

Selection criteria for suites of landscape species as a basis for site-based conservation

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Abstract

Focal-species approaches provide tractable frameworks for structuring site-based conservation, but explanations of how and why focal species are chosen are often lacking. This paper outlines the rationale and selection criteria for one such strategy: the “Landscape Species Approach.” We define five criteria for selecting landscape species (area requirements, heterogeneity, ecological function, vulnerability, and socioeconomic significance) and illustrate the process using data from two landscapes, the northwestern Bolivian Andes and northern Congo. Candidate species from each site were scored and suites of complementary landscape species were assembled. Like all focal-species approaches (and indeed all conservation planning), this approach is not without biases. However, by making our assumptions explicit and allowing evaluation of the inherent biases, we attempt to provide a transparent, replicable method for identifying species around which to structure site-based conservation (landscape species). The process is also useful for identifying data gaps, ranking threats, and setting research priorities. Clear justification and selection criteria should accompany any focal species strategy to allow methods to be replicated, tested, and refined.

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1. Introduction

Prescriptive approaches to conservation address two fundamental questions: *where* to do conservation, and *how* to do it (Margules and Pressey, 2000; Redford et al., 2003). *Where* approaches, also called “priority setting” (Groves et al., 2002), seek to identify optimal or underrepresented places for conservation action. *How* approaches, also referred to as “reserve design” and “reserve management,” attempt to build effective strategies for carrying out conservation action once sites have been identified (Sanderson et al., 2002).

How approaches are generally applied to protect a particular place or “target landscape,” a process we label “site-based conservation.” But defining the spatial limits of a target landscape or the biodiversity that occurs there is often problematic (Redford and Richter,

1999). Focal-species approaches provide a lens through which to evaluate the size, quality and configuration of wild landscapes (Franklin, 1993; Noss, 1996; Lambeck, 1997) and allow conservation practitioners to structure site-based conservation actions according to the needs of species. Unfortunately, the choice of focal species is often arbitrary and rarely justified (Andelman and Fagan, 2000). This paper presents selection criteria and ecological justification for one focal-species approach to site-based conservation: the “Landscape Species Approach” (Sanderson et al., 2002).

Sanderson et al. (2002) define landscape species by their “use of large, ecologically diverse areas and their impacts on the structure and function of natural ecosystems... their requirements in time and space make them particularly susceptible to human alteration and use of wild landscapes” (Sanderson et al., 2002, p. 43). Because landscape species require large, wild areas, it is posited that they serve a significant umbrella function (*sensu* Caro and O’Doherty, 1999); in other words, meeting their needs will provide substantial protection

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for the species with which they co-occur and the wild lands on which both depend.

It is widely recognized that the umbrella function of suites of species is greater than that of individual species (Lambeck, 1997; Andelman and Fagan, 2000; Williams et al., 2000). The landscape species approach maps spatially-explicit requirements for a suite of landscape species, and based on their overlap with human land uses, identifies key threats to be addressed by conservation action (Sanderson et al., 2002).

Here, we present a method for selecting a suite of landscape species around which to structure site-based conservation (see Fig. 1). This and other focal-species approaches, like all conservation planning, carry inherent biases (Lindenmayer et al., 2002), and may be constrained by incomplete or inconsistent data. Recognizing these challenges, we have attempted to develop a method that identifies data deficiencies and

potential biases, so that these may be confronted and where possible dealt with, rather than ignored or unwittingly accepted. This method should not be construed as a comprehensive approach for conservation planning; instead, it serves as a foundation to be combined with other tools like special element mapping (Noss, 1996).

Using data for two tropical sites (described in Section 2), we define the selection criteria and the scoring process (Sections 3.1–4), examine the effects of individual criteria on species rankings (Sections 5–6.1) and present the results for each site (Section 6.2). We conclude by highlighting some challenges encountered and issues that may affect the selection of landscape species at other sites (Section 7).

2. Study sites

Data are presented from the greater Madidi landscape in the northwestern Bolivian Andes (henceforth: Madidi), and the Ndoki-Likouala landscape, in northern Congo Republic (henceforth: Ndoki).

The Madidi site covers an area of approximately 40,000 km² centered on Madidi National Park, Apolobamba Area of Integrated Management, the Tacana Indigenous Reserve, and the Pilon Lajas Biosphere and Indigenous Reserve. Spanning elevations from over 6000 m to around 200 m, Madidi is one of the most species-rich protected area complexes in the world. Highland portions of the Madidi landscape harbor important populations of Andean deer (*Hippocamelus antisensis*) and Andean condor (*Vultur gryphus*), the cloud forests reaching from Apolobamba to Pilon Lajas form the largest intact tract of spectacled bear (*Tremarctos ornatus*) habitat in the Andes, and the lowlands support dense aggregations of white lipped peccary (*Tayassu pecari*), tapir (*Tapirus terrestris*) and jaguar (*Panthera onca*). The major threats in the greater Madidi landscape are unregulated land- and resource-use stemming from rapid colonization and a legal/regulatory framework fraught with internal conflicts.

The Ndoki landscape covers approximately 30,000 km² of lowland Guineo-Congolian forest with large and intact populations of most large mammals (Fay and Agnagna, 1991; Blake et al., 1995). Forest types vary from semi-deciduous forest in the northwest to swamp forest in the southeast. Ndoki harbors important populations of forest elephants (*Loxodonta africana cyclotis*), western lowland gorillas (*Gorilla gorilla gorilla*), chimpanzees (*Pan troglodytes troglodytes*) and bongo (*Tragelaphus euryceros*). The region has extremely low human population density (<1/km²), and until recently has been isolated from modern human influence. Wildlife in the region are partially protected in Nouabalé-Ndoki National Park and Lac Télé-Likouala aux

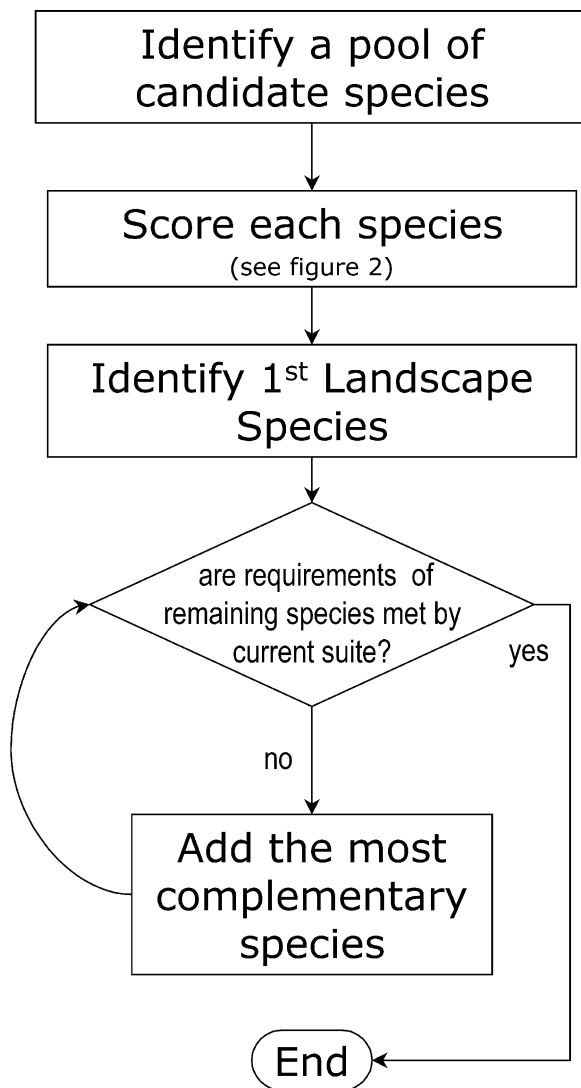


Fig. 1. A schematic summary of the steps included in selection process for a suite of landscape species.

Herbes Community Reserve. Nouabalé-Ndoki National Park is also contiguous with Dzanga Sangha Protected Area Complex and Lac Lobeke National Park in the Central African Republic (C.A.R.) and Cameroon, respectively. Major threats in the Ndoki landscape stem from rapid development of a logging industry throughout the landscape, the associated hunting and export of large volumes of wild meat, and the creation of logging communities, which, in this once sparsely populated region, greatly increases the pressure on forest resources.

At each site, selection teams were assembled from field biologists, management personnel and others with local expertise or knowledge of the species being evaluated. Selection teams met to identify candidate species, score each according to the criteria presented below and build the suite of landscape species, as summarized in Fig. 1. Data for the criteria were drawn from the following sources (listed in order of priority): published data, unpublished data or “gray literature,” and expert opinions.

3. Criteria for scoring candidate species

In the sub-sections that follow we, (1) define the five categories used as selection criteria for suites of landscape species (area, heterogeneity, vulnerability, ecological functionality, and socio-economic significance) and provide the ecological justification for each; (2) describe the scoring process within each category, and (3) discuss how the five category scores were aggregated to form a single metric. This process is schematically summarized in Fig. 2.

Candidate species (numbering 10–25) were selected from the pool of all species in each landscape. Recognizing the potential (and unavoidable) subjectivity that could arise from a biased candidate pool, we included all species with a reasonable chance of being selected (indeed, only larger-bodied vertebrates were considered in the two analyses presented here; see Section 7.2). Care was also taken to ensure that the candidate pool collectively occupied the full range of habitat and land use types in each target landscape (see Section 4).

3.1. Area

Wide-ranging species, or more specifically, species with large home ranges are difficult to conserve and when confronted with human disturbance are generally among the first to go locally extinct (Woodroffe and Ginsberg 1998, 2000; Brashares, in press). Conserving intact habitats large enough for area-limited species has obvious benefits for less wide-ranging species (Noss, 1996; Lambeck, 1997). Besides the risk of local extinction, another implication of large home range size is

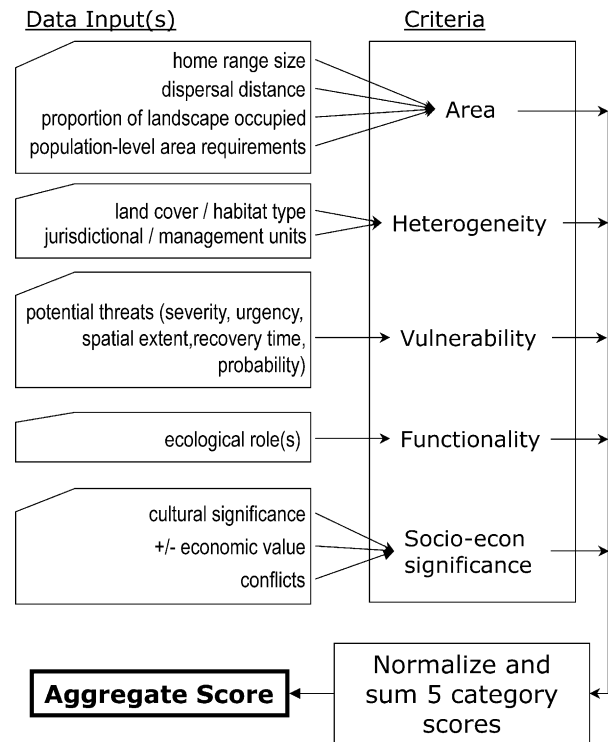


Fig. 2. A schematic summary of the data constituting the five selection criteria for landscape species, and their combination to form an aggregate score. See Sections 3.1–3.5 for descriptions of the data inputs and justification of each criterion and Section 3.6 for a description of the aggregation process.

that many habitat blocks, jurisdictional, or management units may not be large enough to support viable or ecologically functional populations of wide-ranging species (e.g. Ortega-Huerta and Medley, 1999). In these cases landscape connectivity becomes a concern for all management purposes, not just population persistence (Briggs, 2001). Another important variable related to area is dispersal distance (Delibes et al., 2001; Fahrig 2001; Singer et al. 2001). Just as individuals’ movements within their home ranges can functionally link landscape elements, so can dispersal (McCullough, 1996; Brooker et al., 1999; South, 1999; Novaro et al., 2000).

We scored candidate species’ area requirements according to four factors (Fig. 2). Protected areas, and by extension our planning efforts, must strive for long-term viability; home range estimates for seasons or single years generally underestimate longer-term area requirements (e.g. Kenward et al., 2001). Therefore, we used the longest tractable time scales for our home range estimates, being careful to apply the same standard to all species considered. For migratory species moving well beyond the target landscapes, we considered area requirements for the time in which they occupy the landscape.

Unfortunately, home range estimates, particularly site-specific ones, are rare. Facing a paucity of home range data and biases from cross-taxon methodological

differences in home range estimation, the Madidi selection team chose to categorize home range estimates into four bins: <10, 10–25, 25–50, and >50 km². The team considered this a more robust approach because it reduced inter-site and inter-taxon biases in home range estimates. The Ndoki selection team chose to maintain home range as a continuous variable, in part because home range estimates there varied over two orders of magnitude, rendering the potential biases insignificant.

For many species, dispersal distance might be considered one surrogate for lifetime home range, but because so little is known about dispersal, particularly in large mammals (Sinclair, 1992), it was scored separately and as a binary variable. Species likely to disperse farther than 10 km were given a score of one and others a score of zero.

We also considered the proportion of the target landscape occupied by each candidate species. We included this variable because for efficiency, suites of focal-species should be as small as possible (Lambeck, 1997; Andelman and Fagan, 2000), and species occupying only a tiny proportion of a focal landscape may be more appropriately treated as special elements (see Sections 7.1 and 7.3). This value was scored from zero to one as the estimated proportion of the landscape in which the species occurs.

Finally, selection teams considered the area required by a population of each species and whether management units and/or habitat patches within the target landscape must be connected to provide sufficient area for viable populations of each species. Species with population-level area requirements larger than individual habitat patches or management units were given a score of one. Spatially explicit population viability analyses for every species are obviously out of the question, so these scores must be based on the best available data and expert opinions.

3.2. Heterogeneity

Some species require more than just large areas. In many cases wildlife need specific habitat or vegetation types during their life cycle (Leopold, 1933; Dunning et al., 1992; Kozakiewicz, 1995; Mysterud et al., 2001). Bongo (*Tragelaphus euryceros*) for example, range widely in Afro-tropical forest, but probably also require forest clearings for bulk grazing and/or mineral licks (Elkan, P. unpublished data, 1999; Klaus-Hugi et al., 2000). Homogeneous areas, no matter how large, may not be suitable. Planning for species like bongo forces us to evaluate the *composition* of landscape elements, so that all required resources are protected, as well as their *configuration*, to allow individuals to move between necessary elements (Dunning et al., 1992; Lima and Zollner 1996). The importance of configuration may be more general than previously considered; growing

numbers of studies demonstrate “landscape” and “matrix” effects, where species composition or abundance within patches reflects not only the properties of the patch itself, but also those of the surrounding matrix (Hobbs, 1993; Jokimäki and Huhta, 1996; Fahrig and Jonsen, 1998; Cantero et al., 1999; Estades and Temple, 1999; Lidicker, 1999).

A second benefit of considering habitat heterogeneity is that a smaller number of species is required to “cover” a single landscape. This is especially true when heterogeneity is considered in concert with area requirements. Minimizing the number of focal-species reduces modeling, analysis and planning burdens and should streamline conservation efforts (Lambeck, 1997; Caro and O’Doherty, 1999; Fleishman et al., 2000). Heterogeneous habitats also tend to harbor more species than similar-sized patches of uniform habitat (Fox and Fox, 2000), which may make species with heterogeneous habitat requirements better “coarse filters” for biodiversity (Hunter, 1991; Poiani et al., 1998; Andelman and Fagan, 2000). This coarse-filter role may be particularly significant in poorly-studied systems where conservation practitioners may be unaware of significant portions of the biodiversity occurring there (Hunter, 1991).

Jurisdictional heterogeneity is also significant because differences in management can have strong effects on landscape structure (Schonewald-Cox, 1988; Landres et al., 1998), which in turn will affect ecological processes (Wiens, 1992) and necessitate management across jurisdictional boundaries (Landres, 1998; Briggs, 2001). For example, in a single day an individual elephant in the Ndoki landscape may walk from a forestry concession into a national park, across an international border and onto village communal land, all within a single habitat type. Management decisions in each of these areas will affect the conservation of elephants in the others. Therefore, connectivity and compatible management practices among administrative or political units may be just as significant as the connections among habitat types. Considering jurisdictional heterogeneity helps to address the management problems posed by species ranging across management or political boundaries.

We scored heterogeneity using three measures: (1) the proportion of habitat types within the landscape that individuals use, (2) the proportion of jurisdictional types individuals use, and (3) the proportion of habitat types on which the status of the species depends.

The first two measures, the proportions of habitat and jurisdictional types individuals use, were scored by tallying all the habitat or jurisdictional types (within the landscape) that an individual *must* use during its entire life cycle. The term “must” was included to distinguish a species requiring resources in different landscape elements from a widespread generalist whose local distribution

spans a number of habitat (or jurisdictional) types, but for which individuals can complete their life cycles in just one type. The third measure (proportion of habitat types on which the status of the species depends), is a population-level question, considering the number of habitat types on which the *population* depends, regardless of whether *individuals* require heterogeneity or not.

3.3. Vulnerability

Species exposed to a greater number of land uses are more likely to conflict with humans (Newmark et al., 1994), and often experience higher mortality as a result (Woodroffe and Ginsberg, 2000). Planning for sensitive species helps to identify the maximum levels of threat(s) considered acceptable (Lambeck, 1997) and will help maintain the integrity of wild or undisturbed portions of target landscapes. Threats can be characterized according to their severity, urgency, probability of occurrence, and the area they affect (Nature Conservancy, 2000). The number and seriousness of threats affecting each species were also considered in selecting landscape species.

For each candidate species we listed the major land uses within the landscape (recognizing them as potential threats) and scored each according to its effect on that species. Each land use was scored according to the severity (*S*) of its effect on the species, urgency (i.e. the timescale over which it would occur; *U*), the time it would take the species to recover from the threat (*R*), the proportion of the candidate species' local distribution affected (*Pa*), and its probability of occurrence (*Po*). Each category was scored using the criteria in Table 1.

We then summed over all potential threats to combine these measures into a site-specific "vulnerability index" for each candidate species:

$$\sum\{(U + R) \times S \times Pa \times Po\}$$

This index aggregates the threats to each species scaled according to their overall importance. The same is true for threats affecting only a small proportion of the landscape. Severity (*S*), probability of occurrence (*Po*), and proportion of local extent affected (*Pa*), are included as multipliers so that insignificant human activities (i.e. those that have no effect, affect a tiny area, or are exceedingly unlikely) do not contribute to a species' threat index.

3.4. Functionality

Some species have particularly strong effects on the structure and function of natural ecosystems. Beavers create wetlands by damming rivers, tapir and elephants disperse seeds and thin the forest understory, and top predators may control the abundance and composition of prey communities. When these strong effects occur

Table 1
Vulnerability criteria and scores

<i>Severity (S)</i>	
No (or a positive) effect on the species	0
Little measurable but small effect on density or distribution	1
Substantial effect on density or distribution, but local eradication unlikely	2
Serious effects, local eradication a possibility	3
<i>Urgency (U)</i>	
Will not happen in > 10 years	0
Could happen over 3–10 years	1
Could (or will) within 1–3 years	2
Must act immediately (i.e. threat is currently happening)	3
<i>Proportion of local extent affected (Pa)</i>	
0	0
0.01–0.1	1
0.1–0.25	2
0.25–0.5	3
> 0.5	4
<i>Recovery time (R)</i>	
Immediate or < 1 year	0
1–10 years recovery	1
10–100 years recovery	2
100+ years or never for recovery	3
Probability (<i>Po</i>)	0–1

The five components of vulnerability were scored as above for all candidate landscape species and aggregated into a single vulnerability score using the equation in the text.

maintaining healthy populations of ecologically pivotal species will help conserve healthy communities and ecosystems (Estes et al., 1989; Power et al., 1996).

To quantify the ecological significance of candidate species, we considered the number of ecological functions in which they are involved and their significance in each. The overall list of functions may vary between sites but generally includes predation, seed dispersal, seed predation, pollination, mechanical disturbance, strong competitive interactions, and acting as a prey base. Under our scoring system, species playing no role in a particular function received a score of zero for that function, those with a suspected or weak effect received a score of one, species with clear effects a score of two, and species with strong effects were given a score of three. Sub-scores for candidate species were summed over the list of functions for each site to produce an overall functionality score for each species.

3.5. Socioeconomic significance

The final category for selecting landscape species is socio-economic significance. Numerous examples illustrate that the human social environment can drastically affect conservation outcomes (Newmark et al., 1994; Western and Wright, 1994; Painter and Durham, 1995). Wildlife may clash with people because they raid crops

(Naughton-Treves, 1998), prey on livestock (Mishra, 1997; Linnell et al., 2000), transmit diseases to domestic animals or people (Grootenhuis and Olubayo, 1993), or compete for resources (Du Toit and Cumming, 1999). Wildlife may also have important positive benefits, like serving as a cultural icon or totem, forming a significant portion of people's diets, (Bodmer et al., 1997), or providing opportunities for income (e.g. ecotourism).

As a proxy for socio-economic significance, we used five binary scores based on whether the species: (1) is a potential flagship species (for people at the site or elsewhere), (2) has positive local cultural value, (3) has negative local cultural value, (4) has negative economic value, or (5) has positive economic value when sustainably managed. Binary scores for each of the five characteristics were added to form the socio-economic category score. Scoring positive and negative cultural values independently effectively separates species with both positive and negative value from those with one or the other.

3.6. Aggregating selection criteria scores into a single metric

Next, scores from the five categories were aggregated into a single metric to identify the species demonstrating the greatest number of landscape species' attributes to the greatest degree (Fig. 2). Because the maximum possible score within each category varied, category scores were normalized to avoid inadvertent weighting. We calculated normalized scores by scaling each species' score within each of the five categories according to the maximum score in that category, so that all values fell between zero and one. Scores from all five categories were summed to form a single aggregate score for each species.

4. Building the suite

Once all candidate species were scored, we constructed the suite of landscape species for each site, as shown in Fig. 1. The species with the highest aggregate score was selected as the first landscape species at each site. Subsequent species were chosen one at a time from the next five highest-ranked candidates. At each step the species considered most complementary to the existing suite was added as a landscape species, and the species with the next-highest aggregate score was added to the group of five considered for complementarity.

We defined complementarity as minimum overlap in habitat requirements, distributions across jurisdictional units and distinctiveness of threats encountered. To score spatial complementarity for each species we used the proportion of that species' habitat types, jurisdictional types, and threats that overlap with the existing suite. We suggest that in cases where two species are considered equally complementary and their aggregate

scores are not substantially different, degree of vulnerability to threats, trophic position or taxonomic uniqueness can be considered to break ties (such a situation was not encountered by either selection team).

Species were added to the suite until the needs of the most complementary species (i.e. the next species to be added) were already met by the current suite of landscape species. Thus the suite was effectively "closed" when the requirements of the remaining candidate species were captured by those of the suite of landscape species. Our experience at the two sites discussed here (and five others) suggests that suite sizes will range from 4 to 6 species, depending on the diversity of habitats, threats and the size of the candidate species pool.

5. Sensitivity analysis

We conducted a sensitivity analysis to determine whether any of the five selection categories were redundant or had a disproportionate effect on the candidate species aggregate scores.¹ We evaluated the effect of each category by calculating aggregate scores with and without that category. For each we summed the absolute values of changes in all species ranks ("sum of changes in rank" or SCR) with the elimination of that category. The SCR offers a simple measure of the effect of each category on the ranks of candidate species. The size of the candidate pool sets an upper limit on maximum possible change in rank. Because the Madidi and Ndoki candidate pools contained different numbers of species, comparisons between the two pools are not valid.

Pairwise correlations between individual categories' scores, aggregate scores, and overall ranks were also examined to assess redundancy among categories. Redundancy could stem from covariation in species' life history characteristics, which might similarly affect the area, heterogeneity, vulnerability and possibly socio-economic significance scores. As with SCR, correlations were examined only within sites' data sets.

6. Results

6.1. Sensitivity results

The data presented in Table 2 suggest that categories' scores did not covary strongly, and there was no consistent pattern between the Ndoki and Madidi data sets. We concluded that covariation between categories was not a significant concern.

¹ We also evaluated the effects of ranking species within categories, weighting category scores, and varying the numbers of species considered for complementarity. None had substantial effects, so results from these analyses are not reported.

Table 2
Correlation coefficients among categories and aggregate scores

	Heterogeneity	Area	Vulnerability	Socio-econ	Function	Aggregate
Heterogeneity	1					
Area	0.35 / 0.53	1				
Vulnerability	−0.56 / 0.53	−0.23 / 0.36	1			
Socio-econ	−0.09 / 0.45	0.00 / 0.64	−0.29 / 0.77	1		
Function	0.40 / 0.44	−0.21 / 0.67	0.32 / 0.36	−0.45 / 0.44	1	
Aggregate	0.35 / 0.77	0.50 / 0.82	0.33 / 0.76	0.12 / 0.85	0.50 / 0.72	1

Cells contain correlation coefficients among landscape species selection criteria and between each criterion and the aggregate score. Results in each cell appear as Madidi/Ndoki.

The correlations for Madidi show that no variable was substantially more-strongly correlated with the aggregate score than others. All individual categories for Ndoki were more correlated with the aggregate score than the Madidi values, but it appears that none dominated the overall score.

These results were corroborated by the SCR data in Table 3, which showed that removal of area and ecological functionality categories had the strongest effects on Madidi species' ranked aggregate scores, and that the removal of individual criteria had more even effects in the Ndoki data set.

6.2. Scores and suites from the sites

Scores for Madidi and Ndoki species are presented in Tables 4 and 5, respectively. To illustrate the criteria for complementarity and the process of building the suite, we describe this process in detail for the Madidi dataset.

With the highest aggregate score (see Table 4), white-lipped peccary was selected as the first landscape species, "covering" six major habitat types and eight major threats (see Fig. 3). The species with the next five highest scores were then considered for complementarity: jaguar, spectacled bear, condor, catfish and anaconda. Spectacled bears, which addressed four additional threats (50% overlap) and added three additional habitat types (33% overlap), were the most complementary and were added as the second landscape species. The third species added was the Andean condor, covering

two more habitat types (75% overlap) and addressing one other major threat (50% overlap). In adding the fourth species, giant river otter was more complementary (addressing two more threats with 80% overlap) than catfish (addressing one threat, 88% overlap). However, otters only occupy a handful of streams and rivers in the Madidi landscape, so the selection team found it more appropriate to treat them as a special element (see Sections 7.1 and 7.3), and catfish was added as the fourth species. Next, jaguar was added as the fifth species to the suite. While jaguars' spatial distribution is "captured" by the white-lipped peccary distribution, and spectacled bears are persecuted for raiding livestock in the highlands, jaguar address an acute issue in the lowlands: livestock depredation (and the subsequent persecution of carnivores). With the addition of jaguar, the suite of landscape species "covered" all major habitat types for and threats to the next five species considered for complementarity. However, one habitat type, the Ulla Ulla plane *puma*, remained uncovered by the suite. Vicuña, which fell outside the next five species, occupy the Ulla Ulla plane. Given the extraordinary diversity of the Madidi landscape, we chose to override the 'next five species' rule and include vicuña as the final member of the suite of landscape species. (Section 7.2, discusses this and other methodological decisions in more detail.) The selection team examined the remaining candidate species and concluded that their needs were covered by those of the current suite, so the suite was closed with six species: white-lipped peccary, spectacled bear, condor, catfish, jaguar, and vicuña.

Applying the selection criteria to the candidate species for Ndoki produced the scores in Table 5. Elephant dominated the selection scores with the highest scores in every category except vulnerability. Complementarity was scored as in Madidi and produced the following suite of landscape species: elephant, chimp, bongo, forest buffalo and dwarf crocodile.

7. Discussion and conclusions

Scoring each species was a useful exercise even before aggregating species' scores or identifying the suite of

Table 3
Effect of each selection category as measured by SCR

Category	SCR value	
	Madidi	Ndoki
Heterogeneity	8	12
Area	32	14
Vulnerability	19	16
Ecological functionality	20	10
Socioeconomic significance	16	20

The "Sum of Changes in Rank" (SCR) for each category equals the summed absolute values of all candidate species' changes in rank when that category is removed from the aggregate score.

Table 4
Aggregate and individual category scores for Madidi species

Species	Category					Aggregate score
	Heterogeneity	Area	Vulnerability	Function	Socio-econ	
White-lipped peccary * <i>Tayassu pecari</i>	0.78	1.00	0.86	1.00	0.40	4.03
Jaguar * <i>Panthera onca</i>	0.78	1.00	0.86	0.50	0.80	3.94
Spectacled bear * <i>Tremarctos ornatus</i>	0.74	0.60	0.78	0.50	0.80	3.41
Condor * <i>Vultur gryphus</i>	0.88	1.00	0.28	0.25	1.00	3.41
Surubi catfish * <i>Pseudoplatystoma sp.</i>	0.59	1.00	0.90	0.50	0.40	3.39
Anaconda <i>Eunectes murinus</i>	0.67	1.00	0.79	0.50	0.40	3.35
Tapir <i>Tapirus terrestris</i>	0.78	0.60	0.70	1.00	0.20	3.28
Giant river otter <i>Pteronura brasiliensis</i>	0.67	0.60	0.87	0.50	0.60	3.24
Spider monkey <i>Ateles paniscus</i>	0.67	0.40	1.00	0.50	0.60	3.17
Macaw spp. <i>Ara spp.</i>	0.35	0.80	0.85	0.50	0.60	3.10
Puma <i>Puma concolor</i>	1.00	0.80	0.32	0.50	0.40	3.02
Giant river turtle <i>Podocnemis expansa</i>	0.53	1.00	0.96	0.00	0.40	2.89
Vicuña * <i>Vicugna vicugna</i>	0.32	0.60	0.31	0.25	1.00	2.48

Aggregate and individual category scores for Madidi (northwestern Bolivian Andes) according to the criteria listed in Fig. 2 and described in the text.

* Indicates inclusion in the suite of landscape species.

Table 5
Aggregate and individual category scores for Ndoki (Congo) species

Species	Category					Aggregate score
	Heterogeneity	Area	Vulnerability	Socio-econ	Function	
Elephant * <i>Loxodonta africana cyclotis</i>	1.00	1.00	0.92	1.00	1.00	4.92
Gorilla <i>Gorilla gorilla gorilla</i>	0.44	0.67	1.00	0.77	0.71	3.59
Chimp * <i>Pan troglodytes troglodytes</i>	0.55	0.79	0.83	0.69	0.71	3.58
Bongo * <i>Tragelaphus euryceros</i>	0.81	0.85	0.77	0.85	0.29	3.57
African grey parrot <i>Psittacus erithacus</i>	0.72	0.89	0.44	0.31	0.86	3.22
Buffalo * <i>Syncerus caffer nanus</i>	0.48	0.57	0.90	0.54	0.43	2.93
Leopard <i>Panthera pardus</i>	0.11	0.71	0.65	0.77	0.43	2.66
Red river hog <i>Potamochoerus porcus</i>	0.09	0.66	0.61	0.38	0.71	2.45
Hornbill <i>Ceratogymna spp.</i>	0.02	0.96	0.36	0.38	0.57	2.29
Hippopotamus <i>Hippopotamus amphibius</i>	0.22	0.41	0.66	0.46	0.29	2.04
Nile crocodile <i>Crocodylus niloticus</i>	0.09	0.35	0.48	0.62	0.43	1.97
Swamp otter <i>Aonyx congicus</i>	0.57	0.33	0.44	0.00	0.29	1.64
Crowned eagle <i>Stephanoaetus coronatus</i>	0.28	0.51	0.40	0.00	0.43	1.62
Golden cat <i>Felis aurata</i>	0.21	0.19	0.57	0.15	0.43	1.55
Dwarf crocodile * <i>Ostaeolaemus tetraspis</i>	0.09	0.27	0.49	0.23	0.43	1.51
Spot-necked otter <i>Lutra maculicollis</i>	0.06	0.29	0.44	0.00	0.29	1.08

Aggregate and individual category scores for Ndoki (northern Congo) according to the criteria listed in Fig. 2 and described in the text.

* Indicates inclusion in the suite of landscape species.

landscape species. By highlighting data deficiencies, the process was useful for setting research priorities. The data sets and scores were also general enough to be useful in other conservation planning activities. For example, the threats data presented here were summed by species; the same data can also be summed by threat to gauge the relative effects and the numbers of species affected by each threat within the landscape.

The later discussion highlights a number of difficulties and limitations of the method; many are associated with incomplete or low-quality data. Virtually all conservation planning exercises will be plagued by data limitations (Groves et al., 2002); therefore, our goal must be

to use the best available information while keeping the process moving, and as new data are collected, they should be incorporated. The products should be viewed as “living documents” to be updated in an adaptive manner (Holling et al., 2002). Perhaps more significantly, the point of the selection process presented here is to identify the most informative species for site-based conservation; this means that the relative relationships between species are far more important than whether the results are correct in an absolute sense. It is important to acknowledge these limitations and recognize the selection criteria for what they are: a decision-support structure for choosing focal-species, rather than a one-size-fits-all recipe.

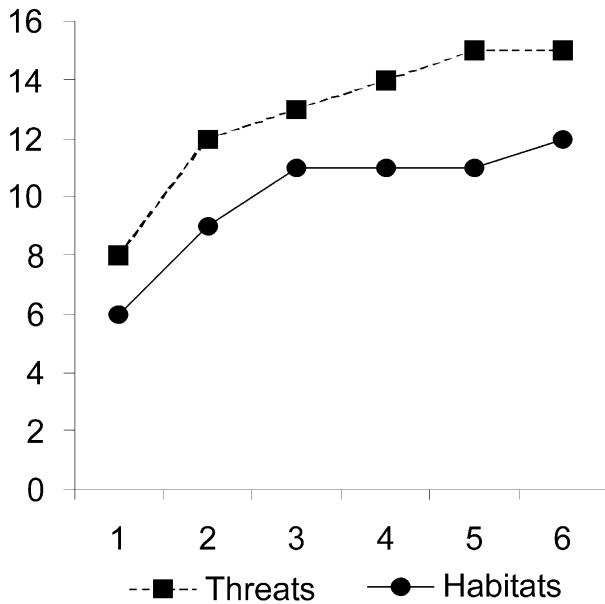


Fig. 3. Cumulative numbers of major threats encountered and habitat types required by the suite of landscape species (vertical axis) as species were added (horizontal axis). Data for Madidi are shown, where 11 major habitat types and 15 major threats were considered.

7.1. Complementarity: the great leveller

Complementarity had a leveling effect on the choice of species for the final suite. Recall that after the first landscape species is selected, the next five highest-ranked species are considered and the species with the least overlap in spatial distribution and threats addressed is added to the suite. This has a number of implications. First, changes (either from different weighting or improvements in data quality) that do not affect a species' rank within the group of candidate species will have no effect on the suite of landscape species chosen. Second, changes that affect species' ranks will only affect the suite of landscape species chosen under two circumstances: (1) if a species is moved out of the group of species considered for complementarity, or (2) if the change moves it below a species with similar spatial requirements and threats.

The vicuña in Madidi illustrates potential complication that can arise if a species whose needs are not met by the current suite falls outside the next five candidates considered for complementarity, but the current suite covers the needs of the five candidates. One might argue that such a species would more accurately be considered a special element (*sensu* Noss, 1996), and its conservation planned outside of the landscape species approach. However, we acknowledge that there is nothing magical about five species to be considered for complementarity. The number of species considered for complementarity, five or more or fewer, should depend on the group of candidate species and ultimately the source fauna. A diverse fauna with greater numbers of wide ranging

species (e.g. an African savanna) might require a larger number of complementary species, than one with less redundancy (e.g. arctic tundra or a desert).

7.2. Data sources, uncertainty and subjectivity

Ideally, all selection data would be rigorously collected at the site being considered, but many management decisions must be made without this luxury. Given this reality, it is essential to track uncertainty in our information throughout the selection process. We recommend scoring all quantitative data according to their statistical and methodological rigor and applicability to the target site. This will (1) maintain transparency and let others appraise the validity of our decisions, and (2) provide an index for setting research priorities. We used the six-point scale in Table 6 to summarize the quality of data used for each criterion. A similar scale could be applied to overall data for each species, category, or even for each datum.

Data availability on habitat types may also affect the selection of species. For example, the data used here are heavily biased toward terrestrial habitat types. This obscures variation in habitat use for aquatic species, so our measures of habitat heterogeneity for aquatic species are too low. This also reduces the extent to which aquatic species complement terrestrial species. For example, even though anaconda and river turtle are known to use different aquatic habitats, they were treated as identical in terms of complementarity because aquatic habitats were lumped in the Madidi dataset. A more extreme example came in considering complementarity for the fourth species added to the Madidi suite; all the habitat types "used" by the giant river otter were already covered by white-lipped peccary and spectacled bear. We acknowledge that, biologically speaking,

Table 6

A proposed scheme for scoring data quality in the selection process

1	Excellent data from the site
2	Excellent data from an ecologically similar site
3	Good data from the site or an ecologically similar one
4	Bad or incomplete data from the site or a similar ecosystem
5	Data only available for a similar species or at an ecologically dissimilar site
6	No data—informed guesses based on sister taxa, allometry, or another indirect source

"Excellent data" were defined as: complete data sets collected using rigorous methods and with reasonable precision and statistical power. Where seasonality and/or interannual variation are significant, data sets should span these temporal scales. "Good data" were defined as data (perhaps collected for another purpose) that suggest a pattern or result relevant to the criterion or parameter being considered, but that do not cover all seasonal or interannual variations, has low power or precision or may be limited in some other way. "Bad or incomplete" data consisted of anecdotes, incomplete datasets, or data collected for substantially different questions, but that are somehow relevant to the question being considered.

this is nonsensical, but this situation is an artifact of the land cover data available and underscores the need to examine data sets and assumptions carefully and to acquire better data as necessary.

As the previous section illustrates, the selection process presented here does not entirely eliminate subjectivity in the choice of focal species or conservation planning. All prioritization schemes carry inherent value judgments and subjectivity (Redford et al., 2003). However, this process forces us to confront the gaps in our knowledge, be explicit about our decision-making, and make our subjectivity more transparent—all essential elements of adaptive management.

Another subjective element of the selection criteria is that they may be biased toward large-bodied generalists. This could arise from the species chosen for the source pool and may be reinforced by the inclusion of area and heterogeneity categories in the selection criteria. Acknowledging the disparities in data availability and conservation interest in vertebrates, we recommend including all potentially suitable candidate species but recognize the difficulties associated with such a strategy. As mentioned earlier, considering only species with a reasonable chance of being selected led the selection teams to evaluate only larger-bodied vertebrates.

We confronted the generalist bias in the context of complementarity. The two simplest definitions of a species' complementarity are: (1) to add the *maximum* number of new habitat types, jurisdictional units or threats to the existing suite of landscape species, or (2) to have the *minimum* overlap between a species' own habitat types, jurisdictional units and threats, and those of the existing suite. We chose the second measure because it would favor a habitat specialist that complements the existing suite over a generalist species that happens to occupy one extra habitat type. While this may lead to larger suites of species, it reduces the chance that the needs of an area-limited habitat specialist will not be met by those of a suite of landscape species.

7.3. Comprehensive planning for conservation

Broadly speaking, there are three primary “tracks” to site-based planning for conservation: wide-ranging species, representation and special elements (Noss, 1996; Noss et al., 2001). Planning to meet the needs of wide-ranging species helps to ensure that conserved habitats are sufficiently large to protect these and other less widely ranging species (i.e. traditional umbrella species concept, sensu Caro and O'Doherty 1999). Representation ensures that all significant habitat or ecosystem types are included in the landscapes being protected. Finally, special elements are unique, often small-scale communities, ecosystem types or species occurrences which warrant individual attention. By constructing a suite of complementary species, the landscape species

approach synthesizes the wide-ranging species and representation “tracks.” The advantage of considering these together is that it functionally links composition (the list of habitat types included) with configuration (Sanderson et al., 2002).

Some species' or communities local distributions may be so limited that they are more appropriately incorporated in site-based conservation as special elements, rather than landscape species (e.g. hippopotamus and Nile crocodile in Ndoki and otters in Madidi). These species, and others with highly restricted distributions—endemics on edaphic anomalies for example—may be more appropriately treated independently when not captured within the areas required by landscape species. All the species considered for the two landscapes discussed here had either relatively broad or exceedingly restricted ranges, so distinctions were easily resolved. We do not offer a fixed criterion for distinguishing special elements from potential landscape species, other than to consider the decision from the perspective of the approach: does considering the species' requirements inform site-based conservation planning by providing guidance on the necessary size, condition or configuration of landscape elements? If so, the species is probably best considered a potential landscape species. If not, the species is probably best considered a special element.

7.4. Testing the criteria

The ultimate test of the criteria presented here will be to determine whether meeting the spatial and resource requirements of a suite of landscape species provides for the majority of sympatric species. A rigorous test of this hypothesis may be impossible; indeed “coarse filter” approaches (Hunter, 1991) have yet to be tested empirically (Groves et al., 2002).

Focal-species approaches are increasingly used to inform conservation action. We have attempted to provide a transparent and reproducible method for identifying sets of particularly informative species for site-based conservation: landscape species. As with the data going into the selection process, the method itself should be viewed as a starting point to be tested, refined and inevitably improved upon. The results should be evaluated in the context of the site and the objectives of conservation practitioners before they are blindly accepted. With clear specification of the goals of reserve design and management, this process will ultimately lead to more efficient and effective site-based conservation.

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