Managing Landscapes for Vulnerable, Invasive and Disease Species

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Abstract

As the focus of conservation moves from protected areas to include working landscapes, management must accommodate multiple goals. In this context, management focused on individual species must be well justified. Conservation practices that target carefully selected individual species can complement pattern-based criteria needed to maintain species richness and ecosystem function by providing mechanistic rationale and quantifiable targets for conservation actions. However, most treatments of landscape design do not consider the compatibility of management for individual disease-causing species, invasive species and threatened species simultaneously. We summarize recommendations from landscape epidemiology and invasion management and evaluate whether they are congruent with general principles for vulnerable species’ protection. Many, but not all, broad strategies for controlling invasive and disease species appear compatible with strategies for protecting vulnerable species. Local circumstances, scale considerations and the relative importance of landscape and other interventions should guide management of trade-offs among conflicting goals.
Keywords: conservation; invasive species; landscape epidemiology; trade-offs; vulnerable species.

Introduction

Landscape design more than ever must target multiple goals as the focus of conservation moves from protected areas to include working landscapes (Daily 1997) and as fewer landscapes remain free of human activities (Vitousek et al. 1997). In this context, landscape management focused on individual species needs to be well justified. Yet individual species-based approaches remain important in conservation, invasive species management and disease control. We examine the rationale for landscape design targeted at individual species. We summarize landscape design recommendations from the fields of landscape epidemiology and invasive species management. Although invasive species include those that harm human health (Invasive Species Council 2001), the invasion literature deals relatively little with human diseases and their vectors. From an ecosystem services perspective, landscape management that effectively reduces human disease threats as well as other impacts of invasive species would be highly desirable (Kremen & Ostfeld 2006). By including landscape approaches to disease management from the medical and public health literature, we sought useful design principles across two literatures concerned with harmful species.

Management to control invasive and disease organisms usually occurs in landscapes that also harbour native and threatened taxa. To address whether simultaneous landscape management for invasive, disease and vulnerable taxa is even feasible, we compare common interventions aimed at suppressing undesired species with landscape management goals typical of vulnerable species conservation. Landscape management both for and against particular classes of species focuses on common key themes including habitat protection; biodiversity maintenance; and management of disturbance, connectivity and fragmentation. In most – but not all – of these areas, broadly defined management strategies for managing invasions and diseases appear compatible with strategies for protecting and enhancing native species.

Why base management on individual species?

Individual species approaches are widely used to both control undesired species and enhance desired species. Vectors that spread life-threatening
diseases like malaria and sleeping sickness can often be more successfully controlled with landscape-management than other techniques in many phases of an epidemic, especially over long time scales (Bos & Mills 1987; Utzinger et al. 2001; Campbell-Lendrum et al. 2005). Control and eradication of invasive species are also usually approached species by species, with emphasis on species that cause significant harm such as introduced mammals on islands (Atkinson 1989) and agricultural weeds (Bossard et al. 2000). Eradications of introduced species have protected scores of threatened taxa (Veitch & Clout 2002). Single-species management efforts are not problem-free, however, because focus on removal of individual species can sometimes lead to unexpected effects or neglect of broader ecosystem restoration goals (Zavaleta et al. 2001). In complex landscapes where ecosystems are under frequent threat of new invasion, management for ecosystem resistance as well as problem species removal could improve outcomes.

Single-species approaches are commonly pursued for vulnerable or threatened taxa, especially those unique to a particular landscape. Single or small groups of species are also a focus of landscape conservation when management for the species in question is expected to meet a range of other needs or defined goals (Noss et al. 1996; Kotliar et al. 1999; Freudenberger & Brooker 2004). The utility of such approaches, variously termed focal or indicator species approaches among others, has been debated for decades (e.g. Landres et al. 1988; Lindenmayer et al. 2002). Focal species approaches have been criticized for ad hoc species selection (Landres et al. 1988) and lack of evidence that conservation priorities based on focal taxa adequately protect other species and values (Simberloff 1998; Andelman & Fagan 2000; Lindenmayer et al. 2002; Smith & Zollner 2005). Methodologies range widely – ‘focal species’ may or may not need to possess umbrella, keystone, flagship and indicator traits or combinations thereof (Simberloff 1998), so the term only broadly refers to individual species whose needs are highlighted in conservation planning.

Limited findings on the performance of focal species approaches suggest that individual species targets can provide mechanistic rationales for landscape design to complement the pattern-based criteria on which species richness and ecosystem function targets are generally based (Sanderson et al. 2002; Coppolillo et al. 2004; Taylor et al. 2005; but see also Andelman & Fagan 2000). Landscape planning that targets both individual species and broader conservation goals might therefore provide the best and most measurable outcomes (Poiani et al. 1998; Dan Doak, personal communication). This area, however, deserves much more study.
Managing both for and against species

Broadly speaking, vulnerable species are thought to benefit from landscape features like corridors, large reserves, protection from anthropogenic disruption, and maintenance of natural disturbance regimes. Are these needs fundamentally different from those of invasive and disease organisms? While invasive species tend to share traits like generalist habits (Patz & Wolfe 2002, Marvier et al. 2004), early age of first reproduction, and small seed or offspring size (Rejmanek & Richardson 1996) that differ from traits typical of vulnerable species (specialist habits, restricted distributions, late age of first reproduction, few offspring, large adult size), exotic and disease species can sometimes move through habitat corridors intended to enhance habitat value for native species (Simberloff & Cox 1987; Loney & Hobbs 1991; Hess 1996). Some exotic species can invade relatively undisturbed forest understoreys or montane parks (Pauchard et al. 2003; Chornesky et al. 2005). Many species threatened in their home ranges are abundant or invasive where introduced, highlighting the limits to generalizing about biological differences between invasive and threatened taxa. However, some commonly suggested landscape management approaches for vulnerable taxa and against harmful species resemble one another – such as maintaining high biodiversity (Shea & Chesson 2002) and minimizing novel disturbances (Hobbs & Huenneke 1992).

Disease species management

Reviews of landscape epidemiology identify 34 diseases with known or suspected ties to landscape change (Patz et al. 2004) and innumerable environmental disturbances that trigger their emergence (McMichael 2001; Patz et al. 2005). For example, Lyme disease illustrates disease emergence driven by habitat fragmentation and biodiversity loss, while malaria exhibits complex responses to disturbance. Lyme disease, the most common vector-borne disease in the USA (Centers for Disease Control and Prevention 2000), has a complex transmission cycle involving many vertebrate hosts of varying competence (transmission ability) and environmental sensitivity. The abandonment of agriculture in the northeastern USA during the twentieth century allowed widespread reforestation and a subsequent explosion of the adult vectors’ most favoured food resource, white-tailed
Invasive and Disease Species

deer (Fish 1993). Today, urban sprawl increases disease risk by fragmenting these forests and thereby increasing the population of a highly competent host, the mouse *Peromyscus leucopus*, while simultaneously reducing the abundance of less competent hosts (Ostfeld *et al.* 2002). This landscape-driven loss of biodiversity produces a risk ‘amplification effect’ (Keesing *et al.* 2006) that may be applicable to many other disease systems including leishmaniasis, Chagas disease, babesiosis and plague (Ostfeld & Keesing 2000). Malaria has a much simpler transmission cycle involving only mosquitoes and people. Difficulty in predicting the effects of environmental change on malaria arise because each of the 30–40 malaria-carrying mosquito species has different competence levels, habitat preferences and adaptability to environmental changes (World Health Organization 1982; Patz & Wolfe 2002). Landscape changes that decrease the abundance of one malaria-carrying mosquito can increase the abundance of another (Lines 1993). Varying responses of malaria-carrying mosquitoes to urbanization, agriculture, deforestation and other landscape changes around the world (Lines 1993) suggest that the landscape epidemiology of malaria is site and species-specific. This phenomenon is even more apparent when multiple diseases are concerned, as in West Africa, where environmental changes that reduced malaria have also increased the spread of tick-borne relapsing fever (Trape *et al.* 1996).

However, landscape management can successfully control disease organisms. Environmental management strategies have successfully reduced malaria in India (Rajagopalan *et al.* 1990), Honduras (Reid 1997) and northern Europe (McMichael 2001). Various strategies, including draining swamps in Indonesia and Zambia, removing algae in Oaxaca, managing river flow in Zambia, clearing forest in Africa (Molyneux 2002), altering irrigation in Asia (Campbell-Lendrum 2005), a suite of landscape management techniques used by mosquito abatement districts in the USA, Singapore and Cuba have successfully reduced many diseases (Bos & Mills 1987). Still, the idiosyncratic environmental responses of many diseases challenge land managers and policy-makers who want to incorporate landscape epidemiology into management. A few region-, disease- or development-specific health assessment guides exist (World Health Organization 1982; Jobin 1999) but are limited in their applicability. We synthesized known mechanisms linking environmental change and disease emergence (summarized in Patz *et al.* 2005) to suggest a focused management and assessment guide (Box 27.1).
Box 27.1 Summary guide to landscape disease management and assessment (based on Patz et al. 2005)

When designing a landscape, one should consider the effects of anthropogenic drivers (A) on the mechanisms of ecological change (B) that can affect the dynamics of environmentally sensitive and dangerous diseases (C).

A. Anthropogenic drivers that affect disease risk
1. Wildlife habitat destruction, conversion or encroachment, particularly through deforestation and reforestation.
2. Changes in the distribution and availability of surface waters, such as through dam construction, irrigation and stream diversion.
3. Agricultural land-use changes, including proliferation of both livestock and crops; deposition of chemical pollutants, including nutrients, fertilizers and pesticides.
4. Uncontrolled urbanization, including urban sprawl, climate variability and change, migration and international travel and trade.
5. Accidental or intentional human introduction of pathogens.

B. Mechanisms of effects of ecological change on infectious diseases
1. Altered habitats or breeding sites for disease vectors or reservoirs.
2. Niche invasions or transfer of interspecies hosts.
3. Biodiversity change (including loss of predator species and changes in host population density).
4. Human-induced genetic changes in disease vectors or pathogens (such as pesticide and antibiotic resistance).
5. Environmental contamination by infectious disease agents.

C. Diseases sensitive to ecological change or with high potential disease burden
1. Malaria across most ecological systems.
2. Schistosomiasis, lymphatic filariasis and Japanese encephalitis in cultivated and inland water systems in the tropics.
3. Dengue fever in tropical urban centres.
Although Box 27.1 provides a synthesis of major historical drivers and threats, it cannot predict the emergence of unknown pathogens or suggest testable hypotheses in the search for ecological principles underlying disease invasion. The following generalities move in those directions, and have each been tested, observed or hypothesized more than once, with those higher in the list appearing more widely accepted and generalizable.

1 Disruption of natural systems can increase the emergence of disease agents whose ecosystem disservices offset the services gained by the disruption, and vice versa. Human actions can both enhance and curb the density-dependent mechanisms that maintain equilibrium in disease transmission cycles (Campbell-Lendrum et al. 2005).

2 Limiting contact between humans, hosts and vectors by reducing host and vector habitat in human settlements and siting human settlements away from these habitats (World Health Organization 1982; Epstein 2002; Patz & Wolfe 2002).

3 Climate change could increase the global infectious disease burden by altering landscapes and vector distribution and behaviour (McMichael et al. 2004).

4 Culture and behaviour that change the environment to a state other than that in which we evolved increase disease risk (the evodeviationary hypothesis, McMichael 2001). Alteration of natural landscapes should be minimized because human populations often lack resistance to new vectors that invade the perturbed area.

5 Greater abundances and diversity of vector and host predators (Ostfeld et al. 2002) and less competent hosts (Keesing et al. 2006), and reduced abundances of generalist vectors and hosts (Molyneux 2003), reduce disease risk.

6 Vectors tend to travel along roads, rivers, valleys and other linear landscape features (Timischl 1984; Molyneux 2003).

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4 Leishmaniasis and Chagas disease in forest and dryland systems.
5 Meningitis in the Sahel.
6 Cholera in coastal, freshwater and urban systems.
7 West Nile virus and Lyme disease in urban and suburban systems of Europe and North America.
8 Cryptosporidiosis in agricultural systems.
Minimizing edge between different habitat types (including human communities) reduces vector and host habitat and contact with humans (Epstein et al. 1994; McMichael 2001).

**Managing landscapes for invasion resistance**

Species invasions are so numerous that a single-species approach is no longer feasible in many landscapes. In ecosystems that harbour many exotics or frequently receive new ones, single-species management cannot keep pace and may simply allow one exotic to replace another (Zavaleta et al. 2001). For example, managers of tamarisk (*Tamarix* spp.) invasion in the western USA increasingly recognize that they must also address exotic Russian olive (*Eleagnus angustifolia*) and Siberian elm (*Ulmus pumila*), which are both replacing tamarisk over growing areas (NISC Tamarisk Analysis Team, unpublished). Yet most invasive species work still focuses on individual species (Marvier et al. 2004). Landscape interventions for individual species, as opposed to straightforward control and removal approaches, most frequently involve manipulating disturbance — including herbivory by wild or domestic animals, fire, flooding and tillage. Interventions more broadly targeted at building landscape resistance to invasion could, but do not yet, include focal species approaches similar to those for conservation.

The limited literature on landscape design for controlling invasions suggests several coarse-scale principles similar to those for disease management. Several authors emphasize the importance of protecting habitat, minimizing fragmentation and minimizing novel disturbances (Hobbs 2000; Marvier et al. 2004; With 2004). However, a habitat conservation approach is limited by inevitable conversion, fragmentation and disturbance even in protected landscapes. Global-change-mediated habitat disturbance will extend the reach of invaders (see Chapter 31), for example into closed-canopy forest ecosystems (Corlett 1992), which will probably experience greater mortality and gap formation with climate change (Graham et al. 1990). Positive invader interactions can also drive cycles of habitat change and invasion — as when gap formation by exotic insect pests creates opportunities for plant invasions in forests.

Beyond general habitat conservation, particular spatial patterns of disturbance might help limit the spread of invasions (With 2004). With’s models suggest that poor dispersers spread more quickly in landscapes where disturbances are clumped or concentrated, but good dispersers spread more quickly through landscapes with dispersed disturbances that serve as stepping stones.
for movement. The implications for landscape design thus depend on the dispersal abilities of invaders relative to desired native species. With’s models also suggest minimizing edge to reduce invader movement from matrix habitats into less-disturbed fragments. However, these recommendations need to be empirically tested before they are widely applied.

Restoring or maintaining site-specific historical disturbance regimes, including flooding, fire and herbivory, could also help bolster invasion resistance (Hobbs & Huenneke 1992; Stromberg & Chew 2002) (Fig. 27.1). This approach is complicated, though, by the ability of some invaders to capitalize on any disturbance. Fire appears to promote many more invasions than it reduces, even in fire-adapted systems like South African fynbos (D’Antonio et al. 1998; D’Antonio 2000; see also Chapter 11). In some cases, maintaining natural disturbance regimes could limit invader establishment or spread at low abundances but might only accelerate the spread of widespread or abundant invaders. For example, pulse flooding is considered crucial to maintaining and restoring riparian communities in the western USA (Stromberg & Chew 2002) but also contributes to rapid spread of invasive tamarisk (T. Carlson, personal communication). In some cases, reintroducing disturbance could boost long-term control only if the invader’s ability to spread is also impaired.

**Natural disturbance regime**

Maintains native species diversity

- **Decrease in frequency/intensity**
  - Decreased diversity of natives (dominance of competitively superior species)

- **Change in type of disturbance**
  - Elimination of natives, enhanced invasions (direct damage to natives; creation of new microsites)

- **Increase in frequency/intensity**
  - Elimination of natives, enhanced invasions (direct damage to natives; creation of new microsites)

Any change in historical disturbance regime may alter species composition by reducing the importance of native species, by creating opportunities for invasive species, or both.

Figure 27.1 Altered disturbance regimes can change communities via multiple pathways. Adapted from Hobbs and Huenneke 1992.
through traditional control methods such as spraying, biocontrol or mechanical removal. Where disturbance management targets individual invasive species, one must also monitor the effects on other exotics and natives. Controlled burns are used to suppress yellow star thistle in several parts of California, but at one site ecologists found that these fires were also suppressing native forbs and increasing overall exotic cover (C. Christian, unpublished). The effectiveness of and trade-offs involved in manipulating disturbance ultimately depend on local circumstances.

Finally, maintaining diversity and composition of native communities appears to enhance resistance to at least some invasions. Invasibility generally increases as a function of extrinsic site factors that support a more diverse native biota and species similar to potential invaders, but declines within a given site if more species are present to compete directly or indirectly with the invader (Shea & Chesson 2002). The mechanisms linking biodiversity maintenance to increased biotic resistance include an enhanced competitive environment, greater resource use that makes fewer resources available to exotics, and more rapid or complete ecosystem recovery from disturbances that could permit invasions (Alpert et al. 2000). Ecologists have successfully explored seeding native competitors into California grasslands, for instance, to suppress individual invaders like yellow star thistle (K. Hulvey, unpublished) or to boost resistance generally (Seabloom et al. 2003), but managers have not yet applied these findings at large scales.

A holistic approach targeted at reducing invasions should include prevention measures as well as containment, control and resistance features on the landscape. New introductions can probably best be prevented through interventions in trade, transport and early detection at points of entry into a region. Consistent detection efforts are also crucial to prevent nascent foci of exotics from spreading (Chornesky & Randall 2003). Exotics that initially fail to establish or spread because of modest incompatibilities with local environmental conditions, susceptibility to local diseases and predators, or inability to compete can take advantage of temporary windows of opportunity or evolve the capacity to overcome these barriers (Cody & Overton 1996; Thompson 1998). Established exotics that fail to spread should not, therefore, be regarded as benign.

Managing for and against species: General principles

Coarse-scale landscape management suggestions emerging from the disease and biological invasions literature are very similar to each other (Table 27.1).
Table 27.1  **Summary of coarse-scale landscape design suggestions for three species-based goals.**

<table>
<thead>
<tr>
<th>Disease control</th>
<th>Invasive species control</th>
<th>Vulnerable species protection</th>
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<tbody>
<tr>
<td>Minimize anthropogenic disruption in general, but some disturbances can improve disease control</td>
<td>Minimize anthropogenic disruption in general, but natural disturbance regimes can sometimes improve invader control</td>
<td>Minimize anthropogenic disruption in general, but natural disturbance regimes can sometimes benefit vulnerable taxa</td>
</tr>
<tr>
<td>Anticipate climate change effects on landscape and species dynamics</td>
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<td>Anticipate climate change effects on landscape and species dynamics</td>
</tr>
<tr>
<td>Maintain biodiversity to dilute and slow disease transmission</td>
<td>Maintain biodiversity for direct and indirect competitive resistance</td>
<td>Maintain biodiversity to maintain species’ interactions and resilience</td>
</tr>
<tr>
<td>Minimize edge to reduce edgehabitat for vectors/hosts and reduce human contact</td>
<td>Minimize edge between disturbed and undisturbed patches to reduce spread into new habitats</td>
<td>Minimize edge: area ratio between disturbed and undisturbed habitat to maximize habitat value for species</td>
</tr>
<tr>
<td>Anticipate and break up movement along linear landscape features, especially if they involve disturbance (roads, rivers) but even if they consist of relatively undisturbed habitat</td>
<td>Anticipate and break up movement along linear landscape features, especially if they involve disturbance (roads, rivers) but even if they consist of relatively undisturbed habitat</td>
<td>Anticipate need for movement through habitat corridors and stepping stones, and need to break up linear disturbance features such as roads</td>
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<tr>
<td><strong>Clump disturbances and conversion</strong> to limit spread of good dispersers, <strong>and/or disperse disturbances</strong> to limit spread of poor dispersers</td>
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<td></td>
</tr>
<tr>
<td>Limit contact between humans, hosts and vectors by keeping their respective habitats separate</td>
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</table>
This should not be surprising; landscape epidemiology explicitly treats disease organisms and vectors as invaders, and epidemiological frameworks are used in the ecological invasions literature, particularly to describe and model spread (Hastings 1996, Higgins et al. 2000). Management suggestions emerging from the disease and invasions literature are also similar, at the conceptual level, to conservation practices typically targeted at protection of vulnerable and threatened taxa. A potential conflict area between desired and undesired species is around corridors and connectivity – which are to be maximized for the protection of vulnerable species but broken up to prevent disease and invasive spread (Hobbs 1992). Relatively little empirical work has been done on the functioning of corridors, and generalizations across species, scales and ecosystem types are difficult (see Chapters 22–25). However, differences between the ways in which vulnerable and invasive species use corridors might be exploited to minimize conflict. With the notable exception of wildlife diseases borne by large mammals such as brucellosis (Bienen 2002), undesired species often make use of linear disturbed features like roads, while vulnerable taxa often require corridors or stepping stones made up of less-disturbed habitat (Beier & Noss 1998). Minimizing connectivity via roads, power lines and other anthropogenic features to reduce disease and exotic spread can be compatible with maximizing connectivity of relatively undisturbed habitat. Connectivity at different scales might also be managed differently to limit disease and exotic species spread while maintaining desired taxa. For instance, to protect an ecosystem type from invasions or disease, it can be useful to protect examples of that ecosystem in more than one area. Across these discrete areas, connectivity may be minimized or intentionally limited to prevent contagion of harmful organisms (Bienen 2002). Within each protected area, however, connectivity may be enhanced to maintain functional communities and viable populations of large, wide-ranging and top predator species.

**Landscape principles**

Is general concordance between management principles for disease, invasion and vulnerable species wishful thinking?

1 There are inevitably trade-offs among the goals of controlling disease, limiting invasive species, and protecting or enhancing vulnerable taxa and communities – for example, when forest clearance abates infectious
disease or when enhancement of riparian function enables invasive species spread.

2 Sometimes the most critical intervention to protect one or more species will not be landscape design. Managers should tackle multiple goals, addressing landscape interventions for those goals that are most responsive or sensitive to landscape configuration relative to other drivers, and managing goals most affected by other factors through other means. The importance of landscape design versus other interventions must be evaluated when trade-offs arise in managing for and against different values and species. For instance, biological invasions can be managed at the stages of establishment and spread through landscape design as well as at the introduction stage through other measures. Similarly, in some cases it is most effective to use pharmaceuticals, bed nets, quarantines and insecticides rather than landscape design to control disease. Finally, vulnerable taxa may in some cases be conserved best through mitigating the direct impacts of harvesting or invasive species, pollution and/or disease as well as or instead of habitat change (Dirzo & Raven 2003).

3 Consider whether interventions at more than one scale can reduce conflicts and trade-offs among goals. Landscapes experience processes at multiple, nested scales and should be managed at more than one scale (Lindenmeyer & Franklin 2002). Hence . . .

4 General principles cannot substitute for an understanding of local particulars – rather, they can guide which local particulars need to be addressed.

Finally, we return to the overarching question of when and whether individual species considerations should be part of landscape design and stewardship. We suggest the following general guidelines:

1 Carefully selected species should be part – but not all – of landscape targets to provide a mechanistic (process) basis for management. Methodologies for including individual species targets vary, and many have not been evaluated for effectiveness. We argue that including both species-level and ecosystem-level or higher-order targets in landscape design allows management to capture complementary process and pattern-based values.
2 Individual species should be selected with complementarity in mind – vulnerable taxa with contrasting needs, as well as species to be managed against. No single species, or any small set of species, will act as a complete umbrella for other species and desired functions on a landscape. Management that targets different kinds of species, including invasive or disease species as well as threatened or vulnerable species, will provide more complete coverage than management based on one or a few flagship species.

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References


