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ECOLOGICAL IMPACTS AND MITIGATION STRATEGIES FOR RURAL LAND MANAGEMENT

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Abstract. Land-use change and land-management practices affect a variety of ecological processes. Land-use impacts on ecological processes include local extirpations, introductions of new species, changes in land-cover extent, changes in juxtaposition of land-cover types, changes to disturbance regimes, changes in vegetation structure and composition, and effects on air, water, and light quality, and noise pollution. Effects of land-use changes on ecological processes are discussed with special reference to changes in rural environments. Our premise is that better understanding of ecological processes improves land management. Mitigation strategies are presented with respect to management of initial ecological conditions, of the changes themselves, and of the altered system. The paper focuses on proactive environmental management efforts and identifies key research issues as (1) quantifying land-use legacies, (2) determining conditions under which land use modifies impacts of other stressors, (3) identifying conditions under which deleterious impacts can be avoided, (4) understanding cumulative impacts of land-use change, (5) improving our understanding of how land use alters resistance or susceptibility to invasion and impacts of pollutants, (6) crafting socioeconomically reasonable incentives for restoring or reducing effects of land-use practices, and (7) accelerating the integration of social and ecological sciences

Key words: aggressive species; biomass; carbon sequestration; cumulative impacts; disturbance; invasion; land cover; land use; legacy; pollution.

Introduction

Land use and land management are prevailing forces on the Earth (Meyer and Turner 1992, 1994, Dale et al. 2000, Watson et al. 2000). Humans alter ecological processes directly and indirectly through land use, management, and policy decisions regarding natural resources (Brookfield 2001). In the United States, food production uses about 50% of the total land area, 80% of the fresh water, and 17% of the fossil energy used in the country (Pimentel and Pimentel 2003). Land degradation via removal of vegetation, soil erosion, salinization, and soil compaction is also severe, but it is difficult to estimate its extent or cost (Dregne 2002). About 60% of the historical wetland area in the Upper Midwestern region of the United States has been drained, largely for agriculture, causing a decline in flood abatement, water quality improvement, and biodiversity (Zedler 2003). Human activities on the land are pervasive in all types of ecological systems on Earth, even those typically thought of as ''pristine'' and not inhabited by *Homo sapiens* (e.g., Chase 1987, Wilkening 2001). Furthermore, rural land use and management affect all ecological processes, often in several ways that together induce changes to ecological composition, structure, and function.

Changes in rural land use in the conterminous United States over the past 50 years (1950–2000) are pronounced. The general trends are increases in human population density, large exurban growth, and conversion and abandonment of agricultural lands (Brown et al. 2005). Implications for biodiversity of these trends are discussed by Huston (2005) and Hansen et al. (2005). Theobald et al. (2005) set forth how ecological science perspectives can improve land-use planning and policy. The goal of this paper is to examine how improved understanding of ecological processes can facilitate progressive and more enlightened rural land management so as to avoid or mitigate undesirable consequences. We begin with a brief review of environmental issues related to land use and management. We then discuss mitigation strategies and end by articulating research questions that need to be addressed to advance the ecological science forming the foundation

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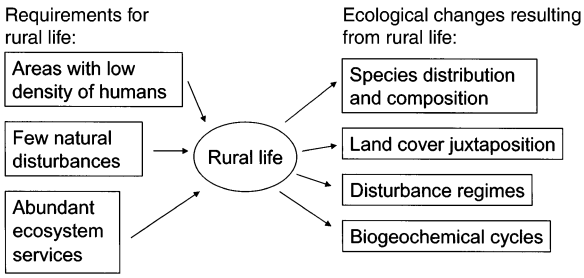

FIG. 1. Requirements for rural life and the ecological impacts.

upon which effective land management is built. In addressing these scientific questions, ecologists can more effectively contribute to the development of proactive and broad-scale land-use and land-management policies.

Terms basic to this discussion are defined (building upon definitions in Dale et al. 2000) as follows. Land cover refers to the ecological state and physical appearance of the land surface. Examples include closed forest, open forest, grassland, and cropland. Change in land cover converts land from one type of dominant vegetation or built environment to another. Land use refers to the land-management practices of humans. Examples are protected areas, timber harvest, row-crop agriculture, grazing, and human settlements. Change in land use may or may not cause a significant change in land cover. Land management is the administration of a given land use by humans. Land management can affect ecological processes without changing the basic land use. For example, management of livestock grazing can be minimal or intensive and regulated or unregulated. Rural living relates to the land uses of agriculture, ranching, and forestry that support country life. Rural life is supported by areas with low density of humans (see Fig. 1a of Brown et al. 2005), few natural disturbances, and an abundance of ecosystem services (e.g., clean water, clean air, etc.) yet induces ecological impacts (Fig. 1).

ECOLOGICAL IMPACTS OF RURAL LAND USE AND MANAGEMENT

Because environmental concerns related to rural land use and management are diverse and occur on a variety of scales, our discussion of impacts is organized around the means by which they affect ecological structure or process. Four major pathways are identified by which land use and management practices can affect ecological processes—changes to species demography and diversity, land cover juxtaposition, disturbance regimes, and biogeochemical cycles. These pathways can occur simultaneously and interact with each other against a backdrop of other stressors that, collectively, can induce dramatic, nonlinear, and self-reinforcing changes. Hence, discerning the means of impact is not always direct, and management based on knowledge of a single pathway may not be adequate when forces interact. Predictions as to the effects of land-management practices are, therefore, likely to be context dependent (Archer and Bowman 2002).

Species changes

Species changes resulting from rural land use and management include changes in demography that may lead to local and selective extirpation or proliferation, introduction of new species, and changes in distribution. Elimination of local populations of a species can occur when land-use practices cause extensive mortality and/or prevent recruitment by altering habitat. For example, replacing forest with suburban homes or farmlands has eliminated populations of lady slipper (*Paphiopedilum villosum* (Lindl) Stein) (V. Dale, *personal observation*), and heavy grazing of rangelands by livestock or wildlife can shift the relative abundance of grasses, forbs, and woody plants, causing the local extinction of some species and a dramatic increase in others (Archer and Smeins 1991, Vavra et al. 1994). Reduction in native earthworm (*Heteroporodrilus mediterreus*) populations as a result of farming can adversely affect soil aeration and other soil processes (Friend and Chan 1995). The spread of agriculture alone has been responsible for the selection of a few crop species that now dominate the Earth's surface. Selection for varieties of plants that resist pests and are easy to grow and harvest has drastically reduced the genetic variation over much of the Earth. For example, 90% of the world's food is provided by 15 crop plant species (Brookfield 2001).

Management actions directed at one species or set of organisms may also lead to unexpected changes in the abundance of other organisms and cause dramatic changes in ecological structure and function. For example, the widespread eradication of prairie dogs, traditionally viewed as competitors with livestock for range forage, may enable the proliferation of undesirable woody plants and, thus, create a whole new set of management challenges and ecological impacts (Weltzin et al. 1997). In another example, xeric habitats associated with urban land uses supported less spider diversity than agricultural fields or residential yards, indicating the importance of incorporating natural habitats into planning of human environments (Shochat et al. 2004).

The widespread use of pesticides associated with farming and other land-management activities has impacted both target and nontargeted organisms. Pesticides include many products, such as insect repellants, weed killers, disinfectants, and swimming pool chemicals, designed to prevent, destroy, repel, or reduce pests of any sort. In the United States, the Environmental Protection Agency (EPA) must evaluate pesticides before they can be marketed to ensure that they protect human health and the environment (especially rare and threatened species). Even so, nontarget species can be killed or adversely affected when these chemicals are used correctly or incorrectly.

Demographic changes can also occur with changes in the size and shapes of habitat. Land uses may bisect populations so that interactions are no longer possible. For example, linear features such as roads can interrupt movements of terrestrial animals (Forman et al. 2002). The long-term consequences of land-use induced disruption on gene flow and population structure are not known. Offerman et al. (1995) suggested a scheme of classifying species according to ''gap-crossing ability'' and using home range size as a means to identify species that are most susceptible to habitat fragmentation.

Another way land use can affect species is via purposeful introduction or unintended spread of aggressive species, which are brought into an area as part of a land-management practice. Examples include the introduction of exotic pasture grasses (e.g., smooth brome (*Bromus inermis*), kleingrass (*Panicum coloratum* L.), and buffelgrass (*Pennisetum cillare* L.)), which are spreading into and displacing native species in rangelands (e.g., D'Antonio and Vitousek 1992, McClaran and Anable 1992). Sometimes particular land-use or management practices foster the spread of certain species. Highways, roads, and right-of-ways along power lines and pipelines can serve as corridors facilitating the spread of exotic plants and animals (Forman et al. 2002). Domestic livestock may promote the spread and establishment of invasive forbs and shrubs in grasslands via seed dispersal and disruption of fire regimes (Archer 1995).

Nonnative species introductions, whether accidental or intentional, have potentially long-lasting ecological and economic impacts (Mooney and Drake 1986, Mack et al. 2000, McNeely 2001) associated with modification of disturbance regimes (Mack and D'Antonio 1998), alteration of biogeochemical cycles (Vitousek and Walker 1989, Le Maitre et al. 1996), reductions in overall species richness (Bock et al. 1986), and, ultimately, species extinctions (Pimm et al. 1995). Biological invasions are internationally regarded as a major threat to biological diversity, second only to habitat loss (Coblentz 1990, Vitousek et al. 1997*a*, Wilcove et al. 1998). Though many papers have cited the impacts of nonnative species on native biodiversity and ecosystem processes, studies quantifying these effects are few (Parker et al. 1999) and largely descriptive or observational in nature (Cronk and Fuller 1995).

These examples illustrate the importance of ecosystem rather than organismic approaches to land use and management. Such a systems approach considers all plant and animals and the physical conditions of their environment, as well as socioeconomic conditions (Holling 2001). Recognition of how land-use practices might foster or impede the spread of organisms is a first step toward developing strategies for containing deleterious organisms and altering barriers affecting the movement of species. For instance, knowing that the fungus killing Port Orford cedar (*Chamaecyparis lawsoniana*) in southern Oregon and northern California is spread by logging trucks, the washing of logging trucks was initiated (Strittholt and DellaSala 2001, Jules et al. 2002). This simple and inexpensive action has proven to be an effective deterrent to the spread of the fungal spores. Control of the spread of deleterious species may require local sacrifices and drastic land-use changes to avert escalation to regional scales. For example, when the Asian long-horned beetle (*Anoplophora glabripennis*) was first introduced into Halifax, Nova Scotia, Canada, there were calls to cut all spruce (*Picea mariana*) trees in a broad swath around the point of infestation as means to prevent their spread (Haack et al. 1977). However, local homeowners did not want to compromise their landscaping and refused to implement this action. The insect eventually killed the trees around those homes anyway, and its spread now threatens the logging industry in northeastern North America.

Changes in land-cover juxtaposition

Changes in land cover that result from land use can alter habitat and the juxtaposition of cover types. Habitat alterations (such as occur with cropland conversions, urban expansion, logging, grazing, construction of dams, water course alterations, etc.) can make a site unsuitable for species that once occupied an area. Local site disturbance can make a place available for new species or ecosystems. Linear features, such as roadside vegetation and fencerows, may enhance the spread of select organisms (Camp and Best 1994). For example, the spread of the gypsy moth (*Lymantria dispar*) is so tightly linked to road networks in the eastern United States that maps of gypsy moth distribution over time delineate roads (Sharov and Liebhold 1998); and coyote (*Canis latrans*) emigrate along highways, taking advantage of road kills for food and culverts for shelter (Clevenger et al. 2001). Conversely, some land uses may constitute barriers to the spread of other organisms. The presence of wolves (*Canis lupus*) is inversely related to road density (Mladenoff et al. 1995).

Changes in the juxtaposition of land-cover types can also impact ecological processes. For example, the expansion of suburban areas and loss of forests in the eastern United States has drastically increased the area of forest edge. As a result, those species that occur in forest edges (such as the native Cowbird [*Molothrus ater*]) are becoming more prolific (Chalfoun et al. 2002). Simultaneously, the erosion of soil from disturbed areas into more pristine areas is more common now (Pimentel and Skidmore 1999, Pimentel 2000). Agricultural land typical erodes soil at rates ranging from 13 $tons \cdot ha^{-1} \cdot yr^{-1}$ to 40 $tons \cdot ha^{-1} \cdot yr^{-1}$ worldwide

(Pimentel and Kounang 1998). Other papers in this issue discuss examples of how patterns of land cover affect biodiversity (e.g., Hansen et al. 2005).

Changes to disturbance regimes

Land-use and management practices alter disturbance regimes (e.g., fire, pest outbreaks, floods, blowdowns) by disrupting the frequency, extent, and intensity of disturbance as well as by instigating new disturbances. Fire severity and time since fire can affect the richness and dominance of nonnative species (Keeley et al. 2003). Curtailment of surface fires exemplifies how disruption of disturbance frequencies can alter fundamental ecosystem properties. Elimination of fire in systems that evolved with frequent surface fires has caused dramatic changes in structure and function (e.g., oak savannas of the Midwest [Peterson and Reich 2001], ponderosa pine [*Pinus ponderosa*] forests of the western United States [Covington and Moore 1994], and longleaf pine [*Pinus palustris*] forests of the southeastern United States [Gilliam and Platt 1999, McCay 2000]). Ironically, increases in disturbance intensity often result from land-management practices designed to control frequent, small, low intensity disturbances. Such controls may create the very conditions that make large, catastrophic disturbances possible. Examples include the recent massive crown wildfires in southwestern U.S. forests (Covington 2000) and outbreaks of the native southern pine bark beetle (*Dendroctonus frontalis*) (Perkins and Matlack 2002).

Land-management practices are often aimed at altering the frequency and intensity of disturbances such as fires (e.g., Keeley 2002) and floods (e.g., Persoons et al. 2002). But reductions in seasonal flooding (Johnson 1994, Friedman and Lee 2002) and in the frequency of low-intensity surface fires (Covington and Moore 1994, Brawn et al. 2001) have altered communities that once depended on these events. Changes in the frequency or intensity of one type of disturbance may be linked to alterations in the frequency or intensity of other disturbances. For example, livestock overgrazing can reduce the amount and continuity of fine fuels to the extent that surface fires are not possible (Madany and West 1983, Baisan and Swetnam 1990, Savage and Swetnam 1990). In recognition of the importance of disturbance in maintaining certain ecological structures and preventing changes to undesirable states, land-management practices may seek to mimic historical conditions to which species in a region are accustomed (Parsons et al. 1999). Examples include the reintroduction of flooding in the Grand Canyon (Powell 2002) and prescribed burning in grasslands, savannas, and certain forests (e.g., Scifres and Hamilton 1993, Arno et al. 1995, Andersen et al. 1998, Agee 2003, Fuhlendorf and Engle 2001, 2004).

Land use and management practices can also increase the susceptibility of ecological systems to other disturbances (e.g., landslides can result from building roads on steep slopes [Swanson and Dyrness 1975] and from land-use modifications [Glade 2003]; heavy grazing can exacerbate wind and water erosion and hence ecological degradation [Tongway and Ludwig 1997]). Knowing the conditions that foster disturbances and when to control them is a challenge facing both the ecological research and environmental management communities.

Changes to biogeochemical cycles

Changes in the cycling of water, nutrients, and energy inevitably occur from land use and management via many pathways. For example, air and water pollution often accompany use of land for industry, transportation, and urban growth. Current emissions of carbon dioxide, nitrous oxide, and methane have increased dramatically due to changes in land management, especially with industrialization and intensification of agricultural practices. Changes in vegetation structure that occur with land use and management alter pathways of energy flow and nutrient cycling. Changes in the extent of forest and impervious land covers can dramatically alter watershed hydrology (e.g., Wissmar et al. 2004). Climate and atmospheric chemistry are directly and indirectly influenced by land cover, via biophysical and biogeochemical aspects of land-surface–atmosphere interactions (Hoffman and Jackson 2000, Aber et al. 2001, Bonan 2002). Even when a land-use practice is no longer in place, its legacies remain (e.g., plow furrows or livestock wastes can have long-term effects on the environment; Bellemare et al. 2002, Foster et al. 2003). We focus on three examples of land-use impacts to biogeochemical cycles (air pollution, greenhouse gas emissions, and changes in vegetation structure and composition) as a way of illustrating the diversity of interacting factors.

Air pollution

Air pollution is the accumulation of solid, liquid, and gaseous compounds in the atmosphere at concentrations that are greater than would naturally occur at a particular location under given meteorological, biological, and geological conditions. Ozone, particulate matter, lead, carbon monoxide, mercury, and sulfur dioxide are just a few examples of air pollutants. These and other pollutants can alter physical and chemical atmospheric cycles and affect human and ecological health. Changes in land use and cover can affect these pollutants through two direct mechanisms (emissions and deposition) and three indirect mechanisms (atmospheric chemistry, physical meteorology, and radiation transfer). These same mechanisms can feed back and affect land cover, and possibly even land use.

Even in an unperturbed natural ecological system, volatile organic compounds (VOCs) emitted by vegetation (Rasmussen and Went 1965, Hewitt 1999) can

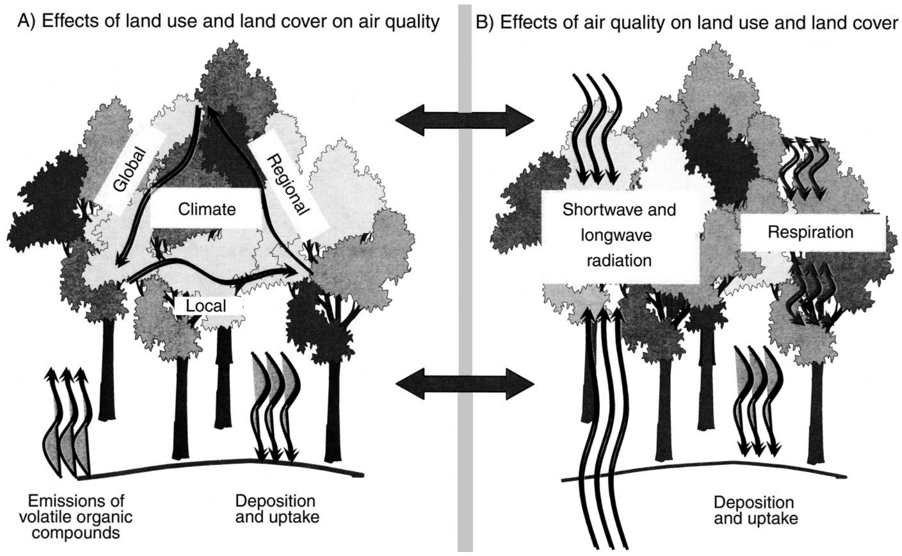

FIG. 2. Depiction of mechanisms by which (A) land use and land cover affect air quality and (B) air quality affects land use and land cover. These feedback mechanisms differ depending on the prevailing vegetation type, soil moisture, temperature, pH, nutrients, and surface permeability.

contribute to the secondary formation of organic aerosols (Hatakeyama et al. 1989, Mazurek et al. 1991, Hoffman et al. 1997). In a polluted, highly oxidizing atmosphere, these gas-to-particle conversions may be accelerated. Regardless, all VOCs are not equal, and different plant species emit different types of VOCs. For example, many deciduous trees are high emitters of isoprene, a compound that can be important in the formation of ozone (Chameides et al. 1988) but is less significant to the secondary formation of organic aerosols. In contrast, many evergreen species are prolific emitters of α- and β-pinene and can contribute significantly to the formation of organic aerosols (Hoffman et al. 1997) but are less important to ozone formation. Thus, for any parcel of land, wholly different emission profiles may be expected within the broad spectrum of potential biotic land covers (Fig. 2) (Lamb et al. 1993, Simpson et al. 1995, Isebrands et al. 1999). Similarly, different land covers have surfaces that differ in two and three dimensions (e.g., size, shape, and orientation) that can affect the rate of aerosol deposition (Wesely and Hicks 2000). In turn, deposition of particles onto foliage can influence rates of photosynthesis and nutrient and contaminant loads (Chameides et al. 1999, Bergin et al. 2001).

Other feedbacks associated with particulate matter include local, regional, and global impacts on longwave and shortwave radiation budgets, which, in turn, influence temperature, precipitation, and photosynthetic active radiation (Cerveny and Balling 1998, Rosenfeld 2000, Chameides and Bergin 2002, Kaufman et al. 2002, Menon et al. 2002). Proliferation or intensification of industrial, commercial, transportation, or residential land uses can further (and sometimes drastically) alter emissions and deposition profiles. The more intensified land uses can open new pathways to primary and secondary particulate matter production and removal, with corresponding climate effects (e.g., some anthropogenic sources may inject large quantities of diesel soot directly into the atmosphere, whereas others may contribute to the secondary formation of sulfate aerosols from the oxidation of directly emitted sulfur dioxide). Thus, cumulative effects on air pollution arising from land use and management depend on initial conditions and actions on the land, but these impacts can be difficult to predict because of the many feedbacks and indirect interactions.

Greenhouse gas emissions

Land-management practices, such as grazing, forestry, and conversion to arable lands, affect trace gas emissions due to alterations in the cycling of nutrients and distribution of organic matter. These changes in land management have substantially contributed to decreased CH_4 oxidation and increased CO_2 emissions as well as N_2O production from soils. CO_2 emissions resulting from land-use change since 1850 are approximately 50% of the contributions due to fossil fuel burning and cement production (Watson et al. 2000). Extensive use of N fertilizers and increased atmospheric loading of N into many regions of the world have probably contributed to the observed increases in atmospheric CH_4 and N_2O (Ojima et al. 1993, Mosier et al. 1997). Increased use of nitrogen has simultaneously contributed to changes in primary production, decomposition, and carbon storage; eutrophication of lakes, estuaries, and coastal areas; acidification of soils, streams, and lakes; and changes in species composition and biodiversity (Galloway et al. 1995, Holland et al. 1997, Vitousek et al. 1997*b*, Smil 1999).

Rural land-use practices associated with land clearing, cultivation, and drainage of wetlands affect the cycling of carbon and nitrogen and enhance the mobilization of nitrogen from soil organic matter. Fertilizer consumption in a number of developing countries has accelerated at a much faster pace than the global average in recent years (United Nations Food and Agricultural Organization 2002). These changes portend large increases in N trace gas emissions, since N volatilization losses can be much higher from tropical and subtropical agricultural soils than from temperate soils (Keller and Matson 1994, Matson et al. 1996). Agricultural practices associated with livestock and poultry production directly and indirectly affect emissions and hydrological efflux of nutrients. The livestock industry produces large amounts of waste material (Nevison and Holland 1997), which are used as organic fertilizer. However, much of the nitrogen associated with animal manure is lost through volatilization to NH_3, accounting for over one-third of global NH_3 emissions to the atmosphere (Bouwman et al. 1997).

Rural land clearing in many developing countries is associated with biomass burning. Biomass burning releases large amounts of carbon dioxide and reactive nitrogen to the atmosphere. These emissions contribute to recent changes in greenhouse gas fluxes from agricultural lands around the world (Crutzen and Andreae 1990, Crutzen and Goldammer 1993, Lindesay et al. 1996). Fluxes of N_2O in some cases remain elevated following biomass burning due to increased nitrate levels in soil and reduction of plant uptake of nitrogen.

Changes in vegetation structure

Another example of the effects of rural land use and management on biogeochemical cycles is via changes in physiognomy. In the process of using natural resources, humans often induce changes in vegetation structure. The most well-known examples are deforestation and desertification. Desertification results from extraction of water in excess of what a region can afford to lose and can lead to long-term change in water availability (Dregne 1983, Verstraete 1986, Schlesinger et al. 1990, Moat and Hutchinson 1995, de Soyza et al. 1998). The process of increasing woody plant density is less well known and, thus, is the focus of our discussion here (Fig. 3).

The proliferation of shrubs and trees in grasslands and savannas has been widely reported in arid, semiarid, and montane regions of North and South America, Australia, and Africa over the past century (Archer et al. 2001, Archer 2003). The causes of woody encroachment are actively debated, but likely reflect changes in climate (amount and seasonality of rainfall), herbivory (increased grazing or decreased browsing), fire regimes (decline in frequency and/or intensity), atmospheric

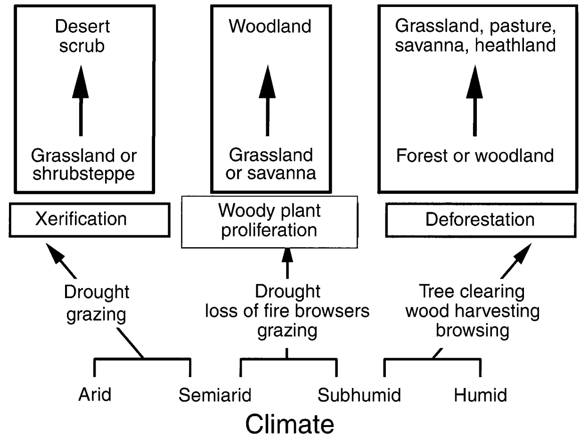

FIG. 3. Diagrammatic representation of processes of biomass change: woody proliferation in relation to desertification and deforestation (adapted from Archer and Stokes [2000]).

CO_2 enrichment, and N deposition (Archer 1994, Archer et al. 1995, van Auken 2000, Kochy and Wilson 2001). It is difficult to assign primacy to these factors, which have likely interacted strongly through time. In drylands, woody plant encroachment occurs when mesophytic grasses are replaced by unpalatable shrubs and trees (e.g., *Larrea*, *Artemesia*, *Prosopis*, *Juniperus*, *Pinus*). There are many ramifications for such vegetation change. Because woody plant proliferation adversely affects grass production and, hence, livestock production (Scholes and Archer 1997), it threatens the sustainability and profitability of commercial ranching and pastoral land uses. In addition, shifts from grass to woody plant domination alters wildlife habitat; and proliferation of deep-rooted woody plants is often assumed to cause ground water depletion and reduce flows of springs and streams. As a result, ''brush management'' in drylands is often a key element of wildlife (Ben-Shaher 1992), livestock (Scifres 1980, Valentine 1997, Bovey 2001), and watershed management.

Early approaches to brush management for livestock grazing typically had the goal of widespread, indiscriminant woody plant eradication; however, the rising cost of fossil fuels coupled with short-lived treatment effects have made large-scale mechanical and chemical treatments of woody plants economically tenuous or unrealistic. In addition, the recognition that herbicides can have deleterious environmental effects and that woody plants provide habitat for wildlife has lead to the advent of more progressive and selective approaches (e.g., Scifres et al. 1985, 1988).

Numerous studies have challenged traditional perspectives on the effects of woody plants on the hydrological cycle in drylands, but this topic remains highly controversial (Belsky 1996, Wilcox 2002). Effects of woody plant encroachment on biodiversity have not been studied but likely vary with species, growth form, and stand development. Field observations indicate that woody plants such as *Juniperus*, *Tamarix,* and *Pinus* can form virtual monocultures. Furthermore, increases in woody plant abundance in drylands can lead to significant changes in ecosystem carbon stocks (e.g., Archer et al. 2001, Jackson et al. 2002, Houghton 2003, Wessman et al. 2004), which may support new land-use drivers as industries seek opportunities to acquire and accumulate carbon credits to offset their CO_2 emissions. Woody plant proliferation in grasslands and savannas may, therefore, shift from being an economic liability to a source of income. However, perverse incentives may result, as land management may shift to promote rather than deter woody plant encroachment. Thus, potential benefits associated with carbon sequestration should be carefully weighed against costs in the form of increases in nitrogen and non-methane hydrocarbon emissions (Guenther et al. 1999, Martin et al. 2003), potentially adverse effects on livestock production, stream flow and ground water recharge, the extirpation of plants and animals characteristic of grasslands/savannas, and, indeed, the local or regional extinction of grassland and savanna ecosystems.

Mitigation Strategies

Clearly, major challenges are associated with understanding and mitigating the negative impacts of rural land use on ecological processes. The complexity of meeting these challenges can be simplified by considering potential impacts before, during, and after land-use change occurs. We first consider the situation where ecological functions are relatively intact. Second, we focus on the land use and management activity itself. The last analysis deals with the ecological system after it has been altered by land use and management practices. This approach to coping strategies reflects the need to envision the land both before and after the transformation, as well as the changes themselves. Because changes to the land typically are intense and cumulative and leave persistent legacies, recovery to initial conditions is often not plausible or requires a long time or high investments. A proactive and broad-scale perspective on land use and management is the most effective approach (Robertson et al. 2004).

Managing ecological conditions

In situations where ecological systems are still relatively intact and the effects of surrounding land-use changes have not had a great impact, land management and policy decisions can be designed to enhance resilience (i.e., the ability to bounce back after a change) or resistance (ability to endure). Sometimes, protecting key areas can reduce ecological vulnerability. Preserved areas should be large enough to protect habitats, species, and the environmental conditions required to support them. Typically, the optimal size of protected areas depends on the home range of species that inhabit the area, topographic conditions, and use of neighboring lands (e.g., Hansen and Rotella 2001). Spatial features of the landscape often need to be considered, and boundaries should be selected based on the configuration of ecological conditions (e.g., catchment boundaries) rather than political or social features (e.g., land owner county or country borders). For example, watershed protection should include the headwaters of streams, and distance to other preserved areas can be important if animals are anticipated to move between them. In his analysis of land planning, Forman (1995) argued that initial land-management decisions should be based on location of water and biodiversity concerns, for these are the most susceptible features.

A second way to enhance resilience and resistance in the face of impending change is by maintaining or establishing species able to tolerate stressful conditions. Via their persistence, such species may reduce ecological impacts. For instance, Freeway Park in downtown Seattle is placed high above the interstate highway and planted with trees and shrubs able to tolerate air pollution and shallow soils. More often, the selection of species for landscaping is based on appearance and cost rather than resistance and resilience. For example, rural and suburban lawns are typically planted with nonnative grasses, which are relatively inexpensive to establish and thrive under moderate traffic and repeated cutting. But other plant species can be more effective at maintaining a diversity of ecosystem services while reducing the need for high volumes of water, pesticides, and fertilizers that ultimately impact groundwater, streams, and atmospheric chemistry (Baron et al. 2002).

Maintaining ecological conditions is the most cost-effective way to protect environmental conditions, for reclamation or restoring dysfunctional ecological systems is costly and often has a low probability of success. However, it is sometimes unrealistic to maintain ecological conditions. Proactive strategies to reduce environmental impacts may be most effective. For example, field tests of pesticide free production (PFP) demonstrate that reduced use of pesticides is becoming a reasonable alternative for farming (Nazarko et al. 2003). Similarly, Integrated Pest Management (IPM) is a proactive practice that reduces pesticide use by a four-step approach that (1) sets action thresholds, (2) monitors and identifies pests, (3) prevents pests from becoming a threat, and (4) controls pests by use of pesticides (only when necessary, and then using less risky chemicals first).

Influencing land use and management practices so they are less harmful to the environment

There are several ways that changes to the land can be managed so that they are less deleterious to ecological services. By organizing the location of land uses within a landscape context, land-use choices, which take advantage of natural features, can be developed to reduce harmful impacts on the ability of the system to provide ecosystem services. Zoning regulations restrict the location of land uses but typically focus only on making adjacent land uses compatible with socioeconomic goals. The extent to which industrial and other intensive activities that cause severe environmental harm are confined to the more resistant or resilient locations varies greatly. Such locations may include areas that support few rare species, have soils and bedrock through which water does not readily percolate, or are not directly connected to groundwater. Salt caverns are examples of such resistant locations.

Another way to reduce impacts of land use is to create sacrifice areas where concentrated and intense land uses occur and can be contained, so that other areas can be spared. This strategy is implemented on many military installations in the United States and is likely one of the reasons these lands support so many endangered species (Leslie et al. 1996). Focusing high human impacts on resistant or resilient locations can diminish the potential for spread of the impact. For example, dense residential development can translate to more natural areas left undisturbed and to placing forest plantations on sites resilient to the repercussions of intense tree management so that sensitive sites can be protected.

An additional strategy is to adopt strategies and regulations that diminish environmental impacts such as water runoff, atmospheric emissions, and loud noises that result from land-use activities. For example, rotation grazing during droughts can decrease the area of bare ground (Teague et al. 2004), which often leads to erosion. Mulching and composting reduce evapotranspiration and enhance soil quality. More broadly, the traditional default approach of ‘‘dilution as the solution to pollution’’ is appropriate only for point sources and where unpolluted areas are large. Now that pollution sources are almost ubiquitous, the dilution approach exacerbates pollution intensity and may induce cumulative effects. Building higher smoke stacks simply causes regional rather than local air quality changes. Furthermore, most environmental regulations only address the rate of emissions and not the effects. For example, air quality management is largely driven by direct regulation of technology: cleaner cars, cleaner fuels, cleaner industry, etc.; but cumulative impacts of many sources of pollution are not addressed.

Even so, coping strategies aimed at technology are a major opportunity for improving environmental effects of land use. For example, resource extraction activities can be designed to reduce sources of environmental problems. The use of whole-tree harvesters (e.g., feller-bunchers) to cut trees results in more beneficial debris left in the forest, less soil disturbance, and more wood sent to market. Technological options on agricultural lands include conservation tillage, integrated nutrient management (which uses manure and compost), precision farming, organic farming, conversion of monoculture to complex diverse cropping systems, meadow-based rotations and winter cover crops, and establishing perennial vegetation along contours of steep slopes (Leopold 1948, Lal 2003). No-till farming can significantly reduce erosion from agricultural lands where erosion occurs largely as a result of rain and wind action on plowed ground (Pimentel and Kounang 1998). Of course technology changes also impact land use itself. With the advent of chain saws, bulldozers, and other large machinery, rates of land-cover change have dramatically accelerated worldwide (e.g., Leopold 1948, Klink et al. 1993).

Furthermore, land use can be managed so that deleterious effects are reduced in size, impacts are less likely to occur, or the size or longevity of their legacies are diminished. For example, breaks in vegetation can reduce the spread of wild fires or insect outbreaks and reduce the size of the impacted area. As another example, pollution resulting from transportation can be diminished by placement of industrial, residential, and commercial uses to reduce transit distance and by embracing mass-transportation systems. The emerging trend seems now to concentrate sources of pollution (e.g., build more dense core cities and maintain outlying green space) and to make those sources cleaner.

Given that land use and management actions that change ecological conditions are sometimes necessary, one benefit of an ecological perspective is that the potential for environmental losses is recognized. Thus, plans can be set in place to reduce negative impacts of land changes to the environment (e.g., by protecting vulnerable resources during construction). Alternatively, losses at one location can be used to bargain for environmental gains at another place (e.g., wetland mitigation).

Managing the land and ecological processes after the land changes

In situations where land use and management have degraded ecological composition, structure, and function, it is necessary to develop coping strategies that promote ecological restoration and mitigate against further harmful impacts. However, restoration to the original ecological state is costly, takes time, may have a low probability of success, and sometimes is not even possible or useful. Land-management practices benefit from recognizing that ‘‘an ounce of prevention is worth a pound of cure.’’ Yet current activities on the land typically do not have a strong ecological perspective. Mitigation and coping strategies for impacted ecological systems need the joint effort of the scientific and decision-making communities to provide restoration of the impacted ecological system or amelioration of the deleterious effects on natural resources. Being able to modify an ecological system requires a high level of cooperation and agreement among landowners and managers. There are a few examples of such cooperation (e.g., the Applegate Partnership in Oregon; the Malpai Borderlands Group in southern Arizona [McDonald 1995]), and ways to develop collaboration have been set forth (Wondolleck and Yaffee 2000). Ecological understanding has provided sound principles for decisions about land use and management (Dale et al. 2000).

Coping with changes to the land requires recognizing that human activities are a part of the rural landscape. Dealing with natural variability requires a perspective that builds on the history as well as social values for of an area (Hunter 1993, Hessburg et al. 1999, Landres et al. 1999). There is a growing literature on desired future conditions, which is often a more achievable goal than trying to reestablish historical conditions (e.g., Gonzalez 1996, Liu et al. 2000). The concept of desired future condition is most meaningful at the scale of a region because it explicitly considers the mix of habitats (type and seral stage) generated by processes that are only observable at the broader scale. To sustain ecological systems and preserve ecological integrity, management must allow for the dynamic processes that accompany disturbance–recovery cycles and protect essential energy and material transfers that take place during changes to the land. When these ecological processes are operative over a broad area, a mosaic of habitat patches exist in various stages of postdisturbance recovery (e.g., Fuhlendorf and Engle 2001, 2004). Given the nonequilibrium nature of ecological systems, the distribution of terrestrial and aquatic habitats is dynamic. As a consequence, desired future conditions include variability as an integral and essential component of habitat and population objectives.

Attainment of desired future conditions can be assessed by a suite of ecological metrics that collectively represent key features of the environment (Dale and Beyeler 2001). For example, one metric might compare the distribution of terrestrial and aquatic habitats following management to that expected under natural disturbance regimes (e.g., Hunter 1993, Landres et al. 1999). A critical management challenge is to ensure that human activities do not increase the frequency or severity of disturbances to such an extent that they surpass the capacity of the ecological systems to recover. To ensure resilience, management practices must not disrupt those energy and material transfers that promote habitat recovery. An appropriate goal for management activities would be to mimic, to the extent possible, natural disturbance events in terms of their severity (i.e., spatial extent and character) and recurrence interval.

Structural changes to ecological systems can sometimes enhance function and artificially speed up the rate of succession or even change the recovery path to an alternative stable state (Leopold 1948, Ludwig et al. 1997, Whisenant 1999). Early stages of succession are most affected by substrate characteristics (soil texture, moisture, and nutrient condition), distance to seed sources, and seed morphology, whereas later stages are largely determined by environmental changes caused by earlier immigrants. Thus, the rate of succession can be enhanced by such actions as establishing plants that promote autogenic recovery and providing nesting sites that attract birds that disperse seed of native plants and hence foster succession. Placing wood and debris in recovering systems can quickly create habitats that allow the reintroduction of a variety of species (Bouget and Duelli 2004). Physical enhancement to soil texture and establishment of berms and retention ponds can dramatically reduce water runoffs, which is often detrimental to aquatic organisms. Restoration of wetlands can provide the services of flood abatement, water quality improvement, and enhancement of biodiversity (Zedler 2003).

Incentives are often needed to implement management practices that are in tune with the environment. Satisfying environmental laws and regulations is the typical goal. With the globalization of the world's economy and the recognition of the relationship between the carbon stored in vegetation and soils and climate change (e.g., Shukla et al. 1990, Dale 1997, Malhi et al. 2002, Antle et al. 2003), carbon credits are emerging as a measure of land-use impacts on the carbon cycle. In this situation, industry provides funding for carbon sequestration efforts associated with improved cropping systems, afforestation, land improvements, and rehabilitation of degraded lands. These efforts have additional positive spin-offs, including decreased erosion, increased soil fertility, and water-holding capacity, and enhanced biodiversity and wildlife habitat. Thus, the use of carbon credits is promoted as a strategy that has both environmental and social benefits, which allows industry to meet emissions standards while providing land managers the funds needed to implement progressive management and restoration practices (Cairns and Lasserre 2004).

Research Needs

Exploring ''causes, mechanisms, and consequences of land use and land-cover change'' is one of the top 10 research topics in landscape ecology (Wu and Hobbs 2002) and has been an ongoing research theme in NASAs Land-Cover Land-Use Change program (more information *available online*)[6] and the Strategic Environmental Research and Development Program's Ecosystems Management Project (more information *available online*).[7] Under this broad topic, we identify seven major areas for research focus:

1) Quantify land-use legacies. This quantification involves characterizing changes in structure, function,

[6] ⟨http://lcluc.gsfc.nasa.gov/⟩

[7] ⟨http://www.cecer.army.mil/KD/SEMP⟩

and composition and determining persistence and spatial extent. It also requires knowing what conditions may influence the type, duration, and extent of land-use legacies.

2) Determine conditions under which land-use change modifies (exacerbates or ameliorates) impacts of other stressors. For example, during droughts the impacts of land-management practices can be more severe. It will be useful to determine conditions under which severe impacts of ''compounded perturbations'' (Paine et al. 1998) are common. The effect of land-use changes on natural disturbances is also an area that needs further investigation.

3) Identify conditions under which increased impacts can be avoided. For example, what properties of ecological systems confer resistance to change or an ability to recover from change (resilience)? What types of farming practices enhance soil water retention?

4) Understand cumulative impacts of land-use change. This understanding requires knowing how and when different land-use changes interact with other stresses to affect the environment. Because cumulative and synergistic effects are so pervasive, defining when such conditions do *not* occur might be the easier task.

5) Improve understanding of how land use alters resistance or susceptibility to invasion and impacts of pollution. Quantitative and experimental studies are needed if we are to develop a robust understanding of the drivers and functional consequences of the spread of nonnative species and environmental pollutants.

6) Craft socioeconomically reasonable incentives for restoring or reducing effects of land practices. Setting environmental goals within their socioeconomic context highlights research needs and the potential for establishing incentives that have global as well as local significance. A key challenge in implementing viable strategies and incentives is the political instability in developing countries. Ways to quantify both realized and potential benefits over large and spatially heterogeneous areas need to be developed, along with methods of monitoring and tracking how well incentives are met.

7) Accelerate the integration of social and ecological sciences. Long-lasting plausible ecological solutions will not be effectively implemented unless multiple goals of society are reasonably met. It is, therefore, necessary for ecologists to work with social scientists a priori to determine how goals for land management are developed and when and how political, economical, or social conditions may constrain management options. Ultimately, ecologists need to achieve a better understanding of political, economic, and social drivers of change in land use if they are to effectively assess and influence present and future land-use options.

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Literature Cited

Aber, J., R. P. Neilson, S. McNulty, J. M. Lenihan, D. Bachelet, and R. J. Drapek. 2001. Forest processes and global environmental change: predicting the effects of individual and multiple stressors. BioScience **51**:735–751.

Agee, J. K. 2003. Historical range of variability in eastern Cascades forests, Washington, USA. Landscape Ecology **18**:725–740.

Andersen, A. N., R. W. Braithwaite, G. D. Cook, L. K. Corbett, R. J. Williams, M. M. Douglas, A. M. Gill, S. A. Setterfield, and W. J. Muller. 1998. Fire research for conservation management in tropical savannas: introducing the Kapalga fire experiment. Australian Journal of Ecology **23**:95–110.

Antle, J., S. Capalbo, S. Mooney, E. Elliot, and K. Paustian. 2003. Spatial heterogeneity and the design of efficient carbon sequestration policies for agriculture. Journal of Environmental Economics and Management **46**:231–250.

Archer, S. 1994. Woody plant encroachment into southwestern grasslands and savannas: rates, patterns and proximate causes. Pages 13–68 *in* M. Vavra, W. Laycock, and R. Pieper, editors. Ecological implications of livestock herbivory in the West. Society for Range Management, Denver, Colorado, USA.

Archer, S. 1995. Herbivore mediation of grass–woody plant interactions. Tropical Grasslands **29**:218–235.

Archer, S. 2003. Proliferation of woody plants in grasslands and savannas: a bibliography. Texas A&M University, College Station, Texas, USA.

Archer, S., T. W. Boutton, and K. A. Hibbard. 2001. Trees in grasslands: biogeochemical consequences of woody plant expansion. Pages 115–138 *in* E.-D. Schulze, M. Heimann, S. Harrison, E. Holland, J. Lloyd, I. Prentice, and D. Schimel, editors. Global biogeochemical cycles in the climate system. Academic Press, San Diego, California, USA.

Archer, S., and A. Bowman. 2002. Understanding and managing rangeland plant communities. Pages 63–80 *in* A. Grice and K. Hodgkinson, editors. Global rangelands: progress and prospects. CAB International, Wallingford, Oxon, UK.

Archer, S., D. S. Schimel, and E. A. Holland. 1995. Mechanisms of shrubland expansion: land use, climate or CO_2? Climatic Change **29**:91–99.

Archer, S., and F. E. Smeins. 1991. Ecosystem-level processes. Pages 109–139 *in* R. K. Heitschmidt and J. W. Stuth, editors. Grazing management: an ecological perspective. Timberline Press, Portland, Oregon, USA.

Archer, S., and C. J. Stokes. 2000. Stress, disturbance and change in rangeland ecosystems. Pages 17–38 *in* O. Arnalds and S. Archer, editors. Rangeland desertification. Kluwer Publishing Company, Dordrecht, The Netherlands.

Arno, S. F., M. G. Harrington, C. E. Fiedler, and C. E. Carlson. 1995. Restoring fire-dependent ponderosa pine forests in western Montana. Restoration and Management Notes **13**:32–36.

Baisan, C. H., and T. W. Swetnam. 1990. Fire history on a desert mountain range: Rincon Mountain wilderness, Arizona. Canadian Journal of Forest Research **20**:1559–1569.

Baron, J. S., N. L. Poff, P. L. Angermeier, C. N. Dahm, P. H. Gleick, H. G. Hairston, R. B. Jackson, C. A. Johnston, B. D. Richter, and A. D. Steinman. 2002. Meeting ecological and societal needs for freshwater. Ecological Applications **12**:1247–1260.

Bellemare, J., G. Motzkin, and D. R. Foster. 2002. Legacies of the agricultural past in the forested present: an assessment of historical land-use effects on rich mesic forests. Journal of Biogeography **29**:1401–1420.

Belsky, A. J. 1996. Western juniper expansion: is it a threat to arid northwestern ecosystems? Journal of Range Management **49**:53–59.

Ben-Shaher, R. 1992. The effects of bush clearance on African ungulates in a semi-arid nature reserve. Ecological Applications **2**:95–101.

Bergin, M. H., W. Greenwald, J. Xu, Y. Berta, and W. L. Chameides. 2001. Influence of aerosol dry deposition on photosynthetically active radiation available to plants: a case study in the Yangtze delta. Geophysical Research Letters **28**:3605.

Bock, C. E., J. H. Bock, K. L. Jepson, and J. C. Ortega. 1986. Ecological effects of planting African lovegrasses in Arizona. National Geographic Research **2**:456–463.

Bonan, G. B. 2002. Ecological climatology. Cambridge University Press, Cambridge, UK.

Bouget, C., and P. Duelli. 2004. The effects of windthrow on forest insect communities: a literature review. Biological Conservation **118**:281–299.

Bouwman, A. F., D. S. Lee, W. A. H. Asman, F. J. Dentener, K. W. Van der Hoek, and J. G. J. Olivier. 1997. A global high-resolution emission inventory for ammonia. Global Biogeochemical Cycles **51**:561–587.

Bovey, R. W. 2001. Woody plants and woody plant management: ecology, safety, and environmental impact. Marcel Dekker, New York, New York, USA.

Brawn, J. D., S. K. Robinson, and F. R. Thompson. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics **32**:251–276.

Brookfield, H. 2001. Exploring agrodiveristy. Columbia University Press, New York, New York, USA.

Brown, D. G., K. M. Johnson, T. R. Loveland, and D. M. Theobald. 2005. Rural land-use trends in the conterminous United States, 1950–2000. Ecological Applications **15**: 1851–1863.

Cairns, R. D., and P. Lasserre. 2004. Reinforcing economic incentives for carbon credits for forests. Forest Policy and Economics **6**:321–328.

Camp, M., and L. B. Best. 1994. Nest density and nesting success of birds in roadsides adjacent to rowcrop fields. American Midland Naturalist **131**:347–358.

Cerveny, R. S., and R. C. Balling, Jr. 1998. Weekly cycles of air pollutants, precipitation and tropical cyclones in the coastal NW Atlantic region. Nature **394**:561–563.

Chalfoun, A. D., M. J. Ratnaswamy, and F. R. Thompson. 2002. Songbird nest predators in forest-pasture edge and forest interior in a fragmented landscape. Ecological Applications **12**:858–867.

Chameides, W. L., and M. Bergin. 2002. Enhanced: soot takes center stage. Science **297**:2214–2215.

Chameides, W. L., R. W. Lindsay, J. Richardson, and C. S. Kiang. 1988. The role of biogenic hydrocarbons in urban photochemical smog: Atlanta as a case study. Science **241**: 1473–1475.

Chameides, W. L., et al. 1999. Case study of the effects of atmospheric aerosols and regional haze on agriculture: an opportunity to enhance crop yields in China through emission controls. Proceedings of the National Academy of Sciences (USA) **96**:13626–13633.

Chase, A. 1987. Playing God in Yellowstone. Harcourt Brace, San Diego, California, USA.

Clevenger, A. P., B. Chruszcz, and K. Gunson. 2001. Drainage culverts as habitat linkages and factors affecting passage by mammals. Journal of Applied Ecology **38**:1340–1349.

Coblentz, B. E. 1990. Exotic organisms: a dilemma for conservation biology. Conservation Biology **4**:261–265.

Covington, W. W. 2000. Helping western forests heal—the prognosis is poor for US forest ecosystems. Nature **408**: 135–136.

Covington, W. W., and M. M. Moore. 1994. Southwestern ponderosa forest structure: changes since Euro-American settlement. Journal of Forestry **92**:39–47.

Cronk, Q. C. B., and J. L. Fuller. 1995. Invasive plants: the threat to natural ecosystems worldwide. Handbook No. 2., WWF Plant Conservation. Chapman and Hall, London, UK.

Crutzen, P. J., and M. O. Andreae. 1990. Biomass burning in the tropics: impacts on atmospheric chemistry and biogeochemical cycles. Science **250**:1669–1678.

Crutzen, P. J., and J. G. Goldammer. 1993. Fire in the environment: the ecological, atmospheric, and climatic importance of vegetation fires. John Wiley and Sons, New York, New York, USA.

Dale, V. H. 1997. The relationship between land-use change and climate change. Ecological Applications **7**:753–769.

Dale, V. H., and S. C. Beyeler. 2001. Challenges in the development and use of ecological indicators. Ecological Indicators **1**:3–10.

Dale, V. H., S. Brown, R. A. Haeuber, N. T. Hobbs, N. Huntly, R. J. Naiman, W. E. Riebsame, M. G. Turner, and T. J. Valone. 2000. Ecological principles and guidelines for managing the use of land. Ecological Applications **10**:639–670.

D'Antonio, C. M., and P. M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annual Review of Ecology and Systematics **23**: 63–87.

de Soyza, A. G., W. G. Whitford, J. O. Herrick, J. O. VanZee, and K. M. Havstad. 1998. Early warning indicators of desertification: examples of tests in the Chihuahuan Desert. Journal of Arid Environments **39**:101–112.

Dregne, H. E. 1983. Desertification of arid lands. Harwood Academic Publishers, Chur, Switzerland.

Dregne, H. E. 2002. Land degradation in the drylands. Arid Land Research and Management **16**:99–132.

Forman, R. T. T. 1995. Land mosaics: the ecology of landscapes and regions. Cambridge University Press, Cambridge, UK.

Forman, R. T., et al. 2002. Road ecology: science and solutions. Island Press, Washington, D.C., USA.

Foster, D., F. Swanson, J. Aber, I. Burke, N. Brokaw, D. Tilman, and A. Knapp. 2003. The importance of land-use legacies to ecology and conservation. BioScience **53**:77–88.

Friedman, J. M., and V. J. Lee. 2002. Extreme floods, channel change, and riparian forests along ephemeral streams. Ecological Monographs **72**:409–425.

Friend, J. J., and K. Y. Chan. 1995. Influence of cropping on the population of a native earthworm and consequent effects on hydraulic properties of vertisols. Australian Journal of Soils Research **33**:995–1006.

Fuhlendorf, S. D., and D. M. Engle. 2001. Restoring heterogeneity on rangelands: ecosystem management based on evolutionary grazing patterns. BioScience **51**:625–632.

Fuhlendorf, S. D., and D. M. Engle. 2004. Application of the fire–grazing interaction to restore a shifting mosaic on tallgrass prairie. Journal of Applied Ecology **41**:604–614.

Galloway, J. N., W. H. Schlesinger, H. Levy, II, A. Michaels, and J. L. Schnoor. 1995. Nitrogen fixation: anthropogenic enhancement–environmental response. Global Biogeochemical Cycles **9**:235–252.

Gilliam, F. S., and W. J. Platt. 1999. Effects of long-term fire exclusion on tree species composition and stand structure in an old-growth *Pinus palustris* (longleaf pine) forest. Plant Ecology **140**:15–26.

Glade, T. 2003. Landslide occurrence as a response to land use change: a review of evidence from New Zealand. Catena **51**:297–314.

Gonzalez, O. J. 1996. Formulating an ecosystem approach to environmental protection. Environmental Management **20**:597–605.

Guenther, A., S. Archer, J. Greenberg, P. Harley, D. Helmig, L. Klinger, L. Vierling, M. Wildermuth, P. Zimmerman, and S. Zitzer. 1999. Biogenic hydrocarbon emissions and landcover/climate change in a subtropical savanna. Physics and Chemistry of the Earth (B) **24**:659–667.

Haack, R. A., K. R. Law, V. C. Mastro, H. S. Ossenbruggen, and B. J. Raimo. 1977. New York's battle with the Asian long-horned beetle. Journal of Forestry **95**:11–15.

Hansen, A. J., R. L. Knight, J. Marzluff, S. Powell, K. Brown, P. H. Gude, and K. Jones. 2005. Effects of exurban development on biodiversity: patterns, mechanisms, and research needs. Ecological Applications **15**:1893–1905.

Hansen, A. J., and J. J. Rotella. 2001. Nature reserves and land use: implications of the ''place'' principle. Pages 54–72 *in* V. H. Dale and R. A. Haeuber, editors. Applying ecological principles to land management. Springer-Verlag, New York, New York, USA.

Hatakeyama, S., K. Izumi, T. Fukuymama, and H. Akimoto. 1989. Reactions of ozone with α-pinene and b-pinene in air: yields of gaseous and particulate products. Journal of Geophysical Research **94**:13013–13024.

Hessburg, P. F., B. G. Smith, and R. B. Salter. 1999. Detecting change in forest spatial patterns from reference conditions. Ecological Applications **9**:1232–1252.

Hewitt, C. N. 1999. Reactive hydrocarbons in the atmosphere. Academic Press, San Diego, California, USA.

Hoffman, W. A., and R. B. Jackson. 2000. Vegetation–climate feedbacks in the conversion of tropical savanna to grassland. Journal of Climate **13**:1593–1602.

Hoffman, T., J. R. Odum, F. Bowman, D. Collins, D. Klokow, R. C. Flagan, and J. H. Seinfeld. 1997. Formation of organic aerosols from the oxidation of biogenic hydrocarbons. Journal of Atmosphere Chemistry **26**:189–222.

Holland, E. A., B. H. Braswell, J. F. Lamarque, A. Townsend, J. M. Sulzman, J. F. Muller, F. Dentener, G. Brasseur, H. Levey,II, J. E. Penner, and G. Roelofs. 1997. Variations in the predicted spatial distribution of atmospheric nitrogen deposition and their impact on carbon uptake by terrestrial ecosystems. Journal of Geophysical Research (Atmospheres) **102**:15 849–15 866.

Holling, C. S. 2001. Understanding the complexity of economic, ecological, and social systems. Ecosystems **4**:390–405.

Houghton, R. A. 2003. Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850–2000. Tellus **55B**:378–390.

Hunter, M. L. 1993. Natural fire regimes as spatial models for managing boreal forest. Biological Conservation **65**: 115–120.

Huston, M. A. 2005. The three phases of land-use change: implications for biodiversity. Ecological Applications **15**: 1864–1878.

Isebrands, J. G., A. B. Guenther, P. Hartley, D. Helmig, L. Klinger, L. Vierling, P. Zimmerman, and C. Geron. 1999. Volatile organic compound emission rates from mixed deciduous and coniferous forecast in Northern Wisconsin, USA. Atmospheric Environment **33**:2527–2536.

Jackson, R. B., J. L. Banner, E. G. Jobbagy, W. T. Pockman, and D. H. Wall. 2002. Ecosystem carbon loss with woody plant invasion of grassland. Nature **418**:623–626.

Johnson, W. C. 1994. Woodland expansion in the Platte River, Nebraska: patterns and causes. Ecological Monographs **64**: 45–84.

Jules, E. S., M. J. Kauffman, W. D. Ritts, and A. L. Carroll. 2002. Spread of an invasive pathogen over a variable landscape: a nonnative root rot on Port Orford cedar. Ecology **83**:3167–3181.

Kaufman, Y. J., D. Tanre, and O. Boucher. 2002. A satellite view of aerosols in the climate system. Nature **419**:215–223.

Keeley, J. E. 2002. Fire management of California shrubland landscapes. Environmental Management **29**:395–408.

Keeley, J. E., D. Lubin, and C. J. Fotheringham. 2003. Fire and grazing impacts on plant diversity and alien plant invasions in the southern Sierra Nevada. Ecological Applications **13**:1355–1374.

Keller, M., and P. A. Matson. 1994. Biosphere–atmosphere exchange of trace gases in the tropics: land use change. Pages 130–117 *in* R. Prinn, editor. Global atmospheric-biospheric chemistry. Kluwer Academic/Plenum Publications, New York, New York, USA.

Klink, C. A., A. G. Moreira, and O. T. Solbrig. 1993. Ecological impact of agricultural development in the Brazilian *cerrados*. Pages 259–282 *in* M. D. Young and O. T. Solbrig, editors. The world's savannas: economic driving forces, ecological constraints and policy options for sustainable land use. Parthenon Publishing Group, Carnforth, UK.

Köchy, M., and S. D. Wilson. 2001. Nitrogen deposition and forest expansion in the northern Great Plains. Journal of Ecology **89**:807–817.

Lal, R. 2003. Global potential of soil carbon sequestration to mitigate the greenhouse effect. Critical Reviews in Plant Sciences **22**:151–184.

Lamb, B., D. Gay, H. Westberg, and T. Pierce. 1993. A biogenic hydrocarbon emission inventory for the U.S.A. using a simple forecast canopy model. Atmospheric Environment **27A**:1673–1690.

Landres, P. B., P. Morgan, and F. J. Swanson. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications **9**:1179–1188.

Le Maitre, D. C., B. W. v. Wilgen, R. A. Chapman, and D. H. McKelly. 1996. Invasive plants and water resources in the western Cape province, South Africa: modelling the consequences of a lack of management. Journal of Applied Ecology **33**:161–172.

Leopold, A. 1948. A Sand County almanac, and sketches here and there. Oxford University Press, New York, New York, USA.

Leslie, M., G. K. Meffe, J. L. Hardesty, and D. L. Adams. 1996. Conserving biodiversity on military lands: a handbook for natural resources managers. The Nature Conservancy, Arlington, Virginia, USA.

Lindesay, J. A., M. O. Andreae, J. G. Goldammer, G. Harris, H. J. Annegarn, M. Garstang, R. J. Scholes, and B. W. Van Wilgen. 1996. International geosphere, biosphere program international global atmospheric chemistry SAFARI-92 field experiment: background and overview. Journal of Geophysical Research-Atmospheres **101**:23 521–23 530.

Liu, G. L., J. D. Nelson, and C. W. Wardman. 2000. A target-oriented approach to forest ecosystem design—changing the rules of forest planning. Ecological Modelling **127**: 269–281.

Ludwig, J., D. Tongway, D. Freudenberger, J. Noble, and K. Hodgkinson, editors. 1997. Landscape ecology, function and management: principles from Australia's rangelands. CSIRO Publishing, Melbourne, Australia.

Mack, M. C., and C. M. D'Antonio. 1998. Impacts of biological invasions on disturbance regimes. Trends in Ecology and Evolution **13**:195–198.

Mack, R. N., D. Simberloff, W. M. Lonsdale, H. Evans, M. Clout, and F. Bazzaz. 2000. Biotic invasions: causes, epidemiology, global consequences and control. Issues in Ecology **5**:1–20.

Madany, M. H., and N. E. West. 1983. Livestock grazing–fire regime interactions within montane forests of Zion National Park, Utah. Ecology **64**:661–667.

Malhi, Y., P. Meir, and S. Brown. 2002. Forests, carbon and global climate. Philosophical Transactions of the Royal Society of London Series A Mathematical Physical and Engineering Sciences **360**:1567–1591.

Martin, R. E., G. P. Asner, R. J. Ansley, and A. R. Mosier. 2003. Effects of woody vegetation encroachment on soil nitrogen oxide emissions in a temperate savanna. Ecological Applications **13**:897–910.

Matson, P. A., C. Billow, and S. Hall. 1996. Fertilization practices and soil variations control nitrogen oxide emissions from tropical sugar cane. Journal of Geophysical Research **18**:533–545.

Mazurek, M. A., G. R. Cass, and B. R. T. Simoneit. 1991. Biological input to visibility reducing aerosol particles in the remote arid southwestern United States. Environmental Science and Technology **25**:684–694.

McCay, D. H. 2000. Effects of chronic human activities on invasion of longleaf pine forests by sand pine. Ecosystems **3**:283–292.

McClaran, M. P., and M. E. Anable. 1992. Spread of introduced Lehmann lovegrass along a grazing intensity gradient. Journal of Applied Ecology **29**:92–98.

McDonald, B. 1995. The formation and history of the Malpai Borderlands Group. Pages 483–486 *in* L. F. De Bano, P. F. Ffolliot, and A. Ortega-Rubio, editors. Biodiversity and management of the Madrean Archipelago: the Sky Islands of southwestern United States and northwestern Mexico. USDA Forest Service General Technical Report RM-GTR-264. Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado, Tucson, Arizona, USA.

McNeely, J. A., editor. 2001. The great reshuffling: human dimensions of invasive alien species. IUCN (The World Conservation Union), Gland, Switzerland.

Menon, S., J. Hansen, L. Nazarenko, and Y. Luo. 2002. Climate effects of black carbon aerosols in China and India. Science **297**:2250–2253.

Meyer, W. B., and B. L. Turner II. 1992. Human population growth and global land-use/cover change. Annual Review of Ecology and Systematics **23**:39–61.

Meyer, W. B., and B. L. Turner II, editors. 1994. Changes in land use and land cover: a global perspective. Cambridge University Press, Cambridge, UK.

Mladenoff, D. J., T. A. Sickley, R. G. Haight, and A. P. Wydeven. 1995. A regional landscape analysis of favorable gray wolf habitat in the northern Great Lakes region. Conservation Biology **9**:279–294.

Moat, D. A., and C. F. Hutchinson, editors. 1995. Desertification in developed countries. Kluwer Academic Publishers, London, UK.

Mooney, H. A., and J. A. Drake, editors. 1986. Ecology of biological invasions of North America and Hawaii. Springer-Verlag, New York, New York, USA.

Mosier, A. R., W. J. Parton, D. W. Valentine, D. S. Ojima, D. S. Schimel, and O. Heinmeyer. 1997. CH_4 and N_2O fluxes in the Colorado shortgrass steppe: 2. Long-term impact of land use change. Global Biogeochemical Cycles **11**: 29–42.

Nazarko, O. M., R. C. Van Acker, M. H. Entz, A. Schoofs, and G. Martens. 2003. Pesticide free production of field crops: results of an on-farm pilot project. Agronomy Journal **95**:1262–1273.

Nevison, C. D., and E. A. Holland. 1997. A reexamination of the impact of anthropogenically fixed nitrogen on atmospheric N_2O and the stratospheric O_3 layer. Journal of Geophysical Research **102**:25 519–25 536.

Offerman, H., V. Dale, S. Pearson, R. Bierregaard, and R. O'Neill. 1995. Effects of forest fragmentation on neotropical fauna: current research and data availability. Environmental Reviews **3**:191–211.

Ojima, D. S., D. W. Valentine, A. R. Mosier, W. J. Parton, and D. S. Schimel. 1993. Effect of land use change on methane oxidation in temperate forest and grassland soils. Chemosphere **26**:675–685.

Paine, R. T., M. J. Tegner, and E. A. Johnson. 1998. Compounded perturbations yield ecological surprises. Ecosystems **1**:535–545.

Parker, I. M., D. Simberloff, W. M. Lonsdale, K. Goodell, M. Wonham, P. M. Kareiva, M. H. Williamson, B. Von Holle, P. B. Moyle, J. E. Byers, and L. Goldwasser. 1999. Impact: toward a framework for understanding the ecological effects of invaders. Biological Invasions **1**:3–19.

Parsons, D. J., T. W. Swetnam, and N. L. Christensen. 1999. Uses and limitations of historical variability concepts in managing ecosystems. Ecological Applications **9**:1177–1178.

Perkins, T. E., and G. R. Matlack. 2002. Human-generated pattern in commercial forests of southern Mississippi and consequences for the spread of pests and pathogens. Forest Ecology and Management **157**:143–154.

Persoons, E., M. Vanclooster, and A. Desmed. 2002. Flood hazard causes and flood protection recommendations for Belgian river basins. Water International **27**:202–207.

Peterson, D. W., and P. B. Reich. 2001. Prescribed fire in oak savanna: fire frequency effects on stand structure and dynamics. Ecological Applications **11**:914–927.

Pimentel, D. 2000. Soil erosion and the threat to food security and the environment. Ecosystem Health **6**:221–226.

Pimentel, D., and N. Kounang. 1998. Ecology of soil erosion in ecosystems. Ecosystems **1**:416–426.

Pimentel, D., and M. Pimentel. 2003. Sustainability of meat-based and plant-based diets and the environment. American Journal of Clinical Nutrition **78**:660–663.

Pimentel, D., and E. L. Skidmore. 1999. Rates of soil erosion. Science **286**:1477–1477.

Pimm, S. L., G. J. Russell, J. L. Gittleman, and T. M. Brooks. 1995. The future of biodiversity. Science **269**:347–350.

Powell, K. 2002. Open the floodgates. Nature **420**:356–358.

Rasmussen, R. A., and F. W. Went. 1965. Volatile organic material of plant origin in the atmosphere. Proceedings of the National Academy of Science (USA) **53**:215.

Robertson, G. P., J. C. Broome, E. A. Chornesky, J. R. Frankenberger, P. Johnson, M. Lipson, J. A. Miranowski, E. D. Owens, D. Pimentel, and L. A. Thrupp. 2004. Rethinking the vision for environmental research in US agriculture. BioScience **54**:61–65.

Rosenfeld, D. 2000. Suppression of rain and snow by urban and industrial air pollution. Science **287**:1793–1796.

Savage, M., and T. W. Swetnam. 1990. Early 19th-century fire decline following sheep pasturing in a Navajo ponderosa pine forest. Ecology **71**:2374–2378.

Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virginia, and W. G. Whitford. 1990. Biological feedbacks in global desertification. Science **247**:1043–1048.

Scholes, R. J., and S. R. Archer. 1997. Tree–grass interactions in savannas. Annual Review of Ecology and Systematics **28**:517–544.

Scifres, C. J. 1980. Brush management: principles and practices for Texas and the Southwest. Texas A & M University Press, College Station, Texas, USA.
Scifres, C. J., and W. T. Hamilton. 1993. Prescribed burning for brushland management: the South Texas example. Texas A & M Press, College Station, Texas, USA.
Scifres, C. J., W. T. Hamilton, J. R. Conner, J. M. Inglis, G. A. Rasmussen, R. P. Smith, J. W. Stuth, and T. G. Welch. 1985. Integrated brush management systems for south Texas: development and implementation. Texas Agricultural Experiment Station, College Station, Texas, USA.
Scifres, C. J., W. T. Hamilton, B. H. Koerth, R. C. Flinn, and R. A. Crane. 1988. Bionomics of patterned herbicide application for wildlife habitat enhancement. Journal of Range Management **41**:317–321.
Sharov, A. A., and A. M. Liebhold. 1998. Model of slowing the spread of gypsy moth (Lepidoptera: Lymantriidae) with a barrier zone. Ecological Applications **8**:1170–1179.
Shochat, E., W. L. Stefanov, M. E. A. Whitehouse, and S. H. Faeth. 2004. Urbanization and spider diversity: influences of human modification of habitat structure and productivity. Ecological Applications **14**:268–280.
Shukla, J., C. Nobre, and P. Sellers. 1990. Amazon deforestation and climate change. Science **247**:1322–1325.
Simpson, D., A. Guenther, C. N. Hewitt, and R. J. Steinbrecher. 1995. Biogenic emissions in Europe. 1. Estimates and uncertainties. Journal of Geophysical Research **100**:22 875–22 890.
Smil, V. 1999. Nitrogen in crop production: an account of global flows. Global Biogeochemical Cycles **13**:647–662.
Strittholt, J. R., and D. A. DellaSala. 2001. Importance of roadless areas in biodiversity conservation in forested ecosystems: case study of the Klamath–Siskiyou ecoregion of the United States. Conservation Biology **15**:1742–1754.
Swanson, F. J., and C. T. Dyrness. 1975. Impact of clearcutting and road construction on soil erosion by landslides in the western Cascade Range, Oregon. Geology **3**:393–396.
Teague, W. R., S. L. Dowhower, and J. A. Waggoner. 2004. Drought and grazing patch dynamics under different grazing management. Journal of Arid Environments **58**:97–117.
Theobald, D. M., T. Spies, J. Kline, B. Maxwell, N. T. Hobbs, and V. H. Dale. 2005. Ecological support for rural land-use planning. Ecological Applications **15**:1906–1914.
Tongway, D., and J. Ludwig. 1997. The nature of landscape dysfunction in rangelands. Pages 49–61 *in* J. Ludwig, D. Tongway, D. Freudenberger, J. Noble, and K. Hodgkinson, editors. Landscape ecology, function and management: principles from Australia's rangelands. CSIRO Publishing, Melbourne, Australia.
United Nations Food and Agricultural Organization. 2002. FAOSTAT: agricultural data. ⟨http://www.apps.fao.org⟩
Valentine, J. F. 1997. Range developments and improvements. Academic Press, Provo, Utah, USA.
van Auken, O. W. 2000. Shrub invasions of North American semiarid grasslands. Annual Review of Ecology and Systematics **31**:197–215.
Vavra, M., W. Laycock, and R. Pieper, editors. 1994. Ecological implications of livestock herbivory in the West. Society for Range Management, Denver, Colorado, USA.
Verstraete, M. M. 1986. Defining desertification: a review. Climatic Change **9**:5–18.
Vitousek, P. M., J. Aber, R. W. Howarth, G. E. Likens, P. A. Matson, D. W. Schindler, W. H. Schlesinger, and D. G. Tilman. 1997*b*. Human alteration of the global nitrogen cycle: causes and consequences. Issues in Ecology **1**:1–15.
Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J. M. Melillo. 1997*a*. Human domination of Earth's ecosystems. Science **277**:494–499.
Vitousek, P. M., and L. R. Walker. 1989. Biological invasion by *Myrica faya* in Hawai'i: plant demography, nitrogen fixation, ecosystem effects. Ecological Monographs **59**:247–265.
Watson, R. T., I. R. Noble, B. Bolin, N. H. Ravindranath, D. J. Verardo, and D. J. Dokken, editors. 2000. Land use, land-use change, and forestry. Special report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.
Weltzin, J. F., S. Archer, and R. K. Heitschmidt. 1997. Small-mammal regulation of vegetation structure in a temperate savanna. Ecology **78**:751–763.
Wesely, M. L., and B. B. Hicks. 2000. A review of the current status of knowledge on dry deposition. Atmospheric Environment **34**:2261–2282.
Wessman, C. A., S. Archer, L. C. Johnson, and G. P. Asner. 2004. Woodland expansion in US grasslands: assessing land-cover change and biogeochemical impacts. Pages 185–205 *in* G. Gutman, A. C. Janetos, C. O. Justice, E. F. Moran, J. F. Mustard, R. R. Rindfuss, D. Skole, B. L. Turner, II, and M. A. Chocharane, editors. Land change science: observing, monitoring and understanding trajectories of change on the Earth's surface. Kluwer Academic Publishers, New York, New York, USA.
Whisenant, S. G. 1999. Repairing damaged wildlands. Cambridge University Press, Cambridge, UK.
Wilcove, D. S., D. Rothstein, J. Dubow, A. Phillips, and E. Losos. 1998. Quantifying threats to imperiled species in the United States. BioScience **48**:607–615.
Wilcox, B. P. 2002. Shrub control and streamflow on rangelands: a process-based viewpoint. Journal of Range Management **55**:318–326.
Wilkening, K. E. 2001. Trans-Pacific air pollution: scientific evidence and political implications. Water, Air, and Soil Pollution **130**:1825–1830.
Wissmar, R. C., R. K. Timm, and M. G. Logsdon. 2004. Effects of changing forest and impervious land covers on discharge characteristics of watersheds. Environmental Management **34**:91–98.
Wondolleck, J. M., and S. L. Yaffee. 2000. Making collaboration work: lessons from innovation in natural resource management. Island Press, Washington, D.C., USA.
Wu, J., and R. Hobbs. 2002. Key issues and research topics in landscape ecology. Landscape Ecology **17**:355–365.
Zedler, J. B. 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. Frontiers in Ecology and the Environment **1**:65–72.