

Prioritizing Ecosystems, Species, and Sites for Restoration

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12.1 Introduction

Ecological restoration is ‘the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed’ (SERI 2004), and is often interpreted as guiding an ecosystem back to an historic reference condition (Egan and Howell 2001) or, somewhat more equivocally, to an historic trajectory of change. From the perspective of biodiversity conservation, restoration ideally will result in an assemblage of species that is well adapted to current and anticipated future site conditions; is diverse (in terms of composition, structure, and function); contains viable populations of species of conservation concern; provides ecosystem services; and is resilient under current and potential future conditions.

Although common definitions of restoration suggest human action, restoration options fall along a continuum from passive to active. In passive restoration, or ‘benign neglect’ (Zahner 1992), we leave a site or landscape alone to heal itself through ecological succession, soil building, and colonization of the area by species that had been extirpated directly or indirectly by humans. Landscapes sometimes recover remarkably well without human meddling, especially if they are large enough to incorporate natural disturbance regimes and other ecological processes. Nevertheless, one should not assume that a landscape with regrown vegetation has, in fact, recovered its native biodiversity and ecological integrity. Vegetation structure, for example, may be abnormal (e.g. stunted) and many native species

may be missing. Natural recolonization of highly disturbed sites by native species may depend on an uncommon coincidence of seed availability, favourable conditions for recruitment, and an absence of competing non-native species (Standish et al. 2007). Herbaceous floras of forests may take centuries to recover after intensive logging or, especially, agriculture (Duffy and Meier 1992; Bellemare et al. 2002; Finn and Vellend 2005). Where profound physical and chemical changes in soils have occurred, human land-use effects may be essentially irreversible. Forests in Western Europe that grew back on Roman agricultural sites from 2000 years ago still show differences in soil nutrients, pH, and plant species composition and diversity (Dupouey et al. 2002; Dambrine et al. 2007). Given increasing documentation of dispersal limitation in ecological communities, it is reasonable to assume that many species with limited dispersal capacity (e.g. ant-dispersed herbs, flightless insects) will never return to isolated sites in fragmented landscapes without human assistance (Brudvig and Mabry 2008).

In many cases some degree of active restoration is needed to guide an ecosystem towards recovery. Active restoration might be as simple and relatively cheap as restoring a characteristic fire regime through prescribed burning (Van Lear et al. 2005). In other cases, however, more intensive management is needed, for example, thinning small trees from overgrown forests, removing woody vegetation that has invaded grasslands, stabilizing eroded slopes, obliterating and revegetating roads, filling canals and ditches, placing large logs into streams in

logged watersheds with trees too small to contribute coarse woody debris, removing invasive exotic species, and reintroducing extirpated native species (Noss and Cooperrider 1994; Friederici 2003). Active restoration can be very costly. Large projects, such as the restoration of the Kissimmee River and Everglades in Florida or the San Francisco Bay delta in California, can each cost many billions of dollars (Holl et al. 2003). In addition, a significant number of restoration projects fail to meet their objectives (Vallauri et al. 2002; Wilkins et al. 2003).

Whether restoration is active or passive, prioritization is necessary to make the best use of available resources. There are too many degraded sites to restore them all, given limitations in funds, manpower, and political clout (i.e. powerful economic interests may oppose 'non-productive' uses of land). Too many populations of species have been extirpated or reduced severely in numbers to recover them all and manage their populations in perpetuity. And too many kinds of ecosystems – forests, grasslands, wetlands, streams, and lakes – in any region have suffered severe declines and degradation. On which species, ecosystems, and sites do we focus our efforts? The answer, in large part, depends on one's goals. The set of sites selected for restoration will depend on the overall conservation goals, which should be stated explicitly, and on the specific criteria used to prioritize sites and measure progress towards goals. In this respect, prioritization for restoration is no different from systematic conservation prioritization generally (Margules and Pressey 2000; Chapters 1 and 3).

Nevertheless, unlike systematic conservation planning as usually defined, the literature of restoration ecology contains relatively few examples of prioritization using site-selection algorithms and based on such principles as complementarity and efficiency (but see Westphal et al. 2003, 2007; Crossman and Bryan 2006). Rather, most priorities in restoration have been determined through the use of expert opinion and ranking or scoring methods that apply generic design criteria such as site size or proximity, or through analysis of economic costs versus economic, social, and ecological benefits (Llewellyn et al. 1996; O'Neill et al. 1997; McAllister et al. 2000; Gkaraveli et al. 2004; Cipollini et al. 2005; Petty and Thorne 2005).

In quantitative conservation prioritization, the network of sites selected by an algorithm to meet conservation goals might vary along a condition spectrum from virtually pristine to highly degraded. In many landscapes most of the selected sites will be far from pristine, many of the target species present in less than viable populations, some species historically present now missing, and key ecological processes dysfunctional. A logical response of managers to this situation is triage: concentrate restoration efforts on those moderately degraded sites that can be restored relatively easily and cheaply, with a high perceived chance of success, and ignore the completely 'trashed' sites (which may be impossible to restore) as well as those in relatively good condition. The critical assumption in the triage approach is that highly degraded sites have crossed thresholds of irreversibility (Palik et al. 2000). A similar triage could be applied to populations. Triage in site selection, however, could result in spatially and functionally disconnected sites, in turn precipitating significant further losses of biodiversity and ecological function. Quantitative methods for optimizing the configuration of restored sites across a landscape have only recently been explored (Crossman and Bryan 2006; Westphal et al. 2007; Chapter 3). Moreover, a persistent problem with triage, and with application of threshold concepts generally, is that irreversible damage is often assumed with little evidence, which may result in important sites being abandoned to development or continued degradation (Bestelmeyer 2006). Given the complexity and uncertainty surrounding non-linear ecosystem dynamics, alternative stable states, and hysteresis (Mayer and Rietkerk 2004; Ives and Carpenter 2007; Strange 2007), triage must be applied cautiously and should never be used to consign species to extinction (cf. Marris 2007).

In this chapter we review approaches to prioritization of ecosystems, species, and sites for restoration; discuss methods of assessing costs, benefits, and scheduling of restoration actions; suggest the need for new methods; and highlight the challenge of restoring the composition and structure of ecosystems into a dynamic and uncertain future. Although prioritization as discussed in this book is primarily spatial and, therefore, focuses on sites, we argue that prioritization of species and ecosystems

is just as important – and should precede – the prioritization and selection of sites in a reserve design. For example, prioritization of species and ecosystems is necessary to develop defensible quantitative goals for site selection and to determine the optimal configuration of sites. Although there is a growing theoretical basis for identifying species and ecosystems of high importance (e.g. highly interactive species, vegetation types crucial to the global carbon budget, species and assemblages likely to be most resilient or adaptable to climate change), quantitative methods for prioritization of species and ecosystems are limited. This is a key area where ecological and evolutionary theory must be made applicable to an urgent conservation problem.

12.2 Priority ecosystems

'Ecosystems', considered practically for conservation planning, are vegetation types and habitats discrete enough to be mapped (Noss 1996). In addition to plant communities at some level of classification hierarchy, ecosystems include aquatic, terrestrial, and subterranean habitats defined by physical (abiotic) parameters such as landform, geology, soils, elevation, aspect, and other characteristics (Kintsch and Urban 2002; Noss et al. 2002; Lombard et al. 2003). Prioritization of ecosystems for protection historically has been based on rarity, attributes of the species assemblage (e.g. richness, endemism), and level of threat. The same criteria could be applied to prioritization of ecosystems for restoration. However, we argue that more defensible criteria for prioritization for restoration are extent of decline in area or quality attributable to human action (e.g. since European settlement), contribution of the ecosystem type to broad-scale ecological processes (e.g. the global carbon cycle, regional-scale disturbance or hydrologic regimes, nutrient transport), and risk of further degradation (which might push a system past a threshold of resilience and into an alternative and less desirable stable state; Beisner et al. 2003).

Probably the most widely used prioritization system for ecosystems in North America is one developed by The Nature Conservancy, which ranks species and ecological communities (i.e. ecosystems) at global and sub-global (e.g. nation,

state, province) scales. This system, managed by NatureServe (<http://www.natureserve.org/>), is designed to take into account many factors that determine vulnerability, but in practice rankings are based primarily on rarity, as this parameter can often be estimated with greatest confidence. For example, a species or community is generally considered Critically Imperiled (G1 at the global scale) if there are 5 or fewer occurrences, Imperiled (G2 at the global scale) if there are 20 or fewer occurrences, and so on. Thus, G1 species are almost entirely narrow endemics; similarly, G1 communities have very restricted distributions. It is important to recognize that many of these species and communities have been rare throughout their evolutionary histories and are not necessarily at greater risk of disappearing today than formerly.

For ecosystems, where function (including ecosystem services; Christensen et al. 1996) may be as important as species composition, a decline in area or quality is arguably a more defensible criterion for prioritization than rarity. The massive decline of a once-dominant regional ecosystem is usually more worrisome in terms of biodiversity reduction and alteration of major ecological processes than is the loss of a localized plant association. Hence, Noss et al. (1995), in a review of the endangered ecosystems of the United States, emphasized the extent of areal decline (i.e. conversion) and degradation in quality (ecosystem composition, structure, and function; Noss 1990) in prioritizing ecosystem types for conservation and restoration. In the United States, major vegetation types that once dominated entire regions – for example, tallgrass prairie, longleaf pine (*Pinus palustris*) savannas, ponderosa pine (*P. ponderosa*) forests – but have been converted to agriculture or to alternative vegetation states due to fire exclusion or other disruption of ecological processes, stood out as primary endangered ecosystems (Noss et al. 1995).

When setting representation targets for ecosystems, it therefore makes sense to consider the historical extent of an ecosystem in addition to its current area (Strittholt and Boerner 1995). Representation targets for severely depleted ecosystems may appropriately be set at considerably more than 100 % of their current area, assuming

that restoration of the ecosystem on lands it formerly occupied is feasible. Curiously, few priority-setting schemes in ecological restoration have applied extent of decline as a criterion. Palik et al. (2000) recognized the logic of assigning priority to ecosystems that were historically abundant but are currently rare; paradoxically, however, their prioritization index gave precedence to currently rare ecosystems that were also historically rare, as well as to the least disturbed examples of these ecosystems (which would presumably be less costly to restore). In general, the restoration literature emphasizes the assignment of priority to sites or management actions rather than to ecosystem types or species.

Areal decline can, in principle, be easily computed as the per cent change from historic (e.g. presettlement) to present area. Nevertheless, the lack of accurate maps of historic vegetation and other habitat distributions, along with problems relating to disparate resolution and data quality, often make such calculations highly uncertain. Historic distributions of ecosystems can be modelled, however, based on historical accounts and known affinities of vegetation with soils, climatic envelopes, and other abiotic parameters (Strittholt and Boerner 1995). Indeed, Palik et al. (2000) found that the identity of ecosystems in their study landscape was highly predictable based only on geomorphic and soil variables, which allowed them to map the potential reference distribution of various ecosystems across the landscape. The extent of ecosystem degradation (qualitative decline in composition, structure, or function) can be estimated by comparison of current conditions at various sites to historic or current reference sites. A large literature has developed on the use of quantitative indices, for example, index of biotic integrity for streams (Karr 1991), floristic quality index for plant communities (Lopez and Fennessy 2002), and bird community index (O'Connell et al. 2000) to assess the departure of local communities from reference conditions. Sorely needed, however, are synthetic measures that take into account the conversion and degradation of multiple communities on a landscape scale – the scale at which restoration programmes must increasingly focus. Also needed are more reliable quantitative measures of the importance of ecosystems in terms of large-scale

ecological processes and the provision of ecosystem services.

Little work has been done to quantify the relative functional importance of particular ecosystem types within the broader mosaic of ecosystems that constitute the regional landscape. Using extent of decline in area or quality as a primary criterion for prioritizing ecosystems for restoration carries the assumption that massive declines generate commensurate losses in functionality. The dominant (e.g. matrix) ecosystem in a natural landscape determines the character of that landscape and probably controls the majority of ecological processes relevant to biodiversity, for example, the disturbance regime, hydrology, and carbon sequestration. Nevertheless, the combination and pattern of ecosystems in a heterogeneous landscape may have emergent properties of functionality; hence the concept of functional landscapes (Noss 1987; Poiani et al. 2000). Animals such as amphibians that use distinct habitats during different phases of their life cycle, and ungulates that migrate seasonally from winter to summer ranges, require a combination of ecosystem types within their normal movement range and without significant barriers in between. Moreover, analogous to the keystone species concept, some relatively uncommon ecosystem types may have a disproportionate influence on landscape structure and function. These 'keystone ecosystems' (deMaynadier and Hunter 1997) may exert such influence on landscapes by shaping disturbance regimes or providing limiting resources to a large number of species. Examples include riparian ecosystems, barrier beaches, disturbance-created gaps of various sizes, and springs. We are aware of no attempts to quantify the functional contribution of suspected keystone ecosystems to landscape dynamics.

Given uncertainty about the relative functional roles of ecosystems in a landscape, a prudent goal for restoration is to mimic the natural relative cover values and spatial juxtaposition patterns of ecosystem types across a landscape to the extent feasible. 'Natural' in this case refers to an historic reference condition that maintained the full complement of native biodiversity. Reference conditions are defined within a range of variability on the basis of historical accounts and/or from extant sites that

remain in relatively high-quality condition (Moore et al. 1999; Egan and Howell 2001; SERI 2004). Innovative modelling approaches to estimating historical ranges of variability include the study by Wimberly et al. (2000), who used a 'landscape age-class demographic simulator' to integrate dendroecological and paleoecological data and estimate historical variability in the proportion of old-growth and late-successional forests in a forest landscape with a simulated historical fire regime. In the case of rare ecosystems, however, information sufficient to reconstruct historical reference conditions may be scarce (Brudvig and Mabry 2008), with remaining reference areas often being protected areas (Sinclair et al. 2002; Scholes and Biggs 2005) that are typically biased in their representation of conditions throughout a region (Margules and Pressey 2000; Scott et al. 2001; Hansen and Rotella 2002). This is not the place to discuss the extensive philosophical and theoretical controversies surrounding the use of historic or extant reference conditions and ranges of variability (Landres et al. 1999) to guide ecosystem reconstruction, but it is clear that future approaches that incorporate ecosystem function and future threats such as climate change are needed to effectively prioritize ecosystems for restoration.

One of the challenges of comprehensive planning is integrating prioritization across ecosystem types. As a preliminary example of the integration of forest, stream, and estuarine restoration priorities, we focus on the Salmon, Siletz, and Yaquina River catchments on the central Oregon (United States) coast, which comprise 178,000 ha of private and public lands. Wimberly et al. (2000) estimated historical means of late-successional forests at 66–76 per cent of the landscape. Using estimates of tree size from Ohmann and Gregory (2002), we identified late-successional forests on the central Oregon coast as stands with large (50–75 cm diameter) and very large (>75 cm) trees. Approximately 30,900 ha of late-successional forests (~25 % of average historic condition) remain in the three catchments (Figure 12.1a).

Linke et al. (2007) assessed the condition and vulnerability of sub-catchments to identify river restoration priorities. We quantify current conditions of

sub-catchments as the ratio between per cent area covered by late-successional forests (presence of large and very large trees) and a 70 % benchmark based on estimates from Wimberly et al. (2000) (Figure 12.1b). Sub-catchment condition is significantly lower than historic forest conditions on private lands (Figure 12.1c). To quantify vulnerability of sub-catchments, we estimated per cent area in private ownership (Figure 12.1d).

Delivery of large wood to streams and estuaries is a critical ecosystem process that provides habitat for numerous aquatic species, including salmonid fishes. Burnett et al. (2007) identify stream reaches with high potential to develop juvenile coho (*Oncorhynchus kisutch*) and steelhead (*O. mykiss*) rearing habitat; both species have high ecological, economic, and cultural value, and both are in serious decline. Some 534 km of streams provide high-potential coho habitat and 606 km of streams provide high-potential steelhead habitat in the study area. Each of the three catchments has an estuary, and all are in need of some level of restoration (Jennings et al. 2003).

We visually identified 36 sub-catchments for prioritization of multi-ecosystem restoration (Figure 12.2), constituting 33 % of the 109 sub-catchments, 31 % (55,200 ha) of the study area, and 51 % (15,900 ha) of the areas with large trees. Prioritization of sub-catchments for restoration was based on the selection of sub-catchments in moderate to good condition (condition ≥ 25) and low to moderate vulnerability (vulnerability ≤ 80). Additional sub-catchments were added to extend restoration areas to the headwaters of major stream branches to increase connectivity (Linke et al. 2007). Condition of selected sub-catchments averaged 45 % of historic late-successional area (10–106 %), compared to 25 % for all sub-catchments. The restoration sub-catchments contain high-potential salmonid habitat: 136 km (25 %) of coho and 220 km (36 %) of steelhead. Restoration of these sub-catchments is therefore likely to have the greatest benefit to salmon. Further prioritization exercises in this study region should include application of site-selection algorithms and inclusion of marine features, when data become available.

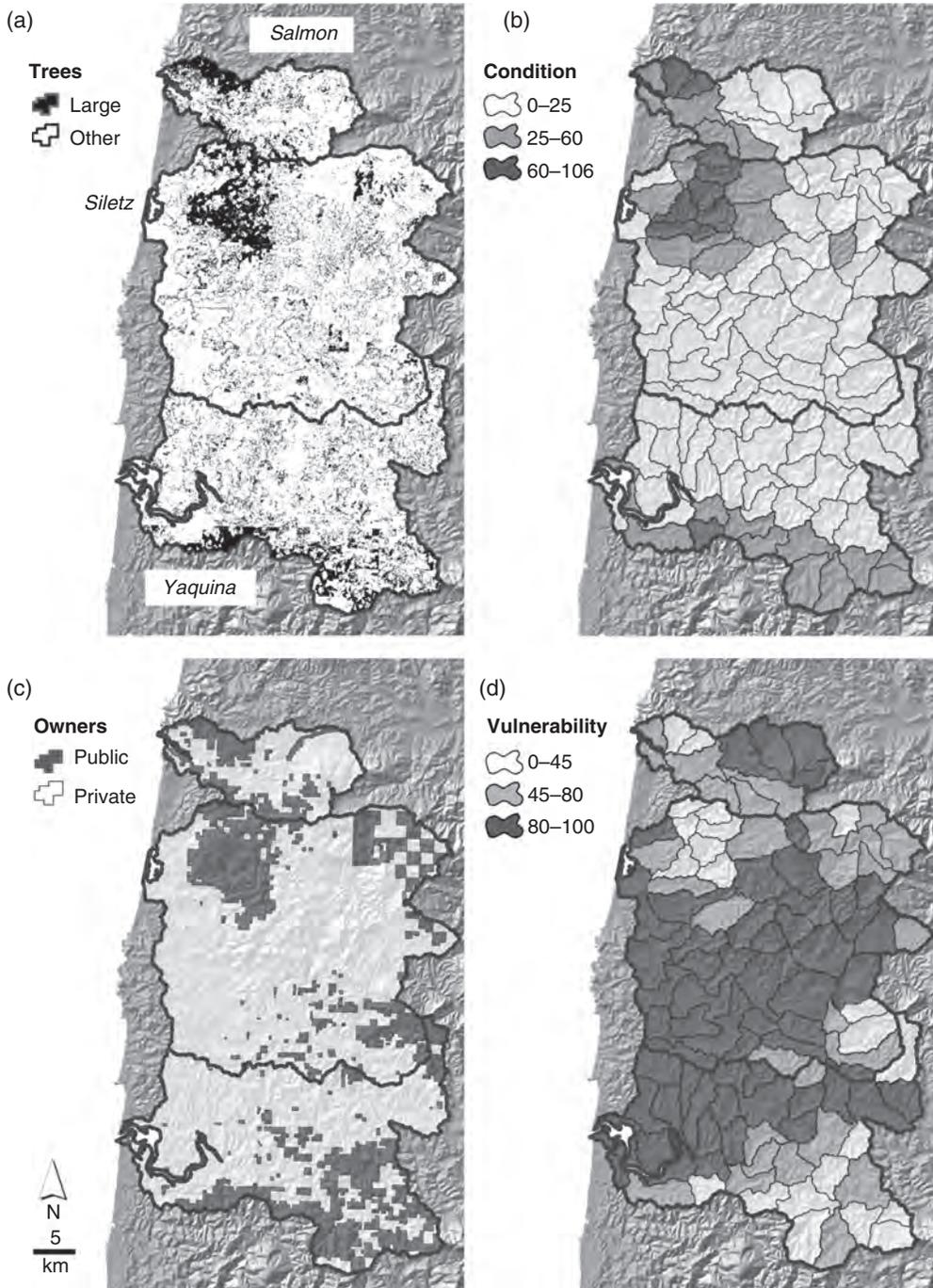


Figure 12.1 Salmon, Siletz, and Yaquina River catchments. (a) Forest stands with large or very large conifers. (b) Sub-catchment condition as the per cent area with large or very large conifers compared to the estimated 70% average extent of such forests. (c) Public and private ownership in the study area. (d) Sub-catchment vulnerability measured as the per cent of sub-catchment area in private ownership.



Figure 12.2 A prioritization of sub-catchments for restoration of forest, stream, and estuarine ecosystems in the study area.

12.3 Priority species

Ecological restoration aims to return ecosystems to their historic trajectories of change, which usually begins with restoring the system to something approximating an historic reference condition. This restoration requires the repopulating of degraded sites with the species that presumably inhabited such sites before disturbance by modern humans. Many combinations of species are possible, with composition and richness determined by, among other things, the site's area, isolation, and specific abiotic conditions. Given that the potential species pool for a site undergoing restoration is enormous, how do we decide what species to maintain on or actively reintroduce to the site?

Theoretical ecologists find restoration ecology interesting because it provides an opportunity to address assembly rules, that is, the factors that determine which species can coexist on a site and the

order in which they may be added to the assemblage (Diamond 1975a; Weiher and Keddy 2001). This controversial body of theory has obvious bearing on ecological restoration, yet empirical testing of assembly rules in the context of restoration projects has only just begun (Temperton et al. 2004). Functional roles of species in ecosystems, species traits related to such roles, and dispersal limitations, which will determine which species must be actively reintroduced to a site, need to be considered. Plant traits can significantly affect their ability to colonize sites and to thrive on sites being restored. For example, in grassland restoration in Britain, the ability of plant species to establish and persist depended on gap colonization ability, competitive ability, vegetation regeneration ability, generalist habits, and association with fertile soils (Pywell et al. 2003). In general, dispersal limitations of particular species represent a significant obstacle to passive restoration, both in terrestrial ecosystems (Kirkman et al. 2004) and in aquatic ecosystems such as streams, where refugia from disturbances such as floods and drought are critical to the process of passive restoration (Lake et al. 2007).

In an exemplary trait-based approach to species prioritization for restoration, Brudvig and Mabry (2008) considered species traits to produce a list of plant species for initial reintroduction to several oak savannas in Iowa (United States). Species traits (exotic versus native status, wetland indicator status, life form, seed-dispersal mode, seed mass, degree of association with intact high-quality habitat [conservatism], and light versus shade affinity) were used as filters to reduce a regional species pool of 893 species to 111 species that were targeted for active reintroduction. These species included native upland plants with passive or ant-dispersed seeds or heavy wind-dispersed seeds. The species considered to be of highest priority for reintroduction were shade-intolerant native perennial grasses and forbs with conservative habitat requirements. These species were considered unlikely to colonize the restoration sites on their own, but were suspected to fill under-represented niches and play important functional roles in the ecosystem (Brudvig and Mabry 2008).

The functional roles of particular species in ecosystems should be a crucial consideration in conservation and restoration, especially given the enormity of anthropogenic stresses and rapid environmental

(e.g. climatic) change. Reviews of recent empirical and theoretical research on the diversity–stability relationship demonstrate persuasively that ecosystems are generally more stable (resistant and/or resilient) when they contain a diversity of functional groups and high richness and redundancy of species within functional groups (Loreau et al. 2001; Elmqvist et al. 2003; Hooper et al. 2005; Ives and Carpenter 2007). Plant functional types based on species traits can be associated with specific ecosystem services, such as provision of livestock fodder and reduction in landslide risk (Quétier et al. 2007). Compensatory mechanisms among plant species in restoration projects may result in the aggregate biomass of species being more stable than the biomass of individual species in response to disturbance or stress (Seabloom 2007).

The role of key species versus diversity itself in maintaining stability is still being debated, but as Ives and Carpenter (2007:62) suggest:

a finding common to many empirical studies is that the presence of one or a handful of species, rather than overall diversity of an ecosystem, is often the determinant of stability... depending on the types of stability and perturbation, different species may play key roles... [therefore] each ecosystem might have to be studied on a case-by-case basis. In the face of this uncertainty and our ignorance of what the future might bring, the safest policy is to preserve as much diversity as possible.

Certain kinds of species stand out, however, as especially critical to the maintenance of diversity and stability within ecosystems. These can be generally referred to as highly interactive species (Soulé et al. 2003). Keystone predators, for example, have been demonstrated to enhance diversity of lower trophic levels in a wide array of ecosystems (Paine 1966; Terborgh et al. 2001; Sergio et al. 2005; Dobson et al. 2006). Reintroducing large carnivores to landscapes undergoing restoration, assuming the presence of adequate prey and security from humans, may help stabilize food webs and prevent large herbivores from decimating vegetation structure and eliminating sensitive plant species and birds that depend on a certain vegetation structure (Berger et al. 2001; Ripple and Beschta 2007). If we expand the keystone concept to encompass all species that play pivotal roles in ecosystems, disproportionate

to their abundance (Power et al. 1996), ecosystem engineers such as beaver and burrowing vertebrates, among many other taxa (Boogert et al. 2006) are priority candidates for restoration. Restoring the gopher tortoise (*Gopherus polyphemus*) to sites in the southeastern United States may enable hundreds of commensal species to exist on sites where they would otherwise be excluded due to lack of burrow habitat (Jackson and Miltrey 1989). By definition a keystone species is often the sole member of a functional group in an ecosystem; hence, if it is missing from a restored site, that function will not be fulfilled. Also of critical importance are dominant or foundation species (Ellison et al. 2005), another class of highly interactive species, which determine the structure and overall character of an ecosystem. Restoring all highly interactive species to ecologically effective population sizes – rather than just minimally viable populations – should be a critical component of any restoration programme (Soulé et al. 2003, 2005).

Nevertheless, knowledge is limited on the relative importance of species in food webs or other ecological interactions and, especially, how to quantify such importance. Most work thus far has been theoretical, conceptual, or natural historic, but the ground has been laid for more quantitative consideration of species priority, at least in terms of relative ranking. For example, research has revealed the architecture of mutualistic species networks in a number of communities and has shown that some species have a much larger number of interactions, or stronger interactions with other species, than others, and that interaction networks are nested, that is, specialists interact with subsets of species with which generalists interact (Bascompte et al. 2003). Quantifying the cumulative interaction strengths of various species would provide much-needed guidance to prioritizing species for restoration. However, we should not be surprised if the interaction strengths and number of interactions of particular species vary depending on their ecological context – that is, they are community or site-specific – hence, it might be unreasonable to expect practical guidance for restorationists from this body of theory anytime soon.

In some cases there may be conflicts between restoring one species over another. For example, in

the Florida Everglades, two federally Endangered species, the snail kite (*Rostrhamus sociabilis*) and the wood stork (*Mycteria americana*) require very different hydrologic conditions (in particular, water level) because of differences in prey and foraging behaviour (Takekawa and Beissinger 1989). In this heavily managed ecosystem, managers are often in a quandary over which species to manage the system for. The recovery of particular species can also conflict with the restoration of an ecosystem, a compelling example being the Mexican spotted owl (*Strix occidentalis lucida*) versus ponderosa pine forest in the southwestern United States (Noss et al. 2006). The owl requires relatively dense stands within the generally more open-canopied ponderosa pine landscape (Beier and Maschinski 2003). Anthropogenic fire exclusion has increased owl habitat in some areas, while also increasing the risk of uncharacteristically severe fires, threatening human lives and property. Fuel-reduction treatments, however, could eliminate nesting habitat for the owl. The key to resolving most of such conflicts is to analyse and manage habitat conditions on a broad regional scale, where multiple objectives can be met simultaneously but on different sites (Noss et al. 2006). In the case of the Mexican spotted owl and ponderosa pine forest, detailed spatial analysis shows that, while real conflicts do exist, they have been exaggerated and can be resolved through well-informed landscape-level management (Prather et al. 2008).

12.4 Priority sites

Priority sites for restoration follow, albeit not necessarily automatically, from the identification of priority ecosystems and priority species. These ecosystems and species are features for which high goals (targets) can be set in the site-selection algorithms discussed throughout this book. Priority sites for ecosystem restoration would constitute, in part, those portions of the landscape where high-priority ecosystems are located, where they might contribute most to landscape ecological functions or ecosystem services, and where they can be restored most cost-effectively. They might also constitute areas that capture under-represented segments of the range of variation (e.g. along an elevation or soil-moisture gradient) within a classified ecosystem

type, or important edaphic interfaces or ecotones between ecosystem types, which are often biological and evolutionary hotspots (Pressey et al. 2003). Priority sites for species restoration (e.g. population recovery, augmentation, or reintroduction) can be identified through a combination of habitat or population models for particular focal species with site-selection algorithms (Carroll et al. 2003, 2006). For most real-world cases we see no need to separate the process of site selection for conservation from site selection for restoration, as restoration is a subset of conservation that addresses sites on the more degraded portion of the 'pristine to trashed' spectrum. For instance, in the Great Sand Hills of Saskatchewan, prioritization of conservation areas to protect sensitive grassland birds, small mammals, and vascular plants from gas development also identified degraded sites where restoration of gas well pads would be expected to benefit species of concern most positively (Great Sand Hills Scientific Advisory Committee 2007).

For identifying priority sites for species reintroduction, one common approach for well-studied species is the extrapolation of habitat models from reference populations to regions where the species has been extirpated (Mladenoff et al. 1995; Carroll et al. 2006). Even better, when population size for a reference area is known and at carrying capacity (perhaps in a protected area), Boyce and McDonald (1999) suggest a method for estimating potential density and population size for other sites, including areas for species reintroduction. For example, Boyce and Waller (2003) estimated a restoration target (potential carrying capacities) of 321 grizzly bears (*Ursus arctos*) in the Bitterroot region of central Idaho and western Montana where grizzly bears had been extirpated since 1932. Over larger regions these approaches may prove useful in prioritizing restoration sites that most efficiently meet a population target based on population viability assessments for the species. Critical to the success of such approaches is the effectiveness of habitat or distribution models in predicting out-of-space and/or out-of-time conditions, which frequently falter when applied to novel environments (Guisan et al. 2006; Moilanen et al. 2006b; Dormann 2007).

Prioritizing restoration efforts within extant ranges of declining species is also a common

conservation problem. Although mechanisms, such as habitat or mitigation banking, cap-and-trade, and 'no-net-loss' principles are increasingly being considered for natural resource management and conservation (Minns 1997; Bonnie 1999), details on how to most effectively identify and prioritize restoration sites, account for time lags, and determine appropriate offset ratios (area restored per unit

area impacted) are less well agreed on (Minns 2006; Morris et al. 2006). Nielsen et al. (2006) demonstrated an approach to identifying restoration sites for a declining grizzly bear population in Alberta, Canada. Source and sink (caused by roads) habitats were predicted for a 132,076 km² area of the Canadian Rockies and Foothills (Figure 12.3) with a no-net-loss principle suggested. Using this approach,

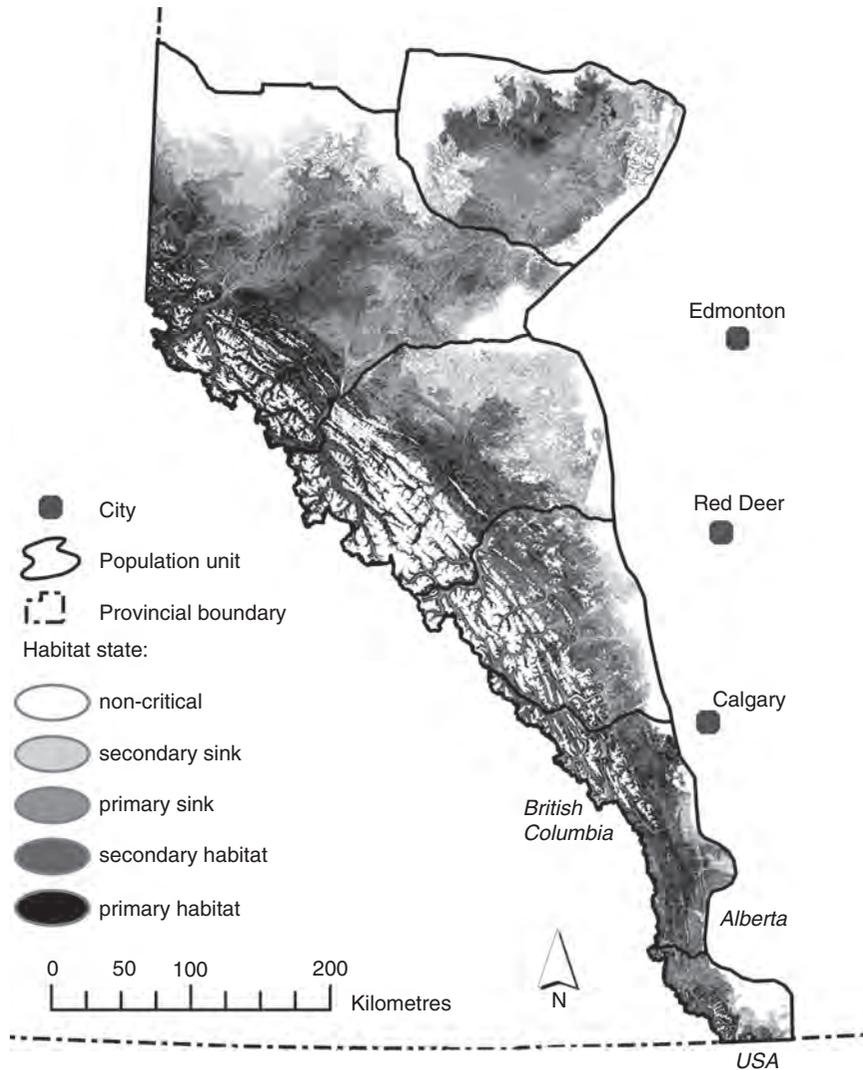


Figure 12.3 Grizzly bear habitat states for a 132,076 km² area along the east slopes of the Canadian Rockies in Alberta, Canada (~55 % of grizzly bear range in Alberta). Habitat states are helping to prioritize conservation and restoration areas, as well being suggested as a method for managing habitat offsets in a cap-and-trade or no-net-loss conservation framework.

Source: Modified from Nielsen et al. (2006).

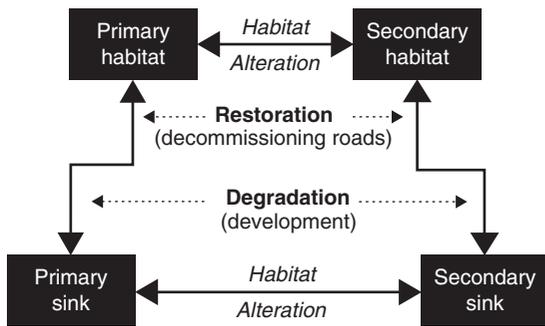


Figure 12.4 Conceptual diagram of the no-net-loss or habitat banking framework for grizzly bear management in Alberta, Canada.

Source: Modified from Nielsen et al. (2006).

developments that reduced habitat quality (forestry, oil/gas, coal mining, etc.) would necessitate restoration through decommissioning of roads elsewhere (Figure 12.4). It is unlikely, however, that offset ratios would be 1:1; landscape configuration also requires consideration. As an alternative to naive offset ratios, Moilanen et al. (2008b) suggested a method to estimate ‘robustly fair’ offset ratios that account for temporal changes in habitat quality and restoration uncertainty.

In contrast to compensatory actions that attempt to maintain existing levels of habitat, a frequent conservation planning problem is the identification of potential conservation sites that maximize habitat and minimize restoration costs (Westphal et al. 2007). For instance, in Alberta, Canada, grizzly bear conservation areas for each of seven management units have been proposed with the intent of minimizing human access to no more than 0.6 km/km² of roads. An objective function that maximizes habitat and minimizes restoration cost can be used to prioritize areas best suited for conservation protection, while also identifying restoration sites and associated costs (Westphal et al. 2007). In the case of grizzly bear conservation areas in Alberta, Figure 12.5 demonstrates the location of a conservation area that maximizes habitat and minimizes restoration costs using simulated annealing in MARXAN (Ball and Possingham 2000a; Possingham et al. 2000) for one population unit near Calgary, Alberta. Total restoration cost for the 2,400 km² conservation area was estimated at US\$157,970.

As noted earlier, most prioritization exercises in ecological restoration address the identification of priority sites and priority management practices. Most have employed expert-based scoring and ranking systems, sometimes aided by more formal decision analysis (Cipollini et al. 2005), which helps to quantify expert opinion and make it more transparent. Nevertheless, few restoration prioritizations have employed objective site-selection algorithms as addressed in this book. Even those approaches that use spatially explicit decision-support models to determine restoration priorities (Twedt et al. 2006) are often highly arbitrary in their scoring procedures and fail to apply well-accepted principles such as complementarity, that is, ‘individual patches (or actions) cannot be ranked in isolation because their value depends on other decisions made in the landscape – the whole is more than the sum of the parts’ (Possingham and Nicholson 2007).

In the interest of space, we do not review the various scoring and expert-based approaches to restoration prioritization here. Rather, we note, briefly, some recent forays into quantitative algorithm-based prioritization of restoration sites. As a first example, Westphal et al. (2003) examined metapopulation viability of an endangered Australian bird, the Mount Lofty Ranges Southern Emu-wren (*Stipiturus malachurus intermedius*). They considered three management options: expanding existing habitat patches, connecting patches by corridors, and creating new patches through restoration. To evaluate these options they used Markov chains to model metapopulation dynamics and stochastic dynamic programming to find optimal spatial solutions. They found stochastic dynamic programming especially useful in determining the optimal sequence of management actions. This study is a good example of the uncommon but useful approach of combining population viability analysis (Chapter 9) and algorithmic site-selection (Chapters 4–5; see also Carroll et al. 2003) as well as comparing protection, connection, and restoration options in a single exercise.

As another example of the application of site-selection algorithms to restoration, Crossman and Bryan (2006) worked in an increasingly familiar landscape setting – one in which conservation of remnant patches of natural habitat is insufficient to meet biodiversity conservation objectives; rather,

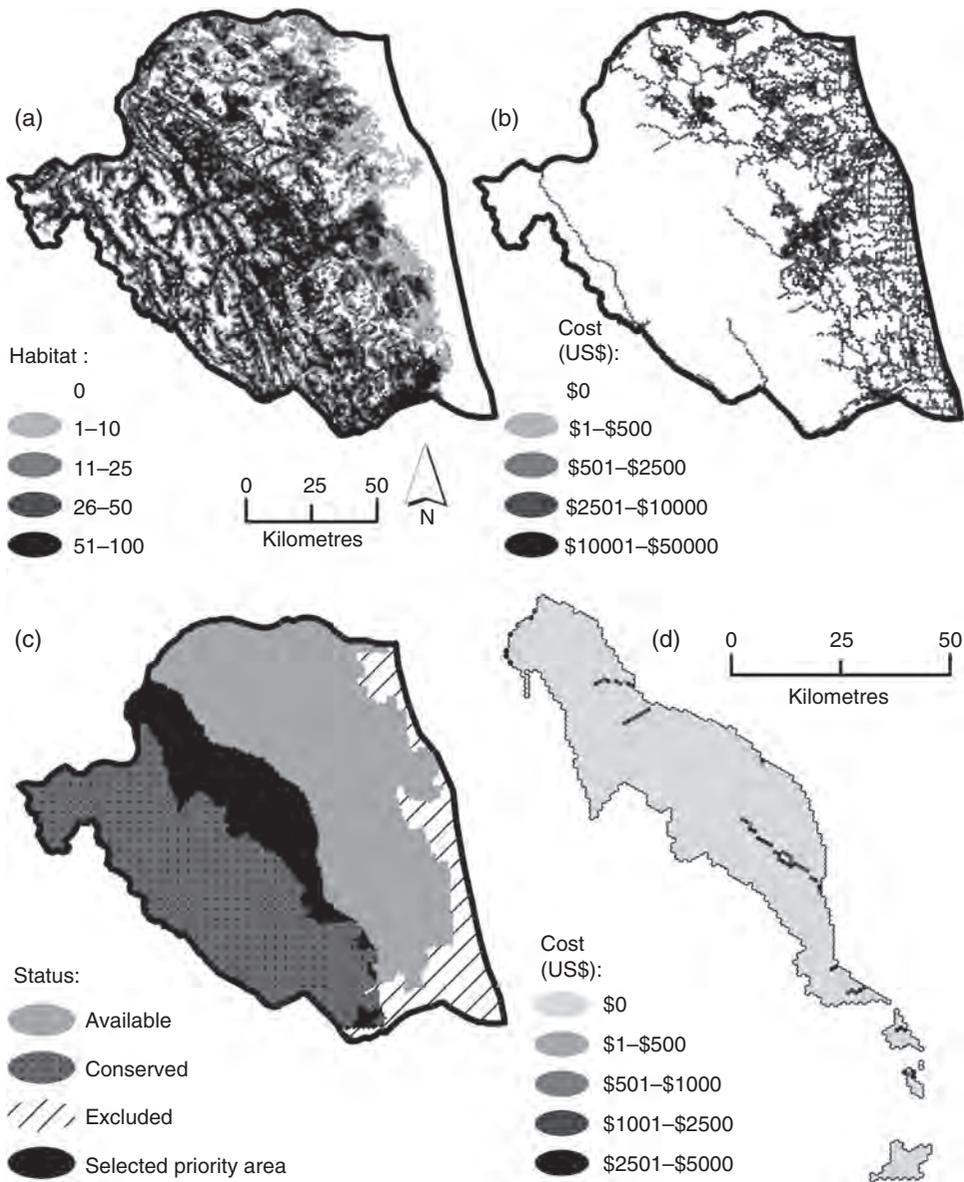


Figure 12.5 Selected grizzly bear priority areas and restoration costs associated with road decommissioning in the Clearwater management unit of Alberta, Canada. (a) Habitat scores for 1 km² hexagon-shaped planning units depict the ranked likelihood of habitat use from a resource selection function. (b) Potential restoration costs defined as the cost of decommissioning roads (US\$5,000 per km) in a planning unit in order to minimize road density to 0.6 km/km². Cost estimates (adjusted for inflation) from Harr and Nichols (1993). (c) Status of planning units used to define areas that were excluded (private lands in the east), already conserved (Banff National Park), available, and selected by MARXAN (maximizing habitat and minimizing restoration costs). (d) Total restoration cost within the selected priority area is estimated at US\$157,970 with planning units requiring restoration illustrated.

large-scale restoration is required. They used integer programming and applied common conservation planning principles (e.g. comprehensiveness, representation, efficiency, flexibility) to identify spatial priorities for restoration, but found that the 'optimal' solution was highly fragmented and likely to be dysfunctional ecologically. Applying a system of impedance surfaces to the solution improved landscape structure by making the network of sites more compact, contiguous, and connected by corridors. Importantly, addition of the impedance layer had only a minor effect on the cost of the solution, as measured in terms of area designated for restoration (Crossman and Bryan 2006).

As a final example, Westphal et al. (2007) combined a greedy heuristic iterative improvement algorithm with a simulated annealing algorithm to determine the optimal configuration of revegetated woodland sites across the Mount Lofty Ranges of South Australia. The goal was to select the suite of sites in the landscape that, if revegetated, would maximize the number of bird species, based on species distribution models. Various scenarios were modelled, including some that included assumptions on required patch size and connectivity (e.g. achieving 1,000 individuals per species and assuming that the utility of a revegetated site for a species is 0 if it will hold fewer than 50 individuals of that species). Because information was lacking on true restoration costs, consideration of costs was limited to acquisition. With the assumption that all sites have the same per area cost, the most efficient solutions contained revegetated patches ranging from 780 to 4,010 ha. Inclusion of information on property values reduced patch sizes and shifted selected sites towards the less expensive parts of the landscape. An important contribution of this paper was the clear demonstration that the network of sites selected for restoration will be different under varying assumptions of economic cost. Previous non-algorithmic restoration prioritizations based on cost-benefit analysis (Petty and Thorne 2005) are less compelling because the 'optimal' solutions in such cases are not, in fact, likely to be close to mathematically optimal solutions because alternative combinations of sites are not considered rigorously, that is, complementarity is ignored.

A lingering concern that deserves more attention in the prioritization of restoration projects – albeit we do not have space to address this concern in detail here – is their likelihood of success. Even with substantial funding, many restoration projects fail partially or completely in meeting their goals (Zedler and Callaway 1999; Wilkins et al. 2003). These failures undoubtedly reflect the relative youth of restoration ecology as a science, and of ecological restoration as a practice, but they also reflect an unwarranted optimism on the part of regulatory agencies that require mitigation for habitat destruction. Many failures arguably could be avoided by improved consideration of abiotic conditions, species traits, dispersal limitation, local adaptation, functional complementarity, and other ecological and evolutionary factors. Nevertheless, restoration failures highlight the need to prevent damage to ecosystems in the first place rather than assuming an arrogant 'don't worry, we can fix it' attitude.

12.5 Restoration challenges for a no-analog future

The historic reference condition model of ecological restoration is complicated by findings from paleoecology that many modern species assemblages actually came together under quite different climatic conditions than what exist today and are very different from what can be expected over the next century. As stated succinctly by Williams and Jackson (2007:475): 'No-analog communities (communities that are compositionally unlike any found today) occurred frequently in the past and will develop in the greenhouse world of the future.' In principle, we could restore species assemblages on particular sites to resemble the communities that are forecasted to exist by models of species' responses to shifting climate envelopes (Chapters 6 and 13). Although such models can be useful as a first approximation of potential future ranges (Pearson and Dawson 2003), they carry a high level of uncertainty, in part because they fail to consider dispersal limitations or interspecific interactions. The surprises likely to come with global warming will likely include shifts of species into different areas of their fundamental niches as interacting species (positive or negative) appear and disappear; the result may be species occupying

strikingly different realized niches from what they occupy today (Williams and Jackson 2007). Nevertheless, we believe it would be prudent for restoration projects to begin experimentally introducing species, and genotypes within species, from lower latitudes and altitudes. There will undoubtedly be tensions between the traditional goals for ecological restoration, based on historical reference conditions, and new goals based on adaptation to climate change (Chapter 13). This tension highlights the need for a thoughtful dialogue and debate that clearly articulates differences in underlying values, assumptions, and goals (Harris et al. 2006).

Given the uncertainty of future climatic conditions, we agree with Rouget et al. (2003) that attention to ecological processes in site selection for restoration will become increasingly important. Sites that, due to unique abiotic conditions such as serpentine soils, edaphic interfaces, or hot springs, are biological and evolutionary hotspots should be favoured because these conditions will persist even as species composition changes – and they will likely allow new biodiversity to be generated. A major concern is that dispersal limitations will prevent many species from moving rapidly enough to keep pace with climatic change. Conservationists may indeed need to take the interventionist

approach of transporting some species into newly favourable habitats. We recommend that this be done with caution, paying particular attention to maintaining functional groups and highly interactive species across sites. If species occupying key functional groups are dying out at previously occupied sites and not dispersing rapidly enough to colonize new sites, then biodiversity can be expected to decline globally unless we assist the colonization of these species (cf. Hunter 2007). Even for species that would be able to disperse rapidly enough to track shifting climatic conditions, habitat fragmentation may make dispersal into suitable habitats impossible. Maintenance and restoration of habitat connectivity at all spatial scales – especially upslope, up-latitude, coast–inland, and across other environmental gradients – is a sensible strategy under all potential future scenarios. Prioritizing such ‘climate change adaptation corridors’ thus becomes a primary challenge for coming decades.

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