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REVIEW

Wildlife studies on the Tongass National Forest challenge essential assumptions of its wildlife conservation strategy

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Abstract

The Tongass National Forest in southeast Alaska, USA, includes the Alexander Archipelago and narrow North American mainland, comprising one of the largest remaining, largely pristine, coastal temperate rainforest in the world. Management of the Tongass has become increasingly challenging because of expectations of a conservation framework designed to maintain viable populations of native wildlife species while decades of extensive clearcut logging of old-growth forests has continued. We used the findings of multiple published studies conducted on the Tongass from 1998 to 2017 to examine 4 assumptions of its wildlife conservation strategy (WCS): forest planning assessments of wildlife viability were realistic, forest management and conservation policies are implemented at appropriate ecological scales, old-growth reserves are effective habitat conservation areas and ensure functional connectivity, and forest-wide standards and guidelines ensure sufficient habitat for sensitive species in managed landscapes. Several ecological field studies, population and spatial analyses and modeling, and statistical analyses revealed that wildlife viability assessments to evaluate forest plan alternatives underestimated the risk of extinction by only examining individual vulnerable species rather than considering joint probabilities across multiple species; the ecological scale of management and conservation policies do not adequately consider area-sensitive vulnerabilities of island communities as evidenced by the increasing risk of extirpation of island endemics whose populations have become isolated and reduced; old-growth reserves are unlikely to maintain viable populations of endemic small mammals in isolation or as functionally connected metapopulations; and a spatially explicit analysis of individual home ranges demonstrated that forest-wide standards and guidelines provide about half the breeding habitat needed by a federally listed endemic raptor, the Queen Charlotte goshawk *(Accipiter gentilis laingi)*, of which only half of that is secure. Thus, assertions that the WCS is properly functioning as designed are dubious because a comprehensive monitoring plan has not been implemented and vital underlying assumptions are not supported by available science. We recommend 3 forest management and conservation policy adjustments: limit size and location old-growth forest harvests, restore forests through intermediate stand management of second growth, and conduct a formal review of WCS elements and assumptions.

KEYWORDS

biodiversity, clearcut logging, ecological scale, island endemics, old-growth forests, reserve network, temperate rainforest, wildlife viability

The Tongass National Forest (Figure 1) is one of the largest, relatively pristine, temperate rainforests in the world (DellaSala et al. 2011, 2022) with 6.7 million ha distributed across ≥20,000 islands and a narrow mainland in southeast Alaska, USA (Everest et al. 1997). It extends from the southern tip of Prince of Wales Island 800 km north to the Hubbard Glacier.

Southeast Alaska is globally recognized for its expansive tracts of intact rainforest that contribute to climate stabilization (DellaSala et al. 2022). Wind disturbance plays a fundamental role in shaping forest dynamics, at large and small scales, and over a continuum dependent on landscape features including exposure, landscape position, and topography. These forests support complete wildlife communities, most notably all-inclusive trophic assemblages that include primary producers to top carnivores (Vynne et al. 2021). The Alexander Archipelago has a terrestrial mammalian fauna with a nested structure that resulted primarily from differential colonization following glacial retreat (Conroy et al. 1999, Sawyer et al. 2019). Regardless of the primary mechanism, habitat loss and fragmentation are expected to reduce diversity of mammalian taxa in southeast Alaska through increasing extinction probabilities (Burkey 1995, Frankham 1998, Crooks et al. 2017, Püttker et al. 2020, Vynne et al. 2021). Furthermore, vital interspecific interactions across ecological communities are altered if a predator, prey, or symbiote is extirpated (Smith 2012*a*, Brodie et al. 2018, Kelt et al. 2019).

In 1997, Tongass planners were commissioned to manage wildlife habitats to maintain viable, widely distributed populations of existing native and desired non-native vertebrate species as directed by the 1982 viability rule of the 1976 National Forest Management Act (NFMA; U.S. Forest Service [USFS] 1982). Procedures

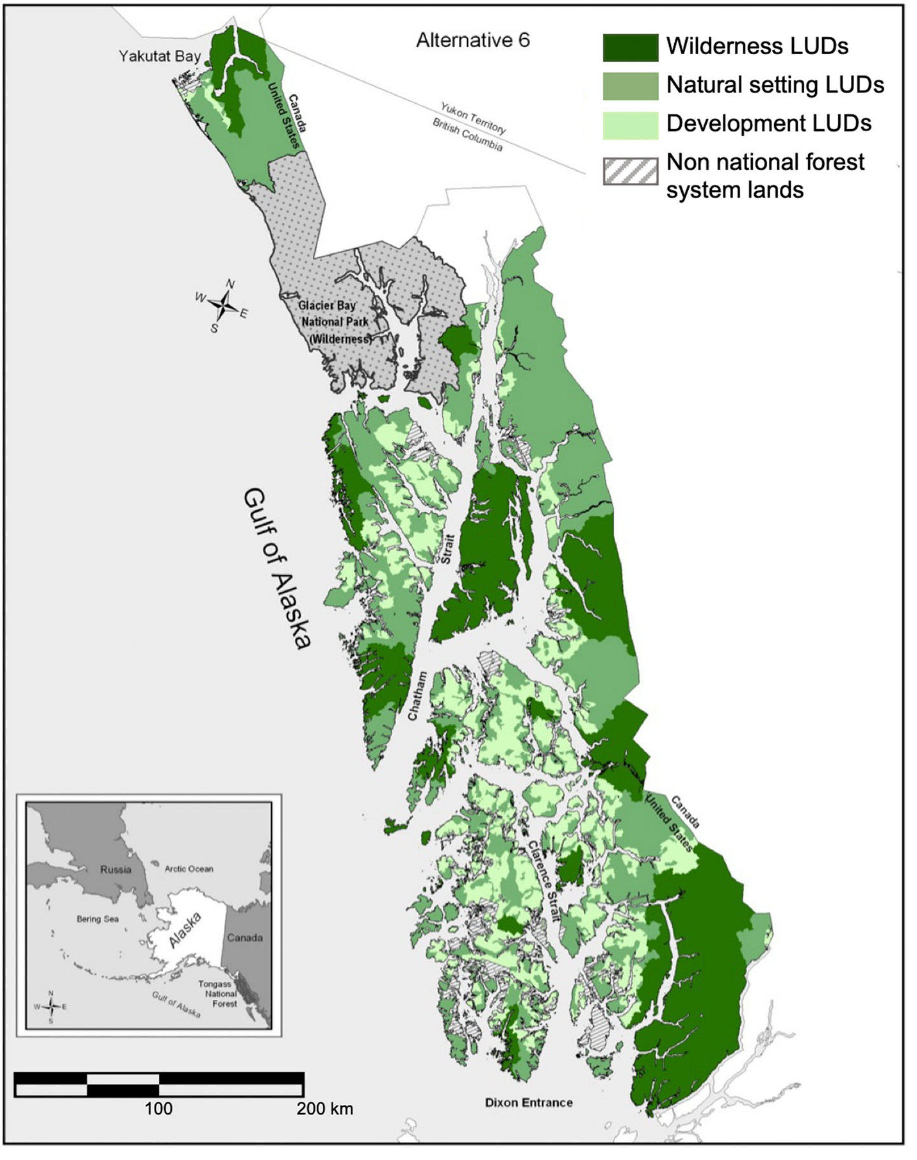

FIGURE 1 The Tongass National Forest in southeast Alaska, USA, extends from the southern tip of Prince of Wales Island 800 km north to the Hubbard Glacier and includes the Alexander Archipelago and a narrow strip of North American mainland encompassing 6.7 million ha. Single ranger districts exceed the size of many national forests in the continental United States. The Wildlife Conservation Strategy old-growth reserve network is depicted as wilderness and natural setting land use designations (LUDs) within Alternative 6 of the 2008 Forest Plan Amendment (USFS 2008). Wilderness LUDs include wilderness areas and National Monuments. Natural setting LUDs includes lands that maintain old-growth forest: congressionally designated unroaded areas; old-growth forest LUDs; remote and semi-remote recreation; municipal watersheds; special interest areas; wild, scenic, and recreational rivers; and research natural areas. Development LUDs include timber production, modified landscapes, scenic viewsheds, and experimental forests; <25% of these lands are suitable for timber harvest. Non national forest system lands represent state, Native, and private lands (USFS 2008).

for implementing NFMA viability provisions are expected to occur through the following processes: 1) describing the ecological context, 2) identifying species of viability concern, 3) collecting information on species of viability concern, 4) identifying species groups, 5) describing conservation approaches, 6) developing land and resource management plan (LRMP) alternatives, 7) evaluating the effects of LRMP alternatives on viability, and 8) conducting monitoring activities.

Historical timber management of the Tongass limited old-growth rainforest available to planners in framing a conservation strategy. A large majority of timber harvests occurred before the 1997 forest plan revision (USFS 2008), with cumulative disturbance and ecological consequences from 5 decades of high grading (i.e., exclusive harvest of the most valuable forests) across the region, including large-tree stands and expansive landscapes with contiguous productive old-growth (timber volume >46.6 m^3/ha) forests (Albert and Schoen 2012). While approximately 79% of the Tongass remains largely undisturbed and undeveloped (stream and shoreline buffers, reserves, wilderness areas), the majority of the unmanaged portion is highly fragmented, composed of ≥20,000 small (<400 ha), uninhabited islands with little opportunity for timber harvest (USFS 1997). The managed portion (~21%) has been subjected to intensive broad-scale disturbance from extensive clearcut logging that produced sharply contrasting land cover types within single landscapes (Figure 2). The highest rates of change occurred among biogeographic provinces and landform associations that originally contained the largest concentrations of highest volume, productive old growth (POG). Although only 12% of POG forests have been logged, large-tree stands were reduced by ≥28%, karst forests by 37%, and landscapes with the highest volume of contiguous old growth by 66.5% (Albert and Schoen 2012).

The USFS responded to this management challenge with a comprehensive, science-based revision of the forest plan (Swanston et al. 1996). The 1997 Tongass Land and Resource Management Plan (TLMP; USFS 1997) combined familiar, previously used elements and processes gleaned from the scientific literature with regional ecological information from journal publications, workshops, expert panels, and agency reports to design a unique, strategic conservation framework (Swanston et al. 1996, Everest et al. 1997, Smith and Person 2007, Smith et al. 2011, Smith 2013). Emulating natural disturbances offered an approach to designing management plans that maintain prevailing ecological conditions (Nowacki and Kramer 1998). Tongass planners chose a management plan alternative that departed substantially from the natural disturbance regime in which ≥95% of canopy openings produced by windstorms average <0.03 ha (Nowacki and Kramer 1998). The 1997 TLMP continued to emphasize

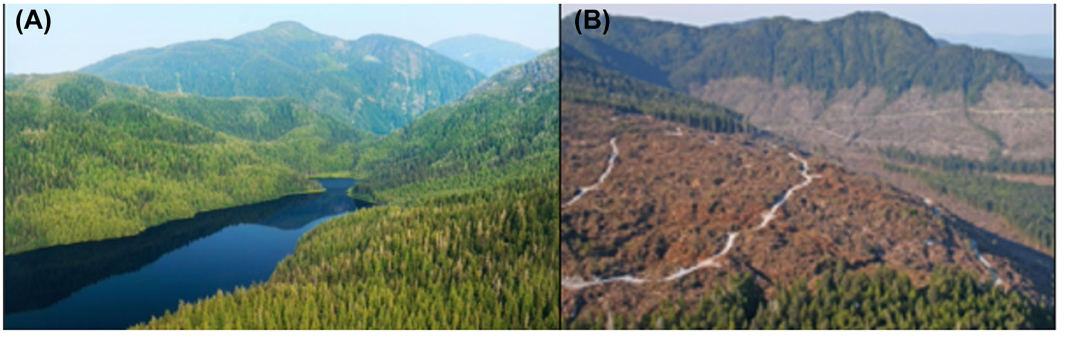

FIGURE 2 Timber management of the Tongass National Forest in southeast Alaska, USA, was dichotomous, producing sharply contrasting unmanaged and managed landscapes of A) intact old-growth temperate rainforests (photo by Alan Wu/Flickr Creative Commons) and B) extensive clearcuts (photo by John Schoen). A small proportion (12%) of the entire Tongass National Forest was logged; still, 67% of the highest volume forests were harvested (Albert and Schoen 2012). Some smaller islands experienced broad-scale disturbance across a significant proportion of the entire distribution of resident endemic mammals (Smith et al. 2011, Smith and Fox 2017).

clearcut logging, adding broad-scale disturbance to expansive landscapes of young, unmanaged, even-aged second growth (USFS 1997).

Despite its size and southeast Alaska's highly fragmented island biogeography, an important and far-reaching central hallmark of the 1997 TLMP is that the Tongass National Forest is largely managed as a single contiguous forest ecosystem. This has been evident from expectations that the Tongass would continue to persist as interconnected late-successional ecosystems across southeast Alaska (Shaw 1999), assessments and summary data would be presented as forest-wide statistics (USFS 1997, 2008, 2016), and management would disregard the variety and significance of unique biota and ecological communities (Smith 2012*a*), including the consequences of disproportionate loss and fragmentation of available habitat from cumulative effects of logging to island endemics (Smith and Person 2007, Smith et al. 2011, Albert and Schoen 2012, Smith 2013, Smith and Fox 2017).

Although amendments to TLMP have occurred since 1997 (USFS 2008, 2016), today the Tongass Wildlife Conservation Strategy (WCS) remains largely intact (USFS 2016). Unfortunately, a continuous decline in funding since 1997 has limited resources and capability to implement a proposed comprehensive long-term monitoring plan for sensitive or other vulnerable species (Smith et al. 2011, Smith 2013, Smith and Fox 2017). Consequently, there is little documentation regarding the implementation of management or conservation actions or corresponding responses and outcomes for intended forest resources.

Until recently, the most apparent alterations or amendments to the WCS have been the removal, movement, or change in composition of designated old-growth reserves (Smith et al. 2011). Small old-growth reserves have remained spatially inexplicit (USFS 2008), obscuring assessments of landscape structure and functional connectivity. In the 2016 forest plan amendment, an immeasurable number of small (many isolated) conservation areas totaling 1.3 million ha of old-growth forest were established across the Tongass, and 8,350 ha of old-growth forest was preserved among 73 watersheds to protect anadromous streams (USFS 2016: appendix D). It remains unclear if the watershed allocation was in addition to the old-growth reserve network expectation that ≥16% of the area be retained as old-growth forest (USFS 1997: appendix K). Regardless, without additional research, including spatially explicit analyses of watersheds and surrounding landscapes, contributions to functional connectivity or essential habitat of sensitive or old-growth obligate wildlife remain uncertain.

The Tongass WCS embodies a complex land and resource management framework intended to maintain biological diversity that comprises numerous elements, some of which are integrated within a hierarchical structure that is susceptible to systemic failure because each component is essential for overall functioning (Smith et al. 2011, Smith 2013). Although some elements have been successfully implemented for select species under specific circumstances elsewhere (USFS and Bureau of Land Management 1994), the Tongass WCS was implemented as an experiment with several essential underlying assumptions (Smith and Person 2007, Smith et al. 2011, Smith 2013). Yet statements asserting or accepting the 1997 Tongass WCS as a scientifically sound foundation from which to base management decisions continued to occur in forest and project planning documents (USFS 2008: D26, USFS 2016: K1–3), meetings, internal communications (W. P. Smith, USFS, personal communications), received emails, internal and cooperator meetings, oral comments during public workshops and meetings, and external communications (e.g., newspaper articles). Such assertions lack sufficient monitoring data or supporting evidence from scientific studies, potentially enabling further threats to essential old-growth ecosystems and native wildlife.

Our objective was to illustrate apparent disparities between designed expectations and documented outcomes of implementing the WCS through an investigation of published wildlife research conducted on the Tongass, including studies explicitly designed to examine inherent assumptions. Specific objectives were to use published results to examine 4 assumptions of the Tongass WCS: forest planning assessments of wildlife viability were realistic, forest management and conservation policies are implemented at appropriate ecological scales, old-growth reserves are effective habitat conservation areas and ensure functional connectivity, and standards and guidelines ensure sufficient habitat for sensitive species in managed landscapes.

STUDY AREA

Southeast Alaska is composed of mainland extending south along the western coast of Canada and >1,000 islands. The area includes fjords and glaciated mountain ranges and a cool, wet (200–600 cm annual precipitation) maritime climate, with mean monthly temperatures ranging from 13°C in July to 1°C in January (Smith and Nichols 2003, 2004). Elevation ranges from sea level to 1,643 m. The region is highly fragmented, with islands ranging in size from <1 ha to 6,670 km^2. The narrow mainland is largely isolated from other large, contiguous landmasses because of mountains, glaciers, and ice fields immediately to the east (Everest et al. 1997). Southeast Alaska is further stratified by 21 biogeographic provinces according to various configurations of physical, climatic, and biotic features. About 4 million ha (60%) is rainforest, of which 2.2 million ha is productive forests (USFS 1997). Coniferous rainforest dominates the landscape from shoreline to about 600-m elevation. The forest canopy is dominated by western hemlock (*Tsuga heterophylla*) and Sitka spruce (*Picea sitchensis*) in uplands but includes shore pine (*Pinus contorta*), mountain hemlock (*Tsuga mertensiana*), western redcedar (*Thuja plicata*), and Alaska-cedar (*Chamaecyparis nootkatensis*) in wetlands (Nowacki and Kramer 1998); remaining areas are riparian, alpine, muskeg, or sparsely vegetated mountain peaks and other rock formations (Smith and Nichols 2003, 2004). About 90% of commercial forests are upland Sitka spruce–western hemlock forests (USFS 1997).

Large trees (>75-cm diameter), downed and decaying wood, snags, and heterogeneous substrates are key components of old-growth rainforest ecosystems (Alaback 1982, Nowacki and Kramer 1998). The understory is dominated by blueberry (*Vaccinium* spp.), especially in canopy gaps (Smith and Nichols 2003, 2004; Smith 2012*a*). Unmanaged forests have a multilayered overstory of uneven-aged trees, dominant trees that generally are ≥300 years old, and structurally diverse understories (Alaback 1982; Smith and Nichols 2003, 2004). These forests vary in structure from scrub, or low-volume, communities of short (<10 m), small (<0.5-m diameter) trees with open canopies and dense, shrubby understories on poorly drained sites (peatland) to high-volume stands with a closed canopy, tall (>60 m), large (>3-m diameter) trees, and a predominantly herbaceous understory on highly productive sites (Harris and Farr 1974, Alaback 1982). The western hemlock-Sitka spruce forest type constitutes most of the closed-canopy forests in the region (Alaback 1982). It is spatially heterogeneous at a fine scale (<1 ha) and typically occurs on low-elevation, well-drained sites, often as a mosaic with fens and muskegs (Smith and Nichols 2003, 2004).

Southeast Alaska has a unique mammalian fauna that is significantly correlated with island isolation and extinction events resulting from differential colonization and island area effects (Conroy et al. 1999). Consequently, southeast Alaska is a hot spot of endemism (Cook et al. 2001), with varying and unique mammal assemblages and ecological communities (Cook and MacDonald 2001; Cook et al. 2001, 2006; Smith 2012*a*). The life history of birds is also influenced by the fragmented nature of the region, most notably northern goshawks (*Accipiter gentilis*) and other species that require large breeding ranges (Smith 2013). Prominent indigenous vertebrates include the Queen Charlotte goshawk (*A. g. laingi*), marbled murrelet (*Brachyramphus marmoratus*), Alexander Archipelago wolf (*Canis lupus ligoni*), brown bear (*Ursus arctos*), American marten (*Martes americana*), Sitka black-tailed deer (*Odocoileus hemionus sitkensis*), and numerous endemic small mammals whose distributions are restricted (MacDonald and Cook 1996, Smith 2005, Cook et al. 2006), including the following island endemics: Prince of Wales Island flying squirrel (*Glaucomys sabrinus griseifrons*), Wrangell Island vole (*Myodes gapperi wrangeli*), and Suemez Island ermine (*Mustela erminea seclusa*).

METHODS

Because the purpose of this review was not a meta-analysis or systemic literature review, we first conducted literature searches focused on our published research papers and the citations within and summarized the findings of numerous studies conducted in Southeast Alaska during 1998–2017. We then used a Web of Science™ word search conducted 31 May 2021 using the keywords Tongass, management, wildlife, and conservation. This search produced 57 results. We expanded our search by reviewing the publications citing relevant sources from this search and by reviewing the sources cited within all of these. We extracted publications from the search output that focused on research or management of wildlife within the Tongass that related to ≥1 of the 4 objectives for this review. We excluded studies of wildlife species that were not identified as a focal or management species within TLMP (e.g., bears, bald eagle [*Haliaeetus leucocephalus*]).

Viability assessments (objective 1)

An initial, integral step in developing a forest plan that prioritizes maintaining biological diversity is establishing a procedure in which planners can objectively evaluate the effect of LMRP alternatives on the persistence of native wildlife (Shaw 1999). The USFS convened numerous risk assessment panels during the 1997 TLMP revision (USFS 1997). Each panel comprised subject matter experts with knowledge of the natural resource under consideration. Seven panels estimated the relative risk that implementation of a range of alternative approaches to management of the Tongass would impose upon continued persistence of select wildlife species across the landscape. Ostensibly, the chosen set of species represented a broad enough range of taxa and ecological lifestyles that the breadth of possible responses to each of the 10 forest plan alternatives under consideration was captured in responses of vulnerable species but with little or no correlation among species' responses to a particular alternative (Shaw 1999).

An eighth panel evaluated old-growth ecosystems, assigning likelihood scores to outcomes that characterized the persistence of interconnected and representative late-successional ecosystems across southeast Alaska according to 3 attributes: abundance and ecological diversity, which considered if "old growth would be equal to or greater than long-term (i.e., 100 years) average and is well distributed across environmental gradients, provinces, and community types;" processes and functions, which considered whether the full range of disturbance processes are represented, and if "stand structure-dynamics and landscape structure-dynamics-age attributes occur across all provinces;" and whether connectivity would be as "effective as it was prior to large-scale timber harvest" (Shaw 1999: 11–12). Assessments from each of the panels became an integral element of the effects analyses that ultimately determined the management policies and actions incorporated in the 1997 TLMP (USFS 1997).

Risk assessment panels evaluated the likelihood of persistence of the northern goshawk, Alexander Archipelago wolf, brown bear, marbled murrelet, American marten, Sitka black-tailed deer, and other terrestrial mammals (Shaw 1999). Other terrestrial mammals included a group of more widely distributed mammals and a group of endemic small mammals whose known distribution in southeast Alaska is restricted (MacDonald and Cook 1996, Smith 2005, Cook et al. 2006). To assess the influence of varying management applications and intensities on wildlife viability, panels examined the marginal risk (individual extinction probability) of vulnerable species under each alternative and focused attention on the taxon with the highest projected risk of extinction with implementation of the alternative. This most sensitive species and its probability of extinction was used to reflect the risk to wildlife population viability for all vertebrates across the planning area for the alternative under consideration (Shaw 1999). This approach has been challenged because of untenable assumptions regarding the interpretation and application of select, individual vulnerable species from viability assessments to conservation planning (Soulé 1987, Smith and Zollner 2005, Jenkins et al. 2021).

Number of species influences the probability of any extinction

Smith and Zollner (2005) detailed an approach to assessing wildlife viability that explicitly considers the risk of any extinction among vulnerable vertebrate species in the planning area, calculated as the "likelihood of at least one success" (Snedecor and Cochran 1980: 115). Thus, the assessed probability of extinction following implementation of an alternative is the joint probability of marginal probabilities, each of which represents the risk to viability of individual species (Smith and Zollner 2005). They created a scenario with multiple hypothetical species at risk using each's corresponding independent marginal probabilities (acknowledging distinctive rather than correlated responses) for each of several management alternatives, similar to procedures used in planning TLMP (Shaw 1999). Smith and Zollner (2005) used this scenario to illustrate how the probability of any extinctions and the probability of the single most likely extinction differ as a function of the number of species examined. Recall, one panel assessed viability risks of a group of 26 terrestrial mammals (Shaw 1999) that included 14 endemic small mammal taxa and 12 additional terrestrial mammal taxa (USFS 2008: appendix D68). Moreover, endemic mammals were the most vulnerable of all wildlife species to future landscape disturbances assessed by the panel (Swanston et al. 1996: 11).

When marginal probabilities are used to calculate the joint probability of any extinction under each management alternative, the risk to wildlife viability is consistently and markedly higher than that obtained from selecting the most vulnerable species at risk (Figure 3). This occurs because the risk of local extirpation increases with the number of extinction-prone species in a region (Smith and Zollner 2005). Furthermore, the 1982 NFMA planning rule to ensure wildlife viability explicitly charges managers with the responsibility of protecting all vertebrates in a planning area, not just selecting species that appear to be the most vulnerable (Shaw 1999, Smith and Zollner 2005). Forest plan alternatives that pose the highest risk to more vulnerable species might not necessarily represent the greatest threat to wildlife communities, in part because of the variety of unique assemblages and their interspecific relationships and dependencies (Conroy et al. 1999, Smith 2012*a*, Smith and Fox 2017, Kelt et al. 2019, Jenkins et al. 2021) but also because of the varying number of sensitive species that occur among southeast Alaska's unique fragmented communities (Conroy et al. 1999, Smith 2012*a*, Colella et al. 2021).

Tongass planners in 2008 did acknowledge the influence of the number of species at risk on the probability of any extinction and compared 1997 assessment panel results among management alternatives with corresponding joint probability estimates and among proposed forest plan alternatives (USFS 2008: D85–86).

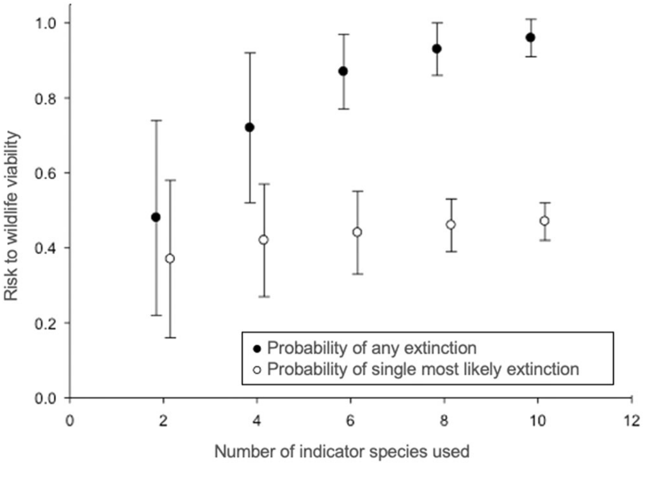

FIGURE 3 Disparity in risk to wildlife viability between estimating the probability of the most likely extinction (extinction of most sensitive species) and the probability of any extinction when assessing the relative risk that implementation of alternative approaches to management of national forest would impose upon continued persistence of indigenous wildlife across the landscape. Figure reprinted with permission from Smith and Zollner (2005).

Still, it remains unclear whether the joint probability calculations used the mean marginal probability of each wildlife assessment panel ($n = 7$) or included the marginal probabilities of each of the 14 individual endemic small-mammal taxa and 12 other terrestrial mammals. The latter seems unlikely because marginal probabilities were not assessed for each of the 26 terrestrial mammals (Shaw 1999). More importantly, the acknowledged higher extinction risks did not appear to raise concerns sufficient to generate discussion about acceptable threshold values of viability risks relative to policy or implementing future management actions or conservation measures (USFS 2008).

Implicit in the Tongass approach is an assumption that similar species are perfectly correlated in how each responds to a management alternative (Shaw 1999). Such an assumption is not ecologically tenable (Szaro 1986, Soulé 1987, Todd and Burgman 1998, Jenkins et al. 2021). Vertebrate faunas comprise diverse ecological assemblages of organisms that include herbivores, granivores, insectivores, carnivores, and omnivores, which often use the environment at different scales and in different ways (Wiens et al. 1993, Lancaster 1996, Kelt et al. 2019). Species in wildlife communities, even those with seemingly similar habitat affinities and life histories, do not respond to disturbance uniformly within habitat patches (Szaro 1986, Laurance 1991, Niemi et al. 1997) or across broader spatial scales (McGarigal and McComb 1995). Consider also the nature of forces that influence wildlife populations; anthropogenic disturbances are additive and extraneous to ecosystems (Püttker et al. 2020). Because wildlife communities evolved under unique environmental circumstances, local populations respond differently to anthropogenic disturbances compared with natural regimes (Wilson et al. 2005). Individual species likely respond differently to the same anthropogenic disturbance (Szaro 1986, Wiens et al. 1993, McGarigal and McComb 1995, Niemi et al. 1997). Within the same community, species of management concern can require strikingly different disturbance regimes and consequently respond divergently to the same management prescription (Smith 2013). Thus, it is unrealistic to expect that a select set of vulnerable species can represent the risk to viability of the entire vertebrate fauna across numerous unique, diverse communities with varying life histories and sensitivities to anthropogenic disturbance and spatial context (Wiens 1989, 1996).

Managing a large-scale island archipelago (objective 2)

Issues of ecological scale are a major concern in wildlife conservation, and many ecological patterns and processes are scale-dependent (Wiens 1989). Perceptions of how populations are spatially subdivided and impressions of extinction and dispersal dynamics depend on the scale at which the population is viewed (Wiens 1996). Conservation of biological diversity also requires maintaining evolutionary diversity (genetic and life-history attributes) of organisms indigenous to a region (Cook and MacDonald 2001, Colella et al. 2021), including the composition, structure, and functions of local ecological communities (Smith 2005, Watson et al. 2018, Grantham et al. 2020). Land management planning for the Tongass National Forest, however, has occurred at the scale of millions of hectares (USFS 1997, 2008, 2016), which is a much broader scale than the contiguous landscapes available to fragmented wildlife populations and ecological communities across an island archipelago and isolated mainland (MacDonald and Cook 1996, Conroy et al. 1999, Cook et al. 2001). Cook et al. (2001) listed 24 endemic mammals, several of which occur only on one or a few islands (MacDonald and Cook 1996, Smith 2005, Cook et al. 2006, Colella et al. 2021). The entire known distribution of the Suemez Island ermine, a small carnivore, is $<160\ km^2$ (MacDonald and Cook 1996). Moreover, several species encompass multiple, genetically distinct lineages (some representing incipient or new species) attributable to independent colonization histories from divergent source populations (Cook et al. 2006). The insular landscapes of the Alexander Archipelago have produced highly endemic populations that should be prudently managed as hotspots of biological and evolutionary diversity. Thus, islands available for timber harvest should each initially be considered an independent biological unit (Cook et al. 2006).

Consequences of management at inappropriate ecological scales among island communities

The Wrangell Island vole is a habitat specialist that achieves its highest densities in old-growth forests (Figure 4A) and is unable to sustain breeding populations in peatland scrub (mixed-conifer) forest (Figure 4B), clearcuts (Figure 4C), or second growth (Figure 4D,E; Smith and Nichols 2004, Smith et al. 2005*a*, Smith and Fox 2017). This red-backed vole is known only from Wrangell and Etolin islands (MacDonald and Cook 1996, Runck 2001). Wrangell Island is 544 km^2 (54,400 ha) and 85% of the island is in Tongass National Forest, of which 72% is available for timber harvest. Approximately 2,700 ha of old-growth forest has been clearcut logged, with a proposed timber project to harvest an additional 16,600 ha (USFS 2019). Moreover, there are no explicit conservation directions or actions to protect this vulnerable island endemic from local extirpations (USFS 1997: 4–87).

On Wrangell Island, voles are sympatric with the Keen's mouse (*Peromyscus keeni macrorhinus*), a habitat generalist that flourishes in old-growth, managed, and scrub forests (Smith and Fox 2017). Keen's mouse can be an intense competitor of voles, with interspecific competition between the 2 species explaining more variation in vole abundance (and vice versa) among habitats than the variance associated within habitats (Smith and Fox 2017). Thus, clearcut logging of old-growth forests on Wrangell Island favors populations of the Keen's mouse by creating habitats that breeding vole populations cannot exploit and further reducing vole abundance across managed landscapes because of increased interspecific competition from increasing mice populations (Smith and Fox 2017). Furthermore, opportunities for voles to reoccupy managed landscapes are limited because broad-scale disturbance can take ≥300 years for ecological succession to achieve old-growth forest conditions (Nowacki and Kramer 1998).

Thus, when forest management is applied indiscriminately across archipeleagos (*defacto* contiguous landscapes), it is implemented at inappropriate ecological scales and thus insensitive to the variation and uniqueness of species composition, phylogeography, life-history attributes, and interspecific relationships among island communities (Cook et al. 2006). The consequences of disproportional habitat loss and fragmentation typical of island endemics results in isolation, local extirpation, and overall reduction of endemic populations, increasing risk of extinction (Burkey 1995, Frankham 1998, Crooks et al. 2017, Püttker et al. 2020, Vynne et al. 2021).

Effectiveness of old-growth reserve system (objective 3)

The WCS has 2 components, each representing sharply contrasting management and landscape conditions. The first is a forest-wide, old-growth reserve network (Figure 1; USFS 1997) in which the reserves and other protected lands are expected to provide sufficient habitat to sustain viable, well-distributed populations of old-growth-obligate wildlife (Iverson and Renè 1997, Smith and Person 2007). This network was intended to serve as a coarse filter to maintain a functional and interconnected old-growth ecosystem (USFS 2008: D6). Coarse filters use the compositional integrity and functional proficiency of landscapes or ecosystems as surrogates to predict or ensure the wellbeing of particular taxa or ecological communities (Jenkins et al. 2021). A second function of the old-growth reserve system is to facilitate functional connectivity of protected lands, which also contributes old-growth structural elements in the development land-use designations of the Tongass planning area within which timber harvest and other anthropogenic disturbances occur over time.

The old-growth reserve network included all non-development lands and a system of large, medium, and small habitat conservation areas (reserves). Islands <400 ha were included and received protection from additional logging (USFS 1997, 2016). Each major watershed is required to have at least a small reserve encompassing ≥16% of its area. The preferred biological objective of a small reserve is to contain ≥50% POG; the minimum prescription is ≥25% POG. Medium reserves were delineated as contiguous landscapes of ≥4000 ha with ≥2000 ha of POG, of which ≥50% must be in the highest volume category. Large reserves must be ≥16,000 ha of contiguous landscape, with ≥50% POG and ≥25% in the highest volume category (USFS 1997: appendix K).

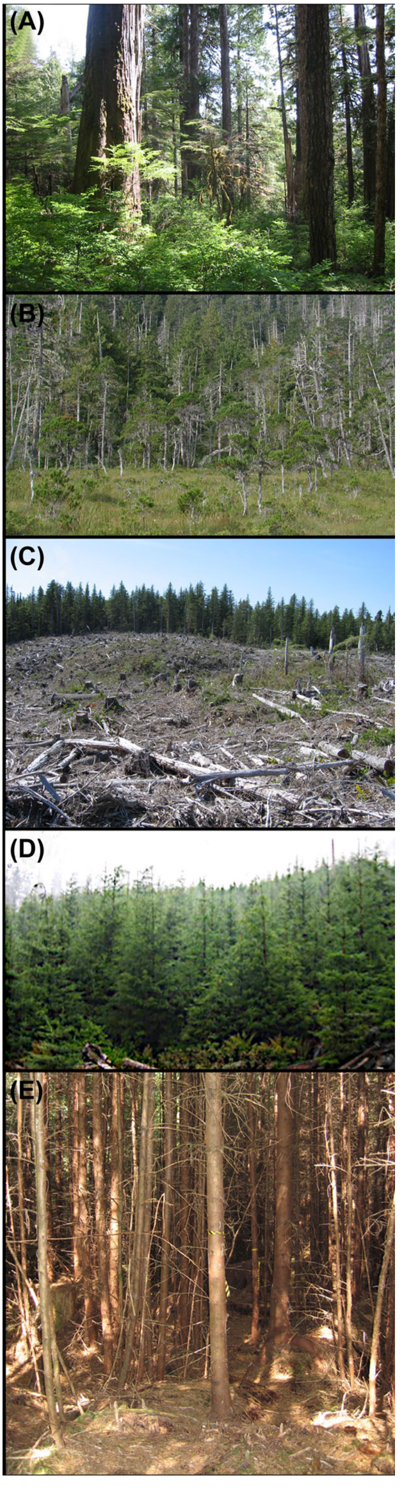

FIGURE 4 Common management stages and corresponding vegetation structure available to endemic small mammals in managed landscapes of the Tongass National Forest, southeast Alaska, USA: A) old-growth forest, B) peatland scrub forest, C) clearcut, D) young (<20 yr) second growth, and E) unthinned older (>40 yr) second growth.

The second WCS component includes active management of the matrix (commercial clearcut logging). Land managed for timber production was expected to contribute little toward maintaining biological diversity (USFS 1997: appendix N). Within the matrix, forest-wide standards and guidelines were implemented to uphold remaining components of the old-growth ecosystem. A standard is a course of action or level of achievement that must be accomplished to achieve forest goals and are mandatory. A guideline is also a course of action that must be followed, but guidelines relate to activities in which site-specific factors might require flexibility and require further analysis. Therefore, forest-wide standards and guidelines serve as fine filters to protect specific resources (e.g., old-growth forests) and functions (e.g., streamside buffers), facilitate connectivity across old-growth forests, and to ensure sufficient habitat for individual sensitive species (USFS 1997: chapter 4). Thus, for wildlife populations to persist in heterogeneous landscapes, either individual habitat patches must be large enough to provide for viable populations in isolation (Smith and Person 2007, Crooks et al. 2017) or the juxtaposition of suitable habitat within the matrix must allow for interpatch movements that facilitate meta-population dynamics (Smith 2012*b*, Fahrig et al. 2021).

Habitat conservation areas

The northern flying squirrel (*Glaucomys sabrinus*) was selected as the design (proxy) species for small old-growth reserves (≥650 ha) in the 1997 TLMP (USFS 1997) because of its assumed dependency on POG. Northern flying squirrels are k-selected, omnivorous, mature-forest obligates (Smith et al. 2004, 2005*b*; Smith 2007, 2012*b*; Holloway and Smith 2011) with specialized gliding locomotion (Scheibe et al. 2006). The underlying premise was that if the Tongass conservation strategy maintained viable and widely distributed populations of flying squirrels across the planning area, it would support other small mammals with similar life histories and habitat needs (Swanston et al. 1996, USFS 1997). Northern flying squirrels inhabit forests along southeast Alaska's mainland coast and occur on ≥15 islands of the Alexander Archipelagos (MacDonald and Cook 1996, Smith 2005, Schoen et al. 2006). The Prince of Wales Island flying squirrel (*G. s. griseifrons*) is an island endemic with reduced genetic variation (Bidlack and Cook 2001) that is considered a subspecies of ecological concern in the Tongass National Forest (Schoen et al. 2006) and is listed by the International Union for the Conservation of Nature as potentially endangered (Hafner et al. 1998).

Small old-growth reserves were expected to function as habitat conservation areas that provide sufficient habitat to facilitate occupancy by flying squirrels, and functionally connected populations interspersed throughout the matrix would behave as a metapopulation (Fahrig and Merriam 1985, Fahrig et al. 2021). Although there was no design requirement to ensure physical connectivity among old-growth reserves or with other non-development land-use designations, the assumption was that POG retained through other features of the conservation strategy (larger old-growth reserves, standards and guidelines) will establish landscape connectivity to facilitate dispersal across the matrix (USFS 2008: D8).

Persistence in habitat conservation areas

Smith and Person (2007) examined whether flying squirrels are likely to persist in isolation over a range of time periods in small habitat conservation areas with varying compositions of old-growth spruce-hemlock and mixed-conifer forests (Figure 4A) consistent with forest plan guidelines for both preferred and minimum habitat objectives (USFS 1997: appendix K). Given these guidelines, Smith and Person (2007) models revealed that the probability of persistence over a planning horizon of 100 years in small habitat conservation areas with the preferred prescription (50%) of spruce-hemlock composition was 0.73–0.77 and for the minimum prescription (25%), probabilities were 0.66–0.71. Furthermore, to sustain isolated populations over long periods (100 yr) with a high level (≥0.95) of confidence, flying squirrels require very large (244,600 ha) reserves of 100% optimum habitat. Medium (2000 ha POG) and large reserves (8000 ha POG) as currently specified (USFS 1997: appendix K) have a <0.90 probability of sustaining viable populations of flying squirrels over the 100-year planning horizon (Smith and Person 2007). These persistence estimates have been evaluated in the field. For example, on Kosciusko Island, flying squirrels were apparently extirpated from a 50-ha remnant patch of old-growth forest surrounded by <50-year-old second growth (E. A. Flaherty, Purdue University, unpublished data), and Shanley et al. (2013) observed that flying squirrels were not found in patches <29 ha and only selected the largest fragments locally and at the landscape scale with the minimum patch size for occupancy of 48 ha. Both suggest that likelihood of persistence is low in these small, isolated patches.

Functional connectivity and dispersal in managed landscapes

Given this uncertainty, Smith et al. (2011) evaluated the efficacy of small reserves as a functionally connected network that provided temporary suitable habitat for flying squirrels dispersing among large and medium reserves. They estimated the number of immigrants required to persist in small reserves for 25 and 100 years, landscape resistance to movement, and maximum effective dispersal distance via least-cost path analysis among small and larger reserves to ensure the required number of immigrants (Pyare and Smith 2005). Landscape resistance and risk of predation were higher in clearcuts (Figure 4C) than mature forests (Smith 2012*b*). Similarly, unthinned second growth (Figure 4E) obstructed visibility of suitable habitat (perceptual range) and impeded gliding (Flaherty et al. 2008, Smith 2012*b*). These dispersal barriers are a significant concern when an estimated 162 dispersers/year are needed to sustain populations for 100 years in small reserves comprising 25% primary habitat and ≥6 juvenile dispersers/year are needed to achieve a 0.95 probability that a breeding pair would reach a patch in which flying squirrels were recently extirpated (Smith et al. 2011).

Considerations of dispersal distance across managed matrix habitat is also important for maintaining persistence of flying squirrels. The maximum effective dispersal distance (Pyare and Smith 2005) for a 0.95 probability persistence over 100 years ranged from 844 m for small old-growth reserves with 25% primary habitat to 1,151 m for small old-growth reserves that comprise 100% primary habitat. Corresponding values for persistence in small old-growth reserves over 25 years were 1,172 m and 1,174 m (Smith et al. 2011). Remarkably, the maximum value of 1,174 m fell well within the distance that juveniles can move through intact landscapes (~7 km) over short time periods (Smith 2012*b*). Unfortunately, most of northern Prince of Wales Island has been clearcut logged (Figure 5), and ≥50% of small old-growth reserves prescribed in the 1997 TLMP for northern Prince of Wales (Figure 5) were isolated and not functionally connected to a source population (Smith et al. 2011).

These results underscore the vital role of immigration in rescuing sinks or facilitating metapopulation viability of northern flying squirrels among unsustainable fragmented populations, and the extent to which permeability of landscape elements can influence dispersal and functional connectivity of subpopulations in a managed matrix (With and Crist 1995, Richards et al. 2002, Pyare and Smith 2005, Smith et al. 2011, Trapp et al. 2019). The expectation that the Prince of Wales Island flying squirrel will function as a metapopulation with successful dispersal among old-growth fragments or reserves in managed landscapes is not supported by the findings of multiple studies examining this island endemic's habitat relations, population dynamics, and dispersal capability, including perceptual limitations, locomotion, energetics, and diet (Smith 2012*b*). Without large trees to facilitate gliding (Vernes 2001), flying squirrels must use quadrupedal (walking or running) locomotion, which is energetically more expensive than gliding (Scheibe et al. 2006, Flaherty et al. 2010*a*) and increases travel time (Byrnes and Spence 2011), leading to increased risk of predation (Smith 2012*b*). Additionally, flying squirrels cannot replenish energy stores by foraging as they disperse across clearcuts and second-growth stands because of the absence of preferred food resources (Flaherty et al. 2010*b*, Price et al. 2017). Finally, flying squirrels are unable to perceive old-growth forests across managed stands and are therefore unlikely to initiate movements across these more energetically expensive and risky land cover types (Flaherty et al. 2008).

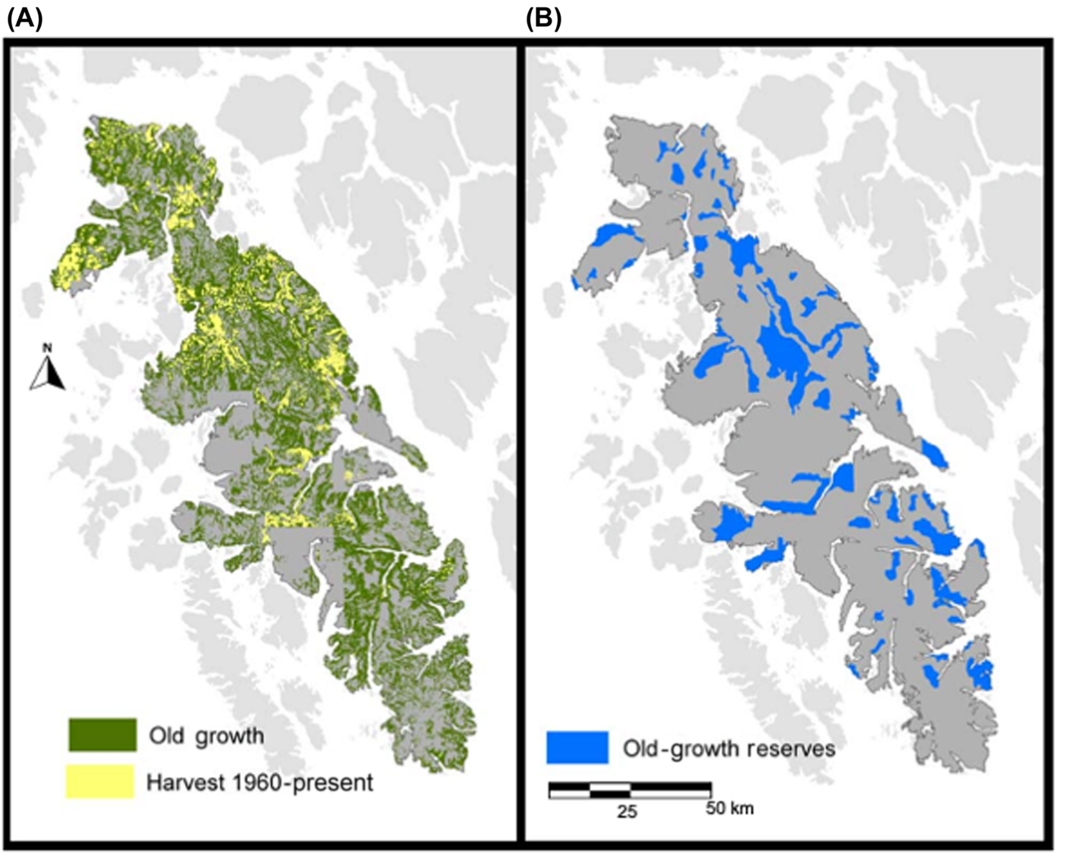

FIGURE 5 Prince of Wales Island, Alaska, USA, depicting the distribution of A) old-growth rainforest and areas logged since 1960 and B) old-growth reserves (USFS 1997).

Forest-wide standards and guidelines (objective 4)

The second component of the WCS uses forest-wide standards and guidelines, which are implemented for the protection or management of different forest resources (USFS 1997: chapter 4). Standards and guidelines apply to all or most areas of the Tongass, are organized by resource conservation status, and are used in conjunction with additional standards and guidelines included within each management prescription (USFS 1997: chapter 3). Standards and guidelines were established to manage locally important habitat for native wildlife (USFS 1997: chapter 4) and sensitive species (USFS 1997: 4–87), especially those that were not explicitly considered by viability assessment panels (Shaw 1999) or selected as ecological proxies in the design of the old-growth reserve network (Iverson and Renè 1997, USFS 1997, Smith 2013).

The northern goshawk was designated a sensitive species and underwent viability risk assessment (Shaw 1999). Goshawks received special consideration on the Tongass largely because of concerns over populations of the endemic Queen Charlotte goshawk (Iverson et al. 1996). Formally described as a metapopulation (Sonsthagen et al. 2012), the Queen Charlotte goshawk's distribution includes Prince of Wales and barrier islands and coastal British Columbia and nearby islands. The United States Fish and Wildlife Service listed all areas with known nests, except Prince of Wales Island, as threatened subpopulations in 2012 (U.S. Fish and Wildlife Service 2012), although all subpopulations are deemed essential for long-term viability (Sonsthagen et al. 2012) and ≥33% of POG on Prince of Wales Island has been converted to second growth (USFS 2008: appendix E; Albert and Schoen 2012). The most imminent threats to breeding populations are loss or fragmentation of nesting or foraging habitat from logging (Figure 2B) without ensuing intermediate stand management (Figure 4E), which eliminates nest trees and reduces prey diversity and availability (Reynolds et al. 1992, Finn et al. 2002, McGrath et al. 2003, Mahon and Doyle 2005, Northern Goshawk *Accipiter gentilis laingi* Recovery Team 2008).

In western North America, breeding home ranges of northern goshawks are spatially configured as a hierarchical sequence of 3 areas (Andersen et al. 2005), all of which need to be considered simultaneously in land use planning (Reynolds et al. 2006, Northern Goshawk *Accipiter gentilis laingi* Recovery Team 2008): nest area, post-fledging area, and foraging area. Nest areas provide alternate nest trees, roost trees, and prey plucking posts, and serve as centers of essential breeding behaviors or life-history events (Reynolds et al. 1992, 1994, 2006). Post-fledging areas surround active nest trees, average 800 ha in southeast Alaska (Iverson et al. 1996), and represent the core-use area of adult female and young goshawks after fledging but before becoming independent of adults and dispersing (Kenward 1982, Kenward et al. 1993, Kennedy et al. 1994). McClaren et al. (2005) suggested the biological role of post-fledging areas and nest areas are similar and to consider them as one functional component. Regardless, the habitat composition of post-fledging areas should be similar to nest areas (Reynolds et al. 2008). Foraging areas comprise the majority of northern goshawk breeding home ranges and are especially important for adults providing food to young and for juveniles prior to natal dispersal. Breeding home ranges in southeast Alaska average 21 km^2 (Iverson et al. 1996). The combined home range of breeding pairs can be much larger than that of individual birds (Boal et al. 2003).

The 1997 TLMP did not incorporate concepts of nest area, post-fledging area, and foraging area habitat management, which underpin conservation planning to sustain viable populations of northern goshawks across its distribution (Reynolds et al. 2006, Northern Goshawk *Accipiter gentilis laingi* Recovery Team 2008). Still, Tongass forest-wide policy is focused on protecting confirmed and probable goshawk nests (USFS 1997: chapter 4); standards and guidelines propose to accomplish this by maintaining an area of ≥40 ha of POG generally centered over the nest tree or probable nest site (Figure 6). Another stated objective is to manage foraging habitat to retain essential features of forest stand structure in areas of timber harvest (Figure 2B) because tree density of unmanaged second growth (Figure 4E) reduces prey abundance and diversity and prevents aerial pursuit of prey by goshawks (Reynolds 1983, Salafsky et al. 2007).

Despite a substantial increase in knowledge since the 1997 TLMP revision, the implications of those new insights to goshawk conservation and land-use policies in southeast Alaska had not been revised in forest plan amendments (Smith 2013). Without long-term monitoring, it has remained unclear whether a network of reserves designed explicitly for other wildlife species (USFS 1997) or protection of goshawk nest trees in landscapes intensively managed for timber, would provide sufficient habitat to sustain breeding populations of the northern goshawk across the planning area (Finn et al. 2002). What is clear from the literature is neither coarse-filter nor fine-filter components of the WCS appear relevant to northern goshawk life history or conservation planning; 40-ha nest buffers (Figure 6) and habitat conservation areas distributed across expansive landscapes of even-aged second growth have never been applied as mitigating measures elsewhere in its distribution (Smith 2013).

Smith (2013) conducted a spatially explicit analysis of contributions of the Tongass WCS to the breeding home ranges of northern goshawks across southeast Alaska. He used 136 confirmed nest-tree locations and empirically derived estimates (Iverson et al. 1996) to delineate corresponding virtual post-fledging areas and female breeding home ranges, within which they calculated the area of 4 cover types and 4 land-use categories. They derived preferred habitat from empirical studies in southeast Alaska (Iverson et al. 1996). About 30% of nests had >51% of post-fledging areas in preferred habitat but >91% of post-fledging area was in an unsecure (unprotected from development) land-use designation; 60% of post-fledging areas had >51% in an unsecure designation, whereas only 16% had >51% in the protected old-growth forest. Among cover types, preferred habitat comprised an average of 39.4% of the post-fledging area. Smith (2013) obtained similar results from an analysis of the female breeding home range but with notable differences. The percentage of the broader landscape that consisted predominantly (>75%) of lands available for development was greater than in post-fledging areas (Smith 2013). The percentage of the total

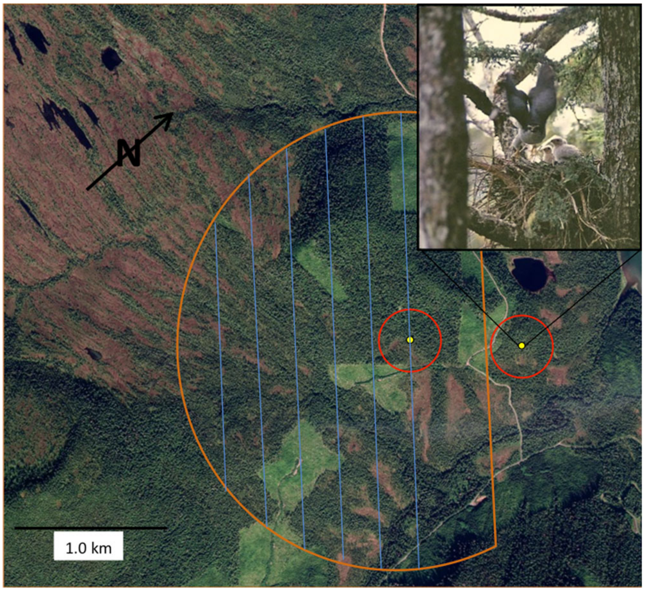

FIGURE 6 Northern goshawk nest sites (yellow spheres) during 1999 to 2001 in managed landscapes of the Tongass National Forest in southeast Alaska, USA (image courtesy of Google Earth), with an active nest (photo by Craig Flatten) in the canopy of old-growth rainforest. Red circles represent circular 40-ha old-growth buffers (360-m radius) prescribed for active goshawk nests by forest-wide standards and guidelines in the Tongass Land Management Plan (USFS 1997, 2008, 2016). Area with blue lines within the orange semi-circle depicts half the typical goshawk post-fledging area (PFA); the mean radius of goshawk PFAs is 1,600 m, whereas the radius of breeding female home ranges averages 2,600 m (Smith 2013). Light green areas along logging roads are recent clearcuts; light brown areas are muskegs.

home range with 26–50% of the total area in preferred habitat also increased compared with post-fledging areas, whereas about half as many home ranges had ≥51% of this broader landscape in preferred habitat as compared with the post-fledging area (Smith 2013). From these analyses, it is clear that the Tongass WCS is not contributing sufficient secure habitat to sustain breeding pairs of the northern goshawk across southeast Alaska.

RESULTS

Based on this review, we conclude that the Tongass Land Management Plan is not meeting expectations of ≥4 essential assumptions of the WCS. Additional empirical evidence from the literature supports a conclusion that the WCS has not met expectations of maintaining an interconnected old-growth forest ecosystem. Extensive high-grading and disproportional harvest of the most productive forest have substantially reduced old-growth forest abundance and diversity (Albert and Schoen 2012). Expansive even-aged clearcuts produced landscapes that support a fraction of the old-growth obligate species and provide little functional connectivity, isolating wildlife communities in many of the remnant old-growth patches (Smith et al. 2011).

The Tongass WCS was implemented as an experimental conservation plan composed of numerous elements, some of which are founded in sound ecological science and theory and were successfully implemented elsewhere with different wildlife species and circumstances. A systematic, comprehensive long-term monitoring scheme was proposed as a means to document implementation of management actions and conservation measures, and to record responses and outcomes of select forest resources (i.e., to evaluate if the WCS was functioning according to expectations). In the absence of monitoring data, we chose to use the results of wildlife studies on the Tongass that were designed to examine the robustness of vital underlying assumptions.

The enormity and complexity of the Tongass present unprecedented management and conservation challenges, most notably the highly fragmented and isolated nature of southeast Alaska. Empirical evidence from the literature provides examples of isolated ecological communities, varying in composition, ecological roles, and relationships among members, and the potentially irreversible consequences of cumulative broad-scale anthropogenic disturbances on old-growth obligate species, many of which are endemic. The Wrangell Island vole and Prince of Wales Island flying squirrel are examples of endemics for which a substantial part of their historical distribution has been clearcut logged, local populations have become extirpated or isolated, and total populations are reduced, all of which influence persistence. Given the proclivity for endemism, the discontinuity of landscapes further stratified among 21 biogeographic provinces, and the diversity of unique plant and animal assemblages with varied ecological functions and dependencies, it is unrealistic to expect that the Tongass can be managed as a single rainforest ecosystem or according to a conservation strategy that relies on isolated old-growth forest remnants scattered across vast landscapes of unmanaged, even-aged second growth (coarse filter) and uninformed, ineffective fine-filter mitigation measures.

The conceptual framework and procedures used by planners to assess the risk to viability of native wildlife underestimated the effects of implementing each of 10 forest plan alternatives across the planning area. Consequently, when forest management planning and implementation are considered in the context of widespread fragmentation, isolation and endemism, ecological scale, variation and complexity of ecological communities, and an incomplete monitoring plan with substantial gaps in data and analyses, serious questions arise about the effectiveness of the WCS in maintaining widely distributed, viable populations of native wildlife, especially old-growth obligate endemics.

A network of old-growth reserves functioning as habitat conservation areas across intensively managed landscapes can be effective in sustaining viable populations of sensitive, old-growth obligate species. Establishing small, medium, and large habitat conservation areas, each designed to sustain proxy species operating at appropriate ecological scales and collectively establishing functionally connected landscapes, is an empirically based coarse-filter approach. Nonetheless, demographic analysis revealed that the size of a habitat conservation area (with 100% POG) required to sustain viable northern flying squirrel populations in isolation over the planning horizon exceeds the size of medium and large old-growth reserves, the preferred prescriptions of which contain only 50% POG. Further analysis demonstrated that landscapes within the matrix were not functionally connected and incapable of facilitating demographic or genetic rescue among small-mammal endemics. Despite having comparably high densities, the viability risk of the Prince of Wales Island flying squirrel is higher today because subpopulations have become isolated, local extirpations have occurred, and the overall population is reduced. Furthermore, because the northern flying squirrel was selected as a proxy, the effects of cumulative habitat loss and functionally discontinuous landscapes have implications for other old-growth obligate small mammals, especially island endemics.

The WCS also includes forest-wide standards and guidelines as a fine-filter approach to retain, replace, or mitigate essential conditions, mostly in managed landscapes. Forest-wide standards and guidelines are essential for sensitive species such as the Queen Charlotte goshawk that require a diversity of land cover types, including mature or old-growth forest. Forest management guidelines throughout its distribution invariably prescribe rotational management of the entire planning area, which produces landscapes that are a mosaic of cover types varying in stand age, structure, and spatial extent, thereby supporting a wide range of potential avian and mammalian prey species. Landscapes across the Tongass are a sharply contrasting dichotomy of old growth and expanses of even-aged second growth, most of which were logged during a few decades with little (<20%) ensuing intermediate stand management. Unfortunately, neither the reserve network nor the prescribed standards and guidelines accomplish the objective of providing sufficient breeding habitat to sustain northern goshawks across the Tongass.

DISCUSSION

To address apparent deficiencies and meet expectations of the 1982 viability rule of the 1976 National Forest Management Act, we propose 3 revisions to forest management and conservation policies. First, further commercial harvests of old-growth forests should emulate the primary natural disturbance regime (wind) in size of canopy gaps, frequency of occurrence, and landscape conditions (e.g., forest stand composition and exposure, canopy structure) and circumstances (e.g., slope, aspect, wind severity and direction; Nowacki and Kramer 1998), which will prohibit commercial broad-scale clearcut logging. This policy will reduce further negative effects to old-growth obligate wildlife, especially island endemics (Cook et al. 2006, Smith and Person 2007, Smith et al. 2011, Smith and Fox 2017), and acknowledge the contribution of southeast Alaska's rainforest in mitigating climate change (DellaSala et al. 2022). Second, restoration of forests throughout the matrix through intermediate stand management of second growth should become a forest management priority, especially on Prince of Wales Island and other islands that support island endemics whose native distributions have been substantially reduced by clearcut logging. Priority should be given to landscapes in which old-growth forests are isolated and to second-growth forests along anadromous streams.

Intermediate stand management will reduce midstory density and expedite ecological succession toward achieving mature forest conditions (Nowacki and Kramer 1998) that will benefit the federally listed Queen Charlotte goshawk (Smith 2013) and increase functional connectivity of managed landscapes for endemic small mammals (Flaherty et al. 2008, 2010*a*, *b*; Smith et al. 2011; Howard 2022). Healthy anadromous streams support salmon populations that provide vital marine nutrients required for forest regeneration and development (Quinn et al. 2018, Schoen 2020). Restoration of riparian forests will directly contribute to the health and diversity of the old-growth forest ecosystem (Schoen 2020).

Thirdly, we recommend the Tongass National Forest undertake a formal review of WCS elements that appear incapable of achieving mandated or desirable expectations because of extensive historical timber harvests, misimplementation of proposed or established policies, or untenable assumptions. The review will require an updated assessment of forest resources to accurately inventory and map habitats (Shanley et al. 2021), and extensive research to document the diversity and life-history needs of southeast Alaska's unique ecological communities (Cook et al. 2006), with an initial focus on populations and habitat of the Queen Charlotte goshawk and island endemic mammals that have experienced substantial broad-scale disturbance (Smith et al. 2011, Smith 2013). Conservation measures need to consider the unique life-history attributes of sensitive species. Recognizing the hierarchical structure of goshawk breeding home ranges is fundamental to designing and implementing an effective conservation plan.

MANAGEMENT IMPLICATIONS

Future conservation and management policies and actions will require consideration of recent research findings (especially from the Tongass) and a comprehensive long-term monitoring plan to evaluate implementations and corresponding responses and outcomes. Clearly, an adaptive management approach that explicitly acknowledges and considers the uniqueness of southeast Alaska's varied landscapes and spatial context, geological history, fauna, and ecological communities will provide insights into the complexities and limitations of imposing established forest management policies and actions. A new paradigm that employs new knowledge with systematically scheduled assessments from monitoring programs will provide timely, meaningful evaluations of the consequences of management actions that can remedy existing deficiencies and improve WCS effectiveness.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

ETHICS STATEMENT

The work summarized in this manuscript did not analyze new data collected from studies of wildlife or humans.

DATA AVAILABILITY STATEMENT

Data sharing not applicable because we did not generate new data. All data referenced in this review are included in published articles cited in the manuscript.

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