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Ecosystem Services and Emergent Vulnerability in Managed Ecosystems: A Geospatial Decision-Support Tool

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Abstract

Managed ecosystems experience vulnerabilities when ecological resilience declines and key flows of ecosystem services become depleted or lost. Drivers of vulnerability often include local management actions in conjunction with other external, larger-scale factors. To translate these concepts to management applications, we developed a conceptual model of feedbacks linking the provision of ecosystem services, their use by society, and anthropogenic change. From this model we derived a method to integrate existing geodata at relevant scales and in locally meaningful ways to provide decision-support for adaptive management efforts. To demonstrate our approach, we conducted a case study assessment of southeast Alaska, where managers are concerned with sustaining fish and wildlife resources in areas where intensive logging disturbance has occurred. Individual datasets were measured as indicators of one of three criteria: ecological capacity to support fish/wildlife populations (*provision*); human acquisition of fish/wildlife resources (*use*); and intensity of logging and related land-use change (*disturbance*). Relationships among these processes were analyzed using two methods—a watershed approach and a high-resolution raster—to identify where *provision*, *use* and *disturbance* were spatially coupled across the landscape. Our results identified very small focal areas of social-ecological coupling that, based on post-logging dynamics and other converging drivers of change, may indicate vulnerability resulting from depletion of ecosystem services. We envision our approach can be used to narrow down where adaptive management might be most beneficial, allowing practitioners with limited funds to prioritize efforts needed to address uncertainty and mitigate vulnerability in managed ecosystems.

Key words: ecosystem services; social-ecological systems; anthropogenic change; resilience; vulnerability; adaptive management; southeast Alaska; even-aged forest management; subsistence.

Introduction

A persistent challenge in management of resource systems is the protection of natural capital that society depends upon, that is, the ecological functions that provide resource stocks and support the

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necessary conditions for human life (Folke and others 1991; Collados and Duane 1999). Land use and resource extraction strongly affect how these ecosystem services (ES) are provided by natural and managed landscapes (Lambin and others 2003). Managers and decision-makers are increasingly challenged to sustain ES flows to stakeholders and economies in the face of many types of social and ecological change, including drivers external to local or regional systems (Haynes and Quigley 2001; Deutsch and others 2003). Degraded or lost ES flows have been associated with environmental crises, economic downturns, and highly polarized conflicts among stakeholders and decision-makers (Carpenter and Gunderson 2001); for example, collapse of fisheries (Carpenter and Brock 2004) and timber-based economies (Trosper 2003); conflicts over water rights and regulation in agricultural systems (Cassman and others 2005); decline of major continental estuaries associated with agricultural land use (Light and others 1995); and widespread species extinctions and invasions (Erlich and Mooney 1983). These outcomes highlight how feedbacks from anthropogenic change in the biosphere—including local actions intended to achieve sustainable outcomes—may result in complex and challenging obstacles to the sustainability of managed ecosystems.

Integrative studies of coupled human-natural systems have provided important insights on the origins, dynamics and uncertainties of these management problems (Gunderson and others 2002). In seeking environmental stability and predictable flows of resources, people may modify natural disturbance regimes and ecological processes in ways that reduce ecological resilience to change (Holling and others 2002; Walker and others 2006). With lowered resilience, ecosystems become more vulnerable to dramatic shifts in the capacity to generate ES flows essential to economic, cultural, and physical well-being (Holling and others 2002). The complexity of these coupled systems creates high uncertainty and unpredictability of dynamics in response to these changes (Walker and others 2004). For example, resource-extractive land use provides immediate societal benefits and positive feedbacks to the proximate drivers of land-use change (Lambin and others 2001), yet these benefits often come with ecological costs that are temporally or spatially displaced because of nonlinear and lagged responses of ecosystems (Ludwig and others 1993). In this way, human impacts on ecosystems often become limiting or transformative factors in societal development, rather than provide the stability intended by management.

Our objectives in this study were to represent these dynamics with a conceptual model, translate model relationships into analytical tools, and apply these tools for preliminary assessment of emergent vulnerability in a managed forest ecosystem. Despite recent frameworks that allow managers to move beyond panaceas and examine specific sources of vulnerability in resource systems (Ostrom 2007), there are few tools or techniques for assessment or prioritization of these vulnerabilities. Prioritization of areas most vulnerable to ES losses can focus management efforts to address critical problems within the constraints of public funds. This article represents our first step toward building the theory and knowledge needed to optimize adaptive management efforts that seek to understand and mitigate vulnerabilities prior to catastrophic change.

Conceptual Model

Social and ecological systems are coupled by multiple types of complex interactions within and across scales (Low and others 1999). To understand emergent vulnerabilities in managed ecosystems, we focused on three interactions: the capacity for ecosystems to generate ES (provision), flows of ES to society (use), and impacts of intensive extractive use (disturbance) (Figure 1).

In an initial state without intensive anthropogenic influences, use and disturbance are small relative to provision. In this state, the ecosystem has a high provisioning capacity whether or not the resulting ES are used by society (Figure 2A). With increasing human use of ES (particularly extractive resource uses), disturbance often increases in parallel with use (Figure 2B). Here ecological resilience is tested by cumulative feedbacks to both the resource stocks and the regenerative capacity needed to sustain the provisioning of ES. Once these feedbacks exceed the resilience threshold of the coupled system, provision capacity decreases nonlinearly (Figure 2C). Eventually, the decline in provisioning of ES reduces potential use by people (Figure 2D). The resulting state, with low provision and use, but with a persistent legacy of high disturbance, may be highly resilient to attempts to restore the initial system condition and its ES flows (de Groot 1992) although some restoration may be possible with management (Lambin and others 2003) and natural recovery after disturbance (Rudel and others 2005).

Vulnerability emerges in the second time-step (Figure 2B), when use and disturbance feedbacks push the system toward thresholds, but before catastrophic change occurs. If these locations of

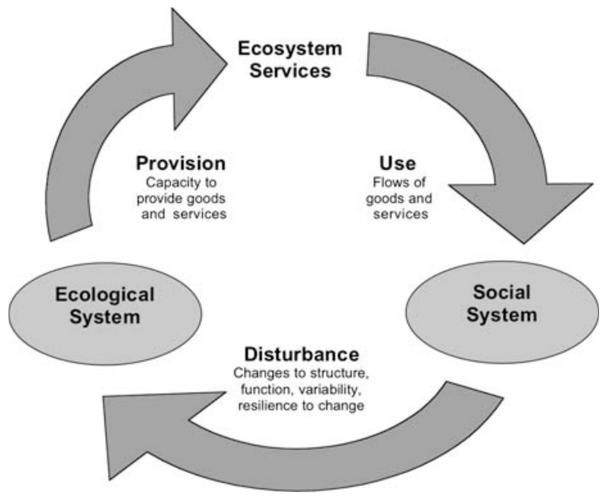

Figure 1. A conceptual model of social-ecological interactions in managed ecosystems. Provision represents the ecological capacity to generate goods and services, commonly referred to as natural capital. Use represents the actual flows of ecosystem goods and services to society, which at a given instance may be lesser, equal to, or greater than provision. Disturbance represents the human modifications of ecosystems that results from extractive use and associated efforts to stabilize environmental conditions and future flows of resources. Feedbacks from disturbance directly affect provision by depleting resource stocks and degrading the regenerative capacity of ecological functions.

high provision, use, and disturbance can be identified, targeted management interventions have a much greater likelihood of preventing loss of ES flows than if these interventions were applied uniformly or haphazardly across the landscape.

Objectives

To translate these concepts into practical applications, we developed an assessment method that integrated numerous datasets reflecting ecological productivity, resource use (ES flows), and anthropogenic disturbance. We conducted a trial assessment of southeast Alaska, as a case study of the cumulative feedbacks of land-use change (even-aged forest management) on important ES flows (fish and wildlife resources) to local communities (see Lambin and others 2003). In southeast Alaska, experts anticipate that ecological capacity to support wildlife and fish populations will decline as forests regenerate into dense second-growth stands and as stream culverts fail and become potential barriers to fish passage or detrimental to rearing and spawning habitats. Land managers face the challenge of understanding these interactions and mitigating undesirable outcomes across the vast (8.8 million ha) region of southeast Alaska, with very limited institutional resources. Thus our analysis was intended to identify and prioritize areas where research and management would be most valuable.

Methods

We developed an assessment methodology with three objectives: (1) describe the spatial patterns of ecological capacity, human use, and anthropogenic disturbance, using existing data sets; (2) integrate this information at relevant scales and in locally meaningful ways; and (3) develop a practical assessment tool that could serve as decision-support for adaptive management efforts. We chose southeast Alaska as a case study, developed a set of indicators based on expert knowledge of local interactions and current management issues, and assembled a regional geodatabase that represented the best available, albeit incomplete, knowledge base for our purposes. Individual datasets, such as forest condition or deer harvest, were measured as indicators of one of three criteria: the ecological capacity to produce fish and wildlife (*provision*); local harvest of fish, wildlife, and related resources (*use*); and intensity of logging, roading, and other types of land-use change (*disturbance*). Spatial relationships among these criteria were analyzed and visualized using two alternative methods—a watershed-based approach and a continuous response (raster) surface. Criteria scores were analyzed across the region using non-parametric analyses and raster arithmetic to identify where *provision*, *use* and *disturbance* were spatially coupled across the southeast Alaska landscape. Lastly, our approach involved numerous assumptions and methodological choices that influenced the assessment outputs. An explanation of each of these methods follows.

Study Area

Southeast Alaska extends from Icy Bay near Yakutat (59°N,140°W) to Dixon Entrance (55°N,130°W) and includes the western portions of the Coast Range on the mainland and the Alexander Archipelago. Defined as a temperate rainforest biome, the region is characterized by its mild maritime climate, island biogeography, coastal glaciers, salmon streams, high faunal diversity, and dense conifer forests. Of a permanent population of roughly 75000, over half reside in two 'urban' centers (Juneau and Ketchikan) and 32 smaller rural communities and Alaska Native villages. Most communities are geographically isolated by water-

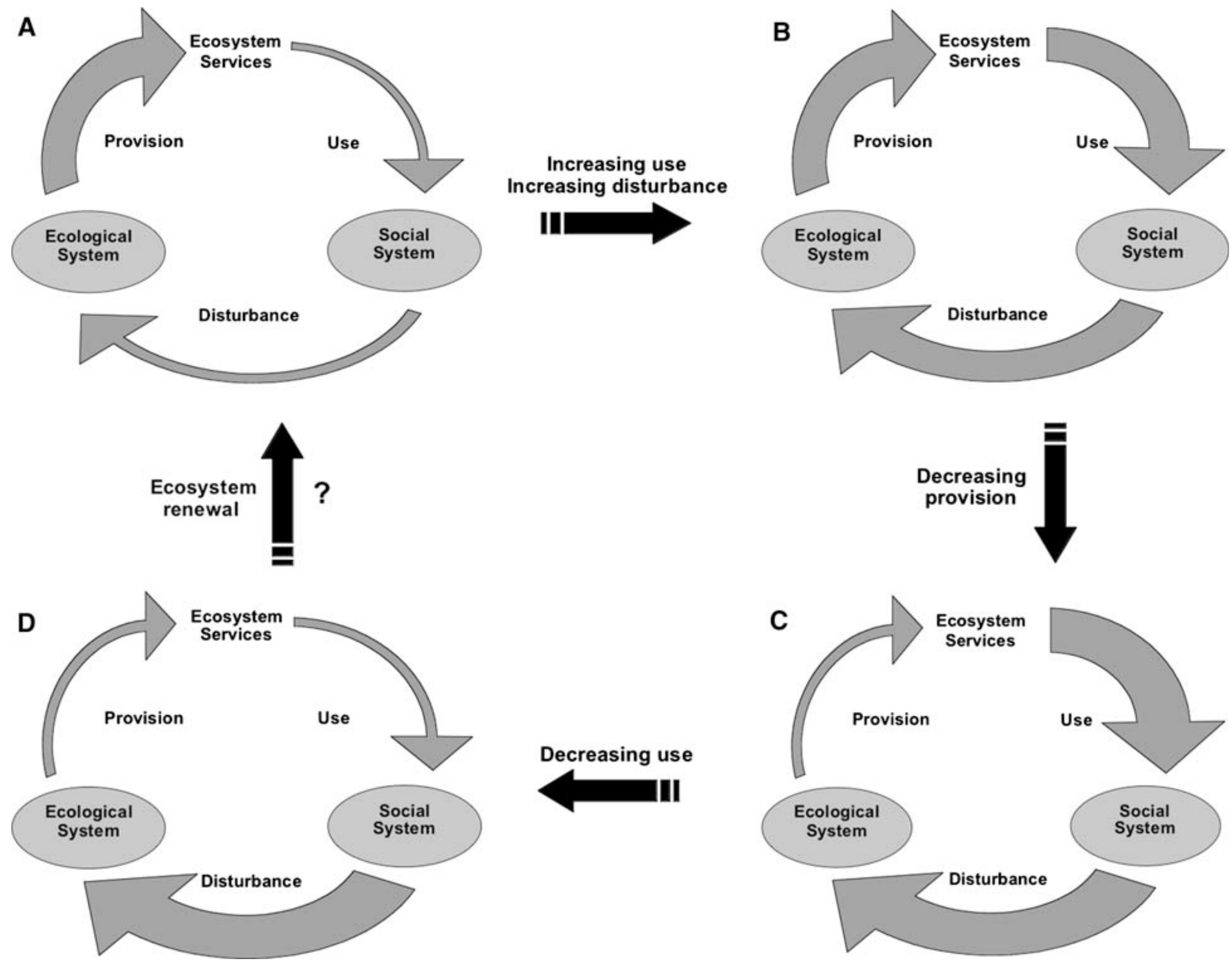

Figure 2. A hypothetical sequence of system dynamics leading to vulnerability in managed ecosystems, via declines in provision and use of ecosystem services. Thickness of arrows represent the relative magnitudes of provision, use, and disturbance at each time step. Dynamics progress from (**A**) an initial state where provision is large, but use and disturbance are relatively small. With increasing extractive use of ecosystem services (**B**) there are concomitant increases in disturbance, but the ecosystem appears resilient to these feedbacks. When cumulative feedbacks exceed ecosystem resilience (**C**), provision capacity declines non-linearly, followed by attendant declines in use (**D**), that is, loss of ecosystem service flows. At the endpoint, the potential for the system to revert to its initial state is unknown. Vulnerability is emergent when feedbacks from use and disturbance were greatest, prior to declines in provision (**B**).

ways, terrain, and lack of a regionally integrated road network. Local use of natural resources is primarily regulated by two agencies (US Forest Service and the Alaska Department of Fish and Game), and roughly 90% of the region is public land. The Tongass National Forest, at 6.2 million ha, comprises about 75% of the region's land area and envelops most communities (Figure 3). In the following sections, we summarize the expert knowledge used to select indicators for preliminary assessment of the southeast Alaska case study.

Ecological Provision

The temperate rainforests of southeast Alaska are among the most productive and biologically diverse ecosystems at high latitudes. Nearly continuous precipitation and mild temperatures support a range of vegetation types including coastal spruce-hemlock forest, deciduous forest and shrubs, muskegs (peat bogs), and alpine tundra (Viereck and Little 1986). Along a productivity gradient, vegetation ranges from large-stature closed-canopy forests on well-drained soils to stunted open-canopy forest and shrub bogs or muskegs on saturated peat soils (Neiland 1971; Mead 1998). The most productive forests in southeast Alaska tend to be mixed spruce-hemlock stands on riparian sites, or well-drained areas underlain by karst limestone formations. These 'big-tree' forests comprise less than 5% of the region by area, but are considered

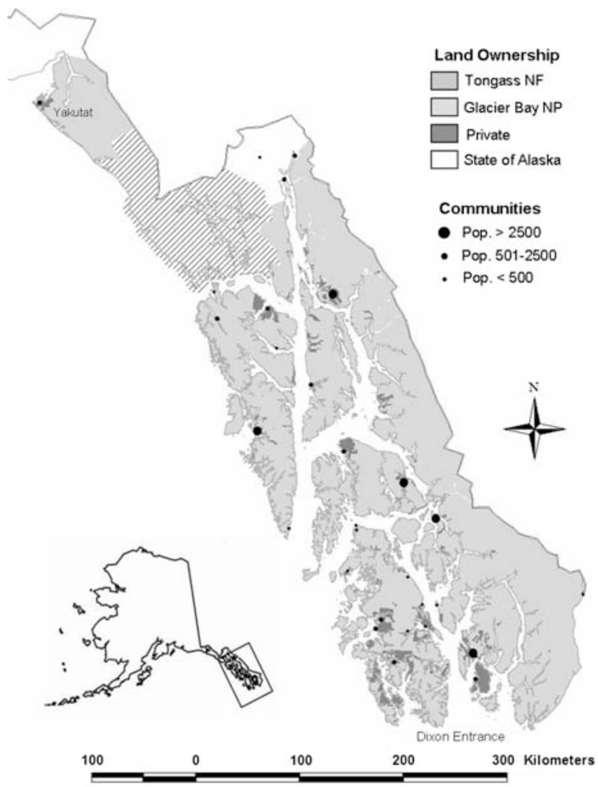

Figure 3. Map of southeast Alaska, with communities and land ownerships.

to be hotspots for species diversity and productivity (Cook and others 2006).

Endemic fauna are adapted to an old-growth forest matrix that includes riparian edges, wetlands, and coastal estuaries (Hanley and others 1989; Hargis and others 1999). Understory vegetation, including *Vaccinium* spp. and several herbaceous and graminoid plants, are important browse for large mammals such as deer and bear. Habitat suitability indices (HSI) for these species reflect the importance of these features, as well as elevation, snow cover, and/or proximity to salmon streams (Hanley and others 2005). Forested watersheds and riparian vegetation also support aquatic habitat conditions for spawning and rearing of five species of Pacific salmon (*Onchorynchus* spp.) and other anadromous fish. Salmonids are keystone species important to stream productivity (Claeson and others 2006; Wipfli and others 1999), terrestrial food webs (Hilderbrand and others 1999), and riparian forests via marine nutrient inputs (Helfield and Naiman 2001; Gende and others 2002).

Windthrow is the prevalent stand-replacing natural disturbance in the coastal temperate rainforests of southeast Alaska (Nowacki and Kramer 1998; Kramer and others 2001). Unlike the taiga of boreal Alaska and Canada, fire is very rare and destructive in these forests. Natural regeneration occurs on exposed mineral soils, nurse logs, and organic soils, and the type of disturbance and substrate strongly influence the species composition of regeneration (Deal and others 1991). In conjunction with gap-phase regeneration, natural disturbances foster a high structural complexity in old-growth rainforests (Alaback 1982), upon which many endemic species are dependent.

Anthropogenic Disturbance

The cumulative impacts of timber harvest and even-aged forest management are the most visible and widespread form of anthropogenic disturbance in southeast Alaska. Since 1954, approximately 250,000 ha of old-growth forests in the Tongass National Forest have been harvested, nearly all by the clear-cut method. Although even-aged management required harvest of all trees in the unit, records indicate 'landscape high-grading' was practiced where the most productive stands were targeted first for harvest. As a result, on some islands logging disturbance may have been concentrated in the forests with the highest capacity to support fish and wildlife populations (Hanley and others 2005).

Overall, the long-term impacts of even-aged forest management are poorly resolved in southeast Alaska, largely because disturbances of this type and magnitude have no historical precedent in the region. Although most harvested areas have initiated rapid regeneration, too little time has passed to understand impacts that may occur over the medium- to long-term. However, ecologists and managers expect that during long periods of regeneration (100–250 years), dense second-growth stands will exclude understory browse vegetation (Alaback 1982; Deal and others 1991) and lack the unique structural features that support habitat of endemic species (DeGange 1996; Willson and Gende 2000). As a result, habitat bottlenecks are a principal concern on islands where even-aged management has been extensive (ADFG 1998; Hanley and others 2005).

Even-aged forest management also commonly has negative impacts on aquatic habitat conditions in salmonid spawning/rearing areas. Harvesting of riparian forests alters stream habitat by increasing light penetration (Meehan 1970; Tyler and others 1973), altering stream chemistry (Singh and Kalra 1977), reducing inputs of large woody debris (Chamberlin 1982), increasing sediment loading due to runoff and soil erosion (Brown and Krygie

1971; Swanson and Dyrness 1975; Beschta 1978), and altering fluvial geomorphology (Wood-Smith and Buffington 1996). Outside of the riparian zones, impacts of timber management on hydrological and nutrient cycles—two closely coupled processes in these mesic soils—and stream crossing structures (for example, culverts) can degrade stream habitat even if riparian habitats remain undisturbed (Chamberlin 1982). Overall these changes reduce habitat quality for spawning and rearing salmonids, especially if roads and stream crossings are not adequately maintained.

Human Use

Residents of southeast Alaska engage in a variety of subsistence practices, ranging in significance from primary sources of food and firewood for rural households, to relatively minor supplements for 'urban' households. In the 32 rural communities of the region, annual gross subsistence harvest (not including firewood) is approximately 5.8 million lbs (2900 tons), equivalent to 271.2 lbs per capita (Alaska Department of Fish and Game, unpublished data) of wild game, fish, seafood, plant foods, birds and eggs, and various non-timber forest products. In addition to nutrition, traditional hunting and gathering practices are integral to the cultural heritage of Alaska Natives; federal law mandates a priority for subsistence over other uses during times of resource shortage.

Several regional industries also depend directly on fish and wildlife populations supported by the ecosystems of southeast Alaska. The commercial seafood industry supports a large proportion of revenue and employment in many rural and urban communities (Hartman 2002). Nature-based recreation and tourism has been the fastest growing industry in the last two decades, now comprising more than 10% of the regional economy (Colt and others 2006). Sport-fishing and hunting supports the guide/outfitter and ecotourism industries, which depend on the availability of fish and wildlife species for harvest and/or observation in their natural habitats. Growth in recreation and tourism has been driven in part by dramatic increases in cruise ship visitation, which supports primarily non-consumptive activities such as sightseeing and wildlife viewing.

Geodata, Indicators, and Criteria

We assembled a southeast Alaska geodatabase using existing datasets with full regional coverage pertaining to biophysical and ecological condition, human use and infrastructure, and local anthropogenic disturbance (Table 1). These geodata included both vector (shape) files that described a discrete feature (public use cabin, stream culvert); and raster coverages, which described a continuous feature (forest productivity, habitat suitability). We adapted a 'criteria and indicators' approach (Canadian Forest Service 2001) to aggregate multiple variables (indicators) into a small number of indices (criteria) for statistical and spatial comparison. Indicators were selected based on data availability and the expert knowledge summarized in the preceding sections. Given the challenge of integrating multiple datasets from different sources in a transparent and logical manner, we explored two approaches for measuring indicators, analyzing criteria, and presenting results; these methods are described below.

Estimating Indicators and Criteria: Watershed Ranking Method

The objective of this approach was a parsimonious and transparent integration of multiple data sets at a resolution commonly used in management planning (watersheds). First, we estimated each indicator for each watershed based on raw or area-weighted values of geodata, such as productive old-growth forest, average deer harvest, number of stream culverts, and so on (Table 1). Next, instead of averaging indicators with different units of measure (for example, hectares, meters, number of sites), we ranked watersheds (n = 1006) by each indicator. Watersheds could have same rank for a given indicator if the values were equal, for example, if percent-harvested forest equaled zero. We then averaged indicator ranks to provide a criteria score for *provision*, *use*, and *disturbance* for each watershed.

Next, we analyzed probability distributions of criteria scores and pairwise correlations among indicators. In addition to basic insights, these analyses helped determine suitability of these data for parametric models; two of three criteria score distributions were non-normal, and the majority of indicators were autocorrelated, both within and across criteria groupings. This suggested the data were inappropriate for parametric models to investigate statistical relationships among watershed criteria scores.

Therefore to describe relationships among *provision*, *use*, and *disturbance* we used alternative analyses that required fewer assumptions, such as K-means clustering and conditional queries of watershed criteria scores. The K-means method clusters samples (watersheds) into a predefined

Table 1. Criteria, Indicators, and Techniques used for Two Methods of Measuring and Aggregating Indicators into Criteria

Criteria	Indicator	Watershed method	Raster method
Provision	Productive forest land[1]	% WS area	Existing raster
	Productive old-growth forest		
	Second-growth forest		
	Habitat suitability[2]	% WS area	Existing raster
	Sitka black-tailed deer		
	Brown bear		
	Black bear		
	Anadromous habitats[3]	Stream length/WS area	Distance function
	King salmon		
	Coho salmon		
	Pink salmon		
	Chum salmon		
	Sockeye salmon		
	Steelhead trout		
	WS fish productivity[4]	N/A	Raster multiplier
	Estuaries[1]	% WS area	Distance function
Use	Fishing/Seafood Harvest		
	Major sport-fishing WS[4]	N/A	Distance function
	Shellfish harvesting sites[1]	n sites/WS area	Distance function
	Hunting[4]	n harvested/WS area	Interpolation (krieg)
	Sitka black-tailed deer		
	Brown bear		
	Black bear		
	Infrastructure[1]	n sites/WS area	Distance function
	Public use cabins		
	Log transfer sites		
	Coastal use[1]	N sites/WS area	Distance function
	Harbors		
	Hatcheries		
	Aquaculture		
Disturbance	Harvested forest[1]	% productive forest land	Existing raster
	Urban land cover[2]	% WS area	Existing raster
	Road-stream crossings[5]	n crossings/stream length	Distance function

[1]*Southeast Alaska GIS Library (2005); US Department of Agriculture Forest Service-Alaska Region.*
[2]*The Nature Conservancy-Alaska, Juneau, AK.*
[3]*Anadromous Waters Catalog (2006); U.S. Fish and Wildlife Service , U.S. Department of Interior.*
[4]*Tongass Resource Assessment (1998); Alaska Department of Fish and Game, Juneau, AK.*
[5]*Geodata produced for this study using an intersect of roads (USDA) and anadromous waters (USFWS).*

number of groups using Euclidean distances (between sample and cluster means) and multiple iterations to achieve robust group convergence. We used clustering to identify natural groupings among the population of watersheds on the basis of their criteria scores.

Analysis using conditional statements identified a group of watersheds that satisfied a specified condition—that is, '*high provision* AND *high use* AND *high disturbance'*—evaluated at a range of sensitivities, or 'benchmarks'. Results of these conditional queries provided groups of watersheds that reflect degrees of coupling among criteria in a spatially explicit manner. Clusters of watersheds identified by *K*-means were compared with results of conditional queries to determine whether the two methods identified similar groups of watersheds.

Estimating Indicators and Criteria: Raster Interpolation Method

The objective of raster interpolation was to generate continuous, high-resolution coverages of criteria scores to capture the spatial heterogeneity of indicators and, if possible, generate normally distributed scores suitable for parametric statistical analysis. For this purpose we used geospatial processing tools and interpolation techniques that introduced several assumptions and artifacts into the analysis. Table 1 lists methods applied for each indicator and the major steps are summarized below. First, we modified existing raster coverages by reclassifying cell values to a scale of 1–10 using geometric intervals; this step applied to the majority of *provision* and *disturbance* indicators. For vector data, such as point (for example, public use cabins) and line features (for example, salmon streams), we calculated distance functions with a maximum of 2.5 km, so that raster cells closer to a given feature were scored higher than cells further away; this method was applied primarily for *use* indicators. We explored a range of maximum distance parameters (from 0.5–20 km) in an informal sensitivity analysis and chose 2.5 km as the global parameter for distance functions because its outputs approximated the median value of *use* criteria scores along this range. For *provision*, watershed salmon productivity (ADFG 1998) was used as a multiplier of the salmon stream raster. Streams in primary salmon-producing watersheds were weighted (5×) greater than secondary (2×) and non-producing watersheds (no multiplier). Game harvest data, coded as a watershed attribute, were processed by converting watershed polygons to center points and interpolating point attributes (for example, total deer harvested) by simple krieging. Lastly, for primary sport-fishing watersheds (a binary attribute), a distance function was calculated from the salmon streams in those watersheds identified for high sport-fishing use (areas closer to the salmon stream scored higher). Together, these methods produced raster coverages for each indicator in Table 1. We aggregated indicator rasters by summing cell values to generate three criteria rasters, which we reclassified (as above) and then summed to produce a single raster that estimated 'social-ecological coupling', an index we defined based on the nexus of *provision*, *use*, and *disturbance* scores.

Assumptions, Proxies, and Limitations

A major challenge in understanding vulnerability of social-ecological systems (SES) lies in the integration of social and ecological information across space and time (Carpenter and Brock 2004; Alessa and others 2007). Coupled SES and their interactions are not static over time, nor are they uniformly distributed across the landscape. Likewise, ecological goods and services are generated and received at a range of spatial and temporal scales (Limburg and others 2002); these processes are often shaped by multiple interacting drivers of change and the legacies of historical factors, such as land-use change (Collados and Duane 1999; Lambin and others 2003). Moreover, available information for measuring SES interactions, such as ecosystem services, rarely provide a complete picture of these interactions at multiple scales (Low and others 1999). To address these complexities and data gaps, our case study analysis of southeast Alaska required several assumptions and proxy measures. In addition to those already mentioned, we highlight three areas where methodological choices influenced our results.

First, in many portions of the methodology we treated watersheds as individual units of analysis. This allowed a focus on the features and processes that could be differentiated at the watershed (or finer) resolution, because only these indicators would influence watershed scores. Watersheds are increasingly used as integrated ecological units to assess and manage resources and ES (Lant and others 2005), even though this approach fails to account for larger-scale interactions among watersheds, such as population movements or fluxes of water, nutrients, and energy. Our current analysis is limited by data availability to a snapshot of current conditions and does not capture any temporal variability or dynamics, which are usually critical to understanding emergent vulnerability in complex systems (Walker and others 2006). We argue, however, that the current approach can be used to understand where vulnerability is most likely to emerge, in conjunction with knowledge of post-disturbance interactions and drivers of change.

Second, existing geodata for fish/wildlife harvest were not comprehensive, so we used the best available proxies. We estimated hunting based on harvest tags associated with permits and surveys and therefore did not include unreported harvests, which are common for rural subsistence hunters (ADFG 1998). Because a majority of permit holders and respondents resided in the population centers of Juneau and Ketchikan, the hunting data in this study poorly represented the rural communities of southeast Alaska. In addition to missing community-level data, there were several important categories of subsistence harvest (for example, fish, seafood, plant materials) missing from the analysis because suitable geodata with full regional coverage were unavailable. Data reflecting non-consumptive activities such as wildlife viewing were also not available; instead we used the proxy of recreation sites (public use cabins). Lastly, commercial fishing occurs mostly in ocean passages and harvested fish could not be spatially linked to specific streams or watersheds. Overall, these information gaps created a considerable bias in the spatial estimation of fish/wildlife ES flows, in favor of ‘urban’ hunters and sport anglers.

A third complication involved selection of indicators related to forest roads. Roads are an important source of access for hunters and anglers, but may deter recreationists seeking remote wilderness experiences (Miller and McCollum 1997). Because roads may simultaneously support and discourage different uses in southeast Alaska, it is unclear how their existence cumulatively affects social capacity to acquire resources, from subsistence deer to amenity values like isolation and scenery. Forest roads also create disturbance, for example, erosion and sedimentation, changes in upland runoff, groundwater flow, stream flow regime and are a vector for invasive plants (Gucinksi and others 2001). However, the impacts of roads are highly variable based on location, type and quality of construction. For these reasons, we elected not to use roads as a stand-alone indicator of either *use* or *disturbance*. In part, the spatial distribution of the hunting and fishing data used in this study strongly reflects the importance of roads (ADFG 1998). Road crossings with salmon streams, based on a spatial intersection of anadromous waters and roads geodata, were measured as an indicator of *disturbance*.

Results

Preliminary Analyses

Both methods generated normal distributions for *provision* scores and long-tailed (non-normal) distributions for *use* and *disturbance* scores. We found that normally distributed *use* scores could be generated using raster interpolation by calculating distance functions with no maximum distance parameter. However, this introduced an unrealistic artifact because a given feature, such as a public use cabin, influenced estimation of recreation use at distant locations many hundreds of km away. Instead we chose 2.5 km as the global distance parameter for raster interpolation, which yielded a *use* raster in which roughly one-third of the raw cell values (criteria scores) equaled zero.

Based on watershed scores, most indicators were weakly to mildly autocorrelated, both within and across criteria groupings. Criteria scores calculated by the watershed ranking method were also weakly correlated, the strongest relationship existing between *use* and *disturbance* (Table 2). As we

Table 2. Pairwise Correlations Among Indicators of Different Criteria, and Among Criteria Scores, Based on 1006 Watershed Scores Calculated Using the Watershed Ranking Method (see *Methods*)

Variable	By variable	r	P
Provision	Use	0.21	<0.0001
Deer habitat	Sport-fishing	0.06	0.041
Forest land	Sport-fishing	0.10	0.0018
Forest land	Deer harvest	−0.07	0.0195
Forest land	Coastal use	−0.07	0.031
Deer habitat	Deer harvest	0.13	<0.0001
Deer habitat	Bear harvest	0.21	<0.0001
Provision	Disturbance	0.25	<0.0001
Estuary	Urban land cover	0.08	0.0128
Forest land	Fish stream × roads	0.07	0.0314
Deer habitat	Fish stream × roads	0.19	<0.0001
Deer habitat	% forest harvested	0.14	<0.0001
Use	Disturbance	0.38	<0.0001
Sport-fishing	% forest harvested	0.09	0.0059
Coastal use	% forest harvested	0.07	0.0251
Bear harvest	% forest harvested	0.15	<0.0001
Recreation sites	Urban land cover	0.24	<0.0001
Coastal use	Urban land cover	0.48	<0.0001
Sport-fishing	Fish stream × roads	0.21	<0.0001
Bear harvest	Fish stream × roads	0.24	<0.0001
Recreation sites	Fish stream × roads	0.18	<0.0001
Coastal use	Fish stream × roads	0.22	<0.0001

Only those correlations significant at $P < 0.05$ are depicted.

previously discussed, long-tailed distributions of criteria scores and autocorrelation among indicators (used to calculate criteria) suggested that parametric models were unsuitable for describing relationships in *provision*, *use*, and *disturbance* among watersheds. Results of non-parametric methods to measure spatial coupling of these criteria, including conditional and cluster analyses, are presented below.

Nonparametric and Raster Analyses

Conditional queries of watershed criteria scores identified where provision capacity, resource use, and localized disturbance were coupled, across a range of sensitivities. We mapped those watersheds that satisfied the conditional statement of high *provision*, high *use* and high *disturbance*, based on a given benchmark (Figure 4). At the highest level of sensitivity (the 5% benchmark), three watersheds satisfied the conditional statement, that is, each scored in the top 5% of all criteria distributions. As sensitivity was reduced, the analysis captured watersheds with lesser degrees of coupling, exhibiting a clumped pattern across the landscape (Figure 4). The 50% benchmark, the lowest sensitivity tested, included all watersheds where *provision* was in the upper 50% and where *use* and *disturbance* were non-zero (due to long-tailed distributions).

Cluster analysis using the *K*-means method was explored as a technique to identify natural groups of watersheds (n = 1006) based on criteria scores.

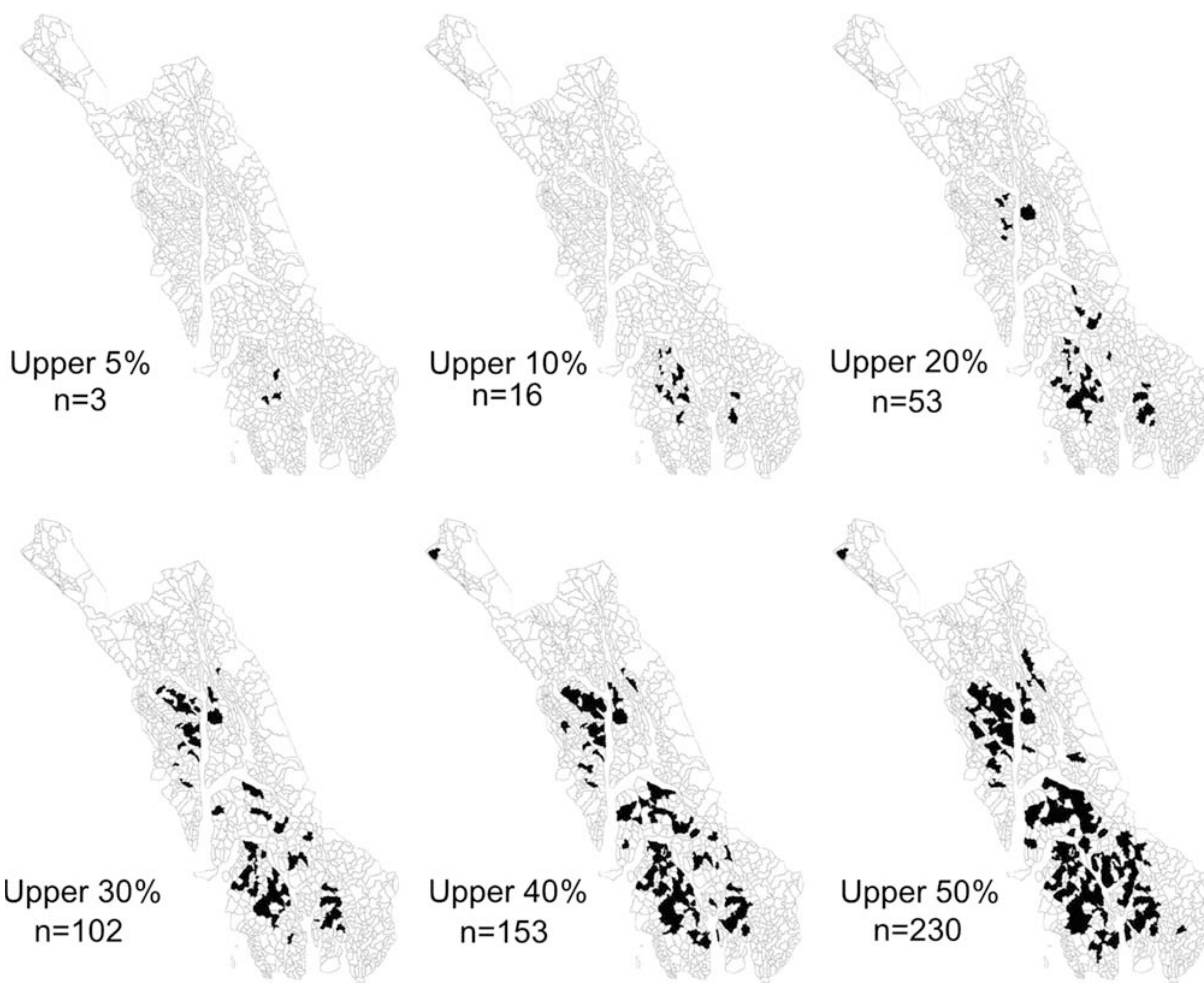

Figure 4. Results of conditional queries of watershed criteria scores calculated using the watershed ranking method. Watersheds filled in solid black indicate where high provision, use, and disturbance scores are spatially coupled, based on conditional statements evaluated at a range of sensitivities, from the upper 5% to upper 50% of the criteria score distributions.

Any number of clusters may be imposed on the data prior to analysis, so we explored sensitivity of results to the pre-defined number of clusters, ranging from 3 to 15. We found that with five clusters identified, a single cluster had significantly higher mean criteria scores than the remaining four clusters (Tukey's HSD, $P < 0.01$). This cluster also captured the vast majority of watersheds identified using conditional statements, that is, all watersheds at the 25% benchmark, and greater than two-thirds of watersheds at the 50% benchmark (Table 3; cluster E). We found that if more than five clusters were imposed onto the watershed criteria data, the *K*-means analysis split 'cluster E' into several smaller groups of watersheds that were statistically equivalent in terms of mean criteria scores (Tukey's HSD, $P > 0.05$). Because no additional information was yielded beyond five clusters, we focused on these results, which suggested the existence of a distinct group of watersheds where *provision*, *use*, and *disturbance* were tightly coupled (Table 3).

Criteria score rasters were summed to generate a continuous, high-resolution surface for estimating the coupling of fish/wildlife ES flows and localized disturbance from land-use change (Figure 5). Summary raster (SES coupling) values were non-normally distributed because the majority of cells had the lowest possible score for *disturbance*, and nearly one-third had the lowest possible score for *use*. Although we identically reclassified criteria rasters and introduced no weighting to the calculation, the 'SES coupling' raster largely reflected the spatial heterogeneity of *disturbance* (Figure 5).

Comparison of Method Outputs

Overall, very similar results were generated by the watershed ranking method (Figure 4) and the ras-

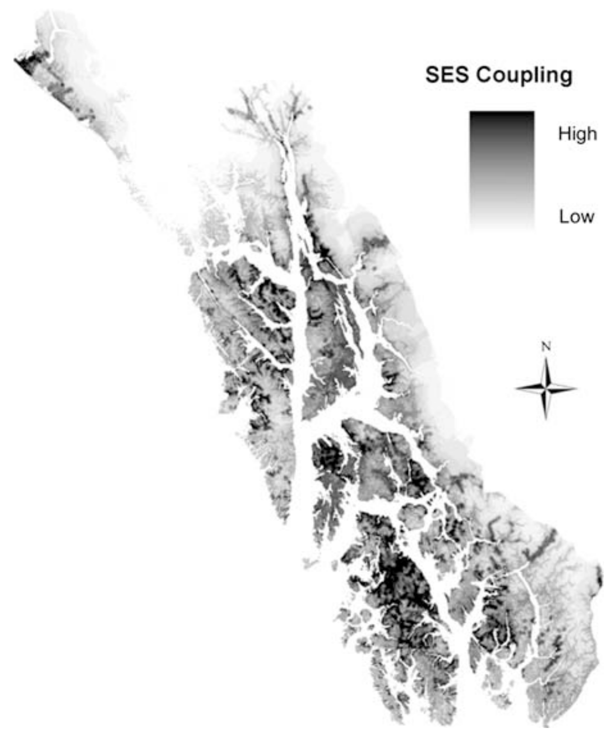

Figure 5. Results of the raster interpolation method for estimation of social-ecological coupling in fish and wildlife interactions in southeast Alaska. Coupling reflects where provision capacity, human use, and anthropogenic disturbance coincide on the landscape, using a grayscale gradient from low (light) to high (dark).

Table 3. Combined Results from *K*-means Cluster Analysis and the Sensitivity Analysis to Benchmarks used in Conditional Queries, Based on Criteria Scores Generated by the Watershed Ranking Method

Cluster	*n*	Benchmark								
		5%	10%	15%	20%	25%	30%	35%	40%	50%
A	283	0	0	0	0	0	1	4	16	63
B	81	0	0	0	0	0	0	0	0	1
C	227	0	0	0	0	0	0	0	0	0
D	229	0	0	0	0	0	0	0	0	8
E	186	3	16	30	53	75	101	122	137	158
Total *n*	1006	3	16	30	53	75	102	126	153	230
Total area (ha × 10^3)		30.34	123.73	226.41	393.51	571.25	769.05	1037.25	1245.05	1894.74
% Region (by area)		0.28%	1.16%	2.13%	3.69%	5.36%	7.22%	9.74%	11.69%	17.79%

Benchmarks reflect the upper percentage of the criteria distribution analyzed. Data depict number of watersheds in each cluster, the number identified at each benchmark that fell within each cluster, and area metrics. Watersheds at selected benchmarks are mapped in Figure 4.

ter interpolation method (Figure 5). In addition to the qualitative similarity in spatial relationships among outputs, a comparison of mean raster values summarized by watershed provided quantitative evidence of similarity in results. Watersheds identified in conditional queries had significantly higher mean raster values than the watersheds not satisfying the conditional statement (Tukey's HSD, all $P < 0.05$).

Discussion

Given the limitations of the data upon which our analysis was based, we interpret the results only as an illustration of our methodology and outputs, rather than as a robust assessment of southeast Alaska. For example, the analysis relied on data that did not fully capture the variables of interest (such as, rural subsistence hunting and fishing) or provide time-series to represent local dynamics (such as, forest regeneration or fluctuation in salmon populations). The largest data gaps pertained to fish/wildlife harvest by rural communities and non-consumptive uses of these resources by residents and visitors. Analogous data limitations characterize most regions where managers seek to minimize SES vulnerabilities. Within the constraints of available data, our results suggested that nearly half of all watersheds were not being used for hunting, fishing, or various forms of recreation. In reality, many of these areas are known to be important for subsistence, recreation, and tourism. These gaps and inconsistencies suggest that our analysis should be used cautiously in implementing management actions and also underscore the importance of investment in data documenting these uses.

By contrast, we have higher confidence in our finding that a majority of watersheds had the lowest possible *disturbance* score, given the high quality of timber harvest records and the large areas of southeast Alaska that have not been actively managed. However, more diffuse anthropogenic impacts, such as climate change and accumulation of organic pollutants in biota were not reflected in our analysis. It is unlikely, however, that inclusion of these diffuse disturbances would have improved our analysis because they were unlikely to have differentiated strongly at watershed scales.

Understanding Social-Ecological Coupling

Our analysis identified areas with high *provision*, *use*, and *disturbance*, but what does the nexus of the high criteria scores signify? We supposed that it indicated tight social-ecological coupling, where threshold declines in ecological capacity might force consequent loss of ES flows. An alternative explanation is that because of the lack of time-series data, temporal changes have already occurred in ecological capacity and ES flows. In other words, locales with high *provision* and *use* might have experienced reductions relative to their prior magnitude but still score higher than non-productive, rarely used locales. Although possible, it seems unlikely that such threshold changes have occurred because evidence of collapses in fish and wildlife populations in southeast Alaska, and the management crises that would likely be associated with such collapses, are lacking. However, without historical data we cannot rule out reductions or shifts in ecological capacity and fish/wildlife harvest, especially at small scales where non-linear dynamics may be masked by apparent stability at larger scales, such as, individual streams in a larger watershed. We suggest the raster method developed in this study provides insights into such fine-scale heterogeneity that is poorly captured by the watershed approach. Overall, despite these issues, our analysis is likely to be robust with respect to current spatial relationships in the existing data, which was our principal objective.

A third possible interpretation of our measure of social-ecological coupling (that is, coincidence of high criteria scores) is that some currently unknown properties of these areas convey high resilience of *provision* and *use*, despite high levels of anthropogenic *disturbance*. Regardless of the implications of coupled *provision*, *use*, and *disturbance*, these areas clearly warrant a high priority for research and mitigation using an adaptive management approach. We suggest this because our analysis illustrated where the strongest disturbance feedbacks may accumulate in the region's most productive areas. If (or when) these feedbacks exceed ecological resilience, we expect loss of provision capacity to occur rapidly and nonlinearly and, because of high levels of use, drive changes in ES flows that will have the most pervasive societal impacts. Once lost, most ES cannot be replaced or substituted by human means (Deutsch and others 2003); they often require natural recovery occurring over long time horizons (Rudel and others 2005), during which ES losses may exceed the adaptive capacity of local economies and cultures (Carpenter and Gunderson 2001; Berkes and others 2003). Therefore, it is preferable to avoid ES losses and the resulting declines in human well-being, rather than attempt to restore lost ecological capacity and social welfare.

Uncertainty and Management

Mitigating vulnerability before undesirable change occurs is extremely difficult, in large part because of the uncertainty associated with complex SES. Our case study of southeast Alaska illustrates the difficulties that managers face under high uncertainty. Like many regions, the drivers and impacts of land-use change in southeast Alaska are emergent from interactions among state factors and multiple drivers of change (Lambin and others 2001). These include a warming climate that will influence forest regeneration, in part by driving widespread decline of long-lived, slow-growing tree species (Beier and others 2008); a regional economy increasingly dependent on amenity migration and tourism (Colt and others 2006); and a governance institution (US Forest Service) increasingly constrained by funds and litigation (Nie 2006). Local experts expect certain changes to occur but have little predictive ability, because this combination of dynamics and drivers are wholly unprecedented in the region. Such uncertainty presents challenges to decision-makers, especially where ES flows are highest and therefore so are the management stakes.

In southeast Alaska, spatiotemporal variation in forest regeneration dynamics and the non-linear responses of wildlife populations to changing habitat conditions (as forests regenerate) remain largely unresolved. As second-growth forests advance into the stem exclusion stage, habitat quality declines for wildlife species important for subsistence and commercial use. Poor habitat conditions are expected to persist for 80–200 years, with recovery time depending on spatially heterogeneous factors such as site productivity and soil drainage (Alaback 1982; Deal and others 1991). As a result, recovery times for forest ES will be highly variable and difficult to predict (Rudel and others 2005). Pre-commercial thinning treatments are the current approach to accelerate successional dynamics to hasten forest recovery. Early thinning results are promising, but limited public funds require managers to prioritize treatments in key areas of concern.

Second, there is concern about the impact of hundreds of stream-road crossings on fish passage and habitat quality, and uncertainty in how crossing structure functionality may degrade over time. Forest roads constructed for logging purposes require regular maintenance and management attention (Swanson and Dyrness 1975). Due to steep, rugged terrain and a very wet climate, culverts and other stream crossing structures in southeast Alaska commonly need repair or replacement every 5–10 years (Flanders and Cariello 2000). Failure of these structures may result in degradation of aquatic habitats and emergent changes in watershed hydrological processes (Chamberlin 1982; Wood-Smith and Buffington 1996; Gucinski and others 2001). Given constraints of public funds, managers must prioritize the culverts of highest importance, where consequences of culvert failure will be greatest.

Third, concerns over maintenance costs (in part) have prompted recent proposals by the US Forest Service to decommission logging roads in several areas. One of these places, Prince of Wales Island (PWI), supports a rapidly growing sport-hunting and guiding industry, however, nearly half of PWI's existing logging roads have been listed for possible closure. Sitka black-tail deer populations on PWI are a vital subsistence resource for communities both on and off the island. Road closures may constrain access for subsistence users, recreationists, and commercial guides that have become accustomed to logging roads over the last several decades (Brinkman and others 2007). Flexible, adaptive decision-making about road closures is constrained by uncertainty of the impacts on different user groups, and how user preferences may change over time.

Overall, these uncertainties are magnified where ES flows and disturbance are most tightly coupled on the landscape. We suggest that the analysis of spatial variability in SES coupling provides a basis for prioritizing research and mitigation efforts within the constraints of limited public funds. For example, if forest management has generated negative feedbacks to the ecological capacity to maintain fish and wildlife populations, while simultaneously creating positive feedbacks to fish and wildlife harvest in the same locales (because of increased access via logging roads) the scenario that emerges—decreasing resource availability and increasing user demand—is one where a forward-looking, adaptive management strategy will be especially valuable. When supplemented with more complete data, our approach can identify where this effort should be prioritized in southeast Alaska, in terms of both research and mitigation strategies, for example, forest thinning, culvert repair, stream restoration, and road maintenance/closure.

Future Directions

Our current method is only a starting point in efforts to detect emergent vulnerabilities and develop an applied understanding of resilience in managed ecosystems. Without better knowledge of system thresholds, we cannot predict emergent vulnerability with any degree of confidence. Recognizing that thresholds are dynamic, multi-scale properties of systems (Anderies and others 2007), we envision the need to disaggregate criteria into individual indicators so that smaller-scale thresholds can be studied to improve understanding of the larger-scale thresholds that are emergent properties of the SES. By unpacking the criteria and focusing on suites of indicators (for example, salmon habitat, harvest, and culvert suitability; or deer habitat, hunting, and road access), the task of estimating thresholds and targeting specific vulnerabilities becomes considerably more manageable. This approach also allows the management system and its social and ecological components to be framed in terms of multi-scale, integrative frameworks for SES analysis (Ostrom 2007).

Final Thoughts

The recent Millennium Ecosystem Assessment presented a broad scientific consensus that ecosystem services worldwide were in decline (MEA 2005). Addressing the social challenges listed in the report will require changes in both the conceptual basis and applied methods of resource management. Instead of broad panaceas, avoiding collapse and fostering resilience in managed ecosystems requires adaptive management that considers many sources of variability and change (Ostrom and others 2007).

However, adaptive management has not been implemented in many places simply because it is too expensive and time-consuming to conduct everywhere. Land managers face challenging problems that require adaptive management, but lack the resources to study an entire region, or to select research and mitigation sites haphazardly. We have described an approach that greatly increases this efficacy by identifying where research is most likely to uncover important understanding about resilience and vulnerability, where mitigation will yield the most benefit, and thus where managers could apply adaptive management for optimal outcomes. Approaches such as ours that identify locales and interactions of concern allow for the focused experimentation and learning needed to estimate thresholds, reduce uncertainty, and build adaptive capacity to unprecedented changes. To this end, transparent and place-specific applications of theory can help decision-makers and practitioners address vulnerability in managed ecosystems.

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Author Contributions

Conceived of or designed study (CMB, TMP, FSC); Performed research (CMB, TMP); Analyzed data (CMB); Contributed new methods or models (CMB, TMP); Wrote the paper (CMB, TMP, FSC).

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