Annals of Arid Zone 43(3&4): 1-2, 2004

Challenges and Opportunities for Conserving Faunal Biodiversity in Arid Ecosystems

Erica Fleishman[1], Ralph Mac Nally[2] and James R. Thomson[2]

[1] *Center for Conservation Biology, Department of Biological Sciences, Stanford University, Stanford, CA 94305-5020, USA,*

[2] *Australian Centre for Biodiversity: Analysis, Policy and Management, School of Biological Sciences, Monash University 3800, Australia*

Abstract: Increasing competition for limited sources of water and primary productivity in arid and semi-arid landscapes poses substantial challenges for conserving the distinctive faunal biodiversity of these ecosystems. Natural climatic severity, human population growth, and anthropogenic climate change constrain opportunities for ecological restoration and rehabilitation. Nonetheless, although it may not be possible to restore systems to a historical or undisturbed state, existing management frameworks can be used to develop efficient strategies for faunal conservation with a high probability of success. We outline the threats to faunal biodiversity in arid ecosystems from climate change, agriculture (cultivation of crops and livestock grazing), modification of fire regimes, non-native invasive species, and localized urbanization. We also examine practical alternatives for addressing those threats. In particular, an integrated approach to valuation of environmental assets that incorporates both socio-economics and ecology is likely to gain community support and provide a realistic foundation for effective adaptive management. We describe how the approach has been implemented for riparian systems in Victoria, Australia, and explore how it can be generalized to conserve faunal biodiversity of arid zones worldwide.

Key words: adaptive management, agriculture, climate change, environmental assets, invasive species, land use

Status of Faunal Biodiversity in Arid Ecosystems

In the eyes of a large proportion of the public, and even many biologists, arid ecosystems appear relatively unappealing. Many deserts and semi-deserts, which we refer to collectively as arid zones, are bitterly cold during the winter months, stiflingly hot during the summer, and, by definition, dry. In terms of faunal species richness and endemism, however, arid ecosystems Belie their popular reputation as ecological wastelands. Australia's Great Sandy Desert, for example, has more species of reptiles than any other ecosystem in the world (Cogger, 1992; Ricketts *et al.*, 1999). The Chihuahuan Desert in North America is believed to support more species of birds, mammals, and butterflies than any other Nearctic ecosystem and is ranked as globally outstanding in species richness of reptiles, birds, mammals, and cacti (Olson and Dinerstein, 1998; Ricketts *et al.*, 1999).

The heterogeneous topography, soils, and climate characteristics of many arid ecosystems, in conjunction with isolation of critical resources such as water, also leads to remarkably high levels of endemism of both terrestrial and aquatic taxa. Within the United States and Canada, the Great Basin and Colorado Plateau have notably high levels of endemism among vascular plants, and the Sonoran Desert has more endemic species of birds than any other ecosystem (Ricketts *et al.*, 1999). Long-term isolation of springs and other sources of water appears to have facilitated the evolution of several hundred endemic species and subspecies of fishes, mollusks, crustaceans, aquatic insects, and plants in the Mojave Desert alone (Hubbs and Miller, 1948; Hershler *et al.*, 2002).

Water is the most limited resource in arid ecosystems (Huxman *et al.* 2004). Increasing exploitation of water by humans means that future conservation of faunal biodiversity in these regions will require concerted human action. In dry landscapes, flowing and standing water provide support for terrestrial as well as aquatic and riparian taxa. For example, approximately 80% of the terrestrial species of animals in arid western North America are facultative or obligate users of riparian ecosystems (Thomas *et al.*, 1979). Concentration of human land uses near reliable sources of water poses threats to native faunal biodiversity that are likely to intensify under current projections of population growth and climate change.

Some 70% of the world's arid lands– more than 36 million km^2– are ecologically degraded (UNCCD, 2004). Thus, although the total extent of deserts and semi-deserts is increasing globally in response to human activities and climate change, the extent and continuity of relatively undisturbed arid lands and the quantity and quality of resources for native animals are decreasing. Moreover, the effects of environmental changes such as shifts in temperature and precipitation, invasion by non-native species, and altered disturbance regimes may be less reversible in arid regions than in other ecosystems (Sala *et al.*, 2000; Smith *et al.*, 2000). Arid ecosystems are thought to have relatively low ecological elasticity (capacity to rebound from disturbance) and ecological resistance (tolerance for perturbation without changing state) to environmental change, and recovery times may be longer than in more mesic landscapes (Sala *et al.*, 2000; Smith *et al.*, 2000).

Studying and Conserving Arid Ecosystems

Several hurdles impede development of the scientific basis for conservation and land-use planning in arid zones. First, baseline climatic severity and unpredictability complicate efforts to quantify biotic responses to natural and anthropogenic environmental change against background "noise" (Houghton *et al.*, 1975; Grayson, 1993). In particular, differentiating effects of human land use from natural variability typically requires long-term studies across large areas. For example, the nominal latitudinal boundary of sub-Saharan vegetation in Africa can move by as much as 200 km between years in response to changes in precipitation (UNCCD, 2004). Species with low tolerance for natural or human-induced environmental variability may have been winnowed out long before contemporary investigations, leaving the more resilient and cosmopolitan species and a homogenized biota (Lockwood and McKinney, 2001). Consequently, we cannot necessarily infer which species have already been extirpated by human land uses or which species may be lost to intensified activities in the future.

Another obstacle to conservation and management-oriented research in arid environments is that many ecologists, drawn to the spectacular biodiversity of the tropics, historically have considered arid landscapes to be biologically austere and unremarkable. Climatic extremes, rugged topography, and low human population density in arid zones also have posed logistic challenges to scientific exploration (Fleishman *et al.*, 2004). As a result, there is little long-term information on species distributions, reference conditions, and land use and land cover change. Historical occurrence records usually are credible, especially if the observer was a reputable naturalist. Poor documentation of survey locations, however, compounded by uneven distribution of sampling effort, means that lack of data rarely can be interpreted as legitimate evidence of absences. One consequence is that records from museums and the archives of resource agencies may misrepresent species distributions across extensive landscapes.

Issues related to popularity, human population density, and accessibility continue to plague present-day surveys. Citizen-based monitoring programs illustrate the disparity in sampling effort per unit area in deserts compared with mesic and human ecosystems. During the 2003 Breeding Bird Survey in the United States, 16 transects were counted in Nevada, an arid state with an area of 286,000 km^2, in comparison with 48 transects in the mesic state of Maryland (32,000 km^2) (Sauer *et al.*, 2003). Birds Australia conducted a continent-wide survey program in the late 1990s, amassing records for 279,000 surveys. A map of survey locations provides a good rendition of roads across the ca 70% of Australia that is arid or semi-arid, whereas the overlay of surveys in agricultural, mesic areas in southwestern and southeastern Australia is a virtual blur of sampled locations (Barrett *et al.*, 2003). Although appreciation of the ecological and aesthetic value of arid lands is increasing, lush tropical systems still seem to garner a disproportionate share of conservation attention and funds.

Here, we describe five major phenomena that directly or indirectly lead to loss and degradation of habitat for faunal biodiversity in arid ecosystems: climate change, agriculture (cultivation of crops and livestock grazing), modification of fire regimes, non-native invasive species, and localized urbanization. Although these environmental changes are not confined to arid zones, they are perhaps the most daunting obstacles to conservation of biodiversity in those regions. We outline the challenges that each threat poses for protection of native animals and their resources, and, where possible, we examine realistic alternatives for meeting those challenges. Finally, we highlight a promising model for ecosystem rehabilitation that is being implemented successfully in Victoria, Australia. Incorporating both socio-economics and ecology, the model appears to be an efficient potential approach for conserving faunal diversity in arid zones around the world.

Climate Change

Climate, the most prevalent environmental driver on natural systems, determines ecological potential in the absence of human land use and in the context of restoration or reconstruction efforts. During the next 100 years, both drought and precipitation events in most arid regions are likely to become more extreme and frequent, posing challenges for the native biota (Rind *et al.*, 1990; IPCC, 2001).

Montane islands, or mountain tops, often have relatively large numbers of native and endemic species (Wilcox, 1980; Myers, 1986, Guisan *et al.*, 1995). In arid landscapes, steep gradients in elevation and land cover frequently isolate montane species from lower-elevation basins and from conspecific individuals in nearby mountain ranges. As existing patterns of temperature and precipitation are modified by climate change, many of these species may face local extirpation unless the species can either shift their distributions to track necessary resources or adapt in situ (Parmesan *et al.*, 1999; Thomas and Lennon 1999; Saether *et al.*, 2000).

Using principles of the equilibrium theory of island biogeography (MacArthur and Wilson, 1967), scientists have tried to predict how montane faunas in desert environments may respond to climate change. In the early 1990s, researchers began to project the impacts of increasing temperature in the Great Basin of western North America on mammals and butterflies. Early forecasts were based on assumptions of strong positive relationships between species richness and area, a weak positive relationship between species richness and isolation from potential sources of colonists, and tight associations between specific assemblages of animals and plants (McDonald and Brown, 1992; Murphy and Weiss, 1992; Boggs and Murphy, 1997). McDonald and Brown (1992) predicted that a 3°C rise in temperature would lead to the extirpation of 9-62% of the montane mammals in various Great Basin mountain ranges and 21% of the mammals in the Great Basin as a whole.

Subsequently, using larger data sets, workers reexamined initial predictions concerning biological consequences of warming trends in the Great Basin (Lawlor, 1998; Grayson, 2000; Grayson and Madson, 2000; Fleishman *et al.*, 2001). It now appears that present-day assemblages are considerably more dynamic than previously thought (Brown, 1971, 1978), and relatively few species are likely to be lost from the Great Basin as a whole (Lawlor, 1998; Fleishman *et al.*, 2001). However, the number of losses at the mountain-range level may differ considerably among ranges (Fleishman *et al.*, 2001).

Whether animals can retreat to potential high-elevation refugia will depend in part on their ability to disperse sufficiently rapidly to deal with climatic changes along the elevational gradient. If food, suitable microclimates, and other resources gradually move higher, animals may be able to move in concert. However, if certain slope exposures (e.g., south- and west-facing slopes) become disproportionately warmer and drier, organisms with limited dispersal ability may be unable to disperse to areas with suitable conditions, and local extinctions may result (Murphy and Weiss, 1992; Fleishman *et al.*, 1998). Moreover, environmental variability will likely be positively correlated with elevation (Hidy and Klieforth, 1990). Therefore, as animals move higher in response to climate change, they probably will encounter progressively more variable weather and more extreme events. Recent work has emphasized that faunal responses to environmental change may depend in part on the speed of change and the extent to which the variance in ecological conditions shifts (Grayson, 2000; McLaughlin *et al.*, 2002).

One strategy for conserving large mammals in arid ecosystems is to establish networks of latitudinal and elevational corridors for migration (Berger, 2004). Corridors should link both currently occupied locations and locations that may become suitable for target species in the future, on time scales of decades to centuries. Native species have a much greater probability of adapting to climate change if access to potential habitat is not limited by urbanization and other intensive human land uses.

Agriculture

Human occupants of arid zones traditionally have coped with fluctuations in climate by being nomadic. As settlements have become more permanent, human capacities to respond to climatic variability have decreased and environmental degradation has increased. At present, most arid-land agriculture, which we define broadly to include both cultivation of crops and livestock grazing, requires irrigation. The resulting alteration of land cover, hydrology, and morphology of natural waterways frequently leads to loss of aquatic and riparian biodiversity. Human appropriation of groundwater often increases the mean vertical distance between the land surface and the water table, which reduces flow into rivers, dewaters springs and seeps, and drives vegetation across often irreversible thresholds to alternative stable states (Chambers *et al.*, 2004).

Extensive use of land for production of food reduces and fragments natural land cover and converts heterogeneous native vegetation to virtual monocultures of non-native food crops (Fensham and Fairfax, 1997; Miller *et al.*, 1997; Bennett 1999). These changes in land cover jeopardize faunal biodiversity in many ways. For example, loss and fragmentation of dry forests and woodlands disproportionately affects woodland-interior species that cannot use edges between land cover types (Mac Nally, 1999). Ecological "release" of native species in response to extensive fragmentation can have cascading effects on biodiversity when the intensity of ecological interactions increases. For example, vast tracts of dry woodlands across southeastern Australia have been cleared for wheat and sheep farming, creating very large numbers of small remnants of native vegetation. These remnants become dominated by a native species of bird, the noisy miner (*Manorina melanocephala*), which aggressively excludes many species of small insectivorous birds. This shift causes local depression of species richness of birds and declines in vegetation health (Grey *et al.*, 1998; Major *et al.*, 2001).

Loss of understory from livestock grazing modifies fire regimes and increases susceptibility to invasion of non-native plants. Conversely, abandonment of historically cultivated land also threatens some species. Relatively small-scale, non-industrialized agriculture can maintain open, early successional habitats that are favored by some taxonomic groups, such as terrestrial invertebrates and small mammals

(New, 1991; Feber *et al.*, 1996; Morrison *et al.*, 1996). Modern, mechanized agriculture, often characterized by extensive monocultures and reliance on industrial biocides, has different effects on faunal assemblages than less intensive methods of cultivation (Erhardt, 1985).

Negative impacts of agriculture on faunal biodiversity – and economic potential – are exacerbated by salinization, desertification, and sodalization. Salinization refers to accumulation of water-soluble salts in soil, which inhibits the ability of native and domesticated vegetation to utilize water. Although salinization can result from natural causes, high rates of evaporation and low levels of precipitation increase the probability that irrigation in dry environments will lead to salinization. Southeastern and southwestern Australia are in the grip of a salinity-driven ecological and economic catastrophe due to the loss of vast numbers of trees (Walker *et al.*, 1993), which has reduced evapotranspiration and led to rising water tables (Pavelic *et al.*, 1997; Greiner, 1998; Stirzaker *et al.*, 1999). As water tables rise, salt is drawn towards the surface, killing native vegetation and causing extensive salt scars (Yates and Hobbs, 1997). Large-scale replanting of deep-rooted vegetation seems to be the only feasible solution to the problem.

Desertification generally refers to loss of vegetation and concomitant erosion that diminish biological and economic productivity and resilience to environmental variability. Desertification, like other forms of disturbance, also may facilitate invasions of non-native plants with little value as forage for either wildlife or livestock. Further, when agriculture must be abandoned, human need for food often increases local hunting pressure.

Sodic soil refers to soil from which much of the chlorine has been washed away, leaving sodium ions attached to particles of clay. Those particles lose their tendency to adhere when wet, leading to erosion or impermeability to water and roots. In addition, runoff from sodic soils increases turbidity and concentrations of nutrients in natural and anthropogenic waterways. In some regions, sodicity is even more extensive than salinity. For example, sodicity affects nearly one-third of the land in Australia (NLWRA, 2002). Soil compaction from overstocking of hoofed ungulates is yet another problem. Germination rates of native grasses, forbs, and trees are quite low in compacted soils (Yates *et al.*, 2000, Maestre *et al.*, 2003), which can stall natural regeneration when grazing pressures are released (Martin and Chambers, 2002).

Although agricultural land uses are not benign, the adverse effects of well-managed agriculture may be more reversible and compatible with preservation of faunal biodiversity than the effects of other land uses, such as urbanization. Irrigation, for instance, creates artificial riparian areas and wetlands that may support some native species, especially when natural aquatic ecosystems have been severely degraded. We found that species richness of butterflies in artificially created riparian areas near the eastern edge of the Sierra Nevada (California and Nevada, USA) was not significantly lower than in native riparian habitat (Fleishman *et al.* 1999). Assemblages of butterflies in artificially created riparian areas, however, were dominated numerically by a few geographically widespread and opportunistic species, and species composition in native habitat was more heterogeneous than in artificial riparian habitat (Nelson and Andersen, 1994; Fleishman *et al.*, 1999). While riparian areas created to support agriculture are not substitutes for comparatively undisturbed systems, their support of some native fauna is important for predicting the impacts of proposed water redistribution schemes on ecological diversity and function.

Fire Regimes

Before European settlement, wildfires and human-ignited fires had major influences on vegetation structure and composition on several continents. Plants and animals were adapted to a shifting mosaic of land cover in which patchy, low-intensity fire events were relatively common. In the arid western United States, for instance, pre-settlement fires are thought to have occurred at intervals from 12-25 years to 100-200 years, depending on the plant community. In arid ecosystems during the past several centuries, however, natural and anthropogenic changes in climate, introduction of livestock grazing, suppression of wildfire, and cessation of human-ignited fire have led to an increase in the density and age-class distribution of trees and a decrease in species richness and per cent cover of grasses, forbs, and shrubs. In the Great Basin, for example, piñon and juniper trees have increased by two to five times in spatial extent and two to twenty times in density since 1870. Fire suppression over extensive areas of New South Wales and Queensland has led to the expansion of perennial woody vegetation, causing changes from relatively open grasslands or woodlands to contiguous scrub thickets (Radford *et al.*, 2001). This is referred to as the "woody weed" problem in Australian rangelands.

As the proportion of woody species relative to herbaceous species increases, due in part to longer fire return intervals, so do fuel loads and fire severity. Accordingly, the probability of high-intensity crown fires that induce shifts from woodlands dominated by native species to grasslands dominated by non-native invasive species is increasing rapidly. At the same time, expansion of invasive plants, especially herbaceous taxa, is decreasing fire return intervals and increasing the spatial extent of fires–conditions that also favor continued spread of invasive woody plants such as *Lantana* sp. (Duggin and Gentle, 1998).

As the human population in arid regions expands into areas formerly categorized as rural or wildlands, the probability that fire will result in loss of human life, property, or recreational and economic opportunities rises. Locations in which people, homes, and other urban and suburban structures lie in close proximity to large amounts of natural vegetation have been dubbed the "urban-wildland interface."

Prescribed fire has been suggested as an efficient way to reduce the probability of high-intensity fires and the concomitant expansion of non-native invasive species. As with any major disturbance, prescribed fire can have undesirable effects on ecosystem characteristics such as infiltration, soil erosion, availability of soil nutrients, stream flow, sediment transport, and water quality (Carignan and Steedman, 2000; Earl and Blinn, 2003). These effects may be transient or enduring depending on the spatial extent and intensity of the fire, soil and vegetation characteristics before and after the fire, topography, and weather patterns. Manipulation of fire and other disturbance patterns also can be prohibitively expensive and politically contentious, and short-term and long-term benefits differ among species. Well-designed experimental burns, especially when conducted by interdisciplinary teams of scientists and land managers, are an excellent source of information on the diverse effects of prescribed fire under different environmental circumstances (Williams *et al.*, 2003).

Invasive Species

Invasive non-native species modify ecosystem processes, species distributions, and population dynamics of native species worldwide, posing major challenges to conservation of faunal biodiversity (Mooney and Hobbs, 2000; Rejmanek, 2000; Palumbi, 2001). In arid ecosystems, the problem is perhaps greatest along rivers and streams, where invasives are changing vegetation structure and composition. Anthropogenic changes in flow regimes, particularly reduction of the frequency and intensity of floods, tend to accelerate invasion of non-native plants and animals while decreasing recruitment of dominant native trees that provide habitat for riparian and aquatic wildlife (Eby *et al.*, 2003; Jenkins and Boulton, 2003; Lytle and Poff, 2004).

Among the most serious potential consequences of species invasions is biotic homogenization. Biotic homogenization disrupts coevolved interactions among species; decreases genetic, taxonomic, and functional diversity and therefore evolutionary potential at multiple levels of biological organization; affects transfer of matter and energy; and may decrease the resilience of an ecosystem to future disturbance (Kinzig *et al.*, 2001; Soulé *et al.*, 2003; Olden *et al.*, 2004). For example, establishment of invasive plants with phenological patterns different from those of native plants reduces the ability of native herbivores to persist in the face of environmental variability.

Human settlement serves as a conduit for introduction of non-native species that compete directly and indirectly with native animals in arid environments. In many rural landscapes, feral livestock, including horses, pigs, goats, and camels, vie with native animals for water and forage. In suburban areas, feral cats, foxes, and dogs are a major source of mortality among native songbirds (Bolger *et al.*, 1991). Predation by feral red foxes (*Vulpes vulpes)* and domestic cats *(Felis cattus*) are among the most damaging effects on biodiversity across the arid and semi-arid regions of Australia (Banks *et al.*, 1998; Banks, 1999). Non-native fishes and amphibians, many of which were introduced for recreational fisheries, also compete with their native counterparts (Dunham *et al.*, 2002). The cane toad (Bufo marinus), introduced into Australia to control cane sugar pests, has spread over almost one-third of the continent and has left a trail of ecological havoc (Lampo and De Leo, 1998).

Understanding how assemblages change in response to the intrusion of non-native species is critical to development of practical strategies for ecological restoration and maintenance. The preferred course of restoration action might seem clear, albeit expensive and ecologically challenging–eradicate the invaders. Developing an effective course of action, however, is not so simple. Many instances of invasions of non-native species that did not result in extinction of native species have been documented (Simberloff, 1981), and many native species of animals can adapt to ecological changes associated with increasing dominance of certain invasives. In the southwestern United States, for example, water availability has been reduced by high evapotranspiration rates of introduced salt-cedar (*Tamarix ramosissima*), which has become established in riparian communities formerly dominated by native trees (Cleverly *et al.*, 1997; Sher *et al.*, 2000). Several species protected by the U.S. Endangered Species Act, most notably the endangered Southwestern Willow Flycatcher (*Empidonax traillii extimus*), are able to exploit areas that now are dominated by *T. ramossissima.*

In many situations, unless native riparian vegetation can be restored immediately following elimination of non-native species, rehabilitation efforts inadvertently may threaten native biota (Zavaleta *et al.* 2001). For example, non-native willows (*Salix* spp.) have been planted extensively to arrest erosion along the banks of streams and rivers in Australia. Some restoration ecologists have argued that the willows should not be eradicated without a concomitant, parallel plan to reestablish native woody riparian vegetation (Cremer, 1999; Smith and Starr, 1999; Williams and West, 2000).

Management responses to invasive species must recognize that during intermediate stages of invasion, certain types of management actions could put native species at risk. For instance, clearcutting effectively has removed thousands of individual *T. ramosissima* from riparian corridors in the Mojave Desert, and therefore has left large areas with neither a woody overstory; a diverse understory of shrubs, forbs, and grasses; nor critical ecological processes to help restore native vegetation. Such severe simplification of the vegetation could have adverse effects on the occurrence and persistence of many faunal groups (Fleishman *et al.*, 2003; Rood *et al.*, 2003). If maintenance of species richness of native animals is a high priority, it may be preferable to remove non-native plants in a patchwork or mosaic pattern and, if possible, to accompany selective removal of non-native plants with focused efforts to reestablish native plants with life forms similar to those that were removed. This will help maximize heterogeneity of plant composition and structure while leading toward eradication of non-native invaders. Moreover, spatially discontinuous eradication efforts may help to minimize evolution of resistance to biological and chemical control agents (Stockwell *et al.*, 2003).

Urbanization

Localized urbanization – the growth and expansion of commercial, residential, and industrial areas and transportation networks – is another major driver of loss and degradation of habitat for faunal biodiversity in arid ecosystems. The spatial configuration of urban growth and specific changes in land use and land cover associated with urbanization differ geographically, and it is nearly impossible to draw generalizations that are applicable worldwide. Contiguous urban sprawl tends to be more prevalent in western countries. In Asia, most land-use changes involve a shift from agriculture to urbanization, and it is more difficult to separate urban from rural uses. Urbanization in Asia often occurs not only occurs in a centralized location but also between and beyond agricultural areas. In essence, agricultural areas become more urban, although humans do not necessarily relocate to metropolitan areas proper. The resulting landscape pattern, termed "desakota" (McGee, 1991), is a heterogeneous matrix of patches of dense human settlement, traditional agriculture, and intensified agriculture.

In the United States, urbanization threatens more species than any other land use (Czech *et al.*, 2000). Urbanization usually results in an increase in impervious surfaces such as concrete, with adverse ecological effects (Walsh *et al.*, 2004), and a reduction in cover of native vegetation. In arid and semiarid ecosystems, urbanization frequently increases vegetational cover, although the structure and composition of vegetation in urban areas (e.g., lawns and non-native trees) is quite different from the original vegetation (Nowak *et al.*, 1996). As a result, species richness often peaks in an intermediate location along an urbanization gradient. For example, golf courses may have more species of birds and butterflies than either nature preserves or business districts (Blair, 1999). However, a high proportion of the species found in locations with intermediate levels of urbanization are relatively widespread, whereas many native species are restricted to locations with comparatively little urban development despite relatively low species richness in those locations (McKinney, 2002).

Urbanization in desert environments can modify the physical environment in ways different from mesic systems. Many cities in temperate regions, such as New York City, have a "heat island effect" in which ambient air temperatures in urban areas are several degrees warmer than in the surrounding landscape (Pickett *et al.*, 2001). In deserts, irrigation and plantings have cooling effects through evaporation (Brazel *et al.*, 2000). The effects of reductions in groundwater and evapotranspiration, common to virtually all urban areas, are particularly severe in arid ecosystems. When urban demands for water exceed local supplies, the speed, magnitude, and spatial extent of water appropriation from surface flows and below-ground aquifers increase rapidly. Urban dewatering of deserts, especially in conjunction with other human land uses and climate change, reduces availability of the most essential resource for native animals.

Frameworks for Management of Biodiversity

The essential components of any effective management process are to identify (1) the entity (or entities) to be managed, (2) the processes influencing that entity, (3) the risks of deleterious effects on the entity and the opportunities for improving the entity, (4) an effective, efficient means by which to monitor the state of the entity, and (5) a process by which management can be modified if the state of the entity is changing in an undesirable way. These five steps represent the simplified foundation of an adaptive management process (see Walters, 1986).

The government in the state of Victoria, Australia, has developed a coherent framework for management of its natural resources. Because the entities to be managed are viewed in different ways by different groups of stakeholders, whose perspectives must be reconciled, the Victorian model eliminates the distinction between natural resources and other social and economic entities valued by stakeholders. Use of the term "asset" (DNRE, 2002) to describe managed entities brings natural resources into a common valuation framework with other goods and services. Assets may be classified as environmental, social, or economic; consensus-based delineation of assets streamlines concurrent management processes for multiple entities. Environmental assets are assigned to one of four major groups on the basis of their rarity, representativeness, naturalness, or large-scale significance. Although the framework primarily has been applied to management of rivers, we believe that with appropriate modification for local ecology and sociopolitical circumstances, the integrated strategy applies equally well to management of resources and habitat for wildlife in arid and semi-arid regions (**Table 1**).

A key issue is the prioritization of management effort (Hobbs and Kristjanson, 2003). Funding, political will, and staff inevitably are too limited to manage simultaneously all environmental and ecological assets. Therefore, we must make value-based judgments about the relative importance of assets, a process often referred to as "triage" (DeKay and McClelland, 1996; Restani and Marzluff, 2001). The Victorian government explicitly adopted the triage approach by attaching high significance to "heritage" rivers, most of which are in outstanding natural condition, and representative rivers, the best examples of rivers of different types (e.g., coastal rivers, rivers flowing through mesic or xeric landscapes) (**Table 2**). Lower management priority is assigned to rivers, or river reaches, that are mostly in good condition and could be improved markedly with relatively little added investment. Although heavily degraded rivers often received the majority of funding for rehabilitation and restoration in the past, the current model of management considerably reduces investment in these areas.

The main lesson of the Victorian framework in the context of conservation management of arid ecosystems is that it is critical to identify not only assets but also spatially-explicit threats to the assets, the probability that those threats will occur, opportunities to ameliorate threats, and the costs involved in risk abatement and restoration. This basic information is essential for prioritizing management efforts across vast arid and semi-arid landscapes.

Recently, the government of Australia invested heavily in measuring assets and threats across the continent's extensive rangelands. Arid and semi-arid rangelands constitute ca 75% of the area of Australia (Commonwealth of Australia 2001), but are occupied by only 13% of its population (2.7 million people). As a result, relatively few observers are available to gather data

Table 1. Valuation of environmental assets for rivers in the state of Victoria, Australia (DNRE, 2002) and a potential valuation of assets for faunal biodiversity in arid ecosystems

Broad classification	Detailed classification	Detailed classification
Rarity	Significant biota (e.g., rare or threatened)	Rare or threatened species
	Significant vegetation	Significant interaction
	Significant wetland or estuary, Significant riparian area,wetland, or other resource	
	Sites of significance (high endemism or species richness)	Site with high endemism, species richness, or taxonomic distinctiveness
Representativeness	Ecologically "healthy" river representative of different kinds of rivers historically found in the state	Area in good ecological condition, representative of different types of land cover historically found in the ecosystem
Naturalness	Macroinvertebrate communities	Terrestrial vertebrates
	Natural riparian vegetation (width, structural intactness, longitudinal continuity)	Terrestrial invertebrates Amphibians Aquatic taxa
	Natural fish populations (expected community, few invasives)	Community characteristicof limited human land use, few non-natives
	Fish migration passage	Migration corridor
	Naturally operating ecological processes (e.g., primary productivity)	Naturally operatingdisturbance processes (e.g., fire, flood)
Large-scale significance	Heritage river (high ecological, cultural, or recreational value)	High aesthetic, ecological,cultural, or recreational value
	Associated with Ramsar wetlands	Associated with international treaties

on many aspects of biodiversity and conservation. Nevertheless, the process of deliberately auditing assets and threats has provided the country with its first snapshot of the state of arid and semi-arid regions, which can serve as a benchmark for assessing future environmental change.

Conclusions

The proliferation of national and international efforts to assess and conserve faunal biodiversity has increased awareness of the rich and distinctive ecological heritage of arid zones. At the same time, the number and magnitude of irreversible threats to biodiversity are rising as technological advances facilitate expansion of human populations into areas that historically were uninhabitable or not permanently habitable. Human population growth and climate change further exacerbate competition for shrinking supplies of water and primary productivity in arid and semi-arid landscapes.

Table 2. Major phases in managing and improving river health in Victoria, Australia (adapted from DNRE, 2002:49)

1	Identify assets
2	Identify assets of high existing value
3	Identify threats to each asset and quantify the risk that each threat will occur
4	Identify opportunities and requirements for restoration
5	Set priorities for protection using a risk-based approach that quantifies the value of the asset, severity of the risk, program cost, ,likely improvement in the state of the asset given the cost, and necessary level of community participation
6	Identify broad actions necessary to alleviate threats such as vegetation clearance, salinization, and urbanization
7	Develop detailed action plans for specific assets and threats
8	Provide medium-term (five years) and longer-term (10 years) targets for expected change in asset condition
9	Develop guidelines for adaptive managment of the asset, including monitoring, reporting, and review and identification of triggers for changes in management if the state of the asset is changing in an undesirable fashion
10	Develop a stakeholder outreach and awareness program

In some was severely degraded arid regions, even if cost were no consideration, there is little potential for ecological restoration and rehabilitation. In other regions, restoration potential is limited because costs generally are prohibitive. However, we believe that the proportion of arid lands that must be discounted is relatively low. Although it may not be possible to restore systems to a historical or undisturbed state, we may be able to enhance resources and minimize threats to native biodiversity. Optimization models, for instance, can be used to develop efficient spatial and temporal strategies for management that are likely to yield the most effective outcomes for faunal conservation under realistic sets of ecological and socio-economic constraints. Pragmatic acknowledgment that healthy ecosystems support human needs and desires as well as native species – and that loss of ecosystem function jeopardizes both humans and native species – is likely to increase stakeholder support for consensus- based conservation and land-use planning. As the Victorian framework for management of rivers demonstrates, creative and thoughtful approaches to adaptive management represent a real hope for maintenance and restoration of arid ecosystems.

Acknowledgments

Thanks to G. Ceballos, G. Luck, K. Seto, and A. Smyth for helpful discussions and data sharing. E.F. acknowledges support of the Nevada Biodiversity Research and Conservation Initiative during preparation of this manuscript. J.T. is supported by the Australian Research Council. This is publication No. 67 from the Australian Centre for Biodiversity: Analysis, Policy and Management at Monash University.

References

Banks, P.B. 1999. Predation by introduced foxes on native bush rats in Australia: do foxes take the doomed surplus? *Journal of Applied Ecology* 36: 1063-1071.

Banks, P., Dickman, C. and Newsome, A. 1998. Ecological costs of feral predator control: foxes and rabbits. *Journal of Wildlife Management* 62: 766–772.

Barrett, G., Silcocks, A., Barry, S., Cunningham, R. and Poulter, R. 2003. The new atlas of Australian birds. Birds Australia, Melbourne.

Bennett, A.F. 1999. Linkages in the landscape. The roles of corridors and connectivity in wildlife conservation. IUCN, Gland, Switzerland.

Berger, J. 2004. The last mile: How to sustain long-distance migration in mammals. *Conservation Biology* 18: 320-331.

Blair, R.B. 1999. Birds and butterflies along an urban gradient: Surrogate taxa for assessing biodiversity. *Ecological Applications* 9:164-170.

Boggs, C.L. and Murphy, D.D. 1997. Community composition in mountain ecosystems: Climatic determinants of montane butterfly distributions. *Global Ecology and Biogeography Letters* 6: 39-48.

Bolger, D.T., Alberts, A.C. and Soulé, M.E. 1991. Occurrence patterns of bird species in habitat fragments: sampling, extinction, and nested species subsets. *The American Naturalist* 137:155-166.

Brazel, A., Selover, N., Vose, R. and Heisler, G. 2000. The tale of two cities—Baltimore and Phoenix LTER sites. *Climate Research* 15:123–135.

Brown, J.H. 1971. Mammals on mountaintops: Non-equilibrium insular biogeography. *The American Naturalist* 105: 467-478.

Brown, J.H. 1978. The theory of insular biogeography and the distribution of boreal mammals and birds. *Great Basin Naturalist Memoirs* 2: 209-228.

Carignan, R. and Steedman, R. 2000. Impacts of major watershed perturbations on aquatic ecosystems. *Canadian Journal of Fisheries and Aquatic Sciences* 57: 1-4.

Chambers, J.C., Miller, J.R., Germanoski, D. and Weixelman, D.A. 2004. Process-based approaches for managing and restoring riparian ecosystems. Great Basin *Riparian Ecosystems: Ecology, Management, and Restoration* (Eds. J.C. Chambers and J.R. Miller), pp. 261-192. Island Press, Washington, D.C., USA.

Cleverly, J.R., Smith, S.D., Sala, A. and Devitt, D.A.. 1997. Invasive capacity of *Tamarix ramosissima* in a Mojave Desert floodplain: the role of drought. *Oecologia* 111: 12-18.

Cogger, H.G. 1992. *Reptiles and Amphibians of Australia*. Cornell University Press, Ithaca, New York, USA.

Commonwealth of Australia, 2001. Rangelands – tracking changes. Australian collaborative rangeland information system. National Land and Water Audit, Land and Water Australia, Canberra.

Cremer, K. 1999. Willow management for Australian rivers. *Natural Resources Management Special Issue*: 1-22.

Czech, B., Krausman, P.R. and Devers, P.K. 2000. Economic associations among causes of species endangerment in the United States. *BioScience* 50: 593–601.

DeKay, M.L. and McClelland, G.H. 1996. Probability and utility components of endangered species: preservation programs. *Journal of Experimental Psychology: Applied* 2: 60-85.

Department of Natural Resources and Environment (DNRE). 2002. Healthy rivers, healthy communities and regional growth. The Victorian River Health Strategy. Department of Natural Resources and Environment, Melbourne, Australia.

Duggin, J. and Gentle, C. 1998. Experimental evidence on the importance of disturbance intensity for invasion of *Lantana camara* L. in dry rain forest-open forest ecotones in north-eastern NSW, Australia. *Forest Ecology and Management* 109: 279-292.

Dunham, J.B., Adams, S.B., Schroeter, R.E. and Novinger, D.C. 2002. Alien invasions in aquatic ecosystems: Toward an understanding of brook trout invasions and potential impacts on inland cutthroat trout in western North America. *Reviews in Fish Biology and Fisheries* 12: 373-391.

Earl, S.R. and Blinn, D.W. 2003. Effects of wildfire ash on water chemistry and biota in south-western USA streams. *Freshwater Biology* 48: 1015-1030.

Eby, L.A., Fagan, W.F. and Minckley, W.L. 2003. Variability and dynamics of a desert stream community. *Ecological Applications* 13: 1566-1579.

Erhardt, A. 1985. Diurnal lepidoptera: sensitive indicators of cultivated and abandoned grassland. *Journal of Applied Ecology* 22: 849-861.

Feber, R.E., Smith, H. and MacDonald, D.W. 1996. The effects on butterfly abundance of the management of uncropped edges of arable fields. *Journal of Applied Ecology* 33: 1191-1205.

Fensham, R.J. and Fairfax, R.J. 1997. The use of the land survey record to reconstruct pre-European vegetation patterns in the Darling Downs, Queensland, Australia. *Journal of Biogeography* 24: 827-836.

Fleishman, E., Austin, G.T., Brussard, P.F. and Murphy, D.D. 1999. A comparison of butterfly communities in native and agricultural riparian habitats in the Great Basin. *Biological Conservation* 89: 209-218.

Fleishman, E., Austin, G.T. and Weiss, A.D. 1998. An empirical test of Rapoport's rule: Elevational gradients in montane butterfly communities. *Ecology* 79: 2482-2493.

Fleishman, E., Austin, G.T. and Murphy, D.D. 2001. Biogeography of Great Basin butterflies: Revisiting patterns, paradigms, and climate change scenarios. *Biological Journal of the Linnean Society* 74: 501-515.

Fleishman, E., Dunham, J.B., Brussard, P.F. and Murphy, D.D. 2004. Explanation, prediction, and maintenance of native species richness and composition in the central Great Basin. *Great Basin Riparian Ecosystems – Ecology, Management, and Restoration* (Eds. J.C. Chambers and J.R. Miller), pp. 232-260. Island Press, Washington, D.C., USA.

Fleishman, E., McDonal, N., Mac Nally, R., Murphy, D.D., Walters, J. and Floyd, T. 2003. Effects of floristics, physiognomy, and non-native vegetation on riparian bird communities in a Mojave Desert watershed. *Journal of Animal Ecology* 72: 484-490.

Grayson, D.K. 1993. The Desert's Past: A Natural Prehistory Of The Great Basin. Smithsonian Institution Press, Washington, D.C., USA.

Grayson, D.K. 2000. Mammalian responses to Middle Holocene climate change in the Great Basin of the western United States. *Journal of Biogeography* 27: 181-192.

Grayson, D.K. and Madson, D.B. 2000. Biogeographic implications of recent low-elevation recolonization by *Neotoma cinerea* in the Great Basin. *Journal of Mammalogy* 81: 1100-1105.

Greiner, R. 1998. Catchment management for dryland salinity control -- model analysis for the Liverpool Plains in New South Wales. *Agricultural Systems* 56: 225-251.

Grey, M.J., Clarke, M.F. and Loyn, R.H. 1998. Influence of the Noisy Miner *Manorina melanocephala* on avian diversity and abundance in remnant Grey Box woodland. *Pacific Conservation Biology* 4: 55-69.

Guisan, A., Holten, J.I., Spichiger, R. and Tessier, L. 1995. Potential ecological impacts of climate change in the Alps and Fennoscandian mountains. Conservatory and Botanical Garden of Geneva, Switzerland.

Hershler, R., Madsen, D.B. and Currey, D.R. (Eds.) 2002. Great Basin Aquatic Systems History. Smithsonian Contributions to the Earth Sciences No. 33.

Hidy, G.M. and Klieforth, H.E. 1990. Atmospheric processes and the climates of the Basin and Range. *Plant biology of the Basin and Range* (Eds. C.B. Osmond, L.F. Pitelka and G.M. Hidy), pp. 17-42. Springer-Verlag, Berlin, Germany.

Hobbs, R.J. and Kristjanson, L.J. 2003. Triage: how do we prioritize health care for landscapes? Ecological *Management and Restoration* 4: S39-S45.

Houghton, J.G., Sakamoto, C.M. and Gifford, R.O. 1975. Nevada's weather and climate. Nevada Bureau of Mines and Geology, Special Publication 2. Reno, Nevada, USA.

Hubbs, C.L. and Miller, R.R. 1948. The zoological evidence: correlation between fish distribution and hydrographic history in the desert basins of western United States. *Bulletin of the University of Utah* 38: 17-166.

Huxman, T.E., Smith, M.D., Fay, P.A., Knapp, A.K., Shaw, M.R., Loik, M.E., Smith, S.D., Tissue, D.T., Zak, J.C., Weltzin, J.F., Pockman, W.T., Sala, O.E., Haddad, B.M., Harte, J., Koch, G.W., Schwinning, S., Small, E.E. and Williams, D.G. 2004. Convergence across biomes to a common rain-use efficiency. *Nature* 429: 651-654.

Intergovernmental Panel on Climate Change (IPCC). 2001. *Climate change 2001: Impacts, Adaptation, and Vulnerability. Geneva, Switzerland.*

Jenkins, K.M. and Boulton, A.J. 2003. Connectivity in a dryland river: Short-term aquatic microinvertebrate recruitment following floodplain inundation. *Ecology* 84: 2708-2723.

Kinzig, A.P., Pacala, S.W. and Tilman, D. (Eds.) 2001. *The Functional Consequences of Biodiversity*. Princeton University Press, New Jersey, USA.

Lampo, M. and De Leo, G.A. 1998. The invasion ecology of the toad *Bufo marinus*: From South America to Australia. *Ecological Applications* 8: 388-396.

Lawlor, T.E. 1998. Biogeography of Great Basin mammals: paradigm lost? *Journal of Mammalogy* 79: 1111-1130.

Lockwood, J.L. and McKinney, M.L. (Eds.) 2001. *Biotic Homogenization: The Loss of Diversity through Invasion and Extinction*. Kluwer Academic / Plenum Publishers, New York, USA.

Lytle, D.A. and Poff, N.L. 2004. Adaptation to natural flow regimes. *Trends in Ecology and Evolution* 19: 94-100.

MacNally, R. 1999. Habitat fragmentation and habitat loss: Secondary, cascading effects and predictability. *Australian Biologist* 12: 138-151.

Maestre, F., Cortina, J., Bautista, S., Bellot, J. and Vallejo, R. 2003. Small-scale environmental heterogeneity and spatiotemporal dynamics of seedling establishment in a semiarid degraded ecosystem. *Ecosystems* 6: 630-643.

Major, R.E., Christie, F.J. and Gowing, G. 2001. Influence of remnant and landscape attributes on Australian woodland bird communities. *Biological Conservation* 102: 47-66.

MacArthur, R.H. and Wilson, E.O. 1967. *The Theory of Island Biogeography*. Princeton University Press, Princeton, New Jersey, USA.

Martin, D. and Chambers, J. 2002. Restoration of riparian meadows degraded by livestock grazing: above- and belowground responses. Plant Ecology 163: 77-91.

McDonald, K.A. and J.H. Brown. 1992. Using montane mammals to model extinctions due to climate change. *Conservation Biology* 6: 409-415.

McGee, T.G. 1991. Southeast Asian urbanization: Three decades of change. *Indonesian Journal of Social and Economic Affairs* 51: 3-16.

McKinney, M.L. 2002. Urbanization, biodiversity, and conservation. *BioScience* 52: 883-890.

McLaughlin, J.F., Hellmann, J.J., Boggs, C.L. and Ehrlich, P.R. 2002 Climate change hastens population extinctions. *Proceedings of the National Academy of Sciences (USA)* 99: 6070-6074.

Miller, J.N., Brooks, R.P. and Croonquist, M.J. 1997. Effects of landscape pattern on biotic communities. *Landscape Ecology* 12: 137-153.

Mooney, H.A. and Hobbs, R.J. 2000. *Invasive Species in a Changing World*. Island Press, Washington, D.C., USA.

Morrison, M.L., Mills, L.S. and Kuenzi, A.J. 1996. Study and management of an isolated, rare population: the Fresno kangaroo rat. *Wildlife Society Bulletin* 24: 602-606.

Murphy, D.D. and Weiss, S.B. 1992. Effects of climate change on biological diversity in western North America: Species losses and mechanisms. In *Global Warming and Biological Diversity* (Eds. R.L. Peters and T.E. Lovejoy), pp. 355-368 Yale University Press, New Haven, Connecticut, USA.

Myers, N. 1986. Tropical deforestation and a mega-extinction spasm. In *Conservation Biology: The Science of Scarcity and Diversity* (Ed. M.E. Soulé), pp. 394-409. Sinauer, Sunderland, Massachusetts, USA.

National Land and Water Resources Audit (NLWRA). 2002. Australia's natural resources: 1997-2002 and beyond. Commonwealth of Australia, Canberra.

Nelson, S.M. and Andersen, D.C. 1994. An assessment of riparian environmental quality by using butterflies and disturbance susceptibility scores. *Southwestern Naturalist* 39: 137-142.

New, T.R. 1991. Butterfly Conservation. Oxford University Press, Melbourne, Australia.

Nowak, D.J., Rowntree, R. McPherson, E.G. Sisinni, S.M. Kerkmann, E. and Stevens, J.C. 1996. Measuring and analyzing urban tree cover. Landscape and Urban Planning 36:49–57.

Olden, J.D., Poff, N.L., Douglas, M.R., Douglas, M.E. and Fausch, K.D. 2004. Ecological and evolutionary consequences of biotic homogenization. *Trends in Ecology and Evolution* 19: 18-24.

Olson, D.M. and Dinerstein, E. 1998. The Global 200: A representation approach to conserving the earth's most biologically valuable ecoregions. *Conservation Biology* 12: 502-515.

Palumbi, S.R. 2001. Humans as the world's greatest evolutionary force. *Science* 293: 1786-1790.

Parmesan, C., Ryrholm, N., Stefanescu, Ć., Hill, J.K., Thomas, C.D., Descimon, H., Huntley, B., Kaila, L., Kulberg, J., Tammaru, T., Tennett, W.J., Thomas, J.A. and Warren, M. 1999. Poleward shifts in geographical ranges of butterfly species associated with regional warming. *Nature* 399: 579-583.

Pavelic, P., Narayan, K.A. and Dillon, P.J. 1997. Groundwater flow modelling to assist dryland salinity management of a coastal plain of Southern Australia. *Australian Journal of Soil Research* 35: 669-686.

Pickett, S.T.A., Cadenasso, M.L. Grove, J.M., Nilon, C.H., Pouyat, R.V., Zipperer, W.C. and Costanza, R. 2001. Urban ecological systems: Linking terrestrial ecological, physical, and socio-economic components of metropolitan areas. *Annual Review of Ecology and Systematics* 32: 127-157.

Radford, I., Nicholas, D. and Brown, J. 2001. Impact of prescribed burning on *Acacia nilotica* seed banks and seedlings in the *Astrebla* grasslands of northern Australia. *Journal of Arid Environments* 49: 795-807.

Rejmanek, M. 2000. Invasive plants: approaches and predictions. *Austral Ecology* 25: 497-506.

Restani, M. and Marzluff, J.M. 2001. Avian conservation under the Endangered Species Act: Expenditures versus recovery priorities. *Conservation Biology* 15: 1292-1299.

Ricketts, T.H., Dinerstein, // Olson, D.M. Loucks, C.J. Eichbaum, W., Della Salla, D., Kavanagh, K., Hedao, P. Hurley, P.T. Carney, K.M., Abell, R. and Walters, S. 1999. *Terrestrial Ecoregions of North America: A Conservation Assessment.* Island Press, Washington, D.C., USA.

Rind, D., Goldberg, R., Hansen, J., Rosenzweig, C. and Ruedy, R. 1990. Potential evapotranspiration and the likelihood of future drought. *Journal of Geophysical Research* 95: 9983-10004.

Rood, S.B., Gourley, C.R., Ammon, E.M., Heki, L.G., Klotz, J.R., Morrison, M.L., Mosley, D., Scoppettone, G.G., Swanson, S. and Wagner, P.L. 2003. Flows for floodplain forests: a successful riparian restoration. *BioScience* 53: 647-656.

Saether, B., Tufto, J., Engen, S., Herstad, K., Røstad, O.W. and Skåtan, J.E. 2000. Population dynamical consequences of climate change for a small temperate songbird. *Science* 287: 854–856.

Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M. and Wall, D.H. 2000. Global biodiversity scenarios for the year 2100. *Science* 287: 1770-1774.

Sauer, J.R., Hines, J.E. and Fallon, J. 2004. The North American Breeding Bird Survey, results and analysis 1966–2003. Version 2004.1. USGS Patuxent Wildlife Research Center, Laurel, Maryland. http://www.pwrc.usgs.gov/bbs/, accessed June 2004.

Sher, A.A., Marshall, D.L. and Gilbert, S.A. 2000. Competition between native *Populus deltoides* and invasive *Tamarix ramosissima* and the implications for reestablishing flooding disturbance. *Conservation Biology* 14: 1744-1754.

Simberloff, D. 1981. Community effects of introduced species. In *Biotic Crises in Ecological and Evolutionary Time* (Ed. M.H. Nitecki), pp. 53-81. Academic Press, Chicago, Illinois, USA.

Smith, S.D., Huxman, T.E., Zitzer, S.F., Charlet, T.N., Housman, D.C., Coleman, J.S., Fenstermaker, L.K., Seemann, J.R. and Nowak, R.S. 2000. Elevated CO_2 increases productivity and invasive species success in an arid ecosystem. *Nature* 408: 79-82.

Smith, T. and Starr, B. 1999. An historical perspective. Monash University, Adelaide, Australia: Cooperative Research Centre for Catchment Hydrology: 573-577.

Soulé, M.E., Estes, J.A., Berger, J. and Martinez del Rio, C. 2003. Ecological effectiveness: conservation goals for interactive species. *Conservation Biology* 17: 1238-1250.