ELSEVIER

Landscape and Urban Planning 58 (2002) 41–56

LANDSCAPE
AND
URBAN PLANNING

www.elsevier.com/locate/landurbplan

A conceptual model for conservation planning based on landscape species requirements

Eric W. Sanderson*, Kent H. Redford, Amy Vedder, Peter B. Coppolillo, Sarah E. Ward

Wildlife Conservation Society, International Conservation Programs, 2300 Southern Blvd., Bronx, NY 10460, USA

Received 2 March 2001; received in revised form 1 October 2001; accepted 5 October 2001

Abstract

Effective conservation planning requires, considering all the complicated biological, social and economic factors which impinge on the ecological integrity of a site, and then focusing inevitably limited conservation resources on those times, places and activities that most impact ecological structure and function. The landscape species concept provides a useful lens for defining conservation landscapes and highlighting potential threats from human activity. This paper outlines a conceptual methodology for landscape conservation being tested by the Wildlife Conservation Society at three sites in Latin America and Africa. Based on the biological requirements of an ecologically functioning population of a landscape species, the "biological" landscape is defined. This landscape is compared to the landscape of human activities through the use of Geographic Information Systems (GIS). Focal landscapes sufficient to meet species requirements are defined and threats from human activity evaluated with respect to biological requirements. A suite of landscape species may be selected depending on resources, leading to multiple, often overlapping, focal landscapes. A hypothetical example is presented.

Keywords: Landscape scale conservation; Conservation planning; Landscape species

1. Introduction

We know from practice that the most effective strategies to conserve biodiversity must account for the complex and diverse needs of wildlife and people. Parks and reserves can effectively protect some elements of biodiversity (sensu Redford and Richter, 1999) and contribute to the conservation of nature, but often strict protection is not possible over sufficient areas. As a result virtually all protected areas are embedded in a landscape in which natural resource exploitation of multiple types occurs (Redford and Robinson, 1995; Brandon et al., 1998; Woodroffe and Ginsberg, 1998). Effective biodiversity conservation must therefore integrate use and protection across the landscape (Arnold, 1995; Fox et al., 1996; Furze et al., 1996; Steiner et al., 2000b). For landscape scale conservation to be socially as well as ecologically sustainable, those strategies must succeed in a mosaic of different land uses that not only conserves biodiversity, but also allows people to make a living (Casimir and Rao, 1998; Milner-Gulland and Mace, 1998; Hoare, 1999; Zander and Kachele, 1999; Thompson and Sorvig, 2000). Such a conservation strategy must integrate land uses as diverse as parks, large forestry concessions, indigenous management areas, extractive reserves, agricultural zones, and

* Corresponding author. Tel.: +1-718-220-6825;
fax: +1-718-364-4275.
E-mail address: esanderson@wcs.org (E.W. Sanderson).

0169-2046/02/$20.00
PII: S0169-2046(01)00231-6

urban areas (McShane, 1990; Forman and Collinge, 1996, 1997; Poiani et al., 1998; Hostetler, 1999; Pino et al., 2000).

However effective conservation planning is not a matter of simply thinking, over larger areas and across different management units. Effective conservation planning must also include consideration of the heterogeneous and dynamic nature of ecosystems (Huston, 1994; Pickett et al., 1997; White and Harrod, 1997; Koehler, 2000). Nature, as landscape ecologists like to say, is not homogenous. Spatial heterogeneity is created by environmental gradients (Austin and Heyligers, 1989; Burnett et al., 1998), disturbance processes (Pickett and White, 1985; Baker, 1992; Turner et al., 1994; Lertzman and Fall, 1998) and organisms themselves (Cumming et al., 1997; Kinnaird, 1998), particularly people (Baker, 1995; Lawrence et al., 1998; Thrash, 1998). Some processes are slow and create gradual changes, for example successional processes (Parker and Pickett, 1998). Other processes create fast or punctuated changes, but those changes may be distributed irregularly across large areas, for example disturbance processes like wind throws and fire (White, 1979; Holling, 1992).

Because nature is not homogenous, most organisms, at one scale or another, depend on heterogeneity for their survival (Pulliam and Danielson, 1991; Hansson et al., 1995; Tilman and Kareiva, 1997). Accordingly neither people, nor animals, use the landscape homogeneously, but rather cluster their activities differentially among landscape elements (Holling, 1992). People distribute their uses in space and time, establish areas of non-use (i.e. protected areas), and live in concentrations that create an uneven mosaic of use (Verlinden, 1997; Schrijnen, 2000; Thomlinson and Rivera, 2000). Similarly some organisms use multiple habitats, depend on temporary concentrations of resources, and move around the landscape in non-random ways and according to spatially and temporally distributed requirements (Kozakiewicz, 1995; Kinnaird et al., 1996; Lima and Zollner, 1996). Protecting organisms means protecting their ability to meet their requirements, inside and outside parks, among different human uses, and in spite of human economic and cultural requirements being met in the same times and places (Harrison and Fahrig, 1995; Bowen and Crouse, 1997; Lambeck, 1997; Naughton-Treves, 1998; Hoare and Du Toit, 1999).

In short, conservation planning requires a landscape sensibility (Noss, 1983). Landscapes, as defined by ecologists, are spatially heterogeneous areas, typically with extents at least a few kilometers across (Turner, 1989; Forman, 1995). There are a variety of different kinds of heterogeneity that can be recognized in the landscape, depending on the goal and scale of the planning effort. Some conservation plans focus on geographic or ecological distinctions, like climate, topography or vegetation types (Scott et al., 1993; Faith and Walker, 1996; Bonan, 2000; Fairbanks and Benn, 2000) which have a scale and complexity quite different from plans focused on accommodating different land uses or the economic ambits of different communities (Steiner et al., 2000a; Treu et al., 2000). The key for conservationists is defining the kinds of heterogeneity that most directly influence the parts of nature under threat.

Effective conservation planning must clearly define biologically relevant landscape elements for planning at the appropriate scales (Poiani et al., 1998; Whited et al., 2000; Wu and Smeins, 2000). What elements are relevant and at what scales is an active area of research in both conservation biology and landscape ecology at this time, leading to an array of different approaches (Franklin, 1993; Simberloff, 1998; Caro and O'Doherty, 1999; Miller et al., 1999; Bellman, 2000; Noss, 2000; Poiani et al., 2000; Steiner et al., 2000a). Some of these approaches emphasize patterns of biodiversity over the landscape with the goal of conserving the most species rich places (e.g. Myers et al., 2000). Others emphasize conservation of ecosystem services and economic viability of local communities as a means toward ecosystem conservation (e.g. Barrett and Arcese, 1995; Alpert, 1996). Still others plan portfolios of conservation sites that insure representation of species, communities and ecological phenomena (e.g. Anderson et al., 1999; Ricketts et al., 1999; Groves et al., 2000). These different conservation approaches are applied at a variety of different scales, from global visions to plans for small, isolated protected areas.

Recently the Wildlife Conservation Society developed an approach to site-based conservation planning based on the species requirements of suites of ''landscape species'' (Redford et al., 2000 and summarized below). This approach chooses certain species whose life history characteristics and biological requirements in space and time make them particularly useful for identifying when and where human uses of the landscape may compromise ecological integrity of the overall landscape (Vedder, 2000). Thus, this approach builds on other species based approaches to landscape conservation such as the focal species approach (Lambeck, 1997), but it differs in not describing not just how to select species to focus conservation effort, but how actually to focus that effort through a conceptual, spatially explicitly methodology for systematically thinking about conservation landscapes in terms of biological requirements and human use. This conceptual model is currently being field tested across three sites in tropical Latin America and Africa and will be refined through experience over the next 5 years.

2. The landscape species concept

Landscape species are defined as biological species that ‘‘use large, ecologically diverse areas and often have significant impacts on the structure and function of natural ecosystems’’ (Redford et al., 2000). Their requirements in time and space make landscape species particularly susceptible to human alteration and use of natural landscapes. The landscape species concept is a way of selecting a few species for conservation attention with the belief that meeting their needs will achieve conservation of other species and of the landscape as whole. As such the landscape species concept is related to other species focused conservation planning techniques based on focal species, umbrella species, keystone species or flagship species (Lambeck, 1997; Caro and O'Doherty, 1999; Miller et al., 1999). The landscape species concept is different however in that the species' requirements are used to define the landscape in which conservation must occur, both the outer limits of such an area (the extent) and the important variation within it (the grain). The landscape species is the lens through which we view the landscape to determine where human activities impinge on the ecological integrity of an area.

The definition of landscape species contains several provisions. The first provision is that populations of landscape species use large areas; that is, the scale of the population's use of the environment is significant in making the landscape species a conservation target. Large of course is relative, but in this sense ‘‘large’’ refers to organisms whose populations use landscapes over distances and through times comparable to human management regimes and resource extraction activities—hundreds to thousands of square kilometers and with population dynamics measured in years and decades. The population's use of the landscape, measured over a time significant for conservation purposes, defines the area in which we will work to ensure its conservation.

The area necessary to ensure the conservation of landscape species population—the conservation landscape—is by the definition, ecologically diverse. The different elements of the landscape spatial mosaic can be defined by ecosystem types, human land uses, or any other combination of species-meaningful land type classifications (e.g. forested areas near roads where there is heavy hunting). A landscape by itself is a scale-less entity; it requires some other entity or process to define its functional scale (Allen, 1998). In this definition, the extent of the landscape is defined by sum of all the areas required to support a population of the landscape species; the grain is defined by the mosaic of areas necessary for that species, using grain and extent in the sense of (Weins, 1989). The different land types that the species uses determine the composition of the ‘‘ecologically diverse’’ landscape.

Because landscape species use relatively large areas that consist of different habitat types and land uses, they encompass the space and resource requirements of many other species, i.e. they serve as umbrella species (Wilcox, 1984). Most significantly, landscape species also require heterogeneous areas that are maintained by ‘‘landscape scale’’ processes such as disturbance (Pickett and Rogers, 1997; White and Harrod, 1997); in other words, landscape species rely on the composition and configuration of landscapes rather than simply large areas. Because of their role in defining landscapes, landscape species are more than just umbrella species, because conservation of the landscape species will lead not only to the conservation of other sympatric species, but presumably also to the conservation of the structure of the landscape, including the ecological functions dependent on that structure. Therefore, many species that do not directly interact with the landscape species may still be conserved through maintenance of functional landscapes.

Landscape species also often have *direct* impacts on the structure or function of natural landscapes (deMaynadier and Hunter, 1994; Johnston, 1995). Species like forest elephants structure ecosystems in a variety of different ways, by localizing and redistributing energy and materials (e.g. aggregations of grazers; seed dispersers), by physically altering the landscape as a form of disturbance agent or structuring it (e.g. elephants, beavers or corals), or by modifying the way other organisms use the landscape (e.g. predators). In these senses, landscape species are often ecologically pivotal or keystone species (Mills et al., 1993; Power et al., 1996). Although landscape species may often play important functional roles in landscapes, the lack of those functions does not preclude them from being classified as landscape species because of their benefits in defining landscapes, as described above.

Landscape species are typically susceptible to human activities in the landscape because of their requirements for large areas and their sensitivity to landscape structure and function. Because of the scale of their requirements, they often come into conflict with human beings who are exploiting resources over similar scales in the same landscape. This susceptibility may not translate into a propensity toward global extinction (dispersal limited species with narrow requirements may be more extinction prone, for example), rather landscape species may indicate their vulnerability by altering their use of time and space (Stern, 1998; Novaro et al., 2000b) or through diminution of the nature and strength of their ecological interactions (Weaver et al., 1996; Novaro et al., 2000a). Their long-term survival depends on their ability to withstand large scale fluctuations in critical resource availability year to year and over spatial scales defined by their mobility, such that when their viability in a landscape becomes threatened, it seems likely that the integrity of the landscape is under threat.

The selection of landscape species approach may bear superficial similarity to the 'focal species' approach (Lambeck, 1997), but there are two important conceptual differences. First, under the focal species approach the suite of species is chosen by identifying the most demanding species with respect to: "area", "resource", "dispersal", and "process" requirements (Lambeck, 1997). In contrast, landscape species are chosen based on an aggregated score incorporating area, heterogeneity in the habitats and land uses they encounter, vulnerability to anthropogenic threats, ecological functionality, and socio-economic significance (Wallace et al., 2001). Candidate landscape species are scored according to these criteria and the suite is built by choosing the one species exemplifying the five criteria and then adding other highly ranked species with complementary habitat requirements. This process continues until the spatial requirements of the most complementary candidate species (i.e. the species with the least spatial overlap) have already been met by the current suite. The second difference is one of emphasis rather than methodology. The focal species approach focuses primarily on selecting the species upon which to base conservation planning. The landscape species approach focuses on how planning efforts should strive to meet and protect the needs of a suite of species. Therefore, the two approaches, Lambeck's focal species and landscape species, may prove to be entirely complementary.

The application of the landscape species approach to conservation is not limited to any specific type of landscape and may be applicable to seascapes (marine environments) as well. The landscapes defined by landscape species may or may not have protected areas; they may or may not be heavily impacted by human activities. In practice the selection of where to apply this conservation strategy will depend on conservation priorities typically developed at larger (national, regional or global) scales. In this way landscape species conservation can be nested within a comprehensive strategy for conservation, connecting global or regional scale conservation priority-setting activities to site-based conservation work.

It seems likely that in most cases, conservation of one landscape species will not be sufficient to gain all benefits in terms of conservation of other species and overall landscape function (Fleishman et al., 2001; Andelman and Fagan, 2000). Rather a suite of species will be required, each of which will define its own conservation landscape. These landscapes will likely overlap in space and time, providing additional information as to which landscape elements are most critical. Ideally the suite of landscape species will provide complementary insights into landscape structure and function, while providing a set of different indicators of success.

The remainder of this paper outlines a conceptual strategy for focusing conservation activities through the landscape species concept as currently being implemented by the Wildlife Conservation Society's Living

Landscapes Program in an adaptive management framework. The strategy consists of identifying the biological landscape of the landscape species, identifying the landscape of human activities which occurs in the same area, intersecting these to identify those places and times where human activity is most likely to threaten the landscape species population, and then redefining the focal landscape which can potentially meet the needs of the target population given necessary conservation interventions. These steps are outlined in Fig. 1 and illustrated in Figs. 2–7.

3. Method and discussion

3.1. Define the biological landscape of the landscape species population

Provided that a general site has been selected for conservation through some coarse-filter priority-setting mechanism, the first goal of the landscape species approach is to identify the landscape of biological requirements for a population of the focal species. In practice the target population may be identified as the

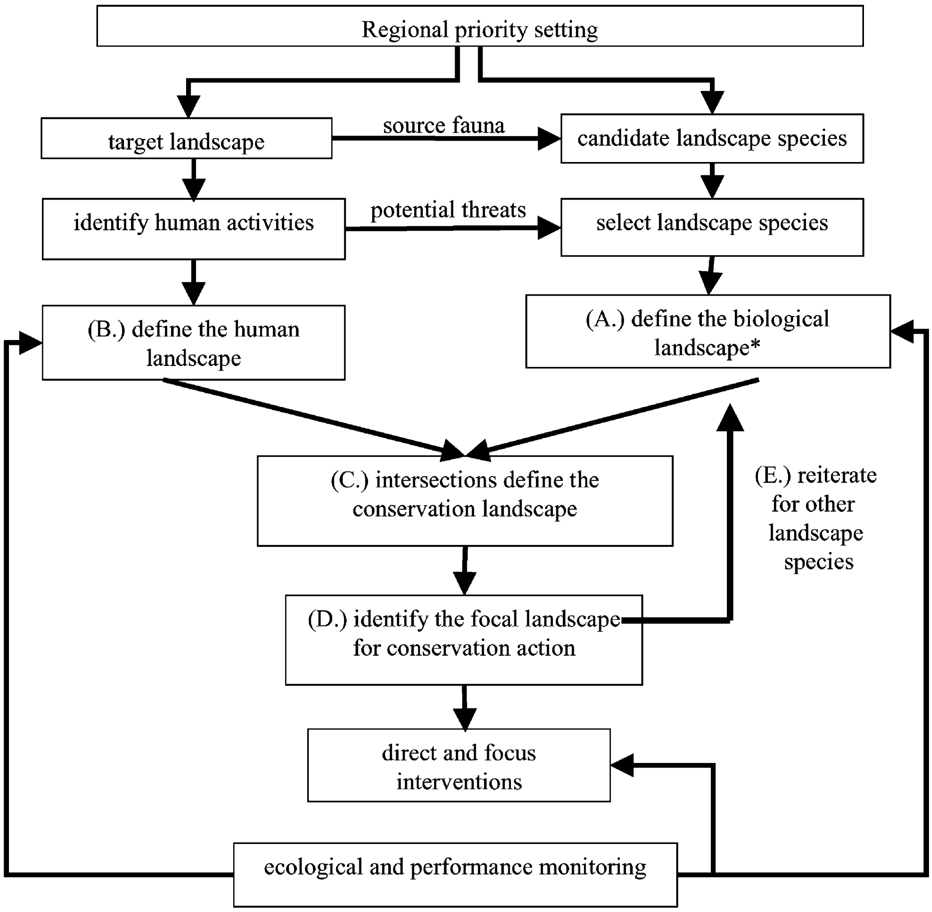

Fig. 1. Flow chart summarizing the steps in implementing a landscape species approach to conservation for one species. Based on some kind of global or regional priority-setting exercise, either a candidate set of species or a target area is designated. Through simultaneous processes human activity in the area is examined as potential landscape species are identified. Based on the requirements of the landscape species, a biological landscape is defined. The biological landscape is intersected in a spatially and temporally explicit manner to define the conservation landscape. A subset of those landscape elements is selected that meet the requirements of the landscape species in the most efficient fashion. Human activities within the focal landscape that conflict with species requirements are considered threats and are the focus of conservation interventions. Ecological and performance monitoring enable adaptive management of this process as new information becomes available. The steps labeled (A)–(E) are the subject of this paper and match the subheadings in the text. Figure adapted from Vedder, 2000 with permission.

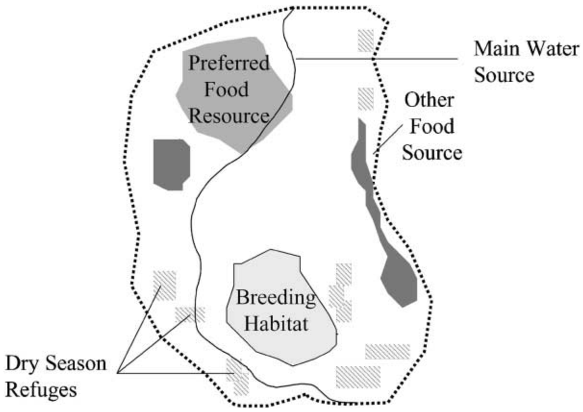

Fig. 2. The biological landscape of species X showing landscape elements where food resources, breeding habitat, and dry season refuges are available. The dotted line outside delimits the landscape conservation area (LCA) for conservation action regarding the target population of species X.

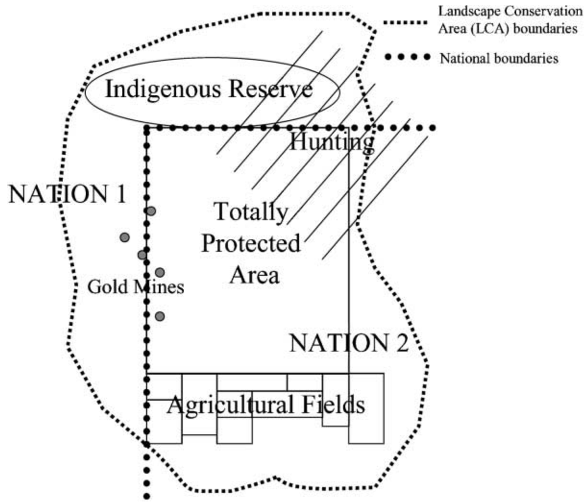

Fig. 3. The landscape of human activities showing that the corresponding area to Fig. 2. Human uses in the landscape include protection, agriculture, gold mining and hunting. Politically the area is divided between two nations. There is an indigenous reserve in nation 1 contiguous with the totally protected area in nation 2.

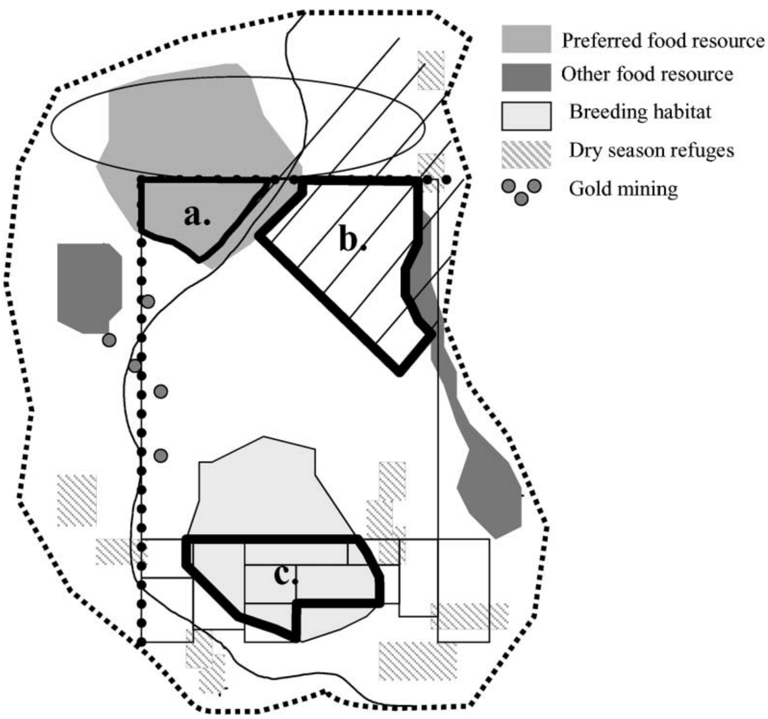

Fig. 4. The union of the biological landscape with the human landscape to generate landscape elements relevant to conservation of species X. Highlighted landscape elements include (a) a 65 km^2 area containing a preferred food resource within the totally protected area, (b) a 90 km^2 area that does not contribute direct to species requirements, but where there is illegal hunting within the protected area, and (c) a 80 km^2 area of breeding habitat outside the protected area and dominated by agriculture. All other parts of the landscape are similarly described in terms of their area, biological importance, human activities, and seasonal use by both people and the landscape species population.

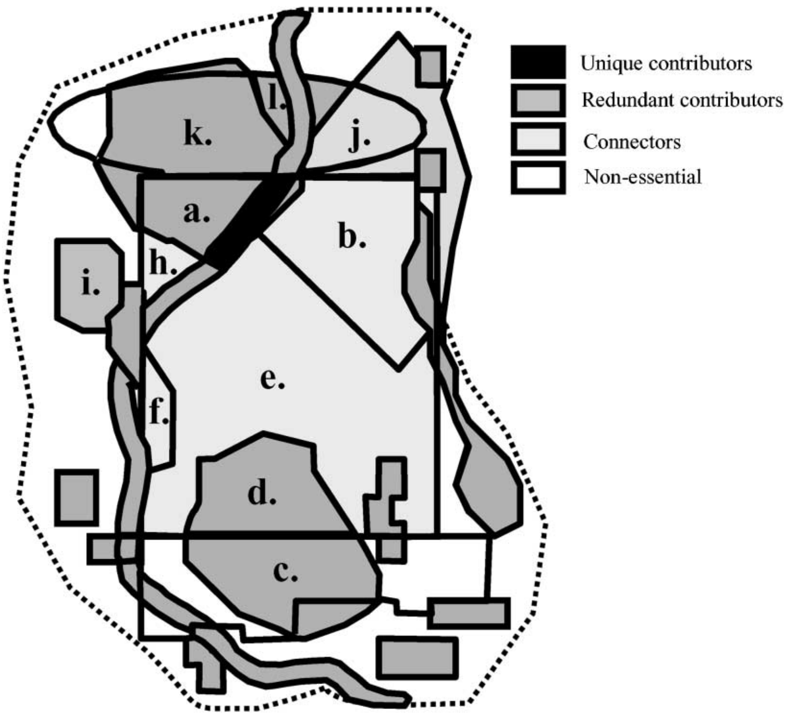

Fig. 5. Each new landscape element is evaluated with respect to the rarity and type of its contribution to the landscape of the species X. Unique contributors are elements that are required by the species and that occur in only one place in the landscape. In this case, landscape element (g) is defined as preferred food resource habitat with access to the river and within protection. Other landscape elements may be required for the species, yet are redundantly available in the landscape. Still other elements may not be required at all, yet are necessary for connectivity. Finally elements are not required nor provide connection are labeled as non-essential with respect to species X.

population occurring within a human administrative unit (like a protected area) or within a more naturally defined unit, like a watershed or mountain range. The specification of the target population should include an explicit estimate of the approximate number of individuals of the species that we wish to conserve. That number may be determined through some mechanism like a minimum viable population analysis, which defines the number of individuals required to maintain demographic stability and genetic diversity (Ruggiero et al., 1994). The population target may also be determined by identifying the number of organisms necessary for a population to maintain all of the ecological functions of that species in that landscape (Redford, 1992; Redford and Feinsinger, 2001). There is some evidence that species can be at minimally viable levels, yet in insufficient numbers to maintain ecological functions that species performed in the past; they are, in a phrase, ''ecologically extinct''. (Estes et al., 1989; Novaro et al., 2000a). Presumably ecologically functioning populations will require a higher population target than a minimally viable population, although how much higher is difficult to know. In practice determining these target population levels is very difficult, however it is only through population targets, even adaptively managed ones, that we can set clear goals for our conservation efforts.

Once the target population level has been identified, the next step is to quantify the landscape requirements for the species to achieve its population target. Quantifying the landscape requirements requires specifying how much area of each landscape element is required and when that element is required, and then mapping those landscape elements for a particular conservation site. Biological requirements for most species can be summarized as food, water, mates, and shelter in sufficient quantities and qualities to sustain the target

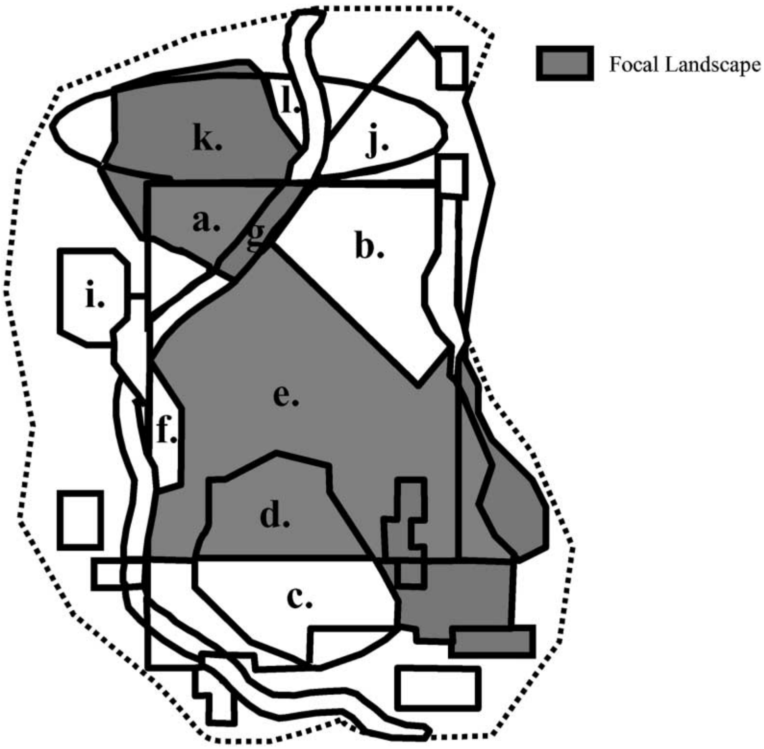

Fig. 6. The focal landscape is defined as the set of landscape elements that meets all the requirements of the landscape species and that insures connectivity for that population. Focal landscapes necessarily include all uniquely contributing elements, but may select among redundant elements, such that the total area of habitat for each requirement is met (Table 1). Multiple focal landscapes could be generated from most landscapes; the primary focal landscape should be the minimum set of landscape elements necessary to support the target population.

population. Often these biological requirements translate spatially into different kinds of habitats, or in a more general terminology, different landscape elements (e.g. food resources, breeding habitat, dry season refuges, etc.).

Defining the biological requirements in this way requires an extensive understanding of the species biology, and most likely, some educated guesswork. Information from wildlife-habitat studies at the site or elsewhere that inform habitat requirements, densities in different habitat types, and movement patterns between habitat types will be particularly useful. In cases where such information does not exist, it may be necessary to make assumptions or guesses based on related species or local information, which must then be monitored in the implementation phase, and adapted through a process of monitoring and further research. A hypothetical list of requirements for a landscape species, illustrating the kinds of information required, is given in Table 1.

The next step is to express the locations of these landscape elements in space, either using maps of the region, or preferably using spatial databases, for example, Geographic Information Systems (GIS), to generate a map of the ‘‘biological landscape’’ (Fig. 2). In this step the generic understanding of landscape requirements (e.g. 100 km^2 of preferred food habitat) is mapped to the specifics of a particular landscape (e.g. forest patches in the protected area). We need to know the overall distribution of each landscape element type and to quantify the amount of each element that is available. In some cases, where the required set of landscape elements changes dramatically with season, different maps may be required for each season. Often this work will require working from vegetation maps, aerial photos, remote sensing imagery or other

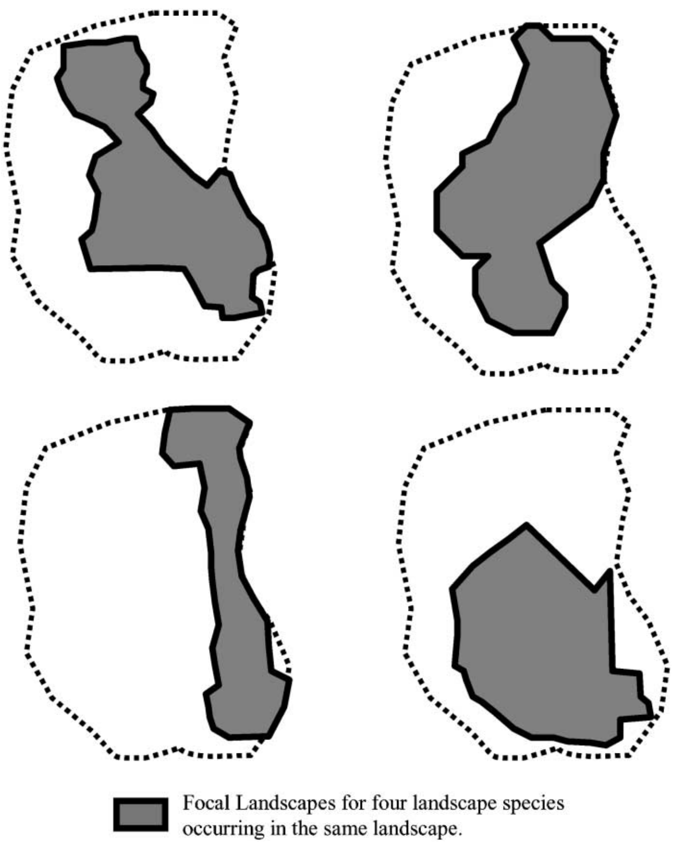

Fig. 7. Landscape species conservation will typically be based on a small suite of species, each of which will have its own focal landscape. The planning methodology outlined in this paper will be reiterated for each species, based on species-specific biological landscapes, but the same human landscape. Different landscape elements with different kinds of human activity may pose quite different threats, depending on the species characteristics. Landscape elements shared among multiple landscape species and under threat will receive higher priority attention.

spatial data sources to create data layers representing each of the important requirements. Actual species locations and movement patterns from biological surveys can additionally guide us to define the biological landscape.

It is important to recognize that the biological landscape mapped in this way is not a fixed, unchanging entity. Rather dynamic ecological processes alter landscape structure, build and destroy landscape elements, and generate new patterns that may influence

Table 1
Landscape requirements of a hypothetical landscape species

Landscape element type	Area of element[a]	Time of requirement
Preferred food resource	100 km^2	November–April
Other food resource	125 km^2	April–September
Dry season refuges	75 km^2	May–August
Breeding habitat	45 km^2	September–October
Main water source (river)	95 km (length)	September–May

[a] Required to meet the requirements of a target population of species X, a hypothetical landscape species. The target population size is estimated to be 500 individuals for this hypothetical population based on demographic and ecological functioning grounds, see text for details.

the conservation of the target population (Pickett et al., 1997). It is necessary to be aware of these changes and if necessary, enlarge the landscape of consideration to include all elements that have bearing on the population's survival in an area. For example, if a species depends on a certain kind of habitat maintained by periodic flooding of a river, it is not sufficient to identify those habitat patches during the analysis, but also to include within the landscape upstream water control structures which determine flooding regimes (e.g. Richter and Richter, 2000). Such extended landscape units need not be contiguous with the biological landscape that maintains the landscape species population, but they must be clearly specified in the analysis. The complete outer extent of the biological landscape defines the landscape conservation area (LCA).

3.2. Define the human landscape of the stakeholder community

When considering how human beings are using a landscape, the first step is to identify who are the stakeholders in the landscape. The stakeholder in a conservation landscape is any person or organization that uses, administers or has an interest in some part of the LCA, regardless of their feelings about conservation. For example, park guards, nearby farmers, drivers using a road through the area, hunters, and people living downstream all have a stake in the landscape.

For each stakeholder group, the type, spatial and temporal distribution of activity in the LCA must be described. Human uses of the landscape typically include various kinds of resource extraction (e.g. timber, non-timber products, hunting, mining, etc.) settlement and agriculture, transportation routes (whether on roads, rails, or rivers) or units set aside with limited or no use, such as the various classes of protected area. It is also important to recognize important political, administrative and/or cultural divisions in the area, as these often have a large part in defining conservation policy and implementation. To the extent possible, future uses should also be accounted for in this step.

Just as with species requirements, human uses are not abstract entities, but have important distributions in time and space that can be represented as maps or through geographic data layers. For the area of the LCA, separate data layers indicating geographic extent and categorized descriptions of the activities of each stakeholder group should be prepared. For example, an analysis of threats might characterize potential threats by their urgency, severity, probability and recovery time (The Nature Conservancy, 2000). Where necessary, these data layers are also partitioned by important temporal patterns in use. Together these data layers represent the map of the "human landscape" (Fig. 3).

3.3. Intersections define the conservation landscape

Once the biological and human landscapes have been mapped, we combine them through an overlay process to fractionate the landscape by human use and biological need. In geographic information system jargon, we spatially union the two landscapes to create a large number of new landscape elements that are defined by the biological requirements of the landscape species and their human use. For example, if a protected area included three different kinds of habitat, then the result of the union would be three different landscape elements, each described as protected habitat of a given type. Each of these resulting landscape elements is given a unique identification for subsequent steps, an appropriate description of its biological and human uses, and a measurement of its area (Fig. 4). If there are important seasonal differences in the landscape, different unions would be performed for each season.

Further the connectivity relationships between landscape elements should be described. Connectivity may be very different for landscape species than for human begins. Biological connection is based on the behavioral response of the species to landscape structure, including factors like the propensity to dispersal and the response to potential barriers (Bennett, 1990, 1999). Human access is an important determinant of the conservation status of an area, however the connectivity of areas follows different patterns for people than for wildlife. People preferentially use roads and rivers that might be barriers to other species' movements. Because of these differences in how human beings and landscape species perceive connection in the landscape, connectivity relationships should be described separately for these two groups.

3.4. Identify the focal landscape for conservation action

The planning process to this point has emphasized the inclusion of all biological requirements with all stakeholder activities in a landscapes, resulting in a large number of landscape elements potentially relevant to conservation of the landscape species population. The next step is to prioritize those landscape elements to focus on a few elements where conservation actions are necessary. Not all human activities are incompatible with biodiversity conservation, nor are all parts of the landscape necessary for species requirements. But for those human uses that do conflict with species requirements, we need to know where and when they are occurring and how they conflict with the landscape species population.

3.4.1. Evaluate the contribution of each landscape element to species requirements

The criteria for prioritizing landscape elements is based on their contribution to the biological requirements of the landscape species and the frequency with which similar elements are found in the landscapes. The most important landscape elements are those that are essential to the survival of the population and which are unique in the landscape. For example, if there is only one patch of suitable breeding habitat for the population with the LCA, it is essential that that landscape element be maintained and conserved so that the population can persist. Other landscape elements may provide critical resources to the species, but there may be similar such elements on the landscape, any one or any combination of which may be satisfactory. That is, some landscape elements may contribute redundantly to species requirements, so that only a subset of these elements are required for maintenance of the population.

A third class of landscape elements are those which do not contribute directly to species requirements, but are important for their connectivity to other essential elements. These "connector" landscape elements connect two or more unique or redundant landscape elements as defined above. The last class of landscape elements are those which do not contribute to species requirements, either through connection or through function (Fig. 5).

3.4.2. Describe minimal sets of landscape elements

Given these classifications of landscape elements, the next step is to define minimal set of biologically adjacent landscape elements required to support the target population. The minimal criteria may be based on minimal area or number of landscape elements, or may be a function of threat, as discussed below. This subset of the landscape elements of the LCA is named the "focal landscape" and represents those areas and times that are minimally necessary to support the landscape species population (Fig. 6). This set will necessarily include all the "unique" landscape elements as well as a sufficient number of redundant elements to meet all biological requirements, and the set of connector units that assures connectivity of the landscape for the species. Multiple focal landscapes may be possible because of a variety of different redundant and connector elements that can be combined in different configurations. These redundant and connector elements provide some flexibility in the conservation planning, because they provide potentially different ways of integrating human uses and biological requirements. If no other focal landscape is not possible, then the importance of the specified landscape elements is heightened.

3.4.3. Identify threats in each focal landscape element

Once the focal landscape is defined, the role of human activities within each element of the focal landscape is considered. For each human use in a given landscape element, an evaluation is made of how that use threatens the contribution of an element to species requirements. Some human uses may not pose a threat because they do not interfere with species needs or because the use occurs at a different time from when the population requires that element, although caution is warranted because sometimes the effects of human activities can be indirect and counter-intuitive (e.g. Miller and Hobbs, 2000). For example, some non-timber product activities may be compatible with species uses of a patch of forest. However human uses that do conflict with species requirements and occur at the same time and place (e.g. hunting) are reclassified as threats to the population and increase the importance of conservation action within that landscape element.

After determining which landscape elements are under threat and the severity of those threats, a re-evaluation of the stakeholders is made to identify the subset of stakeholders associated with potential threats to the landscape species, across the set of focal landscape elements. This subset will be the subset of stakeholders that must be engaged to accomplish conservation in the landscape. In practice there may be a few key players who can leverage maximum conservation advantage.

3.4.4. Evaluate the overall level of threat of the focal landscape

The overall level of threat to the focal landscape is based on all the threats to its component elements. In general threats to unique landscape elements are the most serious and should receive higher ranking than threats to redundant or connector elements. Connector elements, although themselves not contributing directly to species requirements, may be more important than redundant elements in terms of conservation threat, because redundant elements by definition contribute non-uniquely to population maintenance and so might be replaced in new configurations. Connector elements, in contrast, often connect unique sets of elements, and therefore may be of greater importance.

Threats also need to be considered holistically across the landscape. Alternative focal landscape structures may be possible that minimize the level of threat, even though larger areas of numbers of landscape elements are included. Some threats may reach across multiple landscape elements and thus become the focus of a common conservation response. The final selection of the focal landscape should assure the on-going maintenance of the target population with the least amount of conflict. Thus, this planning effort ultimately directs conservation resources to those times and places, which are the greatest threats to the landscape species population.

3.5. Reiterate for other landscape species

As noted before, conservation activities focused on one species are unlikely to completely capture the variety of landscape level phenomena that are the object of overall biodiversity conservation. The selection of landscape species is designed to maximize the potential of contributing to conservation of other species and to the overall landscape. In most cases suites of landscape species will be used to highlight multiple, overlapping focal landscapes. For each species in the suite, steps (A), and (C)–(E) will need to be repeated to determine the focal landscapes. Presumably the landscape of human activities (B) will not change appreciably during different iterations of the analysis, though the evaluation of different activities as threats will be species specific (i.e. a human activity which is a threat to one species may be inconsequential to another). After the series of focal landscapes are developed, they will be overlaid to find the set of landscape elements and threats which compromise the populations of the multiple species (Fig. 7).

4. Conclusions

The great urgency for effective conservation action requires a planning model that is clear, concise and leads rapidly to the most efficient use of limited time, money and effort for conservation. Landscape species provide a cost-effective way to achieve a significant set of conservation goals in the face of the challenge of addressing multiply threatened species and communities, and the difficulty in adequately understanding highly complex ecosystems and landscapes in a timely fashion. There is a presumed efficiency in using a small set of selected species to achieve a broader set of conservation goals, particularly if those species are chosen for maximum conservation benefit. The large scale and heterogeneous requirements of landscape species confer an important umbrella role in securing the conservation needs of many other species, species assemblages, and larger-scale ecological processes. Moreover, a solid understanding of the interactions of landscape species, which often fulfill ecologically pivotal roles in ecosystems, to other species, communities and phenomena, provides insights into the functionality, integrity and resiliency of natural ecosystems. The result is a planning strategy firmly based on biological knowledge that yields a clear cut vision for where and when to implement conservation action.

Acknowledgements

Many of the background ideas for this paper were developed at a conference supported by the Gilman

Foundation and the Wildlife Conservation Society, 11–15 May 2000. The authors gratefully acknowledge the contributions of the participants. We also acknowledge useful discussions with members of the Wildlife Conservation Society's Living Landscapes Program who will be testing this approach in the field, including Robert Wallace, Lilian Painter, Humberto Gomez, Bryan Curran, Fiona Maisels, Paul Elkan, Sarah Elkan, Connie Clark, Jeff Jorgenson, and Amanda Jorgenson. Additional funding for this work has been received from the US Agency for International Development under Grant no. LAG-A-00-99-00047-00, "Biodiversity Conservation at the Landscape Scale".

References

Allen, T.F.H., 1998. The landscape "level" is dead: persuading the family to take it off the respirator. In: Peterson, D.L., Parker, V.T. (Eds.), Ecological Scale: Theory and Applications. Columbia University Press, New York.

Alpert, P., 1996. Integrated conservation and development projects—examples from Africa. Bioscience 46, 845–855.

Andelman, S.J., Fagan, W.F., 2000. Umbrellas and flagships: efficient conservation surrogates, or expensive mistakes? Proc. Natl. Acad. Sci. 97, 5954–5959.

Anderson, M., Comer, P., Grossman, D., Groves, C., Poiani, K., Reid, M., Schneider, R., Vickery, B., Weakley, A., 1999. Guidelines for Representing Ecological Communities in Ecoregional Conservation Plans. The Nature Conservancy, Washington, D.C.

Arnold, G.W., 1995. Incorporating landscape pattern into conservation programs. In: Hannsson, L., Fahrig, L., Merriam, G. (Eds.), Mosaic Landscapes and Ecological Processes. Chapman & Hall, London.

Austin, M.P., Heyligers, P.C., 1989. Vegetation survey design for conservation: gradsect sampling of forests in north-eastern South Wales. Biol. Conserv. 50, 13–32.

Baker, W.L., 1992. The landscape ecology of large disturbances in the design and management of nature reserves. Landscape Ecol. 7, 181–194.

Baker, W.L., 1995. Longterm response of disturbance landscapes to human intervention and global change. Landscape Ecol. 10, 143–159.

Barrett, C.B., Arcese, P., 1995. Are integrated conservation-development projects (ICDPs) sustainable? On the conservation of large mammals in Sub-Saharan Africa. World Dev. 23, 1073–1084.

Bellman, K., 2000. Towards a system analytical and modelling approach for integration of ecological, hydrological, economical and social components of disturbed regions. Landscape Urban Plan. 51, 75–87.

Bennett, A.F., 1990. Habitat corridors and the conservation of small mammals in a fragmented forest environment. Landscape Ecol. 4, 109–122.

Bennett, A.F., 1999. Linkages in the Landscape: the Role of Corridors and Connectivity in Wildlife Conservation. International Union for Conservation of Nature and Natural Resources, Gland, Switzerland.

Bonan, G.B., 2000. The microclimates of a suburban Colorado (USA) landscape and implications for planning and design. Landscape Urban Plan. 49, 97–114.

Bowen, B.W., Crouse, D., 1997. Landscape-level considerations in the marine realm. In: Meffe, G.K., Carroll, C.R. (Eds.), Principles of Biological Conservation, Sinauer Associates, Inc., Sunderland, MA.

Brandon, K., Redford, K.H., Sanderson, S.E. (Eds.), 1998. Parks in Peril: People, Politics, and Protected Areas. Nature Conservancy, Island Press, Washington, D.C.

Burnett, M.B., August, P., Brown, J., Killingbeck, K.T., 1998. The influence of geomorphological heterogeneity on biodiversity. Part I. A patch-scale perspective. Conserv. Biol. 12, 363–370.

Caro, T.M., O'Doherty, G., 1999. On the use of surrogate species in conservation biology. Conserv. Biol. 13, 805–814.

Casimir, M.J., Rao, A., 1998. Sustainable herd management and the tragedy of no man's land: an analysis of west Himalayan pastures using remote sensing techniques. Hum. Ecol. 26, 113–134.

Cumming, D.H.M., Fenton, M.B., Rautenbach, I.L., Taylor, R.D., Cumming, G.S., Cumming, M.S., Dunlop, J.M., Ford, A.G., Hovorka, M.D., Johnston, D.S., Kalcounis, M., Mahlangu, Z., Portfors, C.V.R., 1997. Elephants, woodlands and biodiversity in southern Africa. S. Afr. J. Sci. 93, 231–236.

deMaynadier, P., Hunter, M.L., 1994. Keystone support. BioScience 44, 2.

Estes, J.A., Duggins, D.O., Rathbun, G.B., 1989. The ecology of extinctions in kelp forest communities. Conserv. Biol. 3, 252–264.

Fairbanks, D.H.K., Benn, G.A., 2000. Identifying regional landscapes for conservation planning: a case study from KwaZulu-Natal, South Africa. Landscape Urban Plan. 50, 237–257.

Faith, D.P., Walker, P.A., 1996. Environmental diversity: on the best-possible use of surrogate data for assessing the relative biodiversity of sets of areas. Biodiversity Conserv. 5, 399–415.

Fleishman, E., Murphy, D.D., Blair, R.B., 2001. Selecting effective umbrella species. Conserv. Biol. Pract. 2, 17–23.

Forman, R.T.T., 1995. Land Mosaics: the Ecology of Landscapes and Regions. Cambridge University Press, Cambridge.

Forman, R.T.T., Collinge, S.K., 1996. The 'spatial solution' to conserving biodiversity in landscapes and regions. In: DeGraaf, R.M., Miller, R.I. (Eds.), Conservation of Faunal Diversity in Forested Landscapes. Chapman & Hall, London.

Forman, R.T.T., Collinge, S.K., 1997. Nature conserved in changing landscapes with and without spatial planning. Landscape Urban Plan. 37, 129–135.

Fox, J., Yonzon, P., Podger, N., 1996. Mapping conflicts between biodiversity and human needs in Lantang National Park, Nepal. Conserv. Biol. 10, 562–569.

Franklin, J.F., 1993. Preserving biodiversity: species, ecosystems, or landscapes? Ecol. Appl. 3, 202–205.

Furze, B., De Lacy, T., Birckhead, J., 1996. Culture, Conservation, and Biodiversity: the Social Dimension of Linking Local Level Development and Conservation Through Protected Areas. Wiley, Chichester, New York.

Groves, C., Valutis, L., Vosick, D., Neely, B., Wheaton, K., Touval, J., Runnels, B., 2000. Designing a Geography of Hope: a Practioner's Handbook for Ecoregional Conservation Planning. The Nature Conservancy, Washington, D.C.

Hansson, L., Fahrig, L., Merriam, G. (Eds.), 1995. Mosaic Landscapes and Ecological Processes. Chapman & Hall, London.

Harrison, S., Fahrig, L., 1995. Landscape pattern and population conservation. In: Hansson, L., Fahrig, L., Merriam, G. (Eds.), Mosaic Landscapes and Ecological Processes. Chapman & Hall, London.

Hoare, R.E., 1999. Determinants of human-elephant conflict in a land-use mosaic. J. Appl. Ecol. 36, 689–700.

Hoare, R.E., Du Toit, J.T., 1999. Coexistence between people and elephants in African savannas. Conserv. Biol. 13, 633–639.

Holling, C.S., 1992. Cross-scale morphology, geometry and dynamics of ecosystems. Ecol. Monogr. 62, 447–502.

Hostetler, M., 1999. Scale, birds, and human decisions: a potential for integrative research in urban ecosystems. Landscape Urban Plan. 45, 15–19.

Huston, M.A., 1994. Biological Diversity: the Coexistence of Species on Changing Landscapes. Cambridge University Press, Cambridge, UK.

Johnston, C.A., 1995. Effects of animals on landscape pattern. In: Hannsson, L., Fahrig, L., Merriam, G. (Eds.), Mosaic Landscapes and Ecological Processes. Chapman & Hall, London.

Kinnaird, M.F., 1998. Evidence for effective seed dispersal by the Sulawesi Red-Knobbed Hornbill, Aceros cassidix. Biotropica 30, 50–55.

Kinnaird, M.F., O'Brien, T.G., Suryadi, S., 1996. Population fluctuation in Sulawesi Red-Knobbed Hornbills: tracking figs in space and time. The Auk 113, 431–440.

Koehler, H.H., 2000. Natural regeneration and succession—results from a 13 years study with reference to mesofauna and vegetation, and implications for management. Landscape Urban Plan. 51, 123–130.

Kozakiewicz, M., 1995. Resource tracking in space and time. In: Hannsson, L., Fahrig, L., Merriam, G. (Eds.), Mosaic Landscapes and Ecological Processes. Chapman & Hall, London.

Lambeck, R.J., 1997. Focal species: a multi-species umbrella for nature conservation. Conserv. Biol. 11, 849–856.

Lawrence, D., Peart, D.R., Leighton, M., 1998. The impact of shifting cultivation on a rainforest landscape in West Kalimantan: spatial and temporal dynamics. Landscape Ecol. 13, 135–148.

Lertzman, K., Fall, J., 1998. From forest stands to landscapes: spatial scales and the roles of disturbances. In: Peterson, D.L., Parker, V.T. (Eds.), Ecological Scale: Theory and Applications. Columbia University Press, New York.

Lima, S.L., Zollner, P.A., 1996. Towards a behavioral ecology of ecological landscapes, Trends in Ecology. Trends Ecol. Evolution 11, 131–135.

McShane, T.O., 1990. Wildlands and human needs—resource use in an African protected area. Landscape Urban Plan. 19, 145–158.

Miller, J.R., Hobbs, N.T., 2000. Recreational trails, human activity, and nest predation in lowland riparian areas. Landscape Urban Plan. 50, 227–236.

Mills, L.S., Soule, M.E., Doak, D.F., 1993. The history and current status of the keystone species concept. BioScience 43, 219–224.

Miller, B., Reading, R., Strittholt, J., Carroll, C., Noss, R., Soule, M., Sanchez, O., Terborgh, J., Brightsmith, D., Cheeseman, T., Forman, D., 1999. Using Focal Species in the Design of Nature Reserve Networks. Wild Earth Winter 1998/1999.

Milner-Gulland, E.J., Mace, R., 1998. Conservation of Biological Resources. Blackwell, Malden, MA.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858.

Naughton-Treves, L., 1998. Predicting patterns of crop damage by wildlife around Kibale National Park, Uganda. Conserv. Biol. 12, 156–168.

Noss, R.F., 1983. A regional landscape approach to maintain diversity. BioScience 33, 700–706.

Noss, R., 2000. Landscape species as conservation tools. Wildlife Conservation Society, Bronx, NY.

Novaro, A.J., Funes, M.C., Walker, R.S., 2000a. Ecological extinction of native prey of a carnivore assemblage in Argentine Patagonia. Biol. Conserv. 92, 25–33.

Novaro, A.J., Redford, K.H., Bodmer, R.E., 2000b. Effect of hunting in source-sink systems. Conserv. Biol. 14, 713–721.

Parker, V.T., Pickett, S.T.A., 1998. Historical contigency and multiple scales of dynamics within plant communities. In: Peterson, D.L., Parker, V.T. (Eds.), Ecological Scale: Theory and Applications. Columbia University Press, New York.

Pickett, S.T.A., Rogers, K.H., 1997. Patch dynamics: the transformation of landscape structure and function. In: Bissonette, J.A. (Ed.), Wildlife and Landscape Ecology: Effects of Pattern and Scale. Springer, New York.

Pickett, S.T.A., White, P.S., 1985. The Ecology of Natural Disturbance and Patch Dynamics. Academic Press, Orlando, FL.

Pickett, S.T.A., Ostfeld, R.S., Shachak, M., Likens, G.E. (Eds.), 1997. The Ecological Basis of Conservation. Chapman & Hall, New York.

Pino, J., Roda, F., Ribas, J., Pons, X., 2000. Landscape structure and bird species richness: implications for conservation in rural areas between natural parks. Landscape Urban Plan. 49, 35–48.

Poiani, K.A., Baumgartner, J.V., Buttrick, S.C., Green, S.L., Hopkins, E., Ivey, G.D., Seaton, K.P., Sutter, R.D., 1998. A scale-independent, site conservation planning framework in The Nature Conservancy. Landscape Urban Plan. 43, 143–156.

Poiani, K.A., Richter, B.D., Anderson, M.G., Richter, H.E., 2000. Biodiversity conservation at multiple scales: functional sites, landscapes and networks. BioScience 50, 133–145.

Power, M.E., Tilman, D., Estes, J.A., Menge, B.A., Bond, W.J., Mills, L.S., Daily, G., Castilla, J.C., Lubchenco, J., Paine, R.T., 1996. Challenges in the quest for keystones. BioScience 46, 609–620.

Pulliam, H.R., Danielson, B.J., 1991. Sources, sinks, and habitat selection: a landscape perspective on population dynamics. Am. Nat. 137, S50–S66.

Redford, K.H., 1992. The empty forest. BioScience 42, 412–422.

Redford, K.H., Feinsinger, P., 2001. The half-empty forest sustainable use and the ecology of interactions. In: Reynolds, J., Mace, G., Robinson, J.G., Redford, K.H. (Eds.), Conservation of Exploited Populations. Cambridge University Press, Cambridge, UK, in press.

Redford, K.H., Richter, B.D., 1999. Conservation of biodiversity in a world of use. Conserv. Biol. 13, 1246–1256.

Redford, K.H., Robinson, J.G., 1995. Sustainability of wildlife and natural areas. In: Munasinghe, M., Shearer, W. (Eds.), Defining and Measuring Sustainability. The Biogeophysical Foundations., United Nations University.

Redford, K.H., Sanderson, E.W., Robinson, J.G., Vedder, A., 2000. Landscape Species and Their Conservation: Report From a WCS Meeting, May 2000. Wildlife Conservation Society, Bronx, NY.

Richter, B.D., Richter, H.E., 2000. Prescribing flood regimes to sustain riparian ecosystems along meandering rivers. Conserv. Biol. 14, 1467–1478.

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

Ruggiero, L.F., Hayward, G.D., Squires, J.R., 1994. Viability analysis in biological evaluations—concepts of population viability analysis, biological population, and ecological scale. Conserv. Biol. 8, 364–372.

Schrijnen, P.M., 2000. Infrastructure networks and red-green patterns in city regions. Landscape Urban Plan. 48, 191–204.

Scott, J.M., Davis, F., Csuti, B., Noss, R., Butterfield, B., Groves, C., Anderson, J., Caicco, S., D'Erchia, F., Edwards, T.C., Ulliman, J., Wright, R.G., 1993. Gap analysis: a geographical approach to protection of biological diversity. Wildlife Monogr. 123, 1–41.

Simberloff, D., 1998. Flagships, umbrellas, and keystones: is single-species management passe in the landscape era? Biol. Conserv. 83, 247–257.

Steiner, F., Blair, J., McSherry, L., Ghuathakurta, S., Marruffo, J., Holm, M., 2000a. A watershed at a watershed: the potential for environmentally sensitive area protection in the upper San Pedro Drainage Basin (Mexico and USA). Landscape Urban Plan. 49, 129–148.

Steiner, F., McSherry, L., Cohen, J., 2000b. Land suitability analysis for the upper Gila River watershed. Landscape Urban Plan. 50, 199–214.

Stern, S.J., 1998. Field studies of large mobile organisms: scale, movement and habitat utilization. In: Peterson, D.L., Parker, V.T. (Eds.), Ecological Scale: Theory and Applications. Columbia University Press, New York.

The Nature Conservancy, 2000. The Five-S Framework for Site Conservation: a Practioner's Handbook for site Conservation Planning and Measuring Conservation Success. The Nature Conservancy, Washington, D.C.

Thomlinson, J.R., Rivera, L.Y., 2000. Suburban growth in Luquillo, Puerto Rico: some consequences of development on natural and semi-natural systems. Landscape Urban Plan. 49, 15–23.

Thompson, J.W., Sorvig, K., 2000. Sustainable landscape construction: a guide to green building outdoors. Island Press, Washington, D.C.

Thrash, I., 1998. Impact of large herbivores at artificial watering points compared to that at natural watering points in Kruger National Park, South Africa. J. Arid Environ. 38, 315–324.

Tilman, D., Kareiva, P. (Eds.), 1997. Spatial Ecology: the Role of Space in the Population Dynamics and Interspecific Interactions. Princeton University Press, Princeton, NJ.

Treu, M.C., Magoni, M., Steiner, F., Palazzo, D., 2000. Sustainable landscape planning in Cremona, Italy. Landscape Urban Plan. 47, 79–98.

Turner, M.G., 1989. Landscape ecology: the effect of pattern on process. Ann. Rev. Ecol. Syst. 20, 171–197.

Turner, M.G., Hargrove, W.W., Gardner, R.H., Romme, W.H., 1994. Effects of fire on landscape heterogeneity in Yellowstone National Park, Wyoming. J. Vegetation Sci. 5, 731–742.

Vedder, A., 2000. Minutes of the First Meeting of the Living Landscapes Program, La Paz, Bolivia, 22 September–6 October 2000. Wildlife Conservation Society.

Verlinden, A., 1997. Human settlements and wildlife distribution in the Southern Kalahari of Botswana. Biol. Conserv. 82, 129–136.

Wallace, R.B., Gómez, H., Coppolillo, P., 2001. Criterios de seleccion de especies paisaje en el noroeste de los Andes de Bolivia. In: Proceedings of the V Congroso Internacional de Manejo de Fauna el la Amazonia y Latinoamerica, in press.

Weaver, J.L., Paquet, P.C., Ruggiero, L.F., 1996. Resilience and conservation of large carnivores in the Rocky Mountains. Conserv. Biol. 10, 964–976.

Weins, J.A., 1989. Spatial scaling in ecology. Funct. Ecol. 3, 385–397.

White, P.S., 1979. Pattern, process and natural disturbance in vegetation. Botanical Rev. 45, 229–299.

White, P.S., Harrod, J., 1997. Disturbance and diversity in a landscape context. In: Bissonette, J.A. (Ed.), Wildlife and Landscape Ecology: Effects of Pattern and Scale. Springer, New York.

Whited, D., Galatowitsch, S., Tester, J.R., Schik, K., Lehtinen, R., Husveth, J., 2000. The importance of local and regional factors in predicting effective conservation. Planning strategies for wetland bird communities in agricultural and urban landscapes. Landscape Urban Plan. 49, 49–65.

Wilcox, B.A., 1984. In situ conservation of genetic resources: determinants of minimum area requirements. In: McNeely, J.A., Miller, K.R. (Eds.), National Parks, Conservation and Development, Smithsonian Institution Press, Washington, D.C.

Woodroffe, R., Ginsberg, J.R., 1998. Edge effects and the extinction of populations inside protected areas. Science 280, 2126–2128.

Wu, X.B., Smeins, F.E., 2000. Multiple-scale habitat modeling approach for rare plant conservation. Landscape Urban Plan. 51, 11–28.

Zander, P., Kachele, H., 1999. Modeling multiple objectives of land use for sustainable development. Agric. Syst. 59, 311–325.

Eric W. Sanderson is the Associate Director of the Landscape Ecology and Geographic Analysis Program in the International Conservation Programs of the Wildlife Conservation Society (WCS). Sanderson received his PhD in ecology (emphasis on

ecosystem and landscape ecology) from the University of California, Davis, in 1998. His research interests include conservation landscape ecology, applications of geospatial technologies to conservation, and the historical and geographical context of conservation from site-based efforts to global conservation planning.

Kent H. Redford is the Vice President of Conservation Strategy in the International Conservation Programs of the Wildlife Conservation Society. For 5 years previously, Redford was the Director of Conservation Science in the Latin American and Caribbean Division at The Nature Conservancy and before that, he was a professor at the University of Florida. His PhD in biology is from Harvard University in 1984. He has written extensively on the theory and practice of conservation.

Amy Vedder is Vice President of the Living Landscapes Program of the Wildlife Conservation Society. Previously Vedder was Director of the WCS Africa Program and Coordinator of the Biodiversity Program. Vedder has conducted extensive research on the ecology of mountain gorillas and other forest primates and earned her PhD in biology is from the University of Wisconsin, Madison, in 1989. She has worked on applied conservation field projects for the last 25 years.

Peter ("Pete") B. Coppolillo is a Landscape Ecologist in the Living Landscapes Program of the Wildlife Conservation Society. Coppolillo earned his PhD in ecology from the University of California, Davis, in 1999. His research interests include interactions between wildlife and livestock, and western Tanzania, particularly the area in and around Katavi National Park.

Sarah E. Ward was Program Officer for the Living Landscapes Program of the Wildlife Conservation Society. Ward received her MS in Geography from the University of Wisconsin, Madison, in 1998. Currently she is conducting research on oceanic birds on the French Frigate Shoals in the Hawaiian Islands.