Habitats are vulnerable to catastrophic events	4
6.	Potential exists for inbreeding depression	5
7.	High potential exists for local extripation	5
8.	Capability to disperse is limited or barriers to dispersal exist	5
9.	Geographic distribution is limited within southeast Alaska	4
10.	Geographic distribution is limited to southeast Alaska	3
11.	Geographic distribution is limited outside southeast Alaska	2
12.	Level of knowledge about the species in southeast Alaska is limited	3
13.	Demographic characteristics of the species (e.g., natality and mortality rates) indicate slow rates of increase in the population	3
14.	Size of the population in southeast Alaska is relatively low	3
15.	Size of the population outside southeast Alaska is relatively low	4
16.	Population trend in southeast Alaska is down	3
17.	Population trend throughout the species range is down.	4

Table 2. Species associated with old-growth forest habitats that are recognized as having potential viability and/or distribution concerns in southeast Alaska.

Common Name	Scientific Name
Northwestern great blue heron	*Ardea herodias fannini*
Vancouver Canada goose	*Branta canadensis fulva*
Queen Charlotte goshawk	*Accipiter gentilis laingi*
Boreal owl	*Aegolius funereus richardsoni*
Hawk owl	*Surnia ulula caparoch*
Alexander Archipelago wolf	*Canis lupus ligoni*
Brown bear	*Ursus arctos horribilis*
Marten	*Martes americana*
Prince of Wales river otter	*Lutra canadensis mira*
Mountain goat	*Oreamnos americanus columbiae*
Flying squirrel	*Glaucomys sabrinus*

Table 3. Level of concern associated with viability and/or distribution for 11 species in southeast Alaska.

Species	Evaluation Criteria[a,b]																
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
Northwestern great blue heron	H	L	N	H	H	-	L	L	N	L	L	M	L	M	L	-	N
Vancouver Canada goose	H	H	N	M	L	N	N	L	L	M	M	H	L	M	M	-	-
Queen Charlotte goshawk	H	M	L	H	L	M	H	L	M	H	H	H	H	H	H	H	H
Boreal owl	H	H	N	H	M	N	L	M	L	L	L	H	L	H	M	-	-
Hawk owl	M	L	N	M	L	N	L	L	L	L	L	H	L	-	-	-	-
Alexander Archipelago wolf	M	M	N	H	N	M	M	M	H	H	H	H	L	M	M	-	-
Brown bear	H	H	N	H	N	L	M	M	M	M	M	M	H	M	M	L	H
Marten	H	H	N	H	L	L	H	M	L	L	L	H	L	M	H	L	H
Prince of Wales river otter	H	H	N	M	L	L	M	M	H	H	H	H	L	L	M	L	L
Mountain Goat	H	H	N	H	H	H	H	H	M	M	M	M	M	M	M	L	L
Prince of Wales flying squirrel	H	H	N	H	M	M	M	H	H	H	-	H	L	-	-	-	-

[a]See Table 1 for a description of the evaluation criteria.

[b]Level of concern: H = high concern (3) L = low concern (1)
M = moderate concern (2) N = no concern (0)
- = information not adequate for a rating (2)

Table 4. Ranking of concern associated with viability and/or distribution of 11 species in southeast Alaska.

Species	Rank of Concern by Evaluation Criteria[a]																	
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	Total
Queen Charlotte goshawk	3	6	2	12	5	10	15	5	8	9	6	9	9	9	12	9	12	141
Prince of Wales flying squirrel	3	9	0	12	8	10	10	15	12	9	4	9	3	6	8	6	8	132
Mountain goat	3	9	0	12	12	15	15	15	8	6	4	6	6	6	8	3	4	132
Alexander Archipelago wolf	2	6	0	12	0	10	10	10	12	9	6	9	3	6	8	6	8	117
Marten	3	9	0	12	4	5	15	10	4	3	2	9	3	6	12	3	12	112
Brown bear	3	9	0	12	0	5	10	10	8	6	4	6	9	6	8	3	12	111
Prince of Wales river otter	3	9	0	8	4	5	10	10	12	9	6	9	3	3	8	3	4	106
Boreal owl	3	9	0	12	8	0	5	10	4	3	2	9	3	9	8	6	8	99
Vancouver Canada goose	3	9	0	8	4	0	0	5	4	6	4	9	3	6	8	6	8	83
Northwestern great blue heron	3	3	0	12	12	10	5	5	0	3	2	6	3	6	4	6	0	80
Hawk owl	2	3	0	8	4	0	5	5	4	3	2	9	3	6	8	6	8	76

[a] See Table 1 for a description of the evaluation criteria.

Table 5. Criteria for Habitat Conservation Areas proposed to maintain viable and well distributed populations of wildlife associated with old-growth forests in southeast Alaska.

Species	Habitat Conservation Areas: Large	Medium	Small	Source
Brown bear	40,000 ac 20 mi apart 1 Class I stream (5 females)	--	--	Titus and Schoen 1992
Marten	40,000 ac 25 mi apart 50% vc 4+ 25% vc 5+ (25 repo. units)	8,000 ac 9 mi apart 50% vc 4+ 25% vc 5+ (5 repo. units)	1,600 ac per watershed 50% vc 4+ (1 repo. unit)	Flynn 1992
Boreal owl	--	5,000 ac 10 mi apart vc 4+ (9 pairs)	--	Suring 1992a
Flying squirrel	--	--	1,000 ac per watershed vc 4+ (10-20 pairs)	Suring 1992b
Goshawk	40,000 ac 20 mi apart 50% vc 4+ 25% vc 5+ (8 pairs)	10,000 8 mi apart 50% vc 4+ 25% vc 5+ (2 pairs)	--	Crocker-Bedford 1992
Combined standard	40,000 ac 20 mi apart 50% vc 4+ 25% vc 5+ 1 Class I stream	10,000 ac 8 mi apart 50% vc 4+ 25% vc 5+	1,600 ac per watershed 50% vc 4+	

Table 6. Characteristics of Habitat Conservation Areas (HCA) proposed for the Tongass National Forest in southeast Alaska.

HCA Number	Percent of Volume Class 4	Percent of Volume Class 5+	ASQ	Acres of Tentatively Suitable Forest Land	Total Area (acres)
Large HCAs					
3	22	46	3419470	14695	47207
7	23	47	9326	40	59027
8	30	21	3526753	15156	38646
11	40	15	3375457	14506	41683
15	40	33	0	0	52052
19	29	30	9303	40	34381
23	23	24	88397	380	48715
25	25	29	0	0	53850
28	23	31	298864	1284	45613
38	28	22	0	0	56436
41	29	41	3712033	15952	32918
42	38	16	0	0	62078
45	22	14	576393	2477	50127
46	20	26	2476905	10644	30296
48	31	26	4112054	17671	38221
52	26	18	3122102	13417	41123
56	27	30	3348170	14388	40635
57	20	6	704086	3026	37930
61	31	27	3094739	13299	34553
65	24	25	9317	40	58011
76	23	38	65478	281	40679
81	22	20	27934	120	38574
83	26	20	3634345	15618	38327
84	30	15	0	0	49052
88	21	37	4757731	20446	39011
89	25	32	4346280	18677	38572
93	15	42	1112002	4779	40268
95	11	45	23261	100	33310
96	11	48	9305	40	40049
106	13	53	2094760	9002	39347
119	18	42	1691776	7270	40319
121	22	57	1047052	4500	42386
129	23	12	0	0	43870
132	25	41	0	0	51855
133	36	30	0	0	63790
137	28	41	0	0	69032
138	26	13	2526901	10859	39157
151	8	43	265657	1142	34318

Table 6. Characteristics of Habitat Conservation Areas (HCA) proposed for the Tongass National Forest in southeast Alaska - continued.

HCA Number	Percent of Volume Class 4	Percent of Volume Class 5+	ASQ	Acres of Tentatively Suitable Forest Land	Total Area (acres)
Medium HCAs					
2	13	20	726246	3121	11642
4	42	21	1000604	4300	10520
5	32	23	1154171	4960	9680
6	24	47	1225187	5265	9815
9	23	33	1163814	5001	10743
10	28	15	842402	3620	15403
12	38	19	1032223	4436	10129
13	23	38	1039133	4466	9912
14	25	32	1482124	6369	9594
16	27	21	1115323	4793	10265
17	27	18	1014680	4360	9861
18	29	36	23226	100	6655
20	28	30	973546	4184	10185
21	33	6	566433	2434	10054
22	15	77	1586898	6819	9088
24	28	10	533554	2293	16971
26	17	27	1149775	4941	8982
27	21	21	27790	119	9698
29	14	26	782036	3361	9902
30	15	36	1049070	4508	7913
31	26	13	0	0	14372
32	31	34	1099832	4726	9813
33	15	40	1549204	6657	12131
34	19	32	130804	562	8533
35	21	16	667038	2867	8894
36	29	26	1163660	5001	10861
37	29	38	879599	3780	6520
39	18	19	339473	1459	7714
40	25	20	0	0	15918
43	29	10	0	0	19052
44	12	20	400268	1720	9979
47	12	31	674273	2898	9952
49	32	2	547761	2354	9580
50	48	5	756447	3251	9014
51	13	32	764273	3284	9553
53	27	19	793595	3410	9649
54	27	33	962363	4136	10323
55	28	39	1360141	5845	12926
58	13	6	181534	780	10322
59	22	23	474739	2040	9224
60	20	12	564075	2424	10604
62	26	45	1098059	4719	10234

Table 6. Characteristics of Habitat Conservation Areas (HCA) proposed for the Tongass National Forest in southeast Alaska - continued.

HCA Number	Percent of Volume Class 4	Percent of Volume Class 5+	ASQ	Acres of Tentatively Suitable Forest Land	Total Area (acres)
63	29	35	4651	20	10184
64	38	28	1277273	5489	10960
66	20	26	907354	3899	10778
67	25	20	987791	4245	12854
68	19	22	692896	2978	12450
69	23	24	725874	3119	8959
70	17	67	1403762	6032	9417
71	24	16	679964	2922	10567
72	25	35	1191610	5121	10242
73	33	13	1119635	4811	11718
74	24	29	567735	2440	11979
75	31	30	688887	2960	7701
77	28	9	344537	1481	12265
78	28	19	926147	3980	9540
79	28	24	558616	2401	11382
80	39	13	461117	1982	9668
82	33	14	847211	3641	10475
85	22	45	1498583	6440	9600
86	51	6	4667	20	7990
87	33	30	1008604	4334	11147
90	25	38	1146417	4927	9332
91	18	26	0	0	12034
92	27	44	1495149	6425	8935
94	26	22	0	0	9761
97	27	24	1265052	5436	11272
98	22	25	950034	4083	9987
99	29	40	1174008	5045	10711
100	20	16	822652	3535	10306
101	19	30	841970	3618	10255
102	15	17	487135	2093	8035
103	31	37	1066340	4582	8123
104	33	23	0	0	14820
105	30	37	1361114	5849	9654
107	19	55	1398510	6010	9916
108	29	24	1016251	4367	9656
109	22	26	776297	3336	7851
110	21	39	1173966	5045	10391
111	22	31	967051	4156	10317
112	18	26	520137	2235	5548
113	22	28	1096670	4713	10678
114	33	27	806966	3468	11761
115	25	4	0	0	12059
116	32	28	1166762	5014	10610
117	25	53	0	0	6837

Table 6. Characteristics of Habitat Conservation Areas (HCA) proposed for the Tongass National Forest in southeast Alaska - continued.

HCA Number	Percent of Volume Class 4	Percent of Volume Class 5+	ASQ	Acres of Tentatively Suitable Forest Land	Total Area (acres)
118	37	29	875151	3761	13304
120	25	24	23273	100	12389
122	31	23	0	0	9560
123	30	21	0	0	18611
124	23	26	404352	1738	10695
126	23	32	0	0	12512
127	29	28	1246435	5356	11312
128	14	40	0	0	13654
130	25	9	0	0	15735
131	20	46	0	0	17906
134	17	59	0	0	7457
135	11	2	0	0	13622
136	18	9	0	0	13143
139	32	50	0	0	15608
140	27	18	520340	2236	11160
141	25	43	0	0	17854
142	23	47	0	0	9129
143	24	45	0	0	24534
144	17	6	0	0	12386
145	39	12	0	0	6969
146	24	12	0	0	20086
147	21	4	0	0	14491
148	22	16	0	0	15460
149	38	16	583091	2506	10032
150	23	15	0	0	13485
Average	25	27			
Sum			125463261	539160	2917415

Table 7. Likelihood of maintaining viability and distribution of wildlife associated with old-growth forests on the Tongass National Forest under current and proposed management strategies over the long term (i.e., 100 years) (categories are from Thomas et al. 1990).

Strategy	Viability and Distribution Assessment	Source
Tongass Land Management Plan	Low	USDA Forest Service 1979
Tongass Land Management Plan Revision - Draft EIS	Moderate	USDA Forest Service 1990
Tongass Land Management Plan Revision - Supplement to the Draft EIS	Very low	USDA Forest Service 1991
Conservation strategy incorporating Core-Deme Reserves, HCAs, and management standards	Very high	Crocker-Bedford et al. 1991
Conservation strategy incorporating HCAs and management standards	High	This document

Very High: Continued existence of a well-distributed population on the Forest is virtually assured, even if 1) major catastrophic events occur within the population, 2) research finds that the species is less flexible in its habitat needs or dispersal abilities, or 3) demographic or genetic factors prove to be more significant than assumed in the analysis.

High: Likelihood is high that a well-distributed population will continue to exist on the Forest. Some latitude is allowed for catastrophic events to affect the population or for biological findings that the population is less flexible in its habitat needs or dispersal abilities.

Moderate: Likelihood of continued existence of a well-distributed population is moderate. Limited latitude exists for catastrophic events affecting the population or for biological findings that the population is less flexible in its habitat needs or dispersal abilities.

Low: Likelihood of continued existence of a well-distributed population on the Forest is low. Catastrophic, demographic, or genetic factors are likely to cause elimination of the species from parts or all of its geographic range on the Forest over the long term.

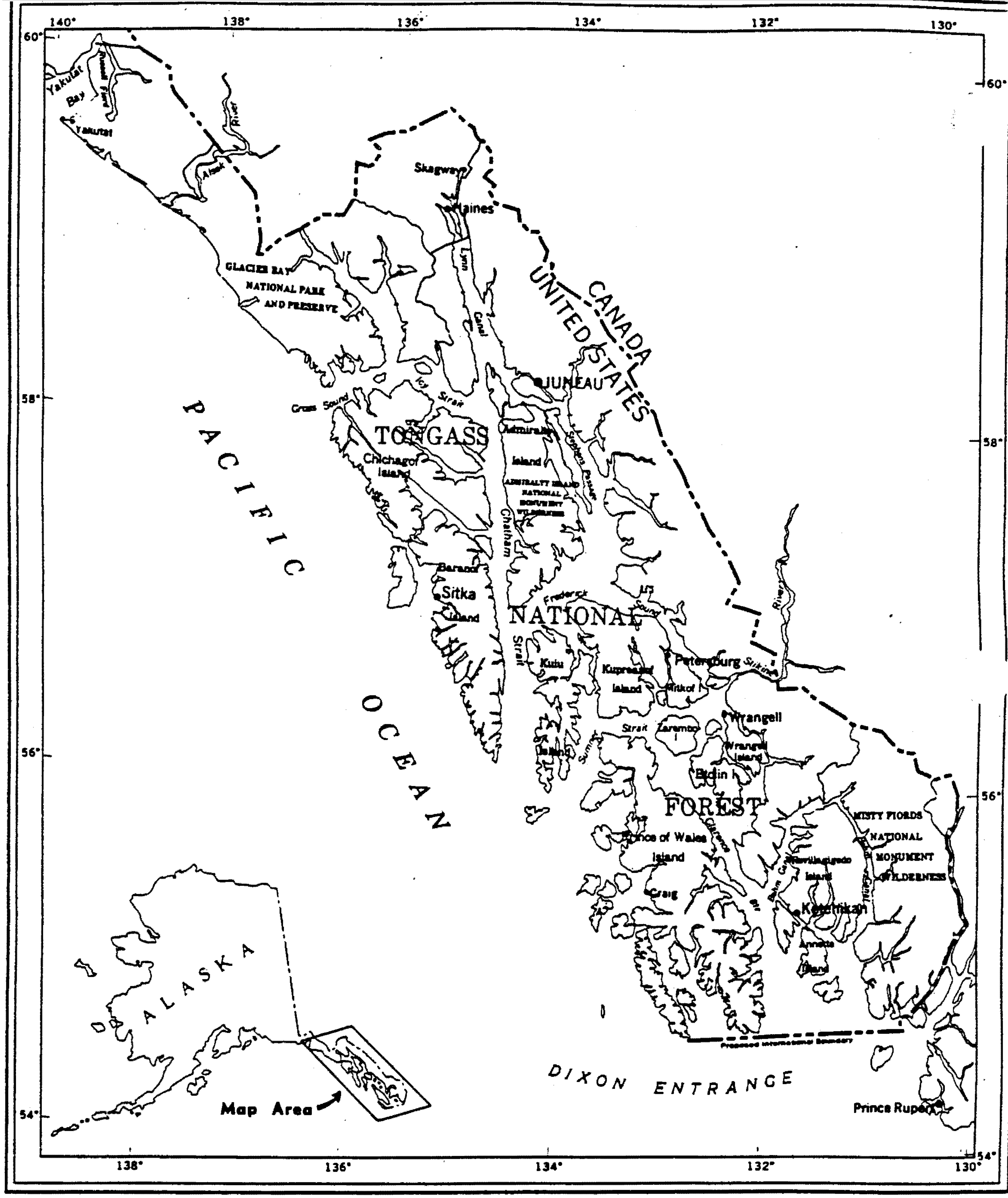

Fig. 1. Map of southeast Alaska.

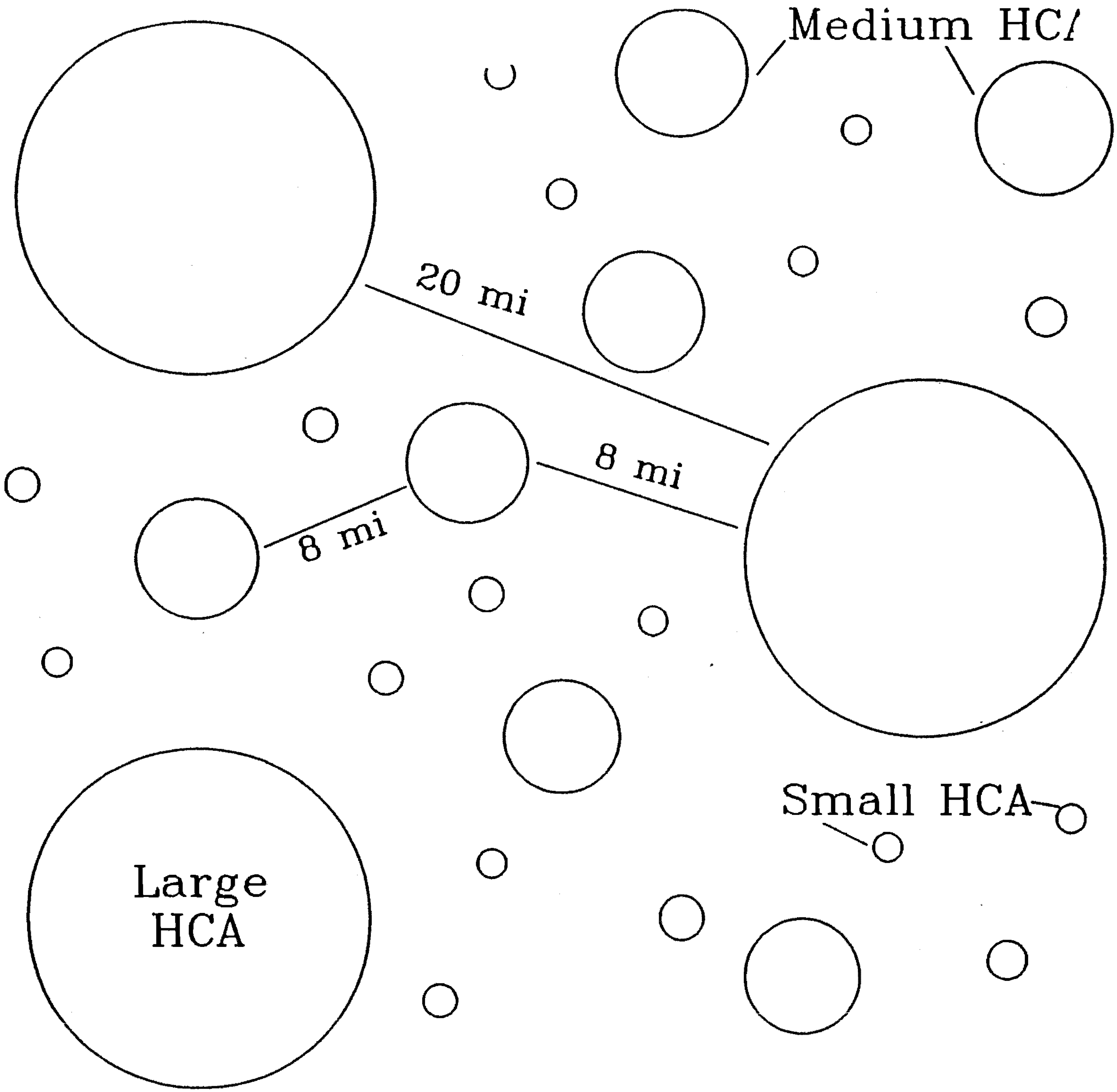

Fig. 2 Proposed Habitat Conservation Areas

APPENDICES

APPENDIX A: CONCEPTS OF CONSERVATION BIOLOGY: AN OVERVIEW

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Conservation biology has emerged as a scientific discipline and as an approach to the management of landscapes (Soule and Wilcox 1980). It brings together the "findings" of pure science and focuses them in applied manner to maintain biological diversity (Thomas and Salwasser 1989). Biological diversity encompasses the whole realm of life from genetic components through complete landscapes (Szaro and Shapiro 1990). One of the primary components of the maintenance of biological diversity is ensuring the continued viability of all species throughout their range. This essentially means that management programs are implemented that ensure that species are not allowed go extinct or are not extripated anywhere they currently occur.

Extinction is more accurately portrayed as a process rather than an event (Wilcox 1986). The process is often initiated by human-induced environmental change (e.g., habitat loss) which causes a reduction in the size, number, and proximity of populations. Stochastic factors associated with population demographics, genetic variability, and environmental variation along with natural catastrophes begin to have critical impacts on population viability and distribution (Shaffer 1981). Individual factors then interact in a mechanism of feedback loops resulting in a downward spiral to extinction.

FRAGMENTATION OF HABITATS

Numerous causes have been identified which lead to the extinction or extirpation of a species (Reid and Miller 1989). However, habitat-related factors were the most common factors reported that were related to classifying species as threatened or endangered (Hayes 1991). Fragmentation of habitats has been characterized as one of the primary causes of extinction of animals (Terborgh and Winter 1980, Wilcox and Murphy 1985). Fragmented habitats are restricted in size and surrounded by a landscape modified from the original and often of little use to the species in question (Wilcove 1987). The original habitat may be modified in either composition or structure (Thomas et al. 1990). The significant point is that the habitat modifications function as a partial or complete barrier to dispersal for species in the original habitat.

The extinction/extirpation process associated with habitat fragmentation may be categorized as follows: 1) the loss of species that were accidentally excluded from the fragment when the fragment was created; 2) the loss of species for which the fragment is not acceptable habitat any longer; 3) the loss of species that do reproduce successfully in the habitat fragment, but which occur as small populations; and 4) the loss of species because of ecological imbalances in the fragments (Wilcove and Wilcox 1986). The following discussion of these 4 categories is taken from Wilcove and Wilcox (1986) and Wilcove (1987).

Initial Exclusion

Because habitats are usually heterogenous and are not uniformly distributed across the landscape (especially in southeast Alaska) species associated with particular habitats are not uniformly distributed. When a landscape is fragmented the pieces of habitat that remain will contain an incomplete sample of all the species that were indigenous to the larger block. It has been suggested that because of in complete sampling a 10-fold decrease in the size of natural habitats often leads to a reduction of 30 to 50% of the species initially present (Wilcox 1980, Diamond and May 1981).

The most effective way of approaching this problem is to use site-specific information on the distribution of species to ensure that remaining habitat encompasses all species. However, it is rare (again, especially in southeast Alaska) that such information is available. The initial exclusion problem can also be addressed, to an extent, by planning to have numerous, large fragments. The more habitat patches there are available, the higher the probability that all species will be included. Also, the larger the patches are, the more species will be included (i.e., the species-area relationship) (MacArthur and Wilson 1967, Diamond and May 1981).

Unacceptable Habitat

When landscapes are fragmented some of the patches will probably be smaller than the minimum home ranges or territories of some species. The survival of these species will depend on maintaining large patches of habitat.

Fragmentation of habitats also carries the risk of losing habitat heterogeneity. An apparent uniform landscape (e.g., Sitka spruce-western hemlock forest) is really a mosaic of many different microhabitats. Individual patches within a fragmented landscape may not have the full range of microhabitats that were originally present. Although affected species may occur within the habitat patches they may not be able to persist because all aspects of their habitat may not be present (Lynch 1987).

Possible approaches to this problem include designing the remnant patches for the needs of the most area-sensitive species (Wilcox 1984, Hayden et al. 1985). Meeting the habitat/area requirements of such species (e.g., marten) may also satisfy the habitat/area needs of many species with smaller home ranges. The suggestion has also been made that conservation strategies should be directed toward those species whose populations are vulnerable to habitat modifications (Burgman et al. 1988). These strategies should be related to the probabilities of extinction for the individual species under specific management practices and environmental conditions. Such an approach requires information on the environment, the demographics and genetics of the population, and the effect of management activities on these factors.

Small Populations

Many species will persist in habitat fragments. However, they will exist as small, isolated populations. Such populations may be vulnerable to 1) natural catastrophes, 2) environmental fluctuations, 3) imbalances in sex ratios and/or age distribution, 4) genetic deterioration, and 5) social dysfunction.

Small populations are much more susceptible to the effects of natural catastrophes such as winter storms, droughts, temporary food shortages, or disease epidemics. A population associated with habitat fragments must survive such events with not only enough individuals but also the right mixture of sexes and the right age distribution. Chance events, such as periods of poor reproduction or excessive mortality of a specific sex or age class may skew sex ratios or age distributions enough so that reproduction is impaired. Small, isolated populations associated with patches of habitat may also suffer from inbreeding and lose their genetic variability through chance events. This may lead to reduced fertility, the development of deleterious traits, or the inability to change with environmental conditions (Allendorf and Leary 1986, Ralls et al. 1986). Some species may require the social interaction provided by large numbers of individuals in order to breed. If such social interaction is not possible because not enough individuals are in proximity to one another in the habitat fragments breeding may be impaired.

An obvious solution to the potential problems associated with small populations is to maintain populations that are large enough that they have little vulnerability to extinction or extirpation. Economic considerations often preclude this approach. Instead, the habitat needs of species that are able to disperse readily may be met by establishing a network of habitat patches placed well within the species dispersal distance of each other (Diamond 1985). This approach must include recognition that populations associated with individual habitat patches will experience periodic extinctions. McLellan et al. (1986) have suggested that isolation of habitat patches may lead to extinction of species independent of reduction of habitat size. However, interchange between populations in the patches will allow the species to persist somewhere in the network (Lynch and Whigham 1984). Habitat patches that are far apart relative to dispersal distances need to be large enough to have high persistence rates; smaller patches must be located closer together to ensure high immigration rates (Lomolino 1986). In general, small, isolated patches tend to permanently lose any residual population. Populations rarely become extinct in large patches. Small and intermediate patches located near other patches experience extinctions but tend to be recolonized repeatedly (Urban and Shugart 1986). Also, immigration that occurs before extinction provides a rescue effect that helps maintain required numbers, population characteristics, and genetic diversity.

Species that do not disperse readily across altered habitats may require the maintenance of corridors between the habitat patches (Harris 1984, Fahrig and Merriam 1985). The modification or conversion of natural habitat will more probably impede dispersal than will isolation caused by unsuitable conditions in the natural landscape (Wilcox and Murphy 1985). It is critical in this approach that the habitat patches be numerous enough and close enough together to ensure that the species of interest will be able disperse throughout the network (Bennett 1987, Swanson et al. 1990). If each habitat patch is large enough to support a breeding population of the species of interest, and if extinction rates in each habitat patch are low in relation to recolonization rates, then a network approach can function to maintain species across their range (Schmiegelow and Nudds 1987, Templeton et al. 1990). However, it should be noted that small populations (i.e., <50 individuals) face the high probability of rapid, localized extinctions (e.g., <50 years for bighorn sheep [Ovis canadensis]) (Berger 1990). So efforts should be made to maintain habitat patches as large as possible. It is also critical that when habitat patches are selected for maintenance that the habitat within the patch be evaluated for its suitability for the species of interest (Soule and Simberloff 1986, Loyn 1987).

Ecological Imbalances

Fragmentation may also disrupt ecological interactions in an affected landscape. The loss of one species from a habitat fragment (e.g., prey species) may result in the loss of other species (e.g., predator species). Ecological imbalances are most likely to occur when large, long-lived species with dominant roles in the ecosystem (e.g., predators) are removed (Terborgh 1988). Fragmentation may enhance habitat opportunities for potential competitors (e.g., red-tailed hawk) over species of interest (e.g., goshawk). The increased area of forest edges resulting from fragmentation alters the climate, vegetation, and animal life of extensive areas of the habitat patches. These changes in turn, may seriously affect the ability of species associated with forest interiors to persist (Wilcove et al. 1986). As a landscape matrix is modified so that the previous dominant habitat is largely replaced by another, competing species produced in the new habitat may become so abundant that they even occupy niches remaining in residual habitat patches.

These concerns may be partially addressed by: 1) maintaining the integrity of habitat patches (e.g., minimize habitat disturbance), 2) maintaining as large a patch as possible, and 3) avoid irregular shapes of patches (i.e., circular shapes are preferred).

DEVELOPMENT OF HABITAT CONSERVATION STRATEGIES

Species that may be placed at risk as a result of habitat modification and associated fragmentation may be maintained through conservation planning. A conservation strategy that maintains habitat across the landscape similar to its historical distribution is the best approach to minimizing the risk of extinction (Thomas et al. 1990). Thomas et al. (1990) provided general guidelines for the development of a conservation strategy based on the work of Diamond (1975), den Boer (1981), Harris (1984), Noss and Harris (1986), and Wilcove et al. (1986). Those guidelines follow:

- Habitat patches should be dispersed in a pattern corresponding to the species geographic distribution to minimize the risk of extinction.

- Large blocks of habitat are better than small ones.

- Blocks of contiguous habitat are better than loose aggregations of fragmented blocks.

- Blocks closer together are better than blocks far apart (i.e., distance between blocks must be well within dispersal capabilities of species in question).

- Habitat between blocks should be suitable for movement and stopovers by the species under consideration to facilitate movement (dispersal) among blocks.

- Patches must contain habitat of high enough quality to support the species of concern even during unusual environmental events (e.g., severe winters, temporary loss of food sources).

- The total area of habitat should be divided into as few patches as possible, but consideration must also be given to distributing the patches widely over the species' range.

- Separate patches of habitat should be grouped equidistant from each other in contrast to a linear distribution.

- Habitat patches should be as nearly circular as possible to minimize internal dispersal distances and edge effects.

A conservation strategy that incorporates these guidelines will provide multiple, extensive, and continuous areas of suitable habitat (Thomas et al. 1990). Those areas of habitat will be distributed across the landscape so that interaction between them commonly occurs. The landscape features between the habitat patches will facilitate interchange among patches. Such a conservation strategy will have a high potential of maintaining the species in question throughout their range.

MANAGING FRAGMENTATION IN SOUTHEAST ALASKA

The potential effects of habitat fragmentation are magnified in southeast Alaska because of the natural island ecosystem. Even the mainland portion of the area is essentially an island that is separated from the rest of the continent by glaciers, ice fields, and mountain ranges. The formation of endemic species or subspecies is common in island systems such as southeast Alaska. Unfortunately, endemic island species are more prone to extinction than those in continental systems (Temple 1985). Probability of extinction is increased because populations of endemic island species are usually smaller than populations of continental species and have smaller geographic ranges. Endemic island species often exist closer to population levels necessary to maintain viability over the long term. Island populations may drop below this level with a much smaller percentage of loss than continental species.

Additionally, endemic island species, especially birds, typically have low intrinsic rates of growth (Temple 1985). This characteristic reduces a population's ability to recover following a reduction in numbers making it more vulnerable to extinction. Island species also develop in a more stable environment that that found in continental systems. As a result, island species become narrowly adapted for a specific set of environmental conditions. When those conditions are changed (e.g., through timber harvest) the species is less able to adapt to the new landscape.

Management of natural resources under a concept of multiple use involves consideration of varying levels of production of timber, minerals, recreation, wildlife, and other resources, each of which may have a different effect on the placement of habitat patches and corridors on the landscape (Blake and Karr 1984). The changing habitat mosaic will have an effect on the demographics of species in the habitat patches, on their dispersal ability, and subsequently on the risk of extinction for the whole population. It is important, therefore, that each management scenario be evaluated closely to determine its effects on the spatial pattern of the landscape (e.g., Franklin and Forman 1987).

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APPENDIX B

Species Accounts and

Individual Conservation Strategies

CONSERVATION OF THE NORTHWESTERN GREAT BLUE HERON IN SOUTHEAST ALASKA

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POPULATION STATUS AND DISTRIBUTION

The great blue heron (Ardea herodias) occurs from southeast Alaska across southern Canada to Nova Scotia south to the West Indies, Mexico, and Galapagos Islands (Campbell et al. 1990). The majority of this range is occupied by 6 subspecies (Palmer 1962). The Northwestern great blue heron (A. h. fannini) breeds along the Pacific coast from Washington State north through southeast Alaska. In Alaska, this bird is found only on a narrow strip of coast and associated islands from Dixon Entrance as far north as Cook Inlet (Gabrielson and Lincoln 1959). This subspecies is characterized as generally smaller and darker than other subspecies (Palmer 1962).

Great blue heron populations appear to be increasing throughout their range in the lower 48 states with the largest increases in the east (Robbins et al. 1986). Gabrielson and Lincoln (1959:103) indicated the Northwestern great blue heron was "...a regular but not common permanent resident" throughout its range in Alaska. It may be more common from Wrangell south than in northern southeast Alaska. The Northwestern great blue heron has also been characterized as an uncommon resident of the North Gulf Coast-Prince William Sound area (Isleib and Kessel 1973). Both Taylor (1979) and Armstrong (1980) considered this bird to be an uncommon breeder in southeast and southcentral Alaska. Current population trends of the Northwestern great blue heron are not known.

PATTERNS OF HABITAT USE

Nesting Habitat

The great blue heron is a colonial nesting species (Custer et al. 1980, Simpson et al. 1987, DeGraaf et al. 1991). Colonies of Northwestern great blue herons have been reported to be as large as 183 pairs in British Columbia (Campbell et al. 1990) and as small as 2 pairs in southeast Alaska (G. Van Hine, U. S. Dep. Agric. For. Serv., pers. commun.). Great blue herons nest in a wide range of vegetation and physiographic settings (Gibbs et al. 1987). However, they tend to select the tallest trees within a forested stand and place their nests near the top. The kind of tree available for nesting appears to be less important than its height (Miller 1943). This may be to avoid predators, provide greater visibility, and allow good flight access (Vermeer 1969, Burger 1979, Gray et al. 1980). Heights reported for 926 nests in British Columbia ranged from 23 ft to 230 ft with 67% of the nests between 56 ft and 98 ft (Campbell et al. 1990). Sitka spruce (Picea sitchensis), Western red cedar (Thuja plicata), Western hemlock (Tsuga heterophylla), pine (Pinus spp.), red alder (Alnus rubra), and black cottonwood (Populus trichocarpa) have all been used as nest trees for great blue herons in western United States and Canada (Jackman and

Scott 1975, Campbell et al. 1990). Single nests and small colonies of 2 to 3 nests of Northwestern great blue herons have been found in large Western hemlock and Sitka spruce trees in old growth upland and riparian areas in southeast Alaska (G. Van Hine, U. S. Dep. Agric. For. Serv., pers. commun.).

Although proximity of nest sites to foraging areas is important (Kushlan 1978), reduction of disturbance appears to be more important in nest site selection by great blue herons (Miller 1943, Gibbs et al. 1987). Isolation of a site may be the most important determinant in nest site selection (Henny and Kurtz 1978). Werschkul et al. (1976) reported an indirect relationship in Oregon between the distance from disturbance and the size of great blue heron colonies, the number of nests occupied within colonies, and the fledging rate. The size of great blue heron colonies was also positively correlated with the distance from roads in Montana (Parker 1980).

Foraging Habitat

Great blue herons feed in a variety of aquatic habitats generally less than 1 ft deep, including marine intertidal areas, estuaries, riparian areas, wetlands, freshwater lakes, and muskegs (Gabrielson and Lincoln 1959, Jackman and Scott 1975, Willard 1977). Foraging areas are generally within 3 mi of nest sites, although foraging flights of up to 18 mi have been recorded (Mathisen and Richards 1978, Parris and Grau 1979, Thompson 1979a). At least 3 feeding behaviors have been reported for great blue herons (i.e., standing, walking slowly, and diving feet first) (Forbes 1987a). Prey items taken include fish, amphibians, snakes, small mammals, crustaceans, leaches, and aquatic and terrestrial insects (Palmer 1962). Cottam and Uhler (1945) reported that 68% of the great blue heron's diet consisted of fish, 8% of insects, 8% of crayfish and other crustaceans, 5% of mice and shrews, and 4% of frogs, snakes, and turtles (as determined from the stomach contents of 189 birds). Northwestern great blue herons have been reported to take similar prey (i.e., fish, frogs, mice, shrews, crayfish, and dragonflies) (Palmer 1962). Great blue herons are opportunistic feeders; variation in the diet is generally associated with differences in foraging areas (Jackman and Scott 1975).

HOME RANGE/TERRITORY

During the breeding season the home range of great blue herons encompasses the nest colony and foraging sites. Since they may travel up to 10 mi from nesting areas to feeding areas, home ranges may be very large (DeGraaf 1991).

The male selects the breeding territory which generally encompasses a small area around a previously used nest (Cottrille and Cottrille 1958, Palmer 1962). Activities that take place within the territory include mating displays, copulation, and nesting. The size of the breeding territory appears related to habitat quality and stage of reproductive cycle. Small colonies with 2 or 3 nests apparently have larger territories than large colonies. The size of territory defended decreases as pair formation progresses. The territory is usually only defended against other great blue herons of both sexes.

Maintenance of feeding territories has been reported during the nonbreeding season (Palmer 1962, Dennis 1971).

POPULATION DENSITIES

Great blue heron colonies range in size from a few nests to hundreds of nests (Campbell et al. 1990). However, the Northwestern great blue heron is rarely found in large numbers, especially in southeast Alaska (Jewett et al. 1953; Gabrielson and Lincoln 1959; P. Schempf, U.S. Dep. Inter. Fish and Wildl. Serv., pers. commun.). Single nests and small colonies of 2 to 3 nests that are widely dispersed appears typical of their distribution in southeast Alaska (G. Van Hine, U. S. Dep. Agric. For. Serv., pers. commun.). Isleib and Kessel (1973) estimated the population of Northwest great blue herons as a few 100 in the North Gulf Coast and Prince William Sound areas. The population in southeast Alaska is probably not much larger.

MOVEMENTS/DISPERSAL

Although some populations of great blue herons migrate between wintering and breeding habitats, the coastal population of Northwestern great blue herons is resident, and exhibits very little movement (Gabrielson and Lincoln 1959, Campbell et al. 1990). Foraging birds may move up to 10 mi from nesting sites or roosts (Gibbs et al. 1987, DeGraaf et al. 1991).

Young birds fledge at about 60 days and leave the nest area permanently between 64 and 91 days of age (Pratt 1970). Some young-of-the-year birds from other subspecies apparently disperse widely after they can fly (Palmer 1962, Campbell et al. 1990). Information on the dispersal of Northwestern great blue herons following fledging is not available.

VIABILITY/DISTRIBUTION CONCERNS

Predation on great blue herons, especially by bald eagles (*Haliaeetus leucocephalus*), may have a significant impact on heron populations (Forbes 1987b). Simpson et al. (1987) reported that great blue heron nests that failed because of predation were nearer to the edge of their colony. The tendancy of Northwestern great blue herons to nest singly or in colonies of 2 or 3 nests may make them more susceptible to predation from the large population of bald eagles in southeast Alaska.

Severe winter weather has resulted in the substantial reduction of breeding grey herons (*A. cinerea*) (Reynolds 1979) and great blue herons (Blus and Henny 1981). Nesting mortality in grey herons has also been correlated with the amount of rainfall (Owen 1960). Bovino and Burtt (1979) have also shown that wind and rain decrease the foraging success of great blue herons. More nestlings may also die of hypothermia in the rain (Forbes et al. 1985). Forbes et al. (1985) suggested that reproductive success was favored by low rainfall, or extensive periods of sunshine, or both. Consistent high levels of precipitation and associated wind in southeast Alaska may be limiting the population of Northwestern great blue herons in this area.

The apparent small and scattered population of Northwest great blue herons in southeast Alaska may also be vulnerable to the effects of land management activities (J. King, U.S. Dep. Inter. Fish and Wildl. Serv. [retired], pers. commun.). Great blue herons are considered a species of "special concern" in British Columbia (Campbell et al. 1990). The species is quite vulnerable to disturbance. The sensitivity of the birds and remoteness of many colonies indicate that disturbance is detrimental (Mathisen and Richards 1978, Thompson 1979b, Stephens 1980). Disturbance may have been a major factor in local declines of great blue herons in the midwest United States and in Canada (Vermeer 1973, Bjorklund 1975, Thompson 1979b, Markham and Brechtel 1979, Kelsall and Simpson 1980). Disturbance associated with timber harvest in Oregon was related to the reduction of nearby great blue heron colonies (Werschkul et al. 1976). Disturbance of great blue herons may lead to: 1) increased mortality of young from exposure or predation, 2) nest desertion, or 3) abandonment of the colony (Vos et al. 1985). The effects of disturbance on Northwestern great blue herons in southeast Alaska may have an even greater influence on the population because of the effects of natural factors (e.g., weather, predation).

CONSERVATION STRATEGY

Protection from disturbance, ensuring the availability of adequate nest sites, and availability of adequate food resources has been successful in increasing numbers of great blue herons (Rickard and Watson 1985, Vos et al. 1985). A similar management strategy implemented in southeast Alaska will help ensure the presence of a viable, well-distributed population of Northwestern great blue herons.

Recently occupied and nest colonies should be protected from ground disturbing activities by a buffer of 1/8 mi of undisturbed habitat. Aircraft should not be permitted to fly within 500 ft elevation within 1/4 mi of recently occupied nests or colonies from the period of egg laying through fledging (i.e., 1 March through 31 July) (Campbell et al. 1990).

A nest or colony is considered to be recently occupied until breeding, nesting, or fledging activities have not been observed in the area for 2 years.

MONITORING RECOMMENDATIONS

Forested areas being considered for timber harvest, road construction, recreation development, mineral extraction, or other ground-disturbing activities should be surveyed to determine if nest or colonies of Northwestern great blue herons are present. When nests or colonies are located and buffer zones are established, the nests or colonies should be monitored yearly to document the effects of land management activities on nesting success.

Counts of nests from the ground during the breeding season provide the most reliable estimates of colony size (King 1978). However, such counts may also result in disturbance to nesting birds (Erwin 1981, Tremblay and Ellison 1979). Counts made from aircraft or from aerial photographs were found to be consistently low but with a high precision (Gibbs et al. 1988). Some workers consider aerial survey techniques to be unreliable (e.g., Hutchinson 1979). It is recommended that until nondisturbing, reliable techniques are developed, post-fledging surveys be made from the ground to assess nesting activity in colonies.

RESEARCH RECOMMENDATIONS

The size and distribution of the population of Northwestern great blue herons in southeast Alaska has not been documented other than through anecdotal accounts. Surveys need to be completed to determine their distribution throughout southeast Alaska and to estimate their population size.

Work on great blue herons in other areas has indicated that they are very sensitive to disturbance, especially during the nesting period. The degree of that sensitivity needs to be established for southeast Alaska so that management guidelines address the local situation.

The status of great blue heron populations varies throughout their range. The productivity and recruitment rates of Northwestern great blue herons in southeast Alaska need to be determined to establish the status of the population.

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VIABILITY ANALYSIS FOR THE VANCOUVER CANADA GOOSE

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SUMMARY

The Vancouver Canada Goose (VCG) (Branta canadensis fulva) was identified as a species with potential viability problems by the Viability Committee. There appears to be no population viability risk to the VCG based upon the species biology and recent MVP literature. Important VCG nesting and brood-rearing habitat is generally associated with low volume old growth in association with poorly drained soils, small wetlands, and riparian areas. Because the VCG is not an area sensitive species, maintenance of additional old-growth habitats is not necessary to maintain viability, but additional habitat would contribute to habitat capability to meet public use demands above viability levels. Two recommendations are made to strengthen the potential to maintain VCG habitat capability to maintain viable populations.

CURRENT STATUS

This subspecies occurs throughout southeast Alaska with an estimated population of 10,000 (AMS 3-686).

MINIMUM VIABLE POPULATION ANALYSIS

MVP guidelines for the VCG include maintaining habitat to support 125 geese in each of 8 geographic areas within the Tongass National Forest to maintain a well distributed viable population within the planning area (TLMP DEIS, 3-554). While putting a quantitative estimate on a viable population is difficult without detailed demographic population data, the recommended total of 1000 individuals is a conservative approximation of the "few thousand" order of magnitude guideline recommended by Soule" (1987) and follows the recommendation of Salwasser et al. (1984) of "...preferably over 1000 adults on the average." as a basic viability conservation strategy.

The second component of ensuring a MVP is that the population be well distributed. Dispersal and the ability to interact with adjacent subpopulations in a metapopulation, thus precluding the isolation of local populations, are key ingredients in developing a viable population strategy (Salwasser et al. 1984). The VCG has the potential dispersal mobility of several hundred miles (Hansen 1962), migrating between Glacier Bay and Oregon. With this degree of mobility, use of large (provincial size) game management units and entire islands as a scale for maintaining a well distributed population of VCG is appropriate.

Habitat capability for the VCG was modeled on the nesting and brood rearing habitat requirements using plant associations (Doyle et al. 1988). In summary habitat requirements generally include low-volume, old-growth stands (ave dbh = less than 10"), usually in poorly drained soils, normally adjacent or near small wetlands, lakes or riparian areas. An analysis was conducted of the volume class rating for each plant association (Pawuk and Kissinger 1989) to evaluate the risk of development of high quality VCG habitat.

The analysis revealed that plant associations with a Habitat Capability Index (HCI) rating of .8 or higher (high quality habitat) (N=6) had a volume class of 3 (non-commercial) or 4 (8-20,000 bf/ac) (AMS 3-643). Those plant associations meeting the high HCI value and overall high quality habitat requirements generally occurred within the Mixed Conifer plant association series (Pawuk and Kissinger 1989). Conversely those plant associations with a Volume Class of 5,6 or 7 (> 20,000 bf/ac) had an average HCI value of .35 (N = 13). This analysis contradicts the evaluation of relative habitat value of Volume Classes for VCG found in DEIS table 3-135 rating VCG habitat in Volume Class 5,6, and 7 as high. This analysis suggests that high volume old growth, most in demand and at risk of development (i.e., timber harvest) is generally lower quality (i.e., lower HCI) VCG habitat.

The current VCG habitat capability for the Tongass National Forest is 13,001 (AMS 3-686), a conservative estimate because capability for Wilderness Areas was not available. This estimate is above the estimated current population of 10,000 suggesting that breeding and brood rearing habitat is not limiting on the Tongass National Forest.

To evaluate if MVPs are achieved in the most limiting TLMP alternative and in a worst case scenario, both total habitat capability and well-distributed criteria were examined for these 2 situations. Alternative C produced the greatest reduction in VCG habitat capability of all alternatives, with an estimated forest-wide habitat capability of 11,500 in the year 2150 (DEIS Table 3-170). In terms of total population, this estimate is ten times greater than the recommended MVP of 1,000. A worst case scenario was also presented in the DEIS (3-604) - assuming that all suitable old growth allocated to a harvest prescription was logged. Under this scenario Alternative C reduced habitat capability to 66% of 1954 levels (14,131) or 9,326 geese. This estimate is also well above the recommended MVP of 1,000.

To determine if the population would remain well distributed under these same 2 situations, the proportional contribution to total capability of each geographical unit was examined (Table 1). This analysis assumes that reductions in habitat capability would be evenly distributed forest-wide. This is a conservative examination because capabilities for Wilderness Areas are not included and would necessarily increase capability estimates in every unit.

In all cases but Yakutat, the VCG population would retain the recommended well distributed habitat capability of at least 125 geese in both Alternative C and the Worst Case scenario. By nature of its smaller size, the Yakutat Unit does not presently have the capability to support the recommended MVP of 125 geese.

An additional level of protection to ensure maintaining the integrity of key VCG habitats is the recently passed Tongass Timber Reform legislation. This provides a mandatory 100 ft buffer on each side of Class I and all Class II streams flowing into Class I streams. These riparian areas are key components to VCG habitat use and will be protected under all proposed plan alternatives.

Evidence also suggests that VCG are often sensitive to disturbance (Doyle et al. 1988). A review of the Wildlife Forest-Wide Standards and Guidelines for Waterfowl (DEIS G-51), revealed that adequate protection measures are available during site-specific project implementation to protect important waterfowl areas from undue disturbance.

The above analysis suggests that a well-distributed, viable population of VCG would be maintained on the Tongass National Forest. The VCG is not known to be sensitive to habitat fragmentation and does not require large blocks of old-growth habitat to achieve maximum habitat effectiveness. Any additional VCG capability produced by maintaining additional old-growth habitats would help contribute to meeting consumptive and nonconsumptive demands for this species.

RECOMMENDATIONS

Yakutat

The current MVP, well-distributed capability recommendation of 125 geese cannot be achieved due to size and habitat limitations in the Yakutat geographical unit. However, since this peripheral VCG population has and is adapting to a unique post-glacial successional habitat zone, the value of the genetic diversity of this population is essential to future adaptability of the VCG population. Rather than recommend a lower habitat capability goal to achieve a well-distributed guideline, management emphasis should strive to maintain 100% of current the current habitat capability in the Yakutat Unit, possibly through strengthened Standards and Guidelines.

Estuarine Habitats

A forest-wide allocation of the Beach Fringe/Estuarine prescription is recommended. Estuarine areas are extremely high quality habitats for VCG, especially in relation to their limited availability forest-wide. The value of these areas has been adequately recognized in the Habitat Capability Model (Doyle et al 1988). Because of the importance of the estuarine habitat, the VCG could disproportionately benefit from a forest-wide allocation of the Beach Fringe/Estuarine prescription. Not only would VCG benefit from the protection of this nesting and essential brood rearing habitat, but the integrity of these areas, both habitat structure and levels of disturbance, would be protected for the critical wintering, molting, and prenesting periods. This recommendation would also serve a number of other advantages:

1. serve as a forested habitat corridor throughout an island to connect habitat fragments, reduce the likelihood of creating insular populations, and provide a functional habitat connectivity for achieving a metapoplation conservation strategy, especially for terrestrial vertebrates with limited dispersal capabilities;

2. maintain nearly 100% of bald eagle habitat capability forest wide;

3. maintain nearly 100% of river otter habitat capability forest wide;

4. significantly contribute to increased habitat capability of deer, brown bear, black bear, and marten as the beach fringe and estuarine habitats are generally the highest quality areas;

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Table 1. Distribution of present habitat capability of 13,001·VCG by the 8 recommended geographical units necessary to maintain a well distributed population. Geozones without estimates are Wilderness areas that should maintain present habitat capability (AMS table 3-192). Capability in Alternative C is that Unit's contribution to well distributed in Alternative C, (% of 11,500). Capability in the Worst Case Capability is that Unit's contribution to the total capability (% of 9326) scenerio of logging all suitable old growth allocated to a harvest prescription.

Unit	Geozones Without Estimates	Total VCG Capability	% of Total Capability	Alt C Capability	Worst Case Capability
1A	K13	1195	9.2	1057	858
2	K14, K15	3264	25.1	2887	2341
1C, Admiralty	C14, C15	478	3.7	423	345
1B, 3	S12	5472	42.1	4840	3926
Chichakof	C03, C12	1058	8.1	935	755
Baranof	C13	749	3.7	424	345
Yakutat	C17	50	0.38	44	35
Chilkat, 1D	C16	555	4.3	495	401

A CONSERVATION STRATEGY FOR THE QUEEN CHARLOTTE GOSHAWK ON THE TONGASS NATIONAL FOREST

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SUMMARY

The Queen Charlotte goshawk (Accipiter gentilis laingi) is endemic to southeast Alaska and coastal British Columbia. Owing to its restricted distribution and low natural densities, its population was never great. Analyses indicate that timber harvesting has reduced the population, to an estimated 200-500 pairs in southeast Alaska. The current average density in southeast Alaska might be between 0.2 and 0.5 pair per 10,000 ac of forested land, including muskegs with scrub forest. Pairs (of other subspecies) usually have home ranges between 4,000 ac and 10,000 ac---the median may be 6,000 ac. Although goshawks have the ability to travel great distances, most dispersal to vacant breeding habitat is less than 30 mi from where a bird was hatched.

Analyses of habitat use have shown similar results throughout the geographical range of the northern goshawk in the United States. Home ranges include stands of large trees for nesting, for goshawk flight space beneath the canopy, and for greater abundance and accessibility of some prey. The sparseness of shrubs and small trees appears to facilitate goshawk flight and prey capture. Also, closed canopies provide preferred microclimate in the nesting stand, possible

inhibition to predators, inhibition to open-forest raptors, and increased productivity of some important prey species. A literature review indicated that goshawk densities tend to decrease with amount of timber harvest, and that goshawks may be heavily impacted by forest fragmentation. Southeast Alaska has always included much habitat which is probably marginal or unsuitable for goshawks, but timber harvesting has added to the habitat fragmentation and biogeography problems of the Queen Charlotte goshawk.

My recommended conservation strategy, for the Tongass National Forest, includes habitat conservation areas (HCAs): Large HCAs---capable of supporting 8 pairs of goshawks---separated by 20 mi or less; and Medium HCAs---capable of supporting 2 pairs of goshawks---at distances no greater than 8 mi from Large HCAs or other Medium HCAs. Any goshawk home range located outside of Large and Medium HCAs, would have the male's 1,600-ac core area protected from timber harvest units. Within the home range beyond there, no more than 5% of the timbered land could be harvested in any one decade, though unsuitable timber stands would be included in the calculation of 5%. Recommendations are provided for goshawk study and monitoring.

DISTRIBUTION AND POPULATION STATUS

Three subspecies of northern goshawks breed in North America (Johnsgard 1990). *A. g. atricapillus* is the widely distributed, pale to medium bluish gray form shown in most North American field guides. The Apache goshawk (*A. g. apache*) is larger and has heavier feet, and is found only in northwestern Mexico and the southern portions of Arizona and New Mexico (Brown and Amadon 1968). The Queen

Charlotte goshawk (A. g. laingi) is darker than A. g. atricapillus (Taverner 1940), and often very blackish (Webster 1988) or extremely dark brown/blue (Crocker-Bedford 1990a). The Queen Charlotte subspecies is slightly smaller (Johnson 1989), and occurs only in southeast Alaska and coastal British Columbia (Webster 1988). Information to date indicates that the Queen Charlotte subspecies is most distinct in the Queen Charlotte Islands and southern southeast Alaska, and grades into the A. g. atricapillus somewhere on Vancouver Island, British Columbia, and perhaps in northern southeast Alaska (Webster 1988).

A preliminary habitat capability model estimated a goshawk decline of at least 30% in southeast Alaska and more than 50% within the subspecific range of the Queen Charlotte goshawk (Crocker-Bedford 1990a). Crocker-Bedford (1990a) estimated the current population of goshawks in southeast Alaska at less than 800 pairs, and the total international population of Queen Charlotte goshawks at far less than 2,500 pairs.

Reviews by Iverson (USDA Forest Service, unpubl. rep.) have indicated that the true population is much smaller. He and other reviewers have suggested that Crocker-Bedford's (1990a) habitat capability estimates were too high and projected population declines were too small. This was because Crocker-Bedford's (1990a) habitat capability model considered low volume, commercial forest as fully suitable habitat, even though such stands in southeast Alaska provide little flight space for goshawks. The preliminary model did not account for the higher habitat value of high volume old-growth forests for prey production and accessibility (goshawk flight space), and the model did not consider the fact that logging has concentrated almost exclusively in stands of higher volume timber. The fragmentation model used by Crocker-Bedford (1990a) also failed to consider effects on goshawk flight space of extensive past removal of large trees near beaches, which induced increased understory. Although data are not extensive (see POPULATION DENSITIES AND TRENDS---Southeast Alaska), the professional estimate of several biologists in southeast Alaska, who have carefully considered the local goshawk situation, is that the actual population of goshawks in southeast Alaska is currently under 500 pairs and might possibly be lower than 200 pairs. (These rough estimates were based primarily on the paucity of sightings relative to time in the field by biologists and birders.)

The USDI Fish and Wildlife Service (1992) has designated the northern goshawk (including all three subspecies) as a Category 2 Candidate Species for Threatened or Endangered Status in the United States.

> "Category 2 includes those taxa for which there is some evidence of vulnerability, but for which there are not enough data to support a listing proposal at this time. Elevation to Category 2 does not mandate initiation of a status review. However, because of the level of concern for the goshawk, the [USDI Fish and Wildlife] Service" has initiated a "status review (50 CFR 424.15) to better understand trends in population size and stability and loss or modification of habitat." (Ibid.:545).

As of February, 1992, goshawks were on the Sensitive Species lists of three Forest Service Regions: Southwest, Intermountain, and Pacific Southwest. In Alaska goshawks have been under consideration for Forest Service Sensitive Species status since 1986 (Sidle and Suring 1986). In March, 1991, the Interagency Wildlife Technical Committee, with representatives from 6 agencies, unanimously recommended Sensitive Species status for the Queen Charlotte goshawk (Samson memo of 4/3/91).

PATTERNS OF HABITAT USE

Goshawk literature is relatively consistent in regards to patterns of habitat use, especially for western coniferous forests. The goshawk has long been recognized as typically being dependent upon extensive forests and large stands of "heavy" timber (Bent 1937:127-128). For this reason goshawks may be adversely affected by timber harvesting, especially near nests: in Oregon (Reynolds et al. 1982, Moore and Henny 1983, Mannan and Meslow 1984); in California (Saunders 1982, Hall 1984, Bloom et al. 1985, Woodbridge 1988, Fowler 1988); in Nevada (Herron et al. 1985 as cited in Fowler 1988); in Idaho and Utah (Hennessy 1978); in Idaho (Patla 1990, 1991); in Montana and Idaho (Warren et al. 1990); in South Dakota (Bartelt 1977); in Arizona (Crocker-Bedford 1987, 1990b, 1991, Crocker-Bedford and Chaney 1988, Zinn and Tibbitts 1990); in New Mexico (Kennedy 1988, 1989); and in general (Jones 1981, Reynolds 1983, 1989). Other studies also supported the importance of dense, large trees in nesting stands, though the authors did not specifically conclude adverse effects from timber harvesting: in Colorado (Shuster 1980); in northern Idaho and Montana (Hayward and Escano 1989); and in the Northeastern States (Speiser and Bosakowski 1987, Falk 1990).

Reynolds (1989:97) stated: "Preferred habitat during the breeding season is older, tall forests---deciduous, coniferous and mixed---where goshawks can maneuver in and below the canopy while foraging and where they can find large trees in which to nest." Two radio telemetry studies in Utah (Fischer 1986) and California (K. Austin, Shasta-Trinity National Forest, pers. commun.) determined that goshawks preferred to forage in tall, mature and overmature trees. Results of Fischer and Murphy's (in prep.) study indicated that the differences in goshawk foraging preference were associated with prey vulnerability and not prey abundance, but Reynolds and others (1991) felt that prey abundance was more important than its accessibility. Crocker-Bedford and Chaney (1988) showed preference (use compared to availability) for nesting in stands of large trees with dense canopies, and suggested such preference was associated with similar stands in the vicinity used for foraging. Most prey species of goshawks inhabit the ground and shrub layer in a forest or are generalists found at any level of the forest (Reynolds and Meslow 1984). Following timber harvest, the change from larger trees to smaller trees may reduce the goshawk's ability to hunt successfully (Reynolds 1989, Gullion 1990, Crocker-Bedford 1990b). Considerable habitat within the home range of a pair of goshawks must be of high enough quality to provide sufficient and accessible prey relative to the time and energy expended while hunting.

Closed forest should be contiguous enough to inhibit open-forest and forest-edge raptors (Crocker-Bedford 1990b). Woodbridge (1988) found adverse effects from forest fragmentation in California in addition to direct habitat losses. In Connecticut, goshawks nested an average of 6 mi from forest clearings, 54-88% farther than the average random point from clearings, and farther from openings than nests of any of the other hawks (Falk 1990). The importance of extensive forest was also found in New York (Speiser and Bosakowski 1987). In contrast, nests in northern Idaho and Montana averaged only 0.25 mi from the nearest opening larger than 3 ac (Hayward and Escano 1989); however, the authors suggested that some results of their study were probably biased because many nests were located during timber harvest operations.

Goshawks utilize relatively large prey. In Oregon half the biomass consumed came from birds larger than 200 g (large woodpeckers, owls, pigeons, quail, grouse and ducks) and from mammals larger than 450 g (large squirrels, rabbits and hares), though other species were also major dietary items (small squirrels, flickers, jays, and thrushes) (Reynolds and Meslow 1984). Ptarmigan (Lagopus spp.) can also be important (Johnsgard 1990). The Queen Charlotte goshawk consumes many northwestern crows (Corvus caurinus) on the Queen Charlotte Islands, and mostly Steller's jays (Cyanocitta stelleri) and varied thrushes (Ixoreus naevius) on Vancouver Island (Johnsgard 1990). Prey remains collected at a goshawk nest on Sumez Island, near Craig, Alaska, were mostly spruce grouse (Dendragapus canadensis) and Steller's jays, along with a greater yellowlegs (Tringa melanoleuca) (collections by author with positive identification by D. D. Gibson, Univ. Alaska Museum). Crows have been seen to be a major prey item near Juneau in southeast Alaska (R. Armstrong, pers. commun.).

In summary, large trees are important for nesting and perching (numerous studies previously cited), for flight beneath the canopy and between tree trunks (Moore and Henny 1983, Reynolds 1989, Crocker-Bedford 1990b, Warren et al. 1990), and perhaps for greater prey productivity (Crocker-Bedford 1990b, Warren et al. 1990, Reynolds et al. 1991). Closed forest canopies provide preferred microclimate in the nesting stand (numerous studies previously cited) and possible inhibition to predators in the nesting stand (Reynolds et al. 1982, Moore and Henny 1983, Crocker-Bedford and Chaney 1988, Crocker-Bedford 1990b). Also, closed canopies may be associated with overall prey productivity

(Crocker-Bedford and Chaney 1988, Crocker-Bedford 1990b, Warren et al. 1990), or at least the abundance of certain key prey (Reynolds et al. 1991). The sparseness of shrubs and small trees appears to facilitate goshawk flight (Moore and Henny 1983, Speiser and Bosakowski 1987, Crocker-Bedford 1990b, Warren et al. 1990, Reynolds et al. 1991) and possibly facilitates prey capture (Reynolds and Meslow 1984, Speiser and Bosakowski 1987, Reynolds 1989, Gullion 1990, Crocker-Bedford 1990b, Warren et al. 1990).

The amount of forest with the above attributes, within a home range, may increase the energy intake to expenditure ratio of goshawks (Crocker-Bedford 1990b, Warren et al. 1990). Still, Reynolds and others (1991) believed that goshawks thrive best when provided a wide variety of stand ages (including both the young and older stands which would be found given a 200-300 year timber management rotation) and given a variety of canopy densities. Although Reynolds et al. (1991) called for various stand conditions to be well interspersed for positive edge effects, Crocker-Bedford (1990b) had provided evidence which implied that inter-specific competition from other raptors increased following forest fragmentation beyond the nesting stand.

HOME RANGE/TERRITORY

Goshawks defend against humans 20-25 ac around each of their nests (Reynolds 1983). Unless habitat is altered, a pair apparently defends against other raptors a territory which surrounds all of the pair's cluster of alternate nests. The territory defended against conspecifics may be larger (Crocker-Bedford 1990b).

The home range of a pair of goshawks is apparently larger than their inter-specific or intra-specific territory. Distances between goshawk nest clusters (Crocker-Bedford 1990b), and between goshawk nests and those of other raptors (Crocker-Bedford unpub. data), are often smaller than the radii implied by the literature on home range sizes. Literature as of 1983 showed goshawk home ranges between 5,000 and 8,000 ac (Reynolds 1983). In northern New Mexico from June through September, Kennedy (1989) found that 3 adult males spent 95% of their time within 4,200, 4,400, and 7,000 ac. Adult females averaged 95% of their time within 3,200 ac for the same period. From July through September in northern California, K. Austin (Shasta-Trinity National Forest, pers. commun.) found that 95% of the radio observations of 5 males occurred within an average of 2,930 ac (range = 1,470-4,550 ac), while 95% of the observations of 5 females occurred within an average of 6,990 ac (range = 3,650-10,420 ac). A home range may be 17,000 ac in fragmented forest (K. Austin, Shasta-Trinity National Forest, pers. commun.)

Home ranges of goshawks possibly overlap greatly where habitat is high quality and continuous, so that goshawk nesting occurs at high densities. For example, in virgin and near virgin locales in northern Arizona, pairs concentrated at about one per 1,100 ac (Crocker-Bedford 1990b), much smaller than any of the home ranges discussed above. On the other hand, some populations are so sparse that much unused area occurs between the home ranges of the nesting pairs. For example, in northern New Mexico, Kennedy (1989) found one occupied territory per 8,800 ac (some occupied only by a female without a male), while the measured home ranges there (see above) were smaller. Home range sizes should not be used to calculate assumed breeding densities, nor should breeding densities be used to calculate assumed home ranges of goshawks.

POPULATION DENSITIES AND TRENDS

The reported breeding densities that are summarized in Table 1 for western North America should be used with caution because survey techniques and intensity varied between studies.

Southeast Alaska

Goshawks in southeast Alaska may have declined by far more than 30% since 1950, to less than 500 pairs, and possibly to even less than 200 pairs (see DISTRIBUTION AND POPULATION STATUS). This indicates a current average density between 0.1 and 0.3 pair per 10,000 acres of total landscape (not including sea), or between 0.2 and 0.5 pair per 10,000 ac of potentially forested land, including muskegs with scrub forest. Pair density in southeast Alaska might average between 0.4 and 0.9 per 10,000 acres of forest having over 8 mbf (thousand board feet) of sawtimber per acre.

Crocker-Bedford's (1990a) model estimated habitat capability at 810 pairs in 1988 in southeast Alaska, much higher than the 200-500 pairs estimated by most biologists familiar with the goshawk data of southeast Alaska. Crocker-Bedford (1990a) had estimated 2.5 pairs of goshawks per 10,000 acres of landscape, where the landscape was predominately (84%) old-growth forest having over 8 mbf/acre.

Where old-growth forest having over 8 mbf/acre accounted for half the landscape, Crocker-Bedford (1990a) had estimated 0.7 pair per 10,000 ac of landscape.

In 1991 in southern southeast Alaska, a team of 2-4 biologists surveyed for goshawks (Gustafson 1991). During 57 person-days the team quickly covered 36,000 ac in locales where goshawks had previously been reported---resulting in 7 sightings in 4 locales, as well as 2 newly found nest sites. In other locales where no goshawk had previously been reported, another 55 person-days of surveys covered 37,000 ac of what appeared to be suitable landscape, but resulted in no sighting of any goshawk and no located nest site.

Of 16 confirmed or highly probable goshawk nest sites within the range of the Queen Charlotte goshawk in southeast Alaska, 8 were clearcut or planned for timber harvest until the goshawk nests were found (Iverson unpubl. rep.). Considering only southeast Alaska, Iverson also noted that

> "there appears to be a clinal variation in goshawk abundance, increasing from north to south with most (81%) observations occurring south of Frederick Sound. From a landscape perspective, it is in this region where most timber harvest has already occurred, and where 74% of the planned timber harvest on the Tongass National Forest will occur in the next 10-15 years."

Washington

Goshawks densities in western Washington are sparser than northern spotted owls (Strix occidentalis caurina), and goshawks are more adversely affected by forest fragmentation than are northern spotted owls (R. Lowell and P. Meehan-Martin, pers. commun.).

Oregon

Reynolds and Meslow (1984) suggested that the lack of nesting goshawks in northwestern Oregon was possibly due in part to the amount of past timber harvesting and wildfires that had occurred there.

Mannan and Meslow (1984) concluded that goshawks could possibly be extirpated from northeastern Oregon if old growth forest stands allocated to timber harvest were actually logged.

California

The breeding population of goshawks in California was estimated to have decreased one-third by 1985, mostly because of timber harvesting, and the decline was continuing at about 1% per year (Bloom et al. 1985). The goshawk was once common during winter in southern California, but is now very rarely seen there (Bloom et al. 1985).

Idaho

In Idaho, Patla (1990) found a loss of nesting sites because of logging, despite standards meant to protect the nesting sites. A more thorough and longer-term analysis (Patla 1991) indicated that timber harvesting within 1/4 mi of protected nest sites resulted in a 75-80% reduction in goshawk occupancy of nesting territories. Actual losses were probably higher because of the harvesting of unknown nest trees. The vacated nests were often taken over by other raptors. These results replicated those found by Crocker-Bedford (1990b) in Arizona.

Northern New Mexico

Goshawks in New Mexico appeared to be "threatened" as a result of low reproductive success and low density (Kennedy 1989). Removal of old growth habitats probably reduced the historic population of goshawks in this area (Kennedy 1988). Four of 16 nesting females were without mates (P. Kennedy, Colo. State Univ., pers. commun.); perhaps because the population density had been reduced so much (i.e., only 1.1 nesting female per 10,000 ac) that the opportunity for pairing was reduced (Lande 1987, 1988).

Northern Arizona

Timber harvesting under a selection-harvest regime, in which one-third of the timber volume was cut, was associated with a decrease in goshawk reproduction (Crocker-Bedford 1990b). Pair occupancy exhibited a measured decrease of 75% relative to the controls, despite nest buffers of 3-500 ac (mean = 95 ac) in the treated locales. Fledglings per nest attempt showed an additional decrease of 75%. Other raptors replaced goshawks in most logged territories but not in any control territory. Goshawk foraging habitat may have been degraded by the loss of large trees and by an increase in shrubs, saplings and small trees. Given the amount of timber harvest on the North Kaibab Ranger District, the goshawk breeding population was estimated to have dropped by half between 1972 and 1987, and by three-fourths since timber harvesting began there (Crocker-Bedford 1990b). The true decline might have been even greater (Crocker-Bedford unpubl. rep. Aug. & Sept. 1991).

These and other data were reanalyzed (Crocker-Bedford 1991) to determine the decline in nesting and reproduction as compared to the amount of timber harvesting from 1973 to 1986 within assumed, circular home ranges (n = 53) of 5,800 ac. Selection harvesting in 10-39% of the stands in a home range was associated, on the average, with 50% less reproduction than in home ranges receiving little or no harvesting. Selection harvest in 40-69% of the stands in a home range resulted, on the average, in an 80% decrease in reproduction. Little occupancy and no reproduction occurred when selection harvest extended over 70% or more of the stands in a home range.

Results from 1988-1990 in the same location, the North Kaibab (Zinn and Tibbitts 1990), indicated an even faster decline than estimated by Crocker-Bedford's studies above. After collecting 1991 data, Reynolds (unpubl. rep.) noted that the decline may have stopped or reversed. However, other data (Reynolds unpubl. data) showed that only 13 of the nests occupied in 1991 were in the 121 goshawk nest trees checked in 1987 for Crocker-Bedford's studies, while 24 (65%) of the nests occupied in 1991 had been found since 1987. Nearly half the territories known to be occupied in 1991 had been found since Crocker-Bedford's studies ended in 1987. It is possible that the one-year population increase, measured in 1991 on the North Kaibab, was partly because of increased survey efforts and surveys in future timber sale areas (R. Reynolds pers. commun.).

Eastern United States

Although significant numbers of goshawks (mostly from Canada) sometimes winter in the eastern United States, breeding densities were greatly reduced throughout the eastern States (Bent 1937). The goshawk was extirpated south of the Lake States and Pennsylvania (Jones 1981). The recovery and maturation of many forests in the East may explain the recent range expansion of the goshawk in the Northeast (Speiser and Bosakowski 1987) and the recolonization of the Appalachian Mountains nearly to Georgia (Johnsgard 1990).

Given the goshawk's persistence in parts of the Northeast despite the extent of past forest harvest there, the concern for the viability of the Queen Charlotte goshawk might appear unwarranted. However, hardwood and mixed broadleaf/conifer ecosystems in the Northeast may produce more usable goshawk prey at earlier stand ages than do western coniferous forest ecosystems. Coniferous forests tend to go through a long second-growth stage with few understory plants, while second-growth broadleaf forests typically continue to produce many herbs and shrubs. The forage, seed and berry production in immature broadleaf forests may support larger prey populations than do immature coniferous forests in the West. Furthermore, prey in young broadleaf forests may be more available, because second-growth coniferous forests include a longer period when canopies extend to the ground, thereby impairing maneuverability by goshawks and providing prey escape cover.

Finally, goshawk populations in the Northeast and Lake States have possibly benefited from the periodic invasions of Canadian goshawks of the same subspecies. No such population reservoir exists for the Queen Charlotte subspecies.

Comparison with Europe

Although goshawks (A. g. gentilis) became extinct in England and southern Europe, they persisted in northern Europe despite significant logging and land conversion there. Such persistence in Northern Europe may be in part because of the lack of a European counterpart to our red-tailed hawk (Buteo jamaicensis), resulting in less competition than goshawks find in open forests on this continent (Moore and Henny 1983, Crocker-Bedford 1990b, Patla 1991).

Persistence of goshawks in northern Europe may also be because of the fact that Old and New World goshawks differ morphologically. They may be different species (Brown and Amadon 1968).

MOVEMENTS/DISPERSAL

Several lines of evidence indicate limited movements of Queen Charlotte goshawks over the subspecies range. Adult goshawks (of other subspecies) do not shift their breeding locations and even adult goshawks from northern latitudes are usually resident on their territories year-around (McGowan 1975, Widen 1985). Non-breeding adults without territories also are usually resident year-around (Widen 1985). In south-central Sweden when food shortages induce adults to cease defending their breeding territories during some winters, adults typically travel only 60 mi from their nests (Widen 1985). Goshawks in western British

Columbia (the Queen Charlotte subspecies) also move little (Beebe 1974 as cited in Johnsgard 1990). In contrast, goshawks in interior Canada may travel 100s of mi during food shortages. When adults do leave their residences for winter, the sexes often go separately (Widen 1985), but even so they appear to return to their same territories in spring as they are pair-bonded until one dies (Brown and Amadon 1968, Palmer 1988, Johnsgard 1990). In short, it is improbable that much genetic interchange occurs because of the movements of adults, nor is it likely that unpaired adults will find mates or vacant habitats much beyond their home ranges.

Dispersal by juvenile goshawks may also be limited relative to the distribution of the Queen Charlotte subspecies and some of the bodies of water within its range. In central Alaska, recoveries of 8 banded juveniles indicated average dispersal of 12 mi (McGowan 1975), though in south-central Sweden 6 of 8 juveniles dispersed over 30 mi (Widen 1985). With a much larger sample size (303 recoveries), Hoglund (1964 as reported in Widen 1985) determined that only 44% of all juveniles in northern Sweden dispersed more than 30 mi. Furthermore, only 4% of the juveniles in Germany dispersed over 30 mi (Glutz et al. 1971 as reported in Widen 1985). Many of the juvenile recoveries were during winter, so in the spring many of the juveniles might have returned to the general vicinities where they were fledged, as do so many birds. In short, juvenile dispersal is probably inadequate to promote full genetic mixing within the subspecies range, and juveniles will probably not discover vacant habitat or mates over 30 mi from their fledging sites.

VIABILITY/DISTRIBUTION CONCERNS

Many citations throughout previous portions of this paper implied that timber harvesting has the potential to adversely affect goshawk habitat. Furthermore, reductions in goshawk breeding densities following logging, even given protection of nesting stands (Woodbridge 1988; Crocker-Bedford 1990b, 1991; Patla 1991; Zinn and Tibbitts 1990), have demonstrated that timber management can negatively affect the forest habitat mixture that is necessary for goshawks beyond nesting stands. One theory, which would explain decreases in reproduction following logging even where nest sites are protected, is that goshawks are unable to expand their home range and foraging efforts enough to fully compensate for the losses of key foraging habitats. The male must provide almost all the food for the entire family from May through July. The male's home range is typically large which implies that, even given relatively abundant food, it is already difficult for him to gather enough food to feed the family. Consequently, the loss of foraging stands may affect reproduction more severely than implied by the simple proportion of a home range that is harvested (Crocker-Bedford 1990b, 1991, Patla 1991).

Goshawk breeding densities in North American coniferous forests, as determined by several studies, were compared to descriptions (usually qualitative) of past timber harvesting (Table 1). An exact comparison was not possible because most authors described land management activities in only broad generalizations. Even so, the consistently lower breeding density associated with amount of timber harvest is another indication of adverse effects of timber harvest on goshawks.

Evidence exists that timber harvesting and land conversion can contribute to the extirpation of goshawks from large regions. Goshawks were extirpated from the southern half of Europe. Also, Jones (1981) believed that goshawks had been severely reduced in the northeastern United States and extirpated south of Pennsylvania. His contention is supported by the fact that where forests have matured, goshawks have expanded their range in the Northeast (Speiser and Bosakowski 1987) and recolonized the Appalachian Mountains almost to Georgia (Johnsgard 1990). The large population reservoir of A. g. atricapillus in Canada may have contributed to the recent partial recovery of goshawks in the eastern United States; however, no such population reservoir exists for the insular Queen Charlotte goshawk.

Goshawks tend to frequent landscapes that include forested habitat which humans value for potential lumber. Half of the known nest sites of the Queen Charlotte goshawk in southeast Alaska have already been clearcut, or were within units planned for timber harvest until the nest sites were found (Iverson unpubl. rep.). For Alaska, Iverson also noted that most Queen Charlotte goshawks appear to inhabit the southern half of southeast Alaska, where most timber harvest is scheduled to occur.

Reed et al. (1986) calculated that at least 610 interbreeding pairs of goshawks are necessary to assure long-term genetic viability. The existing population of Queen Charlotte goshawks in southeast Alaska appears below that figure, and it may be that very little mixing occurs across Dixon Entrance with the birds in Canada. More importantly, other threats usually require that a viable population be considerably larger than that needed simply for genetic viability

(Lande 1988). Indeed, the Queen Charlotte subspecies as a whole, including Canadian birds, meets Mace and Lande's (1991) criteria for "vulnerable" to extinction.

The Queen Charlotte goshawk in southeast Alaska might be surviving as a metapopulation largely isolated from the one(s) in Canada. A metapopulation is comprised of several to many demes which only occasionally exchange individuals, while movement within a deme is more complete. It might be that goshawk demes largely correspond with the Ecological Provinces or Subprovinces which have been designated for the Tongass National Forest (USDA Forest Service 1991a). Each (Provincial?) deme population, within which much exchange occurs, may in turn be comprised of very local groups within which interaction between goshawks is complete. The segregation of a low total population of a subspecies into smaller metapopulations and very small demes, along with population declines, increases the chance of local extirpation and possible extinction of an entire subspecies (Mace and Lande 1991).

Logging has probably contributed to the goshawk's island biogeography problems. Tracts of suitable habitat have shrunk owing to the addition of large areas of second growth and clearcuts to the naturally unsuitable habitats of water and ice. Some islands (e.g. Hecata) have been so altered that they may no longer support even one pair of goshawks. The effect has been to make the patches of suitable landscapes smaller, and farther apart, than the Queen Charlotte goshawk already had to contend with naturally. Having suitable home ranges farther apart, leads to slower recolonization of vacant habitats and causes otherwise suitable home ranges to be often unoccupied (Lande 1987, 1988). Tracts of suitable landscape that are so small that few pairs can be supported, realize an edge effect whereby most juveniles are forced to venture through unsuitable landscape in search of vacant habitat and mates, as opposed to finding homes within their ancestral tract (Thomas et al. 1990). Such forest fragmentation was reducing interactions between separated groups and pairs of spotted owls; thereby increasing the likelihood that individual pairs or groups would disappear, and which if continued could culminate in the extinction of the species (Thomas et al. 1990).

In southeast Alaska (including private and state lands), logging has tended to concentrate in the tracts of landscape that were probably high quality for goshawks---those dominated by higher volume timber stands (Crocker-Bedford 1990a). Most residual tracts of landscape potentially useable by goshawks are naturally fragmented with more unsuitable habitats and more low quality habitats (rock, ice, open water; and shrubby, low volume forests and forested muskegs) or are fragmented with clearcuts and second-growth. As a result, I suspect that most residual tracts of "suitable" landscape support sparser breeding densities (see POPULATION DENSITY AND TRENDS---Southeast Alaska), which may lead to reduced reproduction (Lande 1988). One-fourth of the territorial females were unmated in a sparse population in degraded habitat in New Mexico (P. Kennedy, Colo. State Univ., pers. commun.)

Given the apparently low densities of goshawks in southeast Alaska, it may be that no tract of suitable landscape exists which is large enough and of high enough quality to contain a self-perpetuating group---the critical patch size of Lande (1988). Instead, local persistence more likely relies upon: (1) the probable years of persistence of a group in a tract of landscape, which is related to the number of pairs that can be supported there, as well as their productivity and mortality; and (2) the rate of recolonization of tracts of suitable landscape that lose all their animals or all the individuals of one sex. The likelihood of recolonization of a vacant tract of suitable landscape, as well as mixing between existing groups, is a function of several factors: the number of dispersers (in turn dependent upon the size and productivity of the groups found in nearby landscape tracts); the distance between the tracts; the quality of the biological corridors or habitat matrix between suitable tracts; and the ability of goshawks to disperse. Having "suitable" tracts of landscape which are smaller (or of marginal quality) and farther apart, leads to more frequent extinctions of groups within the tracts and slower recolonization of vacant habitats (Lande 1987, 1988; Thomas et al. 1990).

CONSERVATION STRATEGY

Habitat Conservation Areas (HCAs)

One goal of my proposed conservation strategy is to provide tracts of suitable landscapes that are large enough and productive enough that their groups of goshawks are somewhat self-perpetuating. Another goal is to assure that when local extinction does occur that recolonization from nearby tract(s) is probable or, even better, to have dispersers from a nearby tract "rescue" the group in question before it is lost. Tracts of suitable landscape within an Ecological Province or Subprovince could be sized and distributed in various manners: smaller groups of goshawks (more prone to local extinction) in tracts that are closer together (for more frequent recolonization or rescue); or larger groups (less prone to local extinction) within tracts farther apart (less interaction between groups). Most importantly, extinction of an entire deme (perhaps an Ecological Subprovince) should rarely or never occur, because (1) every deme should be largely self-perpetuating, and (2) following a negative event the deme should be "rescued" by dispersers from an adjacent deme before deme extinction can occur. These same goals may be adequate to assure genetic mixing and to prevent in-breeding depression.

Thomas et al. (1990) proposed a somewhat similar conservation strategy for northern spotted owls because logging was making suitable habitat islands (i.e., tracts of landscape dominated by old-growth forest) smaller and farther apart. They described in detail why maintaining habitat for only individual pairs, or very small groups, of owls would probably not perpetuate the species. Reasons included the likelihood that a pair or small group frequently dies out, after which the relatively small tract of empty habitat probably will not be found by dispersing birds. Also, the sex ratio of a group residing in a small tract of suitable landscape easily becomes unbalanced, and it is unlikely that a dispersing bird of the right sex would find a relatively small tract at the appropriate time. Furthermore, a small tract of suitable old-growth habitat is more prone to competition from early succession and open-country raptors. Finally, a tract of landscape, which is similar in size and composition to the average known home range, may lack some type of habitat that is essential over the long-term, and so a seemingly suitable tract really may not be suitable for long-term survival and productivity of an individual pair of birds.

Thomas et al. (1990) reviewed the literature on a variety of bird species, and determined that 20 pairs were typically necessary in a landscape tract to adequately reduce its extinction frequency. They therefore recommended the protection of tracts of landscape with enough habitat capability for 20 pairs of spotted owls. To ensure recolonization when the species becomes extinct in such tracts, and to ensure adequate gene flow, Thomas et al. (1990) also recommended that the 20-pair tracts be located no farther apart than the dispersal distance of 67% of all juveniles.

The concept used for the spotted owl strategy appears sound for the Queen Charlotte goshawk. Although it is doubtful that the numerical objectives are exactly transferable between the species, data to conduct population modelling for the Queen Charlotte goshawk are lacking. Therefore, both the "20 pair rule" and the "67% rule", developed for spotted owls, might be optimum for use with goshawks until research shows otherwise.

Information does not exist on dispersal of Queen Charlotte goshawks; however, dispersal studies on other subspecies indicated the 67% rule could give distances ranging from 10 to 30 mi (see MOVEMENTS/DISPERSAL). Because the studies on goshawk dispersal have shown divergent results, goshawks may possibly have more plasticity in their capability to seek vacant habitat than do spotted owls. It is therefore recommended that 30 mi be set as the maximum distance between edges of landscape tracts if tracts are large enough for 20 pairs of goshawks.

Thomas et al. (1990) recognized that tracts of landscape, with enough capability to support 20 pairs of birds, may not occur in some areas. For tracts with habitat capability for 10 pairs, they recommended separation no greater than the median dispersal distance of spotted owls, and a distance less than the dispersal distance of 75% of all juveniles. This distance amounted to about half that used for 20-pair tracts. Since adequate research has not occurred on the Queen Charlotte goshawk, I will assume that 15 mi would be the appropriate maximum distance between edges of landscape tracts if all tracts were chosen to support 10 pairs of breeding goshawks.

Also, to serve the objectives of other old-growth species (see other reports this publication), it seems appropriate to use even smaller tracts of suitable landscape, to shorten the separation between the tracts, and to diversify tract sizes. A mixture that might be adequate, for all the species of concern on the Tongass National Forest, uses tracts large enough for 8 pairs of goshawks (here called Large HCAs or Large Habitat Conservation Areas), along with tracts large enough for 2 pairs of goshawks (Medium HCAs). The Large HCAs would need to be less than 20 mi apart. The Medium HCAs would be within 8 mi of Large HCAs, or within 8 mi of other Medium HCAs. No potential forest land anywhere would be over 8 mi from a Large or Medium HCA.

The Interagency Viable Population Committee (this publication) defined HCAs as needing at least 50% old-growth forest of over 8 mbf/ac. HCAs also include at least 25% old-growth forest of over 20 mbf/ac. The Committee also defined the size of the total landscapes: Large HCAs at 40,000 ac or larger, and Medium HCAs as at least 10,000 ac.

If landscapes are chosen which only minimally meet the definitions, then the HCAs might often fail to meet the necessary population objectives for goshawks (8 pairs in Large HCAs and 2 pairs in Medium HCAs). The model of Crocker-Bedford (1990a) had estimated the density of Queen Charlotte goshawks at 0.7 pair per 10,000 ac of total landscape, where the landscape was half old-growth forest with over 8 mbf/ac. At this density a 40,000-ac HCA, that minimally met composition guidelines, would support only 3 pairs of goshawks rather than the requisite 8 pairs for a large HCA. This concern is exacerbated by the fact that the Crocker-Bedford (1990a) model apparently grossly overestimated the population density of Queen Charlotte goshawks (see POPULATION DENSITIES AND TRENDS---Southeast Alaska).

Protection of Individual Pairs

A concerted effort should be made to locate goshawks, and to maintain adequate habitat for all pairs of goshawks, within locales that are scheduled for timber harvest. This is because the remaining total international population of Queen Charlotte goshawks is similar in size to that thought minimally viable (see DISTRIBUTION AND POPULATION STATUS, also Thomas 1990, Mace and Lande 1991). Furthermore, current habitat knowledge may not be adequate to assure selection of HCA's that meet the population objectives of Large and Medium HCAs.

Crocker-Bedford (1990b) speculated that it might suffice to manage goshawks by extending the timber rotation period over 7,500-15,000 ac around each goshawk territory. Reynolds et al. (1991) suggested an even longer rotation (doubling the timber rotation period in Southwest from 120 years to 200-300 years) within each home range estimated at 6,000 ac. Crocker-Bedford (1990b) called for a 420-1,600 ac permanent no-cut buffer around the cluster of nests of each pair. Reynolds et al. (1991) recommended full protection of 180 ac for nest sites within any one territory, as well as very conservative management within a 600-ac post-fledgling area (PFA---see Kennedy 1989).

For Queen Charlotte goshawks located outside of Large and Medium HCAs, I suggest combining the above recommendations. Fully protect the central core area of each adult male (1,600 ac---Kennedy 1989), as this area may be critical for food for the goshawk family. Reduced productivity of pairs following timber harvest has been measured (Crocker-Bedford 1990b, 1991; Zinn and Tibbitts 1990; Patla 1991). Hatching rate may be partly related to the amount of prey collected by the adult male, because if inadequate prey are delivered by the male to the nest the female may be more likely to forage and leave her eggs unattended. The quality of nearby foraging habitat is logically related to the amount of prey delivered to the nestlings, which is closely associated with the number of nestlings that survive long enough to fledge. Therefore, while it would seem important to carefully manage the entire home range for quality of hunting habitat, the male's concentrated foraging area nearest the nest would seem to be the most important habitat within its home range.

Careful management of the post-fledging area (PFA) is logically important (Kennedy 1989, Reynolds et al. 1991). Even so, to my knowledge mortality of fledglings has not been proven to be affected by logging.

The 1,600-ac set aside, which I propose for the core area of the adult male Queen Charlotte goshawk, could also suffice for the Small HCA (1,600 ac) recommended by the Interagency Viable Population Committee (this publication) for each major watershed in the Tongass National Forest. No trees should be harvested within the core area of the adult male, except as required for necessary roads and recreational developments.

Beyond the 1,600-ac core area I suggest that habitat management, for individual pairs of Queen Charlotte goshawks, should somewhat correspond to standards recommended by Reynolds et al. (1991) for the Southwest. Thus, for the estimated home range outside the male core area, if all timber were legally available I would suggest doubling the timber rotation length---to 200 years in most sites on the Tongass National Forest. In essence, within the home range but outside the male core area, this strategy would allow only 5% of the old-growth forest having over 8 mbf/ac to be removed in any one decade. However, because much forest is already off limits to timber harvesting (riparian buffers, unsuitable soils, etc.) the true timber rotation period might not increase significantly. For example, in a home range where half the potentially timbered acreage is already unsuitable for timber harvest (not unusual on the Tongass), then harvesting 5% of the total timber acreage from the half that is suitable would really be harvesting 10% of the suitable acreage in any one decade---equal to the normal 100-year rotation on the Tongass. The effect on long-term timber yield would depend upon the amount of forest already designated unsuitable. All clearing widths (harvesting) for roads should be included when calculating the 5% of the timbered land which could be harvested in any one decade.

The quality of habitat is also important (see PATTERNS OF HABITAT USE). Therefore, any harvesting that occurs in the home range, outside of the male core area, should be proportional to the timber volume classes present or, better, emphasize the harvesting of lower volume stands.

Goshawk home ranges are typically 4,900 to 8,000 ac (Reynolds 1983), but they are rarely round. Kennedy (1989) found that the 95%-use areas of males, during the breeding season when the males provided most of their familys' food, extended as far as 5.0 mi from their nests (equal to a circle of 50,000 ac), and use was recorded as far as 6.2 mi away. Even the most intensively used foraging areas extended as far as 3.1 mi from active nests (equal to a circle of 20,000 ac). Therefore, many errors would probably occur in designating home range boundaries, which would reduce the effectiveness of management for individual, historically known pairs.

Goshawk surveys frequently fail to locate nest sites. Therefore, I recommend implementing the above management strategy anytime there is some evidence of a goshawk nesting territory outside of permanently established HCAs. Evidence to consider includes "the number of sightings, time of year of the sightings, courtship behavior, presence of juvenile birds, presence of plucking posts, and territorial behavior" (USDA Forest Service 1991b), in addition to obvious evidence such as nests.

RESEARCH RECOMMENDATIONS

In the spring of 1991, the Tongass National Forest and the Alaska Department of Fish and Game began a cooperative study on the Queen Charlotte goshawk. If funding persists, the study will last at least 3 years.

The study's goals are to:

Determine habitat associations that are used and those that are preferred;

Determine goshawk locations within timber sale assessment areas;

Recommend methods of timber harvest that maintain a viable and well distributed population.

The study's objectives are to:

Survey for presence and absence in relation to landscape habitat features;

Locate territories and home ranges within timber sale assessment areas;

Evaluate the habitat and cover types within home ranges, and compare these to composition beyond the home ranges;

Determine home range, patch size, and habitat preferences through radio-telemetry;

Determine the dispersal distances, especially juveniles;

Prepare management recommendations that will assist in future forest management.

Develop a data base of known pairs and nest sites, which can later be used as an aid in monitoring population trends and effectiveness of habitat management.

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Table 1. Densities of pairs of breeding goshawks in western coniferous forests, as compared to the intensity of timber harvest.

Number of Pairs/ 10,000 ac land	Timber Harvest	Location	Source
0.0	Much	N.W. Oregon	Reynolds and Meslow 1984
0.1	Much	South Dakota	Bartelt 1977
0.4	Fragmented	California	Bloom et al. 1985
0.5[a]	30% Selected	N. Arizona	Crocker-Bedford 1990b
0.8[b]	Much Selected	N. New Mexico	Kennedy 1989
0.8	Little logging, but much fire	Central Alaska	McGowan 1975
1.3	Limited	California	Bloom at al. 1985
1.5	Limited	Oregon	Reynolds and Wight 1978
3.0	Little	Colorado	Shuster 1976
4.4	Light salvage and selection	N. Arizona	Crocker-Bedford and Chaney 1988
9.0	None	N. Arizona	Crocker-Bedford 1990b

[a] And only 0.5 nestling per pair.

[b] Does **not** include the 25% of all territories where the female was unpaired. Unusually low reproduction even where paired.

17 April 1992 - Review Draft

CONSERVATION OF THE BOREAL OWL IN SOUTHEAST ALASKA

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POPULATION STATUS AND DISTRIBUTION

The boreal owl (*Aegolius funereus*) occurs in a holarctic distribution in boreal climatic zones and mountain ranges (Voous 1960). Few bird species are so characteristic of the northern coniferous forest (Voous 1988). Five subspecies have been recognized: *A.f. richardsoni* in North America; *A.f. funereus*, *sibiricus*, and *magnus* throughout Eurasia; and *A.f. caucasicus* in the Caucasus, western China, and the western Himalayas (Mikkola 1983).

The boreal owl breeds across North America from the tree line to central Canada with scattered populations in the northern and central U.S. Rocky Mountains (Johnsgard 1988:220). Recent surveys suggest that populations may exist throughout the mountains of Oregon and Washington (O'Connell 1987, G.D. Hayward, Colo. State Univ., pers. commun.) Gabrielson and Lincoln (1959) indicated that this bird was a rare resident throughout the forested areas of the mainland in Alaska. Isleib and Kessel (1973) considered the boreal owl a rare resident of the north Gulf Coast-Prince William Sound region. They estimated the population to be no greater than a few hundred individuals in this area. Armstrong (1980) reported the boreal owl to be uncommon in southcoastal Alaska and occurring casually or accidentally in southeast

Alaska. Taylor (1979) listed the species as uncommon in southeast Alaska and as an uncommon breeder in southcentral Alaska. Recent surveys of forest owls conducted during the breeding season have documented the presence of boreal owls throughout southeast Alaska (Table 1). However, rates of detection have been low.

PATTERNS OF HABITAT USE

Nesting habitat

Throughout their range, boreal owls tend to select uneven-aged, old growth habitats with large trees, small canopy gaps, and a shrub understory (Johnsgard 1988:222). All but 2 of 23 nest sites located in Idaho were found in extensive forest blocks (Hayward 1989). Boreal owls from the same study area were reported to use coniferous stands having well developed low and high canopies (Hayward and Garton 1988). Large expanses of forest with unbroken canopies (i.e., second-growth forests) are avoided. Nesting occurs in cavities excavated by woodpeckers (Bondrup-Nielson 1979, Palmer 1986, Hayward et al. in review a). The cavities used by boreal owls generally have entrance holes with a diameter greater than 3 in. This requirement may limit the availability of suitable nest sites for this species in southeast Alaska. Of the cavity excavators present in southeast Alaska, only the northern flicker (Colaptes auratus) and perhaps the hairy woodpecker (Picoides villosus) excavate holes large enough for this owl to use (Harrison 1979). However, suitable nest structures may occur when cavities are enlarged by mammals or in cavities created from broken limbs (G.D. Hayward, Colo. State Univ., pers. commun.).

Foraging Habitat

Boreal owls prey upon small forest-adapted rodents, especially microtines, which are primarily captured nocturnally (Johnsgard 1988:224). In Idaho, redback voles (Clethrionomys gapperi) and northern flying squirrels (Glaucomys sabrinus) were the most important prey items of boreal owls (Hayward et al. in review a). The birds hunt primarily under forest cover where their prey is more available (Norberg 1970, Sonerud 1986, Sonerud et al. 1986). The owls do hunt the edge of clearcuts in the spring following snowmelt but before green-up. Following early spring, boreal owls again hunt in the forest. Boreal owls avoid hunting in the same area on successive nights.

Field studies have shown diverse results in the size of foraging range. Their hunting range extends from a radius of 3900 to 5900 ft around the nest (i.e., 1120 - 2500 ac) (Sonerud et al. 1986). Korpimaki (1987) reported the longest foraging trips made by the males extended up to 2.5 mi from the nest. However, he also indicated that the intensive foraging area around the nest is restricted to approximately 740 ac. Bondrup-Nielson (1978) reported foraging areas that ranged from 250 to 1235 ac. Hayward et al. (1987) reported the mean distance between daytime roosts of male boreal owls and nest sites during incubation and nestling periods to be 1.5 mi. Hayward et al. (in review a) also reported that boreal owls in Idaho frequently hunted over 3 mi from the nest site.

Productivity of habitats directly affects activity level and productivity of boreal owls. Boreal owls utilizing habitats with low prey populations will have to forage longer and over a wider area than owls in more productive habitats. Consequently, productivity of boreal owls nesting in habitats with high prey density is greater than that of owls nesting in habitats with low prey densities (Korpimaki 1988).

Roosting Habitat

Boreal owls tend to roost by perching in trees rather than using cavities (Hayward and Garton 1984, Hayward et al. in review a). Roosts are chosen to provide both thermal and hiding cover. Dense stands of coniferous trees are selected as roost sites (Bondrup-Nielson 1978, Hayward and Garton 1984, Palmer 1986).

HOME RANGE/TERRITORY

Home ranges of boreal owls tend to be large but overlap extensively (Hayward et al. 1987). Year-round home ranges averaged over 5,000 ac in Idaho (Hayward 1989). However, seasonal requirements (e.g., relief of heat stress during summer) were met in different areas necessitating relatively long movements by these birds. Extensive seasonal movements may not be required in southeast Alaska because environmental extremes are moderated by the maritime climate.

Males may defend only small territories within home ranges (5.6 ac as reported by Bondrup-Nielson [1979]). However, Meehan (1980) reported the closest singing males in her study area in interior Alaska to be about 1 mi apart.

Territorial activities are confined to the nest sites which may be in limited supply (Solheim 1983, Johnsgard 1988:226).

POPULATION DENSITIES

Although few estimates of population densities are available, data summarized from Europe indicate that densities are generally low (Johnsgard 1988:223). Densities varied from 1 pair per 3100 ac to 1 pair per 770 ac on a 9.6 mi^2 study area in Finland (Korpimaki 1981). Bondrup-Nielsen (1978) estimated a density of 1 bird per 2800 ac in Canada and Meehan (1980) estimated a density of 1 singing male per 2745 ac near Fairbanks. Density is usually determined by food supply and nest site availability (Korpimaki 1988) The limited number of responses during surveys in southeast Alaska indicate that the density in this area may be lower than all those listed above (Table 1).

MOVEMENTS/DISPERSAL

Periodic population movements occur that may be related to population cycles in small mammals (Johnsgard 1988:224). Adult males appear to remain sedentary while females and young tend to move more readily (Mikkola 1983:267, Schwerdtfeger 1984, Korpimaki and Hongell 1986).

Hole nesting tends to favor residency (Haartman 1968). Since suitable nest sites are limited, adult male boreal owls remain in the vicinity of their nest site throughout the year and through prey fluctuations to maintain their territory. Adult males may change actual nest sites within a 1.8 mi radius (i.e., 7040 ac) from year to year (Lofgren et al. 1986). Eighty percent of adult males moved less than 1.2 mi between successive years and 86% of adult males moved less than 1.9 mi over a 2 to 4 year period (Korpimaki 1987). In another study, the mean distance moved by adult males was 0.6 mi and the maximum movement was 3.1 mi (n = 23) (Korpimaki et al. 1987).

Although adult females are capable of moving long distances in response to fluctuations in the prey base, Sonerud et al. (1988) reported that 69% of adult females moved less than 12.4 mi even during declines of micotines. During periods of high prey populations adult females remained within the previous year's home range. The median dispersal distance between breeding sites by adult females through all phases of the prey cycle was reported to be 2.5 mi (n = 75) (Korpimaki et al. 1987). Hayward et al. (in review a) also witnessed a varied strategy of site tenacity and long distance movements in North America.

Juvenile females tend to disperse further than juvenile males. The median dispersal distance of 3 juvenile males was 3.6 mi (range 3 - 6.8 mi) while the median dispersal distance of 9 juvenile females was 6.2 miles (range 4.8 - 148 mi) (Sonerud et al. 1988). Juvenile females were reported to disperse up to 16 mi before first nesting while juvenile males dispersed up to 2.8 mi (Lofgren et al. 1986). A third study reported median dispersal distance of 55 mi (range <3 - 400 mi, n = 37) for juvenile females and 13 mi (range <3 - 50 mi, n = 13) for juvenile males (Korpimaki et al. 1987).

VIABILITY/DISTRIBUTION CONCERNS

The preceding review of the distribution and natural history of the boreal owl indicates that it occurs throughout southeast Alaska, probably in low numbers, and that its continued viability and distribution in this area may be affected by direct removal of habitats through timber harvest. Meehan and Ritchie (1982) considered forest removal through logging to be the management practice having the greatest impact on boreal owls. Reynolds et al. (1989) indicated that most threats to populations of boreal owls are associated with forest management practices. These threats include: 1) loss of nesting sites through removal of snags and cavity bearing trees, 2) change in the composition and/or abundance of prey as a result of changes in the composition and structure of overstory and understory, and 3) elimination of the patchy structure of mature and old growth forests (Hayward and Hayward 1989, Reynolds et al. 1989).

Viability may be a concern for the boreal owl in southeast Alaska because of its:

1) apparent low density,

2) limited availability of nest sites,

3) selection of habitats that are affected by timber harvest, and

4) apparent large home range size.

CONSERVATION STRATEGY

In the past, habitat protection for raptors emphasized protection of known nest sites, most of which were found accidentally (McCarthy et al. 1989). This approach ignored nest stand and foraging habitat requirements. Management of foraging habitat may be the key component in managing for populations of boreal owls (Garton et al. 1989, Hayward 1989). A more comprehensive management strategy is needed to ensure that wildlife objectives (e.g., continued viability and distribution of boreal owls) are incorporated into the management of forests in southeast Alaska. Survival of the boreal owl depends on how well timber management and habitat management for this owl are integrated (Hayward and Hayward 1989).

A management strategy that should ensure boreal owl viability and distribution of populations throughout southeast Alaska includes providing Habitat Conservation Areas (HCAs) of old growth forest (i.e., at least 8,000 bf per ac) 5,000 ac in size, or larger. The HCAs should be approximately 10 mi apart edge to edge and distributed across the landscape. Each block would provide habitat for 1 to 13 pairs of boreal owls, depending on habitat quality. Reported dispersal distances of juvenile males vary, but they indicate that habitat blocks should not be more than 10 mi apart to ensure continued occupancy of habitats.

An alternative strategy would be to 1) manage the 5,000 ac HCAs allowing a low intensity timber harvest (i.e., maintain 60% of the old growth, harvest 40% using group selection), 2) provide an additional 5,000 ac of old growth forest for travel corridors between blocks with moderate intensity timber harvest (i.e., maintain 40% of the old growth, harvest 60% using group selection), and 3) manage the remainder of the watershed under intensive forest management (i.e., clearcut) (G.D. Hayward, Colo. State Univ., pers. commun.).

When individual nests are located outside of the maintained blocks they should be protected with a 1/2 mi buffer within which timber harvest would not be allowed. This will provide approximately 500 ac of habitat adjacent to the nest. Foraging areas around the nest site have been reported to range from 250 to 4,500 ac (Bondrup-Nielson 1978, Sonerud et al. 1986, Hayward et al. 1987). A 500 ac buffer would maintain a minimum amount of foraging habitat in association with the nest site.

MONITORING RECOMMENDATIONS

Forest owls have been surveyed in southeast Alaska from 1986 through 1990 to determine their occurrence and distribution (USDA Forest Service unpublished data). Most surveys were conducted along roads and consisted of noting responses of owls to broadcasts of recorded conspecific owl calls and songs (Suring 1990). Surveys such as these should be expanded throughout southeast Alaska. Playback surveys conducted each year over a large area may be useful in detecting overall trends (Hayward et al. in review a). Sampling techniques should be further formalized to ensure that results of the surveys provide meaningful information on population trends.

However, playback surveys may not be a useful technique to assess population trends of boreal owls on a local scale or to assess response to habitat change (Lundberg 1978, Hayward et al. in review b). A number of factors may affect the calling rate of boreal owls including time of night, current and past weather conditions, physiological condition of owls, competition for nest sites, and mating status (Hayward et al. in review b). Hayward et al. (in review b) have suggested that nest boxes may be used to assess abundance and productivity of boreal owls for intensive monitoring on a local scale.

RESEARCH RECOMMENDATIONS

Knowledge of boreal owls in southeast Alaska consists of limited information concerning their occurrence and distribution (USDA Forest Service unpublished data). Results of studies conducted in other locations (especially northern Europe and the northern U.S. Rocky Mountains) indicate these owls have an affinity for old growth habitats and that they may be sensitive to forest management practices. Reported home ranges tend to be large for an owl of this size. It also appears that the dispersal capabilities of the males may be limited. Studies should be implemented in southeast Alaska to determine habitat relationships of the boreal owl so that management standards and guidelines can be designed to respond to the specific requirements of boreal owls in this area.

Research efforts should be incorporated with management practices to determine the response of boreal owls and their prey to alternative timber harvest strategies. Reproduction and mortality patterns of boreal owls also need to be determined to so that we may evaluate their population status.

ACKNOWLEDGEMENTS

D. C. Crocker-Bedford, G. D. Hayward, P. F. Schempf, L. C. Shea, and K. Titus provided helpful review comments on earlier drafts of this manuscript.

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Table 1. Summary of search effort and boreal owl detections during owl surveys in southeast Alaska, 1986 - 1990.[a]

Survey period	1986	1987	1988	1989	1990	Total
Number of calling bouts completed	63	108	118	115	131	535
Number of owl observations by location						
Southern Prince of Wales Island	--[b]	1 (.01)[c]	0	0	0	1 (<.01)
Northern Prince of Wales Island	--	--	0	--	7 (.05)	7 (.01)
Revilla Island	1 (.02)	--	--	0	3 (.02)	4 (.01)
Wrangell Island	0	0	--	--	[1][d]	0
Mitkof Island	0	0	0	0	0	0
Baranof Island	0	--	--	--	--	0
Juneau Mainland	--	--	3 (.03)	0	0	3 (.01)
Total	1 (.02)	1 (.01)	3 (.03)	0	10 (.08)	15 (.03)

[a]Source: unpublished data, USDA Forest Service, Juneau, Alaska.

[b] -- indicates that owl surveys were not run in that area during that year.

[c]Numbers in parentheses are number of detections per calling bout.

[d]This detection was not made on a survey route and was not included in the total.

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CONSERVATION OF THE NORTHERN HAWK OWL IN SOUTHEAST ALASKA

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DISTRIBUTION AND POPULATION STATUS

The northern hawk owl (*Surnia ulula*) has a continuous holarctic distribution from Alaska, throughout Canada, across Scandinavia, and through Russia and Siberia (Figure 1, Voous 1988:132). Two subspecies occur in North America; breeding populations of *S. u. caparoch* are distributed across northern North America and *S. u. ulula* occurs accidentally in western Alaska (American Ornithologists' Union 1957).

This bird breeds in Alaska from the tree line in the west and central parts of the State southward through southeast Alaska (Gabrielson and Lincoln 1959). Isleib and Kessel (1973) considered the northern hawk owl an uncommon resident of the North Gulf Coast-Prince William Sound region. They estimated the population in this area to be a few hundred individuals. Armstrong (1980) reported the northern hawk owl to be common in central Alaska, uncommon in southcoastal Alaska, and uncommon to a casual visitor in southeast Alaska. Taylor (1979) listed the species as uncommon in southeast Alaska and as an uncommon breeder in southcentral Alaska. Kessler and Kogut (1985) reported the northern hawk owl to be rare during spring and summer bird surveys conducted in southeast Alaska. Recent surveys of forest owls conducted during the breeding season have further documented the presence of northern hawk owls in southeast Alaska (Table 1).

The population status or trends of this species are difficult to assess because of the remoteness of their habitats and problems in censusing these birds (Johnsgard 1988:150). However, Mikkola (1972) has suggested that the population in Fenno-Scandia has declined throughout the last century. Walker (1974:78) has also indicated that northern hawk owls have declined in North America.

PATTERNS OF HABITAT USE

Nesting Habitat

The preferred breeding habitat of northern hawk owls is coniferous or mixed forest near openings (Mikkola 1983:109). Fifty percent of nests found during a study in Scandinavia were in open areas (i.e., bogs and clearcuts) with scattered trees, 30% were in open spruce forests, and 20% in closed spruce forests or in the ecotone between closed and open forests (n = 16) (Sonerud 1985). Another study in Scandinavia characterized breeding habitat as having sparse vegetation (Ims 1982). Nests in Alberta have usually been located in muskeg areas (Jones 1987). In interior Alaska, northern hawk owls nested in open-canopied forests (i.e., 20% to 60% canopy cover) or at the forest edge (Meehan and Ritchie 1982). Sidle (1985) reported that this bird was associated with open shore pine (Pinus contorta) - western hemlock (Tsuga heterophylla) forests in southeast Alaska.

The most frequently noted sites of nests of the northern hawk owl are natural cavities in trees or snags, open hollows where tops of trees have broken off, and cavities excavated by woodpeckers (Voous 1988:135). Occasionally old stick nests of other raptors or crows (Corvus spp.) are used as nest sites (Pullianinen 1978, Mikkola 1983:109). Nests have been reported to be anywhere between 5 ft and 40 ft above the ground (Pulliainen 1978, Lane and Duncan 1987, Johnsgard 1988:148). The only nest site described in the literature for which measurements were reported was a 21 ft high snag with a 50 in diameter-at -breast-height (Lane and Duncan 1987).

Foraging Habitat

The birds hunt in open areas that have adequate perch sites, such as muskegs and muskeg forests (Mikkola 1983:108). During the breeding season these owls primarily eat voles (Mikkola 1983:108-109). Of 1,451 prey items identified from northern Europe, the great majority were voles (Microtidae); Microtus spp. and Clethrionomys spp. made up 80.8% and other voles 15.6% (Mikkola 1972). Outside of the breeding season a distinct shift is made to avian prey (e.g., 41% voles and 31.8% birds [Mikkola 1972]). Birds as large as willow ptarmigan (Lagopus lagopus) and ruffed grouse (Bonasa umbellus) are included in their diet (Mikkola 1972, Axlerod 1980).

The northern hawk-owl's predominant mode of hunting includes visual searching from a perch followed by a rapid pursuit flight. It is the most diurnal of the owls, sometimes hunting during bright daylight and never hunting in the dark (Johnsgard 1988:147). This owl may, at times, travel more than 0.5 mi from the nest site to obtain prey (Mikkola 1983:107). Observations of a single female during the breeding season indicated she did not move more than 0.4 mi from her nest (Sonerud et al. 1987).

HOME RANGE/TERRITORY

Home ranges reported from Norway varied from 350 to 2,100 ac with an average of 920 ac (Baekken et al. 1987). Territorial behavior by northern hawk owls has been described within their home ranges (Robiller 1982). Territories are apparently large with nest sites well separated from one another (Mikkola 1972). Observations of a pair of birds in central Alaska indicated that their home range may have been less than 250 ac (Kertell 1982).

POPULATION DENSITIES

Reported breeding densities of this owl are very low. Four pairs were reported from an area of 77 mi^2 in Norway (Hagen 1956). Good habitat in Sweden may support about 1 pair per 190 mi^2 (Johnsgard 1988:146).

MOVEMENTS/DISPERSAL

Local breeding densities and distribution of northern hawk owls vary in response to fluctuations of prey populations. Movements are most pronounced in adult females, least in adult males and juvenile females, and intermediate in juvenile males (Byrkjedal and Langhelle 1986).

VIABILITY/DISTRIBUTION CONCERNS

This owl occurs in southeast Alaska, apparently in low numbers. Meehan and Ritchie (1982) considered forest removal through logging to be the single management practice having the greatest potential impact on northern hawk owls. However, its foraging habitat (i.e., forest muskegs, open muskegs) will not be significantly affected by current forest management actions. Although this bird is somewhat flexible in its selection of nest and foraging sites, snags and associated cavities are important for hunting perches and nest sites. Loss of nest sites through removal of snags and cavity-bearing trees as a result of forest management practices has been identified as a significant threat to owl populations (Reynolds et al. 1989).

CONSERVATION STRATEGY

A comprehensive snag retention policy that incorporates the specific needs of the northern hawk owl for nest sites and perches should be developed and implemented throughout the Tongass National Forest to ensure the distribution of birds is maintained. Perch trees should be retained in clearcuts to provide hunting sites. Small patches of trees with snags and potential snag replacement trees should be maintained within clearcuts to provide nest sites.

MONITORING RECOMMENDATIONS

Forest owls have been surveyed in southeast Alaska from 1986 through 1990 to determine their occurrence and distribution (USDA Forest Service unpublished data). Most surveys were conducted along roads during darkness and consisted of noting responses of owls to broadcasts of recorded conspecific owl calls and songs (Suring 1990). Surveys specifically designed to detect northern hawk owls should be developed and implemented throughout southeast Alaska. Sampling techniques should be used that will ensure that results of the surveys provide meaningful information on population trends.

RESEARCH RECOMMENDATIONS

Knowledge of northern hawk owls in southeast Alaska consists of limited information concerning their occurrence and distribution. Results of studies conducted in other locations (especially northern Europe) indicate these owls require snags and cavity bearing trees and that they may be sensitive to forest management practices. Studies should be implemented in southeast Alaska to determine habitat relationships of the northern hawk owl so that management standards and guideline can be designed to respond to the specific requirements of these owls in this area.

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Southern Prince of Wales Island	--[b]	0	0	0	0	0
Northern Prince of Wales Island	--	--	0	--	0	0
Revilla Island	0	--	--	0	0	0
Wrangell Island	0	0	--	--	0	0
Mitkof Island	0	1 (.01)[c]	0	0	0	1 (<.01)
Baranof Island	0	--	--	--	--	0
Juneau Mainland	--	--	1 (.01)	0	0	1 (<.01)
Total	0	1 (.01)	1 (.01)	0	0	2 (<.01)

[a]These surveys were not designed to maximize detection of northern hawk owls (i.e., surveys were run at night when northern hawk owls are least active). Source: unpublished data, USDA Forest Service, Juneau, Alaska.

[b] -- indicates that owl surveys were not run in that area during that year.

[c]Numbers in parentheses are number of detections per calling bout.

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THE ALEXANDER ARCHIPELAGO WOLF

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SUMMARY

The wolf (Canis lupus) was at one time widely distributed throughout the northern hemisphere. Today, North American populations of wolves are limited primarily to Alaska and Canada, with small numbers reported in Minnesota, Wisconsin, and Montana. Within Alaska, the Alexander Archipelago wolf (C. l. ligoni) is considered a separate subspecies (Pedersen 1982). Its range includes the islands south of Frederick Sound and the narrow mainland strip of land lying west of the Coast Mountains and extending from Dixon Entrance northward to Yakutat Bay (Hall 1981). The total population in this region is estimated at 690 individuals (Morgan 1990).

The Alexander Archipelago wolf is generally distinguished from other subspecies by its smaller size and dark pelage. Evidence suggests that wolves moved into southeast Alaska from the south, probably following the post-glacial migration of black-tailed deer (Odocoileus hemionus) from southern British Columbia (Klein 1965). This theory is consistent with the findings of Friis (1985), who documented strong similarities in the cranial characteristics of the Vancouver Island wolf (C. l. crassodon) and the Alexander Archipelago wolf. These wolves probably represent remnant populations of a now-extinct

type that once populated the coastal rainforests of southwestern British Columbia, Oregon, and Washington.

Research on the ecology and habitat requirements of the Alexander Archipelago wolf is limited. Because of the highly dissected, discontinuous nature of its island habitat, pack sizes, territories, and movement patterns are probably much smaller than that exhibited by mainland wolves. Additionally, in southeast Alaska the habitat which supports wolves and their prey is being significantly altered by man. Formerly pristine drainages are being accessed by a rapidly expanding road system (186 mi/year), and timber harvesting is planned on over 1.97 million ac of productive old-growth forest (U.S. Forest Service 1991). Direct mortality can be expected as roads bring man into increasing contact with wolves, and wolves will be affected indirectly by logging-related reductions in Sitka black-tailed deer (*O. h. sitkensis*).

Although wolves are not in danger of extirpation in southeast Alaska, significant long-term declines can be expected in intensively developed areas. In order to maintain viable, well-distributed populations of wolves throughout their current range, the following standards and guidelines are recommended: (1) road densities should be held below 1.0 mi/mi^2 in individual wildlife analysis areas, and (2) habitat sufficient to support at least 5 $deer/mi^2$ should be provided in areas where deer are the primary prey species.

BIOLOGICAL INFORMATION

Physical Appearance

The Alexander Archipelago wolf tends to be darker, smaller, and shorter-haired than wolves in northern and interior areas of Alaska (Mech 1970, Wood 1990). On islands in southern southeast Alaska, the black color phase comprises about 20% of the population, grey/brown wolves about 80%, and white or near-white wolves less than 1% (Wood 1990). On the northern mainland the black color phase is more common, comprising about 50% of the harvest (Alas. Dep. Fish and Game, unpubl. data). Adult wolves in southeast Alaska weigh an average of 87 lbs., and rarely exceed 100 pounds; females weigh about 15 lbs. less than males (Alas. Dep. Fish and Game. 1960, Wood 1990).

Distribution

Wolves occur on the mainland and all large islands in southeast Alaska except for Admiralty, Baranof, and Chichagof islands (Game Management Unit [GMU] 4). Wolves readily swim distances of 0.5-1.0 mi to reach islands in search of prey (Wood 1990); however, their absence on islands in the northern archipelago indicates that wide waterways pose effective barriers. Wolves are most abundant in the southern panhandle (GMU 2 and 1A, including Prince of Wales Island, Revillagigedo Island, and the Cleveland Peninsula) where there is an estimated 1 wolf per 18-26 mi^2 (Wood 1990). Wolves are less abundant in GMU 3 (Kuiu, Mitkof, Wrangell and Kupreanof islands) where there is 1 wolf per 50 mi^2, and least abundant on the mainland (1 wolf per 75 mi^2 in subunits 1B, 1C, and 1D) where ungulates are relatively scarce and the landscape steep and/or ice-covered (Morgan 1990).

Population Size

The dense forest cover of southeast Alaska makes it difficult to assess wolf numbers accurately. The best available estimates are based on field observations, discussions with trappers, and anecdotal information. From these sources, the wolf population in southeast Alaska is currently estimated at 635-690 individuals, distributed among approximately 85 packs (Morgan 1990). Although wolf densities in GMU 2 (1 wolf per 30-40 mi^2) are relatively high by Alaska standards (Ballard et al. 1987), they are low compared to densities reported in British Columbia (1 wolf per 8-11 mi^2) and the lower 48 states (1 wolf per 10-15 mi^2) (Van Ballenberghe et al. 1975, Hebert et al. 1982, Fuller 1990).

Population Trends

Wolf populations are closely tied to population levels of their ungulate prey (Keith 1983, Messier 1985). Packard and Mech (1980) concluded that intrinsic social factors and the influence of food supply are interrelated in determining population levels of wolves. In situations where prey populations are reduced by other factors (e.g., winter weather), predation by wolves can inhibit the recovery of prey populations for long periods of time (Gasaway et al. 1983, Van Ballenberghe and Hanley 1984). Wolves can maintain themselves at low levels, even in the near absence of ungulate prey, by switching to alternate foods such as beaver (*Castor canadensis*) and salmon (*Onchorynchus* spp.). Under those conditions, it may be many years before prey escape this "predator pit" and return to their initial population density (Skoglund 1991).

In southeast Alaska, census and harvest data indicate that populations of both deer and wolves peaked in the mid-1960s, and declined during the 1970s (Alas. Dep. Fish and Game, Douglas, unpublished data) . With increasing deer numbers during the 1980s (Kirchhoff and Pitcher 1988), the wolf population in Unit 2 is currently increasing. Smith et al. (1987) reported the wolf population in Unit 3 to be stable at relatively low levels. Recent surveys suggest increasing populations on portions of Mitkof, Kupreanof, and Revillagigedo islands (Alas. Dep. Fish and Game, unpub. data), probably in response to locally increasing deer populations.

Food Habits

Wolves have evolved into highly effective predators on large mammalian prey such as deer, mountain goats (Oreamnos americanus) and moose (Alces alces) (Mech 1970). In southeast Alaska, deer are the primary prey on most of the islands and selected mainland areas, whereas on the mainland, the primary prey are beaver, mountain goat, and moose (Smith et al. 1986a; Wood 1990). Unlike interior wolves, wolves in southeast Alaska have access to spawning salmon during late summer and early fall (Smith et al. 1986b, Wood 1990). Wolves will also feed opportunistically upon small mammals, waterfowl, seals, and carrion (Garceau 1960a, Smith et al. 1986b).

Predation Rates

The rate at which wolves kill large mammals varies with prey availability, vulnerability, and environmental conditions. A minimum maintenance requirement for active wolves in the wild is approximately 3.7 lbs of meat per day (Mech 1970). Actual rates of consumption are somewhat higher, averaging 4.4 lb/wolf/day in Minnesota (Fuller 1989), and 4.2 lb/day for captive wolves in southeast Alaska (Garceau 1960b). By making certain assumptions about prey characteristics (type, body size, and edibility) predation rates can be calculated.

In southeast Alaska, the mean weight of adult and fawn Sitka black-tailed deer are 93 lb and 43 lb respectively (Johnson 1987). Assuming 75% of the total weight represents edible portions of the carcass (Ballard et al. 1987), there is 70 lb and 32 lb of potential food per adult and fawn respectively. If approximately 58% of deer killed by wolves are fawns (e.g., Hatter 1984, Fuller 1989), the average yield of food per deer killed is 48 lb. Assuming wolves consume 4.2 lbs/day, and 80 % of their diet is comprised of deer (Hatter 1984), the average wolf consumes about 1,226 pounds of deer meat per year, or 25.6 deer per year. This is within the range of 15-30 deer/wolf/year suggested by Van Ballenberghe and Hanley (1984), and approximately equal to the mean kill rate of 25/wolf/year calculated for Vancouver Island (Hebert et al. 1982).

Population Dynamics

Wolves are a relatively prolific species. First breeding is at 22 months of age, and litters of 3-7 young are produced (Mech 1970, Stephenson 1989). Most packs include a pair of breeding adults, as well as adults that may or may not breed. Mean litter sizes in Alaska, as indicated by counts of blastocysts, range from 4.6 to 7.2 depending on prey availability per wolf (Gasaway et al. 1992). Because the reproductive potential for wolves is high, natural control in the form of direct mortality or social factors must operate to limit population size. Natural and man-caused mortality, rather than failure to breed or produce pups, is generally the major factor limiting wolf population growth (Rausch 1967, Fuller 1989). In southeast Alaska, age-specific survival rates are not known; however, adult and yearling survival rates in a heavily trapped and hunted populations in south-central Alaska averaged 0.59, while pup survival rates averaged 0.36 (Ballard et al. 1987).

Dispersal

Pups that survive to adulthood either remain in their natal pack or disperse. In exploited wolf populations, where a high percentage of adult wolves are hunted or trapped, lone wolves are more likely to be accepted into established packs (Ballard et al. 1987). Dispersers that do not join established packs often form associations with other wolves, occupying vacant areas adjacent to established pack territories (Ballard et al. 1987). Dispersing wolves are more vulnerable to hunting and trapping than non-dispersers, and have a higher probability of being killed by other wolves (Peterson et al. 1984).

Wolves usually disperse in singles or pairs, and make several temporary forays from the main pack before leaving permanently (Fuller 1989). The average age of dispersing wolves in south-central Alaska was 30-33 months, with 40% of dispersing females and 50% of dispersing males being <24 months of age (Ballard et al. 1987). In Minnesota, 17% of the adults, 49% of the yearlings, and 10% of the pups dispersed from the pack each year (Fuller 1989). In Minnesota, dispersers made 6 exploratory moves before finally moving 3-62 mi away and (usually) establishing new packs (Fuller 1989). In Alaska, wolves disperse throughout the year, and may travel over 435 mi from their original home range (Ballard et al. 1987). In southeast Alaska, dispersal rates and distances for wolves have not been documented. It is highly probable, that the discontinuous, island nature of the habitat greatly restricts dispersal.

Mortality

In addition to mortality inflicted directly by man, wolves are killed each year by starvation, accidents, disease, parasites, and fighting (Mech 1970). Human-caused mortality is the most important factor, accounting for most mortality in protected and heavily exploited populations alike (Peterson et al. 1984, Ballard et al. 1987, Fuller 1989). Alaskan studies have shown that a 25-40% harvest of the early-winter wolf population can result in declines (Gasaway et al. 1983, Keith 1983, Peterson et al. 1984). In Minnesota, 28% mortality in the winter wolf population resulted in declines (Fuller 1989).

In recent years, the harvest of wolves in southeast Alaska has been relatively low. Over the past 30 years, the total wolf kill in southeast Alaska has varied from a high of 219 in 1967-68 to a low of 68 in 1981-82. Similar patterns are apparent on individual GMUs. For example, 82 wolves were taken in GMU 3 in 1967-68, but fewer than 11 have been taken annually since 1984-85 (1989-90 harvest = 22). In GMU 2 the kill has increased to 40 wolves per year (about 20% of the estimated population), reflecting increased prey populations and improved human access. Of the wolves killed in GMU 2 since 1985, 46% were either shot or trapped along the road system (Wood 1990).

The highest proportional harvest in southeast Alaska occurs in GMU 1A where 26.5 percent of the estimated population was harvested in 1988-89. Region-wide, the estimated harvest rate in 1988-89 was 14.6 percent (Morgan 1990). Harvest rates in this range are probably not high enough to regulate wolves effectively; however, populations are probably less able to withstand high harvest on islands because of natural barriers to in-migration.

CONSERVATION CONCERNS

Wolf populations may decline in portions of southeast Alaska over the next century as a result of 3 factors:

(1) an expanding road system and increasing human population will have a direct impact on wolves through increased shooting and trapping,

(2) clearcut logging associated with the road system will reduce habitat capability for Sitka black-tailed deer, the wolf's primary prey, and

(3) inbreeding within insular subpopulations may result in reduced fitness. The rationale for these conclusions is presented below.

Roads

Studies in Wisconsin, Michigan, Ontario, and Minnesota have shown a strong relationship between road density and the presence or absence of wolves (Thiel 1985, Jensen et al. 1986, Mech et al. 1988, Fuller 1989). Wolves generally are not present where the density of roads exceeds 0.93 mi/mi2, whereas similar areas nearby with fewer roads contain wolves. Mech (1989) reported wolves using 1 area with a road density above this reported threshold (1.23 mi/mi^2), but it was adjacent to a large, roadless area. Excessive mortality experienced by wolves in the roaded area was compensated for by individuals which dispersed from the adjacent roadless area.

The primary threat of high road densities comes from the accessibility they allow humans who deliberately, accidentally, or incidentally kill wolves by shooting, snaring, or trapping (Van Ballenberghe et al. 1975, Mech 1977, Berg and Kuehn 1982). Despite legal protection for wolves, Fuller (1989) found that 80% of identified mortality in his study area was human-caused. Mech (1989) reported 60% human-caused mortality in a roaded study area (even after full protection), whereas human-caused mortality was absent in an adjacent study area without roads.

The current road density over most of southeast Alaska does not approach the critical threshold level (i.e., 0.9 mi/mi^2), and in many areas never will. Wilderness areas, roadless areas and sparsely forested and/or mountainous lands will not be roaded or logged, and wolves will persist at varying levels much as they have in the past. Where extensive road systems are planned, however, wolves will be at some risk.

Three bioregional provinces in southeast Alaska (Kupreanof, N. Prince of Wales, and central Prince of Wales) support both wolves and deer, are connected by road or ferry to large population centers, and are undergoing intensive road-building and logging. The planned road network for the 50-year sale area for the Ketchikan Pulp Company, for example, will be approximately 2.5 mi/mi^2, roughly 2.5 times greater than the threshold wolves reportedly tolerate. Because of illegal killing, wolves may have trouble persisting in densely roaded areas even with complete regulatory protection from hunting and trapping.

Prey Availability

In addition to the threat posed by increased access, logging permanently reduces the capability of the habitat to support deer, particularly during winters of deep snow. (Wallmo and Schoen 1980, Schoen and Kirchhoff 1990). Not surprisingly, wolf populations decline as their ungulate prey base declines (Gasaway et al. 1983, Peterson et al. 1984, Fuller 1989, Janz 1989).

In northern Minnesota, Fuller (1989) determined that in the absence of hunting, the deer:wolf ratio necessary to maintain a stable deer population was approximately 90:1. In southeast Alaska, deer populations can be expected to decline when the finite rate of increase drops below 20 % per year (i.e., 1.2), and hunting exceeds 20 % of the annual increment (Van Ballenberghe and Hanley

1984). Assuming previously calculated kill rates of approximately 25 deer per wolf per year, and the above rates of finite increase and hunting loss, at least 156 deer are needed per wolf to maintain equilibrium of predator-prey system in southeast Alaska (Van Ballenberhe and Hanley 1984).

Mech (1977) reported that in a declining deer herd, surviving deer inhabited overlapping edges of wolf-pack territories. There, wolves tended not to hunt in order to avoid fatal encounters with their neighbors. Klein (1981) has also suggested that in southeast Alaska where single wolf packs often occupy entire islands, the potential for wolves to reduce deer numbers is increased because there are fewer inter-territory buffer zones. Assuming the deer:wolf ratio needed for equilibrium is 156:1, the minimum deer density needed to sustain wolves in GMU 2 (1 wolf/32 mi^2) at equilibrium is approximately 5 deer per mi^2.

Genetic Considerations

Individual subpopulations of wolves in southeast Alaska are relatively isolated by the island nature of the Alexander Archipelago. Although wolves readily swim small distances, their absence from Admiralty, Baranof, and Chichagof Islands, as well as some of the outer islands, shows their inability (or disinclination) to swim long (e.g., > 2 mi) distances. As logging and road-building proceed, wolves are expected to decline. Over several generations, inbreeding in isolated, increasingly small subpopulations may lead to declines in genetic hetrozygosity and fecundity (Soule 1980). The degree to which wolves suffer from inbreeding depression is subject to debate (Theberge 1983, Shields 1983, Laikre 1991). If wolves are susceptible to inbreeding depression, the problem will be most pronounced on remote islands where genetic exchange with other wolves is limited. Inbreeding depression is suspected of contributing to the rapid population decline of wolves on Isle Royale (Peterson 1989, Wayne et al. 1991).

RECOMMENDATIONS

Although wolves are not in immediate danger of being eliminated from southeast Alaska, significant declines are expected in several biogeographic provinces over the long term. Steps should be taken to ensure that populations remain well distributed throughout their current range. The following recommendations reflect what I consider to be the minimum standards necessary to meet this objective.

1.) Where roads are joined to communities (e.g., ferry and road access to > 1,000 people), road density within individual wildlife analysis areas (WAAs) should not exceed 1.0 mi/mi^2. In WAAs which adjoin wilderness or roadless areas > 40,000 ac, road densities should not exceed 1.25 mi/mi^2. Roads which are made inaccessible to human traffic through gating or barricading after timber harvest are not considered in this density calculation. Because the coastline provides similar waterborne access to these same wolves, the miles of skiff-accessible beach should be added to road miles when calculating "road density."

2.) Habitat capability necessary to provide for equilibrium populations of predators and prey should be maintained wherever possible. Equilibrium prey populations shall be determined on a site-specific basis, based on expected deer predation rates and deer mortality from hunting (Keith 1983). As a general rule, where deer are the primary prey item for wolves (i.e., on most islands and the southern half of Cleveland Peninsula), sufficient habitat capability should be maintained to support at least 5 deer/mi^2.

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A PLAN FOR MAINTAINING VIABLE AND WELL-DISTRIBUTED BROWN BEAR POPULATIONS IN SOUTHEAST ALASKA

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SUMMARY

The history of the extirpation of brown bears (Ursus arctos) from many regions of North America demonstrates that brown bears have a high potential for population viability problems. Resource managers in southeast Alaska have the opportunity to learn from historic and current pressures on brown bears so that viable and well-distributed populations are conserved for the future. The brown bear is a management indicator species (MIS) for National Forest lands in Alaska. The Tongass Land Management Plan (TLMP) Revision Draft Environmental Impact Statement (DEIS) identified eight geographic units each of which needs to maintain viable and well-distributed brown bear populations. These minimum viable population size estimates vary from 125 to 250 individuals (TLMPR-DEIS, p. 3-553; and USFS Technical AMS, R10-MB-89, pp. 568-773). The 'well-distributed' portion of the minimum viable population analysis requires careful attention because resource extraction activities could result in the extirpation of the brown bear from portions of their range. This problem exists because brown bears require large tracts of undisturbed landscapes. A

series of Monte Carlo simulations demonstrated that a population of 250 bears would be extirpated given continued high human-induced mortality rates that have and are occuring on northeast Chichagof Island. Old growth standards and guidelines and general forest-wide standards and guidelines should provide for intact large habitat conservation areas (watersheds) so that brown bear habitat remains well-distributed over the Forest. The units to maintain viable and well distributed brown bear populations in the present TLMP DEIS are too large. We believe that the combination of Large Habitat Conservation Areas (HCAs) along with appropriate planning on all value comparison units is required to assure well-distributed and viable populations of brown bears. These attributes include Large HCAs (40,000 ac) that are unroaded and spaced at least every 20 mi. These Large HCA's must include high-volume riparian old-growth forests and at least one salmon spawning stream. A brown bear management program is required in all value comparison units that include attributes such as bear access to salmon streams that have forest buffers, few roads, and a program that limits human access. Resource management standards and guidelines to maintain well-distributed and viable populations must include:

1) planning guidelines that include site-specific habitat capability modeling, the clustering of development activities, and cumulative impacts assessments for site-specific plans,

2) the establishment of large, undisturbed habitat conservation areas with limited access in association with intensively managed areas,

3) the maintenance of 100 meter forest buffers along important bear-fishing streams,

4) the continued implementation of solid waste management programs and firearms policies in industrial camps,

5) a progressive, apriori, road closure program, and

6) a program of limited access to cutting units and roads except for ongoing timber extraction activities.

INTRODUCTION

Over the last century, both the distribution of numbers of brown bears have declined across North America. Owing to this decline, various Canadian and American agencies have or are currently developing plans for the conservation, management or recovery of certain brown bear populations. Stable brown bear populations currently occupy southeast Alaska. Yet, there are increases in the rate of change to the old-growth forested landscape that are changing brown bear habitat. In particular, the human access to what were formerly pristine areas translates to more bear-human encounters. Consequently, there is a need to develop a broad-based conservation plan that may allow for viable brown bear populations into the future. This chapter reviews brown bear ecology in southeast Alaska and recommends conservation measures for the long-term maintenance of of these populations across the landscape. We develop a process for applying the biological requirements of brown bears to the design of a conservation plan (e.g., Murphy and Noon 1992). This process was initiated to develop and recommend management standards that would insure viable and well-distributed populations on the Tongass National Forest that are consistent with the National Forest Management Act and within the concept of a multiple-use planning process.

CURRENT STATUS

The brown bear in southeast Alaska is a Management Indicator Species (MIS), for the TLMP Revision. Sidle and Suring (1986) discuss the brown bear bear as a MIS for the National Forest lands in Alaska using the selection criteria that the brown bear is an "emphasis species" since it is hunted, and that it is a "special interest species."

POPULATION DISTRIBUTION, STATUS AND ISOLATION

Taxonomy

Ursus arctos has a holarctic distribution and wide local variation occurs in size, skull morphology, and pelage color (Craighead and Mitchell 1982). This local variability previously led to a wide array of specific and subspecific descriptions especially among Alaska's coastal islands (Merriam 1918, Hall and Kelson 1959). Two North American subspecies are presently recognized (Rausch 1963) with large variation across their range. *U. a. horribilis* includes all brown/grizzly bears of continental North America, including the islands of southeast Alaska, and U. a. middendorffi includes the brown bears of Kodiak, Afognak, and Shuyak islands.

Distribution

Within southeast Alaska, the brown bear occurs on Admiralty, Baranof, and Chichagof islands and on the coastal mainland.

Status

North America - Peek et al. (1987) estimated the North American brown/grizzly bear population between 52,000 and 63,000, with about 65% occurring in Alaska. In the continental U.S., the decline in number of brown/grizzly bears was so substantial that they were classified as threatened in 1975 under the Endangered Species Act. Presently, fewer than 1,000 are estimated in Montana, Wyoming, Idaho, Washington and Colorado (Servheen 1990). These fragmented populations receive a high interagency management priority for maintaining and increasing populations (Strickland 1990).

Over their holarctic range, there is a history of long-term populaton declines and fragmentation of brown bear populations (Servheen 1990). This is especially true in Europe (e.g., Camarra 1983, S*rensen et al. 1990) and the western U.S. Alaska and portions of Canada have the remaining widespread and stable brown/grizzly bear populations.

Southeast Alaska - The density of southeast Alaska's brown bear populations are among the highest measured. For example, Schoen et al. (in review) determined that Admiralty Island has as many as 1,700 brown bears over the 4,403km^2 island. Brown bears are also abundant on Baranof and Chichagof islands, while somewhat lower densities occur on the mainland of southeast Alaska.

Isolation

Populations of brown bears on the northern islands of southeast Alaska are effectively isolated from the mainland. Radio-collared brown bears from Admiralty and Chichagof islands have not moved to other islands (L. Beier, J. Schoen, K. Titus, unpubl. data). Inter-island movements probably occur between Baranof and Chichagof islands. This indicates that each island should be considered a metapopulation and managed for viability separately. For islands like Admiralty, the isolation is probably sufficient in that one or more bears from another locale do not enter the population per generation.

PATTERNS OF HABITAT USE

Southeast Alaska brown bears use a variety of habitat types on an annual basis. Denning habitat includes both low elevation old growth forest and cave dens in alpine or subalpine habitat (Schoen et al. 1987). After emergence from the den in April and May, many brown bears travel to, and use lowland, old-growth forests and coastal sedge meadows (Schoen and Beier 1990) where their diet is dominated by sedges (Carex sp.), skunk cabbage (Lysichitum americanum), and other green vegetation and roots (McCarthy 1989). From mid-June through mid-July most bears use forested slopes and subalpine and alpine meadows, where freshly emergent vegetation is available and where adult bears mate. Brown bears concentrate along salmon streams and associated riparian forest habitat from mid-July until spawning ceases in a particular stream. Habitat mosaics that contain riparian old-growth forest interspersed with devil's club (Oplopanax horridum) and current (Ribes sp.) patches, positioned within easy travel to and from salmon streams are selectively used in greater proportion than their occurrence. Habitat use along salmon streams is determined by features such as species of salmon present, fish catchability, number of fish, forest cover, and distance to other salmon streams. Brown bears depart salmon streams and riparian forest habitats in late summer or early autumn and use avalanche slopes, high elevation forests, and subalpine meadows prior to denning (Schoen and Beier 1988, 1990).

HABITAT CAPABILITY MODEL

Schoen et al. (In press) developed a brown bear habitat capability model for southeast Alaska as an aid to the forest planning process where the consequences of site specific changes in the habitat could be predicted with varying levels of certainty. This empirically derived model is based on quantitative data from long-term radio-telemetry studies on Admiralty and Chichagof islands. Professional judgement and informed consensus were used for evaluating the capability of some man-induced habitat types (e.g., clearcuts) and the reduction of capability of certain habitats (e.g., reduced habitat value near communities).

Riparian old-growth forest was found to have the highest habitat capability for supporting brown bear populations. This habitat type had high use and low availability. Schoen et al. (In press) also reduced the capability of various habitats to support brown bear populations based on the effects of human activity and development. There is ample evidence that human activity and development attributes are among the most important determinants relating to the capability of habitats to support brown bears (e.g., Archibald et al. 1987, Mattson et al. 1987., McClellan and Shackleton 1988, 1989; McClellan 1990, Schoen 1990). The brown bear habitat capability model reflects this lowered habitat capability near zones of human activity or disturbance. As in other areas, southeast Alaskan brown bear populations have undergone high mortality rates when roads and logging camps are developed in pristine habitats (Titus and Beier 1992).

SCALE OF RESOLUTION

Brown bears respond to habitat changes at the landscape level (Schoen 1990), hence the scale for assessing their viability is necessarily large. We suggest that large habitat conservation areas (watersheds) be considered for an initial assessment of viability to 100 years. One convenient way of thinking of the appropriate scale for analyzing viable and well-distributed brown bear populations might be to use the old-growth provinces and sub-provinces described by Samson et al. (1989).

Rationale

Bear biology - Brown bears in southeast Alaska have large, overlapping home ranges averaging 100km^2 (~25,000 ac) for male brown bears on Admiralty Island

(Schoen and Beier 1990). Female home ranges were much smaller, averaging $37km^2$ (~9,000 ac) on Admiralty Island and $25km^2$ (6,000 ac) on Chichagof Island. Although these home ranges are large, they are smaller than those measured for brown bears in other regions (e.g., Miller 1987, Blanchard and Knight 1991). Most life requisites such as adequate old-growth forest patches, salmon streams, berry patches, alpine and denning habitat will be contained within the annual home range. Long-term brown bear viability is dependent on undisturbed 'reservoirs' of adequate size (e.g., Knight and Eberhardt 1985, Horejsi 1989, McLellan and Shackleton 1989, McLellan 1990). Specific rules for determining the size of areas for maintaining well-distributed and viable brown bear populations do not exist, which is not unlike other species and regions (Grumbine 1990). We suggest the watershed as a conveniently-sized scale. The size of that area is 40,000 ac., approximating the mean size of a value comparison unit for Game management Unit 4. An area this size would contain at least 5 female home ranges.

POPULATION DEMOGRAPHY

Brown bears have the lowest reproductive rates among all terrestrial land mammals in North America. Although highly polygamous (Craighead and Mitchell 1982), females do not begin breeding until about age 5, and breeding intervals may average three or more years (e.g., Knight and Eberhardt 1985, Reynolds et al. 1987, Eberhardt 1990, Schoen and Beier 1990). Specifically in southeast Alaska, Schoen and Beier (1990) found that no females < age 7 produced a litter and the mean age for a female with her first litter was 8.1 years. On Admiralty and Chichagof islands Schoen and Beier (1990) found the mean interval between successful litters (was 3.9 years, an interval somewhat higher than assumed elsewhere (Eberhardt 1990). One demographic characteristic important to a viability analysis recorded by Schoen and Beier (1990) was that several adult females failed to produce young for five to six year periods. In terms of population modeling, examination of the data provided by Schoen and Beier indicates that of their marked Admiralty Island females, 82% did not produce cubs in any given year. From the published reproductive data, it appears that Southeast Alaska brown bears begin breeding later and have longer breeding intervals than that found for some other brown bear populations.

Determining annual survival and mortality rates of brown bears in forested habitats is difficult because of censoring and the costs associated with determining the fate of all individuals. Nevertheless, Schoen and Beier (1990) found that a minimum of 28% of the 95 brown bears captured on their Admiralty and Chichagof islands study areas died during their studies. Eighty-two percent of their non-capture related mortalities were the result of some human-factor. This indicates that southeast Alaska brown bear demographics are strongly influenced by humans, even in roadless areas such as Admiralty Island. Age-specific and sex-specific differences in survival have not been examined for southeast Alaskan brown bears.

The age-structure of southeast Alaska brown bears can be evaluated by assuming that the bears capture by Titus and Beier (1992) were an unbiased sample of bears $\geq$ age 4. Of the 21 males captured, 10 (48%) were age 6, 4 (19%) were between 7 - 10, and 7 (33%) were > age 10 (Table 3). Of the 30 females captured, 12 (40%) were age 6, 8 (27%) were between 7 - 10, and 10 (33%) were > age 10 (Table 3). This age structure is similar to that reported for the northcentral Alaska range (Reynolds 1990). From a conservation standpoint, it is important to note that brown bears are long-lived and that a significant portion of the adult population is composed of bears >10 years old. These demographic data stress the importance of careful management insomuch as the consequences of an error will be high (Miller 1990a). This is because few cubs are produced in any given year, at least 8 years are required for females to become important contributors to the next generation, and loss of too many adults will slow the ability of the population to provide for adequate recruitment.

VIABILITY RISK ASSESSMENT

Brown bears are a species for which viability can be easily jeopardized given:

1) historical evidence that this process has occurred over most of this species' range (e.g., Servheen 1990),

2) the species has a low reproductive rate,

3) population declines are invariably the result of man-induced mortality, and

4) viable populations occur most frequently in large tracts of undisturbed or seldom disturbed landscapes.

Varying levels of logging, mining, and roadbuilding in southeast Alaska raise concern for brown bear population viability. The probability of extirpation is positively correlated with the amount of human development and the magnitude of the resource extraction activities across the landscape. Two examples follow, one of which was developed with a viability risk assessment. We performed a viability risk assessment on the northeast portion of Chichagof Island where there are concerns over long-term brown bear population viability.

Admiralty Island

Much of Admiralty Island is set aside as a National Monument wilderness and one can reasonably assume that the probability of extirpation will remain low over this area. If one were to rank the necessity of performing a detailed viability analysis for Admiralty Island, this area would likely rank lower than other areas where have intensive forest management activities are scheduled.

Northeast portion of Chichagof Island

This area has undergone substantial roadbuilding and timber harvest in the recent past and this pattern continues. Resource managers have high interest in predicting brown bear population viability in such areas. Intensive forest management activities will result in an increase in brown bear mortality. There was a strong association ($r = 0.79$, $P < 0.01$, $n = 11$ years) between annual numbers of brown bear deaths and one attribute of timber harvest, namely roadbuilding (Figure 1). These factors led us to perform a detailed viability risk assessment (Appendix A1).

Given the high association between roads and human-induced brown bear mortality we advocate forest management standards and guidelines that mitigate population reductions associated with forest management. These reductions in habitat capability caused by roads are incorporated into the model of Schoen et al. (In press).

Simulation results indicated that the probabilities of local extinction for brown bears are worthy of detailed consideration given past man-induced mortality rates (Appendix A1). Given the low reproductive rates by some individuals, we found that simulated southeast Alaska bear populations declined faster when site-specific data from southeast Alaska were used (Figure 2). This was in contrast to other simulated populations that used data from other regions for comparison.

There are a variety of ways to address questions of population viability (e.g., Shaffer and Samson 1985). For example, if the minimum viable population size is set at 250, brown bear populations will not remain viable with modest adult mortality rates. This assumes that males and females are removed from the population at equal rates. Another way of examining the viability question is to examine the number of simulations in which the population went extinct after 100 years (Figure 3). Results were not the straight inverse of population size as extinction rates for configuration 3 rise more sharply at lower mortality rates than for the other configurations.

Understanding these simulated population sizes and extinction rates might be best illustrated by relating them to historic mortality rates. The human-induced brown bear mortality rates that occurred on northeast Chichagof Island aid in relating these simulations to the on-the-ground situation and the need for conservation planning. From 1984-88 a mean of 15.8 brown bears/year were known to be harvested on northeast Chichagof Island (ADF&G brown bear sealing certificate data). This total known kill includes both sport havest and defense of life or property deaths. We might conveniently assume that the study area contains 250 bears. Although this known harvest rate seems to only be 6.3% of the population, it must be placed in an appropriate context indicating a high potential for a long-term population decline under such a harvest. The unknown kill was not considered (Schoen 1990) nor was the natural mortality. If we consider the unknown kill to be 2% (K. Titus and L. Beier, field notes), and the natural adult mortality to be a conservative 1%, then we are within the bounds of a declining population given the inexact nature of modeling, and estimation of mortality and reproduction.

Second, the proportion of females harvested during this period was higher than normally found elsewhere in southeast Alaska. Fifty-six percent of the autumn brown bear from 1980-1987 on northeast Chichagof Island were females. Third, the location of most of this this harvest was closely linked to the Hoonah road system indicating that brown bears in some watersheds were highly harvested while other remaining pristine watersheds had little or no harvest.

Modelers of brown and polar (Ursus maritimus) bear population dynamics found that about 1.6-2.0% of the adult females can be harvested to support a sustainable population (e.g., Taylor et al. 1987). The known harvest of adult female brown bears on northeast Chichagof Island probably exceeded this level during the mid-1980's. These modeling studies combined with the simulations presented here, and history of brown bear population declines indicate that brown bear conservation planning needs to be an integral part of forest planning and in particular, timber and mineral extraction activities. Resource planners and decision-makers should use this information about southeast Alaska's brown bears in terms of a risk analysis to aid their decision making for maintaining minimum and well-distributed populations (Samson et al. 1985). Actually, managers should strive to manage bears on a sustained yield basis (Miller 1990a) and maintain their distribution across the landscape.

WELL-DISTRIBUTED POPULATIONS

A viability analysis can only determine the number of individuals required to sustain a population for some period of time under a given set of circumstances. The National Forest Management Act also requires that populations be well-distributed across the planning area. For species such as the brown bear in southeast Alaska, problems with maintaining well-distributed populations over time are likely to be more difficult to meet than just maintain brown bears per se. For this reason, bear management efforts should emphasize those areas where roads and people will gain easy access to areas that were formerly difficult to access. For brown bear populations to remain well-distributed, at least some of their required habitats have to be maintained on every value comparison unit.

VIABLE POPULATION CONSERVATION STRATEGY

Progressive planning and implementation efforts are necessary to maintain well-distributed brown bear populations in southeast Alaska. The conservation strategy for southeast Alaska does not address the maintenance of brown bear populations that are at or near the original carrying capacity of the habitat. To assure stable, productive, and huntable brown bear populations will require more resource management measures than discussed below. An overall planning goal should maintain viable populations distributed throughout the planning area as well as maintain much larger populations where bear management will emphasize hunting or viewing. The purpose of this plan however, is to maintain viable populations if brown bear numbers are reduced over time. This strategy contains the minimum measures required to insure that brown bear populations will persist, although they would be reduced in number and they could not be hunted and quality viewing would be unlikely.

Habitat Conservation Areas

We use the term Large Habitat Conservation Area (HCA) as our planning unit. These areas are also known as a value comparison unit (VCU) in Forest Service planning and as a minor harvest unit by the Alaska Department of Fish and Game. Each Large HCA should be at least 40,000 ac. and distributed nor farther than 20 miles to another Large HCA. A Large HCA is one important component for maintaining viable brown bear populations. The northeast portion of Chichagof Island provides one convenient example of a land area that should be used to assure viable and well-distributed populations. At least one Large HCA needs to be established on this area (Figure 4). On northeast Chichagof Island, most watersheds have already undergone extensive roading and accompanying timber harvest. Attributes of management intensity important to brown bears have varied greatly in each of these watersheds, yet brown bears still occur throughout the subprovince. Only 2 watersheds still have areas where bear-human conflicts are currently low. These watersheds include the Game Creek and Seagull Creek watersheds. These watersheds will be roaded and timber harvest will occur in 1992, forclosing the opportunity of a Large HCA on the northeast portion of Chichagof Island. Examination of Figure 4 also indicates that 7 adjacent watersheds have a high potential for bear-human conflict. Within these 7 watersheds, much low elevation riparian forest habitat has been removed, and bear use of these areas is high and associated with important salmon streams. These watersheds include Gartina Creek, Spasski Creek, Suntaheen/Whitestone watershed, Iyouktug Creek, Seal Creek, Freshwater Creek complex, and Kennel Creek. No measures were planned or instituted in any of these VCU's (watersheds) to mitigate bear-human conflicts and assure adequate bear habitat into the future.

Using the northeast Chichagof example, one can understand the need for conservation measures (Schoen 1990) and cumulative planning (e.g., Westman 1985). Brown bear management needs to be considered in every watershed so that functional habitat is distributed across the landscape and viability is assured. Telemetry data (Schoen and Beier 1988, 1990, K. Titus and L. Beier unpubl. data from northeast Chichagof island) indicate that brown bears regularly travel among watersheds, particularly when moving to salmon streams, and when travelling to and from denning habitat. This plan will not assure viable populations in every watershed, but a combination of measures are needed to assure overall viability. A plan should have been developed for avoiding the persistent and high potential for bear-human conflicts that occurred and will continue to occur on the 7 watersheds previously mentioned. In addition to the maintenance of Large HCA's, standards and guidelines will reduce or minimize sources of brown bear mortality that are known to create viability problems.

BROWN BEAR VIABILITY STANDARDS AND GUIDELINES

Developing and implementing reactive programs for increasing brown bear populations after they approach or fall below viability is extremely costly (e.g., Interagency Grizzly Bear Guidelines 1986, U.S. Fish and Wildlife Service 1990). Given the high brown bear densities that presently occur in many areas of southeast Alaska, resource manager need to be conservative in developing programs that will maintain viable populations. Resource managers also should recognize that managing for small but viable populations is risky (e.g., Shaffer and Samson 1985, Grumbine 1990). If populations were allowed to decline to low levels, many user groups (e.g., hunting guides, wilderness guides, tourists, resident hunters) and other industries (e.g., timber subcontractors) would suffer. The standards and guidelines discussed below are to ensure viable and well-distributed brown bear populations are maintained in Southeast Alaska.

Some of these standards and guidelines may best be instituted at the Forest Plan level, while the implementation of others might best be accomplished with more site specific plans.

General Procedures

Identify, rank, and map areas with a high potential for bear-human conflict. Justification - Resource managers need to be able to understand the relative merits of selecting a Large HCA for bears among competing interests.

Model potential changes in the capability of the habitat to support brown bears given various levels of resource extraction on each watershed. Justification - Understanding the long-term viability of such a long-lived, k-selected species requires that the public and resource management professionals have reasonably accurate predictions.

Identify, rank, and map areas with high potential for providing high-quality brown bear refugia. These will be drawn on for use as potential Large HCA.

Establish one Large Habitat Conservation Area (Large HCA) of at least 40,000 ac of functional brown bear habitat at least every 20 miles across brown bear range in southeast Alaska. The Large HCA should be unroaded, have some old-growth forest habitat, and contain at least 1 km of salmon spawning habitat that is accessible to brown bears.

Perform spatial and temporal cumulative impacts analyses during forest planning to assess potential bear-human conflicts and mortality risk. Justification - A wealth of scientific evidence exists indicating that brown bear populations decline significantly from the cumulative effects of widespread resource development. An understanding of brown bear viability will benefit from, and the National Environmental Policy Act requires, a cumulative effects assessment to understand and predict the consequences of projects. Since brown bears have large home ranges, populations are likely to be effected by >1 site-specific forest plan. Use of appropriate modeling (e.g., Schoen et al. In press) is one analysis method.

Monitor the application of standards and guidelines to assure that they are properly and effectively used. Justification - A built in check to evaluate how well the program is working.

Specific Standards and Guidelines

Some standards and guidelines were adopted from Schoen and Beier's (1990) discussion of preliminary management guidelines for intensive land development in brown bear range, and the grizzly bear management guidelines contained in the Interagency Grizzly Bear Guidelines (1986).

Food and solid waste should be handled and disposed of using appropriate and approved methods (e.g., State of Alaska Department of Environmental Conservation, U.S. Environmental Protection Agency) to minimize attracting bears (Schoen and Beier 1990). Fuel-fired incinerators should be a requirement at all logging camps. Justification - Bear-human conflicts will be minimized or even eliminated by careful waste management.

Siting of new seasonal and permanent camps, mineral exploration and operational facilities, log dumps, and transfer facilities should never be located <1.6km from sites of seasonal brown bear concentrations (Schoen and Beier 1990). Justification -Bear-human conflicts can be minimized by keeping people away from bears.

Operating plans for mineral exploration and development, concessionaire special use permits, and timber extraction should include plans for protecting brown bear habitat and reducing bear-human conflicts. Exploration and development should avoid times and seasons when bear-human encounters are likely. Justification - Bear-human conflicts can be minimized by planning activities that enter brown bear range.

Industrial and recreational development should be concentrated rather than dispersed across the landscape. Justification - Concentrated development will minimize bear-human interactions and maintain a few key bear refugia.

A portion of timber sale receipts collected for post sale area improvement (Knudsen-Vanderberg, K-V Act Funds) should be used to 1) enhance brown bear habitat by closing roads and reducing human accessibility, and 2) monitoring brown bear use of cut and nearby uncut areas. This type of K-V monitoring might be accomplished with K-V funds when the original post-sale improvements were supported by K-V funds. Justification - There are costs associated with closing roads and enhancing brown bear habitat. K-V funds are one source to meet this planning requirement and allowing the habitat capability of the habitat conservation area to be increased. K-V funds are one source to understand the effectiveness of any post timber extraction habitat enchancment.

Close roads to selected cutting units except for the timber extraction activities. No recreational opportunites by using cars/trucks or off-road vehicles should be allowed in high bear density areas. Some seasonal exceptions might be allowed. Timber and road contractors should have the same restrictions to road access for recreational use as the general public. Justification - Easy public access to major tracts of pristine habitat causes increases in man-induced brown bear mortality. Cumulatively, this increased mortality has led to declines in brown bears in many regions.

Employee policies regarding the carrying of firearms and on the job hunting as adopted by mining compancies in Southeast should be considered more widely.

A minimum of 100 m buffers of uncut timber should be retained along important salmon-bear-use streams in areas that will undergo intensive timber harvest. Justification - Riparian old-growth forest along salmon streams are among the most highly selected habitats for southeast Alaska brown bears in late summer. Of the late summer radio-collared brown bear habitat use within 1.6km of an anadromous salmon stream, 73.5% and 74.7% occurred within 200m of the stream on study areas on Admiralty and Chichagof islands respectively (n = 221 telemetry locations $\leq$ 1.6km from salmon stream in late summer; data reanalyzed from Schoen and Beier 1987). Brown bear use of salmon streams where buffers were not in place was almost non-existent (Schoen and Beier 1987, 1990, L. Beier and K. Titus unpubl. telemetry data from Spasski Creek, Chichagof Island).

Roads should not be built parallel and close (<200m) to important salmon-bear streams. Bear-human encounters will be higher if roads are built near streams.

In cases of brown bear-human conflict, District Rangers and state personnel should be immediately notified and jointly identify the cause of the conflict. The problem should be corrected or mitigated.

CONSERVATION PLANS IN OTHER REGIONS

This viability plan for southeast Alaska brown bears is less rigorous than adopted for the threatened grizzly bear in the lower 48 states. Few specific guidelines are adopted for dealing with problem bears in southeast Alaska. In contrast, costly measures are invoked in areas such as the Yellowstone ecosystem and Denali National Park to save individual problem bears. The Interagency Grizzly Bear Guidelines have many specific measures for maintaining and improving habitat and restricting recreational opportunities in grizzly bear core areas. The conservation plan for southeast Alaska does not impose such specific restrictions but promotes guidelines for maintaining populations as a whole.

MONITORING RECOMMENDATIONS

Despite the wealth of informaton on bear biology and management, few attempts have been made to develop population monitoring programs (but see Harris 1986). This problem is acute in southeast Alaska were logistics and cover make attempts at indexing bear numbers difficult. One indirect method of monitoring bear numbers is to sample attributes of the hunter-kill over time (e.g., Miller 1990b). Other monitoring methods are aimed at surveying the population directly. Both the Forest Service and the Alaska Department of Fish and Game have requirements to monitor wildlife populations. The difficultly in developing a sound monitoring program for bears and many other forest wildlife species should not be underestimated (Verner 1985).

Specifically in southeast Alaska, brown bear population time-trends can be monitored by aerial alpine surveys, provided that 1) this habitat type occurs in the area of interest (Schoen and Beier 1990), 2) site specific data are not required, and 3) experienced personnel are available to design and/or conduct the surveys. Unless mark-recapture population estimates are combined with alpine surveys, there will be no opportunity to correct the index to some population estimate. In areas of high brown bear density and interest (e.g., Greens Creek Mine), repeated mark-recapture surveys are one of the few methods for obtaining point estimates over time. Schoen and Beier (1990) recommend that density estimates be obtained before the impact and then 5 and 10 years later. Schoen and Beier (1990) evaluated an infrared scanner for enumerating brown bears using riparian habitat along anadramous fish streams during late summer. This monitoring method showed promise, but the costs will be high for further technique development.

Other methods for monitoring bear numbers and/or intensity of use should be explored. Requirements for a useful survey and monitoring method include that it be sensitive to population or habitat use changes (e.g., being able to detect a 20% change with a 95% accuracy), the method is repeatable and labor and logistics are not prohibitive. Data should be collected so that they are amenable to modern trend analyses (e.g., Sauer and Droege 1990) and so that the statistical power is understood and translated to policy- and decision makers (e.g., Peterman 1990).

Methods that should be explored further include the use of an infrared scanner, photographic methods, and counting bear tracks, trails, and day-beds. Schoen and Beier (1990) counted day beds and found them to be a useful index of bear activity. These type of monitoring programs should be instituted in association with resource extraction activities so that before and after data are acquired.

RESEARCH RECOMMENDATIONS

Much of the applied research should center on the development of reasonable, extensive survey techniques that do not require expensive telemetry studies. At the onset these monitoring techniques will require research to evalute the precision and accuracy of the methods.

Other research needs include:

1) an understanding the the ecology of brown bears at dumps and how they interact with the 'wild' population,

2) the long-term use or non-use of clearcuts as the habitat changes,

3) the utility of road closure programs to enhance bear habitat by increasing its capability through a reduction in bear-human conflicts,

4) brown bear use of salmon streams under varying levels of salmon escapement.

5) determining population thresholds to habitat disturbance.

6) continuing to validate and revise the brown bear habitat capability model.

7) continued study of brown bear population viability including the genetic implications of population reduction.

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Appendix A1. Viability Risk Assessment Example - Northeast Chichagof Island

A high brown bear harvest rate occurred from 1980-87 on a ~1,000km^2 area of northeast Chichagof Island between Tenakee Springs and Hoonah. Northeast Chichagof Island brown bear populations will likely decrease and may not remain viable into the future. These risks exist because of the development of a road network and activities associated with timber extraction that promote bear-human conflicts. To examine the viability risk of this brown bear population, we modeled population size over time using POPDYN4 by J.W. Grier of North Dakota State University (Grier 1980 a, b, Grier and Barclay 1988).

This model was used for exploratory purposes. This model may not be the best available although it is among the easier to implement in terms of data requirements (cf. Taylor et al. 1987). I used a standard exponential growth model with demographic stochastic events. Survival of each individual bear was determined by a stochastic routine but environmental and genetic stochastic events were not part of the model. Reproduction was calculated for each female stochastically using probabilities that we provided. All individuals in the simulation were kept track of separately over time as individuals died and others entered the population (Grier 1989). The stochastic nature of this program operated by determining the mortality for each individual by comparing a randomly generated number with the input probabilities provided at the onset of a given simulation. Each individual remained or was discarded from the cohort depending on whether the random number was > or < the input probability. Other approaches exist (e.g., Craighead et al. 1974, Taylor et al. 1987, Eberhardt 1990).

A total of 250 brown bears was assumed as the starting population size. This population size was chosen for several reasons. First, the TLMP Revision DEIS uses this value for each of the three ABC islands. Second, for the example from northeast Chichagof Island, 250 brown bears may be a reasonable representation of the actual number of individuals present (K. Titus and L. Beier, field notes). Finally, the simulations become unwieldy with sample sizes much larger than this.

The objective of these simulations was to evaluate population size after 100 years given varying levels of adult mortality. For long-lived species, patterns of adult female survival are among the most important demographic considerations for maintaining viable populations. Modeling results provide resource specialists with insights about population size over time given forest management practices that alter bear survival in predictible ways. Specifically for brown bears, reducing adult mortality rate is a management option.

Assumptions for all simulations - constant functions -

- closed population
- no density dependent effects
- reproduction and survival are stochastically modeled
- maximum number of young per female -- 3
- assume 125 females and 125 males in the population at the onset of each simulation
- 100 Monte Carlo interations per design factor

- each simulation estimates a population for 100 years

Parameterization -

- constant functions were first input
- three configurations of adult female natality (births, b_i) were used
    - configuration 1
        - age of first reproduction - 5
        - estimated first year mortality rate - 28% (Knight and Eberhardt 1985)

        66% produced 0 young in a year

        15% produced 1 young in a year

        12% produced 2 young in a year

        7% produced 3 young in a year (percentages empirically derived from a brown bear literature; No estimates were made of among-year variability in cub production.)

    - configuration 2
        - age of first reproduction - 5
        - estimated first year mortality rate - 28%

        75% produced 0 young in a year

        12% produced 1 young in a year

        9% produced 2 young in a year

        4% produced 3 young in a year (percentages empirically derived from brown bear literature)

- configuration 3 - age of first reproduction - 7
  - estimated first year mortality rate - 24%

  76.3% produced 0 young in a year

  7.4% produced 1 young in a year

  15.6% produced 2 young in a year

  0.74% produced 3 young in a year

  Configuration 3 is based on data derived from Schoen and Beier (1990).

- annual mortality rates (m_i) for adult bears varied at 2% increments from 6-16%; This is range of mortality over which meaningful population changes occurred.

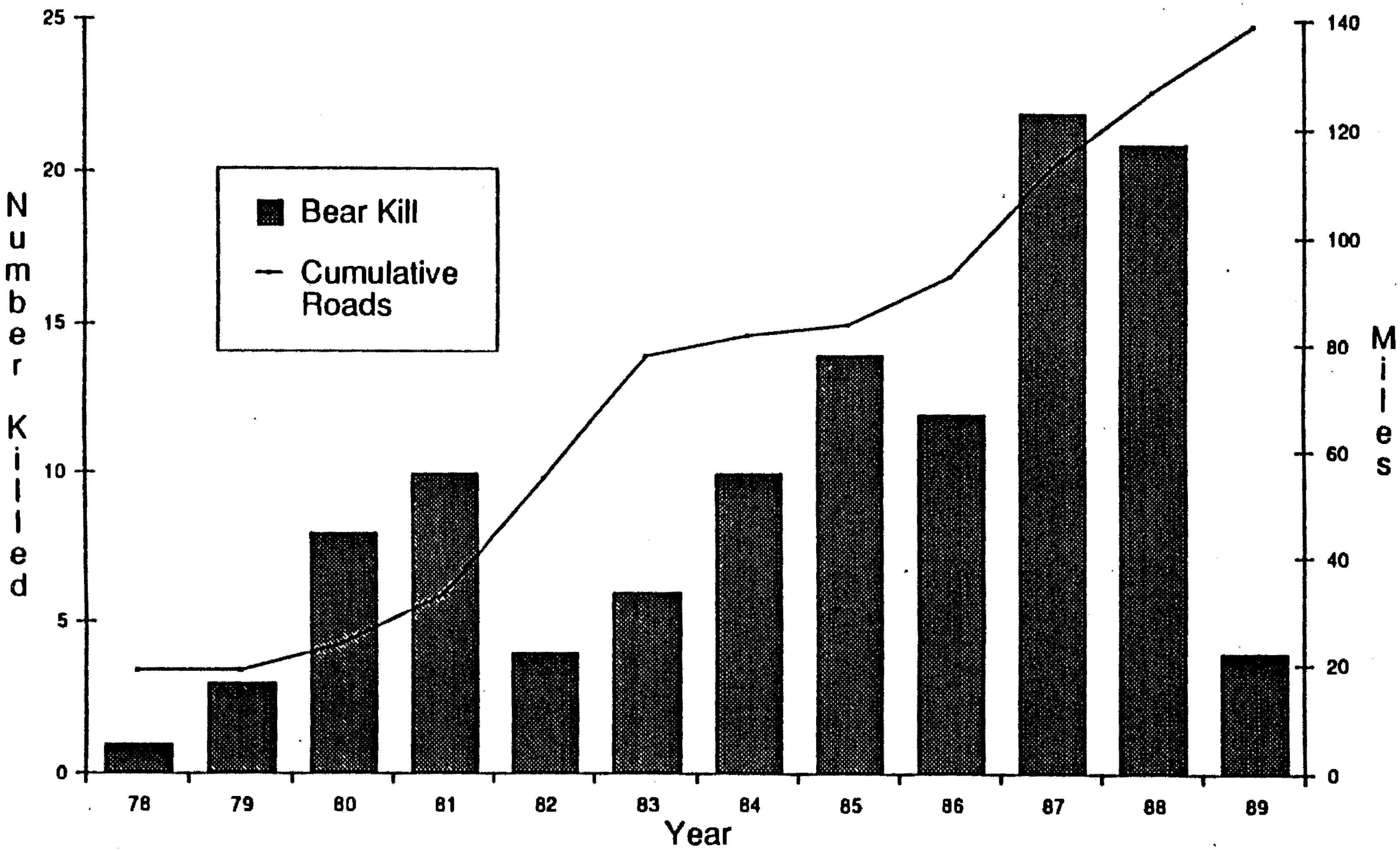
Simulation Results - A total of 2,400 100-year simulations were conducted. Results are expressed as the mean remaining populaton size after 100 years per parameterization and simulation. For visual convenience, population size was expressed on a logarithmic scale (Figures 2 and 3).

Configuration 1 - These simulations assumed an optimistic breeding interval and a relatively large number of 3-cub litters. Assuming that adult mortality can be maintained < 14%, the population increases over time, and there is no probability of extinction (Figure 2). Adult mortality rates $\geq$ 14% resulted in a population decline (= 52; SD = 27) and and extinction probability () of 0.13.

Configuration 2 - These simulations were paramaterized similarly to those of configuration 1 except that a breeding interval of four years was assumed. Like configuration 1, the age of first breeding was assumed to occur at 5. With this lowered reproductive rate, the simulated population declines with an adult mortality rate $\geq$10% (for m_i - 10%, - 209, SD - 65, - 0). Even with a modest 12% adult mortality rate the simulated population was always estimated at less than viability (assuming 250 is viability) and extinction occurred in 6% of the simulations (Figure 2).

Configuration 3 - This simulation used reproductive parameters indicative of a highly k-selected brown bear population and was based on long-term studies from Admiralty Island (Schoen and Beier 1990). These reproductive parameters were lower than used in models of other populations such as the Yellowstone grizzlies (Knight and Eberhardt 1985). Given the best available data from southeast Alaska, one can determine that an initial population of 250 brown bears has a high proability of not remaining viable under moderate mortality rates. For example, a 10% annual adult mortality rate would result in a mean populaton decline over 100 years (Figure 3; - 81, SD - 31, - 0).

LIST OF FIGURES



Bear Kill
Cumulative Roads
Number Killed
Miles
Year
0
5
10
15
20
25
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20
40
60
80
100
120
140
78
79
80
81
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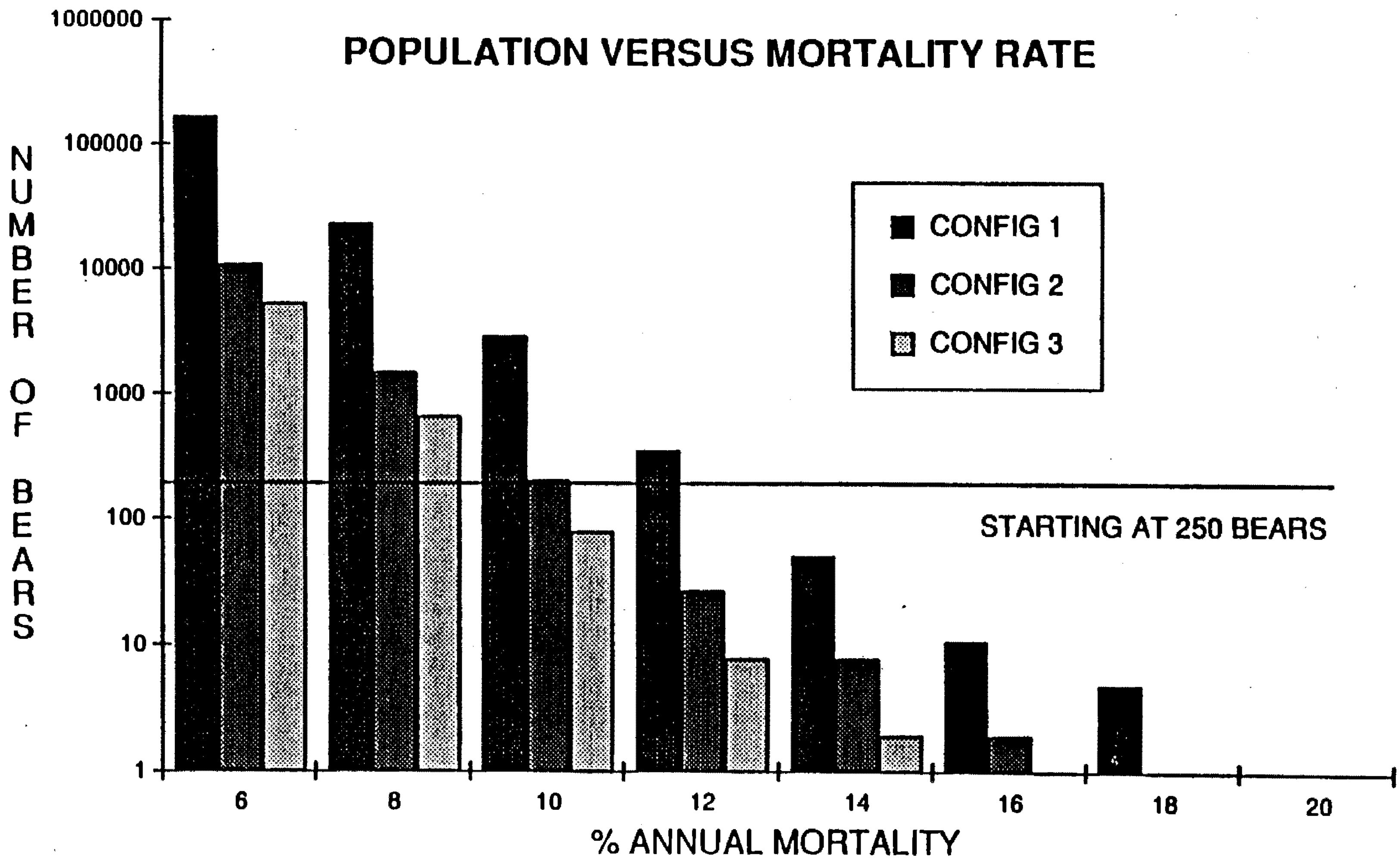
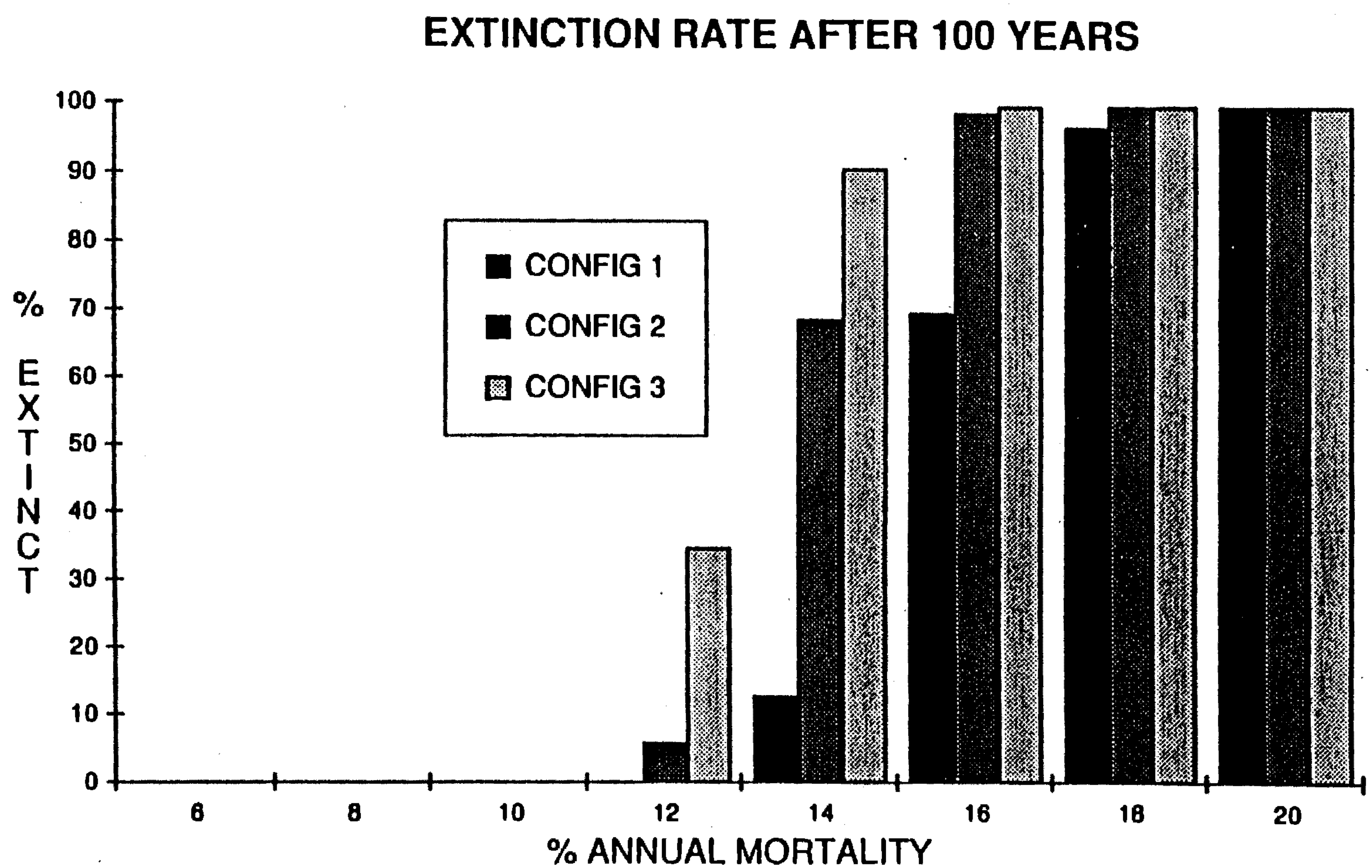
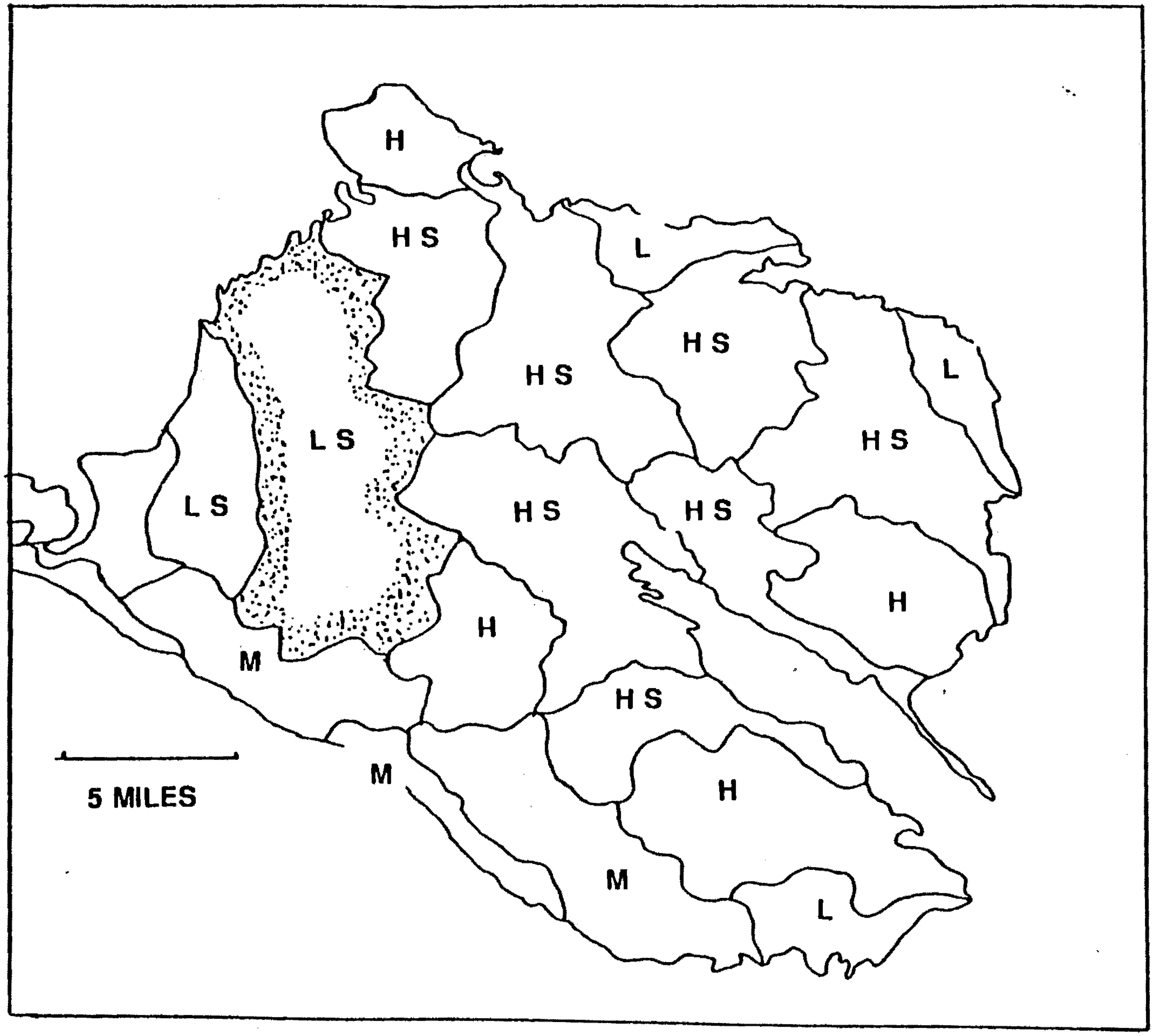


POPULATION VERSUS MORTALITY RATE
NUMBER OF BEARS
1000000
100000
10000
1000
100
10
1
CONFIG 1
CONFIG 2
CONFIG 3
STARTING AT 250 BEARS
6
8
10
12
14
16
18
20
% ANNUAL MORTALITY



EXTINCTION RATE AFTER 100 YEARS
% EXTINCT
100
90
80
70
60
50
40
30
20
10
0
CONFIG 1
CONFIG 2
CONFIG 3
6
8
10
12
14
16
18
20
% ANNUAL MORTALITY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A STRATEGY FOR MAINTAINING WELL-DISTRIBUTED, VIABLE MARTEN POPULATIONS IN SOUTHEAST ALASKA

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INTRODUCTION

This paper reviews the status and biology of martens (Martes americana) in southeast alaska and makes recommendations on habitat conservation measures needed to insure that viable populations are maintained well distributed across the entire area of their current distribution. During October 1990, an Interagency Scientific Committee was assembled by the Team Leader, Tongass Land Management Plan Revision Team, to develop and recommend management standards that would insure well-distributed, viable populations be maintained on the Tongass National Forest (TNF) consistent with the requirements of the National Forest Management act of 1976 (NFMA). This report provides the biological basis and the rationale for the proposed conservation strategy. The strategy was based on reasonable assumptions, expert opinion, and empirical observations. The strategy has been designed to address viability and distribution concerns over the long term. The recommendations for maintaining marten population viability have been incorporated into an overall conservation strategy for old-growth associated wildlife in southeast alaska (Suring et. al. 1992).

Rules and regulations adopted to implement NFMA directs the USDA Forest Service to manage wildlife habitats to maintain viable populations of existing native and desirable non-native vertebrate species (USDA Forest Service 1982). additionally, the regulations state "In order to insure that viable populations will be maintained...habitat must be well-distributed so that those individuals can interact...". For this strategy, "well distributed' was more specifically defined to mean that a species has a high likelihood of occurring in each third-order watershed (i.e. >4,500 ha [10,000 ac]) within its current range. Although introduced to many areas in southeast Alaska (Burris and McKnight 1973), martens have been determined to be a desirable non-native vertebrate species in these areas (Sidle and Suring 1986).

The pioneering work by the Interagency Spotted Owl Scientific Committee (Thomas et. al. 1990) on a conservation strategy for the northern spotted owl (*Strix occidentalis caurina*) has provided an excellent model for the application of biological data to the development of a reserve design (Murphy and Noon 1992). Although less biological information was available for martens in southeast Alaska, the approach and appropriate concepts were adapted from the northern spotted owl conservation strategy and applied to marten conservation in southeast Alaska.

DISTRIBUTION AND POPULATION STATUS

Formerly, martens occurred throughout the coniferous forest zone of North America from Alaska across most of Canada, New England, the Allegheníes, the Great Lakes area, the Rocky Mountains south to New Mexico, the Sierra Nevadas, and the Cascade mountains (Clark et al. 1987). Although still common throughout much of its historic range, martens were extirpated from much of the southeastern portion, including southern Ontario, southern Quebec and Prince Edward Island, most of New England, and portions of the Great Lakes region (Clark et al. 1987). Martens have been reintroduced into several areas to reestablish populations (Clark et al. 1987). A natural reinvasion has occurred in northeastern Minnesota and adjacent Ontario (Clark et al. 1987).

Marten populations have declined with the removal of habitat, usually by logging, and unrestricted trapping (Strickland and Douglas 1987). In the northern and western sections of their range, martens have maintained much of their natural distribution although local populations have been occasionally depleted or extirpated. Protection measures, including sanctuaries and closed seasons, have allowed some marten populations to persist (Strickland et al. 1982).

Although indigenous on only the mainland and a few islands, martens are now common throughout most of southeast Alaska (Johnson 1981). During 1930-1950, martens were introduced to Prince of Wales, Chichagof, and Baranof islands (Burris and McKnight 1973, Johnson 1981). Although no records of transplants to Admiralty Island exist, martens may have escaped from nearby fur farms on Windfall or Pleasant island (Beier 1987). Natural populations occur on Kuiu, Kupreanof, Mitkof, and Revillagigedo islands. The limited natural distribution of martens indicates that the geography of southeast Alaska provided many natural barriers and restricted dispersal.

The marten was selected as a management indicator species on the Tongass National Forest (TNF) because forest management activities were expected to affect population abundance and marten pelts represented significant economic value to local residents (Sidle and Suring 1986, USDA Forest Service 1990). Because the TNF encompasses more than 90% of the land area in southeast Alaska, the management of these lands has a major impact on wildlife in the region.

Taxonomy and Form

The marten in North America is a member of the Order Carnivora, Family Mustelidae, Genus _Martes_, Subgenus _Martes_, and Species _Americana_ (Clark et al. 1987). Marten, or American marten, are appropriate common names, but pine marten should be reserved for the European form (_M. martes_). Fur traders often refer to the marten as Canadian, or American, sable to provide a link with the valuable Russian sable (_M. zibellina_). The Holarctic martens - the pine marten (_M. martes_), the Russian sable, and the Japanese marten (_M. melampus_) along with _M. Americana_ - are closely related (Clark et al. 1987). anderson (1970) considered these martens a "superspecies" because of similar morphology, habits, and habitat. Fourteen subspecies of marten are generally recognized that can be separated into 2 groups, "_Americana_" and "_caurina_," that differ in cranial characters and fossil history (Clark et al. 1987). The _caurina_ type occupies the Pacific coast from southeast Alaska to northern California, the northern mountain ranges of the western United States, and several island groups; the _Americana_ type ranges across eastern, central, and northern North America (Giannico and Nagorsen 1989).

In southeast Alaska, the taxonomy of martens is unclear. according to Hall (1981), the mainland is occupied by 2 subspecies of the *Americana* group - *M. a. kenaiensis* north and west of Lynn Canal and *M. a. actuosa* from northern Lynn Canal south to about the Canadian border - while the islands are all listed as *M. a. nesophila* of the *caurina* group. Hall (1981) did not recognize that most of the island populations were established from martens transplanted from the mainland. Giannico and Nagorsen (1989) found 3 morphological groups among the Pacific coast martens that they examined - a Queen Charlotte Islands group, a southeast Alaska group, and a Vancouver Island and coastal British Columbia group. They concluded that the subspecies *nesophila* should be applied only to Queen Charlotte Islands populations, and Vancouver Island and coastal British Columbia martens were aligned with *M. a. caurina*. Because martens in southeast Alaska showed some affinities with the *Americana* subspecies group, Giannico and Nagorsen (1989) suggested that the *caurina* and *Americana* types may integrate here. additional work needs to be done in southeast Alaska to determine whether any areas have unique genetic forms.

In 1934, 10 martens, captured on the mainland near Behm Canal, were released on Prince of Wales Island and 7 martens captured near Cape Fanshaw were released on Baranof Island (Elkins and Nelson 1954). Between 1949-52, 22 martens were released on Chichagof Island near Pelican - 6 were captured on Baranof Island, 1 near Ketchikan, 3 near anchorage, 6 in the Stikine River drainage, 4 on Wrangell Island, and 2 on Mitkof Island (Elkins and Nelson 1954).

Martens are long, slender-bodied, furred animals. Pelage color varies greatly from nearly blond to almost black. Sexual dimorphism is pronounced with males being up to 60% heavier than females (Clark et al. 1987). In southeast Alaska, weights of males averaged about 1100 g while adult females weighed about 850 g (Flynn 1991).

Population Numbers

Martens are difficult to count accurately, so few estimates of population density or trend exist. Most marten studies have made inferences from trapping results. In the Yukon, archibald and Jessup (1984) estimated 0.6 resident martens/km^2 on their study area during the fall with an overwintering density of 0.4 resident martens/km^2. Because all other martens left the study area during late winter, archibald and Jessup (1984) concluded that the observed overwintering density reflected carrying capacity. The initial version of a habitat capability model for the TNF (Suring et al. 1988) used a density figure based on trapping results from Prince of Wales Island. The model assumed an average density of 0.8 martens/km^2 on Prince of Wales Island with a maximum density of 1.5 martens/km^2 in the best habitats. Based on estimated average home range size of radio-collared martens, the density of resident martens on Chichagof Island was estimated at 0.4 martens/km^2 (Flynn 1991). In the current version of the model, habitat capabilities have been reduced 32% to reflect the results of the Chichagof Island studies.

Population Biology

In most areas, the breeding season occurs during early summer (Strickland and Douglas 1987). Martens are induced ovulators with delayed implantation.

after an active pregnancy of about 27 days, the young are born during late spring, probably during april. Counts of corpora lutea have been found to be a good estimator of litter size during the period of delayed implantation (Strickland and Douglas 1987). although corpora lutea counts usually average about 3, the proportion of yearlings pregnant in a population appears to vary among study areas and years of study (Clark et al. 1987, Strickland and Douglas 1987, Bissonette et al. 1988). Little information exists on litter sizes at birth or survival rates of young martens. Most longer-term studies have found marten populations to fluctuate substantially among years of study (Weckwerth and Hawley 1962, Thompson and Colgan 1987b, Bissonette et al. 1988). These fluctuations have been caused by scarce food (Weckwerth and Hawley 1962, Thompson and Colgan 1987b) and disease (Bissonette et al. 1988).

HABITAT RELATIONSHIPS

Most studies of marten habitat relationships have found that mature, coniferous forests provide optimal habitat for martens (Weckwerth and Hawley 1962, Koehler et al. 1975, Mech and Rogers 1977, Soutiere 1978, Steventon and Major 1982, Spencer et al. 1983, Hargis and McCullough 1984, Thompson and Colgan 1987a, Snyder and Bissonette 1987, Bissonette et al. 1989, Buskirk et al. 1989). Magoun and Vernam (1986) reported an important exception, finding martens using post-fire successional stages in interior Alaska. Because of the relatively recent recognition of the ecological significance of old-growth forests (Schoen et al. 1988, Thomas et al. 1988), most previous studies did not classify marten habitats in terms of old-growth condition. Recent studies have recognized old-growth forest types, and documented the importance of old-growth forests as marten habitat (Spencer et al. 1983, Snyder and Bissonette 1987, Bissonette et al. 1989, Buskirk et al. 1989).

Old-growth conifer forests provide martens with important habitat components, including overstory canopy cover, snags, fallen logs, trees with large, exposed root systems, and abundant understory (Clark et al. 1987). Old-growth forests often support abundant small mammal prey because of the lush shrub and forb vegetation and structural diversity of the understory. Overstory cover provides martens with protection from potential avian predators (Clark et al. 1987). The fallen logs, decadent trees, and large snags in old-growth forests provide martens with important resting microsites which have been found to be important for thermoregulation, especially in winter (Buskirk 1984, Buskirk et al. 1989). Because martens store little fat (Buskirk and Harlow 1989), thermal loss needs to be conserved during winter, especially while resting (Buskirk et al. 1988). In Wyoming, martens rested under coarse woody debris below the snow during cold weather (Buskirk et al. 1989). Snags, large live trees with cavities, and down wood probably provide natal den sites (Clark et al. 1987, Jones and Raphael 1990).

Little is known about the habitat needs for denning and the rearing of young. Few marten dens have been described, and the rearing of young has not been studied. The young-rearing period could be a critical stage in the life history of martens. Thompson and Colgan (1987a) found marten reproductive performance reduced during food-scarce years because young females delayed breeding and older females did not rear young.

Several studies have documented the negative impacts of logging on marten populations (Soutiere 1978, Campbell 1979, Steventon and Major 1982, Snyder and Bissonette 1987, Thompson and Colgan 1987a, Bissonette et al. 1989). These impacts include the removal of overstory cover, the loss of coarse woody debris (standing snags and down wood), reductions in prey abundance and hunting efficiency, greater habitat fragmentation, and increased human access. Thompson and Colgan (1987a) found that martens in logged areas had substantially larger home ranges, especially during periods of low prey availability. Bissonette et al. (1989) recognized the need for landscape-level management for martens including the maintenance of large patches of suitable habitat and movement corridors connecting the patches.

Several models have been developed to evaluate marten habitat (allen 1982, Patton and Escano 1983, Spencer 1982, Ritter 1985, Suring 1987, Suring et al. 1988, Lofroth and Banci 1991). These models recognize the importance of late-successional coniferous forests with overstory canopy cover, standing dead wood, and large coarse woody debris. The model developed for southeast Alaska (Suring et al. 1988) also incorporates a factor relating road density to the effectiveness of habitats to provide escape cover from humans.

The southeast Alaska habitat capability model (Suring et al. 1988) uses timber type, elevation, and physiographic type to describes habitats for martens. These stand-level habitat attributes were chosen because they have been mapped on the TNF and available in the USFS's geographic information system (GIS). The more ecologically-based plant associations (Marten 1989) were not used because the reliability of the mapped database was uncertain. The habitat capability model assumed that old-growth forest stands classified as timber volume class 5 (VC5) and greater on low elevation, upland sites provided good habitat for martens. VC5+ stands in beach fringe or riparian areas were ranked as the best habitats. These stands have habitat attributes considered important for martens. all stands above 250 m in elevation were considered to have a reduced habitat capability with all stands above 460 m having no value. Clearcuts and second-growth stands were assumed to have little value as marten habitat, and nonforest areas have no value.

The habitat relationships in this model are currently under study on northeast Chichagof Island, southeast Alaska, and preliminary recommendations have been made on adjustments to the model's habitat capability coefficients (Flynn 1991). The field research found that radio-collared martens preferred VC6 stands greater than the model predicted and clearcuts were used less. also, radio-collared martens preferred low elevation uplands.

Food Habits

Although martens are opportunistic feeders and their diet includes a wide variety of plant and animal matter, most studies have found small mammals to be important foods (Clark et al. 1987, Strickland and Douglas 1987). Voles, especially Microtus sp., usually comprise the highest proportion of the diet (Clark et al. 1987). Nagorsen et al. (1989) found small mammals, deer carrion (*Odocoileus hemionus*), birds, and salmonid fish the major food items of martens on Vancouver Island. In the Yukon, Slough et al. (1989) found marten diets comprised mostly microtine rodents. Marten population declines have been related to population declines of prey species (Weckwerth and Hawley 1962, Thompson and Colgan 1987b). Small mammal populations, especially

Microtus sp., often fluctuate greatly among years. although little work has been done on marten food habits in southeast Alaska, small mammals probably provide most of the diet. The distribution of small mammals in southeast Alaska is quite variable; generally species richness is greater on the mainland with few species on the islands (Hall 1981). Little is known about population abundance or fluctuations in numbers of small mammals in southeast Alaska.

SPATIAL RELATIONSHIPS

Home Range

Martens populations have been reported as being composed of residents, temporary residents, and transients (Clark et al. 1987, Thompson and Colgan 1987b). An individual that shows site fidelity is considered to have a home range (Spencer et al. 1990) and be a resident (Flynn 1991). Transients show no site fidelity and are often dispersing juveniles. The occupancy of a home range provides an individual with several important advantages including living in a small, well-known area that provides all the necessities of life (Vaughan 1972). Also, the less an animal must range, the chance of encounters with predators is reduced.

Home ranges for martens have often been reported (Buskirk and McDonald 1989), but exact comparisons among studies are difficult because researchers have often used different data collection and analytical procedures. Also, an operational definition of home range (Spencer et al. 1990) was seldom used.

Most studies have found martens to exhibit the common mustelid spatial pattern; home ranges of males and females overlap with intrasexual intolerance (Clark et al. 1987). Typically, the home ranges of males are about 2 to 3 times larger than those of females. Buskirk and McDonald (1989) reported mean home range size for males varied from 0.9 to 19 km and female home ranges varied from 0.6 to 13 km for 9 studies that used radio telemetry and the 100% convex polygon analytical method. a summary of 8 studies prepared by USDA Forest Service staff reported mean home size of 4.6 km for males and 3.1 km for females (USDA Forest Service 1990a). While home range size varied among the studies reviewed, Buskirk and McDonald (1989) found no obvious pattern between home range size and geographic location. Few studies have related home-range size to habitat quality or resource abundance (Buskirk and McDonald 1989). Thompson and Colgan (1987b) found home ranges of martens in cut-over areas were larger in area compared with uncut areas, and average home range size increased substantially during scarce-food years, especially in cut-over areas.

Archibald and Jessup (1984) found home ranges of male and female martens in the Yukon to average 6.2 and 4.7 km^2. In Ontario, Thompson and Colgan (1987b) found the home ranges of females in uncut areas averaged 4.4 km^2 during scarce-food years and 1.0 km^2 during abundant-food years. In cut-over areas, female home ranges averaged 12.7 km^2 during scarce-food years and 3.1 km^2 during abundant-food years (Thompson and Colgan 1987b) .

Home range size of radio-collared martens on Chichagof Island during 1990-91 averaged 6.2 km^2 (range - 3.2 to 11.3 km^2) for 6 resident male martens, and 3 resident female martens home ranges averaged 4.4 km^2 (range - 3.2 to

5.2 km^2) (Flynn 1991). The home ranges of these animals had little intrasexual overlap. Based on limited small mammal trapping, prey population appeared to be at "moderate" levels during 1990-91 (Flynn 1991). Based on studies in Ontario (Thompson and Colgan 1987b), marten home ranges would be expected to expand substantially during a poor-prey year. Thus, a male's home range area is expected to expand by at least 50% during a poor-prey year to about 9.0 km^2 and a female's to about 6.4 km^2.

Movements

Little data have been collected on dispersal distances because radio-collared martens have been difficult to track over large areas and the birth sites of captured martens are seldom known. In the Yukon Territory, archibald and Jessup (1984) had a male and female killed 8.5 and 10.0 km from their known home ranges. On Chichagof Island, the maximum distance traveled from capture sites averaged 26.1 km (N = 8, SD = 11.4) for radio-collared transient male martens and transient females averaged 22.5 km (N = 4, SD = 7.1) (Flynn 1991). The maximum distance traveled recorded for a male was 42 km, and a female moved 32 km. Based on the Chichagof Island information, 68% of transient martens would be expected to move at least 15 km. Slough (1989) found 30 transplanted martens to move an average of 34.5 (males) and 22 km (females) from release sites in the Yukon. Transplanted martens would be expected to show atypical movement patterns.

Studies have reported that martens seldom cross large open areas (Clark et al. 1987). Bissonette et al. (1988) observed a marten crossing a clearcut more than 250 m (800 ft) wide only once. although martens may occasionally swim short distances, large bodies of water probably act as movement barriers. Few islands in southeast Alaska that are separated by more than 60 m of salt water have natural populations. Forested corridors, especially riparian and beach fringe zones, are believed to be important to facilitate movements and dispersal (Clark et al. 1987, Bissonette et al. 1989). Martens will use relatively narrow corridors if the travel distance is short. Several radio-collared martens have moved to the west side of Port Frederick where a 100 x 200 m (330 x 660 feet) forested strip is the only land connecting the northeast lobe of Chichagof Island with the remainder of the island (Flynn 1991)). Bissonette et al. (1988) recommended that corridors be at least 200 m (660 feet) wide, and riparian zones be managed as travel corridors when available.

HABITAT MANAGEMENT

Southeast Alaska

The conservation of habitats in southeast Alaska has been strongly influenced by Congressional action. In 1980, the Alaska National Interest Lands Conservation act (aNILCa) established several large wilderness areas and extended Glacier Bay National Park. These land allocations have protected some marten habitats from commercial exploitation, but the legislation also provided several subsidies to logging companies that promoted the clearcutting of lands unprotected by the legislation. More recently, the Tongass Reform act of 1990 allocated more lands to wilderness and a new nonlogging land status, required 100-foot, no-cut buffers along certain streams, and repealed many of the ANILCA provisions that subsidized logging on the TNF.

Current habitat management on the TNF is directed by the Tongass National Forest Management Plan (TLMP) which was adopted in 1979 and amended in 1985. Because adoption of TLMP predated the implementation of the National Forest Management act of 1976 (NFMA), none of the requirements of NFMA were incorporated. Marten habitat requirements received little attention during the TLMP planning process. Under the plan, individual watersheds on the TNF with martens present were identified. Within occupied watersheds, all forested habitats were considered marten habitat; all forested habitats were considered equal in quality. The plan prescribed that 25% of existing upland old-growth habitat would be "retained" in each drainage. additionally, 50% of existing old-growth forest in the beach fringe would be retained. TLMP did not identify where the retained habitat was located, or provide any direction on retention layout within the landscape.

The retention concept of habitat management as implemented by TNF personnel under TLMP has not worked because:

No specific standards or guidelines for retention layout existed that considered forest type, landscape configuration, or habitat block size;

No consideration was given to access changes caused by infrastructure development (e.g. roads);

The lands allocated to wildlife habitat retention had to meet multiple species objectives, so high-profile species received the most consideration (e.g. eagles, deer, and bears);

Low quality habitats were often included as retention because of their lower commercial value as timberlands; and

Habitats retained for wildlife habitat during one planning period have often been allocated to timber production during a subsequent timber sale.

Western United States

The habitat needs of martens, and furbearers in general, have received little attention from land management agencies across the western United States. Recently, the status of several furbearers, especially martens and fishers, has been challenged by the public (USDA Forest Service 1990a). In response to public concerns and NFMA, martens have been selected as management indicators on several national forests and management guidelines have been drafted. although a thorough review of all standards and guidelines was beyond the scope of this paper, a few efforts are worth noting.

Forest Service staff in Region 5 (California) have drafted preliminary interim guidelines for identifying and managing marten habitats (USDA Forest Service 1990a). These guidelines make several important assumptions including:

1. Habitat areas need to support a reproductive unit that can contribute individuals to the population (i.e. 1 adult male and 2 adult females in

high quality habitats and 1 adult male and 3 adult females in low quality habitats);

2. Home ranges can be used to determine marten spatial needs and home range size is negatively related to habitat quality;

3. a reproductive unit needs 570 ha (1,400 ac) of high quality habitat, 770 ha (1,900 ac) of moderate quality, or 930 ha (2,300 ac) of low quality habitat;

4. Riparian corridors are important travelways and foraging areas;

5. Absence of roads is preferred;

6. Forested corridors should be 46 to 90 m (150-300 feet) wide; and

7. Habitat units should be within 5-16 km (2-6 mi) depending on size of unit.

although these standards and guidelines present some good concepts, they fail to adequately recognize or evaluate the problem of population fragmentation.

Yukon Territory

Management guidelines for martens have been developed by Fish and Wildlife Branch staff of the Yukon Territory government to maximize harvests (Slough and Smits 1985). In areas of good marten habitat, they recommend traplines be spaced at least 9.6 km, so a reserve of about 94 km^2 remains between them. Each reserve would completely contain the home ranges of at least 3 resident males and provide habitat for about 20 adult females. If the habitat is poorer in quality or the amount of good habitat is limited, they recommend that the same size reserves be maintained, but the trapping effort along the trap lines should be reduced to protect adult females.

VIABILITY/DISTRIBUTION CONCERNS

Although most of the original forested land in southeast Alaska was in an old-growth condition, industrial-scale logging has converted large areas of old-growth forest habitat into clearcuts and second growth. Logging of old-growth forest habitats on the Tongass National Forest causes a reduction of marten habitat capability. The timber harvest has been focused in high-volume, old-growth stands at low elevation (USDA Forest Service 1990). In addition to the absolute amount of high quality habitat removed by logging, the dispersed-setting harvest pattern has fragmented the landscape (Samson et al. 1989). About 162,000 ha (400,000 ac) of old-growth habitats have already been logged on the Tongass National Forest, and the current TLMP schedules an additional 0.7 million ha (1.7 million ac) (USDA Forest Service 1990). Also, many miles of new road will be constructed to facilitate timber extraction. Many areas are now extensively roaded, most notably the northern portions of Chichagof and Prince of Wales islands. Road systems open previously inaccessible lands to human activity, including trapping. Martens are relatively easy to capture by trappers and vulnerable to overharvest (Strickland and Douglas 1987). Before roading, the interiors of islands act as reservoirs for marten populations. Animals from unexploited interior areas are able to disperse into overexploited areas near the beach fringe after the trapping season ends. Extensive roading results in most marten home ranges being intercepted by roads, resulting in the entire population being vulnerable to overharvest.

Past and potential future loss of habitat capability on the TNF raises a concern for marten population viability and establishes the need for a conservation strategy. The status and natural history of martens indicates that their viability and distribution in southeast Alaska may be jeopardized by the modification of habitats by logging activities because:

1) habitat quality is greatly degraded by clearcut logging;

2) habitat fragmentation can further degrade marten habitat;

3) martens are highly vulnerable to overharvest by trapping; and

4) marten trapping is relatively difficult to manage.

CONSERVATION STRATEGY

A multi-faceted conservation strategy is recommended for martens in southeast Alaska. an adequate amount of habitat needs to be maintained in the proper distribution to ensure the long-term survival of the species. Long-term survival will occur only if the species can persist during low periods of population cycles. Protected habitats need to be arranged, so the species remains well-distributed across southeast Alaska. Conservation areas need to be connected by travel corridors, so animals can move among protected habitat areas. Additionally, the martens in protected habitat areas need to be protected from overharvest, especially during low population years.

Large blocks of habitat that are capable of supporting a number of reproductive martens, and spaced closely enough to facilitate dispersal between blocks, are far more likely to ensure viable populations and distribution of martens than the same amount of habitat fragmented across the landscape. These large blocks of protected habitat are called Habitat Conservation Areas, or HCAs (Thomas et al. 1990). Martens in HCAs supporting multiple reproductive animals are less vulnerable to random fluctuations in birth and death rates, more resistant from small-scale natural disturbances, and more secure from human disturbances. Martens in larger blocks of high quality habitat will be less vulnerable than martens in smaller blocks of poorer habitat.

In order to ensure long-term population viability of martens in southeast Alaska, a network of HCAs should be established on the Tongass National Forest. Three categories of HCAs are recommended. a Large Habitat Conservation area (HCA) would be capable of supporting 50 resident martens (at least 25 females) during a poor-prey year. The Large HCAs should be at least 16,000 ha (40,000 ac) in total area, but not exceed 32,000 ha (80,000 ac). The habitat should be composed of at least 8,000 ha (20,000 ac) of VC4+ old-growth forest, including at least 4,000 ha (10,000 ac) of VC5+ old-growth forest. These HCAs should be located about every 40 km (25 mi), or one LHCA in each physiographic subprovince. The martens in Large HCAs should have high short-term viability and act as reservoirs for adjacent areas with low short-term viability. A distance of 40 km (25 mi) was chosen because this distance is near the maximum dispersal distance recorded for martens.

Medium HCAs could be of 2 types. Medium HCA1s would be capable of supporting 10 resident martens (at least 5 females) during a poor-prey year be located about every 15 km (9 mi). The Medium HCA1s should be at least 3,200 ha (8,000 ac) in size, but not more than 6,400 ha (16,000 ac) and composed of at least 1,600 ha (4,000 ac) of VC4+ old-growth forest, including at least 800 ha (2,000 ac) of VC5+ old- growth forest. If a suitable Medium HCA1 can not be found within 15 km (9 mi), then a Medium HCA2, capable of supporting 20 resident martens (at least 10 females) during a poor mi-prey year should be located every 25 km (16 mi). The Medium HCA2s should be at least 6,400 ha (16,000 ac) in size, but not more than 13,000 ha (32,000 ac) and composed of at least 3,200 ha (8,000 ac) of VC4+ old-growth forest, including at least 1,600 ha (8,000 ac) of VC5+ old-growth forest. The martens in these HCAs would have relatively low short-term viability, but could be recolonized frequently from other HCAs. The distance of 15 km (9 mi) is within the estimated dispersal distance of at least 68% of radio- collared transient martens, and 25 km (16 mi) is within the mean dispersal distance.

A Small HCA, which should support at least 2 resident martens (at least 1 female) during a poor-prey year, should be established in each watershed greater than 40 km^2 (15 mi^2). These HCAs should be at least 650 ha (1,600 ac) in size and composed of at least 325 ha (800 ac) VC4+ old-growth forest, including at least 160 ha (400 ac) of VC5+ old-growth forest. The Small HCA should not contain more than 10% nonforested area. Small HCAs would provide habitat for a single reproductive unit.

All HCAs should be connected by forested corridors to facilitate dispersal. Old-growth forest riparian and beach fringe habitats should be used as corridors where available. Corridors should be at least 100 m (330 feet) wide if the travel distance is greater than 100 m (330 feet).

All Large and Medium HCAs should be mapped in the forest plan. HCAs should be selected based on the degree of connectivity to adjacent HCAs. Small HCAs and travel corridors should be allocated in the forest plan and identified during project planning.

The construction of roads should be minimized in HCAs and corridors. Roads needed for forest management activities should be routed along the boundaries of Large and Medium HCAs. any roads should be approved by an interagency team before construction, and the future management of the road should be clearly stated in planning documents. Approved roads should be closed except for timber extraction activities. These roads should be closed to marten trapping and all access closed during the open trapping season.

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CONSERVATION OF THE PRINCE OF WALES ISLAND RIVER OTTER IN SOUTHEAST ALASKA

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DISTRIBUTION AND POPULATION STATUS

The northern river otter (*Lutra canadensis*) was originally found over much of North America (Toweill and Tabor 1982). Northern river otters occur throughout most of Alaska except on the aleutian Islands, Bering Sea islands, and arctic coastal plain (Manville and Young 1965, Hall 1981). The 19 subspecies described for North America (Hall and Kelson 1959) have been consolidated into 7 subspecies (Hall 1981). *L. c. pacifica* occurs throughout interior Alaska; *L. c. kodiacensis* occurs only on Kodiak Island; *L. c. mira* (Prince of Wales Island river otter) occurs on the islands and mainland throughout southeast Alaska (Hall 1981). The Prince of Wales Island river otter was originally described as a separate species (i.e., *L. mira*) (Goldman 1935, Hall and Kelson 1959). Reviews of the taxonomic status of river otters indicated that river otters from southeast Alaska are distinctly different morphologically from interior river otters (Zyll de Jong 1972, Fagen 1986). However, maintaining species status for *L. mira* may overemphasize the variation observed between populations (Fagen 1986).

Early reports following settlement of North America indicate that river otters were abundant throughout their range (Melquist and Dronkert 1987). Human encroachment, habitat destruction, and overharvest have resulted in the decline and disappearance of river otters from about 2/3 of their original range (Jenkins 1983, Melquist and Dronkert 1987).

The Alaska Department of Fish and Game manages the harvest of river otters throughout southeast Alaska. Total harvest in this area was reported to be over 500 animals during the 1989-90 season (Alaska Department of Fish and Game 1991). Potential capability of habitats to support river otters in southeast Alaska is estimated to be approximately 6,800 animals (USDA Forest Service 1990:3-583). Populations of river otters have been characterized as stable or increasing in Alaska (Endangered Species Scientific authority 1978).

PATTERNS OF HABITAT USE

River otters have adapted to a variety of habitats throughout North America but they are always closely associated with aquatic environments. Coastal habitats are especially productive because of the variety and abundance of food items available for river otters (Larsen 1984, Stenson et al. 1984). Habitat selection by river otters along the coastline in southeast Alaska appears related to the availability of food resources and adequate cover (Home 1982, Larsen 1983, Woolington 1984). Beaches characterized by convex shorelines, short intertidal lengths, and the presence of bedrock substrate were selected by otters possibly in response to presence and availability of prey. Cottids, Scorpaenids, and Hexagrammids occurred most frequently in otter diets in southeast Alaska (Larsen 1984). These fish often occur in intertidal areas with fairly steep beaches which are often located adjacent to convex shorelines (Hart 1973). River otters hauling out on and crossing beaches with short intertidal lengths with rocky substrates are also less exposed to potential predators than they would be on beaches with long intertidal lengths consisting of fine particulate substrate.

Although beach characteristics affected river otter use of habitats, river otters did not utilize beaches with preferred foraging characteristics when these areas were adjacent to clearcuts (Larsen 1983). Five to 20 year old clearcuts were used less than expected by river otters while forested habitats were used in proportion to availability. This was apparently because of dense shrub growth, extensive slash, and lack of an overstory canopy in clearcuts. River otters in southeast Alaska tended to select areas for use that were relatively free from extensive vegetative debris and dense shrub growth, and preferred sites with a canopy closure of >50%. Of the 4 family groups observed in detail by Home (1982) in Glacier Bay, 3 were associated with forested habitats at least 180 years old. The fourth family group used a 50 year old successional stand. Old growth hemlock forests provided habitat for 9 of 12 family groups not observed in detail by Home (1982). The remaining 3 family groups were associated with willow (<u>Salix</u> sp.) - alder (<u>Alnus</u> sp.) communities.

River otters in southern southeast Alaska made extensive use of natural cavities within 75 ft of beaches as daytime resting sites (Larsen 1983). The burrows most often used were formed by the roots of large conifer trees and decaying snags. Cavities under snags were used as burrows more often than any other structures. The mean diameter at breast height of all trees and snags associated with cavities used as burrows was 34 in. River otters in Glacier Bay consistently occupied burrows under the roots of trees within forested areas (Home 1982). One burrow located during Home's (1982) observations was under the roots of a single tree near the edge of a cliff.

Throughout most of the year the majority of river otter activity occurs within 100 ft of the shoreline (Larsen 1983, Woolington 1984). However, from May through July female river otters use inland habitats generally within 0.5 mi of the coastline as natal denning sites (Woolington 1984). Natal dens occurred on well drained sites near streams in old growth habitats. Stream courses were used as travel corridors between natal den sites and foraging areas on the coastline.

A proportion of river otters periodically move into inland habitats associated with streams and lakes (Home 1982, Larsen 1983, Woolington 1984). Otters apparently travel extensively throughout stream and lake systems utilizing areas with greatest food availability (Melquist and Hornocker 1983). Streams in southeast Alaska support populations of sculpins (Cottus spp.), which are the most available food item for river otters in this area (McLarney 1968, Mason and Machidori 1975, Larsen 1984).

HOME RANGE/TERRITORY

Size and use of home ranges by river otters is influenced by habitat quality, prey availability, weather, topography, reproductive cycle, and conspecifics (Melquist and Hornocker 1983). Adult males generally have the largest home ranges, especially during mating seasons (Melquist and Dronkert 1987). Lactating females restrict their movements within their home range. Home ranges tend to be large in those areas where food and cover are widely dispersed (e.g., mountainous areas and stream habitat) and small in those areas where food and cover are closely interspersed (e.g., marine coastal areas) (Melquist and Dronkert 1987). In southeast Alaska an adult male used an area of 8 mi^2 with a total shoreline length of 25 mi; a yearling male used an area of 9.6 mi^2 with a total shoreline length of 11.8 mi; and an adult female used an area of 3.4 mi^2 with a total shoreline length of 13.5 mi (Larsen 1983). Woolington (1984) reported shoreline lengths used by 2 adult males in southeast Alaska of 2.1 mi and 14.8 mi; shoreline lengths used by 4 adult females ranged from 1.8 mi to 5.6 mi. Shoreline lengths ranged from 1.9 mi to 5.9 mi for 4 family groups also in southeast Alaska (Noll 1988). Individual river otters in each family group, which were made up of adult otters, juveniles, and pups, were found to share home ranges with nearly identical boundaries.

Territoriality in river otters appears to vary from population to population. River otters in Idaho, alberta, and Louisiana exhibited extensive home range overlap (Melquist and Dronkert 1987). River otters in coastal marine habitats in Texas and Alaska appeared territorial with very little overlap in home range (Home 1982, Larsen 1983, Foy 1984, Woolington 1984, Noll 1988). Foy (1984) suggested that territorialism may exist in well-established populations that have an evenly distributed food supply.

POPULATION DENSITIES

Densities of river otters in southeast Alaska have been estimated to be 1 river otter per 1.28 mi of coastline (Home 1982), 1 river otter per 1.24 mi (Larsen 1983), 1 river otter per 0.73 mi (Woolington 1984), and 1 river otter per 0.62 mi of coastline (Noll 1988). The mean of these 4 estimates is approximately 1 river otter per 1 mi of coastal shoreline.

MOVEMENTS/DISPERSAL

Movements of river otters generally follow the coastline in southeast Alaska, although they may occasionally cut across peninsulas (Home 1982, Larsen 1983, Woolington 1984). As noted earlier, Larsen (1983) found evidence of river otters using freshwater habitats. Home (1977) observed otters moving between coastal and inland waters. Woolington (1984) documented movements of female river otters up to 0.5 mi inland along stream courses to establish natal dens. The longest straight-line movements by river otters during a 24-hour period documented by Larsen (1983) were 3.2 mi for an adult male and 2.9 mi for a yearling male. River otters were observed to swim across 1.9 mi of open salt water (Larsen 1983).

River otters usually disperse in April and May at 12-13 months of age (Melquist and Hornocker 1983). However, not all subadults leave their natal areas. Information is limited on the dispersal patterns of river otters in southeast

Alaska. A juvenile male otter emigrated from his natal home range at 22 months of age (Noll 1988).

VIABILITY/DISTRIBUTION CONCERNS

The Prince of Wales Island river otter is a unique form of river otter found only in southeast Alaska and western British Columbia. Information from furbearer harvest statistics and estimates of habitat capability indicate that viability of this animal is not threatened in southeast Alaska.

This animal is strongly associated with saltwater beach fringe and freshwater riparian habitats (Larsen 1983, Woolington 1984, Noll 1988). Although availability of food appears to have the greatest influence on habitat use, adequate shelter (e.g., dens, burrows, and resting sites) must also be available (Melquist and Hornocker 1983, Melquist and Dronkert 1987). Larsen (1983) found in southeast Alaska that sites adjacent to clearcuts were not used by river otters even if good foraging habitat was available. Analysis has been completed which indicates that under some land management scenarios for the Tongass National Forest habitat capability for river otters may be reduced by as much as 65% on some portions of the Forest (USDA Forest Service, unpublished data). These findings indicate that if extensive timber harvest occurs in beach fringe habitats throughout southeast Alaska, the distribution of river otters would be affected.

CONSERVATION STRATEGY

The Prince of Wales Island river otter uses a narrow strip of habitat adjacent to saltwater, estuaries, streams, and lakes. Maintaining the saltwater beach fringe (i.e., 500 ft from mean high tide), estuary fringe (i.e., 1000 ft from mean high tide), and riparian habitats associated with streams and lakes throughout the river otter's range in southeast Alaska will help to ensure that the river otter's current distribution will be maintained.

MONITORING RECOMMENDATIONS

Scent-station indices, winter ground and aerial track counts, mark and recapture techniques, and field-sign surveys have been suggested for use in monitoring populations of river otters (Melquist and Dronkert 1987). When these techniques are used with information from other sources (e.g., fur dealer reports, trapper questionnaires, fur sealing certificates) it is possible to evaluate the status of the population density and distribution of river otters.

Evaluations of scent-station indices have shown mixed results. Robson and Humphrey (1985) concluded that scent-station indices may be useful for a 1-time detection of river otters but that habituation and loss of interest in the scent stations made the technique unsuitable for monitoring populations. Clark et al. (1987) had better success with visitation rates with the scent-station technique. They speculated the differences may have been because of different habitats, population levels of river otters, scent attractant used, or construction of the scent station. Clark et al. (1987) considered the scent-station technique to be an effective method to determine the distribution of otters and changes in that distribution. However, they did not recommend its use as a technique to determine changes in otter population densities. Both studies indicated that surveys of field-sign provided information similar to scent-station indices with much less investment of time and money.

Observation of tracks in the snow has been proposed as a technique to estimate populations of river otters (Reid et al. 1987). This technique is not suitable for most of southeast Alaska primarily because of inconsistent snow cover. Its applicability may also be limited in marine environments because of the tendency of river otters to travel in the water (Woolington 1984); therefore remaining undetected by counts of tracks in the snow.

Mark-recapture techniques are usually not successful with river otters because of the low capture rate. Knaus et al. (1983) suggested the use of radioactive materials to "label" scats of injected river otters. This may be a workable technique but it requires the capture and handling of animals, elaborate equipment (i.e., scintillation counter), and extensive field effort involving multiple visits.

Until reliable and efficient monitoring techniques are developed, the best approach to monitoring river otters in southeast Alaska may be to establish surveys of field-sign to document their distribution and subsequent changes in distribution. This information should be evaluated with information collected from trappers on harvest levels of river otters to provide an indication of population status and distribution.

RESEARCH RECOMMENDATIONS

Additional information on demographic parameters, feeding habits, habitat use patterns, home range requirements, dispersion and movement patterns of river otters is needed in southeast Alaska to ensure that viable populations are maintained throughout the area. The habitat capability model currently being developed for river otters in southeast Alaska needs to be evaluated to ensure it adequately represents habitat relationships in this area (Suring et al. 1987). Information on population structure and recruitment are needed to develop population models that may be used to estimate the long-term viability of the species. Dispersion of river otters is essential to maintaining well distributed populations throughout southeast Alaska. However, information on this aspect of the natural history of river otters is almost nonexistent. Availability of such information is important to evaluate the effectiveness of conservation efforts for the species. Development of reliable and efficient monitoring techniques for river otters in southeast Alaska is also essential so that response of populations to management actions may be determined.

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CONSERVATION OF MOUNTAIN GOATS IN SOUTHEAST ALASKA

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DISTRIBUTION AND POPULATION STATUS

Four subspecies of mountain goat (Oreamnos americanus) have been described in western North America (Hall 1981). O. a. missoulae occurs from western Montana, through Idaho, and north into southwest alberta and southeast British Columbia. O. a. americanus is found in the Cascade Mountains from central Washington State into southwest British Columbia. O. a. columbiae is found throughout northern and western British Columbia, into Yukon Territory and Northwest Territory, and on the mainland in southeast Alaska. O. a. kennedyi is found along the coastal areas of Prince William Sound and adjacent inland areas. Mountain goats have been introduced into the Black Hills of South Dakota, several areas of Montana and Colorado, northeastern Oregon, and northwest Washington State. In Alaska, mountain goats were introduced to Baranof Island in 1923, Kodiak and Chichagof islands in the early 1950s, and Revillagigedo Island in 1983 (Burris and McKnight 1973, Smith and Nichols 1984).

The population of mountain goats in Alaska has been estimated at 15,000 to 25,000 animals; one-third of these reside in southeast Alaska (Ballard 1977, Johnson 1977, Fox et al. 1989). Suitable habitats for mountain goats on the Tongass National Forest in southeast Alaska have been estimated to have the capability to support nearly 6,300 animals (USDA For. Serv., unpubl. data). Populations in parts of southeast Alaska appear to be increasing (Smith 1984). Annual harvests of mountain goats in Alaska have recently averaged about 400 animals, with an average of 170 animals taken in southeast Alaska (Alaska Department of Fish and Game 1991).

PATTERNS OF HABITAT USE

A variety of vegetative food items are eaten by mountain goats throughout the year. These include foliage and seed heads of grasses, sedges, and rushes; foliage, stems, and flowers of forbs; leaves and twigs of shrubs and trees; leaves of ferns; and the entire aerial portion of mosses and lichens (Wigal and Coggins 1982). Foraging sites and forage composition change throughout the year.

Mountain goats have demonstrated a preference for shrub communities associated with south-facing avalanche slopes in the early spring (Schoen and Kirchhoff 1982). The herbaceous understory is one of the first areas to initiate plant growth in the spring. Rhizomes and new shoots of forbs and ferns in this community provide mountain goats with highly nutritious forage (Klein 1953, Hieljord 1971).

As snow melts during the summer, mountain goats move to higher elevation subalpine and alpine areas to feed on plants emerging from melting snowbanks (Fox 1978, Schoen and Kirchhoff 1982, Smith 1986). The new growth of sedges and forbs abundant in these areas are selected (Hieljord 1971).

Food available to mountain goats during winter is much more restricted than during other seasons (Fox and Smith 1988). Accumulation of heavy wet snow in the alpine and subalpine areas, especially in southern southeast Alaska, covers available forage and forces mountain goats to lower elevation forested areas (Smith 1986). Conifers, lichens, mosses, and shrubs are the plant species which comprise the bulk of mountain goats diet during winter (Fox et al. 1989). In some areas of northern southeast Alaska the snow is dryer and lighter. In these areas snow is blown off of ridge tops exposing plants and allowing the mountain goats to forage at higher elevations. Alpine forbs and graminoids continue to be important components of the mountain goats' diet throughout the winter in these areas.

Behavioral strategies of mountain goats to avoid predators, particularly gray wolves (Canis lupus), also affect habitat use by mountain goats. Mountain goats generally move into steep and broken terrain characterized by the presence of cliffs, when approached by gray wolves (Fox and Streveler 1986). Fox (1983) reported most use of habitats by mountain goats in southeast Alaska was within 660-980 ft of cliffs. McFetridge (1977a) also reported that 95% of observations of mountain goats were within 980 ft of escape terrain during October and November. Hieljord (1971) estimated that mountain goats on Kodiak Island and in the Kenai Mountains spent most of their time within 900 ft of escape terrain during summer. Smith (1986) reported that 95% of all relocations of radio-collared mountain goats in southern southeast Alaska were within 1,300 ft of cliffs and that all relocations were within 2,600 ft. The need for escape terrain to be in close proximity to food resources is a critical factor in delineating habitat for mountain goats.

HOME RANGE/TERRITORY

Year-round home ranges of mountain goats in southeast Alaska generally vary from 4 mi^2 to 8 mi^2, with the maximum recorded being nearly 35 mi^2 (Smith 1986). Areas of use within home ranges tended to change with the season of the year (Schoen and Kirchhoff 1982). However, overlap of seasonal ranges was extensive. The seasonal separation of ranges is generally because of a vertical migration with low elevation habitat being used in the winter and spring and high elevation habitat being used in summer (Fox 1978, Schoen and Kirchhoff 1982, Smith 1986, Fox et al. 1989). Adult males had larger home ranges and more distinct seasonal ranges than did females (Fox et al. 1989). The mean distances between centers of summer and winter ranges for males and females were 1.8 mi and 1.2 mi, respectively (Schoen and Kirchhoff 1982).

POPULATION DENSITIES

Population densities of mountain goats in southeast Alaska range from 1.3 per mi^2 to 10.9 per mi^2 and average about 3.9 per mi^2 (Fox 1984, Smith and Bovee 1984). Smith and Bovee (1984) estimated the density of mountain goats on winter range in southern southeast Alaska to be 11.4 animals per mi^2. These densities are consistent with those reported for other populations throughout their range (Fox et al. 1989).

MOVEMENTS/DISPERSAL

Studies of mountain goats through radiotelemetry have indicated that females tend to be sedentary and use relatively small home ranges (Smith 1986). As indicated previously, the mean distance between centers of summer and winter ranges was 1.2 mi for females (Schoen and Kirchhoff 1982). Female mountain goats also show high fidelity to summer and winter ranges from one year to the next and do not explore new areas (Rideout 1977, Schoen and Kirchhoff 1982, Smith and Raedeke 1982).

Seasonal movements of male mountain goats tend to be longer than those of females (i.e., 1.8 mi mean distance between centers of summer and winter ranges) (Schoen and Kirchhoff 1982). Males also exhibit lower fidelity than females to summer and winter ranges from one year to the next. This may result from extensive movements (e.g., more than 10 mi) by males during the rut (Smith 1986). Several studies have found that males move between ridges occupied by female goats during the rut (Geist 1964, Smith 1976, Schoen and Kirchhoff 1982, Smith and Raedeke 1982).

Young mountain goats appear to establish home ranges within or adjacent to the ranges of the females groups in which they were reared (Geist 1971, Schoen and Kirchhoff 1982). Long distance dispersal of mountain goats is generally not the case. However, 1 subadult female was observed to disperse over 45 mi on the Cleveland Peninsula in southern southeast Alaska (Smith and Raedeke 1982).

VIABILITY/DISTRIBUTION CONCERNS

Mountain goats are more sensitive to habitat change and hunting pressure than any other big game species in North America (Chadwick 1983). Studies throughout their range in North America have reported significant declines in populations of mountain goats following modification of habitats and disturbance from human activities (Chadwick 1973, Quaedulieg et al. 1973, Kuck 1977, Phelps et al. 1983).

The amount and distribution of escape terrain within suitable winter habitat is the key factor in mountain goat use of winter ranges (Fox et al. 1989). Any management activity that has the potential of reducing the quality or quantity of winter range will probably have a significant impact on goat populations in the area.

McFetridge (1977b) indicated that use of suitable habitats by mountain goats may also be reduced as a result of human activities. Chadwick (1973) reported that mountain goats will abandon otherwise suitable habitat following initiation of human activities. Five of 7 populations of mountain goats evaluated in British Columbia experienced population declines (Pendergast and Bindernagel 1977). Four of the declining populations were accessible by road; none of the stable populations were accessible by road. The potential for adverse affects of timber harvest and mining activities on mountain goats and their habitats throughout southeast Alaska currently exists (Schoen and Kirchhoff 1982, Smith and Raedeke 1982, Fox 1983, Smith 1986).

The small size and patchy distribution of mountain goat subpopulations in southeast Alaska and the limited movement of these animals provides a high potential for inbreeding or periodic extinction of subpopulations in this area (Smith and Raedeke 1982). Habitat alteration, human activity, and illegal hunting associated with management activities may reduce movements of male mountain goats during the rut and increase mortality in all mountain goats. The resulting effects of genetic isolation and increased harassment and mortality may lead to extinction of subpopulations of mountain goats. Berger's (1990) review of persistence in populations of bighorn sheep (Ovis canadensis) showed that 100% of populations with fewer than 50 individuals became extinct within 50 years and populations with greater than 100 individuals persisted for at least 70 years. Although persistence studies are not available for mountain goats, the results from the bighorn sheep study are indicative of potential persistence problems for mountain goats. Such problems may eventually result in significant changes in the distribution of mountain goats throughout southeast Alaska.

CONSERVATION STRATEGY

Measures should be taken within areas of mountain goat habitat that may be affected by management activities to protect and maintain distinct subpopulations and populations of mountain goats.

Essential winter habitat for mountain goats should be identified during project planning (i.e., forested areas within 1,300 ft of escape terrain). Efforts should be made to maintain at least 80% of the potential habitat capability in these areas as described by the habitat capability model developed for southeast Alaska (Suring et al. 1987).

Essential winter habitat, kidding areas, and other sites important to the maintenance of populations of mountain goats should be identified and protected from modification and disturbance. Camps, mineral exploration and operation facilities, log dumps and transfer facilities, and other facilities should be located more than 1 mi from essential habitats. Resource exploration and development activities and construction and use of roads should be seasonally restricted within essential habitat areas to avoid disturbance of mountain goats. Activities within essential winter habitat should be restricted from 1 November through 1 May. Activities within kidding areas should be restricted from 1 May through 1 august.

Travel corridors used by mountain goats between important wintering sites should be identified and maintained, especially when they occur in forested areas (Fox et al. 1989).

MONITORING RECOMMENDATIONS

Surveys should be conducted annually in the fall with aircraft in areas occupied by mountain goats throughout southeast Alaska. Surveys should be conducted to provide information on total numbers, distribution, elevational and seasonal density, age ratios (kids, yearlings, and adults), sex ratios (adult male to adult female), and habitats used (Wigal and Coggins 1982). Surveys of the same areas may be conducted from the ground or water in the spring to ensure as complete a count as possible and to monitor habitat use (Fox 1984, Wood 1990). Hunter and harvest information may also be obtained through hunter reports required by the Alaska Department of Fish and Game through the hunt permit registration process.

RESEARCH RECOMMENDATIONS

It has been suggested that all areas used by mountain goats in winter (except for travel lanes) are those within 2,600 ft of escape terrain that provide adequate forage (Suring et al. 1987, Fox et al. 1989). This hypothesis needs verification. Additional information is needed on habitat selection by mountain goats at all times of the year to improve the identification of habitats essential to their survival. More information is also needed on the effects of land management activities on the structure of populations of mountain goats relative to their ability to persist over time. This requires that monitoring techniques be further refined to provide needed information on population dynamics.

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CONSERVATION OF FLYING SQUIRRELS IN SOUTHEAST Alaska

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POPULATION STATUS AND DISTRIBUTION

Northern flying squirrels (Glaucomys sabrinus) occur in forested regions throughout most of northern North America (Wells-Gosling and Heaney 1984). Twenty-five subspecies have been recognized throughout its range (Hall 1981). The Prince of Wales flying squirrel (G. s. griseifrons) was described from 2 specimens taken in 1927 near Lake Bay on the northeast part of the island (Howell 1934). Its distribution is limited to Prince of Wales Island. The Alaska coast flying squirrel (G. s. zaphaeus) was described from 6 specimens taken in 1903; the type specimen was taken from Helm Bay on Cleveland Peninsula (Osgood 1905). Specimens have also been collected from Bradfield Canal, Etolin Island, Wrangell Island, and the Nass River in British Columbia (Howell 1918, Cowan 1937). The Richardson flying squirrel (G. s. alpinus), common in western Canada, has been reported north of Juneau on the mainland (Manville and Young 1965).

During the 1920s and 1930s flying squirrels were considered to be scarce on Prince of Wales Island (Howell 1934). The presence of flying squirrels was subsequently verified through trapping in 1956 (McGregor 1958) and from 1977 to

1979 (Van Horne 1981, 1982). However, estimates of population size were not made.

PATTERNS OF HABITAT USE

There have not been any studies completed and published on the natural history of flying squirrels in southeast Alaska. However, there is a fairly rich literature on this subject from studies conducted in eastern North America (Wells-Gosling and Heaney 1984). Only the food habits of flying squirrels have received attention in the Pacific Northwest. Although it is difficult to project the preference of flying squirrels for specific habitats available in southeast Alaska from studies conducted elsewhere, it is possible to identify characteristics of habitat that appear important.

Denning Habitat

Flying squirrels are associated with old growth forests throughout their range (McKeever 1960, Weigl and Osgood 1974, Weigl 1978). One of the most important attributes of old growth forests for flying squirrels is the availability of den sites in natural tree cavities or in woodpecker excavations (Weigl and Osgood 1974). Several dens, cavities, or external nests are used by each flying squirrel, however, the number of cavities or nests required is not known (Cowan 1936, Carey 1991). Individual flying squirrels have used from 1 to 13 dens in interior Alaska and up to 7 nest sites in Oregon (Mowrey and Zasada 1984, Carey 1991). Snags containing nest cavities used by flying squirrels averaged 35 in. diameter-at-breast-height (d.b.h.); live conifers with nests averaged 49 in. d.b.h.

Flying squirrels also use nest sites outside of tree cavities. However, several authors have suggested that such nests do not provide adequate protection in northern latitudes with severe winters (Cowan 1936; Sollberger 1943; Muul 1968, 1974; Goertz et al. 1975). Mowrey and Zasada (1984) noted that as temperatures dropped flying squirrels in interior Alaska moved from cavities to external nests in witches' brooms (Arceuthobium spp.).

Foraging Habitat

Although many observations have been made of food items consumed by flying squirrels, their food habits are not well documented. Northern flying squirrels cannot be maintained on a diet of white spruce seeds, so this kind of food source is probably not important to squirrels in the wild (Brink and Dean 1966). There is a strong indication from studies in the west and northwest that fungi and lichens may be the major or only foods eaten by northern flying squirrels (Cowan 1936, McKeever 1960, Maser et al. 1978, Mowrey et al. 1981, McIntire and Carey 1989). These food items are commonly available only in old growth forests (Maser et al. 1978, Rochelle 1980)

Forests with a well developed shrub layer are preferred by flying squirrels (Jordan 1948, Sonenshine and Levy 1981). Flying squirrels actively avoided forest stands without a fairly dense shrub layer in those studies where this habitat characteristic was evaluated. There was speculation by the observers that shrubs provided protection from predators when the squirrels moved about on the ground while foraging.

HOME RANGE/TERRITORY

Home ranges of flying squirrels have been reported to range in size from 20 to 75 ac (Weigl and Osgood 1974, Mowrey and Zasada 1985). Carey (1991) reported an estimated home range of 1.9 ac (for a 10-day period). Home ranges of individual animals overlap (Mowrey and Zasada 1985). However, females have been reported to defend an entire home range while males do not exhibit any defense (Madden 1974).

POPULATION DENSITIES

Reported population densities range from 0.135/ac in interior Alaska to 4/ac in more favorable habitat further south (Jackson 1961, Mowrey and Zasada 1985). Carey (1989, 1991) reported densities of 0.2/ac in old growth and 0.08/ac in second growth forests on the Olympic Peninsula; 0.8/ac in old growth and 0.4/ac in second growth forests in southwestern Oregon; and 0.8/ac in both old growth and second growth forests in the Oregon Cascade Range.

MOVEMENTS/DISPERSAL

Foraging movements recorded for flying squirrels in interior Alaska ranged from 0.6 to 1.2 mi (Mowrey and Zasada 1985). Carey et al. (1991) reported the mean maximum distance moved between subsequent recaptures of flying squirrels was 325 ft. Information on the dispersal of young animals from their natal sites is not available.

VIABILITY/DISTRIBUTION CONCERNS

Carey (1991) has suggested that the limiting factors for flying squirrels are availability of food, presence of adequate shelter (i.e., cavities), and presence of predators. The habitat attributes related to these limiting factors are associated with old growth forests in southeast Alaska (i.e., snags, fungi and lichens, a well developed shrub layer, and a well developed overstory canopy). Hokkanen et al. (1982) reported a strong relationship between old, mature spruce-dominated forests and high populations of flying squirrels. These authors attributed a wide-spread population decline in flying squirrels to intensive forestry practices resulting in extensive second growth stands. The size and frequency of areas of unsuitable, open habitat appeared to have a direct effect on isolation of populations and their extinction. The only report of extensive use of second growth forests by flying squirrels was in an area where nest boxes were readily available (Goertz et al. 1975). Rosenberg and Raphael (1986) also reported a negative response of flying squirrels to fragmentation of forests.

Snags may be retained following timber harvest to maintain nest sites and the shrub layer develops rapidly following clearcutting, providing 2 aspects of the habitat preferred by flying squirrels. However, the squirrels' source of fungi and lichens as food would not be readily available (Rochelle 1980, Maser and Trappe 1984). Squirrels may also be more vulnerable to predation from avian predators without a protective forest canopy. Once the existing snags deteriorate it will also be many years before a second growth stand will develop suitable nest sites.

Viable populations of the Prince of Wales flying squirrel will probably be maintained in existing Wilderness areas, lands unsuitable for timber harvest, and other forested areas that will not be harvested (Fay and Sease 1985). However, extensive timber harvests without considerations for the habitat and dispersal needs of flying squirrels throughout southeast Alaska may result in extirpations throughout portions of its range, leaving gaps in its distribution.

CONSERVATION STRATEGY

An apparent method to maintain the distribution of flying squirrels throughout areas where extensive timber harvests are planned to occur (e.g., Prince of Wales Island) is to retain forested stands large enough to maintain small subpopulations of squirrels. It is also important to ensure that travel corridors exist so that movement of squirrels is possible between subpopulations (Mowrey and Zasada 1985).

A 75 ac area has been suggested as the patch size of old growth forest necessary to meet the needs of one flying squirrel (Mowrey and Zasada 1985). Since home ranges of flying squirrels may overlap somewhat, portions of the 75 ac patch will provide habitat for more than 1 squirrel. Rosenberg and Raphael (1986) reported that habitat patches larger than 120 ac were required before use by flying squirrels was ensured. A 1,000 ac patch of old growth forest with at least 8,000 bf per ac is assumed to provide habitat for 20 to 40 flying squirrels in southeast Alaska. At least 1 patch should be maintained in each major watershed (i.e., 10,000 ac) to ensure that the distribution of flying squirrels is maintained.

Relatively uninterrupted corridors of old growth forest between habitat patches may be required to ensure interchange of flying squirrels. Old growth forests in riparian areas and beach fringe would serve well as travel corridors. If such areas are not available, stringers of old growth forest should be maintained between habitat patches. Breaks in travel corridors should generally not exceed 65 ft to ensure that flying squirrels can glide across the openings (Mowrey and Zasada 1985). Openings in the travel corridors greater than 100 ft should contain large, scattered trees to provide launching and landing points for the flying squirrels.

MONITORING RECOMMENDATIONS

Since flying squirrels are sensitive to timber harvests, populations in areas where timber harvest is occurring should be monitored to ensure their distribution is maintained. Trapping with live traps is the most effective way to estimate numbers and trends of populations of flying squirrels (Carey et al. 1991). Trapping in spring or fall is recommended using 30 ft by 30 ft to 40 ft by 40 ft grids, 130- to 160-ft spacing with 2 traps (ground and tree) per station, and 2 3- or 4-night trapping periods separated by 3 nights (Carey et al. 1991:13).

RESEARCH RECOMMENDATIONS

The flying squirrels in southeast Alaska were described from very few specimens before the advent of modern analytical techniques. Their taxonomic status therefore, needs to be verified. The distribution of flying squirrels throughout southeast Alaska also needs to be documented. The habitat relationships of flying squirrels in southeast Alaska have not been established. Because of their apparent sensitivity to timber harvest in other areas, the effects of forest management activities on flying squirrels in southeast Alaska should also be determined. Mowrey and Zasada (1984) recommended that corridors for dispersal be incorporated into management plans for flying squirrels. However, it is not clear what constitutes a good corridor for flying squirrels and how they use corridors (a.B. Carey, U.S. For. Serv., pers. commun.). Research is needed to clarify this, and other, aspects of the conservation strategy recommended for flying squirrels in southeast Alaska.

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APPENDIX C: GLOSSARY[a]

Adfluvial - fish which ascend from freshwater lakes to breed in streams

Allowable Sale Quantity (ASQ) - the maximum quantity of timber that may be sold in each decade from suitable scheduled lands cover by a National Forest Land Management Plan

Anadromous - fish ascending from oceans to breed in freshwater

Blowdown - trees felled by high winds

Class I stream - streams with anadromous or adfluvial fish habitat

Core - a defined area that includes the center of activity of a pair including the nest site, if known

Corridor - a defined tract of land, usually linear, through which a species must travel to reach habitat suitable for reproduction and other life-sustaining needs

Deme - a local, genetic population

Demographics - characteristics of a population (e.g., size, density, birth rates, death rates)

Dispersal - the movement, usually 1 way, and on any time scale, of plants or animals from their point of origin to another location where they subsequently produce offspring

Dispersal distance - a straight-line distance that an individual travels from its birth place until it stops dispersing (assumed to be a breeding site) or dies

Environmental analysis (Ea) - a document prepared by a federal agency in which anticipated environmental effects of a planned course of action or development are evaluated

Environmental Impact Statement (EIS) - a document prepared by a federal agency in which anticipated environmental effects of a planned course of action or development are evaluated

Fragmentation - process of reducing size and connectivity of stands that comprise a forest

Geographic Information System (GIS) - a computerised database and mapping system

Habitat capability - capacity of a habitat to support an estimated number of a species

Habitat Conservation Area - a contiguous block of habitat to be managed and conserved for breeding pairs, connectivity, and distribution of species of concern

Home range - the area to which the activities of an animal are confined during a defined period of time

Tentatively suitable - commercial forest land that is producing or is capable of producing industrial timber harvests and: 1.) has not been legislatively or administratively withdrawn from production, 2.) harvest may occur without irreversible damage to watersheds, 3.) there is reasonable assurance of restocking 5 years after harvest, and 4.) responses to timber harvest can be adequately predicted.

Land Use Designation (LUD) II - lands under this designation are managed in a roadless state to retain their wildland character. Timber harvest on these lands is limited to salvage operations to protect other resources

Matrix - habitat remaining outside of Habitat Conservation areas

Metapopulation - a population comprised of a set of populations that are linked by migrants, allowing for recolonization of unoccupied habitat patches after local extinction events

Monitoring - a process of collecting information to evaluate whether or not objectives of a management plan are being realized

Old growth - a forest stand with moderate to high canopy closure; a multilayered, multispecies canopy dominated by large overstory trees; a high incidence of large trees with large, broken tops, and other indications of decadence; numerous large snags; and heavy accumulations of logs and other woody debris on the ground

Population - a collection of individuals that share a common gene pool

Population viability - probability that a population will persist for a specified period of time across it range despite normal fluctuations in population and environmental conditions

Rescue effect - periodic immigration of new individuals sufficient to maintain a population that might otherwise decline toward extinction

Species - a group of actually or potentially interbreeding populations that are reproductively isolated from other such groups

Stochastic - random, uncertain; involving a random variable

Subpopulation - a well-defined set of interacting individuals that comprise a proportion of a larger, interbreeding population

Subspecies - an aggregation of local populations of a species inhabiting a geographic subdivision of the range of a species, and differing taxonomically from other populations

Territory - the area an animal defends, usually during the breeding season, against intruders of its own species

Viability - ability of a population to maintain sufficient size so that it persists over time in spite of normal fluctuations in numbers; usually expressed as a probability of maintaining a specific population for a specified period

Wildlife Analysis Area (WAA) - a division of land developed by the Alaska Department of Fish and Game for analysis and management of wildlife populations

Windthrow - a tree or group of trees uprooted by the wind

[a]Sources include:

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